Visioning Place-Based Resource Recovery: the Twin Falls, Idaho Water Resource Recovery Facility

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Authorization to Submit Thesis

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Abstract

In 2012, WEF adopted a new name for wastewater treatment plants (WWTPs) – Water Resource Recovery Facilities (WRRFs) – with a vision to change the industry from an emphasis on treatment to one that maximizes recovery of valuable resources present in the wastewater. For example, resources that can be recovered or produced from wastewater include biogas, which can be utilized via combined heat and power (CHP) systems to generate electricity; fertilizer-grade nitrogen and phosphorus via struvite crystallization; reclaimed water for non-potable and potentially potable uses; fertilizer-grade biosolids; and potentially bioplastics. Many reports and peer-reviewed publications have mused about the concept of resource recovery, and specific processes have been evaluated that could be employed to produce specific products. However, little-to-no attention has been given to the analysis and evaluation of real, place-based WRRF scenarios. Real WRRF examples and scenarios that present and analyze environmental and economic impacts/benefits are needed. This study does exactly that re-envisioning the City of Twin Falls, Idaho WWTP as a WRRF. Twin Falls has several proximate industrial waste streams that could be beneficially leveraged for enhanced resource recovery - from such industries as Chobani, Glanbia, and Lamb Weston, as well as dairy operations, among others. Instead of pre-treating these waste streams, bypassing the resource recovery opportunity, this study integrates these resource streams into a re-envisioned Twin Falls WRRF.

To facilitate development of WRRF scenarios in this study, we employed the SUMO© model (Dynamita Process Modeling) to evaluate and assess resource recovery alternatives leveraging four proximate waste streams: dairy manure, and industrial wastewaters from TASCO, Chobani, and Lamb Weston. A base model of the existing Twin Falls WWTP was developed and calibrated utilizing facility operational data. Environmental life cycle assessment was performed using the EPA TRACI model to assess the sustainability of several configurations that integrate different combinations of resource recovery technologies. The overall goal was to determine which combination of resource recovery technologies and proximate waste streams would be optimal for the Twin Falls WRRF.

Several of the studied alternatives presented positive benefits, both environmentally and economically. Environmental benefits were largely due to the use of power produced onsite via CHP offsetting power produced from the grid. This result was surprising, as the fuel sources used for electricity production by Idaho Power are approximately only 25% fossil fuel based. Struvite production onsite was found to have some positive environmental impacts due to an offset of mineral fertilizer use. The effects of chemical production used in struvite production, however, had a relatively high quantity of associated emissions, resulting in a net environmental loss. This study also found that the environmental benefits of phosphorus removal were miniscule in the prevention of eutrophication of water.

Acknowledgements

This thesis would not have been manageable without the help and mentorship of several individuals. I would like to first acknowledge my major professor, Dr. Erik R. Coats, whose mentorship and guidance throughout this research has been invaluable. I would also like to acknowledge the support of Jason Mellin, Nick Guho, and Alex Crozes for sharing their expertise and providing insights that assisted with some of the more difficult aspects of this work. I also extend my gratitude to het members of my committee, Dr. Patricia Colberg and Dr. Patrick Wilson, for their support along the way. Additionally, I would like to thank Dr. Imre Takacs for his assistance with SUMO process modeling software; and Shawn Moffit, Bill Leaf, and other and Twin Falls WWTP staff with Jacobs for providing the data that made this work possible.

Dedication

To my parents for always pushing me to succeed and supporting me with my educational endeavors.

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List of Abbreviations

ADM1	Anaerobic Digestion Model No. 1
AD	Anaerobic Digestion
AS	Activated Sludge
ASM	Activated Sludge Model
ATP	Adenosine triphosphate
BOD	Biochemical Oxygen Demand
CHP	Combined Heat and Power
CNG	Compressed Natural Gas
COD	Chemical Oxygen Demand
DO	Dissolved Oxygen
EBPR	Enhanced Biological Phosphorus Removal
ELCA	Environmental Life Cycle Assessment
ffCOD	Filtered and flocculated COD
GAO	Glycogen Accumulating Organism
GBT	Gravity Belt Thickener
HRAAD	High rate anaerobic-aerobic digestion
IC	Internal Combustion
IWA	International Water Association
LCA	Life Cycle Assessment
LCIA	Life Cycle Impact Assessment
MAP	Magnesium Ammonium Phosphate
MBBR	Moving Bed Biofilm Reactor
mgd	Million gallons per day
MMC	Mixed microbial consortia
NPDES	National Pollutant Discharge Elimination System
OHO	Ordinary Heterotrophic Organism
OUR	Oxygen Uptake Rate
P	Phosphorus

PAO	Phosphorus Accumulating Organism
PHA	Polyhydroxyalkanoate
PHB	Poly(3-hydroxybutyrate)
PHB-coV	Poly(3-hydroxybutyrate-co-3-hydroxyvalerate)
PHV	Poly(3-hydroxyvalerate)
RAS	Return Activated Sludge
rbCOD	Readily Biodegradable COD
RQ	Research Question
SBR	Sequencing Batch Reactor
sCOD	Soluble COD
SND	Simultaneous nitrification and denitrification
TKN	Total Kjeldahl Nitrogen
TRACI	Tool for the Reduction of Chemical and Other Environmental Impacts
UASB	Upflow Anaerobic Sludge Bed
VFA	Volatile Fatty Acid
WAS	Waste Activated Sludge
WASSTRIP	Waste Activated Sludge Stripping To Recover Internal Phosphate
WEF	Water Environment Federation
WRRF	Water Resource Recovery Facility
WWTP	Wastewater Treatment Plant

1. Introduction

Historically, wastewater management has been implemented with a treatment-centric view, driven principally by regulations, the need for environmental health protection, and cost, usually with the least expensive option selected (Novak et al. 2015). Indeed, wastewater treatment – especially for non-municipal entities – has been implemented largely because the discharge of raw wastewater to the environment is no longer allowed under U.S. law. However, conventional waste management practices and strategies have become too myopic and outdated, while concurrently not capturing the true value of the resource that is wastewater (Peccia and Westerhoff 2015). Recognizing the need for change, there is an emerging paradigm shift, termed resource recovery, to envision wastewater as a resource with several potential outcomes (Guest et al. 2009). Many real opportunities exist, some that should be leveraged as mandatory while others that could be implemented more strategically.

The greatest intrinsic value of wastewater is the water itself. Many areas in the world are water stressed, and the ability to produce reclaimed water from wastewater can significantly improve situations in which water access is limited (Daigger 2008, Daigger 2009). Wastewater treatment systems also have the ability to sustainably produce valuable fertilizers, either in the form of struvite or as biosolids (Le Corre et al. 2009). Another significant opportunity for resource recovery is in the form of energy. Typically, this is realized through anaerobic digestion of primary and/or waste activated sludge, which results in the production of biogas rich in methane. The methane can be harvested for its energy and used to heat the digesters as well as offset some of the other energy demands at a water resource recovery facility (WRRF). Worldwide, wastewater treatment is one of the largest users of energy. Wastewater does, however, contain a substantial amount of energy before being treated. In fact, others have suggested that wastewater contains enough energy that WRRFs have the potential to be net energy producers (McCarty, Bae and Kim 2011).

Beyond the more “conventional” resources that could be produced from wastewater, bioplastics are yet another form of resource recovery that shows potential. Specifically, polyhydroxyalkanoates (PHAs) are an intracellular carbon storage granule that can also serve as a bioplastic. PHAs are currently produced using pure cultures, but the potential to produce PHAs using mixed microbial cultures, such as those enriched in WRRFs, and wastewater also exists (Coats et al. 2007).

In the wastewater industry, there has much conceptual visioning of resource recovery. For example, the Water Environment Federation (WEF) has adopted the term ‘Water Resource Recovery Facility’ and advocates its use when referring to what has conventionally been known as Wastewater Treatment Plants (Fulcher 2014). Envisioning this advanced, more holistic concept demonstrates both the need and desire for the industry to become more progressive and proactive. However, adoption of resource recovery technologies remains limited. Research has shown that there are many barriers to overcome in adopting resource recovery – including regulations, economics, and social acceptance. Of these barriers, cost appears to be the primary factor that drives municipalities away from resource recovery, as a positive return on investment is necessary. However, many of the benefits of resource recovery are difficult to monetize and are not typically included in cost-benefit analyses, which is almost exclusively treatment driven (Coats and Wilson 2017).

While many have philosophically, socio-politically, or otherwise mused or written about resource recovery, WRRFs, and potential barriers to implementation (Smith et al. 2014, Guest et al. 2009, Puyol et al. 2017, Coats and Wilson 2017), to date there has been no real evaluation, assessment, and demonstration of place-based WRRFs. Because these critical analyses are needed, this study developed, evaluated, and assessed real scenarios for the application of resource recovery technologies by a WRRF located in Twin Falls, Idaho. The southern Idaho region has tremendous potential for resource recovery due to large numbers of industries that generate wet organic wastes, including dairies and dairy and food processing plants. These industries produce high strength wastewaters that contain substrates conducive to resource recovery. However, current industrial waste management practices are quite conventional; that is, each facility operates under an industrial pretreatment permit requiring discharge of their effluent to the city's sanitary sewer system. Clearly, in the context of resource recovery, this legacy, treatment-centric approach results in unrealized opportunities; moreover, their infrastructure, energy, and capital-intensive footprint erodes profitability. To implement resource recovery, the value contained in these types of waste streams must be transferred to and integrated into the city's WRRF.

Research conducted in this study evaluated, applied, and optimized technologies to recover and produce four different high-value resources - reclaimed water, struvite, biogas, and bioplastics - leveraging proximate industrial waste streams. The WRRF in Twin Falls currently operates at an average day maximum month flow of 8 million gallons per day (mgd), focusing on ammonia and biochemical oxygen demand (BOD) removal in compliance with the facility's NPDES waste discharge permit. Solids handling consists of thickening, anaerobic digestion, and dewatering to produce class B biosolids, which are utilized by local area farms as a fertilizer (note: the city bears all costs in producing and providing this fertilizer). Biogas produced is currently used for the heating of the digesters, with the excess flared. Ultimately the Twin Falls WRRF is designed and operated in a manner very consistent with the industry norm.

The focus of this research was to quantify and evaluate resource recovery opportunities through process modelling of the Twin Falls WRRF. Additionally, environmental life cycle assessment (ELCA) was used as a tool to evaluate the effect of resource recovery in increasing sustainability with respect to P-recovery, energy production from biogas, and use of PHAs as bioplastics.

1.1 Research Questions, Hypotheses, and Objectives

Described below are the research questions (RQs) that ultimately drove this study, along with coupled hypotheses and associated research objectives.

RQ 1:	Several proximate high-strength organic-rich waste streams exist within the Twin Falls area. For some agro-industrial facilities (e.g., Chobani) pretreatment currently occurs through upflow anaerobic sludge beds (UASBs) to reduce the chemical oxygen demand (COD) load on the downstream Twin Falls WRRF, while dairies commonly rely on lagoons and land disposal. As an alternative to pretreatment, these waste streams could be used to enhance resource recovery at the Twin Falls WRRF. A central challenge, however, is transport of the various waste streams to the WRRF; waste characteristics including strength and volume might not be amenable to transport via the city's sanitary sewer collection system. This challenge necessarily raises the following question: <u>What are the industrial waste stream characteristics and how might the waste characteristics dictate transport to the WRRF?</u>
Hypothesis 1:	Agro-industrial waste streams proximate to the Twin Falls WRRF will require separate transport to the WRRF (i.e., not commingled in the municipal sanitary sewer collection system) in order to retain maximum value for resource recovery.
Objective 1:	Evaluate and establish the most effective and feasible method of transporting agro-industrial waste to the Twin Falls WRRF to maximize the overall sustainability and economics of proposed centralized resource recovery.

RQ 2: Leveraging proximate agro-industrial waste streams, what combination of resource recovery opportunities could be achieved at the Twin Falls WRRF?

Hypothesis 2: Integrating proximate agro-industrial waste streams, the Twin Falls WRRF could be re-configured to maximize resource recovery for the concurrent production of struvite, nutrient-rich biosolids, CHP, and bioplastics, while concurrently achieving compliance with its NPDES permit.

Objective 2: Utilizing SUMO process modelling software, develop comprehensive process scenarios for maximum resource recovery from agro-industrial waste streams integrated with the municipal wastewater. Several different process configurations will be considered, culminating in a complete inventory analysis that can be used to compare the different configurations.

RQ 3: Success of a resource recovery platform at the Twin Falls WRRF that integrates municipal and agro-industrial waste streams must be measured from both an environmental and economic perspective. In this regard, what is the optimal WRRF configuration that integrates proximate industrial waste streams and produces resources of highest environmental and economic benefit, and how does this scenario compare with current wastewater management strategies?

Hypothesis 3: Wastewater resource recovery accomplished through strategic integration of municipal and agro-industrial wastewaters will be environmentally and economically feasible and justified over current conventional wastewater treatment practices.

Objective 3: Utilize EPA’s Tool for the Reduction of Chemical and Other Environmental Impacts (TRACI) model to perform comparative environmental impact assessments based on the inventory analysis of the process configurations determined in Objective 2. Conduct comparative economic analyses to determine which configurations have the most potential and feasibility for application at the Twin Falls WRRF.

2. Literature Review

2.1 Genesis of the Wastewater Resource Recovery Concept

In 2013, WEF adopted the term Water Resource Recovery Facility (WRRF) in place of Wastewater Treatment Plant (WWTP), which exemplifies the changing industry's desire to focus on the resources that can be obtained from wastewater processes, as opposed to a treatment-centric focus (Fulcher 2014). However, the idea of wastewater containing something of inherent value is not new. Nearly a century ago, Clark discussed the value of fertilizing material and ammonia present within wastewater and sludges associated with its treatment (Clark 1930). Engineers in European cities in the 19th century noted the fertilizing value of nitrogen and phosphorus in wastewater, going so far as to suggest its use as a replacement for nonrenewable mineral fertilizers (Sewage of Towns 1867). Thus, resource recovery has been on the minds of those in the wastewater industry for a long time, and its lack of widespread adoption suggests the need to incorporate more than simple economics into the decision-making process.

This chapter describes the tools utilized in this study and discusses the specific resources targeted for recovery, along with the technologies available for recovery. Life cycle assessment is a comparative tool that was used extensively in this research to compare different process configurations of resource recovery integration. Phosphorus recovery was investigated through crystallization, with enhanced biological phosphorus removal as a mainstream process enhancement to improve effluent quality and realize greater phosphorus recovery. Anaerobic digestion was explored to increase biogas output which can be used with combined heat and power systems to create electricity. Lastly, bioplastic production was considered by coupling microbes present in WRRF treatment processes with waste carbon sources.

2.2 Life Cycle Assessment

Life Cycle Assessment (LCA) is a tool that can be applied to generate useful metrics that allow policy makers, designers, etc. to make more informed decisions regarding a product or process relative to the environmental impacts. LCA dates back to environmental concepts that emerged in the 1970s, with a consistent framework developed in the 1990s. Rather than simply referring to something as "environmentally sustainable," LCA quantitates metrics to provide a sounder basis for "sustainable" when comparing alternative processes or products. LCA is largely focused on the environment as ELCA, but can be applied to other areas as well, such as with life cycle cost (LCC). The four stages of LCA according to ISO 14040 and 14044 are presented below and illustrated in Figure 2.1.

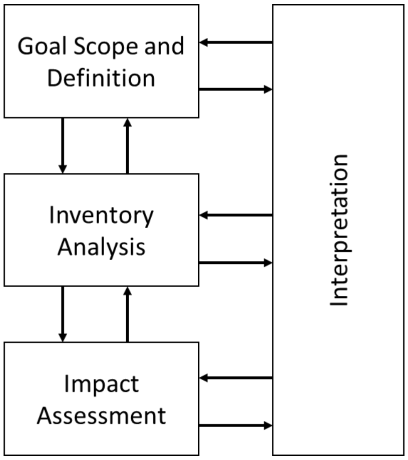


Figure 2.1: Life Cycle Assessment Stages

2.2.1 Example Wastewater LCA Studies

LCA has been an integral tool in numerous wastewater treatment-based studies. Several of these studies were reviewed in order to better understand the implementation of LCA within the water industry. These studies provided much insight in addressing many of the challenges associated with LCA, such as the difficulty in the selection of a functional unit; the majority of studies opt for a volume of wastewater due to their treatment centric nature. Many studies utilized TRACI for the impact assessment, further cementing its usefulness in addressing the needs of this study. Below are short summaries of the studies utilized in this process, with a focus on the individual aspects of LCA: goal and scope, functional unit, inventory analysis, impact assessment, the interpretation of the data collected.

In 2013, Corominas et al. reviewed 45 journal articles in which LCA was applied to studies in the water and wastewater fields. Several potential areas for improvement were found, including better adherence to ISO standards, more consistency in system boundary and functional unit, methodology for impact assessment, and shifting focus from pollutant removal to resource recovery. Several areas of the research presented herein were informed by conclusions drawn from this work. For example, the system boundary was expanded to include all aspects of treatment, including pretreatment. Sludge disposal was also included due to its large transportation footprint. Further expanding on sludge disposal is the usefulness of biosolids in fertilizing; Corominas discusses the substitution of synthetic fertilizer with biosolids, which is an idea also utilized in this research.

Remy et al. (2016) used LCA to evaluate different resource recovery alternatives from municipal wastewater in Germany, with a focus on solids handling and energy use. Solids treatment improvements investigated utilizing thermal hydrolysis to pretreat sludge before digestion, resulting in greater biogas yields. Hydrothermal carbonization was also explored to increase sludge dewaterability prior to incineration, resulting in greater energy recovery. Phosphorus recovery from sludge was also considered, comparing struvite precipitation, sludge incineration, and sludge leaching of phosphorus using chemicals. It was found that while struvite is relatively easy to recover, it is limited to recovery of approximately 16% of total phosphorus contained within the sludge; much of the phosphorus present remains in the sludge and is not hydrolyzed. Other alternatives saw higher phosphorus recovery but required significantly more energy. The ultimate conclusion from this work was that there are several viable pathways to significantly lower the energy consumption at WWTPs, and a one-size-fits-all approach is not a viable strategy for the implementation of energy recovery technologies.

Postacchini et al. (2016) performed an LCA on the treatment of primary effluent using activated sludge (AS), trickling filters, and high rate anaerobic-aerobic digestion (HRAAD).

The functional unit utilized was a volumetric flow of wastewater. LCA databases and collected data were utilized for the inventory analysis, with TRACI and IMPACT 2002+ used for the impact analysis. In achieving treatment, it was found that HRAAD and trickling filters had much lower environmental impacts than activated sludge based on both TRACI and IMPACT 2002+; AS exhibited greater environmental impact, principally due to the intensive energy demands of maintaining aerobic environments in AS systems. The impacts of HRAAD and trickling filters were comparable when using IMPAC 2002+, but trickling filters were shown to have less environmental impact with TRACI. Further data analysis revealed that cutting down on sludge production lowered environmental impacts significantly with both TRACI and IMPACT 2002+, especially with regards to HRAAD.

Ishii and Boyer (2015) investigated urine source separation with centralized wastewater treatment, focusing on nutrient recovery as struvite. The functional unit was selected to be 1,920 m^3 of urine. SimaPro was used for inventory analysis and TRACI for impact assessment. It was found that urine source separation with magnesium oxide for struvite precipitation had higher economic benefits and lower environmental impacts than centralized wastewater treatment.

In a subsequent study, Landry and Boyer (2016) compared urine source separation, centralized wastewater treatment, and centralized wastewater treatment with ozone disinfection. The LCA focusing on nonsteroidal anti-inflammatory drug removal. The functional unit was selected to be 11,184 m^3 of urine, with the scope including conveyance, storage, pharmaceutical management, and nutrient management. TRACI was used for impact assessment. They concluded that centralized wastewater treatment coupled with ozonation was both more expensive and had more environmental impacts than centralized wastewater treatment alone. Scenarios utilizing urine source separation had lower environmental impacts, but because of the greater removal of pharmaceuticals, had similar economic costs.

Rahman et al. (2016) used LCA to compare several nutrient removal processes to achieve three different levels of effluent quality, ranging from 8 mgN/L to 1 mgN/L of total nitrogen and 1 mgP/L to 0.01 mgP/L of total phosphorus. The system boundary included treatment from influent pumping to effluent discharge, with a functional unit of $1m^3$ of influent wastewater. BioWin process modeling software was use to determine process chemical and energy requirements and effluent quality. Their study assumed a 10 mgd influent flow with average wastewater characteristics obtained from the literature. SimaPro was used to complete the inventory analysis, and TRACI was used for the impact analysis. While increasing treatment levels were found to be environmentally beneficial on a local level, they identified some negative externalities that are not often considered in decision making.

Coats et al. (2011c) compared EBPR with chemical phosphorus removal. The functional unit was selected as 10 million gallons of treated effluent. Literature reported values for medium strength wastewater were used, with BioWin process modeling to compare two desktop scenarios of EBPR and chemical phosphorus removal using alum. Another part of this study used data obtained from two full-scale wastewater treatment plants operating in the Portland, OR metropolitan area – one of which realizes EBPR, the other performing chemical phosphorus removal. TRACI was used to quantify environmental impacts. It was found that EBPR is less environmentally harmful than chemical phosphorus removal, considering all of the externalities with production of electricity for treatment and pumping, production of alum, and sludge disposal.

Lassaux et al. used LCA to investigate water usage in the Walloon Region of Belgium. Several different scenarios were evaluated with LCA using a functional unit of 1 m^3 of water leaving the tap. Three reference scenarios were used, representing different expectations of the expansion of water infrastructure and considering effects of wastewater discharge without treatment and with biological nutrient removal. Using Eco-indicator 99 for impact assessment, it was found that higher levels of treatment at WWTPs are beneficial, and phosphorus and nitrogen removal should be encouraged.

As a final example of LCA and wastewater, Smith et al. (2014) focused on energy recovery from biogas with CHP systems, comparing conventional activated sludge coupled with anaerobic digestion, high rate activated sludge coupled with anaerobic digestion, aerobic membrane bioreactors with anaerobic digestion, and anaerobic membrane bioreactors. Typical literature values for medium strength wastewater were used with GPS-X process modeling software to determine effluent quality and biogas production. The functional unit was five million gallons per day of influent wastewater and TRACI was used for impact assessment. This study concluded that anaerobic membrane bioreactors exhibit a high potential in terms of energy recovery, but are disadvantaged because they discharge an effluent high in nutrients and dissolved methane. Overall this study highlights the shortcomings of mainstream anaerobic treatment when compared to other treatment options that also recover energy illustrates areas where development of the technology is needed.

2.3 Candidate Wastewater Resources for Recovery

Within the context of a WRRF, there are several resources available for capture. Of specific interest to this study are phosphorus recovery through crystallization, energy recovery via biogas and CHP, and bioplastic production as PHA.

2.3.1 Phosphorus and Phosphorus Recovery

Phosphorus is essential to life, with no other element able to replace its role. It is an integral component of nucleotides, which form the basis for nucleic acids (i.e., RNA and DNA). Phosphorus is also contained in adenosine triphosphate (ATP), the molecule responsible for energy transfer in cells. The natural P cycle occurs slowly, on a geologic timescale. The element accumulate in living organisms, which ultimately release P upon death. This phosphorus makes its way into aquatic environments, undergoing sedimentation. Through geologic processes, phosphorus deposits are made available as phosphate rock (P_2O_5) (Steen 1998). The natural phosphorus cycle is significantly slower than, for example, the nitrogen cycle, which is very rapid due to the processes of denitrification – converting N to N_2 gas which is released to the atmosphere (Smil 2000).

In contrast to the natural phosphorus cycle, human activities have created an anthropogenic phosphorus cycle, in which we mine phosphate rock largely for use in fertilizers to grow crops. Because phosphate rock sees no natural replenishment, it has been estimated that concentrated reserves from which extraction is economically viable will be depleted within the next 50 to 100 years (Steen 1998).

To support global anthropogenic activities, Cordell et al. (2009) estimates that 17.5 Mt of phosphate rock is mined annually. Ultimately, much of this P is lost to runoff and overuse of fertilizer and so cannot be easily recycled. However, some of these losses can be mitigated

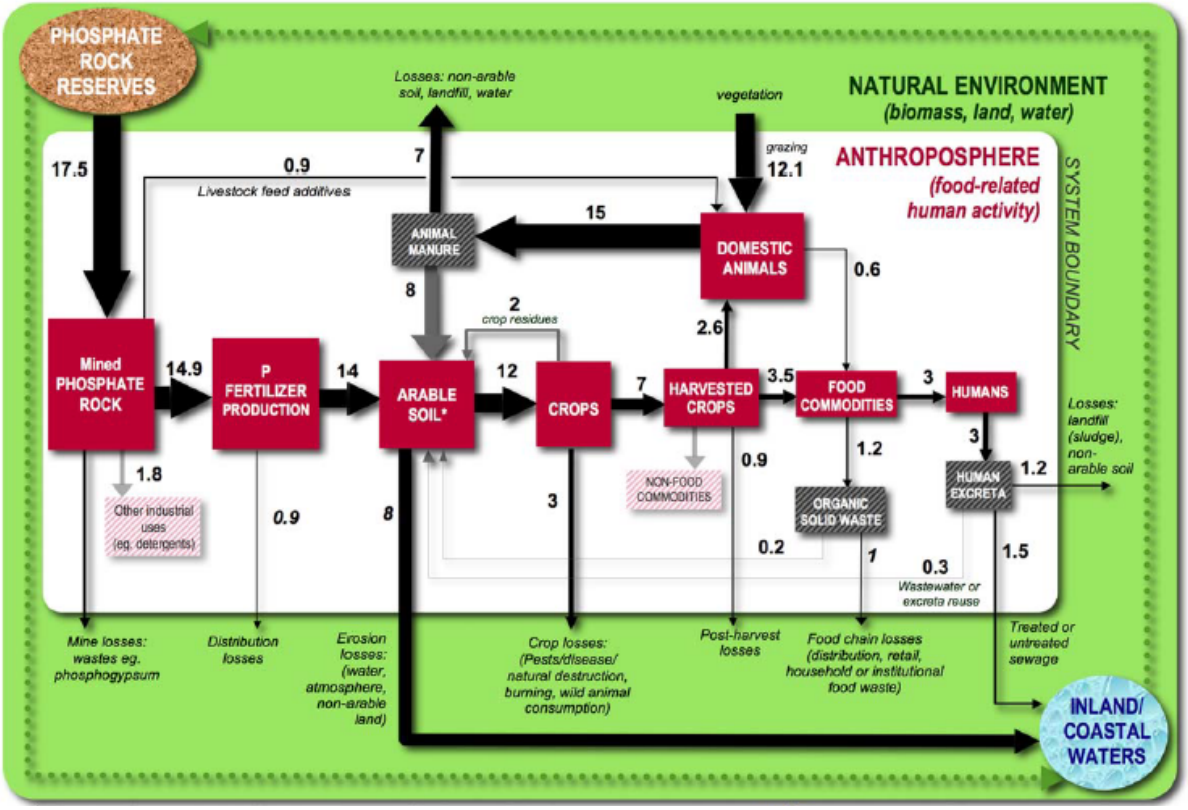


Figure 2.2: Global Phosphorus Cycle (Cordell, Drangert and White 2009)

via phosphorus recovery from waste streams (Childers et al. 2011). Smil et al. (2000) estimates that globally, 1.5 Mt of phosphorus is released into raw wastewater annually, which presents a waste stream with an opportunity to recover phosphorus (Smil 2000). Figure 2.2 details the phosphorus cycle with mass flows and illustrates anthropogenic effects and where losses occur (Cordell et al., 2009)..

