THESIS

EVALUATION OF COST EFFECTIVE APPROACHES FOR NUTRIENT REMOVAL IN URBAN STORMWATER AND WASTEWATER: CITY OF FORT COLLINS CASE STUDY

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ABSTRACT

EVALUATION OF COST EFFECTIVE APPROACHES FOR NUTRIENT REMOVAL IN URBAN STORMWATER AND WASTEWATER: CITY OF FORT COLLINS CASE STUDY

To respond to pending regulation that affects effluent nitrogen and phosphorus standards from urban watersheds, this study compares existing nutrient discharges from wastewater and stormwater sources in Fort Collins, Colorado and evaluates the benefits and costs of nutrient removal strategies identified in both sectors as a guide to urban planners. Six alternative wastewater advanced nutrient removal technologies were modeled in BioWin® to be integrated with the existing modified Bardenpho unit. Approximately 1,500 stormwater control measures (SCMs) are implemented in Fort Collins at present; however, not all provide water quality treatment. Two alternative stormwater scenarios were evaluated using the Simple Method and include: 1) retrofitting existing flood control SCMs to provide treatment, and 2) implementing SCMs to treat runoff from currently untreated impervious areas. Treatment level, environmental impacts, and 20-year lifecycle costs were determined for all alternatives and compared within a multi-criterion decision analysis (MCDA). Existing wastewater discharges of nitrogen and phosphorus are 2.0 and 1.5 times larger than those from stormwater, respectively. Removal efficiencies from these discharge nutrient levels were found to be between 7.5% and 30% for wastewater and 20% and 35% for stormwater. Although wastewater alternatives had large ranges of potential costs, all were determined to be more cost effective ($/lb. removal) than the stormwater scenarios. Struvite precipitation in all MCDA scenarios is the most advantageous alternative, followed by ammonia stripping and extended detention basin (EDB) retrofits.

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1.0 INTRODUCTION

Nutrient pollution is a widespread problem and concern in the United States; the Environmental Protection Agency (EPA) has estimated that there are now more than 15,000 waters that do not meet state nutrient standards and more than 7,000 that are impaired due to excess nutrient concentrations (Shapiro, 2013). Nutrients enter natural water bodies in a multitude of ways, including stream bank erosion, runoff from agriculture, stormwater, and discharges of treated municipal wastewater. Of these sources, agriculture is by far the greatest net contributor throughout the United States (Puckett, 1995); however, the contributions from stormwater and wastewater are not trivial and cannot be disregarded in urban environments.

In Colorado, effluent nutrient standards have become more stringent with the approval of Regulation 85. Passed in 2012, this regulation will have a compliance date in 2023 and require all discharge permits for wastewater treatment plants (WWTPs) to contain nutrient standards of 15 mg/L total nitrogen and 1 mg/L total phosphorus in addition to more frequent monitoring requirements (CDPHE, 2012). Recognizing the role of stormwater in urban nutrient discharge, and the high costs associated with wastewater treatment facility processes, Regulation 85 also provides the opportunity of water quality trading between point and nonpoint sources (*Nutrients Management Control Regulation* , 2012). This approach has been implemented in multiple locations in the United States with success in lowering pollutant concentrations and is believed to be a sustainable solution to regulations that allow for the optimization of cost-effective approaches, yield water quality results equal or greater to those required, and achieve multiple environmental and economic benefits.

1.1 Objectives

This study uses Fort Collins, Colorado as a case-study to evaluate and compare cost effectiveness of wastewater and stormwater nutrient removal technologies.

Objectives of the analysis are to:

1. Identify Fort Collins' annual load of nutrients generated both from stormwater discharge and wastewater treatment plants
2. Determine the potential nutrient load reductions from alternative wastewater treatment and stormwater control implementation scenarios
3. Characterize the costs, performance, and efficiency of each treatment technology such that other municipalities may determine the optimal options for them
4. Assess the potential for nutrient trading when more cost effective nutrient abatement measures are available in the stormwater sector

1.2 Background

1.2.1 Wastewater Technologies

The goal of nutrient removal at wastewater treatment plants is to reduce total nitrogen and total phosphorus concentrations to below levels established by state environmental regulatory agencies, selected as the maximum value that still protects aquatic and human health and allows for the water's intended use. Nutrient removal can be accomplished through a number of processes including:

- Modified Bardenpho process
- Centrate and Recycle Activated Sludge Reaeration Basin
- Anaerobic Ammonium Oxidation
- Selective Adsorption
- Electrodialysis
- Ammonia Stripping
- Struvite Precipitation

Processes within WWTPs are described as either primary treatment (removing suspended solids from the waste stream) or secondary treatment, which targets nutrients and organic compounds. Since all influent wastewater flows along this treatment line, it is referred to as the mainstream process. Solids settled out from primary and secondary clarifiers are treated as well, eventually dewatered and removed for land application. The liquid generated, called centrate, is recycled to secondary treatment. Although centrate is a much smaller flow rate than the plant influent, it contains much larger concentrations of nitrogen, phosphorus, and biological oxygen demand (BOD) (Rittmann & McCarty, 2001). By removing nutrients from the centrate stream, treatment pressure in the mainstream process is decreased, potentially allowing lower aeration and/or sludge recycling rates. For these reasons, many nutrient removal technologies focus on treating centrate prior to remixing with the mainstream wastewater. Figure 1 illustrates the location of sidestream treatment within the wastewater treatment process.

Both biological and physicochemical methods are commonly used in nutrient removal. Biological processes utilize microorganisms that metabolize nitrogen, phosphorus, and carbon. Different species of microorganisms are used to exploit different removal pathways, but all require specific environmental factors to grow optimally. Temperature, pH, and dissolved oxygen must be regulated and monitored closely, as most bacteria thrive within a narrow range of conditions and will slow or even reverse nutrient removal if these are not met.

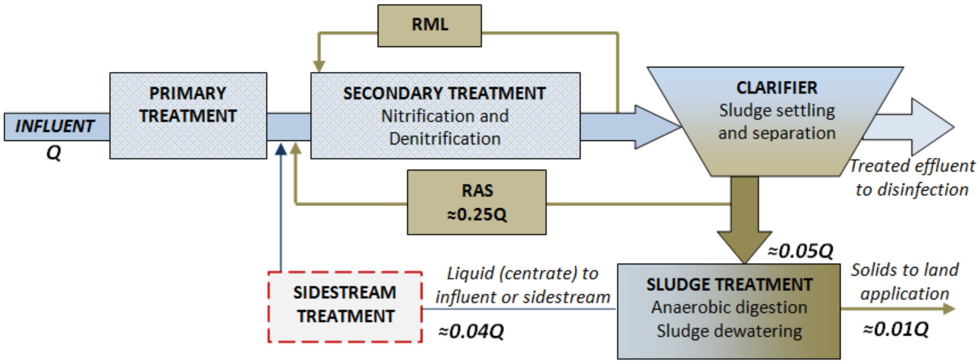

Figure 1: Illustration of processes within a wastewater treatment plant

Phosphorus is more difficult to remove than nitrogen, as this process is also highly dependent on the solids retention time of the biologically active sludge (Rittmann & McCarty, 2001). However, once biological removal is initiated, very little to no chemical addition during secondary treatment is required. Physicochemical processes exploit chemical properties of nitrogen and phosphorus to separate them from the wastewater stream. In many cases, this includes the addition of a chemical to facilitate a chemical reaction, whose product is settles out easily, or a mechanical process to change the properties of the pollutant. The extent of nutrient removal depends on the amount of treatment applied and is relatively easy to control (Sengupta & Pandit, 2011). In addition, some processes make possible the recovery of nitrogen or phosphorus and turning these nutrients into serviceable fertilizer. Table 1 summarizes the technology types included in this analysis and their role within the wastewater treatment process.

Table 1: Summary of wastewater nutrient removal processes

Treatment Technology	Location	Treatment Type	Nutrient Recovery
Modified Bardenpho	Mainstream	Biological	No
CaRRB	Sidestream	Biological	No
ANAMMOX	Sidestream	Biological	No
Selective Adsorption	Sidestream	Physicochemical	Yes (Phosphorus)
Struvite Precipitation	Sidestream	Physicochemical	Yes (Phosphorus)
Electrodialysis	Sidestream	Physicochemical	Yes (Nitrogen)
Ammonia Stripping	Sidestream	Physicochemical	Yes (Nitrogen)

1.2.1.1 Modified Bardenpho

Nitrification and denitrification are the driving biological processes for Bardenpho nitrogen removal. In the first step of nitrification, bacteria oxidize ammonia to nitrite, which is then oxidized to nitrate. This reaction must take place in an aerobic reactor due to its high oxygen demand. Denitrification reduces nitrate and nitrite to nitrogen gas, where it is released from the wastewater stream. Microorganisms use nitrate, produced during nitrification, as the electron oxidizing molecule and require an environment where oxygen is absent (Rittmann & McCarty, 2001). Carbon must also be available from the wastewater influent can be a limiting factor in the reaction. Thus, traditional BNR process trains include at least one aerobic reactor preceded by an anoxic zone so that BOD can be utilized by the denitrifiers. A portion of the sludge generated in the aerobic reactor is recycled back to the influent of the anoxic basin to provide the stream with the nitrates necessary for denitrification. To achieve phosphorus removal, Bardenpho must be modified with an anaerobic reactor prior to the mainstream Bardenpho process, in which electron acceptors including oxygen and nitrate are not present (Figure 2). While initially causing a release of phosphorus from microorganisms, this step primes bacteria for additional phosphorus uptake in the aerobic stage, resulting in a net phosphorus removal (Rittmann & McCarty, 2001).

Despite excellent nitrogen removal, phosphorus removal is not as efficient, since it remains in the system via sludge processing and recycling. Multiple treatment plants have also reported that biological phosphorus removal can hinder solids dewatering efficiency (CH2M HILL, 2014). These, along phosphorus removal's strong dependence on specific process settings, lead to process stability issues; standard effluent phosphorus concentrations may not be consistently met.

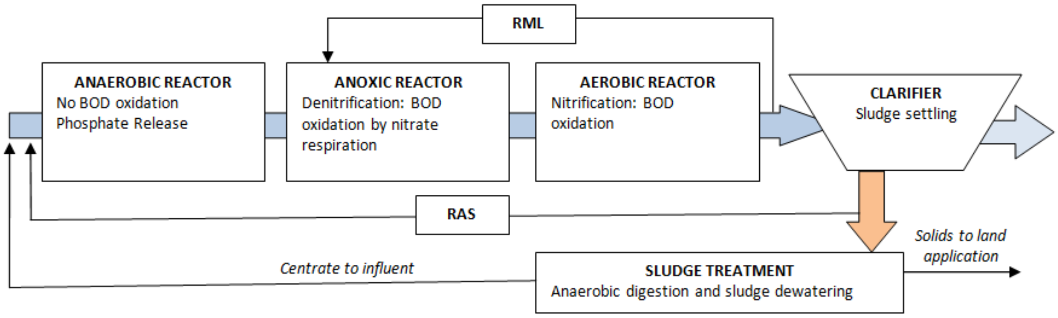

Figure 2: Process Illustration of Modified Bardenpho

1.2.1.2 Centrate and RAS Reaeration Basin (CaRRB)

Centrate and RAS reaeration basin (CaRRB) technology provides additional nitrification of the centrate wastewater stream, allowing for increased denitrification and nitrogen removal (carollo, 2012). It is not a complete treatment, but works jointly with biological nutrient removal in the mainstream process (Figure 3). CaRRB is known by other names as well, such as bioaugumentation regeneration (BAR), bioaugumentation batch enhanced (BABE), and mainstream autotrophic recycle enabling enhanced N-removal (MAUREEN); the common trait among them is the mixing of high ammonia sidestream with RAS (FWR, 2014). Thus, no seeding is required during the startup of CaRRB.

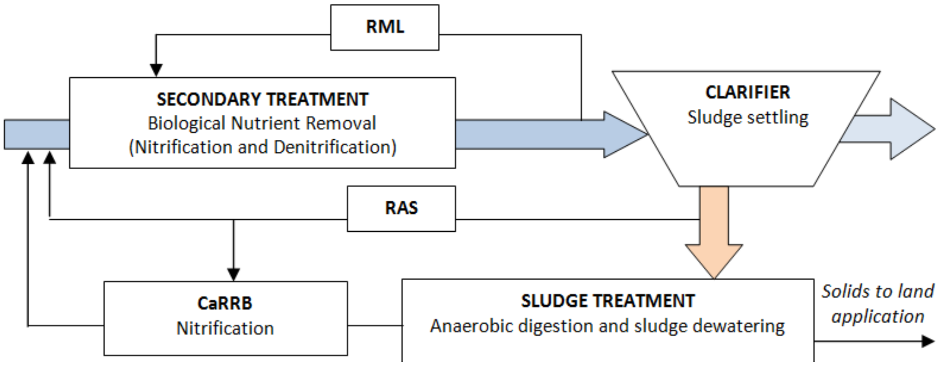

Figure 3: Process Illustration of CaRRB

1.2.1.3 Anaerobic Ammonia Oxidation

Anaerobic Ammonia Oxidation (ANAMMOX) is an alternative biological nutrient removal process which uses microorganisms capable of reducing ammonia in a reaction with nitrite to produce nitrate gas. Although capable of removing up to 90% of nitrogen from the wastewater, it does not remove much phosphorus.

Nitritation is required before the ANAMMOX reactor, which is aerated to maintain an oxygen concentration between 0.5 and 1.5 mg/L (Musabyimana, 2008). The ANAMMOX process requires low energy input and produces less sludge than traditional BNR. Studies have estimated that BNR oxygen requirements can be reduced by as much as 60% (Dapena-Mora et al., 2004). Used as a sidestream process, ANAMMOX allows for reduced nitrification and denitrification in the main treatment train and reduced sludge recycle rates. ANAMMOX is a comparatively more complex operation, and is very sensitive to influent water quality. However, it is readily automated (FWR, 2014) and companies such as Veolia supply pre-made systems and media with established bacteria to expedite startup. Figure 4 illustrates the placement of ANAMMOX process within the wastewater treatment process.

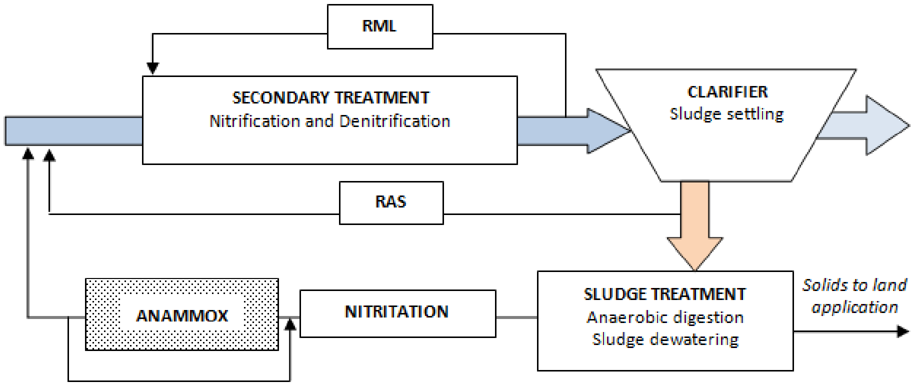

Figure 4: Process Illustration of ANAMOX

1.2.1.4 Selective Adsorption

Selective adsorption (Figure 5) is a chemical process that can be used to remove either phosphorus or nitrogen. Wastewater treatment facilities most commonly use it to remove phosphorus, which is harder to remove biologically. Adsorbents capable of removing phosphorus include lime, aluminum, or various ferric compounds such as polymer anion exchangers with iron oxide (Sengupta &

Pandit, 2011), hydrated ferric oxide, and granular ferric hydroxide. A reactor basin must be installed to mix the adsorbent with the influent stream. Treatment efficiency and fertilizer revenue will depend on the adsorbent type.