While phosphorus recovery from wastewater systems alone cannot replace depleting phosphate reserves and support all human demands, it can help close the anthropogenic phosphorus loop while concurrently improving environmental water quality. Moreover, P recovery from wastewater minimizes future P recovery efforts, which would otherwise need to focus on highly diffuse P reserves present in the water environment. While Mayer et. al. (2016) suggests that the economic cost of recovering phosphorus alone does not justify its widespread adoption, when considering the total value of P recovery, more incentives emerge. This viewpoint illustrates the importance of economics in decision-making, while also suggesting that a more holistic view must be considered to appreciate the full effect of phosphorus recovery.

In WRRFs, phosphorus is typically recovered from side stream processes (i.e., wastewater streams generated internally in WRRFs that have anaerobic digestion) that are rich in nutrients. Two physiochemical treatment processes result in recovering P in two forms: struvite ($NH_4MgPO_4 \cdot 6H_2O$) and calcium phosphate ($Ca_3(PO_4)_2$). Both methods utilize chemicals to precipitate different crystalline structures containing phosphorus.

2.3.3.1 Hydroxyapatite and Struvite

Calcium phosphate most commonly crystalizes as hydroxyapatite ($Ca_5(PO_4)_3OH$). A commercial process for hydroxyapatite recovery was successfully pioneered in the Netherlands in the 1980s (Piekema and Giesen 2001). To produce hydroxyapatite, the formation of calcium carbonate must be avoided as it interferes with the crystallization of hydroxyapatite. Pretreatment with a strong acid is required to lower the pH of the solution to around 5, inhibiting the formation of calcium carbonate. Air stripping is employed as the next step, which serves to remove CO_2 from solution, serving to further inhibit calcium carbonate formation in the subsequent stages of the process. The pH is then increased to between 8 and 9 before the addition of quicklime (CaO), which serves as the calcium source for the reaction. Calcium phosphates have similar chemical composition to phosphate rock, allowing them to serve as a direct substitute in synthetic fertilizer production; however, they have minimal value applied directly as a fertilizer, and so require additional processing (Cabeza et al. 2011). Moreover, due to the high price of the chemicals required for calcium phosphate precipitation, this process is not often employed, as facilities that physiochemically recover phosphorus tend to opt for the more economical precipitation of struvite (Metcalf and Eddy 2013). Calcium phosphates will not be further considered in this research due to their economic unfavorability and the additional processing needed to convert them to fertilizer.

Another phosphorus-based chemical complex that can be produced at a WRRF is struvite - also referred to as magnesium ammonium phosphate (MAP). Struvite is a crystal composed of equimolar ratios of magnesium, ammonium, and phosphate. In WRRFs that utilize anaerobic digestion, struvite is traditionally considered a nuisance as it uncontrollably builds up on piping, in pumps, and on dewatering equipment. Mitigation of struvite buildup is achieved through addition of chemicals that either precipitate one of the reactants in the struvite-forming reaction or lower the pH so that struvite cannot precipitate (Metcalf and Eddy 2013). Intentional struvite precipitation for nutrient recovery can also alleviate the issue of uncontrolled struvite precipitation. Due to its low water solubility, struvite can serve as a slow-release fertilizer with high N and P and so has high potential economic value. Struvite production via precipitation is achieved in a process very similar to that of calcium phosphate. Centrate streams from dewatered anaerobic digester biosolids typically contain high concentrations of ammonia and phosphate, simply requiring the addition of a magnesium source and use of pH control to precipitate struvite.

In achieving struvite production in a wastewater environment, pH is one of the most important parameters to control. Optimal pH for struvite precipitation is about 10.3; however, in practice, pH is typically held between 8 and 8.8 to minimize chemical usage (Ohlinger, Young and Schroeder 2000). The most common method of achieving the desired pH level is through chemical addition of sodium hydroxide, but success has been achieved using other bases as well (Le Corre et al. 2009). Air stripping is another method that can be used to increase pH, which works by shifting the carbonate equilibrium; this method has potential to offset chemical costs, but is dependent on electricity costs.

Magnesium is generally present in very low quantities in wastewater, so to achieve the proper ratio of magnesium to phosphate, supplemental magnesium is usually required. Common sources of magnesium are MgCl, $Mg(OH)_2$ or seawater (Le Corre et al. 2009). Magnesium chloride is often the first choice for municipalities, as it is widely available due to its ubiquitous use in de-icing roads in winter. However, magnesium chloride is typically more expensive than magnesium oxide or hydroxide. Magnesium hydroxide has the added benefit of also increasing the pH of the bulk solution; this may be a good option for many municipalities because it is relatively inexpensive and much less additional chemical is required for pH adjustment (Münch and Barr 2001). The primary disadvantage of magnesium hydroxide is that pH adjustment and magnesium addition are coupled and so cannot be adjusted independently without the use of other chemicals (Halim et al. 2018). Magnesium oxide (MgO) can also be used, as it creates magnesium hydroxide upon contact with water. The biggest drawback in using magnesium hydroxide is its low water solubility; with a solubility product constant (K_{sp}) of 5.61×10^{-12}. Long retention times are needed to allow the magnesium hydroxide enough time to dissociate into water and precipitate. Lastly, while seawater is a good source of magnesium, its use is limited to facilities located near the sea (Nenov et al. 2016). While raw seawater has been shown to be effective, the reject water from nanofiltration has much higher magnesium concentrations and can serve as an inexpensive substitute for other magnesium sources (Lahav et al. 2013). It may, however, contain other foreign ions that could potentially interfere with struvite crystallization such as calcium, sulfates, sodium ions, and chloride (Yetilmezsoy et al. 2017).

Liquid streams from the dewatering of anaerobic digestion slurries serve as excellent candidates for struvite crystallization due to their high content of phosphorus and ammonium. Figure 2.3 (Münch and Barr 2001) illustrates a process schematic with struvite integrated into the liquid stream of the dewatering unit. These so-called ‘sidestreams’ may have phosphorus concentrations in the range of 100-300 mg P/L; when coupled with enhanced biological phosphorus removal (EBPR), the concentration can exceed 600 mg P/L. Ammonia concentrations typically range from 600–900 mg N/L (Lahav et al. 2013). EBPR coupled with struvite crystallization has a high potential to sustainably realize low effluent phosphorus concentrations in WRRFs (Münch and Barr 2001).

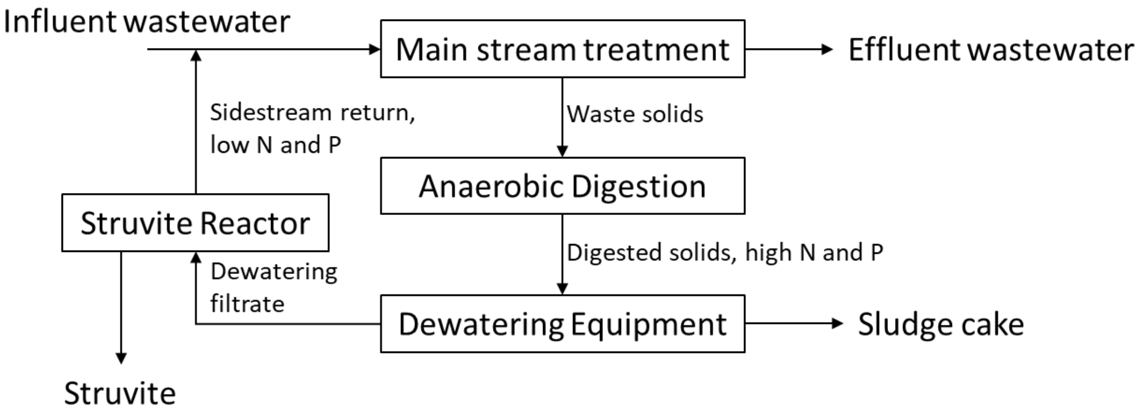


Figure 2.3: EBPR Process Flow Diagram with Struvite Production Reactor

As struvite is composed of an equimolar ratio of magnesium to ammonium to phosphate, magnesium is dosed to maximize phosphorus removal. A target dose for the molar ratio of magnesium to phosphorus is 1.2, which allows for rapid crystal growth and low effluent phosphorus. Effluent phosphorus from struvite precipitation reactors typically ranges from 10 to 25 mg P/L, as higher P recovery requires more chemical to maintain supersaturated conditions (Metcalf and Eddy 2013).

Struvite production requires supersaturated conditions, meaning the product of the molar concentrations of ammonia, phosphate, and magnesium must exceed the solubility of struvite. Supersaturation allows nucleation of two types: homogenous nucleation when struvite spontaneously forms and heterogenous nucleation, when struvite crystals grow on foreign or introduced particles. The supersaturation ratio is the product of the molar concentrations of the reactants divided by the solubility of struvite. Ideal reactor conditions maintain a low supersaturation ratio to prevent homogenous nucleation from occurring, as this prevents an excess of fine particles that are not desired in the final product; lower supersaturation ratios have been shown to decrease the spontaneous formation of fine particles, while still sustaining struvite formation. To better control nucleation of struvite and prevent fine particles from forming, a recycle is often employed to introduce smaller particles on which struvite may crystallize. Crystallization can be achieved in a completely mixed reactor that uses spontaneous struvite precipitation (i.e., no nucleation sites), but this is uncommon due to its large footprint.

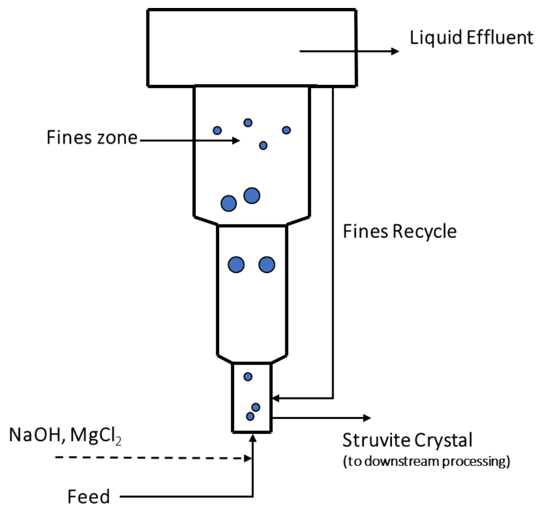


Figure 2.4: Ostara Struvite FBR Operational Schematic

Commercially-available fluidized bed reactors (FBRs) are often employed to precipitate and recover struvite. FBRs are inverted conical reactors with small diameters toward the bottom and larger diameters at the top. As flow occurs in the upwards direction, the upflow velocity is highest at the bottom of the reactor and is slowest at the top. This results in a particle size distribution with the finest particles towards the top of the reactor and the larger particles near the bottom, where they are removed for final processing. Figure 2.4, adapted from Ostara (Ostara 2018b), illustrates one example of an FBR application.

The Pearl process, developed by Ostara, utilizes the liquid stream from dewatering, which is pumped upwards into an FBR with a hydraulic retention time (HRT) of about 1 hour (Figure 2.4). Magnesium chloride is dosed to achieve the desired magnesium concentration, and sodium hydroxide is added to maintain the desired pH. Effluent low in phosphorus (< 25 mgP/L) is collected from the top of the reactor. Recycling of fines is implemented to provide

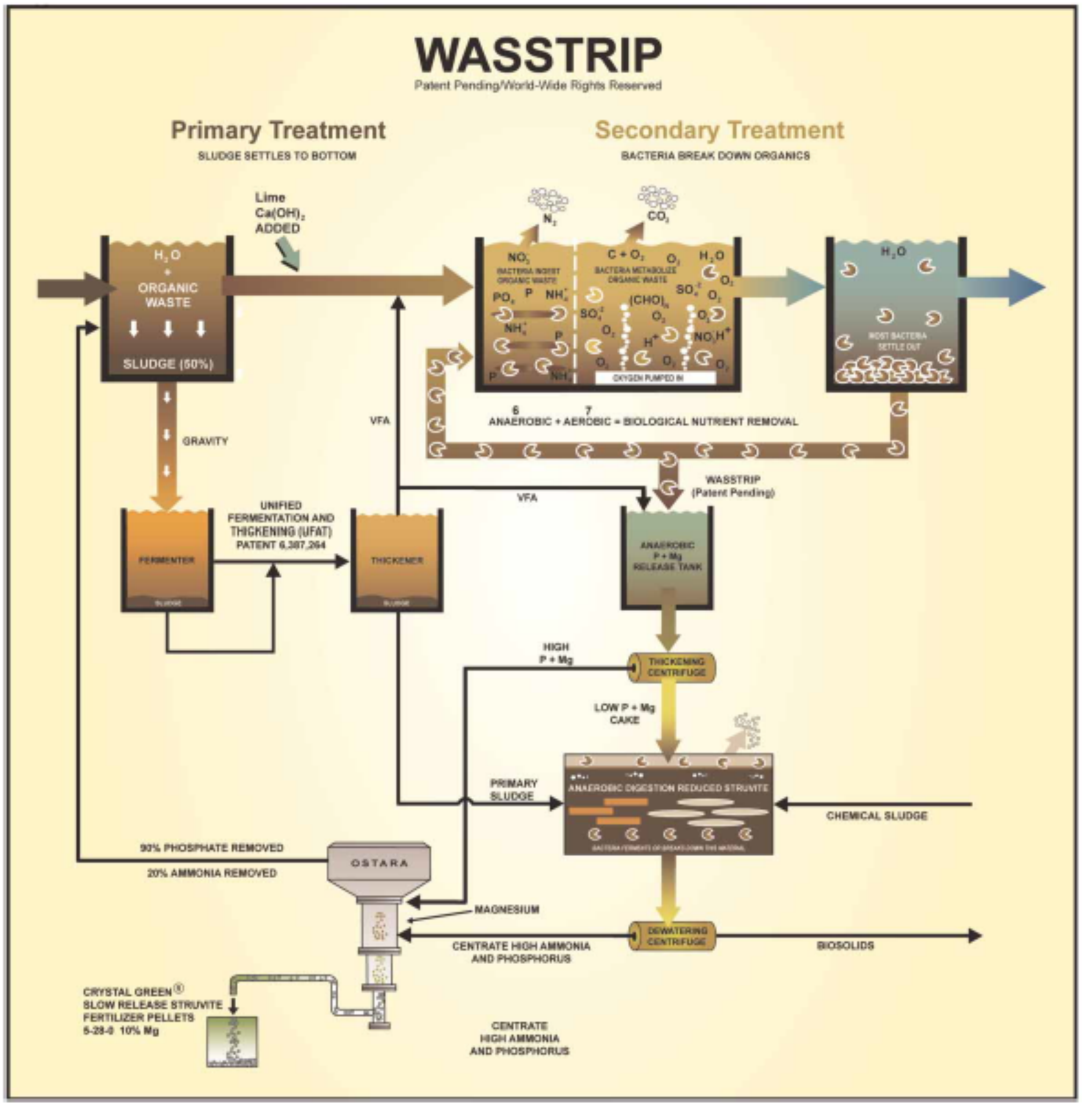


Figure 2.5: WASSTRIP Process Illustration (Ostara)

a nucleation site on which crystals grow and maintain the proper HRT and velocity profile. The final product, consisting of struvite particles with the largest possible diameter, are removed from the bottom of the reactor in batches. The struvite is sifted and dried to remove water (Ostara 2018b).

To mitigate problems of unintentional struvite precipitation within the digester itself, Ostara, in collaboration with Clean Water Services (Hillsboro, Oregon, USA), has developed the Waste Activated Sludge Stripping to Recover Internal Phosphate (WASSTRIP) process, illustrated in Figure 2.5. WASSTRIP works by holding P-rich WAS in an anaerobic tank to elicit a release of internal phosphorus into the bulk solution – induced either endogenously or intentionally through VFA addition (Cullen, Baur and Schauer 2013). Magnesium and potassium are also released by cells into the bulk solution, as they are cations used to balance charges from polyphosphate stores associated with the EBPR process (Arvin 1985). Effluent from WASSTRIP enters solids separation, where the liquid stream that is rich in phosphorus and magnesium, is sent to the struvite reactor. Thickened sludge low in phosphorus and magnesium enters anaerobic digestion, which has minimal struvite formation potential due to a lack of magnesium (Ostara 2017).

Ostara, based in Vancouver, BC, has 17 struvite recovery facilities installed at WRRFs throughout North America and Europe (Ostara 2018a). At the Clean Water Services WRRF in Durham, OR, Ostara Pearl and WASSTRIP processes were employed to remove phosphorus from digester centrate prior to discharge into the headworks of the plant. EBPR process stabilization and resiliency were realized, struvite formation potential within the digester was reduced, and a new revenue stream for offsetting operational costs was established (Cullen et al. 2013).

Several other commercialized processes for struvite crystallization exist, with most of them some variant of an FBR process. AirPrex was developed in Germany to avoid struvite buildup on dewatering equipment by directly utilizing the sludge stream. Magnesium chloride is added to the base of the reactor, which operates with an HRT of approximately 8 hours. The reactor is aerated to strip CO_2, raise the pH, and induce mixing. Struvite is collected after settling to the bottom of each reactor, and effluent sludge flows to dewatering (CNP 2017). Figure 2.6, provided by CNP, illustrates the AirPrex process.

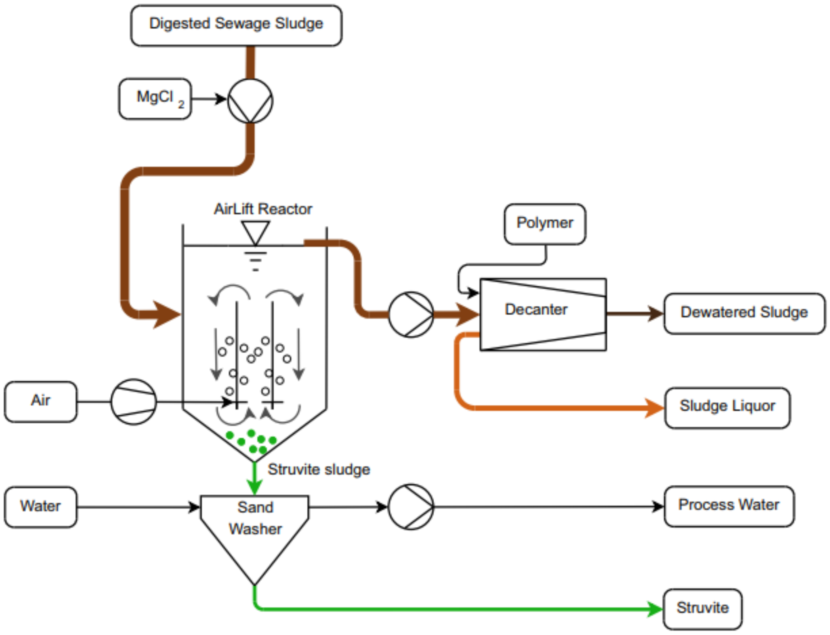


Figure 2.6: AirPrex Struvite Production Process Diagram (CNP)

Seattle-based Multiform Harvest (now owned by Ostara) developed a struvite process that utilizes a conical FBR with a gradually increasing diameter (i.e., diameter does not change at discrete intervals). Magnesium chloride and sodium hydroxide are added, and the larger struvite particles are collected from the bottom of the reactor. No internal recycle is utilized; nucleation is achieved primarily on foreign particles present in the side stream. Figure 2.7 illustrates the Multiform Harvest process diagram. Multiform Harvest struvite recovery technologies are currently employed at four WRRFs in the United States (Multiform Harvest 2018).

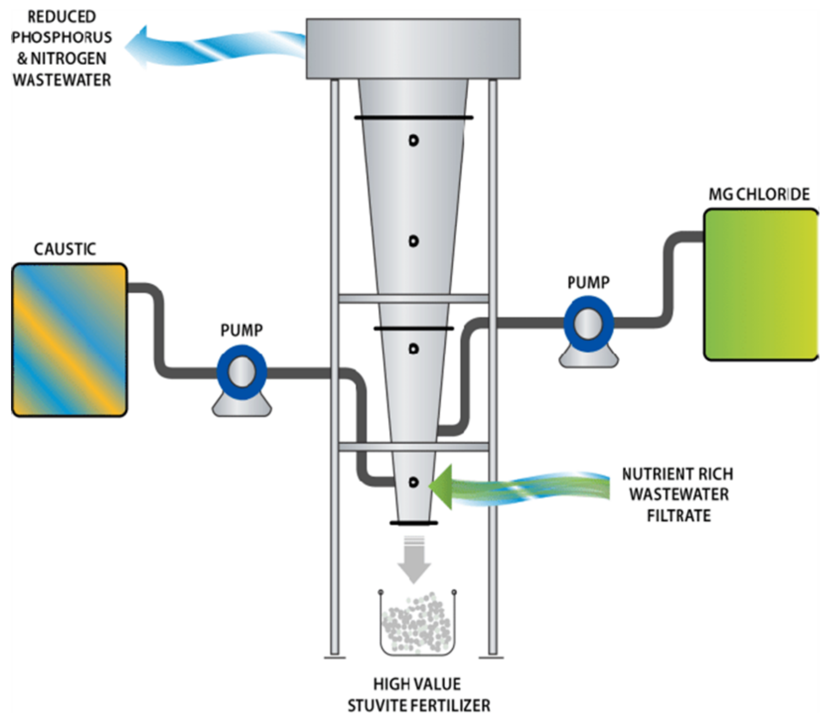


Figure 2.7: Multiform Harvest Struvite Production Process Diagram

2.3.1.2 Enhanced Biological Phosphorus Removal

While struvite production captures phosphorus from high N and P waste streams internally produced at a WRRF, ultimately this is just a fraction of the raw wastewater phosphorus entering a WRRF; maximum P recovery requires other upstream technologies to improve recovery. In this regard, two process approaches can be employed to capture raw wastewater phosphorus: chemical precipitation using trivalent metal salts and biological phosphorus accumulation. If the goal is resource recovery, the enhanced biological phosphorus removal (EBPR) process is the only viable way to recover phosphorus as a valuable commodity; EBPR has also been demonstrated to be much more sustainable than chemical P removal (Coats, Watkins and Kranenburg 2011c).

EBPR functions by selecting for phosphorus-accumulating organisms (PAOs). The process depends on using a VFA-rich substrate and an anaerobic contact zone before aeration (Gerber et al. 1986). In the anaerobic zone, PAOs hydrolyze internal stores of polyphosphate to produce the necessary ATP to consume and store VFAs (Wentzel et al. 1985). Like most other heterotrophs, PAOs are not able to utilize VFAs for growth without a terminal electron acceptor; however, they do have the advantage of being able to store VFAs as PHA, which serves as an internal carbon and electron storage polymer (Wentzel et al. 1990). PAOs are enriched in the MMC by alternating anaerobic and aerobic phases, as the ability to store carbon gives PAOs an advantage over other microorganisms.

In bulk solution, the anaerobic zone is characterized by an increase in phosphorus concentration and a decrease in VFA concentration. Aerobically, PAOs uptake phosphorus from bulk solution using energy from PHA oxidation and convert it to internal polyphosphate. At the end of the aerobic zone, the mixed liquor undergoes solids separation; the solids now contain the PAOs that are rich in P. The liquid effluent, which is low in phosphorus, may undergo further treatment before discharge. Figure 2.8 illustrates PAO metabolism under both anaerobic and aerobic conditions. The net result is an effluent phosphorus concentration that can be below 0.1 mg/L in a well optimized system.

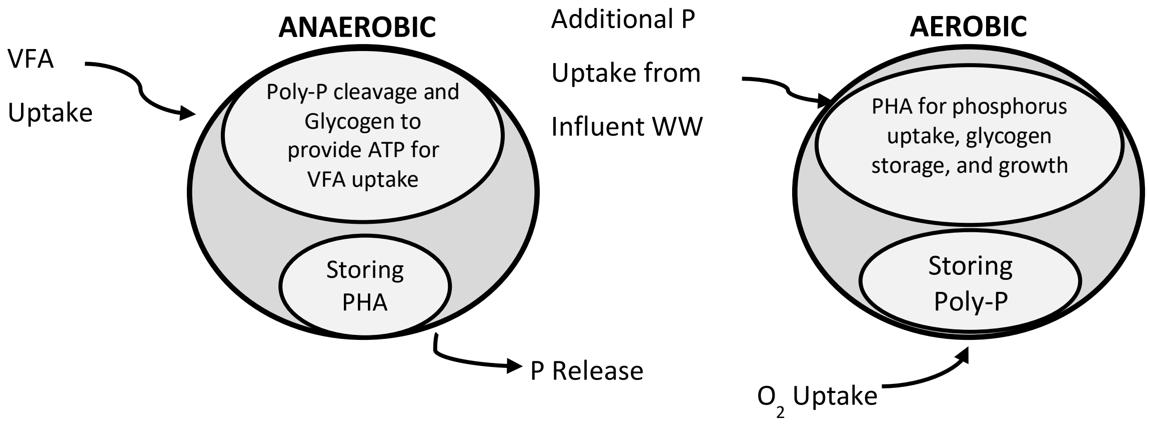


Figure 2.8: PAO metabolism illustration for EBPR

The introduction of RAS to the anaerobic zone can cause operational issues due to the presence of nitrate. Nitrate acts as a terminal electron acceptor, creating an anoxic zone which is not sufficient for inducing EBPR (Wentzel et al. 1990). Additionally, the reduction of nitrate to nitrogen gas requires carbon and has the potential to utilize a significant portion readily biodegradable carbon or VFAs in the influent – the latter may be disruptive to EBPR.

While some influent wastewaters contain sufficient VFAs for EBPR, many are VFA deficient. Primary sludge fermentation is a commonly employed method to produce additional VFAs, yielding a waste stream high in acetate and propionate. The fermentation of other waste carbon sources such as dairy manure or industrial wastes present additional potential VFA sources. Lastly, VFAs such as acetic acid can be purchased directly and added to the treatment process to induce EBPR.

2.3.2 Biogas and Renewable Energy Production

Biogas is a resource that can be produced at WRRFs via anaerobic digestion (AD). Typical AD-produced biogas contains roughly (by volume) 65% methane, 30% carbon dioxide, and traces of other gases such as hydrogen sulfide, water vapor, and hydrogen gas. The methane-rich biogas can be utilized as a resource in several different ways (e.g., electricity production; heat; vehicle fuel). At most WRRFs operating ADs, the biogas is often used solely for digester heating; excess biogas is typically flared, which serves only to prevent methane pollution into the atmosphere (Combined Heat and Power Partnership et al. 2011).

One approach to capture the energy value of biogas involves utilizing a combined heat and power (CHP) system; CHP systems serve to provide needed heating to anaerobic digesters, while concurrently producing electricity that can be sold to the grid or used to offset electricity use within the facility (Combined Heat and Power Partnership et al. 2011). Alternately, biogas can be purified to natural gas pipeline quality, termed gas-to-grid. However, gas-to-grid systems typically have poor environmental performance and high financial risk when compared to CHP systems (Mills et al. 2014). Another method of utilizing biogas is as compressed natural gas (CNG), which requires natural gas purification followed by compression. CNG is most commonly used for vehicle fleets; while CNG is theoretically an option, there are limited natural gas powered vehicles in the Twin Falls region, and thus CNG is not presently a feasible use for biogas. Gas-to-grid systems and CNG upgrading will not be investigated further in this research; the study instead focuses on CHP systems for biogas utilization.