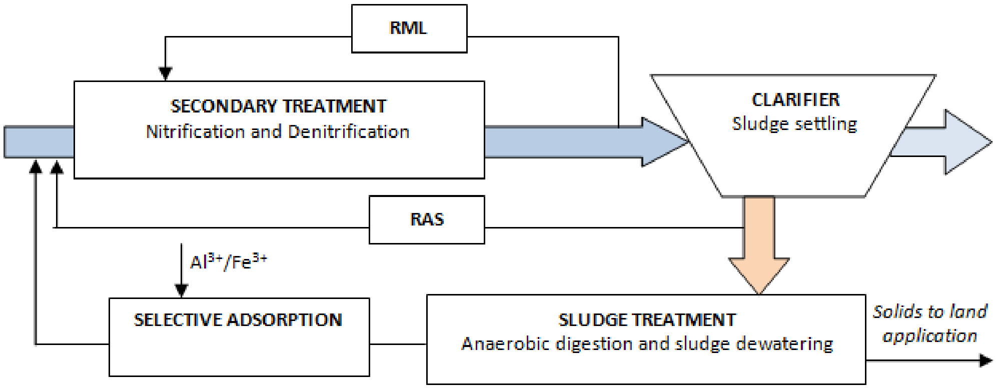

Figure 5: Process Illustration of Selective Adsorption

1.2.1.5 Struvite Precipitation

The struvite precipitation process (Figure 6) requires a reactor to mix magnesium oxide or magnesium chloride with the influent waste stream; some marketed tanks provide for the settling of struvite to occur within the same reactor, but other process set-ups will require a clarifier following the mix tank to settle and collect the struvite (Forrest, Fattah, Mavinic, & Koch, 2008; Wang, Burken, Asce, Zhang, & Surampalli, 2005). The influent stream must be dosed with magnesium oxide (MgO), and the size and quality of the struvite pellets formed are affected by the Mg:P ratio (Forrest et al., 2008).

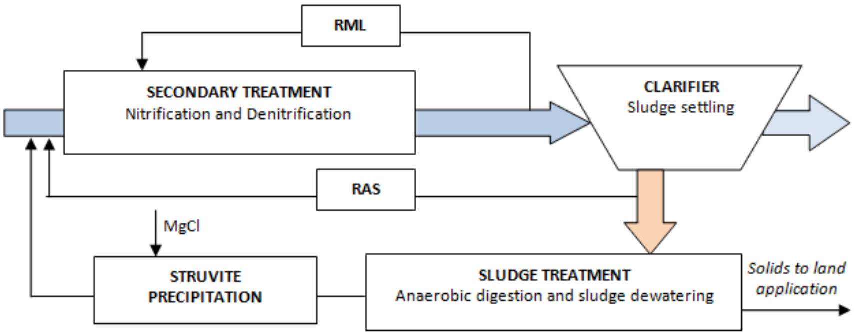

Figure 6: Process Illustration of Struvite Precipitation

1.2.1.6 Electrodialysis

Electrodialysis removes ammonia from centrate through the application of an electric potential across selectively permeable membranes (Valero, Barceló, & Arbós, 2011). The membrane configuration

(University of Stuttgart, 2009), shown in Figure 7, separates the anion and cation streams and determines the amount of pollutant to be removed. The current diverts targeted ions, established by the membrane media, from one flow stream into the other.

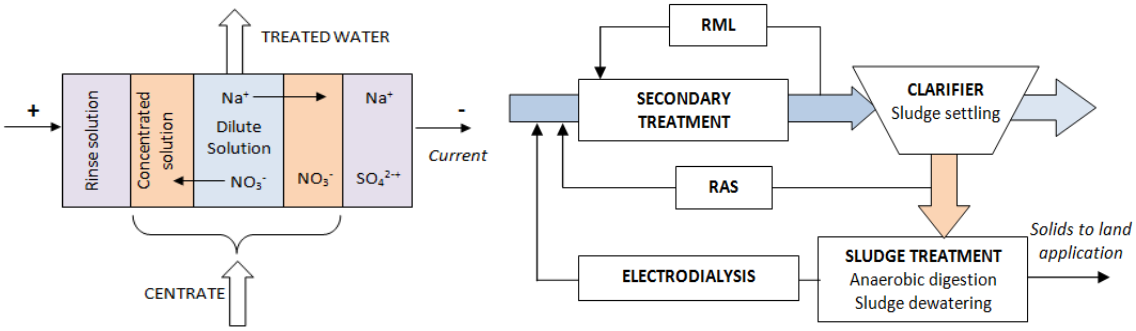

Figure 7: Process Illustration for Electrodialysis

1.2.1.7 Ammonia Stripping

Ammonia stripping utilizes the ammonia liquid-gas equilibrium to remove nitrogen from wastewater. As air is supplied to the wastewater stream, the aqueous-phase ammonia concentration decreases while that of the gaseous-phase increases. Usually, an upflow reactor is used, in which air is pumped through the top of the tank to be mixed with the wastewater. To achieve nitrogen removal, ammonia must be very concentrated within the waste stream; therefore, this treatment is only efficient as a sidestream reactor (Tetra Tech, 2013). Because ammonia stripping removes nitrogen from the wastewater, additional treatment is unnecessary in the mainstream treatment process unless phosphorus removal is desired. Gas-phase ammonia can be precipitated from the air stream with sulfate, generating a potential source of fertilizer (Musabyimana, 2008; Tetra Tech, 2013). Figure 8 illustrates an ammonia stripping tower. Figure 8 illustrates both the basic function of an ammonia stripping tank and the treatment's role within the wastewater treatment plant.

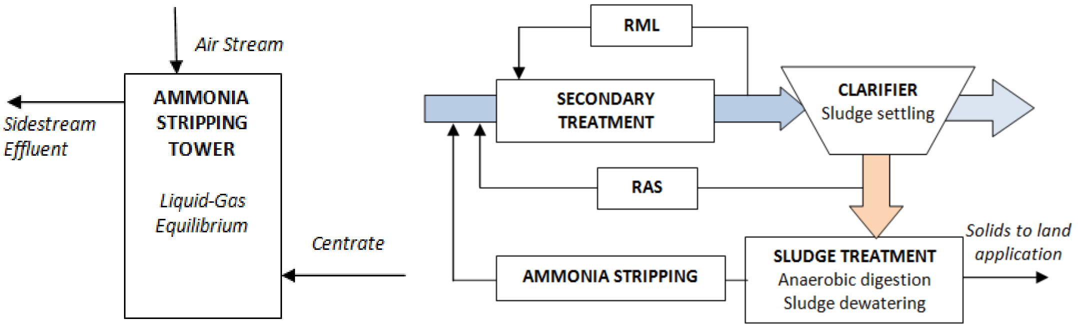

Figure 8: Process Illustration of Ammonia Stripping

1.2.2 Stormwater Best Management Practices (BMP)

Stormwater best management practices (BMPs) are constructed facilities designed to reduce detrimental impacts of urban stormwater on lakes, rivers and streams. Prior to the 1990s, stormwater BMPs in Fort Collins were designed primarily to prevent downstream flooding (Olson, 2014). Starting in the 1990's, the detention basin design was modified to provide pollutant removal in addition to flood control. Both approaches capture stormwater runoff and slow their discharge to downstream receiving waters to reduce erosion, flooding, and infrastructure damage; however, while flood control structures slow the discharge of their captured stormwater, they do not contain storm volumes an adequate time for settling or infiltration. Stormwater controls that provide water treatment detain the water quality capture volume (WQCV), determined by the amount of imperviousness within the contributing area. Although Fort Collins now requires that all new developments treat the WQCV (*Ordinacne No. 152*, 2012), many areas within the city do not have any BMPs that provide water quality control.

1.2.2.1 Bioretention (Porous Landscape Detention)

Bioretention BMPs (Figure 9) capture and filter stormwater runoff through an engineered filter media designed to remove pollutants and support a variety of vegetation. Pollutants are removed from the runoff water column through adsorption to the filter media and, to a lesser extent, absorption by vegetation roots. In addition, bioretention can promote infiltration of stormwater runoff into groundwater. They can be retrofitted into the landscaped areas of most existing developments; including parking lot islands and street medians/parkways. Figure 9 shows two examples of bioretention implemented to reduce and treat urban runoff.

Figure 9: Examples of Bioretention Basins along a) a parking lot median and b) a roadside parkway

1.2.2.2 Extended Detention Basins

Extended detention basins (EDBs) are a modified version of traditional flood control detention basins that capture stormwater runoff and slowly release it over a period of 24-48 hours. During the time that runoff is stored in the EDB, particulate pollutants are removed from the runoff water column via settling. Existing flood control detention basins can be retrofitted into EDBs to improve pollutant removal through the replacement of the outlet structure. Figure 10 provides an example extended detention basin and flood control basin, and highlights the difference between their outlet structures.

Figure 10: Examples of a) extended detention basin and b) flood control basin

2.0 METHODOLOGY

2.1 Wastewater Methods

To estimate the nutrient removal resulting from sidestream treatment units, sidestream treatment flow rates and effluent water quality were integrated with mainstream nutrient removal and sidestream sludge treatment process models.

2.1.1 BioWin Set-Up and Calibration

BioWin, a wastewater treatment simulator that incorporates both physicochemical and biological processes developed and supplied by EnviroSim Associates Ltd., was used to model the existing and alternative nutrient removal technologies at DWRF. The BioWin model of DWRF was created and calibrated to existing water quality by Link Mueller (Mueller, 2015). Process flow diagrams exported from BioWin are included in Appendix A. Throughout the year, the influent flow rate varies; an average of 11.4 MGD was used in this analysis. Table 2 lists additional volumes and flow rates obtained from the treatment plant and used for calibration.

Table 2: Drake Water Reclamation Facility Plant Flow Rates and Volumes

DWRF Volumes and Flow Rates		
DWRF Average Influent	11.4	MGD
Anaerobic Zone	0.43	Million gallons
Anoxic Zone	0.45	Million gallons
Aerobic Zone (2)	0.85	Million gallons/basin
Final Clarifiers	2,420,000	Square feet
Anaerobic Digesters	2.56	Million gallons
Mixed Liquor Return	2Q	MGD
Return Activated Sludge	4.5	MGD
Waste Activated Sludge	0.12	MGD

The solids retention time calculated for the baseline model, and used for the alternative analyses, was 12.48 days. The model includes the primary clarifier, mainstream nutrient removal system (Modified Bardenpho), and solids treatment. Several process simplifications to the current DWRF treatment system layout (Figure 11) in the BioWin representation (Figure 12). This was the case for the primary clarifiers and the dual secondary treatment system.

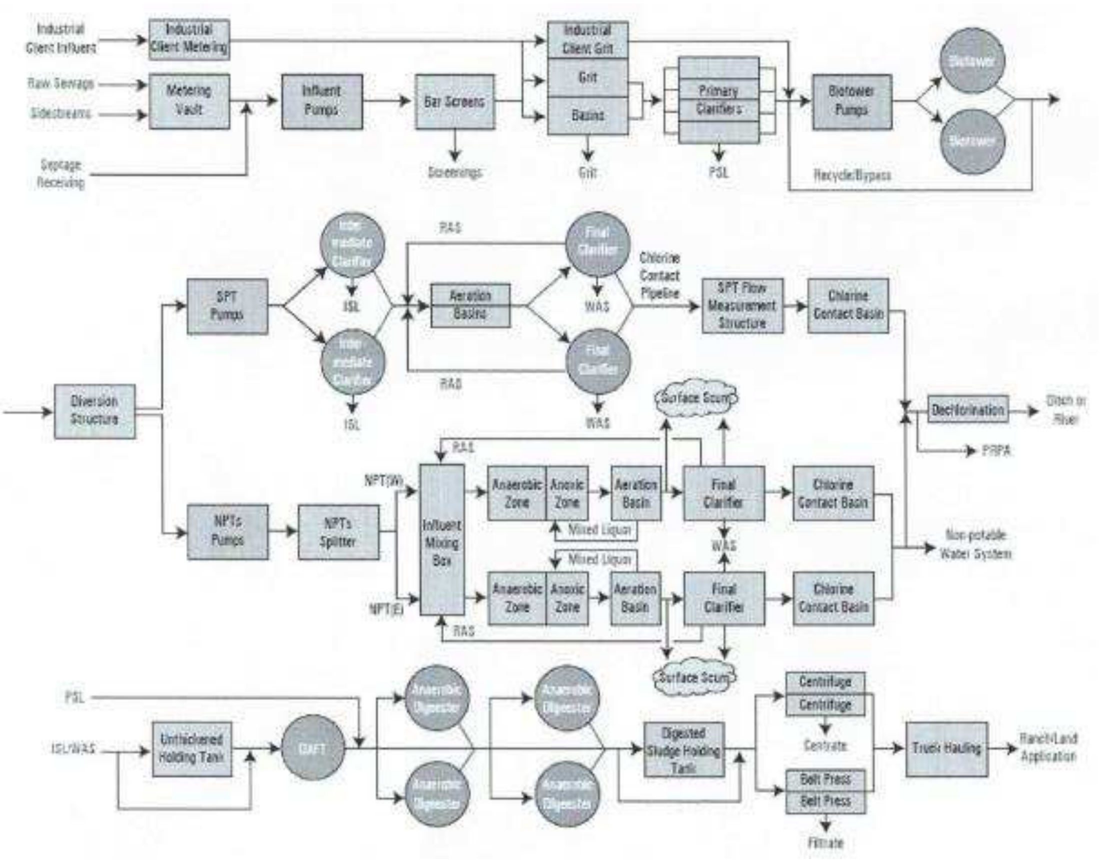

Figure 11: DWRF Treatment process, figure produced by MWH for DWRF in 2008

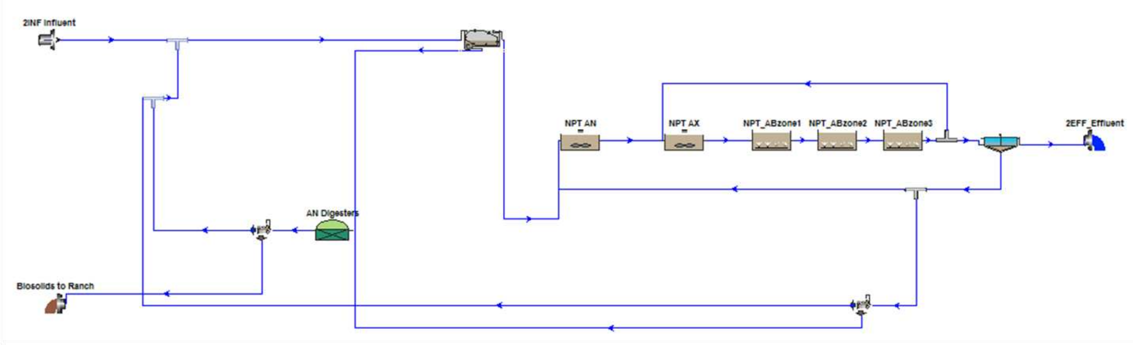

Figure 12: Baseline BioWin model, generated by BioWin (as screenshot)

Four processes were individually calibrated prior to the system calibration: the primary clarifier, the secondary system (including the anaerobic, anoxic, and aerobic reactors), the dissolved air flotation tank (DAFT), and the centrate dewatering unit. All element properties were determined using data available from the from the DWRF Supervisory Control and Data Acquisition (SCADA) system for the month of September 2014 Number of samples for effluent values varied considerably for different constituents, ranging from 2 for TKN, and 66 for TSS.

During the system calibration, biological stoichiometric and wastewater fractions assumed in the program were assessed with respect to literature values and SCADA-calculated values. BioWin default parameters that affect the growth, decay, and metabolic rates of microorganisms are listed in Table 3; theses were parameters were kept at their default values. Table 4 lists the four wastewater fractions – of COD, TN, and TP - that were altered to best represent the processes observed at DWRF.