Economic viability of biogas systems is a function of the quantity of biogas produced, as many processes that utilize biogas do not scale to fit small systems. The EPA suggests that CHP systems are not economically viable in WRRFs with influent flow rates of less than 5 mgd or less than 5 dry tons of solids per day (Combined Heat and Power Partnership et al. 2011). This is due to the high capital cost and poor scalability of the biogas utilization technologies as well as the high construction costs of anaerobic digesters. Additionally, the payback period for investment in CHP is largely a function of electricity prices; in areas with expensive electricity, the payback period can be as short as a few years, while in regions with low-cost electricity, it can be nearly impossible to achieve payback. Research has pointed toward anaerobic membrane bioreactors as a preliminary alternative to address issues such as scalability of digestion systems; however, there is much research to be done to mitigate the risk and uncertainty associated with the development of a new system (Smith et al. 2014, Puyol et al. 2017).

2.3.2.1 CHP Technologies

The most common technology for CHP is the internal combustion (IC) engine; microturbines, combustion turbines, and fuel cells are also used, but to a much lesser extent (Qi, Beecher and Finn 2013). Internal combustion engines have been widely adopted due to their versatility; they range from several kilowatts to around 5 megawatts. Operationally, IC engines have quick startup times and respond rapidly to operational changes with a relatively high turn-down ratio. The principal issues with IC engines are noise and the potential requirement for cooling if excess heat produced is not utilized (U. S. Environmental Protection Agency et al. 2007). Modern IC engines have electrical efficiencies of 37%-42% and thermal efficiencies of 35%-43% (Brown and Caldwell et al. 2010).

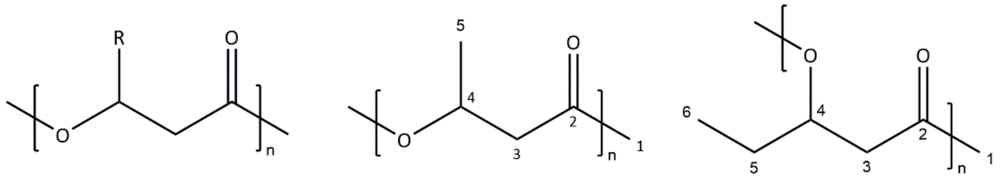
Fuel cells work by converting methane in the biogas to hydrogen gas using steam. Hydrogen enters the anode of the fuel cell where it dissociates into protons and electrons. Protons pass through the electrolyte into the cathode where they react with oxygen to produce water. Electrons flow from the anode into a wire, creating an electric current. Electrons then move to the cathode, contributing to the reaction by which protons and oxygen form water and steam, which is used for the methane conversion to hydrogen and as a heat source for co-generation. Compared to IC or turbine engines, fuel cells have much higher capital and operational costs. Fuels cells have higher electrical efficiency, but typically have a smaller capacity (Brown and Caldwell et al. 2010). Research on fuel cells is ongoing, but there is limited field experience with their operation.

Microturbines are suitable for small applications, up to 250 kW. Of the aforementioned alternatives, they have the lowest electrical efficiency, between 26% and 30%; thermal efficiency is from 30% to 37% (Brown and Caldwell et al. 2010). Due to their simpler design and fewer moving parts, microturbines have the potential for lower maintenance requirements.

Combustion turbines share much of their design with microturbines, but they have much higher energy production capacities as well as a slightly greater electrical efficiency due to their larger size. Electrical efficiencies are typically between 36% and 37%, with thermal efficiencies between 30 and 45% (Brown and Caldwell et al. 2010). One of their main advantages is the lack of a need for cooling. They are more suited to use in very large facilities and do require specialized maintenance.

2.3.3 Bioplastics

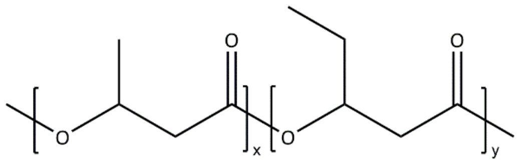
Polyhydroxyalkanoates (PHAs) are an intracellular form of carbon storage polymers synthesized by microorganisms (Kadouri et al. 2005). These polymers have characteristics and qualities similar to thermoplastics such as polypropylene or polyethylene, with the added advantage of biodegradability – making them a more environmentally-friendly type of plastic (Chen and Patel 2012). The structures of a general PHA monomer, as well most common monomers that appear in WRRF systems, are illustrated in Figure 2.9. Poly(3-hydroxybutyrate) (PHB) is known for its brittle properties; however, when combined with poly(3-hydroxyvalerate) (PHV) to form poly(3-hydroxybutyrate-co-hydroxyvalerate) (PHB-coV), PHB exhibits more desirable properties not unlike petroleum-based thermoplastics (Doi 1990).



Polyhydroxyalkanoate

Poly(3-hydroxybutyrate)

Poly(3-hydroxyvalerate)



Poly(3-hydroxybutyrate-co-3-hydroxyvalerate)

Figure 2.9: Structures of Common PHA Monomers

Commercial PHA production is currently realized using pure microbial cultures and synthetic substrates (typically based on corn sugar), requiring specialized processes that create a product cost up to three times higher than petroleum-based plastics (Digregorio 2009). In other words, commercial PHA production within a WRRF setting has yet to be realized.

Achieving economically sustainable PHA production is something that can potentially be realized at a WRRF through the utilization of mixed cultures of microorganisms already in the system that are fed carboxylic acid-rich wastewater. Carboxylic acids, which include acetic acid and 2-6 carbon volatile fatty acids (VFAs), can be readily produced through anaerobic fermentation of organic waste carbon sources (Rabinowitz and Oldham 1986, Romenesko 2017, Coats, Gregg and Crawford 2011a). Due to the resultant carboxylic acid speciation coupled with the microorganisms present in wastewater systems, PHBV will be the predominant type of PHA synthesized in a WRRF, with different amounts formed depending on the mixture of VFAs in the substrate.

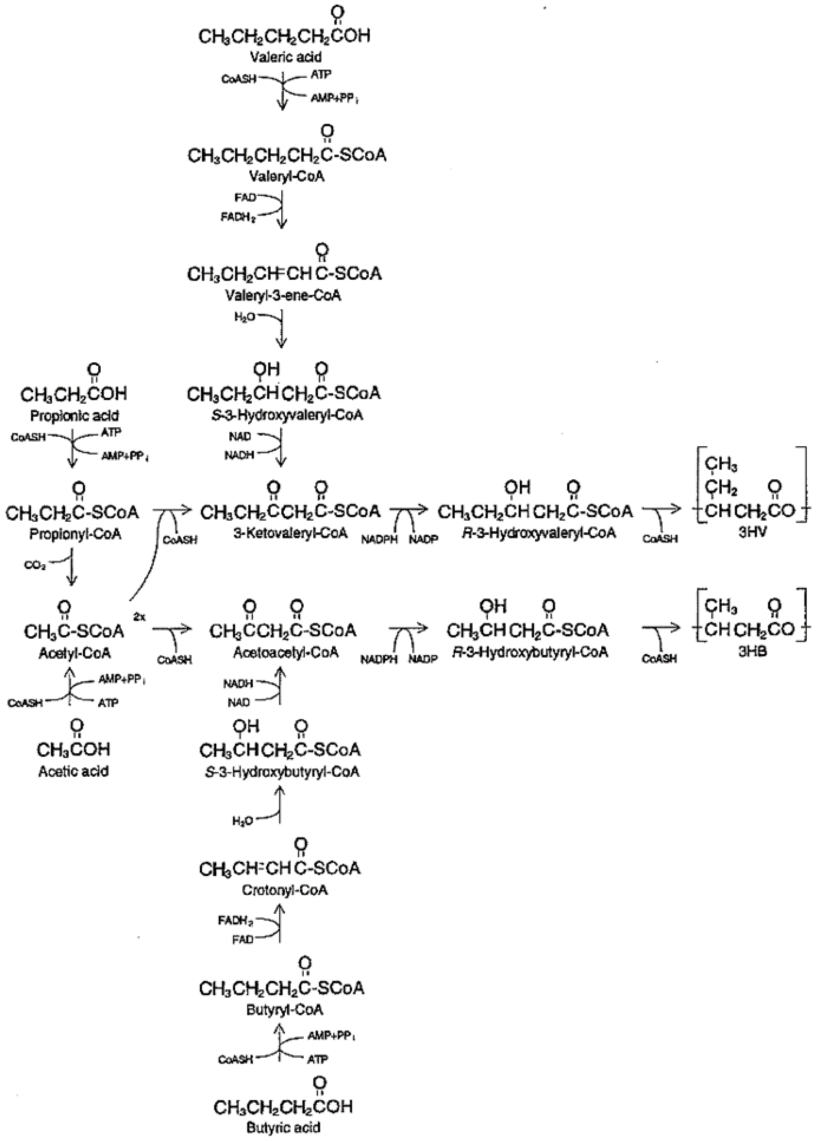


Figure 2.10: Metabolic Synthesis of PHA from common VFAs

In the context of a WRRF, it should be noted that PHA is synononmous with PHBV, with PHA the preferred nomenclature in this study as it is the more general term for this class of polymers. As illustrated in Figure 2.10 (adapted from Braunegg et al. 1998), odd number of carbons tend to produce PHV whereas even number of carbons yield PHB.

2.3.3.1 PHA Production Strategies

In current WRRF applications, PHA is produced via metabolic pathways associated with phosphorus-accumulating organisms (PAOs) during EBPR processes; VFAs are consumed anaerobically and stored as PHA, which is then utilized aerobically. To obtain the energy needed to drive the anaerobic uptake of VFAs, PAOs must hydrolyze their internal stores of polyphosphate for ATP, resulting in an increase in phosphorus concentration in the bulk solution. A cyclical anaerobic-aerobic process selects for PAOs in the WRRF MLSS, allowing them to multiply and ultimately consume all the phosphorus that is released under anaerobic conditions, as well as any additional phosphorus that may be present in the raw wastewater. On a cell dry mass basis, PHA accumulation within a PAO under these conditions generally does not exceed 3% (w/w basis; (Coats et al. 2011b, Probst 2016), which is not enough product to be viable for commercial application. While the WAS from EBPR processes can be used as a source for PHA-producing microorganisms that can be fed a VFA rich substrate, research conducted in this study (Appendix A) suggests that the intracellular concentration of PHA attainable using this method is still not optimal for PHA production.

Aerobic dynamic feeding (ADF) is an alternative engineered strategy for achieving PHA production using mixed microbial cultures and carboxylate-rich wastewater. An engineered ADF environment induces PHBV synthesis associated with feast-famine conditions (Lemos, Serafim and Reis 2006). The feast-famine conditions create metabolic stress for the MMC that ultimately selects for microorganisms that are able to produce excess quantities of PHBV, which they use for energy upon depletion of soluble substrates. The metabolic feast-famine response is generally illustrated in Figure 2.11; as shown, bulk solution carboxylate concentrations decrease as microorganisms consume the substrate, which is stored intracellularly as PHA. The ADF process can be used to enrich for PHA-producing microorganisms because PHA provides a critical source of electrons and carbon during famine conditions, allowing for cell growth and cell maintenance. The ADF process selects for PHA-producing microorganisms since they maintain a competitive advantage over ordinary heterotrophic organisms, which have less potential for survival during the famine period.

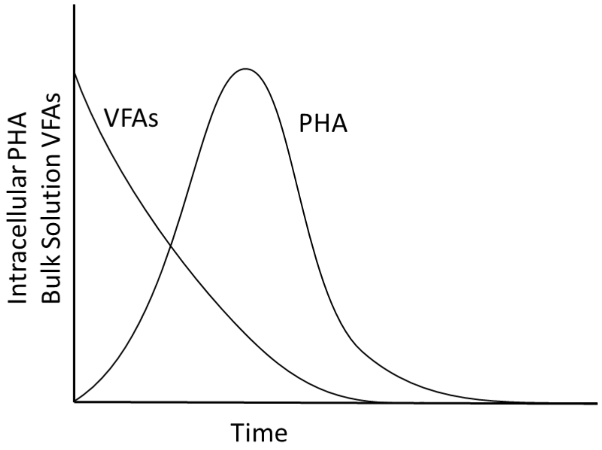


Figure 2.11: Feast-Famine PHA Synthesis Illustration

Commercial PHA production at a WRRF can be achieved by enriching for a mixed microbial consortium (MMC) of PHA-producing organisms using ADF conditions. The envisioned system would be separate from the main WRRF and would consist of an enrichment reactor coupled with a production reactor. The enrichment reactor would be operated in batch mode on a 24-hour cycle, once daily receiving a pulse of VFA rich substrate (Coats et al. 2011b). The waste from this enrichment reactor would contain high concentrations of PHA-producing microorganisms, which under extended feast periods in a production reactor, could be leveraged to produce commercially viable quantities of PHA. Figure 2.12 illustrates how this process would be visioned, with raw dairy manure as the initial substrate. Dairy manure would be mixed with water to form a pumpable slurry that would be sent to a fermenter to be converted into a VFA-rich substrate. Solids separation would separate the VFA-rich liquid stream and residual fermented solids. Residual solids from this process would be anaerobically digested; the VFA-rich substrate would then be used to drive PHA production, as it is rich in nutrients needed for growth and capable of being readily metabolized to PHA. To induce feast-famine conditions in the “Enrichment” stage, substrate would be fed in a single, short duration pulse to a sequencing batch reactor (SBR). The VFAs would be rapidly consumed as part of the "feast" metabolic response. During the remaining reaction time, famine conditions would be present – no VFA-rich substrate would remain and generally nominal quantities of readily biodegradable carbon would be available in the wastewater. The microbes that stored VFAs as PHA would now have an internal carbon storage polymer, allowing them to survive under the famine conditions. This process configuration ultimately would enrich for an MMC with a high proportion of PHA-storing microorganisms (Coats, Brinkman and Lee 2017). This completely mixed reactor would then be volumetrically wasted, and the microbes present would serve as an inocula for the second stage, the "Production" reactor. In this stage of the process, VFA-rich substrate would again be pulse fed to the MMC, based on substrate reaction kinetics, to ensure a sustained 'feast' response. By feeding a high quantity of VFAs, high PHA quantities would be realized. After this stage, the solids would be separated for further processing, and the

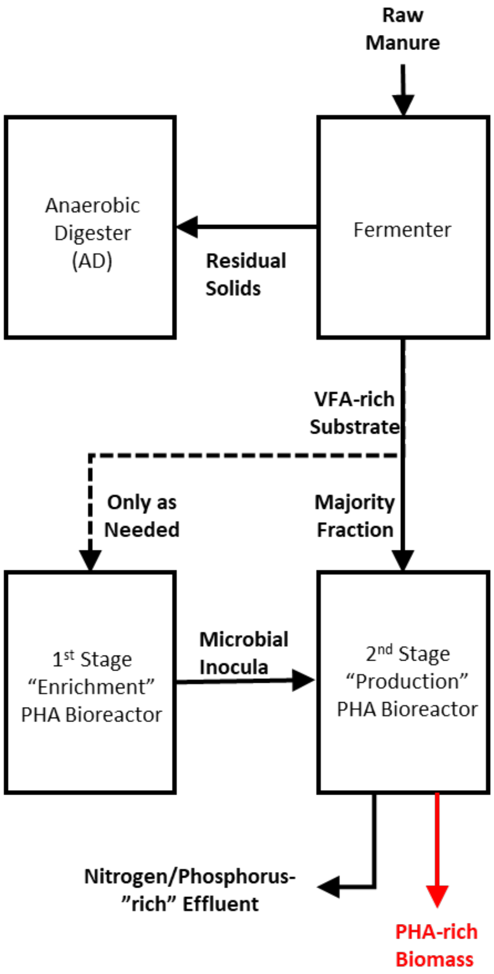


Figure 2.12: Process Schematic of PHA Production from Dairy Manure

liquid effluent, rich in phosphorus, ammonia, and nitrate, would be returned to the WRRF for further treatment.

3. Methodology

The goal of this research was to evaluate and investigate the feasibility of implementing resource recovery at the Twin Falls, Idaho WRRF, leveraging proximate agro-industrial wastes integrated within the existing municipal wastewater stream. Feasibility analyses focused principally on environmental life cycle assessment, and secondarily on economics. This chapter outlines and describes the methodologies employed to develop necessary data to complete the feasibility analyses.

As a first step in completing this research, a model of the Twin Falls WWTP was created using Dynamita's SUMO wastewater process modeling software (hereinafter referred to as 'SUMO') based on current infrastructure; the model, which is highly parameterized, was then calibrated using field-collected data for both summer and winter operating conditions. All biological wastewater treatment models, including SUMO, are principally parameterized on COD, nitrogen, and phosphorus fractions; while some model parameters were characterized as part of the WWTP assessment, Twin Falls WWTP staff do not characterize numerous constituents related to process modeling, thus certain assumptions had to be made in order to calibrate the SUMO model of the existing WWTP. The influent wastewater to Twin Falls also contains high quantities of industrial wastewater, thus parameter assumptions were considered WWTP-specific; such assumptions might be less appropriate for an influent consisting solely of municipally derived wastewater.

After the model was calibrated to reflect actual WWTP performance, alternate configurations to achieve resource recovery – thus transitioning to a WRRF – were implemented into the model and optimized to produce results consistent with the literature and experience of others. The calibration process allowed for the quantification and assessment of any process and/or operational changes on the treatment performance and effluent quality associated with differences between scenarios in the internal recycle (e.g., addition of ammonium from dairy manure) and the removal of certain constituents in the form of resource recovery (e.g. phosphorus removal via struvite precipitation).

The LCIA was performed as described in section 2.2.2 to quantify flows crossing the system boundary in each alternate scenario. SUMO was used to inform on effluent quality, biosolids and struvite production, and biogas production. PHA production was modeled using the DAIRIEES model (Guillen 2017), due to a lack of a formal process metabolic model; the recycle of nutrients associated with PHA production was integrated into the WRRF and included within SUMO. Additional research was performed to provide data regarding fertilizer production and the associated replacement of a fertilizer with biosolids and struvite based on nitrogen and phosphorus content. Electricity production for the region was also considered with regards to offsets from CHP and any increases in electricity usage at the facility associated with additional pumping, etc.

3.1 Description of Current Twin Falls WWTP and System Operations

Twin Falls discharges treated effluent into the Snake River in accordance with their NPDES permit, with limitations on BOD_5, TSS, and ammonia. The permit does not limit total nitrogen including nitrate, nor does it contain stringent phosphorus limits – the latter being anticipated in a future NPDES permit. The current Twin Falls WWTP is operated in a manner consistent with industry norms – with processes designed to target the oxidation of BOD and ammonia as well as removal of solids. Figure 3.1 illustrates the current process.

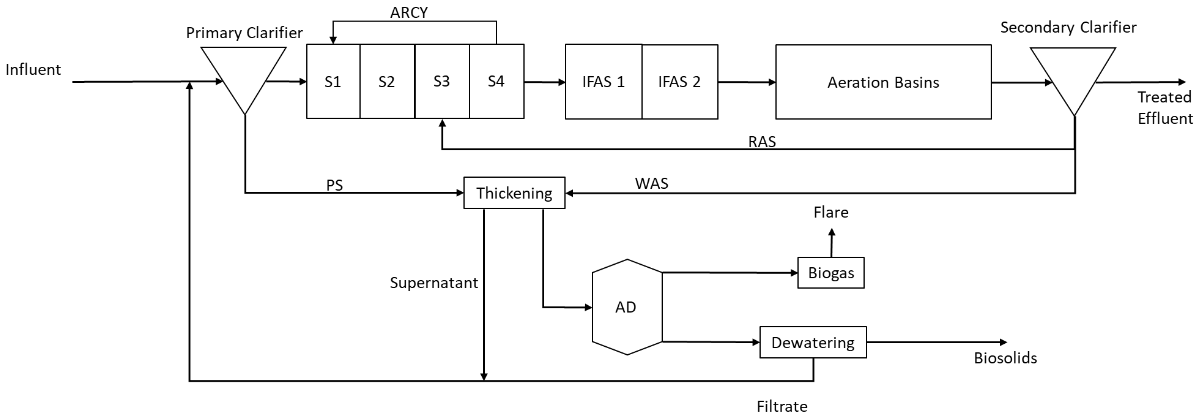


Figure 3.1: Twin Falls WWTP Current Operations Process Schematic

Raw wastewater enters the facility and undergoes preliminary treatment including screening and grit removal, following by primary clarification. The facility operates three primary clarifiers; however, due to the hydraulics of the system, either primary clarifiers 1 and 2 are operational, or the larger primary clarifier 3 operates alone. Primary sludge is pumped from the primary clarifiers continuously. Primary effluent enters the first selector basin (S1). There are four unaerated selector basins in series, operated in a manner that mirrors that of the Virginia Initiative Process (Metcalf and Eddy 2013). Return activated sludge (RAS) enters into the third selector (S3) where it is denitrified. In the fourth selector (S4), an anaerobic/anoxic recycle pump (denoted by Twin Falls as ARCY) pumps mixed liquor back to S1 this allows the primary effluent to blend with denitrified mixed liquor, providing anaerobic conditions necessary for EBPR. The selector basins were constructed as part of a recent WWTP expansion that also includes the next stage of treatment: Integrated Fixed film Activated Sludge (IFAS).

Downstream of the selector basins there are two IFAS trains, each with two basins in series. The basins include small plastic media that supports biofilm microbial growth, as well as mixed liquor rich in microorganisms. The fixed film aspect of the system helps to maintain nitrification in the basins as well as absorb shock loads associated with pretreatment failures at industrial wastewater producers. Following IFAS there are four aeration basins in parallel. These basins are long and narrow (i.e., plug flow), and include the capability to turn off aeration in the middle section so that it can serve as a “swing zone,” promoting denitrification. Mixers are present to maintain complete mixing in the absence of aeration. Since the construction of the IFAS basins, only two aeration basins have been needed to maintain treatment and permit compliance, meaning the facility likely has significant excess treatment capacity. Aeration is accomplished with ceramic fine-bubble diffusers, with low-pressure air provided by four multi-stage centrifugal blowers rated for a total capacity of 30,000 scfm. Aeration basin effluent mixed liquor is settled out in one of four secondary clarifiers, with the settled sludge either returned as RAS or wasted as WAS. The secondary clarifiers utilize Stamford baffles to prevent solids overflow; the two new clarifiers utilize the COPS system (WesTech Engineering, Salt Lake City, UT, USA) for solids removal, with spiral scrapers on the older clarifiers. Secondary effluent undergoes UV disinfection before discharge to the Snake River.

WAS and primary sludge (PS) are combined and co-thickened to approximately 6% solids (weight basis) using a gravity belt thickener (GBT) before realizing treatment via anaerobic digestion. Anaerobic digestion of PS and WAS occurs using two completely mixed mesophilic anaerobic digesters operated in series. Biogas is collected and used in boilers to maintain temperature within the digesters; excess biogas is flared. Class B biosolids are produced and used at farms in the area.

Historically, Twin Falls has been able to comply with its NPDES permit without issue. Table 3.1 shows average influent and effluent values for several parameters collected by facility staff from October 2015 to June 2018.

Table 3.1: Twin Falls Influent and Effluent Characteristics

Parameter	Influent	Effluent	Removal %
BOD_5 (mg/L)	364.2	4.6	99%
TSS (mg/L)	263.41	6.92	97%
TP (mgP/L)	10.72	5.04	53%
NH_4 (mgN/L)	42.86	0.16	99.6%
TKN (mgN/L)	59.15	2.67	95%
NO_3 (mgN/L)	0.25	15.9	N/A
pH	8.24	8	N/A

3.2 Proximate Waste Streams

Twin Falls is uniquely positioned to achieve resource recovery in large part due to the proximate waste streams, several of which are already received at the WWTP. Table 3.2 shows data collected by the city for each of its permitted industrial dischargers.

Table 3.2: Industrial Wastes Treated at WWTP

Entity	**Flow (mgd)**	**BOD_5 (mg/L)**	**TSS (mg/L)**	**FOG (mg/L)**
AmeriPride	0.13	178.3	70.4	110.5
Chobani	0.80	145.0	164.2	24.2
City of Kimberly	0.30	302.6	285.1	24.6
Clif Bar	0.01	659.2	444.1	23.9
Eagle Eye	0.01	143.0	410.4	2.8
Glanbia	0.51	601.5	244.2	89.3
Independent Meat	0.11	84.3	91.6	17.3
KapStone	0.01	401.7	197.0	3.1
Lamb Weston	1.76	2305.1	510.5	93.2
Total	3.65	1268.0	349.4	69.3

Of note, several of these waste streams are pretreated prior to discharge into the city's sanitary sewer collection system. Chobani owns and operates a UASB; the values shown in Table 3.2 are based on effluent from the UASB discharged into the city's sanitary sewer collection system. Lamb Weston also has a UASB pretreatment facility which is operated by the city. The pretreatment of waste in UASBs converts much of the carbon in the wastewater to methane, which is burned and ultimately released as carbon dioxide. This high strength carbon rich waste has a high potential for use at the plant in terms of driving EBPR or PHA production. Lastly, Clif Bar waste is pretreated in a moving bed biofilm reactor (MBBR) which is operated by the city. Of note, Glanbia does not pretreat its waste; rather they operate a surge tank to release high strength waste slowly such that the city's WWTP can accommodate the load.

Beyond waste streams already discharged into the city's sanitary sewer system, TASCO has a sugar beet production facility in the Twin Falls area; sugar beet wastewater is exhibits high concentrations of VFAs, with low concentrations of nitrogen and phosphorus. The facility operates under its own discharge permit, land applying effluent. While not discharged to the current WWTP, TASCO does have a unique waste that could be leveraged in terms of resource recovery. Characteristics and flows of TASCO's flume wastewater are listed in Table 3.3.The concentration of VFAs listed is based on the testing of one sample. Previous tests have indicated a higher VFA concentration closer to 5,000 mg/L, which would be of a higher value. The results of this study are thus highly dependent on this VFA concentration, which is subject to a high degree of uncertainty.

Table 3.3: TASCO Wastewater Characteristics

Avg. Flow, gal/day	**40,000**
COD, mg/L	80,000
sCOD, mg/L	77,770
VFAs, mg/L	1,360
NH_4, mgN/L	20
PO_4, mgP/L	0.2
TSS, mg/L	1,720
VSS, mg/L	1,580

Lastly, southern Idaho is the heart of the Idaho dairy industry; many large dairies are in close proximity to Twin Falls. While research on specific dairy facilities was not conducted, for the purpose of this study it was assumed that a re-configured WRRF would have ready access to manure equivalent to that produced by a 5,000 head dairy. From this facility, dairy waste, including manure, could be trucked in and fermented to produce a high concentration of VFAs – useful in driving EBPR metabolisms or the carbon intensive PHA production process. Scenarios studied that utilize dairy manure consider its use for resource recovery – they do not consider its importation for the sole purpose of treatment.

Of the available proximate waste streams, four sources were selected to be further studied in integrating resource recovery into the Twin Falls WWTP: dairy manure, TASCO, Chobani, and Lamb Weston. Of the existing industrial dischargers, Chobani and Lamb Weston currently realize the greatest loss in resource recovery potential due to their usage of UASBs. This study proposes the conversion of these two UASB facilities to fermenters; this would serve to hydrolyze the organic material present in the industrial wastewater, yielding a mixture of VFAs. Due to the high flow rate of these dischargers, trucking in the waste in concentrated form is not feasible – thus discharge of the VFA rich wastewater into the collection system must take place. TASCO waste and dairy manure can be trucked in and handled on site.

3.3 Wastewater Process Modeling

Formal and universally adoptable mathematical modeling of activated sludge systems began with an International Water Association (IWA) task force in 1982, ultimately leading to the creation of a suite of biological WWT model structures known as the suite of Activated Sludge Models (ASMs) (Henze et al. 2000); collectively, the ASM's provide a common framework for the modeling of conventional secondary treatment activated sludge systems. The ASM framework utilizes a matrix format, referred to as a Gujer matrix, to present models; as illustrated, each column represents a state variable and each row represents a process. The last column in the matrix presents the process rate equation, by which each stoichiometric element in the row is multiplied. Process rate equations for microbial growth processes are principally based on empirically observed Monod kinetic relationships; as such, ASM models are considered "black box" in that they do not represent actual process metabolisms. To develop and execute the complete model, the matrix is solved as a mass balance in each column.

The first iteration of activated sludge models, ASM1, was released in 1987 and included carbon and nitrogen removal. ASM utilized 19 parameters to model 8 separate processes: aerobic heterotroph growth, anoxic heterotroph growth, aerobic autotroph growth, decay of heterotrophs and autotrophs, ammonification, and hydrolysis of organic nitrogen and carbon. ASM1 had a few known limitations at the time, including a lack of pH modeling, the exclusion of nutrients toward cell growth, and a lack of EBPR (Carlsson 2010).

With ASM2, published in 1994, ASM1 was expanded to include EBPR; subsequently ASM2 was further modified in 1999 as ASM2d, which modeled new understanding of the role of PAOs during denitrification. ASM2d also takes advantage of more modern computing capabilities – many simplifications were implemented into ASM1 to ease computations. For example, lysis of different heterotrophs, autotrophs, and PAOs is all modeled separately.

ASM3 was also created to be modular in nature – its base model is simple but allows for the integration of separate modules that include EBPR or pH modelling, for example. ASM3 was expanded to include endogenous respiration – the additional oxygen demand to oxidize new substrate created by cell lysis.

In 2002, IWA released Anaerobic Digester Model No. 1 (ADM1), which models the biological processes that occur in anaerobic digesters. This is a separate model from the ASMs, but it is similar in its framework. ADM1 and ASM1-3 use the same state variables so that the models can exchange information and be operated jointly, combining activated sludge and anaerobic digestion.

3.3.1 Model Selection

Research herein utilized the SUMO process modeling package by Dynamita. In addition to the ASM model structures, SUMO includes several in-house biological models, which vary slightly when compared to the IWA models, but are fundamentally based on a mass-balance approach to wastewater treatment system modeling. The full package simulator includes modeling of settling, pH, precipitate formation, gas transfer, etc., which can be combined with many biological models to produce full plant simulation.

SUMO has three full plant models, the latter two of which include anaerobic digestion: Mini_Sumo for oxygen uptake rate (OUR) and sludge production, Sumo1 for one-stage nitrification and denitrification, and Sumo2 for two-stage nitrification and denitrification. SUMO software also has the ability to import and utilize other models, such as the ASMs.

Sumo1 uses 61 parameters to model 70 different processes. Cell growth, maintenance, and decay for aerobic, anaerobic, and anoxic environments are included for many different types of organisms including ordinary heterotrophic organisms (OHOs), glycogen accumulating organisms (GAOs), nitrifying organisms, hydrogenotrophic methanogens, and acidoclastic methanogens. The model also includes hydrolysis, fermentation, mineral precipitation reactions (e.g., struvite), chemical phosphorus removal, and a gas transfer model.

For EBPR, SUMO utilizes the Barker-Dold model (Barker and Dold 1997), which is similar to ASM2d in that PAOs can uptake soluble phosphorus in anoxic conditions, thus contributing to denitrification. Both models use a factor to reduce the proportion of PAOs capable of using nitrate as a terminal electron acceptor; to-date no research has elucidated details and characteristics on denitrifying PAOs, and most models either assume that a fraction of PAOs can denitrify (i.e., dPAOs) or that the PAOs operate at fractional efficiency with nitrate as an electron acceptor, relative to oxygen. The Barker-Dold model also accounts for differences in PAO metabolism under aerobic versus anoxic conditions – modeling lower phosphorus uptake under anoxic conditions for the same PHA utilization when compared aerobically, something that ASM2d does not model.

Sumo2 was used in conducting research in this thesis. Sumo2 was chosen because the IFAS basins contain several layers of biofilm in which oxygen limiting conditions are likely – thus contributing to the production of some nitrite that is not immediately oxidized to nitrate.

3.3.2 Model Constituents and Parameters

Carbon is one of the most important parameters in any wastewater treatment process; it is commonly measured with two different tests: five-day biochemical oxygen demand (BOD_5) and chemical oxygen demand (COD). While regulations such as NPDES permits are written to the BOD_5 standard, practitioners and researchers tend to prefer COD, as this measurement represents a true estimate of electrons available for biochemical oxidation-reduction reactions. BOD_5 testing is slow, requiring five days to obtain a value that measures the oxygen used to oxidize the sample biochemically. COD tests work using dichromate

(Cr_2O_7), which oxidizes all oxidizable substances present in a sample. While COD measures a much higher value of oxidization than occurs biochemically, it is a quick and reliable test. Conversion factors between BOD and COD are often experimentally determined for specific wastewater so that parties of interest have a quick method of estimating the BOD of a sample. Additionally, products of known chemical composition (e.g., VFAs) can be stoichiometrically converted to a COD value, as all the carbon present in a molecule will be converted to carbon dioxide.

Biological wastewater treatment, and corresponding process models, depend on different fractions of COD such as soluble COD (sCOD) and readily biodegradable COD (rbCOD). rbCOD is difficult to truly measure and is often substituted with filtered flocculated COD (ffCOD), which can be determined via a chemical method that uses zinc sulfate to remove non-readily biodegradable material. ffCOD has been shown to be an effective surrogate for rbCOD, providing consistent and reproducible results (Mamais, Jenkins and Prrr 1993). VFAs also contribute to the COD of a wastewater and are easily converted to a COD value. Figure 3.2 (Melcer and Water Environment Federation 2003) illustrates the different COD fractions contained in wastewater.

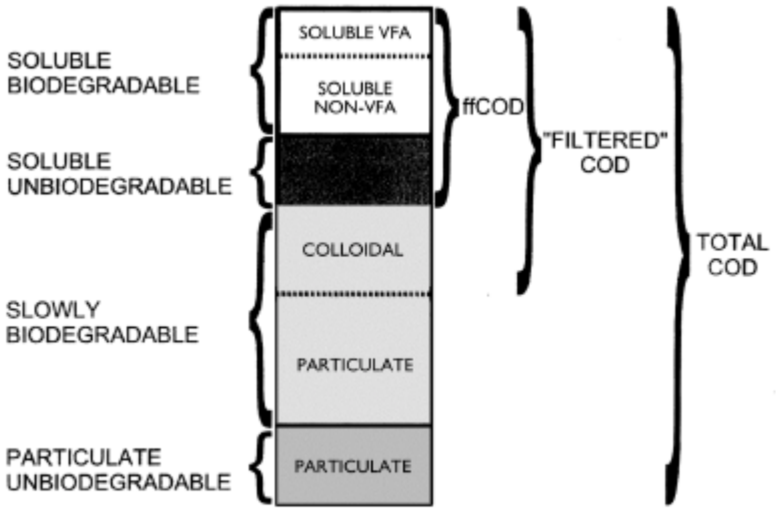


Figure 3.2: Schematic Representation of COD components for Municipal Wastewater (Source: Melcer et al. 2003. Reprinted with permission. © Water Environment Research Foundation

Nitrogen is measured in influent wastewater as total Kjeldahl nitrogen (TKN), which includes ammonia (NH_3) and organically bound nitrogen. Phosphorus in influent is measured as total phosphorus, consisting largely of orthophosphate, which is considered to be the same as soluble reactive phosphorus. Additional influent phosphorus is in the form of organically bound phosphorus. Organically bound phosphorus and nitrogen are separated first into biodegradable and nonbiodegradable categories, and each of those into soluble and particulate. These parameters are very difficult to measure separately, so typical fraction values are generally used. The soluble nonbiodegradable portions are of importance, as they are not affected by biological processes or by settling, leaving them as effluent into receiving waters.

Included in the SUMO software package is an influent tool which uses Microsoft Excel to estimate influent parameters and fractions based on known wastewater characterization data, comparing results to typical ranges. Data obtained from Twin Falls WWTP operational staff indicate a high fraction of soluble nonbiodegradable COD, which was estimated by measuring effluent sCOD from the WWTP containing a negligible amount of BOD, indicating that while the biochemically oxidizable substances have been removed, some soluble nonbiodegradable constituents present in the influent are unchanged throughout the treatment processes. Soluble nonbiodegradable COD accounts for roughly 35% of sCOD, which is higher than the typical range of 12% to 15% - due to the large amount of industrial waste treated at the facility.

Table 3.4 summarizes the average and sample standard deviation of influent flows, loads, and fractions, based on eight sets of data collected by facility staff. All ionic components in the process model were left at default values, as measurements are not routinely taken at the facility. The full table including default unchanged values, and individual measurements from each dataset can be found in Appendix B.

Table 3.4: Typical Twin Falls WWTP Influent Wastewater Characteristics

Influent components	**Average**	**S.D.**
Flow rate, mgd	7.9	0.3
Total COD, mgCOD/L	987.3	187.6
TKN, mgN/L	57.6	14.0
Total phosphorus, mgP/L	10.7	1.2
Nitrite and nitrate (NOx), mgN/L	0.13	0.1
pH and alkalinity		
Alkalinity, mg$CaCO_3$/L	667.5	71.5
pH	7.9	0.2
Influent fractions		
Fraction of VSS/TSS	86.5	4.1
Fraction of filtered COD in TCOD	54.4	11.8
Fraction of flocculated filtered in TCOD	41.5	12.4
Fraction of VFA in filtered COD	1.0	2.9
Soluble nonbiodegradable organic fraction (SU) in sCOD	40.8	16.9
Particulate nonbiodegradable organic fraction (XU) in TCOD	9.4	3.9
OHO fraction in TCOD	4	1.1
Endogenous product fraction (XE) in TCOD	18.1	3.7
Colloidal nonbiodegradable organic fraction (CU) in colloidal COD	17.5	3.8
Fraction of NH_x in TKN	54.5	10.7
Fraction of PO_4 in TP	60.3	6.1
Fraction of N in SB	3.4	1.2
Fraction of N in XU	0.9	0.2
Fraction of P in SB	0.7	0.4
Fraction of P in XU	0.1	0.0

The data presented in Table 3.4 indicates that many influent constituents are comparable to values typically associated with 'high strength' wastewater (Metcalf and Eddy 2013). COD, for example, is listed in Metcalf and Eddy as high strength with a concentration of 1,016 mg/L. Twin Falls sees influent COD concentrations around this number, indicating its high strength. Furthermore, influent total phosphorus is considered high strength at a concentration around 11 mgP/L, and Twin Falls sees an average of 10.7 For ammonia, Twin Falls experiences an average of about 31.4 mgN/L, which falls between the medium strength and high strength concentrations of 20 and 41 respectively. Influent TSS concentration to Twin Falls averages 261 mg/L, which compares to a medium strength concentration of 195 mg/L, and a high strength concentration of 389 mg/L. Lastly, the average influent BOD_5 concentration of 432 mg/L is higher than the typical high strength value of 400 mg/L. Overall, Twin Falls experiences higher concentrations of wastewater constituents than is typically seen in domestic wastewater.

Temperature data was provided by Twin Falls facility staff. While WWTPs often see wide temperature swings between summer and winter months, Twin Falls sees a more consistently warm temperature year-round, likely due to the large volumes of industrial wastewater discharged into the sanitary sewer collection system. As such, a separate model was not calibrated to differentiate between different seasons. Table 3.5 details both the influent and in-basin temperatures for 2017.

Table 3.5: Wastewater Temperatures

Month	**Influent Temperature, °C**	**Basin Temperature, °C**
January	14.5	21.9
February	13.2	21.1
March	13.5	20.1
April	15.5	21.8
May	19.4	25.9
June	21.8	26.0
July	24.6	27.2
August	21.8	27.5
September	19.0	25.2
October	13.9	25.5
November	13.4	23.7
December	11.9	20.9

3.3.3 Model Calibration

Model calibration was performed consistent with guidance and recommendations of Melcer et. al. (Melcer and Water Environment Federation 2003). Physical plant data (e.g., basin dimensions), operational data (e.g., recycle flow rates, DO concentrations), and influent loading data was provided by facility staff. Once this data was input, a simulation was performed to ascertain which kinetic and stoichiometric model parameters required adjustment. A tiered approach was taken to adjusting parameters; one parameter was adjusted at a time to ensure that it did not interfere with other parameters that required adjusting. The simulation was re-run after each parameter adjustment until the error comparing the model predicted data to the field collected data was minimized. Table 3.6 illustrates the analytes and sampling location of data points to be compared between known data sets collected by operators. These points were selected based on data already collected that allowed for a level of detail sufficient for accurate modeling.

Table 3.6: Datapoints used for Model Calibration

Analyte / **Location**	**TSS**	**VSS**	**COD**	**sCOD**	**ffCOD**	**BOD_5**	**TKN**	**NH_4**	**NO_3 + NO_2**	**TP**	**PO_4**
Influent	x	x	x	x	x	x	x	x	x	x	x
Primary Effluent	x	x	x	x		x		x			x
S1				x				x	x		x
S3				x				x	x		x
IFAS 2A	x	x		x				x	x		
IFAS 2B	x	x		x				x	x		
IFAS Effluent				x				x	x		x
Middle Aeration Basin	x	x		x				x	x		x
Aeration Basin Effluent				x				x	x		x
Secondary Effluent	x	x	x	x	x	x	x	x	x	x	x

Initial simulations after input of physical plant characteristics showed several differences in predicted model output versus measured data. For example, the model exhibited higher levels of EBPR activity, complete nitrification in the IFAS basins, and rapid denitrification of RAS. EBPR in SUMO was induced by high levels of fermentation of rbCOD to VFAs in the anaerobic basins – something not consistent with field data. To mitigate this discrepancy, fermentation growth rates for OHOs were nullified; however, to avoid effects on AD modeling, a local parameter for the OHO growth rate was separately set for the anaerobic digester, ensuring proper AD modeling.

The IFAS operations also required significant changes in the model to reflect what is realized at the facility. The diffusion factor for half saturation coefficients in biofilms was increased to reflect the lower biological activity associated with higher half saturation values. The number of biofilm layers was modeled as two, which does not include bulk solution. Additionally, the IFAS basins were modeled with a bulk solution DO of 1.25 mg/L, which is significantly lower than the 5 mg/L that typically observed at the facility. The use of this tactic to ensure IFAS model calibration was not unexpected, as oxygen transfer is a common engineering challenge in modeling biofilms (Takács et al. 2007). The lower modeled DO in bulk solution solved what appeared to be an issue with oxygen transfer in the biofilm, as SUMO predicted higher DO levels in the outer biofilm layer, with near anaerobic conditions on the inner layer – something that was not able to be measured analytically. As evidence of this phenomena and the need to reduce bulk solution DO, the amount of nitrate measured in bulk solution was lower than the quantity of ammonia introduced – indicating some simultaneous nitrification-denitrification (SND) activity, which under the lower DO SUMO then modeled correctly. Indeed, SND has been observed to occur in biofilms with low DO environments (Barnard, Dunlap and Steichen 2013).

To mitigate the slower nitrification rate observed at the facility, the half saturation coefficient for AOB growth on ammonia was increased plant-wide to values within a typical range as reported by Hauduc et. al. (Hauduc et al. 2010); this allowed for ammonia profiling throughout the plant to be consistent with measured data.

Denitrification of RAS was initially estimated by SUMO to be very rapid, evidenced by the near zero predicted nitrate concentration in the third selector basin which receives RAS as contrasted with the typical value measured at the facility of 2.4 mgN/L. As the facility does not operate any mainstream denitrification, it might be expected the microbes present are not adapted to using nitrate as a terminal electron acceptor, thus realizing a significant anoxic growth reduction. The default value in SUMO for this parameter for OHOs of 0.6 was lowered to 0.1 to reflect that nitrate is not reduced as quickly as the model suggests. Many studies calibrate this value in the context of a mainstream denitrification (Melcer and Water Environment Federation 2003, Hauduc et al. 2010); as Twin Falls operates with BOD removal and nitrification only and does not denitrify, this value is of little consequence.

Appendix C details the calibrated model ultimately used in this study, comparing SUMO predicted values with the average and standard deviation of operator collected data. Melcer et. al. recommends that model predicted values differ between 5% and 20% from predicted values for steady state solutions (Melcer and Water Environment Federation 2003). However, due to the high variability in many of the field collected data points, a more statistically valid method of confirming the model predictions was selected. Specifically, to determine if the SUMO estimate was statistically the same as the measured data, hypothesis testing was performed using a two-sided t-test with an alpha of 0.05. The results of this hypothesis testing are listed in Table 3.7.

Table 3.7: Hypothesis Testing of SUMO calibration

Analyte	**Number Rejected**	**Total**	**Percent Rejected**
TSS	0	6	0%
VSS	0	6	0%
COD	1	3	33%
sCOD	3	9	33%
ffCOD	0	2	0%
BOD	0	3	0%
TKN	0	2	0%
NH_4	1	9	11%
NO_3+NO_2	2	8	25%
TP	0	2	0%
PO_4	2	9	22%
Total	9	59	15%

A total 15% of points estimated by SUMO were not statistically the same as the measured data. Soluble COD had the highest number of points that fell outside of the range of being statistically significant; this is likely due to a combination of the estimation of COD fractions of the influent wastewater and the high quantity of soluble unbiodegradable COD seen at the facility due to its high industrial wastewater fraction. Ammonium and phosphorus datapoints differed from measured data when compared to the primary effluent. This is likely due to the time of sampling falling during a period when the dewatering filtrate was not being pumped back to the headworks of the facility.

More recent data provided by Jacobs (not included in this study) indicated that these ammonium and phosphorus concentrations are typically higher than was measured during the 2015 sampling events. The profile of phosphorus at the facility is illustrated in Figure 3.3 as a box plot; the red line represents the SUMO predicted value, the thick black line represents the average value of the field collected data, the thin black lines at the bottom and top of the box represent the lower and upper 25% of the collected data respectively.

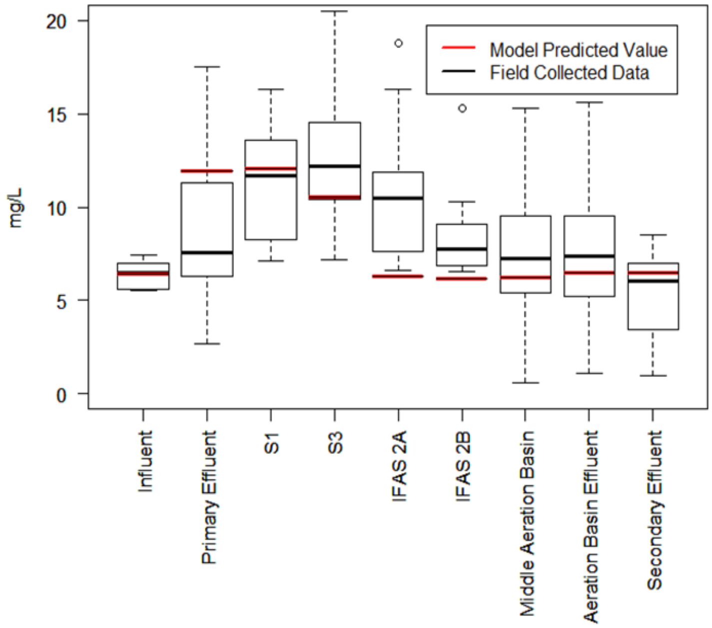


Figure 3.3: Phosphate Profile

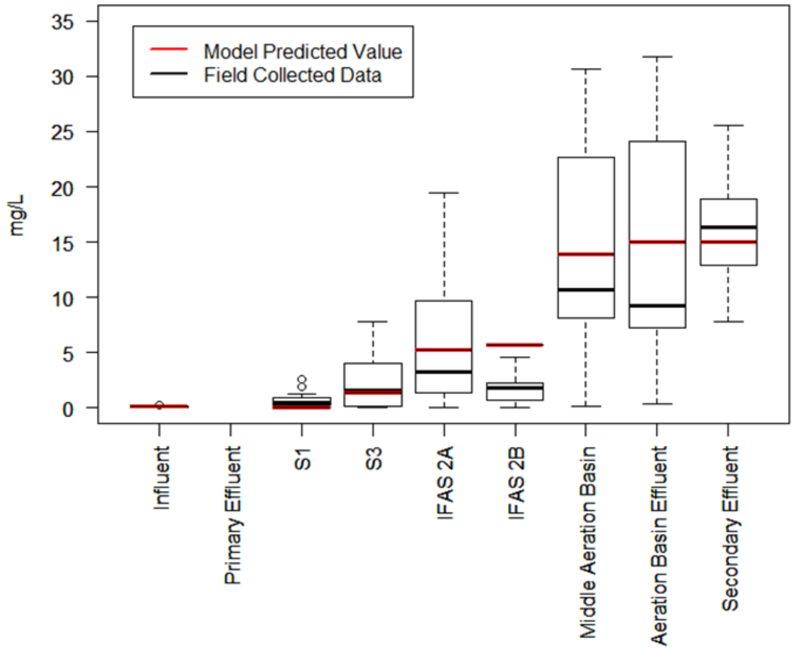


Figure 3.4: Nitrate Profile

Calibration of nitrate also proved difficult due to the IFAS basins performing some simultaneous nitrification and denitrification, likely due to the deep anoxic biofilm layers containing nitrate and little to no oxygen. Figure 3.4 shows a boxplot of nitrate data, which helps illustrate how field collected data shows a decrease in nitrate between the two IFAS basins. Figure 3.5 is a similar boxplot, with data for ammonia.

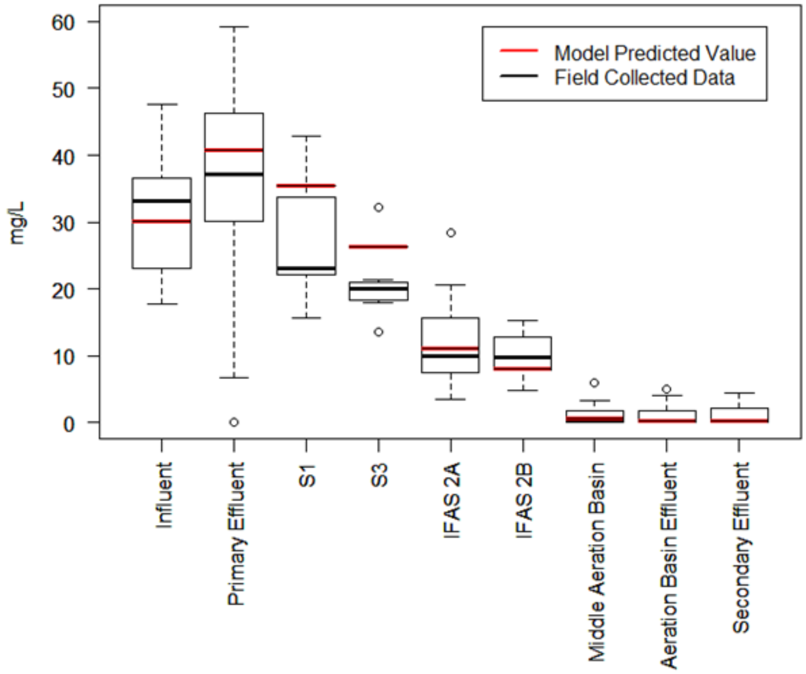


Figure 3.5: Ammonia Profile

Overall, while the calibration was not perfect, the model nonetheless captures most of the intricacies of the facility including the limited phosphorus removal, and nitrification and denitrification. Model calibration served to minimize the error while allowing meaningful conclusions about the data to be drawn. More critically, the calibrated model sufficiently serves the primary purpose of this research – modeling conversion from a WWTP to a WRRF, integrating proximate waste streams.

3.4 Resource Recovery Scenarios

This study established, evaluated, and compared eight different resource recovery alternatives and process configurations, as well as different ways of integrating proximate waste streams for driving specific processes. Table 3.8 summarizes the respective processes and complete resource recovery alternatives integrated into each scenario. Each scenario is then further described with an illustration (Figures 3.6 – 3.13) of the process schematic for each alternative to demonstrate flows crossing the system boundary in each scenario.

Table 3.8: Alternate Scenario Matrix

Scenario		EBPR	CHP	Struvite	PHA
1	Base Operations				
2	Base Operations		x		
3	Base Operations			x	
4	a: PS Fermentation b: UASB Conversion	x		x	
5	a: PS fermentation b: UASB conversion c: Dairy manure integration	x	x	x	
6	a: Dairy manure for PHA, PS ferm for EBPR b: Dairy manure for PHA, UASB for EBPR	x	x	x	x
7	a: TASCO for PHA, EBPR driven by PS ferm b: TASCO for PHA, EBPR driven by UASB conversion	x		x	x
8	Dairy manure for PHA		x		x

Scenario 1 represents the base case (i.e., current operation) and serves as the control to which other alternatives were compared.