Table 3: BioWin Assumptions for Biological and Process Parameters

Microorganism Type	**Maximum Specific Growth Rate, μ_{max}**	**Substrate Half-Saturation Constant, *K***	**Aerobic Decay Rate, *b***	**Anaerobic Decay Rate, b_{an}**	**Anoxic Decay Rate, b_x**	**Yield, *Y***
Ammonium Oxidizing Bacteria (AOB)	0.9 d^{-1}	0.7 $mgNH_4/L$	0.17 d^{-1}	0.08 d^{-1}	0.08 d^{-1}	0.15 mgCOD/mgN
Nitrite Oxidizing Bacteria (NOB)	0.7 d^{-2}	0.1 $mgNO_2/L$	0.17 d^{-1}	0.08 d^{-1}	0.08 d^{-1}	0.09 mgCOD/mgN
Anaerobic Ammonia Oxidizers (ANAMMOX)	0.1 d^{-3}	2.0 $mgNH_4/L$ 1 $mgNO_2/L$	0.019 d^{-1}	0.0095 d^{-1}	0.0095 d^{-1}	0.114 mgCOD/mgN
Ordinary Heterotrophic Organisms (OHO)	3.2 d^{-4}	5 mgCOD/L	0.62 d^{-1}	0.233 d^{-1}	0.131 d^{-1}	0.66 mgCOD/mgCOD

Table 4: Altered Kinematic Parameters for BioWin Calibration

Kinetic Parameters	**Value Applied**	**Default Value**
Fraction of readily biodegradable COD (gCOD/g total COD), Fbs	0.163	0.160
Fraction of unbiodegradable soluble COD (gCOD/g total COD), Fus	0.057	0.050
Fraction of particulate organic nitrogen (gN/g organic N), Fnox	0.052	0.050
Fraction of phosphate to total phosphorus (gPO_4-P/gTP), Fpo4	0.556	0.500

Impacts of various technologies on nutrient removal were evaluated while keeping the RML flow rate constant at 2Q. RAS was kept at 1 percent of the influent flow rate (0.01Q, equal to 0.12MGD) and approximately equal to the rate observed at DWRF. The mean cell residence time of all scenarios was modeled by BioWin to be 12.48 days. BioWin models were run in steady-state. Figure 13 compares the plant effluent (prior chlorination) modeled by BioWin and observed at DWRF. Number of samples for effluent values varied considerably for different constituents, ranging from 2 for TKN, and 66 for TSS.

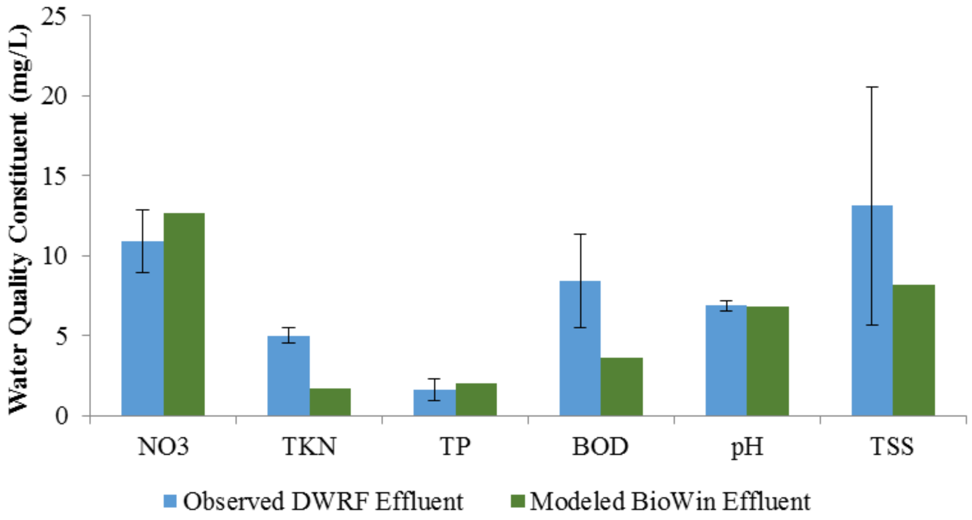

Figure 13: Comparison of modeled vs. observed effluent water quality, error bars represent the standard deviations of observed values

The modeled effluent BOD value is lesser than the calculated average. The final BioWin model did not predict TKN well, and yields an effluent values approximately half that of the value seen at DWRF during September 2014. Because the observed TKN for this time period consisted of only 2 samples, additional data was requested from DWRF to verify the average TKN effluent concentration. Between June 18, 2014 and August 6 2014, a period during which BOD concentrations were consistently less than 15 mg/L and no process changes took place, the average of 6 TKN observations is 1.83 mg/L, with a standard deviation of 0.46 mg/L. These values correlate much more closely with what is predicted by the BioWin model. Because further efforts to resolve this issue resulted in nutrient concentrations more poorly estimated observed values, the inaccurate BOD predictions were accepted in the final model.

2.1.2 BioWin Modeling of Nutrient Removal Alternatives

BioWin specializes in modeling biological treatment processes and does not have model elements specifically suitable for ammonia stripping, electrodialysis, selective adsorption, or struvite precipitation. Therefore, the impacts of these physicochemical processes and ANNAMOX were determined by inputting into BioWin an additional influent that represented the flow and constituent concentrations of the treated centrate (Figure 14). Thus, the effect of each sidestream unit on plant effluent nutrient concentrations could be determined even when specific model elements were not available for a particular treatment process.

The recycle pipeline from the dewatered sludge treatment to the plant influent was removed so that the total influent was not increased (Figure 14). This enabled accounting for nutrient removal in sidestream treatment processes. Actual BioWin schematics are included in Appendix A.

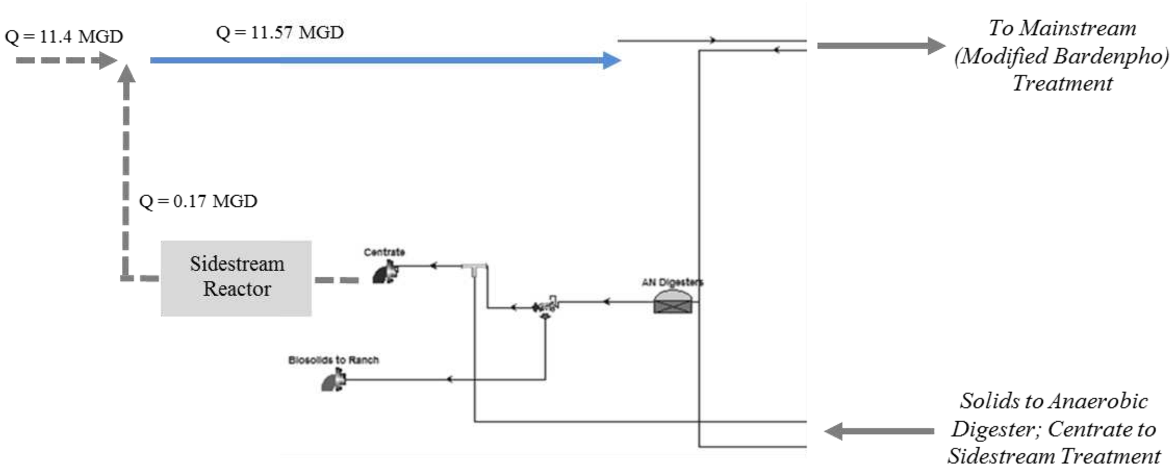

Figure 14: Representative BioWin schematic for physicochemical processes

Biowin values for the centrate water quality – including TKN, nitrate, total phosphorus, dissolved oxygen, COD, magnesium, calcium, alkalinity, and pH – were attained from the baseline model, possible because BioWin calculates and can generate reports for individual elements or connections (representing pipes) in the model. Centrate is generated by dewatering units both before and after the anaerobic digester treatment and is recycled back to the secondary treatment influent; it was the combined centrate in the fully-connected baseline model from which the parameters listed above were obtained. In the physicochemical and ANAMMOX alternatives, these water quality input parameters were input into a new influent element, to be mixed with the main plant influent. Figure 14 shows these elements and their connectivity in the alternatives' models. When physicochemical sidestream processes were modelled, the centrate was disconnected from the influent and allowed to discharge as an "effluent". Although not shown as BioWin elements in the figure, there are two influents – one for the main stream and one for the centrate stream – which combine prior to primary clarification.

It was imperative that the centrate quality of the simulated treated centrate effluent (added as an additional influent to the model) represent the centrate quality calculated by the model after solids treatment. Initially, the fractions of nutrient values were not matching the centrate water quality as predicted by BioWin (Table 5). Since the centrate influent values were known to be the correct steady-state concentrations, only the wastewater fractions of the centrate influent could be modified. Three of these fractions were changed for the centrate influent (i.e. input stream of treated centrate) only, and kept constant for all physicochemical and ANAMMOX scenarios (Table 5).

Table 5: Altered wastewater fractions for treated centrate combined with main influent

WW Fraction	Default	Model
Fbs (gCOD/ gTotal COD)	0.27	0.170
Fnox (gN/ gOrganic N)	0.25	0.050
Fpo4 (gPO4-P/ gTP	0.75	0.995

It should be noted that the default values describe a general plant main influent and are not representative of centrate, which generally contains higher concentrations of its constituents and has already undergone some treatment from the solids treatment process. Figure 15 displays the baseline centrate water quality values and the centrate influent values for the current mainstream RML rate. This process was repeated for additional RML rates, without further variation of the wastewater fractions; for all RML rates, the baseline centrate water quality correlated well with the disconnected centrate represented by BioWin. Underflow from the secondary clarifiers was kept constant and specified as volumetric flow rates, as opposed to percentages, to ensure consistency of centrate production and effluent discharge

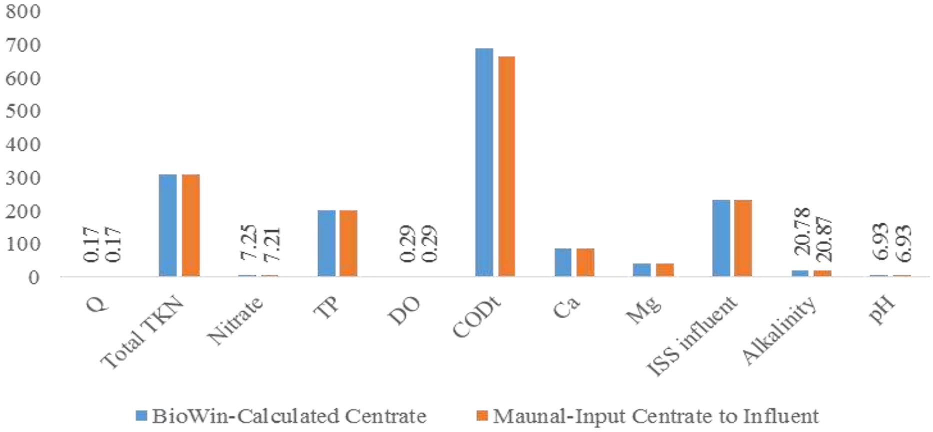

Figure 15: Centrate post-solids treatment vs. manual-input to influent to compare BioWin and User Input

. Spreadsheet calculations determined the necessary process changes to achieve the selected level of nutrient removal. Section 4 discusses these methods in more detail. Table 6 lists the process variables for the studied technologies that have the most impact on nutrient removal.

Table 6: Process Variables for Wastewater Nutrient Removal Technologies

time	Variable	Scenarios (Variable Value)				
		1	2	3	4	5
CaRRB	RAS input into sidestream reactor (MGD)	0.05	0.10	0.13	0.20	0.30
ANAMMOX	Sidestrream reactor volume (MG)	0.7	0.9	1.5	2.0	2.5
Ammonia Stripping	Stripping Tower Height (ft.)	12.5	15	27.5	42.5	50
Electrodialysis	Power (kW)	118	169	236	287	303
Selective Adsorption	Addition rate of aluminum (tons/year)	66	95	131	161	168
Struvite Precipitation	Addition rate of magnesium oxide (tons/year)	150	212	299	361	383

For each of the seven alternatives analyzed, five simulations were run to determine the range of treatment efficiency by varying these parameters. CaRRB was varied through parameters that could be changed directly in BioWin. The remaining alternatives' parameters were varied within spreadsheet models. Table 4 shows how these changed for each set of simulations. Methodology on how these were determined is discussed in Section 4 of this report.

2.1.3 RML Rates in Alternatives Analysis

The initial process alternative considered for wastewater treatment was the variation of recycle mixed liquor (RML) rate using only the Modified Bardenpho treatment. Nitrogen effluent concentrations decrease and phosphorus concentrations increase as the recycle mixed liquor rate is increased (Figure 16), due to differences in their biological removal mechanisms. Phosphorus concentrations increase as RML increases; its removal is achieved by introducing stress to aerobic heterotrophic organisms triggering a dormant state in which they release phosphorus in the anaerobic zone. Later, in the aerobic reactor, these same organisms take advantage of the increased oxygen (electron acceptor) and while metabolizing organic material, uptake more phosphorus than was initially released (Rittmann & McCarty, 2001). With the increased addition of nitrate to anoxic zone through the RML, there is less opportunity for organisms to uptake phosphorus. This results in higher effluent phosphorus concentrations.

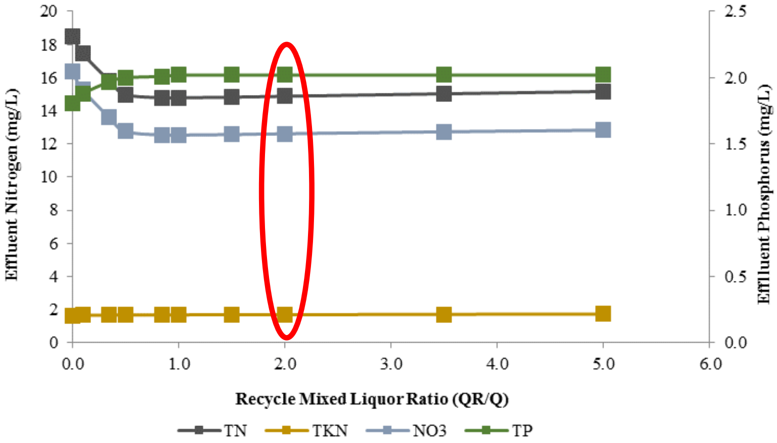

Figure 16: Modified Bardenpho effluent nutrient response to RML; current rate indicated by red circle

The expected relationship is that of increased nitrification (lower effluent nitrogen concentrations) as RML is increased; however, concentrations of TN, TKN, and nitrate are shown in Figure 16 to increase slightly at RML flows greater than 2Q. To test whether the limited nitrification is a result of a carbon limitation, the model was evaluated using varying concentrations of influent COD and mainstream RML (Figure 17).

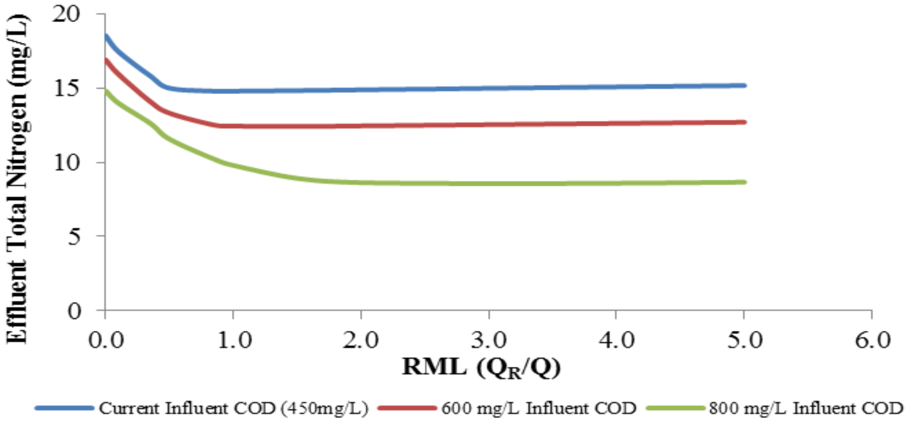

Figure 17: Effect of influent COD in Effluent Total Nitrogen (mg/L)

While increasing the influent carbon improved treatment efficiency, in each case, as the RML is increased beyond 1.0Q the effects on total nitrogen are almost indescribable; the concentrations vary by less than 1.0mg/L. Hence, the modified Bardenpho process at DRWF is carbon limited, at the current influent COD concentration. RML has more of an impact when influent COD is increased to 800 mg/L.