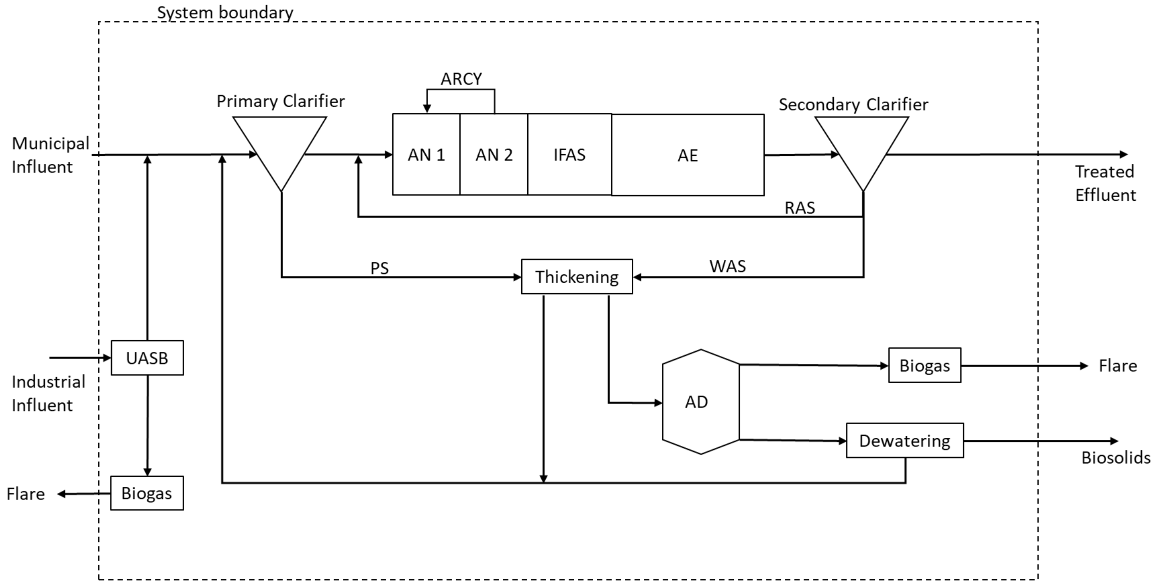


Figure 3.6: Scenario 1

Scenario 2 integrates CHP into the current biogas handling procedures, producing electricity that can offset usage at the facility. No additional considerations are made for the

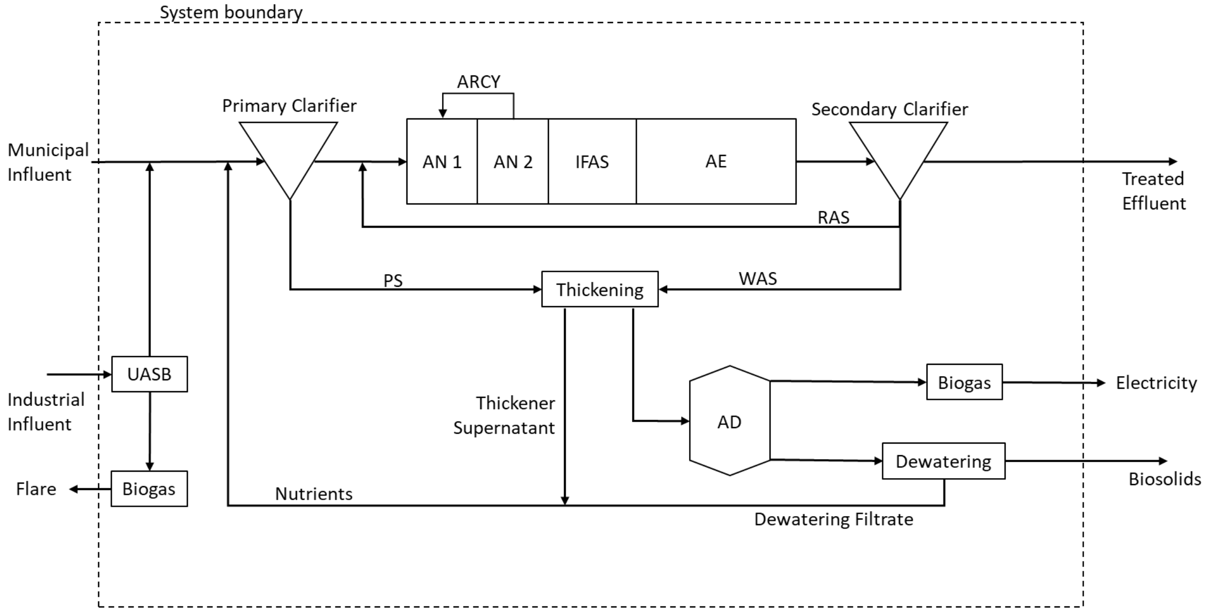


Figure 3.7: Scenario 2

production of additional biogas at the WRRF, and biogas produced through UASB pretreatment is still flared.

Scenario 3 integrates struvite crystallization using the phosphorus rich dewatering filtrate stream – significantly reducing the phosphorus load returned to the headworks of the facility.

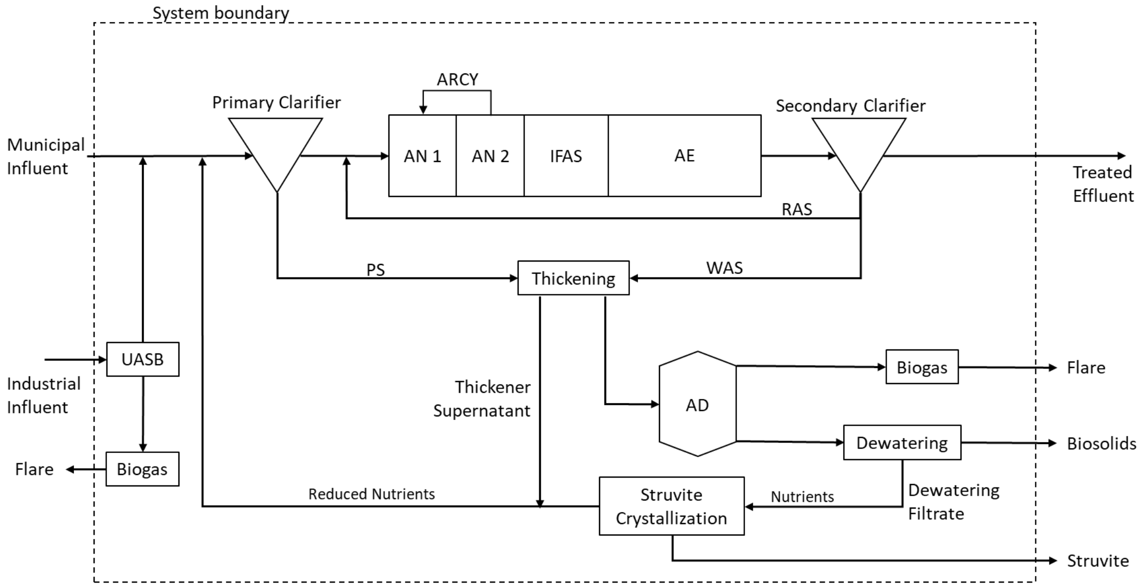


Figure 3.8: Scenario 3

Scenario 4 integrates EBPR and struvite production; this scenario can be readily compared to Scenario 3 to investigate how struvite crystallization is enhanced through EBPR. Two sub-scenarios are included in Scenario 4: (4a) integrates primary sludge fermentation as a VFA source to drive EBPR, while (4b) utilizes carbon rich waste streams at Chobani and Lamb Weston by converting the current UASBs into fermenters on-site – thus sending a VFA rich substrate into the municipal collection system which will ultimately drive EBPR once it reaches the current facility. In driving EBPR, the existing WWTP process configuration must be modified; RAS will be introduced to the first selector basin and the anaerobic recycle (ARCY) will not be utilized.

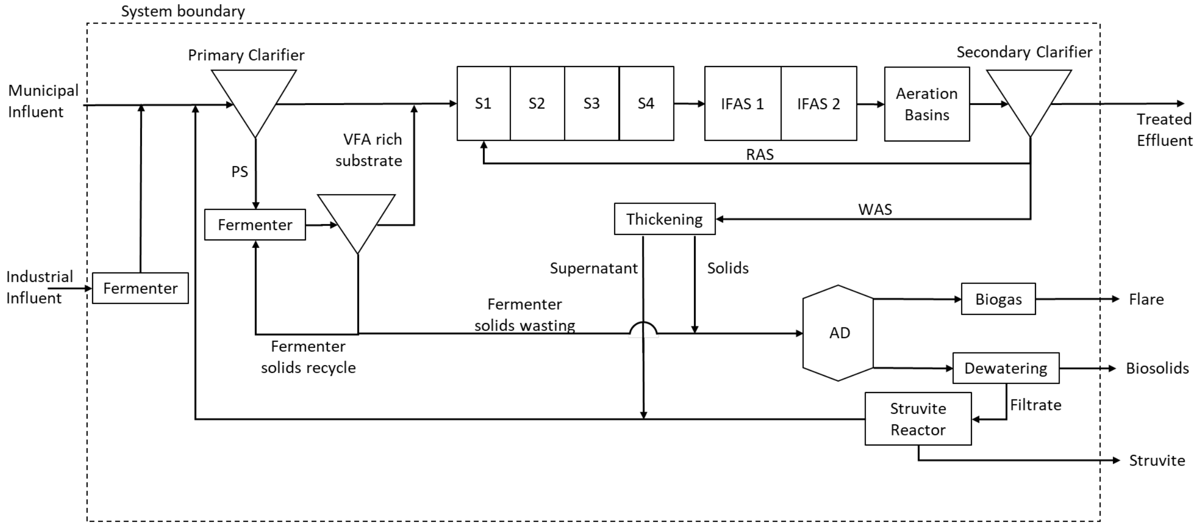


Figure 3.9: Scenario 4

Scenario 5 achieves EBPR, CHP, and struvite crystallization – representing mainstream resource recovery integration and processes widely in use at WRRFs around the globe. Struvite crystallization serves to both provide a beneficial resource (fertilizer) as well as stabilize the EBPR process by eliminating a significant portion of the internal phosphorus recycle. The sub-scenarios integrating proximate waste mirror that of scenario 4 with primary solids fermentation and UASB conversion to drive EBPR, but also includes a new sub-scenario (5c) that incorporates dairy manure. Dairy manure will be fermented to produce VFAs to drive EBPR; residual solids from the fermentation process are anaerobically digested – producing more biogas and thus enhancing CHP over the other two sub

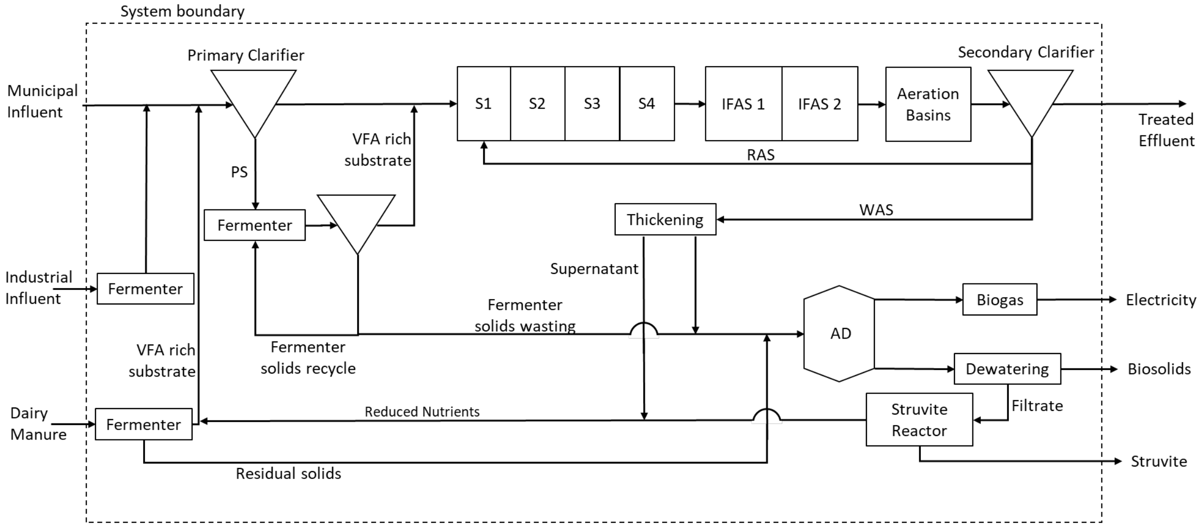


Figure 3.10: Scenario 5

scenarios. The additional load of nutrients from the dairy fermenter are accounted for in this scenario, as are the additional nutrients associated with the digestion of residual solids from the fermenter.

Scenario 6 integrates EBPR, CHP, struvite crystallization, and PHA production. As PHA production and EBPR are both very carbon intensive, while PHA production also requires a more concentrated VFA stream, the only two waste streams available to drive PHA production are TASCO and dairy waste. However, as CHP is considered in this scenario, TASCO would not be able to enhance digestion as dairy manure can – thus dairy manure is the only external substrate used for PHA in both sub-scenarios. EBPR is driven by primary sludge fermentation in sub-scenario (6a) and UASB conversion in sub-scenario (6b).

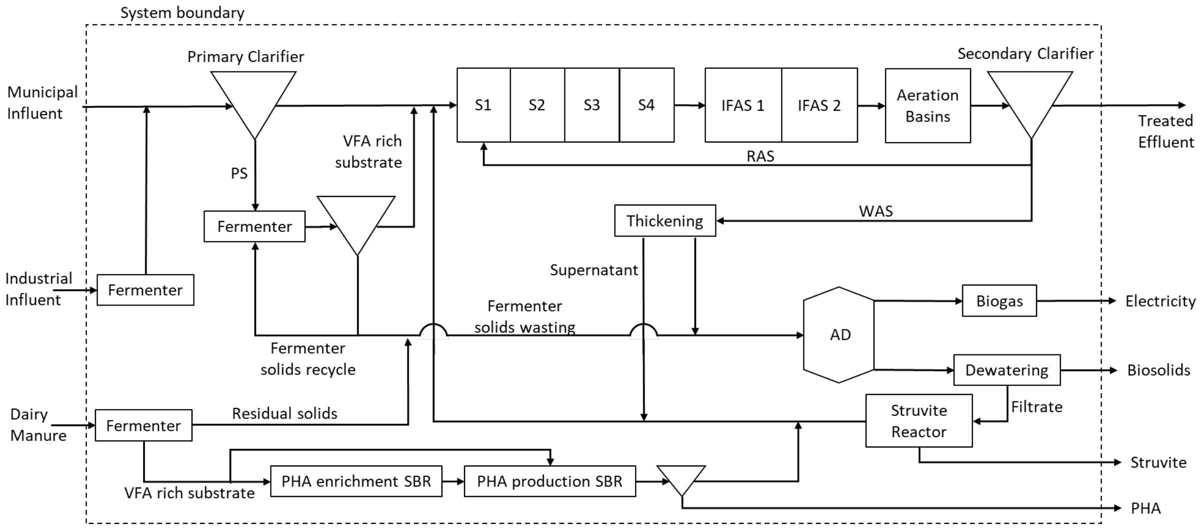


Figure 3.11: Scenario 6

Scenario 7 integrates EBPR, struvite, and PHA. The lack of CHP is of note, as it is not expected to be a large resource in the area. In Idaho, electricity is both relatively inexpensive and relatively clean (Idaho Power 2018). In this scenario, the TASCO wastewater rich in VFAs and deficient in nutrients is used to operate PHA production. PHA enrichment requires nutrients (N and P), in which TASCO wastewater is deficient – thus a different VFA source is required to maintain the PHA enrichment reactor. Primary sludge fermentation is used as the VFA source to drive EBPR and maintain the PHA enrichment reactor in sub-scenario 7a, and UASB conversion was used for this purpose in sub-scenario

7b. Not shown in the diagram is the diversion of a small portion of VFA rich primary effluent to the PHA enrichment reactor for sub-scenario 7b.

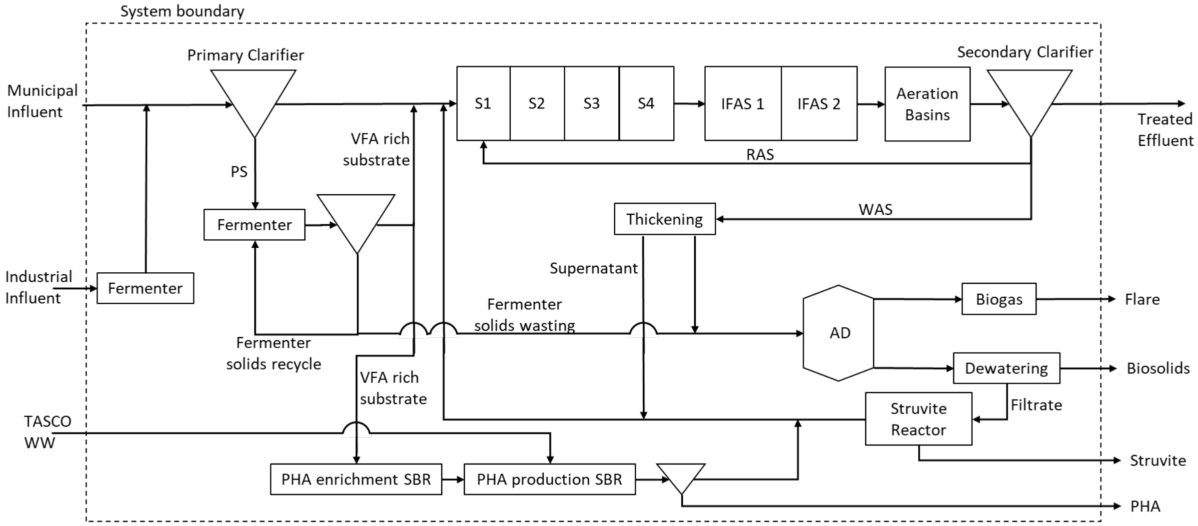


Figure 3.12: Scenario 7

Scenario 8 integrates PHA production and CHP utilizing dairy manure as substrate. This configuration increases biogas production from the digestion of residual solids from dairy manure fermentation, thus enhancing CHP over scenario 2. PHA production is driven by the VFA rich substrate produced from fermentation.

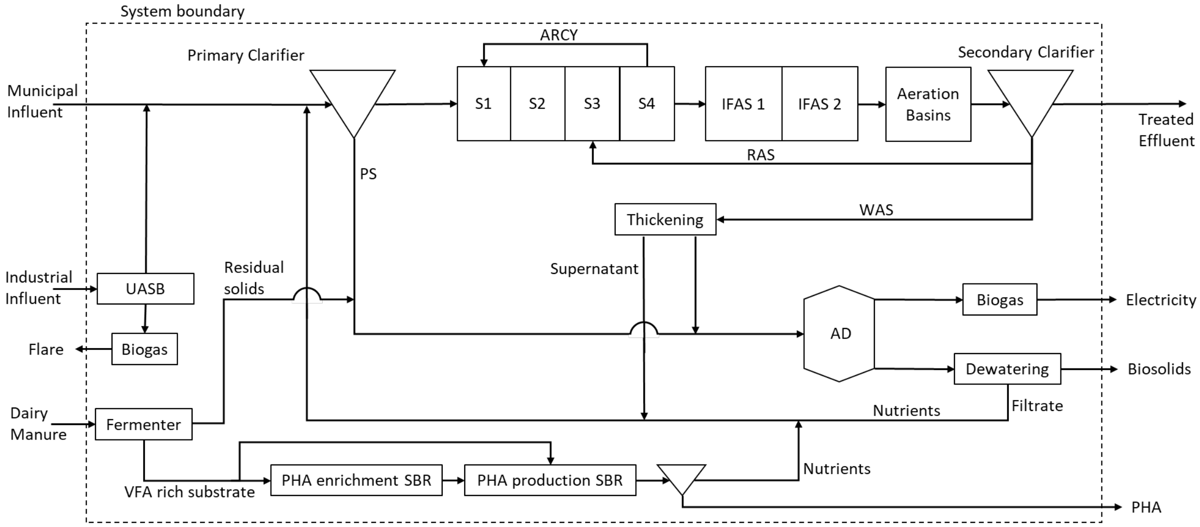


Figure 3.13: Scenario 8

3.4.1 Scenario Modeling

Struvite production was modeled as a single CSTR with no solids recycle. While this does not represent the actual process that would be implemented (i.e., an FBR), this approach was necessary because it utilizes what SUMO is capable of modeling to produce an equivalent result including struvite production, chemical requirement, and nutrient recycling.

PHA production could not be successfully modeled in SUMO, as the kinetic and stoichiometric parameters for PHA producing microorganisms are different than for PHA storing microbes within the SUMO model. To-date the ADF PHBV production model has not been integrated into any commercial process models. Instead, a simple stoichiometric approach was employed to model potential PHBV production. Yield values of 0.6 gPHA/gVFA were assumed for TASCO flume water, based on preliminary tests performed by the UI Environmental Engineering Lab. PHA production using dairy fermenter liquor was based on the DAIRIEES model (Guillen 2017), which is a simply mass balance based systems model. It should be noted that with any PHA production process located at a WRRF, the supernatant from solids separation will need to be included in the treatment process, with phosphorus available for struvite production. In the scenarios in which PHA production was included, a separate state variable stream was created in SUMO, combining with the raw influent wastewater to ensure that the treatment process is modeled to include the additional loads of N and P associated with the inclusion of dairy waste. The characteristics for this state variable were estimated within SUMO from dairy fermenter liquor; the influent dairy fermenter liquor was followed by an aerobic basin to oxidize the VFA substrate that would be used for PHA production. The effluent from this aerobic basin was solids separated, with the liquid stream characteristics now low in VFAs and still high in nutrients converted to state variables.

CHP production was based on the biogas output from SUMO (methane percentage and biogas flow rate). It was assumed that an internal combustion engine with electricity generation efficiency of 40% was utilized. To obtain a value of electricity production, the lower heating value of methane was used with the biogas flow, methane percentage, and 40% electrical efficiency. Alternatives involving the integration of dairy manure augment the anaerobic digesters with additional waste solids to increase biogas output. Residual solids accounting for 40% of the volumetric flow from the dairy manure fermentation process (20,000 gpd) are fed to the digester (Stowe 2014); this was modeled in SUMO using an additional influent stream. The characteristics of this stream are based on data collected by the UI environmental engineering lab, which routinely collects data for total solids, volatile solids, N, P, and VFAs. These values are listed in Table 3.9. To convert these values to a COD basis, a value of 2.3 gCOD/gVS was assumed.

Table 3.9: Dairy Fermenter Residual Solids Characteristics

Parameter	**Value**
Q, gal/day	20,000
TS, %	1.5
VS, %	1.2
COD, mgCOD/L	27,600
NH_4, mgN/L	200
PO_4, mgP/L	12.6
VFAs, mgCOD/L	4,250

In scenario 5c, dairy manure is used to both drive EBPR and enhance AD. In this case, dairy fermenter liquor is used to ensure a stable and resilient EBPR process.

In all alternatives, EBPR was driven by VFAs produced via primary solids fermentation, conversion of UASBs to fermenters, or dairy fermenter liquor. Primary solids fermentation was performed in SUMO using an anaerobic CSTR with a thickener, maintaining an 8-hour HRT and 5-day SRT of the primary solids fermenter (Romenesko 2017). Conversion of UASBs to fermenters at Chobani and Lamb Weston was modeled in SUMO independent of the entire WRRF to keep the full plant model as simple as possible. Influent data to the UASBs is collected by Jacobs staff; the fermentation of these waste streams was modeled in SUMO using a fermenter with the same retention time characteristics as used to model primary solids fermentation. The output of this fermentation system was combined with characteristics of the existing wastewater and accounted for degradation of VFAs that is expected to occur within the collection system. Appendix D details a bench scale experiment conducted to estimate the reduction in VFAs associated with travel through the collection system. The total COD (1,870 mgCOD/L) of this new influent was much higher than the COD associated with current UASB pretreatment (987 mgCOD/L). To maintain BOD removal and nitrification in compliance with the NPDES permit, all aeration basins had to be utilized, which increased blower demands and electricity usage; additionally, the plant SRT had to be lowered from 4.5 days to 4 days to maintain an MLSS concentration below 3,000 mgTSS/L, ensuring permit compliance with solids removal in the secondary clarifiers. All scenarios utilizing conversion of UASBs were capable of achieving phosphorus removal, as well as removal of all constituents necessary for permit compliance.

The liquid stream from the dairy fermenter was assumed to account for 60% of the volumetric flow of the manure slurry used for fermentation. Characteristics of this dairy fermenter liquor are listed in Table 3.10, based on data collected by Stowe (Stowe 2014).

Table 3.10: Dairy Fermenter Liquor Characteristics

Parameter	**Value**
Q, gpd	30,000
TSS, mg/L	3,200
VSS, mg/L	2,720
COD, mgCOD/L	13,570
NH_4, mgN/L	200
PO_4, mgP/L	12.6
VFAs, mgCOD/L	4,250

3.5 Life Cycle Assessment

This section details the process of life cycle assessment as illustrated in Figure 2.1.

3.5.1 Goal and Scope Setting

Setting the goal and scope is the first step in preparing and conducting an LCA, and involves establishing an appropriate functional unit and associated system boundary. The functional unit is the basis of comparison between products/processes and should be selected to allow an equitable evaluation between alternatives. In LCAs on wastewater projects, the functional unit is often a volume of wastewater being treated (Corominas et al. 2013, Smith et al. 2014, Coats et al. 2011c, Lassaux, Renzoni and Germain 2007, Rahman et al. 2016); however, simple volume is not necessarily an absolute functional unit for WRRFs. The system boundary also needs to be clearly defined, as there are many levels of investigation that can occur which can often lead to further investigation. Setting boundaries can prove a challenge as the LCA should be in depth enough to draw meaningful conclusions, yet the investigations into the effects of secondary processes and emissions should not be exhaustive.

The functional unit for this study was selected to be 1,000 lb of influent COD; a volume of wastewater was not selected due to its treatment-oriented nature. By incorporating a mass of COD as the functional unit, the LCA fully incorporates and accounts for the integration of proximate wastes, which exhibit much higher COD concentrations relative to the influent flow rate. Indeed, the conversion from a WWTP to a WRRF realizing varying degrees of resource recovery implementation is better referenced with a mass of COD. Based on the system boundary including all pretreatment in addition to WRRF treatment, the influent COD is constant in all scenarios that do not import additional wastes. However, scenarios that import dairy manure or TASCO wastewater thus have a higher total influent COD, which is accounted for in the LCY with the functional unit. A sample calculation illustrating how the functional unit was integrated is included in Appendix E.

3.5.2 Life Cycle Inventory Analysis

Step two is the life cycle inventory analysis (LCIA), which involves accessing and quantifying the inventory of flows that cross the established boundary. While LCIA can be a challenging and cumbersome process, its usefulness is that it encompasses intrinsic material and energy flows that are typically not considered in traditional decision-making processes. In the case of this project, for example, data that may be necessary to conduct the LCA may include quantifying fertilizer manufacture and transportation that is offset with the usage of biosolids and struvite, quantifying the emissions associated with production of electricity used for the processes studied, or the emissions offset for the production and transportation of plastics associated with their replacement with bioplastics. Completing the LCIA is achieved first through data analysis, or in the case of this study, through process modeling. LCIA is necessary for analyses that answer questions such as relative sizes of emissions, helping decision makers to locate areas that benefit from process improvements. Several tools to assist in performing the inventory analysis exist, commonly referred to as LCA databases. These include software such as ecoinvent and ReCiPe, all of which have been successfully integrated into different wastewater LCA studies.

3.5.3 Life Cycle Impact Assessment

The third step, impact assessment, measures the impact from the data collected during the LCIA stage, and provides comparative results. There are several ways to assess LCIA, but in wastewater projects the EPA Tool for Reduction and Assessment of Chemicals and Other Environmental Impacts (TRACI) model has been proven successful with its midpoint analysis method (Coats et al. 2011c, Postacchini et al. 2016, Landry and Boyer 2016, Rahman et al. 2016, Ishii and Boyer 2015).

TRACI is an ideal tool for projects in the United States, as it was designed with US regulations in mind (Bare 2011). TRACI uses nine different impact categories which were selected to be comprehensive and manageable: ozone depletion, global climate, acidification, eutrophication, smog formation, human health particulate, human health cancer, human health noncancer, and ecotoxicity. A list of around 3,000 chemicals specifies a normalized equivalent in each category; the sum of each is calculated to determine the final score in each category. Other impact assessment tools are also available, such as IMPACT 2002+ and CML 2001. Additional software packages such as umberto, SimaPro, and GaBi also exist which bundle LCA databases with impact assessment tools, presenting an all in one package to perform LCA.

3.5.4 Interpretation

The last step is the interpretation of data which should include identification of significant issues, checks for completion including sensitivity and consistency checks. Lastly, it should include conclusions, limitations, and recommendations.

4. Results and Discussion

This chapter presents and discusses results of the LCA investigations and analysis. The inventory analysis presents results for all the flows crossing the system boundary, utilizing SUMO for effluent quality, resource production, and chemical requirements. The inventory analysis also includes data for the offsets associated with the use of the resources produced at the WRRF. The impact assessment was performed using the EPA TRACI model; results are summarized for each scenario. Lastly, an economic analysis was performed to estimate the net present value of each scenario.