2.2 Stormwater Methods

Three stormwater nutrient-loading scenarios were evaluated: the existing condition, the retrofitting of existing storm control measures (SCMs), and the implementation of new measures. Precipitation records obtained from the National Climate Data Center (NCDC) were used to calculate Fort Collins-specific runoff depths for the pollutant analysis (NOAA, 2014).

2.2.1 Simple Method

In each scenario, the Simple Method (Chandler, 1994) was used in a spreadsheet model to calculate the annual load of nutrients discharged through stormwater. This method calculates stormwater pollutant loads from small (less than a square mile) urban areas. Equation 1 gives the Simple Method equation for calculating pollutant load from urban stormwater:

$$L = P * A * R_V * P_r * C * 0.226 \qquad \text{Equation 1}$$

L is the estimated pollutant load in pounds

P is the average annual precipitation depth

A is the subcatchment area in acres

R_v is the runoff volume coefficient

P_r percent producing runoff

C is the effluent event mean concentration

Fort Collins' average annual precipitation depth is 11.94 inches, calculated using long-term data from the NCDC (NOAA, 2014). In each storm, the initial rainfall infiltrates or is stored in surface depression and thus does not contribute to pollutant discharge. This is accounted for in the simple method by multiplying the annual rainfall by the percentage that produces runoff (P_r). Fort Collins' runoff-producing factor is 0.925, calculated by identifying individual storm events within the NRCS data and assuming a minimum runoff-producing precipitation depth of 0.1 inches.

The runoff coefficient (R_V) accounts for a site's soil type, slope, and land cover on the amount of runoff produced per storm and is calculated by (Chandler, 1994):

$$R_V = 0.05+0.009(I) \qquad \text{Equation 2}$$

The equation used to calculate the runoff coefficient is dependent on the site's percent impervious area (I), defined as the zones in which rainfall cannot be infiltrated by the land surface. Impervious areas include roads, rooftops, and sidewalks. The total site area (A) can be obtained from site plans or, as in this analysis, from aerial maps input into GIS. Land-use designations from the Fort Collins website (fcgov.com) were used to identify smaller subcatchments with uniform imperviousness and land cover. The influent pollutant concentration (C) is estimated in this analysis as the event mean concentration, or the average concentration of pollutant discharged by a runoff-producing storm. Individual storm pollutant concentrations vary a great deal; however, median concentrations are appropriate for long-term analysis. Colorado-specific data for total phosphorus and total nitrogen runoff concentrations were obtained by the Urban Drainage and Flood Control District (UDFCD) and are shown in Table 7 (Clary et al, 2014).

Table 7: UDFCD Measured Event Mean Concentrations

Constituent	TP mg/L	TN mg/L
Open Space	0.41	3.4
Commercial	0.36	3.45
Residential	0.56	5.06
Industrial	0.35	3.56
Highway	0.39	3.78

A GIS model was created with land-use designations from the Fort Collins website (fcgov.com) and soil group data from the National Resources Conservation Service (NRCS) soil database (SSRUGO) (NRCS, 2013). To prepare the data for the cost-analysis tool, BMP-REALCOST (Olson et al, 2013), a GIS tutorial called "Step-by Step Instructions for Using ArcGIS to Delineate Subcatchments and Extract Hydrologic Parameter Values for the UD-BMPCosts Tool" (Olson et al, 2013) was followed, with the exception of slope-analysis instructions. A database of Fort Collins' existing SCMs, including outlet locations and drainage areas, was obtained from the city. These values were sorted to identify the total acres of area treated by each SCM type in each land use category.

It was assumed that 85% of annual runoff from each drainage area is treated by its SCM and that SCMs were sized to treat the appropriate water quality capture volume (WQCV). The following SCM effluent nitrogen and phosphorus concentrations were obtained from data in the International BMP Database (Wright Water, 2012), shown in Table 8. Although influent nitrogen concentrations may vary, it is assumed that for each SCM the effluent concentrations will remain constant regardless of land use and drainage size. The median values reported by the International BMP Database were used in this analysis. The remaining 15% of annual runoff was assumed to bypass the SCM and have nutrient concentrations equal to those reported by the UDFCD. Finally, the total annual nitrogen and phosphorus loads were calculated by doing a flow-weighted average of the effluent concentrations.

Table 8: International BMP Database BMP Mean Effluent Concentrations

BMP Abbr.	**BMP Description**	**TP_{Eff} *mg/L***	**TN_{Eff} *mg/L***
EDBD	Extended Detention Basin	0.220	2.370
EDBD/FC	Extended Detention Basin - Flood Control	0.220	2.370
EDBW	Extended Detention Wetland Basin	0.130	1.280
FCBD	Flood Control Basin	--	--
FCBW	Flood Control Wetland Basin	--	--
PC	Permeable Pavement	0.090	1.490
PLD	Bioretention	0.090	0.900
SF	Sand Filter	0.090	0.850
STC	Storm Interceptor	0.120	2.220
WL	Wetland	0.080	1.190
NONE	No basin	--	--

2.2.2 Scenario 0: Existing SCMs

Approximately 800 SCMs are reported to be in the Fort Collins city limits. Table 9 lists the types of BMPs and the number of acres treated by each. The existing nutrient discharge was estimated using the Simple Method, as described previously. No runoff reduction was included, as it was assumed that all runoff treated by SCMs is discharged through outlet structures to stormwater the conveyance system. Of the BMPs listed, only porous landscape design (also called bioretention basins) and wetlands provide runoff reduction, if stormwater is not captured by underdrain systems. The areas treated by these BMPs comprise 2.6% of the total area in Fort Collins. Since not all runoff generated from these areas would be eliminated, the current runoff reduction is considered negligible.

Table 9: Fort Collins BMP types and areas treated

BMP Type	**No. BMPs**	**Area Treated**	
No treatment	N/A	21,518	acres
Extended Detention Basins	211	4,761	acres
Flood Control Basins/ Flood Control Wetland Basins	518	17,353	acres
Permeable Concrete	3	3	acres
Porous Landscape Design (Bioretention Basins)	7	7	acres
Sand Filters	6	11	acres
Stormceptors	16	43	acres
Wetlands	25	1,176	acres
Total	786	44,872	acres

2.2.3 Scenario 1: Retrofitted Extended Detention and Flood Control Wetland Basins

Of the ten SCM types included in the Fort Collins SCM database, two only provide flood control and do not remove any substantial level of stormwater pollutant. These are flood control detention basins and flood control wetland basins. In the first scenario, these basins are retrofitted to capture and treat the stormwater capture volume. Such changes would involve replacing the outlet structures to include an orifice plate so that pollutant settling may occur before discharge (CH2M HILL & City of Colorado Springs City Engineering Division, 2002). With these upgrades, the flood control detention basins and flood control wetland basins become extended detention basins and extended wetland detention basins, respectively. In the spreadsheet model, the treatment types of these SCMs were revised and the associated treatment calculated.

2.2.3 Scenario 2: Implementing Bioretention Basins

The SCM database also included areas that are not treated by any existing SCMs. These areas were included in the spreadsheet model and delineated into land-types as well. Scenario 2 adds bioretention basins in these areas, so that runoff generated from all previously untreated acres are treated In the spreadsheet model, the treatment type of land areas indicated as not draining to SCMs was modified and the treated runoff volume was altered to 59.5% of total annual runoff, expressed in Equation 2 by multiplying 0.595 to Rv. This is equivalent to 30% of SCM influent infiltrating through the media to groundwater.

2.3 Evaluation Methods

Not all decision factors considered by utilities are economic. There are social and environmental impacts caused by any project managing a common resource such as water. Therefore, an MCDA analysis was used to provide a preliminary look at these considerations and benefits. Assumptions were made through engineering-based judgment to assign weights to the criteria and identify appropriate sub-criteria. These were grounded upon both common and unique issues brought up in the literature analysis of the alternatives. The use of four different weighting sets was deemed to provide a broad perspective to the impact of the various factors. While the data used to evaluate each alternative is based on literature review, the purpose of the MDCA is a preliminary exercise meant to show broad trends and potential routes of future research. Multi-criterion decision analysis (MCDA) allows for the quantitative comparison of alternatives, evaluated through many different criteria, to identify an optimal solution (Zopounidis & Paradalos, 2010). This analysis method is ideal for the comparison of relatively dissimilar alternatives, as it permits flexibility in both evaluation measures and descriptors. Criteria, upon which the decision will be made, were chosen by identifying common utility concerns and factors indicating the project effectiveness. These are listed in Table 10**.**

Table 10: Criteria for Evaluation of Wastewater and Stormwater Alternatives

Criteria	**Sub-criteria**
Treatment Efficiency	Cost per pound TN removal Cost per pound TP removal Total TN removal Total TP removal Treatment variation
Costs	Capital cost Rehabilitation cost Maintenance costs Potential revenue from fertilizer
Nutrient Recycling	Local recycled fertilizer demand Volume of sludge reduction
Operational Complexity	Monitoring requirements Maintenance requirements Start-up period Waste to landfill
Impact Within Treatment Setting	Odors and nuisances Aesthetic appeal Additional pollutant removal
Public Perception	Perception of Progressive Practices

Economic, social, and environmental impacts from each criterion were considered through sub-criteria. The basic data for each alternative is included in Appendix D. Much of the value of MCDA in decision making comes from the ability to assign weights, or relative importance factors, to each of the criteria. Three sets of stakeholder interests were considered while choosing importance factors, by engineering judgment, for this analysis: economic-driven, operational complexity focused, and removal efficiency guided. A fourth weighting set represents the stakeholder position of all criteria being of equal importance. Table 11 lists the relative importance factors used in the MCDA analysis. The assignment of a “4” to a particular category indicates that this criterion is four times as important as those given a “1”, and so on for other values. Equal weighting of subcriteria was maintained through all assessments.

Table 11: Relative Importance Factors of Evaluation Criteria

Criterion	**Weighting Set 1**	**Weighting Set 2**	**Weighting Set 3**	**Weighting Set 4**
Treatment Efficiency	2	1	3	1
Installation	3	1	1	1
Nutrient Recycling	1	1	1	1
Operation	1	3	1	1
Impact within Treatment Setting	1	2	2	1
Public Perception	1	1	1	1

The two types of MCDA used in this analysis are the weighted average method (WAM) and the preference ranking organization method for enrichment of evaluations (PROMETHEE); they differ in the valuing process to compare alternatives. WAM is a value-based method which provides a score to each criterion for each alternative based upon their relative strengths (Zopounidis & Paradalos, 2010). For each criterion, the alternatives will be scored with a decimal number corresponding to *how much* better or worse it is comparted to the other alternatives. The PROMETHEE method, in contrast, is an out-ranking method, and compares each alternative pair-wise with every other alternative (Smet & Lidouh, 2013). Either a one or a zero is assigned based on whether it is better or worse. In many cases, the results from these two methods will only slightly vary. However, if large differences exist or if word scales are used, the results may be distinctly different. The eight sets of MCDA results were analyzed together to determine the alternatives favorable to the greatest number of stakeholders.

3.0 COST METHODOLOGY

Planning-level cost estimates were calculated for a 20-year planning horizon for both wastewater technologies and stormwater controls and include capital and maintenance costs. For each of the technologies considered, initial and ongoing costs included construction costs, maintenance costs, and rehabilitation costs. Potential revenue from nutrient recovery was also considered for wastewater sidestream processes when applicable. The considered costs were combined in each case to obtain a total net present cost (NPC)[1] for the 20-year project using Equation 3 and assuming an average inflation rate (r) of 4.6% (Olson et al., 2013).

$$NPC = K + \sum_{y=1}^{PH} (M - A)(1 + r)^y + \sum_{y_r} R_y (1 + r)^y \qquad \text{Equation 3}$$

Where capital (one-time) costs are represented by K, and annual operative expenses are equal to the difference of maintenance costs (M) and potential monetary gains from by-product use (A) for each year (y). Rehabilitation costs (R_y) are adjusted for inflation for the year(s) in which they will be incurred (y_r), a subset of the total years in the planning horizon. Recurring costs are adjusted for inflation and summed over the duration of the planning horizon (PH). This method of calculating costs allows for consistent comparisons between the alternatives, while ignoring potential benefits from implementing the technology (Griffin, 2005). It assumes that the interest rate over the planning horizon is zero, as is customary in the calculation of costs incurred by private companies. This approach is appropriate for this analysis, as both wastewater and stormwater sectors are paid for by tax-generated funds. Since the City government is not a for-profit entity, it cannot tax excessively or hold excess money to save in a bank, and thus cannot practice long-term investment as a private company would do, expecting federal bonds. Each of the cost components considered (capital, maintenance, revenue, and rehabilitation) are calculated

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[1] In most economic analyses, the net present value (NPV) is reported and is calculated as the difference between the net present revenue and cost; thus, a negative value is obtained if there is a net expense. In this paper, all alternatives have a net monetary cost. Therefore, although potential revenue is included, Equation 4 most accurately describes the net present cost and is stated as a positive expense.

for each alternative through unit costs; these values will be unique for each alternative and are expressed in terms of volume treated, mass of chemicals need, or as a similar cost-per-unit.

3.1 Capital Costs (*K*)

For both stormwater and wastewater cost analyses, capital and maintenance costs involve primarily equipment and labor expenses, although the relative magnitude of each of these components is appreciably different for the two sectors. Capital costs, the first variable in Equation 4, were calculated with Equation 4. In it, *E* is the engineering and contingency cost percentage and *I* is the installation cost percentage.

$$K = (1 + E)(1 + I)X + L \qquad \text{Equation 4}$$

For both wastewater and stormwater calculations, contingency costs are estimated to be 30% of the total equipment costs. Installation costs are estimated to be 45% of the equipment costs, a percentage recommended by the EPA (EPA, 2009), in the wastewater alternatives. Stormwater construction cost estimates include installation costs; therefore, *I* is entered as 0 in Equation 5 for these scenarios. The sum of equipment costs (*X*) is calculated from unit costs, the components of which are listed in Table 12. Land costs (*L*) are also included, and include opportunity costs of using existing space within DWRF.

Table 12: Capital cost component comparison and summary

Wastewater Cost Components	Stormwater Cost Components
Land costs (opportunity cost)	Land costs
Basin and pipe addition costs	Construction costs (estimates obtained for each BMP type from manufacturer quotes)
Chemical storage and pump costs	
Aeration equipment costs	
Installation costs	
Engineering and contingency costs	

3.2 Maintenance Costs (M)

Table 13 lists the maintenance components included in the cost analysis. These are included in the total cost equation as unit costs multiplied by size or treatment variables, such as gallons treated, mass of nutrient removed, or oxygen concentration requirements. Wastewater unit costs for both capital and maintenance costs are described in additional detail in the following section. Stormwater maintenance costs are further arranged into fixed and variable costs, the details of which are included in the discussion of the BMP REALCOST costing model.

Table 13: Maintenance cost component comparison and summary

Wastewater Cost Components	Stormwater Cost Components
Sludge recycling costs	Inlet/Outlet cleaning
Chemical addition costs	Nuisance control
Aeration power costs	Lawn mowing and care
Additional energy costs	Sediment removal
Labor costs	Annual cleanup and planting
Sidestream mixing and pumping costs	

3.3 Rehabilitation Costs (*Ry*)

Rehabilitation costs for both stormwater and wastewater are calculated as a percentage of capital costs, incurred at specified intervals during the planning horizon.

3.4 Revenue from By-Product Reuse (A)

Nutrients are a pollutant in treated wastewater or drinking water, but are necessary in agriculture to obtain greater yields. Several of the alternative sidestream wastewater technologies considered in this analysis separate nitrogen and/or phosphorus from the waste stream such that they may be post-treated and reused. The revenue from the potential sale of fertilizer is estimated as a function of the mass and quality of nutrient-rich by-product.

3.5 Other Cost Considerations

Administrative costs were not included in the analysis. Administrative costs would not vary much among alternatives, and the additional costs are not expected to produce large increases in each sector's existing budget. Particularly that for wastewater operation, as the sidestream treatment process is a small portion of the entire system. Feasibility or planning-level assessments must include these expenses; however, for order-of-magnitude comparisons this assumption is reasonable.

3.5.1 Cost Adjustments for Time

All unit costs (U) for both the wastewater and stormwater analyses were converted to 2014 dollars by Equation 5. This is true for all unit costs within the capital, maintenance, rehabilitation, and revenue calculations.

$$X = U * \frac{ENR\ CCI_B}{ENR\ CCI_P} \qquad \text{Equation 5}$$

Where (X) is the present cost, and $ENR\ CCIB/\ ENR\ CCIP$ is the ratio between the base and present Engineering News Record Construction Cost Indices. The CCI for 2014 is 9834 (ENR, 2014), base year CCIs varied by unit cost. This equation accounts for the inflation that has occurred between the base year and the present.

3.5.2 Cost Comparisons

Each of the separate scenarios for the sidestream treatment alternatives involves a varied operational or infrastructural component and thus they incur varied costs. How these were calculated is described in detail in Section 4.0 of this report and the final ranges of costs per pound removal and total mass removal are given in the results section. However, for further comparison with stormwater alternatives, it is convenient to select a representative cost for each wastewater alternative. The scenario with the greatest cost-efficiency was selected for this technology representation. Results from the treatment-cost calculations generated three groups of results: one for each RML rate analyzed. The lowest possible RML that does not affect the effluent concentration was selected to represent costs. Of the scenarios within the selected RML, the point most closely representing the average among those that resulted in effluent nutrient concentrations less than the current average were included in the side-by-side comparison of results was selected as representative. An additional comparison was made which includes only the scenarios in which the Regulation 85 nutrient standards were met. For the first of these comparisons, the effluent nutrient scenarios were only selected only on the basis of the nutrient treated by the side-stream technology. In the second comparison, both nutrient types were considered.

4.0 COST ESTIMATES AND DATA

4.1 Wastewater Cost Estimates

A spreadsheet model was created to compare nutrient removal sidestream process lifecycle costs. Capital, maintenance, and revenue expenses for wastewater technologies were primarily variable costs that depend on the level of treatment or volume of influent. Unit costs, adjusted to 2014 dollars, were obtained primarily from literature estimates and are listed in Table 14.

Table 14: Unit Costs for Wastewater Nutrient Removal Alternatives

Item	*Unit Cost 2014 Dollars*		*Source*
Capital Costs - Basin Additions, Chemical Storage and Pumps, Aeration Equipment			
Land costs	$0.86	ft^2	(Olson, Urbonas, et al., 2013)
Installation costs	45%	Equipment Costs	(EPA, 2009)
Tank Costs	$1.32	$/gal	(Hydromantis Environmental Software Solutions, n.d.)
WW Piping Costs	$0.40	$/gal	*30% of building costs*
Stripping Tank	$46,374.29	$/ft.	(Hydromantis Environmental Software Solutions, n.d.)
ED Membranes	$3,071,205.50	$/MGD	(Leitz & Boegli, 2001)
Storage Tank	$2,025.96	$/1K tank	(Colorado University, 2002)
Pumps	$14,181.71	$/pump	(Colorado University, 2002)
Blower	$593,590.95	$/ea.	(Hydromantis Environmental Software Solutions, n.d.)
Maintenance Costs - Recycling, Aeration, Chemicals, Energy, Labor, and Sidestream			
Mainstream RML	$1,764.87	$/MGD	(Hydromantis Environmental Software Solutions, n.d.)
Mainstream RAS	$1,775.47	$/MGD	(Hydromantis Environmental Software Solutions, n.d.)
Sidestream RML	$1,764.87	$/MGD	(Hydromantis Environmental Software Solutions, n.d.)
Mainstream Power	$166,364.46	$/mgO2/L	(Hydromantis Environmental Software Solutions, n.d.)
Sidestream Aeration	$166,364.46	$/mgO2/L	(Hydromantis Environmental Software Solutions, n.d.)
Hydrated Lime (Ca(OH)2)	$0.24	$/lb.	(Hydromantis Environmental Software Solutions, n.d.)
Aluminum (Al2(SO4)3)	$0.36	$/lb.	(Hydromantis Environmental Software Solutions, n.d.)
Ferric Chloride	$0.48	$/lb.	(Hydromantis Environmental Software Solutions, n.d.)
Magnesium Chloride	$287.01	$/ton	(Seymour, 2009)
Anit-scalant	$16.89	$/gal	(Leitz & Boegli, 2001)
NiSO4 (ED)	$7.45	$/lb.	(Pruyn, Harrington, & Smith, 1970)
Additional Electricity Cost	$0.04	$/kWh	(Citry of Fort Collins, 2014)
Annual Labor	$35.00	per hour	Engineering Judgment
Sidestream Mixing	$46,597.03	$/MGD	(Colorado University, 2002)
Sidestream Pumping	$40,519.16	$/MGD	(Colorado University, 2002)
Membrane Pumping	$10,605.08	$/kgal	(Pruyn et al., 1970)
Potential Revenue - Fertilizer By-Product			
Good Quality	$574.01	$/ton	(Seymour, 2009)
Fair Quality	$344.41	$/ton	Estimated from (Seymour, 2009)
Poor Quality	$114.80	$/ton	Estimated from (Seymour, 2009)
Rehabilitation Costs - Equipment Replacement			
Piping and Tanks	40% Equipment costs every 20 years		(Sewer Infrastructure Advisory Group, 2013)
Blowers and Pumps	80% Equipment costs every 15 years		(Sewer Infrastructure Advisory Group, 2013)
ED Membranes	100% Equipment costs every 7 years		(Pruyn et al., 1970)
Stripping Tank	50% Equipment costs every 7 years		Engineering Judgment

For each technology, a range of costs was calculated that is associated with the input parameter changes highlighted in Table 4. These produced different levels of treatment; thus, cost-treatment curves cold be developed. Appendix B includes unadjusted literature unit costs. Rehabilitation costs account for equipment life expectancies which are lower than the treatment planning horizon used in this analysis; they are estimated as percentages of the initial equipment cost and informed by literature estimates (EPA, 2009).

4.1.1 Alternatives Cost Determinations

The variables affecting cost for Modified Bardenpho, CaRRB, and ANAMMOX are primarily basin size, aeration rate, or recycle rate, which are directly entered into the spreadsheet cost model without intermediate calculation. However, the four physicochemical processes modeled in this analysis do not have specific BioWin model elements relating process input to effluent water quality. Therefore, the process design variables – including chemical addition rates, tower heights, and electricity demands – were calculated as a function of nutrient removal percentages in spreadsheet models, using literature-based methods. Unit calculations for each technology are as described below, and full calculation sets are included in Appendix C.

Centrate and RAS Reaeration Basin (CaRRB)

Implementation requirements for sidestream CaRRB that affect capital costs include the reactor tank installation and complimentary piping and wastewater pumping. In addition to the internal recycle of the sidestream unit, part of the mainstream RAS is diverted to the CaRRB reactor. Aeration is necessary to achieve nitrification; due to the small size of the tank, one blower is deemed sufficient to achieve the desired dissolved oxygen concentration (Air Force, 1988).

Anaerobic Ammonium Oxidation (ANAMMOX)

ANAMMOX infrastructure requirements include a pre-treatment nitritation reactor, which is aerated at a rate of 0.5mg/L dissolved oxygen to avoid complete nitrification and nitrate toxicity and a second tank that contains the ammonia oxidizers (Musabyimana, 2008). The hydraulic retention time

(HRT) of the initial bioreactor was the controlling factor used in this analysis, and was varied through the basin volume. No chemical additions are necessary (AECOM, 2012).

Ammonia Stripping

Tower height and air flow requirements associated with specified nitrogen removal efficiencies were found following the method outlined by Huang and Shang (Huang & Shang, 2006), in which an influent and effluent ammonia (aqueous) concentration are specified, as well as an influent ammonia (gaseous) concentration. Influent ammonia concentrations were kept constant while those of the effluent were varied through each scenario. The influent gaseous concentration is dependent on influent aqueous concentration and identified through a nomograph provided by Huang and Shang. The effluent gaseous ammonia concentration was then determined. The flow rate, area loading rate, and tower geometry (area and length) are specified and required in determining the required tower height.

Electrodialysis

The current needed to achieve a given level of nitrogen removal was calculated using Equation 6 (Shaffer & Mintz, 1980):

$$I = \frac{zFQ_f(C_{inlet}^d - C_{outlet}^d)}{N\epsilon} \qquad \text{Equation 6}$$

Where ε is the current utilization efficiency (assumed to be 0.9), N is the number of cell pairs (2 pairs), z is the charge of the ion (+1), F is Faraday's constant (96485 amps), Q_f is the diluate flow rate, $C^d{}_{inlet}$ is the diluate ED cell inlet concentration, and $C^d{}_{outlet}$ is the diluate ED cell outlet concentration. The current was then converted to required power requirement (kilowatts) by the assumption that 170 volts are applied in the sidestream treatment. Anti-scalant and nickel sulfate, used to sustain the electro-chemical potential, are added to the treatment tank to maintain efficient pollutant removal (Valero et al., 2011).

Struvite Precipitation

For each desired effluent centrate phosphorus concentration, the required magnesium chloride was calculated with a mass-balance ratio. Ideal Mg:P molar ratios are reported to be between 1 and 1.5

mol/mol (Forrest et al., 2008; Wang et al., 2005). In this analysis, 1.5 mol/mol was used for a conservative estimate, and the required chemical addition was calculated by stoichiometric conversion.

Selective Adsorption

The amounts of aluminum sulfate needed to achieve required phosphorus concentrations were calculated in a similar method as that of struvite precipitation. A molar ratio of 1.75 moles Al/mol P (Sengupta & Pandit, 2011) was used to calculate the annual mass of aluminum sulfate required for selective adsorption.

4.1.2 Capital Costs

Capital unit estimates are given in Table 15. Because Modified Bardenpho is the current treatment design at DWRF, no capital costs are associated with this scenario. Capital costs are primarily dependent on the influent flow rate, which stays constant throughout this analysis. The exceptions are ANAMMOX and ammonia stripping, whose processes are reliant on tank size. Although DWRF currently owns the space in which a sidestream treatment would be placed, implementing such technology would present opportunity costs. These were estimated as the land use value for that area. It was assumed that a sidestream unit would take approximately 100,000 square feet of space. At a land value cost of $0.86 per square foot – equivalent to the cost of undeveloped land (Olson, 2013) – the total opportunity cost is approximately $86,000. The value for undeveloped land was selected because the space within DWRF is not practical for commercial, residential, or industrial production purposes. These unit estimations were multiplied by the unit costs listed in Table 14 to obtain the total unit capital costs, listed in Table 16.

Table 15: Capital Unit Quantities for Sidestream Wastewater Nutrient Removal Alternatives

CaRRB	ANAMMOX	Electrodialysis
200,000 (gal) Tanks	0.7 – 2.5 (million) (gal) Tanks	200,000 (gal) Tanks
170,000 (gal) Piping	170,000 (gal) Piping	170,000 (gal) Piping
2 (ea.) Pumps	2 (ea.) Blowers	0.17 (MGD) ED Membranes
1 (ea.) Blowers	100,000 sqft land	2 (ea.) Storage Tank
100,000 sqft land		2 (ea.) Pumps
		1000,000 sqft land
Ammonia Stripping	**Selective Adsorption**	**Struvite Precipitation**
170,000 (gal) Piping	200,000 (gal) Tanks	200,000 (gal) Tanks
12.5 - 50 (ft.) Stripping Tank	170,000 (gal) Piping	170,000 (gal) Piping
1 (ea.) Storage Tank	1.0 (ea.) Storage Tank	1 (ea.) Storage Tank
1 (ea.) Pumps	1.0 (ea.) Pumps	2 (ea.) Pumps
1 (ea.) Blowers	100,000 sqft land	1 (ea.) Blowers
100,000 sqft land		100,000 sqft land

For each alternative, the individual unit costs were added to generate a total cost, which is also displayed. A range of costs are given in the table for ANAMMOX and ammonia stripping, corresponding to the range of treatment modeled and presented in the discussion section of this report.

Table 16: Sidestream Capital Costs (*K*) for Wastewater Nutrient Removal Alternatives

Capital Cost Item	CaRRB	ANAMMOX	Electrodialysis	Ammonia Stripping	Selective Adsorption	Struvite Precipitation
Land	$86,000	$86,000	$86,000	$86,000	$86,000	$86,000
Engineering and Contingency	$588,024	$839,007	$548,912	$763,490 - $1,772,131	$236,691	$589,199
Installation costs	$429,536	$624,264	$399,191	$67,574 - $2,318,715	$156,950	$430,448
Tank Costs	$264,996	$927,486 – 3,312,449	$264,996	$0	$264,996	$264,996
WW Piping Costs	$676,574	$67,574	$67,574	$67,574	$67,574	$67,574
Stripping Tanks	$0	$0	$0	$579,679 - $2,318,715	$0	$0
ED Membranes	$0	$0	$1,228,482	$0	$0	$0
Chemical Storage Tank	$0	$0	$4,052	$2,026	$2,026	$2,026
Pumps	$28,363	$0	$28,363	$14,182	$14,182	$28,363
Blowers	$593,591	$1,187,182	$0	$593,591	$0	$593,591
TOTAL	**$2,058,084**	**4,550,351 – 9,391,351**	**$1,921,193**	**$2,672,214 - $6,202,457**	**$828,419**	**$2,061,197**

4.1.3 Maintenance Costs

Most treatment curve calculations were created by varying operational inputs; in Table 17 all maintenance cost units are provided, including the ranges of selected treatment variables. The first three items for each (RML, RAS, and Mainstream aeration) are costs of continuing the mainstream BNR treatment, albeit with a lower RML flow rate. Because these are constant through each sidestream process scenario, they are included in a separate table section (*sidestream processes: mainstream component*).

Table 17: Sidestream Operation and Maintenance Unit Quantities for Wastewater Nutrient Removal Alternatives

CaRRB		ANAMMOX		Electrodialysis	
0.05-0.30	(MGD) Recycle	0.1	(MGD) Recycle	1000	(gal) Anti-scalant
2	(mgO2/L) Aeration	0.5	(mgO2/L) Aeration	140000	(lb.) NiSO4
40	(hr./wk.) labor	40	(hr./wk.) labor	730 - 1,876	1000 (kWh) Electricity
0.17	(MGD) Mixing	0.17	(MGD) Mixing	0.17	(hr./wk.) labor
0.17	(MGD) Pumping	0.17	(MGD) Pumping	170	(kgal) Membrane Pumping
Ammonia Stripping		**Selective Adsorption**		**Struvite Precipitation**	
0.89 - 2.29	(mgO2/L) Aeration	0.1	(MGD) Recycle	0.1	(MGD) Recycle
6300	(lb.) Ca(OH)2	353,398 - 908,731	(lb.) Al2(SO4)3	0.3	(mgO2/L) Aeration
40	(hr./wk.) labor	40	(hr./wk.) labor	55 - 140	(lb.) MgCl
0.17	(MGD) Pumping	0.17	(MGD) Mixing	40	(hr./wk.) labor
		0.17	(MGD) Pumping	0.17	(MGD) Mixing
				0.17	(MGD) Pumping

These can be compared with maintenance requirements of solely using Modified Bardenpho. Labor rates necessary for operation and maintenance were assumed to vary little between nutrient removal technologies, as this operational expense is minor compared to power and chemical costs (Daw & Hallett, 2012). Wastewater pumping rates are also identical throughout all scenarios due to the constant centrate flow rate in all analyses. The total costs listed in Table 18 for each sidestream treatment include the mainstream treatment maintenance costs as well as sidestream unit costs.