4.1 Scenario Modeling Results

As described in section 3.5.2, the first step in the life cycle inventory analysis was to generate data for each scenario based on effluent quality and the resources produced. SUMO was utilized to predict effluent quality, biogas yields, struvite production, and biosolids production; as discussed in section 3.4.1, PHA production was based on the DAIRIEES model for scenarios utilizing dairy manure as the substrate, while a simplified stoichiometric approach was employed for the TASCO substrate. Table 4.1 summarizes the effluent quality and resource production for each scenario described in Table 3.8.

4.2 Life Cycle Inventory Analysis

Life cycle inventory analysis was used to quantify the flows and emissions crossing the system boundary as illustrated for each scenario (Figures 3.6–3.13), in accordance with ISO 14040 standards. The system boundary was selected to include all aspects of treatment, including treatment at the WRRF, pretreatment at other off-site locations, and biosolids transportation. Chemical use and their transportation as well as imported waste streams were also considered. Each scenario was separately quantified with emissions associated with effluent quality (BOD_5, PO_4, TSS, etc.) as listed in Table 4.1. Emissions associated with the substitution/offsets of synthetic fertilizer with biosolids and struvite and of petroleum

Table 4.1: Effluent Quality and Resource Production

	Scenario												
	1	**2**	**3**	**4a**	**4b**	**5a**	**5b**	**5c**	**6a**	**6b**	**7a**	**7b**	**8**
Treatment Emissions													
COD, mg/L	183	183	183	183	188	183	188	184	181	189	249	255	183
BOD$_5$, mg/L	2.3	2.3	2.3	2.6	2.3	2.6	2.3	2.3	2.8	2.3	2.4	2.4	2.3
TSS, mg/L	5	5	5	5	5	5	5	5	5	5	5	5	5
NH$_4$, mgN/L	0.2	0.2	0.2	0.6	0.5	0.6	0.5	0.1	0.5	0.7	0.3	0.9	0.2
NO$_3$, mgN/L	14.9	14.9	11.8	11.7	4.3	11.7	4.3	15.1	10.7	6.1	11.2	4.2	17.2
TP, mgP/L	6.3	6.3	1	0.6	0.6	0.6	0.6	0.7	0.6	0.6	0.7	0.7	8.6
PO$_4$, mgP/L	5.8	5.8	0.4	0	0	0	0	0.1	0	0.1	0	0	8.1
Biogas, scfm	141	141	146	110	260	110	260	203	139	268	114	281	142
Biogas methane, %	64.2	64.2	64.2	59.7	61.1	59.7	61.1	62.4	59.5	61	59	62.1	62
Resources Produced													
Electricity, kW	0	579	0	0	0	420	1020	814	531	1051	0	0	566
Biosolids, ton/day	5.7	5.7	6.5	6.8	10.9	6.8	10.9	13.3	9.5	12.5	7.6	11.6	6.9
Struvite, lb/day	0	0	3110	3023	977	3023	977	0.4	1808	2471	2992	1561	0
PHA, lb/day	0	0	0	0	0	0	0	0	790	790	266	266	790

plastics with PHA were also quantified. It was assumed that diammonium phosphate (DAP) with an NPK value of 18-45-0 was replaced with biosolids and struvite – both applied on a P-limiting basis. DAP is one of few synthetic fertilizers with a similar ratio of N to P as struvite and has been selected as the synthetic fertilizer offset through the use of struvite in several studies (Sørensen, Dall and Habib 2015, Foley et al. 2010). Emissions associated with the production of DAP were based on an LCA study performed by Manjare et al. in 2011. An additional assumption was that DAP is transported to the Twin Falls area 1400 miles via truck from a warehouse in Savage, MN, where The Mosaic Company operates a warehouse that stores DAP. Few suppliers of DAP were able to be located in the United States; The Mosaic Company was selected due to its proximity to Twin Falls, as other suppliers found are located in the South and on the East Coast.

Offsets associated with the use of bioplastics (PHA) were evaluated based on a study by Harding et al. in 2007 that compared the production of PHA with three petroleum plastics: polypropylene, high density polyethylene, and low density polyethylene. Their study assumed that the PHA produced in the visioned Twin Falls WRRF exhibits characteristics associated with a 50:50 mixture of high-density and low-density polyethylene. Transportation of petroleum plastics and PHA was not considered in this analysis, as it was assumed the two would cancel each other out.

Emissions associated with production and transportation of chemicals needed for struvite crystallization were also considered in the study. Sodium hydroxide production emissions for both air and water were quantified based on an LCA performed by Thannimalay et al. (2013), which investigated sodium hydroxide production in Malaysia. It was assumed that sodium hydroxide for this study would be transported 228 miles from NuChem, a chemical distributor based in Salt Lake City, UT. Magnesium hydroxide was assumed to be purchased and transported as magnesium oxide and combined with water on site to produce magnesium hydroxide. It was assumed that the product was produced at US Magnesium in Salt Lake City, UT, and transported 222 miles to the Twin Falls WRRF. Emissions associated with the production of magnesium oxide are largely associated with energy demands for its manufacture; it was assumed that one ton of magnesium requires 2948 kWh of electricity, based on a recent study on magnesia production in China by Li et al. (2015).

Electricity source data obtained from Idaho Power was used to quantify emissions associated with electricity production (Idaho Power 2018). Table 4.2 summarizes the percentage of electricity produced based on several electricity sources, with the numbers normalized to eliminate a 5.5% “other” category. Several scenarios considered in this study implemented process changes that affected the net electricity use at the Twin Falls WRRF: the removal of the pumped ARCY flow (scenarios 4, 5, 6, and 7), the increase in blower demands associated with maintaining all aeration basins online (scenarios 4b, 5b, 6b, and 7b), and the production of electricity from CHP (scenarios 2, 5, 6, and 8). The calculation of power savings from removing the ARCY flow was based on Phase 2 data provided by Jacobs (CH2M Hill 2013), which lists the power of the ARCY pump as 25 hp (18.6 kW). The increase in blower load was estimated based on an airflow increase of 7,500 scfm, which is the capacity of a single blower as listed in the Twin Falls 2013 Facility Plan provided by Jacobs. The outlet pressure was assumed to be 8.5 psig, with a conservatively estimated temperature of 100°F. The total estimated increase in power based on these assumptions was 288 hp (215.3 kW). Electricity production from CHP was estimated based on the biogas flow rate and methane content as well as the lower heating value of methane at standard temperature and pressure (960 B/ft^3). Electricity use for the production of magnesium hydroxide and associated emissions were based on fuel source data for Rocky Mountain power, which operates in Salt Lake City, UT (Rocky Mountain Power 2017).

Table 4.2: Idaho Power and Rocky Mountain Power Electricity Sources

Fuel	Idaho Power	Rocky Mountain Power
Hydro	52.39	7.93
Coal	18.28	65.87
Wind	10.02	9.57
Natural Gas	8.37	11.82
Solar	3.23	3.96
Geothermal	1.43	0.44
Biomass	0.78	0.41
Total	100.00	100.00

Emissions for electricity production from each fuel source were based on a 2012 study by Turconi et al. (Turconi, Boldrin and Astrup 2013). Their study quantified, by source of electricity, the CO_2-eq, NO_x, and SO_2 emissions on a kilogram per MWh produced based on 167 LCA studies on different modes of power production, producing results for power generation that are applicable worldwide. Table 4.3 summarizes the values found by Turconi et al., which were used in this study.

Table 4.3: Emissions Based on Energy Source

Energy Source	**CO_2-eq**	**NO_x**	**SO_2**
Hard Coal	825	0.60	3.37
Lignite	1050	0.95	3.80
Natural Gas	690	2.00	0.17
Oil	715	1.00	4.43
Nuclear Power	19	0.03	0.02
Biomass	69	69.25	0.49
Hydropower	11	0.03	0.02
Solar Energy	102	0.28	0.21
Wind	22	0.07	0.06

In the scenario 7, TASCO wastewater was utilized to drive PHA production. It was assumed that the wastewater is trucked 7.7 miles from the TASCO facility south of Twin Falls to the WRRF. To achieve the estimated flow rate of 39,000 gallons per day, four trucks per day are needed, totaling 61.6 miles per day per round trip. Transportation of dairy manure was assumed to require a truck traveling 20 miles one way to the WRRF. To achieve transportation of 340,000 kg (750,000 lb) of manure per day, 10 trips per day are required in heavy trucks, which totals 400 miles per day. Biosolids disposal was also considered: a 25-mile trip with one trip and a single truck necessary was assumed. The emissions associated with transportation were based on 2015 EPA estimates for medium and heavy-duty trucks transporting products (EPA Climate Leadership 2015). Table 4.4 details the emissions associated with transportation for medium and heavy-duty trucks on the basis of emissions per ton per mile.

Table 4.4: Transportation Emissions

Vehicle Type	**CO_2 (kg/ton-mile)**	**CH_4 (g/ton-mile)**	**N_2O (g/ton-mile)**
Medium- and Heavy-Duty Truck	0.146	0.0015	0.0014

The total emissions associated with flows crossing the system boundary for each scenario (effluent quality, transportation, chemical production, plastic and fertilizer substitution/offset, and electricity production) based on the functional unit were then calculated for input into TRACI. Appendix F details the values for all chemicals associated with each scenario and used in the impact assessment.

4.3 Life Cycle Impact Assessment

Figure 4.1 illustrates the emissions for each scenario for each environmental impact category. In addition to the previously discussed impact categories, there are several other impacts that relate to ecotoxicity and human health hazards; these categories are very specific (e.g., human health characterization factor for emission to urban air, cancer) and subject to high degrees of uncertainty. As a result, figures for ecotoxicity and human health hazards are not shown, but the full data set is tabulated in Appendix G.

For every environmental category in Figure 4.1, each scenario was assessed based on six contributing factors: transportation, plastics offset, effluent, struvite chemicals, electricity change, and fertilizer offset. The transportation category includes local transportation of biosolids, dairy manure, and TASCO wastewater, as well as transportation of sodium hydroxide and magnesium hydroxide for struvite production and the reduction in transportation of synthetic fertilizer (DAP). “Plastics offset” is the reduction in emissions associated with the usage of PHA in lieu of petroleum-based plastics. “Effluent” is the result of effluent wastewater constituents of COD, P, and N discharged to the Snake River. “Struvite chemicals” are the emissions associated with the production of magnesium hydroxide and sodium hydroxide. Electricity change is the net change in electricity at the WRRF due to implementation of CHP, removal of the ARCY pump, and changes in aeration demands. Lastly, “fertilizer offset” is based on emissions avoided due to the replacement of DAP with biosolids and struvite. While six factors were selected, only two appear visible in Figure 4.1; electricity offset and struvite chemicals are several orders of magnitude greater than the other four factors. These factors were grouped into an “other” category so that they are more visible.

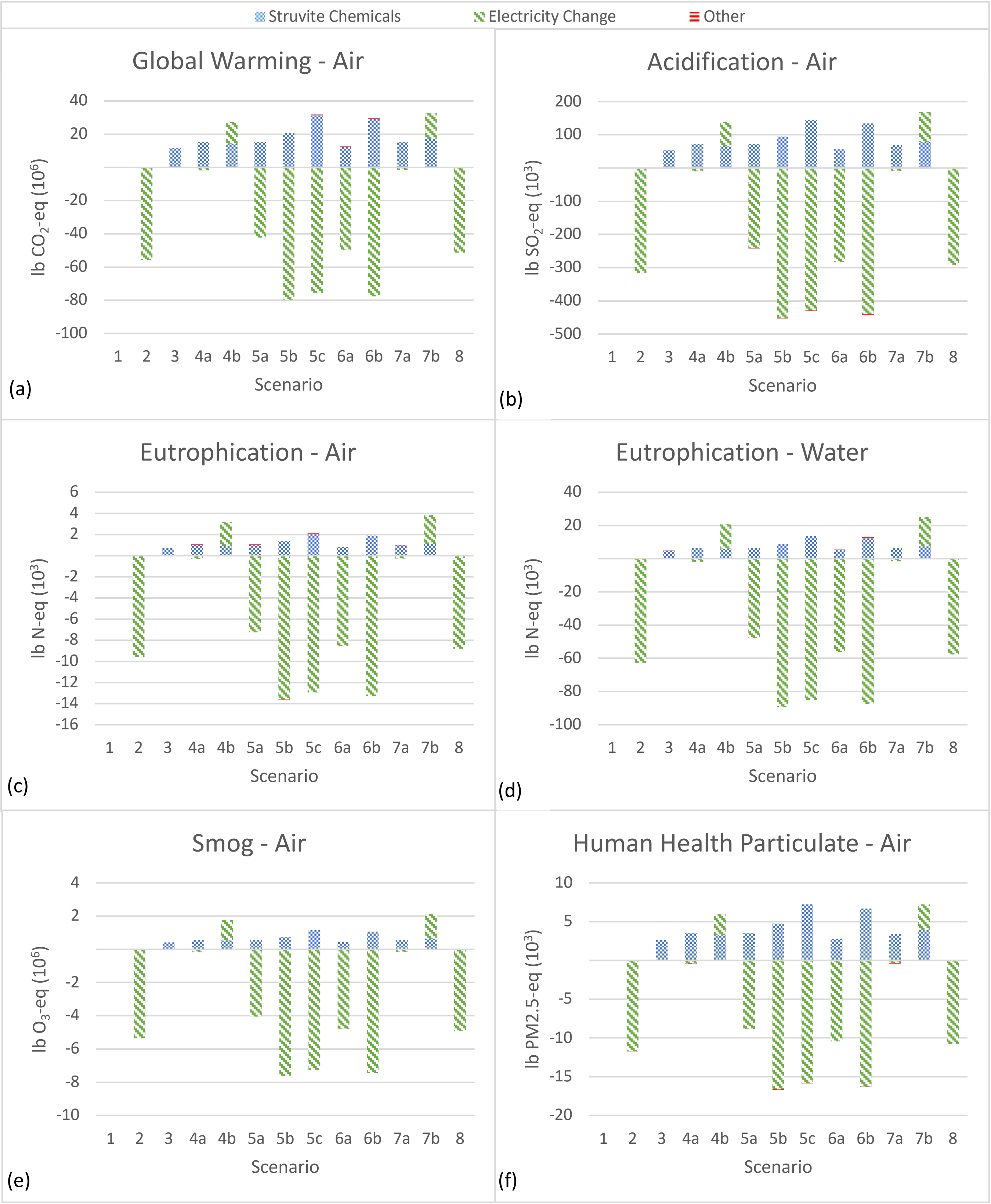


Figure 4.1: TRACI Emissions by Category

4.3.1 Scenario 1 – Base Case

Scenario 1 is the base case and all associated emissions are due to i.) a fertilizer offset from the use of biosolids and ii.) the effluent wastewater.

4.3.2 Scenario 2 – Base Case with CHP

Adding CHP to the base case (Scenario 1) shows promising results in all categories. The use of CHP to reduce electricity demand offsets a significant quantity of emissions associated with using the current sources electricity at the Twin Falls facility. Although Idaho Power maintains a significant portion of its power portfolio as relatively 'clean' hydropower, coal and natural gas still account for roughly 30%, both of which are associated with higher amounts of carbon dioxide, nitrogen oxides, and sulfur dioxide emissions. The offset of these emissions is associated with a reduction in emission equivalents in all categories: global warming – air, acidification – air, eutrophication – air, eutrophication – water, smog – air, and human health particulate – air.

4.3.3 Scenario 3 – Base Case with Struvite Production

Scenario 3, add struvite recovery to the Twin Falls WRRF, exhibits more negative environmental impacts in all categories when compared to the base case (Scenario 1). Chemical requirements for struvite formation are environmentally taxing due to the energy and material requirements involved in its production. Transportation of these chemicals is also environmentally negative; however, the transportation impacts of chemicals alone is more than offset by the reduction in synthetic fertilizer transportation. Overall the net environmental effect is negative with the addition of struvite crystallization alone; while effluent concentrations of total phosphorus from the WRRF is reduced by 84%, the negative environmental effects of chemical use to achieve struvite precipitation outweighs any improvement in effluent quality from the Twin Falls facility

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4.3.4 Scenario 4 – EBPR with Struvite Production

Scenario 4 focuses on creating a resilient EBPR process, utilizing VFAs to drive EBPR and struvite crystallization to reduce the recycle of phosphorus from the digesters to the headworks of the WRRF. Scenario 4a, utilizing primary solids fermentation, outperforms Scenario 4b in every category; it utilizes UASB conversion to produce VFAs,. In Scenario 4b, the increased quantity of readily biodegradable carbon reaching the WRRF (measured as COD or BOD) requires all aeration basins to be on line to maintain treatment, greatly increasing the electricity demands at the WRRF. Since the increase electricity for aeration demands is not offset by CHP (not included in this scenario), 4b results in the second worst impacts in all environmental categories. Similar results were found in a study by Bisinella de Faria et. al. (2015), which found that enhanced primary clarification helped improve biogas production to move toward a positive energy balance. Scenario 4a utilizes anaerobic treatment of carbon (UASBs) to prevent the necessity of having all aeration basins on line; however, it is still the third worst performing scenario with respect to global warming, acidification of air, and human health particulate categories, and the fifth worst with respect to eutrophication of air and water and smog air. Overall, this scenario summarizes the differences between treating wastewater carbon anaerobically (Scenario 4a) versus aerobically (Scenario 4b).

4.3.5 Scenario 5 – EBPR with Struvite Production and CHP

The differences, between scenario 4a and 4b are significantly changed with Scenario 5, which adds CHP to biogas that is produced on site at the WRRF. Scenarios 5a and 5b are identical to 4a and 4b, with the exception that the biogas produced is used to drive CHP and produce electricity. Like Scenario 4, UASB conversion requires all aeration basins to be on line, increasing electricity usage due to high blower demand; however, the addition of CHP generates more electricity than is required by the increased blower demands. This increase in electricity production from superior AD performance and UASB conversion outweighs the increase in electricity demand associated with the increase in aerobic treatment required at the WRRF. In contrast, primary solids fermentation (5a) reduces the quantity of easily digestible primary solids fed to the anaerobic digester, thereby reducing digester biogas output and electricity production. These results demonstrate that carbon can be utilized for either nutrient removal or energy production, but not both.

The subject of carbon management is widely discussed within the literature, and it is understood that while carbon is necessary to support biological removal of N and P, the aerobic oxidation of carbon prevents energy recovery from being truly realized (Jimenez et al. 2015, Sancho et al. 2019). Scenarios 4b and 5b, utilizing UASB conversion for EBPR, better manage available wastewater carbon by preventing the conversion of carbon to carbon dioxide at the pretreatment facilities, thereby allowing more carbon to be used for energy production as evidenced by the biogas flow rate in Table 4.1. Overall, Scenarios 5a and 5b perform significantly better than 4a and 4b in all categories due to the addition of CHP. Scenario 5c, integrating dairy manure, performs in a manner very similar to that of 5b. However, the additional phosphorus load from dairy manure increases the required amount of struvite chemicals. Additionally, the digester biogas output is less than in Scenario 5b, resulting in lower benefits from CHP. The decrease in biogas production was unexpected; it was initially assumed that the increase in substrate fed to the digester from the dairy fermentation process would increase biogas production. Possible reasons this was not found include a lack in digester capacity and the high quantity of nonbiodegradable substrate, the latter of which is likely due to an error in characterizing the dairy manure. Coats et al. (2012) have suggested that co-digestion utilizing thickened, pre-fermented dairy manure increases biogas output, which suggests that the SUMO model is not well equipped to model dairy manure.

4.3.6 Scenario 6 – EBPR with Struvite Production, CHP, and PHA Production

Scenarios 6a and 6b are nearly identical to 5a and 5b; however, they add PHA production utilizing dairy manure. As illustrated in Figure 4.1, Scenario 6a achieves better environmental performance than Scenario5a in all categories; conversely, Scenario 5b performs better than Scenario 6b in all categories, with top environmental performance in all categories. The difference between Scenarios 6a and 6b is quite interesting: while biogas and power output between the two were very similar, 6b receives increased quantities of carbon and phosphorus from the dairy manure, thus both requiring more chemicals for the struvite crystallization process and an increase in aeration demand to maintain treatment. This suggests that PHA production might not always have a net environmental benefit, since the increase in loading to the facility requires more energy and chemicals to maintain treatment, with little environmental benefit associated with the plastic offset.

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4.3.7 Scenario 7 – EBPR with Struvite Production and PHA Production

Scenario 7, utilizing TASCO wastewater to drive PHA production, achieved results equal to or worse than Scenario 3, which likewise did not utilize CHP. As seen with other non-CHP scenarios, the lack of electricity production hindered Scenario 7 significantly; CHP was not considered in Scenario 7, as TASCO wastewater would not produce any substrates to enhance digestion. Scenario 7b, coupling TASCO wastewater for PHA production with UASB conversion to drive EBPR, was the worst performing scenario in all environmental categories, as the increase in electricity for aeration is not offset by CHP.

4.3.8 Scenario 8 – CHP and PHA Production

Without a focus on phosphorus removal and recovery, Scenario 8 achieved surprisingly high environmental performance in several categories by focusing on dairy manure for PHA production while enhancing anaerobic digestion to support greater CHP and electricity output. The result was unexpected, as this scenario does nothing to improve effluent quality by removing phosphorus. In fact, additional phosphorus is added in dairy manure, resulting in higher effluent P than even the base case. Figure 4.1: TRACI Emissions by Category suggests that the environmental benefits for Scenario 8 all come from adecrease in required electricity from the grid. Effluent wastewater constituents in the quantities discharged have negligible impacts on all categories when compared to changes in electricity demand and use of chemicals for the struvite process, which is especially surprising in the eutrophication-water category. The overall sustainability of struvite production was relatively low in all categories, which was also a wholly unanticipated result. That said, Scenario 8 performs very similarly to Scenario 2, which employed only CHP with the base case. The production of bioplastics, like improvements in effluent quality, also has negligible impacts in all environmental categories and across all scenarios. In total, the “other” category routinely had raw values of emission equivalents in each impact category on the order of 10^{-2} to 10^{1}; the only exception was in the transportation category in scenarios where the import of dairy manure or TASCO wastewater were considered. Electricity change, on its own, routinely achieved a magnitude of 10^{6} or 10^{7}, suggesting the importance of energy production in an LCA.

4.3.9 Sensitivity Analysis

A sensitivity analysis was performed to better ascertain the effects of certain parameters on the results of the LCA. Guven et. al. (2018) performed a sensitivity analysis on parameters with known uncertainty or with important global issue. The first parameter tested was changing the sources of electricity used, performed in a similar manner to that Guven. While Idaho Power has a relatively clean footprint of energy fuel sources, it still maintains approximately 25% of its sources from fossil fuels. In 2019, Idaho Power announced its plan to achieve 100-percent clean energy by 2045 by eliminating coal and natural gas and replacing them with solar and wind (Idaho Power 2019a). To illustrate the effects of this change, the LCA was altered by changing Idaho Power’s fuel source percentages and exclude fossil fuel sources by increasing wind and solar. The results were expected to significantly decrease the positive effects of CHP and the negative effects of increased power usage, while giving the other contributing factors (e.g., struvite chemicals, fertilizer offset, effluent, plastic offset, and transportation) a larger input into the total environmental effects as they would no longer be dwarfed by the electricity category. The results of the sensitivity analysis (provided in Appendix H) suggest that this is the case. Figure H.1 illustrates that the environmental impacts of electricity use are lessened significantly; however, they still have a large impact on environmental emissions. The emissions from struvite chemical production become the predominating category in environmental emissions as electricity emissions are on average between scenarios approximately three times lower than emissions from struvite chemicals. This differs significantly from the original result in which electricity emissions were approximately 2.5 times higher than struvite chemicals on average. The largely invisible “other” category includes all four remaining factors (transportation, fertilizer offset, effluent, and plastics offset) and illustrates their negligible environmental effects compared to electricity and struvite chemicals, even when electricity impacts are reduced.

Struvite chemicals are another factor that had a significant impact on the environmental viability of resource recovery, as the negative effects of chemical usage significantly outweighed the positive effects of fertilizer offsets. The two studies utilized for chemical production were based on production in Malaysia and China (Thannimalay 2013, Li et al. 2015). While the process utilized may be similar between these Asian countries and what might be found in the United States, the emissions associated with their production in the United States could be less, due to stricter U.S. environmental regulations. Between sodium hydroxide and magnesium hydroxide, magnesium hydroxide had a much larger effect, and its associated emissions are all due to energy use. The second set of parameters changed for the sensitivity analysis, thus decreasing the effects of these chemicals by assuming that all the energy use required for magnesium hydroxide is renewable and that the emissions associated with the production of sodium hydroxide estimated by Thannimalay et al. (2013) were reduced by 10%. Unfortunately, no studies on the production of chemicals in the United States could be located, and access to LCIA databases that would have provided provide more accurate data was not possible. Figure H.2 illustrates the different results associated with these changes. While the factor for struvite chemicals was significantly reduced, the electricity factor was the only dominating factor. As a result, the other category includes all five other factors.

The last sensitivity analysis performed was a simple combination of the previous two: replacing all fuel sources with 100% renewables and assuming the effects of struvite chemical use were significantly reduced. Figure H.3 illustrates that even with the reduction in emissions from electricity and struvite chemicals, they are both still by far the dominant factors in contributing to emissions across all categories. The other categories removed from this figure (transportation and effluent factors), while still small, are visible when compared graphically to the other factors. Plastics offset, however, is still negligible compared to the other factors and is not visible in Figure H-3.

4.3.10 Data Interpretation and Discussion

The overall results from the impact analysis indicate that production of electricity results in far more positive environmental impacts than focusing on effluent quality or resource production alone. Figure 4.1 illustrates in all categories and across all scenarios that electricity change is the most important factor, followed by struvite chemicals. Fertilizer offset and transportation are the only other factors visible and minimally at best. The effects of both plastics offset and effluent are not visible in any category, as electricity and struvite chemicals are several orders of magnitude greater in impact. Many of the scenario results support the notion that carbon should be preferentially used for energy as opposed to nutrient removal (Rahman et al. 2016); however, permit requirements and the situation at any specific facility may dictate otherwise. It is important to note that the best environmental performance occurred when the maximum combination of resource recovery implementation and phosphorus removal was achieved.

The sensitivity analysis revealed that emissions from struvite chemical production and electricity generation are the largest contributing factors in each category and across all scenarios by several orders of magnitude. A change to fully renewable energy sources would go a long way in reducing total emissions; under this scenario, however, CHP would still have significant environmental benefits, as there are still some emissions associated with renewable energy. Additionally, the chemicals used in struvite production still have a high environmental cost, even if significantly lowered emissions from their production can be achieved.