Table 18: Annual Sidestream Maintenance Costs (*M*) for Wastewater Nutrient Removal Alternatives

CaRRB		ANAMMOX		Electrodialysis	
$88 - $529	(MGD) Recycle	$172	(MGD) Recycle	$176	Recycle
$332,729	(mgO2/L) Aeration	$83,182	(mgO2/L) Aeration	$16,892	(gal) Anti-scalant
$72,800	(hr./wk.) labor	$72,800	(hr./wk.) labor	$1,043,000	(lb.) NiSO4
$7,921	(MGD) Mixing	$7,921	(MGD) Mixing	$75,040 - $29,200	(kWh) Electricity
$6,888	(MGD) Pumping	$6,888	(MGD) Pumping	$72,800	(hr./wk.) labor
				$4,242,030	(kgal) Membrane Pumping
$420,427 - $420,868		**$170,968**		**$3,088,197 - $3,010,771**	
Ammonia Stripping		**Selective Adsorption**		**Struvite Precipitation**	
$148,064 - $380,975	(mgO2/L) Aeration	$176	(MGD) Recycle	$176	(MGD) Recycle
$1,503	(lb.) Ca(OH)2	$126,426 - $325,094	(lb.) Al2(SO4)3	$49,909	(mgO2/L) Aeration
$72,800	(hr./wk.) labor	$72,800	(hr./wk.) labor	$15,785 - $40,181	(lb.) MgCl
$6,888	(MGD) Pumping	$7,921	(MGD) Mixing	$72,800	(hr./wk.) labor
		$6,888	(MGD) Pumping	$7,921	(MGD) Mixing
				$6,888	(MGD) Pumping
$229,255 - $462,165		**$214,212 - $412,880**		**$153,481 - $177,877**	

As in Table 17, mainstream unit costs are listed in a separate table component to provide comparison to regular BNR operation. These costs were determined by multiplying unit amounts in Table 17 by unit costs provided in Table 14. The costs displayed here are yearly maintenance costs and are not adjusted to a net present cost.

4.1.4 Potential Revenue

Side-stream sludge production, calculated from literature values (Sengupta & Pandit, 2011; Wang et al., 2005) as a function of chemical dosage and wastewater flow rate, was used to estimate the fertilizer production from electrodialysis, ammonia stripping, and selective adsorption. The mass of struvite – which can be directly applied as fertilizer – produced per mole of phosphorus removed is available as a literature estimate (Wang et al., 2005) and was used as the basis for its revenue calculations. Table 19 gives fertilizer production estimates for each sidestream technology and shows the revenue estimates for the nutrient recycling technologies.

Table 19: Potential Annual Nutrient Generation and Revenue (*A*) through By-Product Reuse

Electrodialysis		Ammonia Stripping		Selective Adsorption		Struvite Precipitation	
130 - 200	(ton) Fair Quality	130-200	(ton) Fair Quality	15 - 38	(ton) Fair Quality	149 - 384	(ton) Good Quality
$44,773 - $68,882		$44,773 - $68,882		$5,166 - $13,088		$85,528 - $220,421	

These are annual revenues and are not adjusted yet through Equation 2. The nitrogen-containing by-product from electrodialysis, ammonia stripping, and selective adsorption are not as valuable a fertilizer as that from struvite precipitation; thus, the revenue predicted for these sidestream treatments are less than that for struvite.

4.1.5 Rehabilitation Costs

Rehabilitation costs were calculated for each mechanical, electrical, or infrastructure item as a percentage of equipment costs. Table 20 shows treatment-specific cost estimates for sidestream (in addition to mainstream treatment) rehabilitation costs.

Table 20: Rehabilitation Costs (*R*) for Wastewater Sidestream Nutrient Removal Alternatives

Rehabilitation Cost Item	CaRRB	ANAMMOX	Electrodialysis	Ammonia Stripping	Selective Adsorption	Struvite Precipitation
Tanks and Piping	$266.056	$160,058	$269,297	$55,680	$267,677	$267,677
Blowers and Pumps	$663,418	$1,266,327	$30,254	$648,291	$15,127	$663,418
ED Membranes	$0	$0	$1,491,728	$0	$0	$0
Stripping Tank	$0	$0	$0	$828,112 - $3,312,449	$0	$0
TOTAL	**$929,474**	**$2,062,375 - $3,970,349**	**$1,791,280**	**$1,532,083 - $4,016,420**	**$282,804**	**$931,095**

4.2 Stormwater Cost Estimates

Costs of stormwater control measures were evaluated through BMP-RealCost, a spreadsheet cost tool developed in 2010 by the Urban Drainage and Flood Control District in collaboration with Colorado State University (UFCD, unpublished). Administrative costs were removed from the analysis for more accurate comparisons with wastewater technology costs. For the cost analyses, it is assumed that BMP effectiveness does not change over its lifetime, and that regular maintenance is performed.

4.2.1 Alternatives Cost Determinations

Stormwater scenario capital costs include both fixed and variable costs that depend on individual BMP size. The bioretention scenario includes land costs as well, which were determined as a function of both land use and BMP volume. Tables 21 and 22 list the unit costs for extended detention basins and bioretention basins. Land cost values were obtained from BMP-REALCOST and represent average land values for each land use type.

Table 21: Unit Capital Cost Variables

Capital Cost Variable	Extended Detention Basins	Bioretention Basins
Base Cost	$10,000	$10,729
Unit Cost	$0	$9.93

Table 22: Bioretention Land Use Unit Costs

Land Use Type	Bioretention Land Cost
Undeveloped/Open land	$0.86 per ft^2
Residential use	$3.16 per ft^2
Commercial use	$4.60 per ft^2
High density use	$40.25 per ft^2

Maintenance activities for extended detention basins include inlet and outlet cleaning, nuisance control, outlet maintenance, lawn care, and sediment removal from both the forebay and basin. Each of these must occur with a different frequency, which is accounted for in the calculation of annual maintenance costs. Bioretention basins require only annual clean up and planting. Table 23 lists these values for extended detention basins and bioretention basins.

Table 23: Unit Maintenance Cost Variables

Maintenance Cost Variable	Extended Detention Basins	Bioretention Basins
Constant Costs	$1,849 per BMP	$0 per BMP
Variable Costs	$2,782 per AF	$0.62 per CF

4.2.1.1 BMP-RealCost Model

The BMP-RealCost tool determines the planning-level lifecycle costs of stormwater controls implemented in urban environments. In order to calculate BMP implementation costs, BMP-RealCost first determines the number and size of BMPs within each subcatchment based on specified land cover and drainage area values. It also allows for the analysis of BMP effectiveness in subcatchments uniform in land cover, imperviousness, soil type, and slope.

Scenario 1

Scenario 1 model predictions for both the number of BMPs and the capital cost per BMP were over-written with GIS data and engineering judgment as this scenario requires only rehabilitation of pre-existing stormwater structures.

Scenario 2

Bioretention basins are sized according to their calculated water quality capture volume (WQCV), calculated by Equation 7 (CH2M HILL & City of Colorado Springs City Engineering Division, 2002):

$$V = A * a(0.91EI^3 - 1.19EI^2 + 0.78EI) \qquad \text{Equation 7}$$

Where A is the runoff-contributing area, a is the drawdown time coefficient, and EI is the effective imperviousness, here equivalent to total imperviousness. The drawdown time is 0.8 for bioretention basins (UDFCD, 2011). It was assumed that no area in the subcatchment is left untreated. The number of BMPs required (N) is calculated by Equation 8:

$$N = (CA * I_r) \qquad \text{Equation 8}$$

Where CA is the total impervious area in sub-catchment area, and I_r is the percent imperviousness of the subcatchment. The BMP numbers within each land use type for each scenario is provided in Table 24.

Table 24: Stormwater Scenario Unit Quantities

Land Use	Scenario 1			Scenario 2		
	BMP Size	Units	No. of BMPs	BMP Size	Units	No. of BMPs
Commercial	0.41	AF	86	1191.80	CF	2289
Industrial-Heavy	0.68	AF	49	1295.47	CF	801
Industrial-Light	0.27	AF	31	1295.47	CF	495
Parks, Cemeteries	2.72	AF	6	1679.67	CF	88
Residential - AP	n/a	n/a	n/a	1191.80	CF	36
Residential - MU	0.54	AF	41	1140.98	CF	380
Single Family 1000	0.67	AF	24	1197.90	CF	1258
Single Family 2000	1.11	AF	77	1245.16	CF	821
Single Family 4000	2.22	AF	4	1466.23	CF	337
Undeveloped	n/a	n/a	n/a	1945.97	CF	10

4.2.2 Capital Costs

Table 25 shows the unit and total costs for stormwater scenario capital costs.

Table 25: Capital Costs (*K*) for Stormwater Nutrient Removal Scenarios

Capital Cost Variable	Extended Detention Basins	Bioretention Basins
Base Cost	$3,180,000	$69,899,435
Unit Cost	$0	$80,175,551
Land Cost	$0	$109,001,975
TOTALS	**$3,180,000**	**$259,076,961**

4.2.3 Maintenance Costs

Table 26 displays the unit and total operational costs for both stormwater scenarios. These estimates do not include net present value conversions, so they are lower than final reported values.

Table 26: Maintenance Costs (*M*) for Stormwater Nutrient Removal Scenarios

Maintenance Cost Variable	Extended Detention Basins	Bioretention Basins
Constant Costs	$587,982	$0
Variable Costs	$627,421	$5,005,926
TOTALS	**$1,215,403**	**$5,005,926**

4.2.4 Potential Revenue

No potential revenue (*A*) from nutrient recycling or other activities is possible through the implementation of stormwater controls included in this analysis.

4.2.5 Rehabilitation Costs

Rehabilitation costs (*R*) describe the percent of construction costs necessary to restore the BMP as it reaches its long-term pollutant removal capacity and occur at intervals equal to the expected design life of the BMP. For extended detention basins this is 80% every 35 years due to infrastructure replacement (Olson, Roesner, Urbonas, & Mackenzie, 2013). Rehabilitation costs for bioretention are assumed to be 100% of construction costs every 15 years and include the replacement of filter media and replanting of vegetation (Olson, Roesner, et al., 2013).

4.3 Impacts to Budget and Rate Payers

For each alternative, the impact to the city budget and utility consumers was estimated as both total costs per person and by estimating a household's likely bill increase per month. Consumer-normalized capital costs (K_P) were calculated using Equation 9. Equation 10 was used to calculate a similar estimate for operating costs.

$$K_P = \frac{K}{P}$$ Equation 9

$$M_P = \frac{NPC - K}{PH * P}$$ Equation 10

Where K is the capital cost of the alternative, PH is the planning horizon (20 years), P is the Fort Collins population, and G is the total gallons treated over the planning horizon. NPC is the net present cost; by subtracting capital costs from this value, one obtains the net present operation and maintenance costs. Fort Collins has a population of about 150,000 and this value was used in each scenario.

Because consumer utility rates are frequently calculated using the estimated usage rate in mind, an alternative evaluation was performed to better compare the impact of sidestream treatment and BMP implementation on monthly consumer bills. Consumer charges per 1,000 gallons treated (r) were calculated using Equation 11. The resulting monthly bill increase per household (r_h) was found using Equation 12.

$$r = \frac{NPC}{P * (\frac{w_r}{1000}) * 30.4}$$ Equation 11

$$r_h = r * \left(\frac{w_r}{1000}\right) * h * 30.4$$ Equation 12

In both equations, w_r is the wastewater generation rate per person, assumed here to be 70gpcd; this is divided by 1,000 to yield the treatment demand in kilo-gallons. The number of people per household is represented by h, assumed to be 3 people. The term 30.4 is the average number of days per month. Calculated percent increases of consumer charges for wastewater alternatives include only the wastewater portion of a household's utility bill, not the entire water statement.

Generally, stormwater utility bills are not variable upon volume of water treated but the area and land use type of the property. Therefore, stormwater rates (per volume use) were not included in the analysis; however, they may be compared with wastewater values as expenses per person. Table 27 displays the expected costs and changes to a single-family residence utility bill. The current rates for wastewater treatment are included, as are the capital and maintenance costs per person calculated for the Modified Bardenpho (currently technology) alternative. Costs per capita for the remaining wastewater alternatives include the continued operation of the mainstream BNR and thus are not in addition to the current costs. This presentation style was chosen so that the reduced requirements of mainstream treatment could be included in the evaluation of sidestream costs. The costs per capita increase summarize the amount by which the sidestream technology will cause rates to increase per person per month.

Table 27: Impact of technology alternatives to sector budget and rate payer charges

Scenario	Cost per Capita		Monthly Utility Rates		
	Capital	O&M (y^{-1})	Per 1000gal -cap	Per household	Per capita increase
Current Rate (M. Bardenpho)	$0	$96	$3.44	$21.97	N/A
CaRRB	$14	$93	$3.68	$23.53	$7.84
ANAMMOX	$20	$38	$1.54	$9.84	$3.28
Selective Adsorption	$6	$40	$1.59	$10.17	$3.59
Struvite Precipitation	$14	$42	$1.69	$10.78	$3.59
Ammonia Stripping	$41	$102	$4.09	$26.14	$8.71
Electrodialysis	$13	$646	$25.32	$161.74	$54
EDB Retrofits	$21	$9	N/A	N/A	N/A

5.0 RESULTS

5.1 Mass-Loading and Cost Results

Total annual nitrogen and phosphorus discharge from wastewater and stormwater are presented in Figure 18; these results represent the current nutrient loading conditions for both sources, including runoff treatment from existing SCMs and nutrient removal by the Modified Bardenpho process at DWRF. For both nutrients, the average mass loadings from wastewater and stormwater are within the same order of magnitude: Fort Collins stormwater contributes approximately 28% of the nitrogen load and 40% of the total point and nonpoint phosphorus load. The phosphorus load from stormwater is substantial, indicating potential to reduce phosphorus concentrations in receiving water bodies through stormwater controls that remove phosphorus.

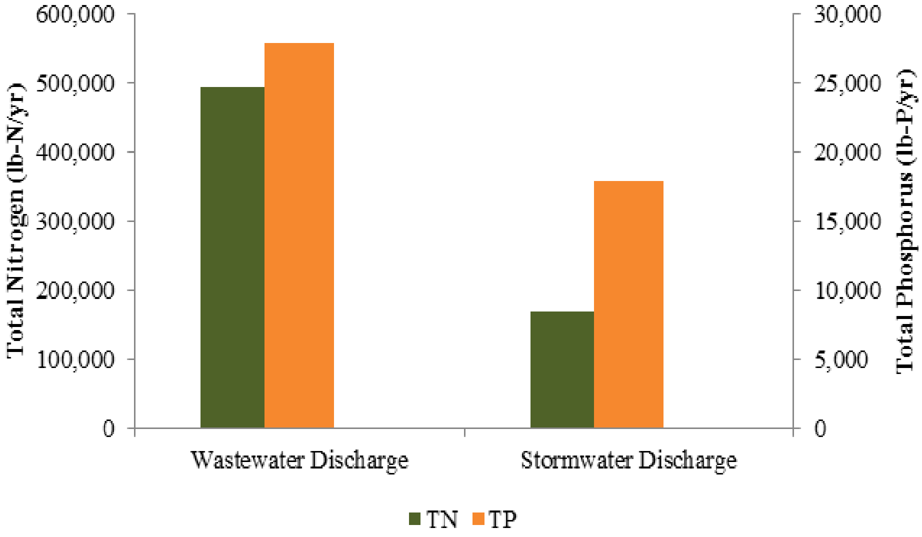

Figure 18: Existing average annual nutrient discharge from wastewater and stormwater

Baseline total effluent concentrations of nitrogen and total phosphorus are 14.9 mg/L and 2.02 mg/L, respectively. All alternative treatment processes reduced their target nutrient concentrations below these levels for all process variations (Figure 19). Most were not effective at reducing the non-target nutrient, although many technologies are predicted to provide limited reduction. The extents of total nitrogen and total phosphorus removal with CaRRB treatment are inversely related. As WAS to the

CaRRB reactor increases, TN is reduced 18% to 29% of its baseline concentration and TP increases 6 to 12 percent above its baseline value. ANAMMOX consistently removes nitrogen and phosphorus 21% and 3% below baseline concentrations. Of the nitrogen-targeting technologies, ammonia stripping and electrodialysis removed the least nitrogen, compared to baseline values and removed total phosphorus to concentrations to about the values achieved by ANAMMOX. Struvite precipitation was the only technology to remove both phosphorus and nitrogen by a considerable extent. Its nutrient removal potential ranges from 45% to 75% below baseline for TP concentrations and from 2.3% and 7.4% below baseline for TN concentrations.

All wastewater alternatives are sidestream processes, and although a technology may decrease centrate nutrient concentrations further than the displayed values, effluent concentrations are also highly dependent on influent wastewater quality. The two sidestream biological processes – CaRRB and ANAMMOX – show the least variability in effluent nutrient concentrations; despite process variations which alter nitrification or nitritation rates, the modeled nitrogen and phosphorus remain within a 1mg/L range for each recycle mixed liquor scenario. Struvite precipitation and selective adsorption show similar effects on the plant effluent water quality. Ammonia stripping and electrodialysis, modeled together in BioWin, produced little variation in phosphorus concentrations.

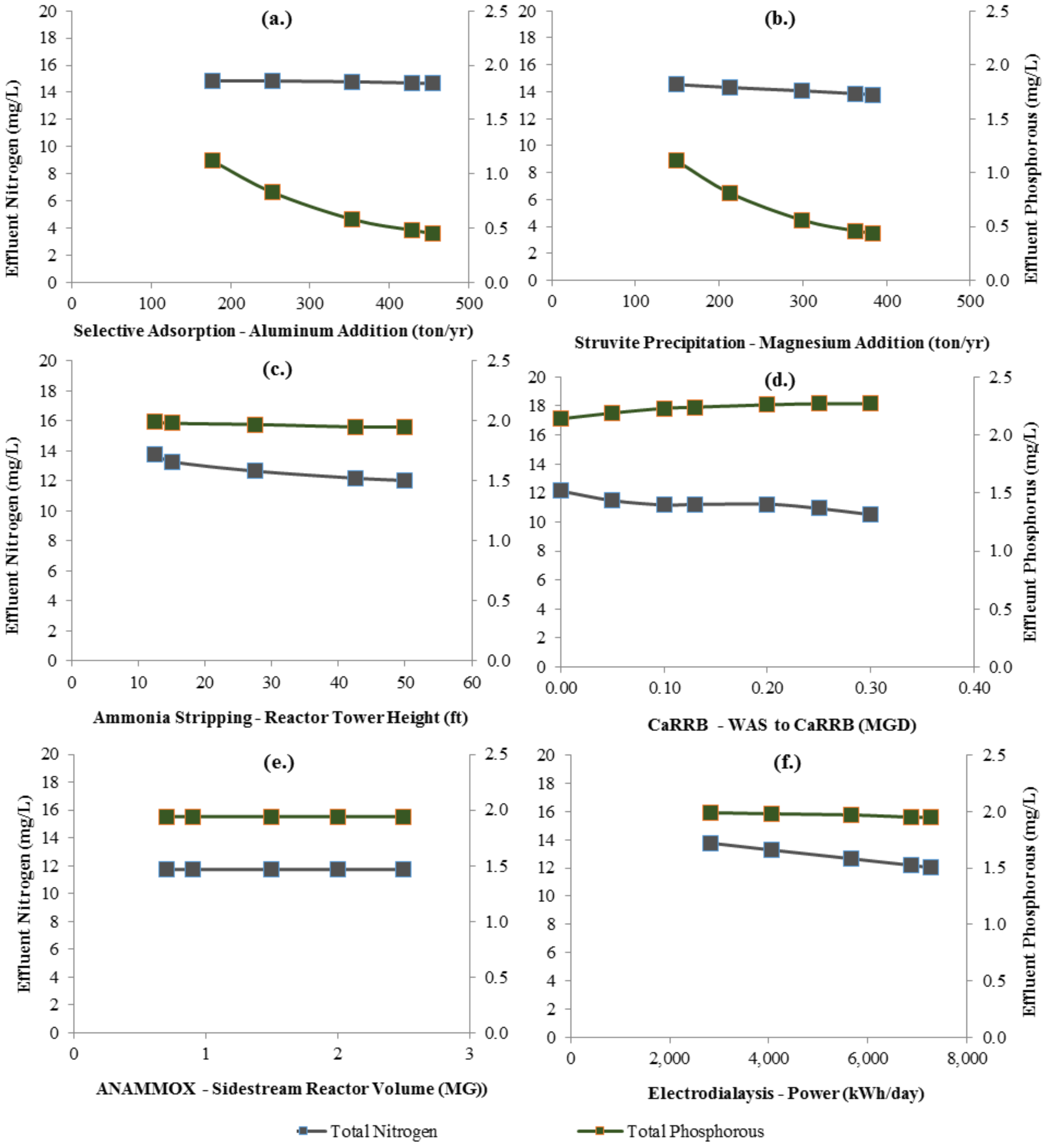

Figure 19: Modeled effluent concentrations from (a) selective adsorption, (b) struvite precipitation, (c) ammonia stripping, (d) CaRRB, and (e) ANAMMOX, and (f) electrodialysis; when sidestream treatment is not in place, baseline effluent total nitrogen and total phosphorus values are 14.9 mg/L and 2.02 mg/L, respectively

To evaluate the effect of RML on sidestream nutrient removal and verify that the system remains carbon-limited, the ammonia stripping alternative scenarios were modeled with 5Q, 2Q, and 0.25Q mainstream RML rates (Figure 20). Ammonia stripping was chosen as the technology to demonstrate this relationship because it does not require carbon in its own process, and so does not introduce additional influences to the carbon-nitrification relationship. Electrodialysis also meets this criteria, but has a cost-treatment efficiency much lower than that of ammonia stripping.

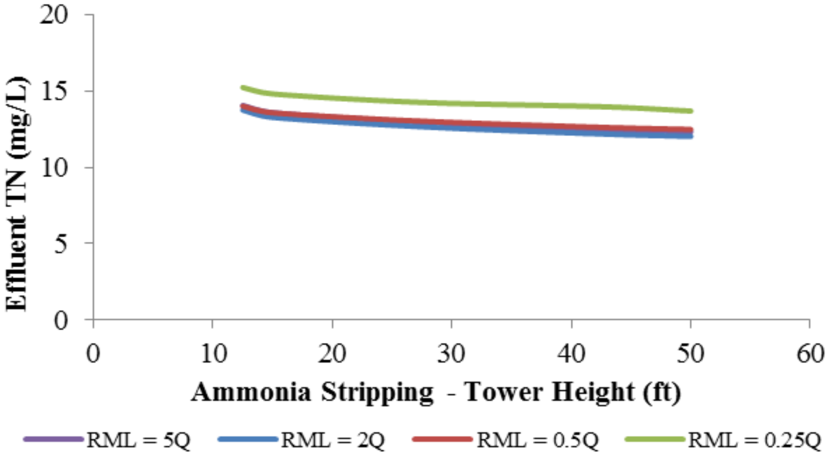

Figure 20: Effect of varying RML on ammonia stripping treatment efficiency

As RML increases, the effluent nitrogen decreases; however, between 0.5Q and 5Q, the changes to the effluent are once again nearly imperceptible. Therefore, the influent COD continues to have a larger impact on effluent nitrogen than RML. Comparing the effect of carbon addition in Figure 17 and the effect of sidestream treatment in Figure 20 reveals that increasing the influent COD, at a RML flow rate of 2Q, can be as effective as the sidestream alternatives at reducing effluent nitrogen. At 800 mg/L influent COD, the nitrogen removal exceeds that predicted for any sidestream alternative. Since the system is carbon limited and the increased RML does not practically effluent nitrogen, all other processes were modeled only with the current mainstream RML rate.

Costs of sidestream treatment varied among the alternatives, and also with the level of treatment (Figure 21). Electrodialysis and CaRRB costs varied the least as process parameters were changed, despite the more varied level of nitrogen removal compared to ANAMMOX.

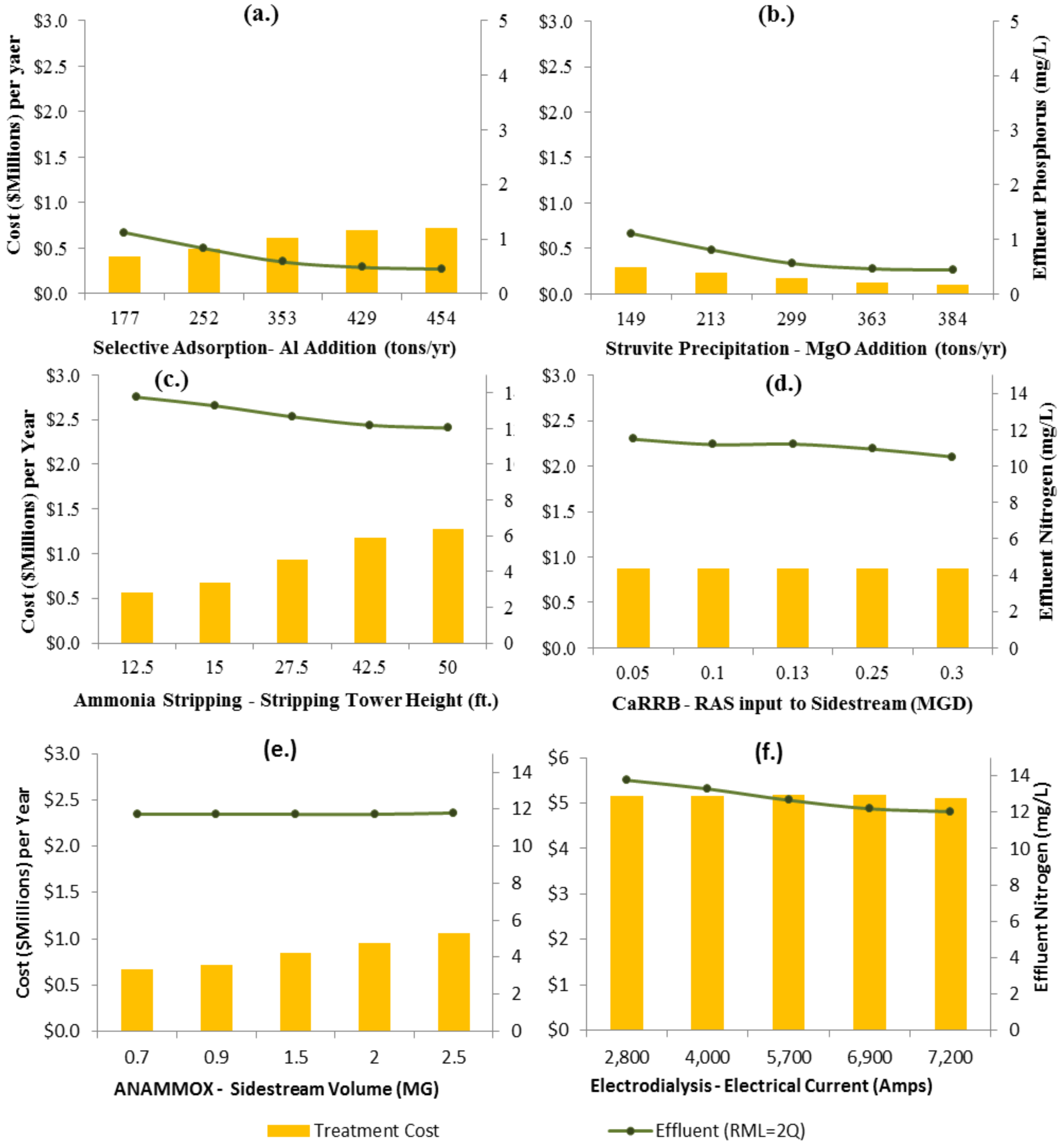

Figure 21: Modeled effluent concentrations from (a) selective adsorption, (b) struvite precipitation, (c) Ammonia Stripping, (d) CaRRB (e) AMAMMOX, and (f) electrodialysis; note the changed scale for (f)

The trend of cost per mass of nutrient removed is unique for each alternative (Figure 22); some alternatives' main variable cost is part of the capital cost, while for others it is the maintenance costs. Additionally, the alternatives vary in the degree their treatment effects the effluent nutrient concentrations.

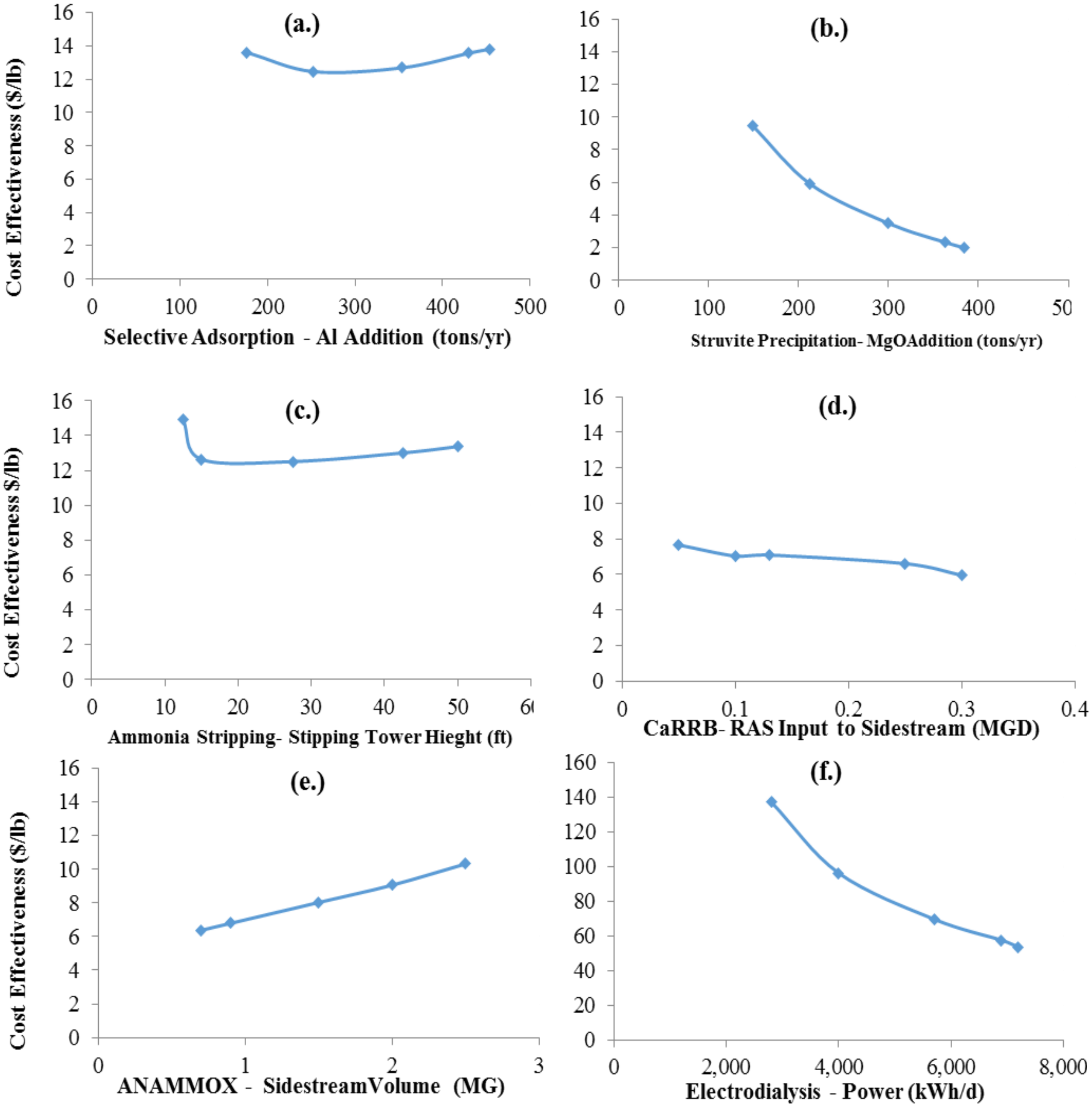

Figure 22: Modeled cost efficiencies from (a) selective adsorption, (b) struvite precipitation, (c) Ammonia Stripping, (d) CaRRB (e) AMAMMOX, and (f) electrodialysis; note the changed scale for (f)

Costs per pound removal, in excess of existing effluent concentrations, for the alternative wastewater technologies are listed in Table 28. A range of cost-per-pound results are reported for the wastewater technologies, representing in minimum and maximum removal obtained through sidestream process variation and recycle mixed liquor alterations.