Several different conclusions can be drawn from the data:

1. As Idaho Power operates with high levels of hydropower and renewable energy (only 25% fossil fuel sources), it was expected that electricity use impacts would be low; however, the data revealed that this was not the case. The eutrophication of water category was initially expected to be highly dependent on effluent phosphorus. Scenario 1 has 6.3 mgP/L of effluent total phosphorus and a 14.9 mgN/L of effluent nitrate, resulting in a nearly negligible value of 14 lb N-eq for eutrophication of water. In contrast, Scenario 8 has an effluent total phosphorus of 8.6 mgP/L and an effluent nitrate of 17.2 mgN/L, yet -57,730 lb N-eq for eutrophication-water. Emissions associated with the bioplastic offset for Scenario 8 had no effect on eutrophication of water; instead, the biggest contributor to this category was nitrogen oxides, which for Scenario 8 were wholly associated with the offset in electricity production. Similar results were found by Rahman et. al. (2016) in that environmentally positive local effects due to improved effluent quality can create additional negative impacts on overall global emissions associated with the inputs required to maintain higher levels of treatment. Sensitivity analysis further illustrates that the effects of offsetting energy use through CHP yields greater environmental benefit than improved effluent quality (i.e., removal of nutrients), even when renewable sources of electricity are fully utilized. The effect of energy use on treatment quality would be further pronounced outside of the study area in a region utilizing a higher percentage of fossil fuels for electricity production.
2. Carbon management strategies do not need to necessarily prioritize energy production over nutrient removal, or vise versa. Mindful use of carbon is paramount in operating in an environmentally healthy manner. Using UASBs to produce biogas for flaring represents the loss of a resource of high potential value. Comparing Scenario 5a to 5b illustrates this concept well; CHP is utilized in both scenarios at the WRRF; however, 5a flares the biogas produced at the UASBs, which then requires that primary solids be used to produce VFAs to drive EBPR. Fermented primary solids have a much lower value for AD, lowering potential electricity output. To

mitigate this, CHP systems could be installed at the pretreatment facilities to utilize the UASB biogas; however, the capital costs and additional maintenance requirements would likely be far greater than a single centrally located system.

3. Three technologies to produce resources of both economic and environmental value were investigated: struvite, bioplastics, and electricity. The production of electricity was shown to be the single most important resource in this study, especially from an environmental perspective. As biogas is already being produced, a significant portion of the necessary framework for its production is already in place. However, economically CHP has more uncertainties than it might in other locales as electricity prices in Twin Falls are well below the national average. Struvite and biosolids usage were shown to be capable offsetting a significant quantity of synthetic fertilizers, suggesting that sustainably sourced fertilizers and fertilizer alternatives make a difference environmentally. However, the environmental benefits of struvite and biosolids revealed in this study are very dependent on the transportation offset of the synthetic fertilizers, suggesting that local sourcing of fertilizer is as important, if not more important, than the type of fertilizer used. Production of PHA is heavily dependent on which external substrate is used. Additionally, when comparing Scenarios 5 and 6, no meaningful conclusions can be drawn regarding the implementation of PHA production at a WRRF. Scenario 6b performs similarly to 5b in eutrophication and smog categories, and slightly worse in the global warming, air acidification and human health particulate categories. Scenarios 5a and 6a also perform similarly, with 6a performing slightly better than 5a in four of the six environmental categories.

4.4 Research Questions Revisited

What are the industrial waste stream characteristics and how might the waste characteristics dictate transport to the WRRF?

As discussed in section 3.2, four proximate waste streams were considered for integration at the WRRF: dairy manure, TASCO wastewater, Chobani wastewater, and Lamb Weston wastewater. Lamb Weston and Chobani both have high flow rates, with daily flow averages of 1.76 mgd and 0.8 mgd respectively. These high flow rates do not reasonably allow the wastewater to be separately to the WRRF through truck, making the existing sewage collection system the preferred method of transporting the waste to the WRRF. Dairy manure supply was not comprehensively investigated; rather, it was simply assumed that a dairy was located within a “reasonable” distance to the facility. A 5,000 head dairy was assumed, from which 750,000 lb (340,000 kg) per day of wet manure was estimated, based on the DAIRIEES model. Typical maximum weight for trucks is 80 tons, meaning 5 one-way trips from the dairy to the WRRF would be required per day. The TASCO facility on the south side of Twin Falls produces a wastewater rich in VFAs and devoid of nutrients. Its high COD is not ideal for EBPR; however, TASCO flume effluent wastewater has shown potential as a substrate for PHA production. While transport through the existing collection system would be logistically ideal, the waste would be of little value if diluted. As a result, separate transport of this waste via truck is necessary. TASCO estimates that 39,000 gallons per day of waste are available to be utilized for a PHA process. Trucks hauling liquids such as water have a typical capacity of around 10,000 gallons, meaning 4 one-way trips from the TASCO facility to the WRRF would be necessary each day.

The hypothesis coupled with this research question was that separate transport of proximate waste streams is required to retain maximum value for resource recovery. This was found to be partially true, as both dairy manure and TASCO wastes do require separate transport to realize PHA production at the WRRF. However, the conversion of UASBs to fermenters does not require separate transport, as the VFA-rich substrate must be first mixed with the main stream of treatment to induce EBPR at the WRRF.

<u>Leveraging proximate agro-industrial waste streams, what combination of resource recovery opportunities could be achieved at the Twin Falls WRRF?</u>

The four different resource recovery opportunities and process enhancements selected for further investigation in section 3.4 are electricity production through CHP, struvite crystallization, bioplastic production, and enhanced P recovery through EBPR. Table 3.8 details each combination of these opportunities applied and the integration of proximate waste. Modeling of each of these scenarios suggests that the WRRF will be capable of maintaining treatment in compliance with its NPDES permit; the removal of phosphorus to a level below 1 mgP/L is also possible with the addition of EBPR, suggesting that the facility will be able to biologically removing phosphorus in the future once a TMDL limiting phosphorus is added to the middle section of the Snake River.

The facility does also have some room for expansion to accommodate the reconfigurations needed to support these scenarios. The footprint for struvite FBRs is small and should be sited to support the process before the dewatering filtrate is sent to the plant drain. CHP engines should be located near the existing digesters to ensure biogas handling is safe and operationally as convenient as possible. Lastly, PHA production facilities should be considered. To ease delivery of either manure or TASCO wastewater, a site at the top of the canyon would be ideal as a transfer station for TASCO wastewater, or a location to create a dairy manure slurry by mixing it with water; this would prevent trucks from having to drive down in the canyon, as well as provide some additional head to lower pumping requirements associated with PHA production at the WRRF.

What is the optimal WRRF configuration that integrates proximate industrial waste streams and produces resources of highest environmental and economic benefit, and how does this scenario compare with current wastewater management strategies?

While all the studied configurations can achieve resource recovery and meet NPDES permit requirements, only a few scenarios were able to achieve a net environmental benefit for the six environmental categories in TRACI: scenarios 2, 5a, 5b, 5c, 6a, 6b, and 8. Of these scenarios, 5b and 6b both perform significantly better than the others. Scenarios 5b performs slightly better than 6b. As a result, the best overall configuration in this study is that of Scenario 5b, utilizing UASB conversion for EBPR and integrating struvite crystallization and CHP. UASB conversion to fermentation ensures carbon is not converted to biogas and flared early in the treatment process, leaving it available to drive EBPR; this ensures that the maximum carbon from the primary clarifiers is anaerobically digested, as opposed to being fermented for EBPR. Scenario 5b also adds less complexity than the next best scenario, 6a, which uses dairy manure to drive PHA production. The uncertainty in the future of commercial PHA production on mixed cultures, combined with the increase in cost associated with PHA production make Scenario 5b preferable to Scenario 6a.

When compared to current wastewater management strategies employed at the Twin Falls WWTP, Scenario 5b makes better use of carbon, opting not to convert it to carbon dioxide as soon as possible by flaring UASB biogas. The complexity associated with operation of a fermenter as opposed to a UASB should be negligible; safety requirements would also be expected to be reduced as flammable biogas would no longer be produced. Phosphorus recovery would be achieved, which will likely be necessary in the future due to more stringent permit requirements that are expected to be enacted in the near future. Lastly, the phosphorus recovery process would maintain resiliency through the addition of struvite crystallization, removing most of the phosphorus from the dewatering filtrate stream that is currently returned to the headworks of the plant. While this would add some operational, logistical, and economic complexity, struvite crystallization technologies have been shown to produce a valuable resource that is capable of offsetting the costs associated with its installation and operation.

4.5 Preliminary Economic Analysis

As a complement to the ELCA, a preliminary economic analysis was conducted to illustrate some of the costs and revenues associated with the operation of a WRRF (Coats and Wilson 2017). Capital costs, operational costs, and revenues were all estimated for CHP, struvite crystallization, and PHA production. It was assumed that any costs associated with EBPR (UASB conversion or primary solids fermenter) were negligible. Capital costs for CHP were estimated to be $5,000 per kW (Combined Heat and Power Partnership et al. 2011). To estimate the cost reduction in electricity savings, it was assumed that the WRRF pays $0.05

per kWh as a Schedule 9 business (Idaho Power 2019b); it was further assumed that the additional operational expenses associated with operation and maintenance of CHP was negligible. The capital costs for the purchase and installation of struvite crystallization reactors was estimated to be $1,800,000 for a WRRF treating the wastes equivalent to 100,000 persons, which is close in size to the Twin Falls WRRF. Price estimates for struvite capital cost were based on conversion of 1,417,000 Euros in 2006 to United States dollars in 2019, based on a study by Montag et. al. (Montag, Gethke and Pinnekamp 2009). Struvite was assumed to sell for $0.50 per pound based on economic evaluation performed by the nearby city of Boise, ID for its struvite production (Barbeau, Kresge and Bower 2009). The cost of producing struvite was estimated to be $150 per tonne (Forrest et al. 2008). PHA cost estimates were based in part on the DAIRIEES model (Guillen 2017); however, as no commercial PHA production has been achieved using mixed cultures at a WRRF, these costs are subject to a high degree of uncertainty. Capital costs were estimated to be $2,000,000, while operational costs from DAIRIEES were estimated to be $250,000 per year. DAIRIEES also does not account for the costs associated with the transportation of manure. Manure and TASCO transportation costs were assumed to be $0.05 per gallon, which was added to the estimated operations and maintenance. Lastly, it was assumed that the PHA produced at the facility would be sold at $2.00 per pound. Table 4.5 summarizes the aforementioned cost values (capital costs included in calculations but not shown), producing a net present value for each scenario based on a 20-year return period and 6% discount rate.

Table 4.5: Preliminary Cost Estimate

	Revenue and Savings			Operations and Maintenance			Net Present Value
Scenario	**CHP**	**Struvite**	**PHA**	**CHP**	**Struvite**	**PHA**	
1	$0	$0	$0	$0	$0	$0	$0
2	$253,646	$0	$0	$0	$0	$0	-$13,797
3	$0	$567,374	$0	$0	$374,467	$0	-$412,631
4a	$0	$551,625	$0	$0	$364,072	$0	-$351,210
4b	$0	$516,840	$0	$0	$341,114	$0	-$215,559
5a	$184,004	$551,625	$0	$0	$364,072	$0	-$361,220
5b	$446,672	$516,840	$0	$0	$341,114	$0	-$239,856
5c	$356,620	$564,290	$0	$0	$372,431	$0	-$420,002
6a	$232,359	$606,630	$576,700	$0	$400,376	$328,131	-$1,429,429
6b	$460,469	$445,829	$576,700	$0	$294,247	$328,131	-$814,751
7a	$0	$552,209	$193,888	$0	$364,458	$328,131	$3,186,266
7b	$0	$486,253	$193,888	$0	$320,927	$328,131	$3,443,477
8	$247,689	$0	$576,700	$0	$0	$328,131	-$864,543

While preliminary, the cost analysis suggests that the production of resources at a WRRF has the potential to offset the costs associated with the production of said resources, providing not only economic benefit to the WRRF, but also environmental benefit. The exception in this study is scenario 7, utilizing TASCO waste for PHA production. As mentioned previously, the concentration of VFAs in TASCO wastewater was tested to be significantly lower than previously tested VFA concentrations. Due to the low number of samples tested over time by the University of Idaho Engineering Lab, the estimated value has a high uncertainty. If testing could ascertain a higher VFA concentration, similar to that of a previous test, nearly five times the PHA production could be realized, which suggests that the economics of PHA production with TASCO waste could also be favorable.

5. Conclusions

This thesis has presented a unique perspective on the conversion of a wastewater treatment plant into a water resource recovery facility. Twin Falls is uniquely positioned in this regard due to its ability to integrate proximate waste streams. This study investigated the integration of four proximate waste streams (wastewater from Chobani, Lamb Weston, TASCO, and dairy manure) to achieve resource recovery in the form of electricity, struvite, bioplastics, as well as improving effluent quality by removing phosphorus biologically. By utilizing the EPA TRACI model to compare environmental performance with respect to global warming, acidification of air, eutrophication of air, eutrophication of water, smog production, and particulate production, it was found that the best resource recovery integrating wastewater management practice was the scenario in which pretreatment of Chobani and Lamb Weston wastewater was altered from UASB pretreatment to one in which the wastewater is fermented, producing VFAs to drive EBPR at the WRRF. By operating in this manner, anaerobic digestion is maximized to produce biogas, which can be used for electricity production. Struvite crystallization helps maintain a resilient EBPR process, while creating a resource of agronomic value. Overall, this thesis has established that resource recovery integration is possible while concurrently achieving permit requirements, demonstrating that the conversion of a WWTP to a WRRF yields great environmental benefits.

5.1 Future Work

This study utilized desktop modeling to evaluate place-based resource recovery. Several areas for improvement in future work were identified. More collaboration with the Twin Falls WWTP is recommended to further verify data, as well as improve on the assumptions made in model calibration. The analytical measurement of several variables was not possible in this study, and future work would greatly benefit from more analytical data.

Additionally, data for emissions associated with the production of sodium hydroxide, magnesium hydroxide, and diammonium phosphate, were based on global estimates.

Future work should consider consulting with local and regional chemical suppliers and manufacturers to be able to accurately determine where the products are produced as well as validate that the quantity of emissions associated with their production are accurate. One possible solution to issue is to utilize LCA database software, which was not available for use in this study.

As noted by Solon et. al. (2019), wastewater treatment modeling does not sufficiently incorporate resource recovery. The inclusion of a metabolic or kinetic model for the production of PHA is an area of possible improvement. Additionally, the model often had trouble reaching a steady state solution in scenarios that included struvite crystallization; chemical inputs to the struvite process had to be manually estimated and then input into the model. Combined with the difficulty in reaching a solution, this process proved quite cumbersome in estimating both the struvite precipitation and the chemical requirements for both magnesium and pH adjustment. As adoption of struvite crystallization technologies continues, the inclusion of a separate unit to model this product would be ideal to allow the modeler to select the chemicals used for both pH adjustment and magnesium source, as well as select a desired effluent P from the struvite system to be recycled to the headworks.

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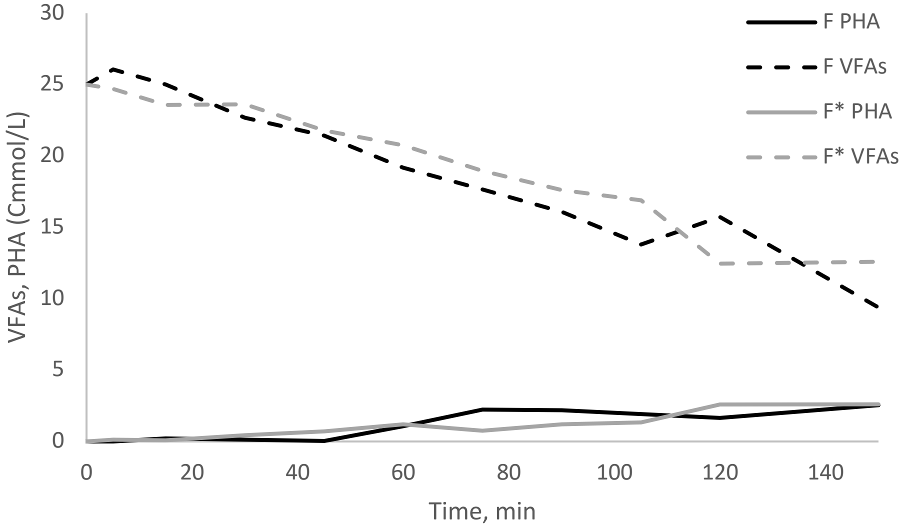
Appendix A: Waste activated sludge PHA batch test

To better understand the potential for the accumulation of PHA within an MMC from an EBPR system, 0.5 L of waste activated sludge (WAS) was taken from 2 separate reactors, labeled F-EBPR and F*-EBPR. F-EBPR has been operating for 10 years, fed a mixture of real wastewater collected from the Moscow, ID WRRF and fermentate rich in VFAs provided by a lab scale primary sludge fermenter with sludge from the Pullman, WA WRRF. This setup routinely achieves a low effluent phosphorus number of 0.1 mgP/L. F*-EBPR operates identically to F and achieves very similar results. WAS was taken from each reactor at the end of the anaerobic period as to ensure PHA enzymes are activated (Probst 2016). Aeration was achieved using ceramic fine bubble diffusers to target dissolved oxygen (DO) levels of 6 mg/L. Each reactor was pulsed with 0.45 L of fermenter liquor to achieve a bulk solution VFA concentration of 25 Cmmol/L. Samples were collected in 15-minute intervals, measuring for bulk solution VFAs and intracellular PHA. DO and pH were monitored continuously using a Hach HQ40.



At the end of 150 minutes, intracellular PHA percentages on a dry weight basis were 3.2% and 2.9% for F-EBPR and F*-EBPR respectively. The maximum theoretical yield based on the quantity of VFAs consumed and the MLSS is 0.269 and 0.168 for F-EBPR and F*-EBPR respectively. This suggests that the MMC does not contain a significant amount of PHA producers, resulting in ineffective PHA production if using solely the MMC. The conclusion drawn from this experiment is that PHA producers must be separately enriched for in the side-stream to produce a commercially viable quantity of PHA.

Appendix B: Influent Characteristics of all Operator-Collected Data

Key parameters	Default	Average	DS 1	DS 2	DS 3	DS 4	DS 5	DS 6	DS 7	DS 8
COD of biomass in volatile solids	1.42	1.43	1.42	1.42	1.5	1.42	1.42	1.42	1.42	1.42
COD of biodegradable substrate in volatile solids	1.8	2.07	1.6	2.1	2.2	2.2	2.3	1.15	3.2	1.8
General stoichiometry										
COD of particulate unbiodegradable organics in volatile solids	1.3	1.51	1.3	1.5	1.5	1.3	1.65	1.1	2.4	1.3
COD of endogenous products in volatile solids	1.42	1.43	1.42	1.42	1.5	1.42	1.42	1.42	1.42	1.42
COD of PHA in volatile solids	1.67	1.67	1.67	1.67	1.67	1.67	1.67	1.67	1.67	1.67
Yield on ultimate BOD	0.95	1.00	1.05	0.95	1.10	0.95	0.95	1.10	0.95	0.95
Fraction of BOD5 to ultimate BOD in soluble biodegradable substrates (-)	0.90	0.93	1.00	0.90	1.00	0.90	0.90	0.90	0.90	0.90
Fraction of BOD5 to ultimate BOD in colloidal biodegradable substrates (-)	0.60	0.63	0.70	0.60	0.70	0.60	0.60	0.60	0.60	0.60
Fraction of BOD5 to ultimate BOD in particulate biodegradable substrates (-)	0.50	0.53	0.60	0.50	0.60	0.50	0.50	0.50	0.50	0.50
Influent components										
Flow rate	N/A	7.9	8.38	7.61	7.65	8	7.97	8.15	7.64	7.87
Total COD	N/A	987.3	810	915	1308	844	1238	890	1020	873.6
TKN	N/A	57.6	77.3	65.9	65.1	63.6	41.8	44.9	38.5	64
Total phosphorus	N/A	10.7	11.1	11.4	12.3	11.2	10.6	8.48	9.23	11
pH and alkalinity										

Alkalinity	N/A	667.5	750	690	710	700	670	690	520	610
pH	N/A	7.9	7.95	7.92	7.89	8.09	8.04	7.8	7.57	7.68
Influent fractions										
Fraction of VSS/TSS	84.86	86.51	88.72	86.67	89.47	91.38	77.90	85.71	87.39	84.85
Fraction of filtered COD (SCCOD, 1.5 µm, incl. colloids) in total COD (TCOD)	40.48	54.43	57.16	43.93	57.65	47.63	64.46	75.73	40.00	48.90
Fraction of flocculated filtered (SCOD, wo colloids) COD in total COD (TCOD)	20.24	41.48	32.59	34.64	41.67	35.90	55.98	61.24	24.00	45.79
Fraction of VFA in filtered COD (SCCOD, 1.5 µm, incl. colloids)	11.76	1.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	8.29
Fraction of soluble unbiodegradable organics (SU) in filtered COD (SCCOD, 1.5 µm, incl. colloids)	11.76	40.76	48.81	75.37	38.46	35.07	29.95	19.29	31.86	47.28
Fraction of particulate unbiodegradable organics (XU) in total COD (TCOD)	14	9.375	14	7	7	5	7	7	14	14
Fraction of heterotrophs (OHO) in total COD (TCOD)	5	4	5	5	3	3	3	3	5	5
Fraction of endogenous products (XE) in total COD (TCOD)	20	18.1	20	20	15	20	10	20	20	20
Fraction of colloidal unbiodegradable organics (CU) in colloidal COD (SCCOD-SCOD)	20	17.5	15	20	15	20	10	20	20	20
Fraction of NHx in total Kjeldahl nitrogen (TKN)	69.8	54.5	43.6	58.6	50.1	54.6	59.6	39.4	55.6	74.4
Fraction of PO4 in total phosphorus (TP)	58.1	60.3	50.3	65.4	53.3	62.5	66.5	66.0	60.1	58.2

Fraction of N in readily biodegradable substrate (SB)	4	3.4	4	4	4	4	2	4	1	4
Fraction of N in particulate unbiodegradable substrate (XU)	1	0.9	1	1	1	1	0.5	1	0.5	1
Fraction of P in readily biodegradable substrate (SB)	1	0.7	1	1	0.5	1	0.2	0.1	0.5	1
Fraction of P in particulate unbiodegradable substrate (XU)	0.1	0.1	0.1	0.1	0.1	0.1	0.01	0.01	0.1	0.1
Ionic components										
Calcium	150	150	150	150	150	150	150	150	150	150
Magnesium	15	15	15	15	15	15	15	15	15	15
Potassium	16	16	16	16	16	16	16	16	16	16
Anions (as chloride)	300	300	300	300	300	300	300	300	300	300
Sodium (strong cation)	110	110	110	110	110	110	110	110	110	110
Influent constituents (usually zero or otherwise calculated)										
Dissolved oxygen (O2)	0	0	0	0	0	0	0	0	0	0
Methanol (MEOL)	0	0	0	0	0	0	0	0	0	0
Stored polyhydroxyalkanoates (PHA)	0	0	0	0	0	0	0	0	0	0
Stored Glycogen (GLY)	0	0	0	0	0	0	0	0	0	0
Anaerobic endogenous decay products	0	0	0	0	0	0	0	0	0	0
Phosphorus accumulating organisms (PAO)	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Glycogen accumulating organisms (GAO)	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Anoxic methanol utilizers (MEOLO)	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1

Aerobic nitrifying organisms (NITO)	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Acidoclastic methanogens (AMETO)	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Hydrogenotrophic methanogens (HMETO)	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Nitrite and nitrate (NOx)	0	0.126	0.118	0.298	0.107	0.065	0.183	0.094	0.144	0
Dissolved nitrogen (N2)	16	16	16	16	16	16	16	16	16	16
Stored polyphosphate (PP)	0	0	0	0	0	0	0	0	0	0
Dissolved methane (CH4)	0	0	0	0	0	0	0	0	0	0
Dissolved hydrogen (H2)	0	0	0	0	0	0	0	0	0	0
Ferric ion (Fe3)	0	0	0	0	0	0	0	0	0	0
Ferrous ion (Fe2)	0	0	0	0	0	0	0	0	0	0
Aged unused hydrous ferric oxide (HFO,old)	0	0	0	0	0	0	0	0	0	0
P-bound hydrous ferric oxide, high surface (HFO,H,P)	0	0	0	0	0	0	0	0	0	0
P-bound hydrous ferric oxide, low surface (HFO,L,P)	0	0	0	0	0	0	0	0	0	0
Aged used hydrous ferric oxide, high surface (HFO,H,P,old)	0	0	0	0	0	0	0	0	0	0
Aged used hydrous ferric oxide, low surface (HFO,L,P,old)	0	0	0	0	0	0	0	0	0	0
Calcium carbonate (CaCO3)	0	0	0	0	0	0	0	0	0	0
Amorphous calcium phosphate (ACP)	0	0	0	0	0	0	0	0	0	0
Struvite (STR)	0	0	0	0	0	0	0	0	0	0
Vivianite (Vivi)	0	0	0	0	0	0	0	0	0	0

Appendix C: Model Calibration Comparing Predicted Values and Measured Data

Analyte	Prim. Eff.		1		S3		IFAS 2A		IFAS 2B		Middle AB		AB Effluent		Sec. Eff.	
	Field	SUMO	Field	SUMO	Field	SUMO	Field	SUMO	Field	SUMO	Field	SUMO	Field	SUMO	Field	SUMO
TSS	92 ± 40.4	125.0					2562.9 ± 803.5	2433.7	2591.4 ± 696.5	2430.1	2338.6 ± 320.4	2419.8			66.6 ± 77.6	5.0
VSS	86.4 ± 39.9	107.7					2258.6 ± 654.3	2006.8	2220 ± 565.8	1998.3	2035.7 ± 297.6	1984.7			65.8 ± 69.9	4.1
COD	768.3 ± 158.3	719.1													382.1 ± 150.6	182.7
sCOD	513.1 ± 159.8	520.3	416.4 ± 150.6	359.7	454.9 ± 256.6	302.1	340.2 ± 96	179.7	388.5 ± 206.7	176.7	327.2 ± 171.4	176.7	294.3 ± 104.1	176.7	214 ± 63.3	176.7
BOD	240.7 ± 137.6	319.1													25.7 ± 30.8	2.3
TKN (unfiltered)															10 ± 8.8	10.0
NH4	38.1 ± 13	40.3	29.9 ± 16.5	34.8	20.6 ± 5.3	25.6	11.7 ± 6.9	11.9	10.1 ± 3.5	2.8	1.2 ± 1.7	0.2	1.1 ± 1.6	0.1	1.2 ± 2	0.1
NO3+NO2			0.7 ± 0.7	0.0	2.4 ± 2.7	0.7	6.2 ± 6.7	4.9	1.8 ± 1.5	10.8	14 ± 10	14.2	13.8 ± 10.3	14.8	17.1 ± 5.9	14.8
TP (unfiltered)															6.5 ± 2.3	5.2
PO4	8.3 ± 3.8	11.2	12.7 ± 6.3	13.5	13.8 ± 6	11.7	10.7 ± 3.8	8.1	8.7 ± 2.9	6.7	7.7 ± 3.7	5.1	7.7 ± 3.8	4.7	4.8 ± 3	4.7

Appendix D: VFA degradation within collection system

Several scenarios studied in this thesis require the conversion of UASB pretreatment to a fermenter, thus sending a VFA rich substrate through the collection system to the WRRF where it can be utilized to drive EBPR. Municipal wastewater traveling through the collection system is expected to contain some level of heterotrophic microorganisms, which would be able to utilize the minimal dissolved oxygen in the wastewater to consume some fraction of the VFAs present.

To account for this, two one-liter reactors were set up to measure the concentration of VFAs over time. Raw wastewater was obtained from the Moscow, ID WRRF and mixed with fermenter liquor from a lab operated fermenter operated with primary solids obtained from the Pullman, WA WWTP. One reactor was operated with a volumetric mixture of 90% wastewater and 10% fermenter liquor, the other at a ratio of 75% wastewater and 25% fermenter liquor. The reactors were gently mixed to allow some surface oxygen transfer, mimicking the environment expected in a sewage collection system. Sample were collected every 1.25 hours until the 6.25 hour mark, with one final sample collected later at 23 hours. The data is plotted in Figure D.1 with the ratio of the VFA concentration at each time point to the initial VFA concentration. Based on the geography of Twin Falls and a conservatively estimated sewer flow velocity of 1.5 ft/sec, it was assumed that the duration of travel to the WRRF would be approximately 12 hours. Based on this estimate a value of 60% was selected, meaning 60% of the VFAs produced at the fermenter would remain at the treatment facility.

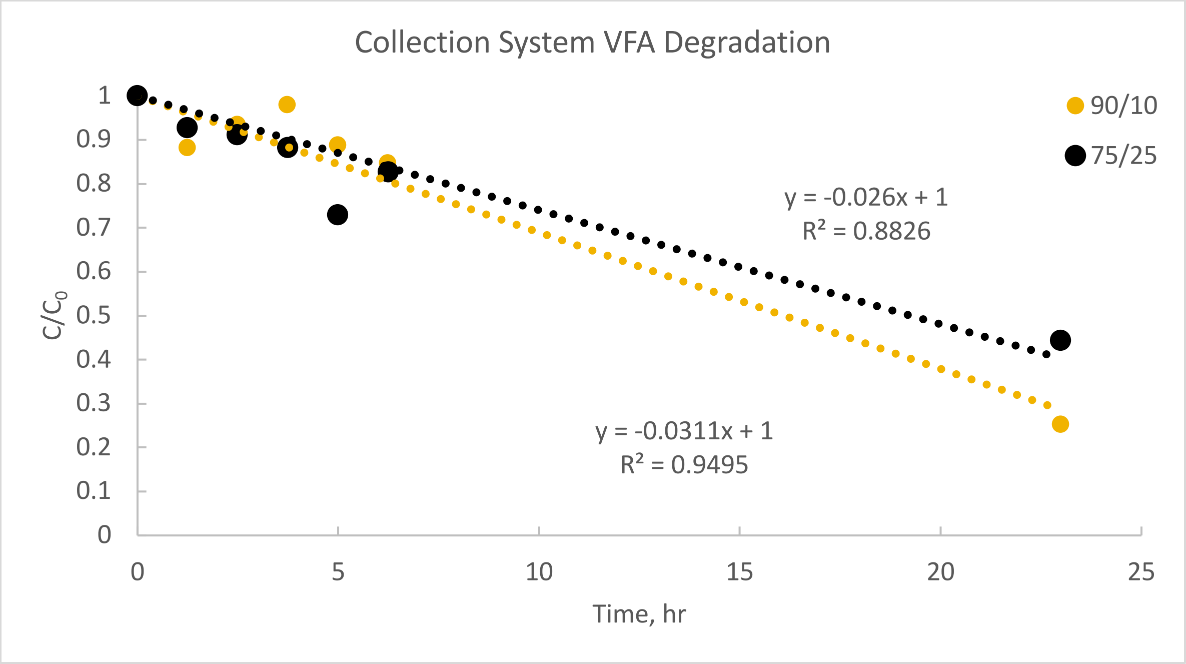


Figure D.1: Collection System VFA Degradation

Appendix E: Sample Calculations

The calculation below illustrates how the functional unit was integrated into the LCA using effluent phosphorus. The product of effluent flow rate and phosphorus concentration is multiplied by the functional unit and divided by the influent mass flow of COD in pounds per day, and units are converted to cancel out remaining dimensions. The result in this case, is a mass of effluent phosphorus associated with the functional unit, 1,000 pounds of influent COD.

7,890,000 ~~gal~~	0.55 ~~mg~~ P	3.785 ~~L~~	1,000 ~~lb COD~~	~~day~~	lb
~~day~~	~~L~~	~~gal~~		127,262 ~~lb COD~~	453,592 ~~mg~~

= 0.28 lb P

Appendix F: Chemical flow for each scenario (TRACI input)

Scenario	1	2	3	4a	4b	5a	5b	5c	6a	6b	7a	7b	8
Carbon Dioxide	-7.35E-01	-5.60E+07	1.16E+07	1.37E+07	2.72E+07	-2.69E+07	-5.90E+07	-4.42E+07	-3.78E+07	-4.86E+07	1.35E+07	3.30E+07	-5.15E+07
Methane	-7.55E-03	-7.55E-03	-5.15E-03	-3.63E-03	-3.10E-03	-3.63E-03	6.26E-04	1.83E+00	1.83E+00	1.83E+00	2.13E-01	2.14E-01	1.82E+00
Nitrous Oxide	-7.05E-03	-7.05E-03	-4.81E-03	-3.39E-03	-2.90E-03	-3.39E-03	5.85E-04	1.71E+00	1.70E+00	1.71E+00	1.98E-01	2.00E-01	1.70E+00
VOCS	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	-2.04E-05	-2.04E-05	-6.07E-06	-6.07E-06	-2.04E-05
Nitrogen Oxides	0.00E+00	-2.16E+05	1.73E+04	1.61E+04	7.13E+04	-1.41E+05	-2.76E+05	-2.45E+05	-1.75E+05	-2.57E+05	1.65E+04	8.63E+04	-1.98E+05
PM10	0.00E+00	0.00E+00	9.78E-09	9.78E-09	1.01E-08	9.78E-09	1.47E-08	1.61E-08	1.15E-08	1.15E-08	8.12E-09	1.42E-08	0.00E+00
PM2.5	-2.46E-02	-2.46E-02	-9.99E-02	-1.03E-01	-8.18E-02	-1.03E-01	-1.19E-01	-1.15E-01	-1.03E-01	-1.02E-01	-9.39E-02	-9.56E-02	-2.79E-02
Ethane	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	-5.32E-02	-5.32E-02	-1.58E-02	-1.58E-02	-5.32E-02
Ethylbenzene	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
1,1,1-Trichloroethane	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Benzene	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	-2.22E-05	-2.22E-05	-6.59E-06	-6.59E-06	-2.22E-05
Toluene	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	-2.05E-05	-2.05E-05	-6.07E-06	-6.07E-06	-2.05E-05
dimethylphthalate (DMP)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	-1.49E-05	-1.49E-05	-4.42E-06	-4.42E-06	-1.49E-05
Phenol	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	-2.63E-06	-2.63E-06	-7.82E-07	-7.82E-07	-2.63E-06
Dichloroethanes	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	-6.43E-07	-6.43E-07	-1.91E-07	-1.91E-07	-6.43E-07
Chloroform	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Mercury(II)	0.00E+00	0.00E+00	2.19E-11	2.19E-11	2.25E-11	2.19E-11	3.28E-11	3.60E-11	-5.25E-08	-5.25E-08	-1.56E-08	-1.56E-08	-5.26E-08
Arsenic(III)	0.00E+00	0.00E+00	1.88E-11	1.88E-11	1.93E-11	1.88E-11	2.82E-11	3.10E-11	2.21E-11	2.21E-11	1.56E-11	2.73E-11	0.00E+00
Cadmium(II)	0.00E+00	0.00E+00	1.61E-12	1.61E-12	1.65E-12	1.61E-12	2.41E-12	2.64E-12	1.89E-12	1.89E-12	1.33E-12	2.33E-12	0.00E+00
Chromium(III)	0.00E+00	0.00E+00	3.51E-11	3.51E-11	3.61E-11	3.51E-11	5.26E-11	5.77E-11	4.12E-11	4.12E-11	2.91E-11	5.09E-11	0.00E+00
Nickel(II)	0.00E+00	0.00E+00	3.69E-11	3.69E-11	3.79E-11	3.69E-11	5.53E-11	6.07E-11	4.34E-11	4.34E-11	3.06E-11	5.36E-11	0.00E+00
Lead(II)	0.00E+00	0.00E+00	8.63E-11	8.63E-11	8.88E-11	8.63E-11	1.29E-10	1.42E-10	1.01E-10	1.01E-10	7.16E-11	1.25E-10	0.00E+00
Sulfur Dioxide	0.00E+00	-1.66E+05	4.15E+04	4.99E+04	8.92E+04	-7.05E+04	-1.62E+05	-1.12E+05	-1.05E+05	-1.27E+05	4.91E+04	1.08E+05	-1.53E+05
Sulfur Oxides (SOX)	0.00E+00	0.00E+00	1.30E-07	1.30E-07	1.34E-07	1.30E-07	1.95E-07	2.14E-07	1.53E-07	1.53E-07	1.08E-07	1.89E-07	0.00E+00
Biological Oxygen Demand	0.00E+00	0.00E+00	1.20E-07	1.20E-07	1.24E-07	1.20E-07	1.80E-07	1.98E-07	1.41E-07	1.41E-07	9.98E-08	1.75E-07	0.00E+00
Phosphoric Acid	-1.58E-01	-1.58E-01	-6.40E-01	-6.57E-01	-5.24E-01	-6.57E-01	-7.63E-01	-7.34E-01	-6.62E-01	-6.53E-01	-6.01E-01	-6.12E-01	-1.79E-01

Ammonia	-5.48E-03	-5.48E-03	-2.22E-02	-2.28E-02	-1.82E-02	-2.28E-02	-2.65E-02	-2.55E-02	-2.30E-02	-2.27E-02	-2.09E-02	-2.13E-02	-6.20E-03
Chemical Oxygen Demand	9.45E+01	9.45E+01	9.42E+01	9.44E+01	6.67E+01	9.44E+01	9.73E+01	8.99E+01	8.82E+01	9.23E+01	1.07E+02	1.09E+02	8.91E+01
Ammonium	9.59E-02	9.59E-02	1.10E-01	3.33E-01	1.60E-01	3.33E-01	2.33E-01	6.14E-02	2.67E-01	3.35E-01	1.08E-01	3.88E-01	9.10E-02
Nitrate	7.72E+00	7.72E+00	6.09E+00	6.03E+00	1.54E+00	6.03E+00	2.25E+00	7.39E+00	5.23E+00	2.96E+00	4.82E+00	1.81E+00	8.37E+00
Phosphate	3.27E+00	3.27E+00	5.08E-01	2.86E-01	2.09E-01	2.86E-01	3.05E-01	3.26E-01	2.76E-01	2.82E-01	2.99E-01	2.82E-01	4.19E+00

Appendix G: Full TRACI Results Table

Scenario	Global Warming Air	Acidification Air	HH Particulate Air	Eutrophication Air	Eutrophication Water	Ozone Depletion Air (kg CFC-11 eq / kg substance)	Smog Air
1	-3.0253728	-0.16496	-0.02501	1.427115	14.02948	0	-0.00011
2	-56018017	-317155.4	-11704	-9562.36	-62827.9	0	-5353817
3	11635418.2	53534.531	2657.817	764.6195	5028.896	0	4.28E+05
4a	13714688.3	61193.09	3167.874	712.1937	4684.43	0	398616.6
4b	27221941.5	139071.28	5964.18	3157.167	20748.54	0	1767406
5a	-26919767	-168865.6	-5322	-6225.2	-40900	0	-3484946
5b	-58963964	-355774	-11916.3	-12239.3	-80417.6	0	-6851559
5c	-44167689	-283613.2	-8611.57	-10862.5	-71371.1	0	-6080916
6a	-37815294	-227004.5	-7663.89	-7733.78	-50812.9	0	-4329427
6b	-48617695	-306227.1	-9588.76	-11366.8	-74684.8	0	-6363171
7a	13536916.6	60698.332	3121.306	732.5001	4818.473	0	410009.6
7b	32955554.7	168362.71	7220.368	3822.174	25120.32	0	2139660
8	-51474367	-291434.9	-10754.9	-8786.41	-57729.6	0	-4919636

Scenario	Ecotox. CF, Em.airU, freshwater	Ecotox. CF, Em.airC, freshwater	Ecotox. CF, Em.fr.waterC, freshwater	Ecotox. CF, Em.sea waterC, freshwater	Ecotox. CF, Em.nat.soilC, freshwater	Ecotox. CF, Em.agr.soilC, freshwater	CF Flag Ecotox
1	0	0	0	0	0	0	0
2	0	0	0	0	0	0	0
3	6.48E-07	6.56E-07	1.41E-06	3.95E-27	8.25E-07	8.25E-07	0
4a	6.48E-07	6.56E-07	1.41E-06	3.95E-27	8.25E-07	8.25E-07	0
4b	6.66E-07	6.75E-07	1.45E-06	4.07E-27	8.49E-07	8.49E-07	0
5a	6.48E-07	6.56E-07	1.41E-06	3.95E-27	8.25E-07	8.25E-07	0
5b	9.71E-07	9.85E-07	2.12E-06	5.93E-27	1.24E-06	1.24E-06	0
5c	1.07E-06	1.08E-06	2.32E-06	6.51E-27	1.36E-06	1.36E-06	0
6a	-0.00101	-0.00097	-0.00872	-4.3E-07	-0.00137	-0.00137	0
6b	-0.00101	-0.00097	-0.00872	-4.3E-07	-0.00137	-0.00137	0
7a	-0.0003	-0.00029	-0.00259	-1.3E-07	-0.00041	-0.00041	0
7b	-0.0003	-0.00029	-0.00259	-1.3E-07	-0.00041	-0.00041	0
8	-0.00101	-0.00097	-0.00872	-4.3E-07	-0.00137	-0.00137	0

Scenario	Human health CF, Emission to urban air, cancer	Human health CF, Emission to urban air, non-canc.	Human health CF, Emission to cont. rural air, cancer	Human health CF, Emission to cont. rural air, non-canc.	Human health CF, Emission to cont. freshwater, cancer	Human health CF, Emission to cont. freshwater, non-canc.	Human health CF, Emission to cont. sea water, cancer	Human health CF, Emission to cont. sea water, non-canc.	Human health CF, Emission to cont. natural soil, cancer	Human health CF, Emission to cont. natural soil, non-canc.	Human health CF, Emission to cont. agric. Soil, cancer	Human health CF, Emission to cont. agric. Soil, non-canc.	CF Flag HH carcinogenic	CF Flag HH non-carcinogenic
1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
3	1.64E-13	1.9E-11	1.67E-13	1.99E-11	1.1E-14	8.36E-13	2.83E-15	3E-13	6.27E-15	4.98E-13	4.56E-13	5.49E-11	0	0
4a	1.64E-13	1.9E-11	1.67E-13	1.99E-11	1.1E-14	8.36E-13	2.83E-15	3E-13	6.27E-15	4.98E-13	4.56E-13	5.49E-11	0	0
4b	1.68E-13	1.96E-11	1.72E-13	2.05E-11	1.13E-14	8.6E-13	2.92E-15	3.08E-13	6.45E-15	5.12E-13	4.7E-13	5.65E-11	0	0
5a	1.64E-13	1.9E-11	1.67E-13	1.99E-11	1.1E-14	8.36E-13	2.83E-15	3E-13	6.27E-15	4.98E-13	4.56E-13	5.49E-11	0	0
5b	2.45E-13	2.86E-11	2.5E-13	2.99E-11	1.65E-14	1.25E-12	4.25E-15	4.49E-13	9.41E-15	7.47E-13	6.85E-13	8.23E-11	0	0
5c	2.69E-13	3.14E-11	2.75E-13	3.28E-11	1.81E-14	1.38E-12	4.67E-15	4.93E-13	1.03E-14	8.2E-13	7.52E-13	9.04E-11	0	0
6a	-3.7E-10	-4.3E-08	-3.8E-10	-4.5E-08	-1.2E-11	-7.5E-10	-5.6E-12	-5.9E-10	-6.9E-12	-5.3E-10	-1.1E-09	-1.2E-07	0	0
6b	-3.7E-10	-4.3E-08	-3.8E-10	-4.5E-08	-1.2E-11	-7.5E-10	-5.6E-12	-5.9E-10	-6.9E-12	-5.3E-10	-1.1E-09	-1.2E-07	0	0
7a	-1.1E-10	-1.3E-08	-1.1E-10	-1.3E-08	-3.5E-12	-2.2E-10	-1.7E-12	-1.8E-10	-2E-12	-1.6E-10	-3.1E-10	-3.7E-08	0	0
7b	-1.1E-10	-1.3E-08	-1.1E-10	-1.3E-08	-3.5E-12	-2.2E-10	-1.7E-12	-1.8E-10	-2E-12	-1.6E-10	-3.1E-10	-3.7E-08	0	0
8	-3.7E-10	-4.3E-08	-3.8E-10	-4.5E-08	-1.2E-11	-7.5E-10	-5.6E-12	-6E-10	-6.9E-12	-5.3E-10	-1.1E-09	-1.2E-07	0	0

Appendix H: Sensitivity Analysis

Figure H.1: Sensitivity Analysis with Reduced Emissions due to Renewable Energy

Figure H.2: Sensitivity Analysis with Reduced Struvite Chemical Production Emissions

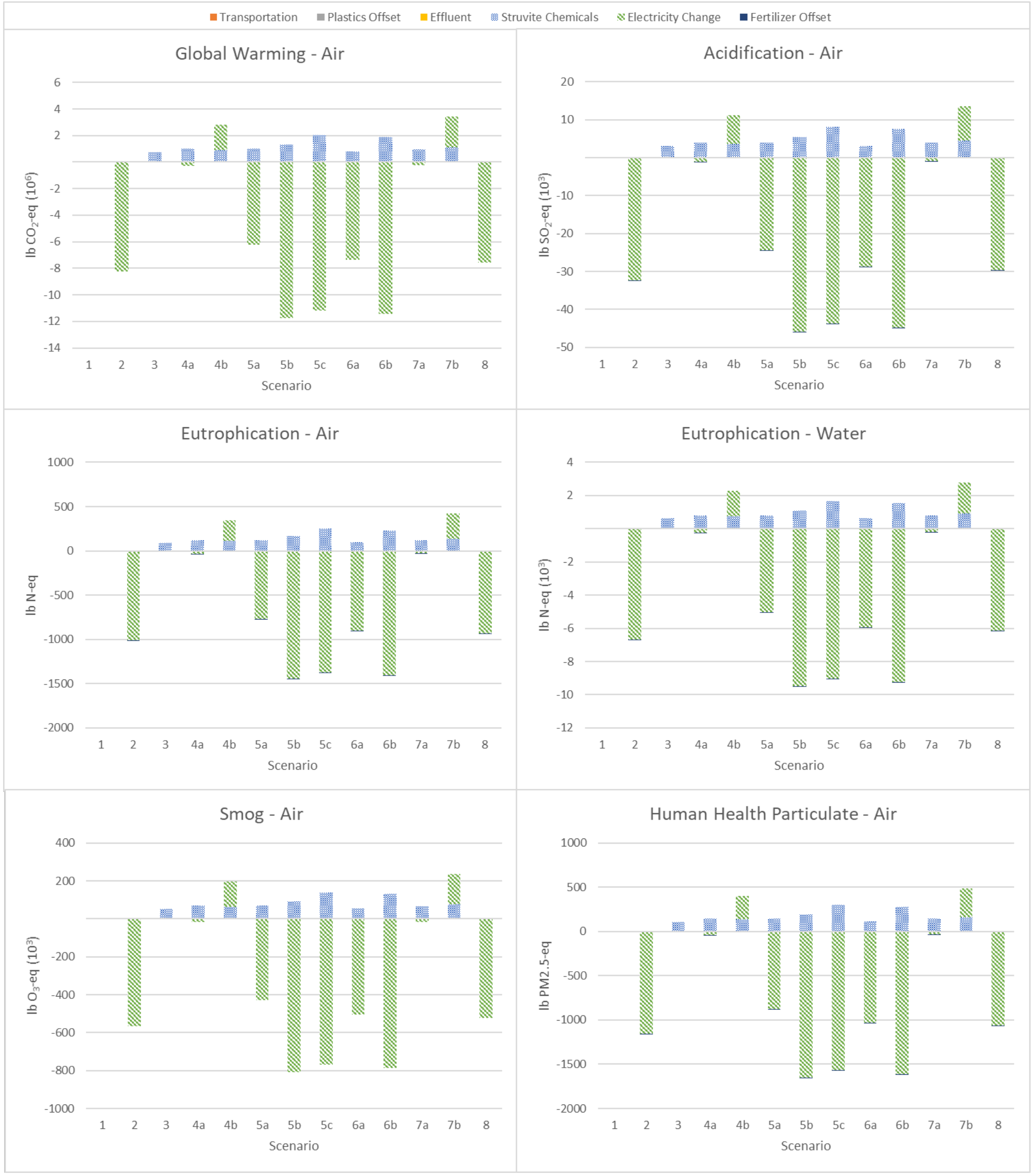


Figure H.3: Sensitivity Analysis with Reduced Emissions from Electricity and Struvite Chemicals

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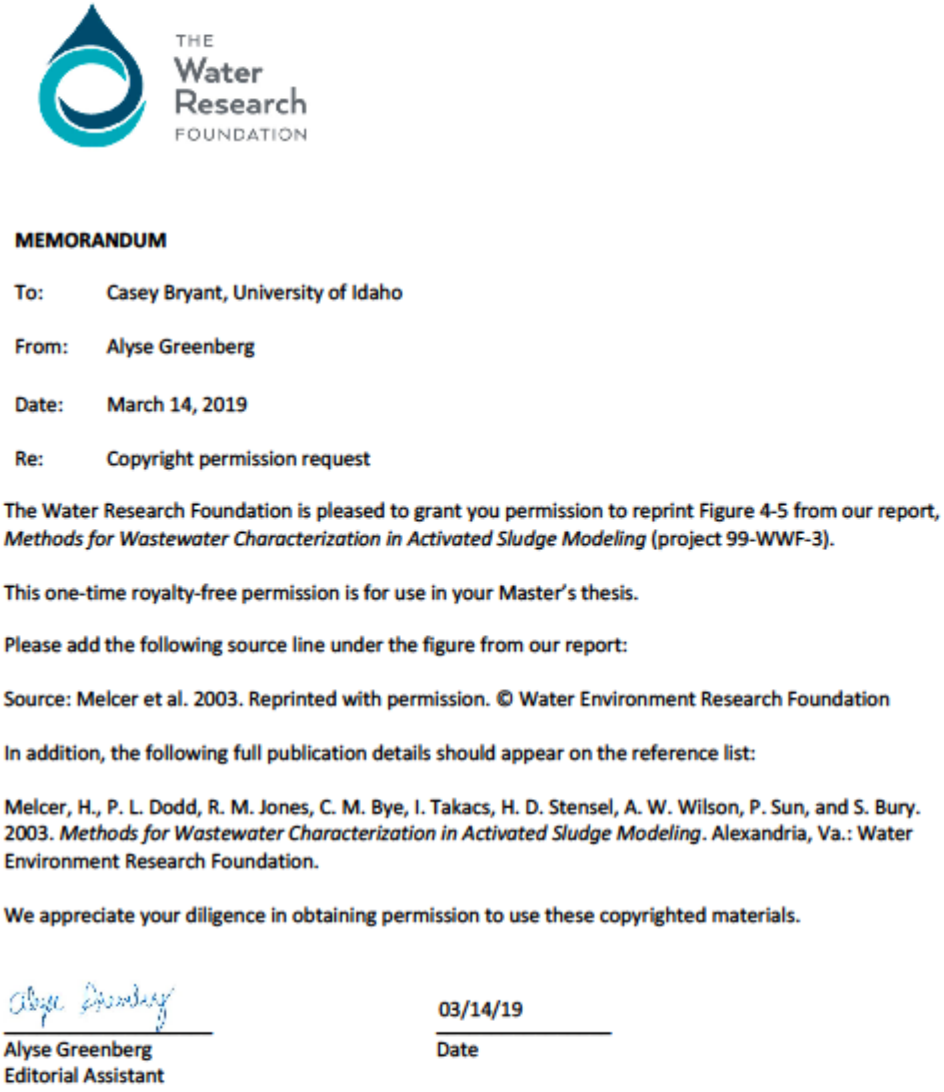

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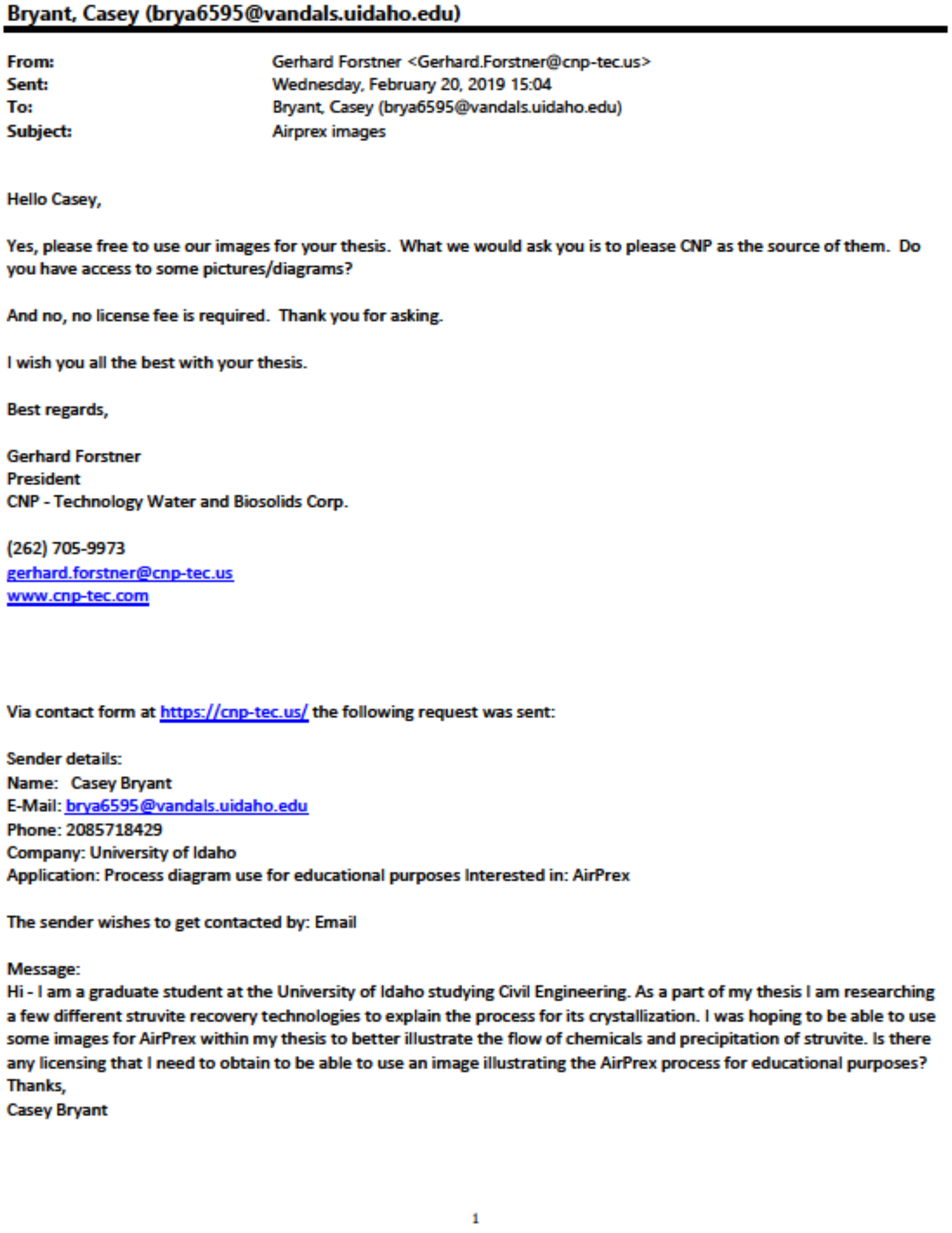

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