Table 28: Nutrient Removal Cost-Effectiveness for Sidestream Processes

Wastewater Technology	Removal Effectiveness	
CaRRB	6.0 – 7.7	$/lb.-N
ANAMMOX	6.4 – 10.3	$/lb.-N
Ammonia Stripping	13.4 – 14.9	$/lb.-N
Electrodialysis	53.6 – 137.2	$/lb.-N
Struvite Precipitation	2.0 – 9.5	$/lb.-P
Selective Adsorption	12.5 – 13.8	$/lb.-P

In each of the alternative stormwater scenarios, nitrogen and phosphorus were reduced in nearly identical percentages from the existing condition (Figure 23). Retrofitting EDBs, including retrofits to all land use types, decreased nitrogen and phosphorus loading by 20% and 22%, respectively. Implementing new PLDs reduced nitrogen and phosphorus loading by 31% and 32%, respectively.

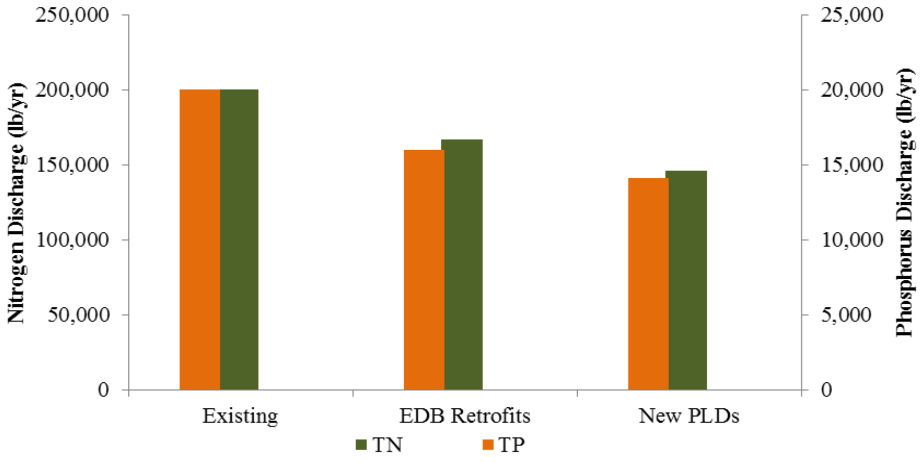

Figure 23: Stormwater Nutrient Loads from Alternative Scenarios

Table 29 lists the costs per pound removal, compared to existing discharges, of the alternative stormwater treatment scenarios. Both cost per unit nitrogen and cost per unit phosphorus are given, although they are not additive costs.

Table 29: Nutrient Removal Cost-Effectiveness for Stormwater Alternative Treatment Scenarios

Stormwater Technology	Nutrient Removal Cost-Effectiveness

SCM Retrofits (EDB)	$410	$/lb.-P	$50	$/lb.-N
SCM Implementation (PLD)	$1,570	$/lb.-P	$160	$/lb.-N

To make a comparison of the various nutrient removal approaches, the highest efficiency scenario, in other words lowest cost-per-pound removal, was selected for each alternative (Table 30).

Table 30: Sidestream treatment summary for wastewater alternatives (corresponding to figures 24 and 25)

Treatment Summary	CaRRB	ANAMMOX	Ammonia Precipitation	Electrodialysis	Struvite Precipitation	Selective Adsorption
Effluent Nitrogen	11.48	11.4	12.03	12.03	14.32	14.84
% Sidestream TN Removal	23%	21%	20%	20%	60%	60%
Effluent Phosphorus	2.19	1.94	1.95	1.9	0.83	0.81
% Sidestream TP Removal	0%	0%	0%	0%	82%	82%

The demonstrated technologies all produce effluent water quality above (lower total nitrogen or total phosphorus) that of existing DWRF discharges. For both nitrogen and phosphorus, the cost-efficiencies of the wastewater sidestream technologies were both more cost-efficient and provided more removal of the target nutrient than the stormwater alternatives considered (Figures 24 and 25).

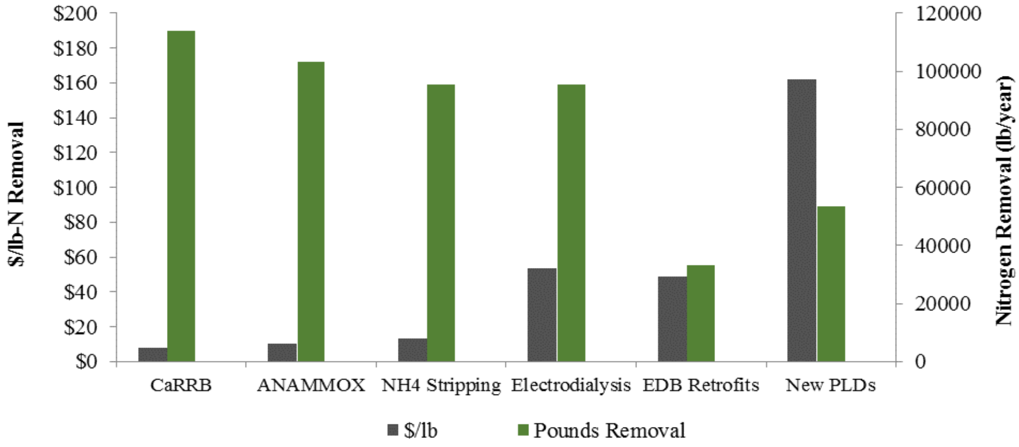

Figure 24: Nitrogen removal comparisons between wastewater and stormwater

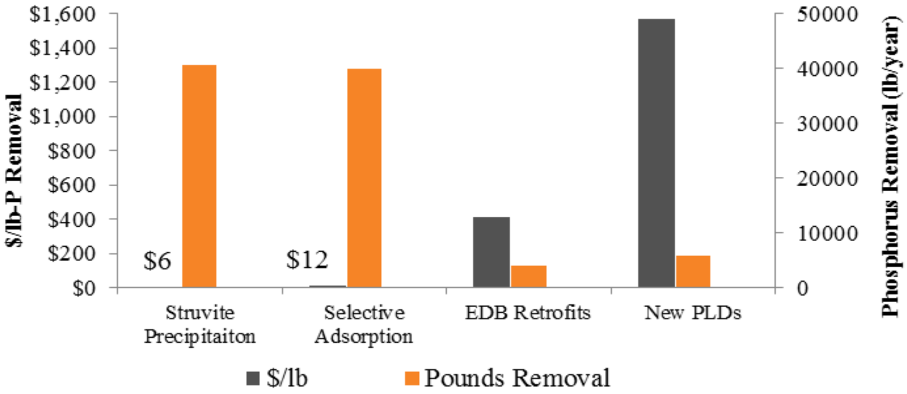

Figure 25: Phosphorus removal comparisons between wastewater and stormwater

Phosphorus concentrations required by Regulation 85 were only met in the scenarios realizing 90% sidestream phosphorus removal in the selective adsorption and struvite precipitation alternatives. Both of these resulted in effluent TP concentrations less than 1 mg/L (Figure 26). The Regulation 85 standard of 15 mg/l total nitrogen was met in each wastewater scenario; a result that is unsurprising given that the current system achieves between 7 and 12mg/l with only BNR. It should be noted that only the struvite precipitation scenario, at the highest removal efficiency, resulted in both nitrogen and phosphorus concentrations below both current DWRF water quality and Regulation 85 standards. Not all sidestream treatment alternatives result in effluent quality better than the current average, however. As can be seen by CaRRB and ANAMOX results, sidestream treatment can interfere with the performance of the mainstream treatment.

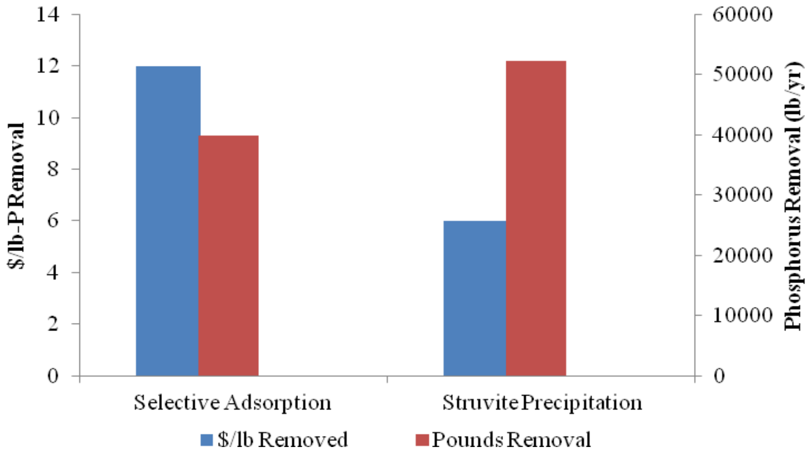

Figure 26: Phosphorus removal comparisons for scenarios meeting Regulation 85 standards

Among the wastewater side-stream scenarios, the technology achieving the most nitrogen removal is CaRRB. This is also the technology that is the most cost-effective of all sidestream processes targeting nitrogen. Struvite precipitation was found to be the most effective technology for removing phosphorus, and has both a higher cost-efficiency and mass-removal rate compared to selective adsorption. None of the nitrogen-targeting sidestream treatments were effective at removing phosphorus. Similarly, selective adsorption does not remove any additional nitrogen. Therefore, the most cost-effective technology that considers both nitrogen and phosphorus removal is struvite precipitation.

In all comparisons, wastewater control scenarios are capable of achieving larger amounts of nutrient removal than stormwater scenarios for nitrogen and phosphorus. Despite the lower potential removal for retrofitting EDBs, this alternative has a cost-efficiency similar to that of electrodialysis. Stormwater scenario 2 involves constructing numerous PLDs and therefore is heavily capital-intensive and has a total cost on the order of those seen by wastewater solutions. Although the total removal of nitrogen is between 30 and 100 percent of that achieved by sidestream technologies, the removal of phosphorus is only 10 percent of that seen in the physicochemical processes. For this reason stormwater controls are not found to be a viable option for cost-effective phosphorus removal.

5.2 Financial Impact to Rate Payers

Capital and operational expenses were normalized using the Fort Collins population to determine the projected impacts to consumer rates and city budget (Figure 27). The average annual fee for wastewater, assuming 70gpcd generation, is $918 per household. Commonly cited rates of water generation range between 50 and 100 gpcd; a value of 70gpcd was chosen as a representative but conservative estimate for rate increases (*EPA*, 2011). The corresponding average stormwater charge is approximately $420. In Figure 27, the capital costs are not calculated per year, but are assumed to be a cost incurred at one time. Maintenance costs, however, are shown as per-capita-year over the planning horizon.

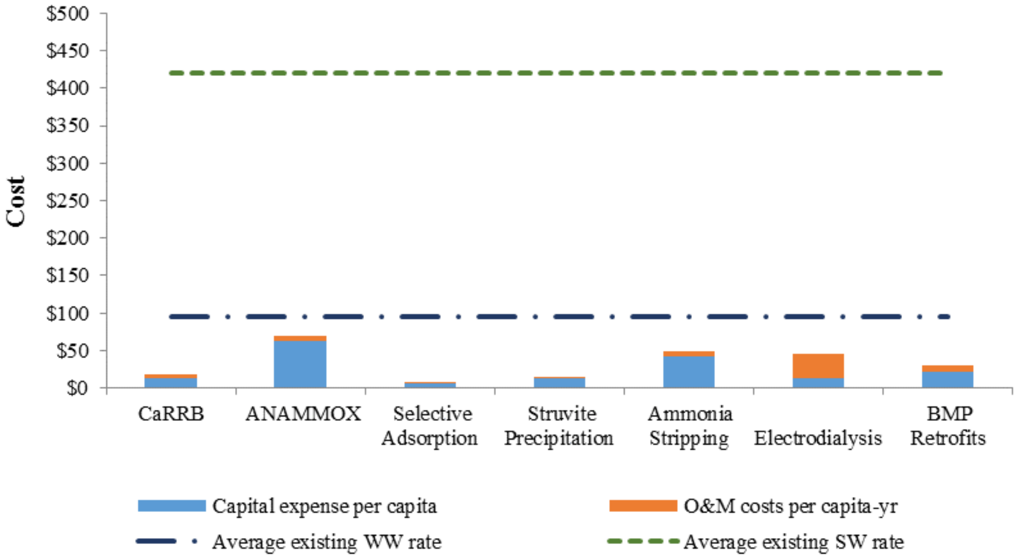

Figure 27: Capital, operational, and total annual costs per capita, compared with current utility rates

In Figure 28, the resulting potential increase to consumer rates are calculated both per thousand gallons – as wastewater fees are normally determined – and per capita-month. As before, an average per capita wastewater generation of 70 gallons per day is assumed. Stormwater maintenance costs (per capita-month) are $0.78 and $5.03 over the 20-year planning horizon for alternatives one and two, respectively.

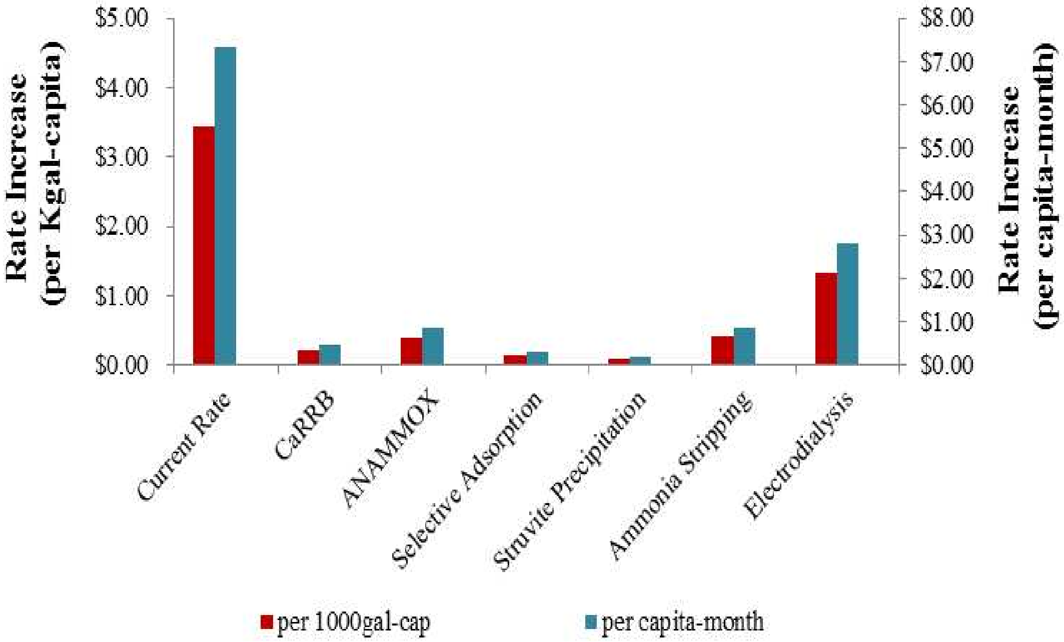

Figure 28: Operation and maintenance (variable) rate increases for alternative technologies

All wastewater technologies shown have rate increases between $0.09 and $1.32 per thousand gallons used. While retrofitting of EDBs would result in an average monthly rate increase comparable to the wastewater scenarios, at $2.62 per month per household, the increase resulting from implementing bioretention basins is more than 14 times this amount.

5.3 Multi-Criterion Decision Analysis Results

Eight MCDA analyses were conducted, encompassing each combination of relative importance factors and sub-criteria rankings for both the WAM and PROMETHEE methods. The rankings of each criterion remained constant through the various model analyses and can be used to compare specific criteria among the alternatives (Figure 29). However, the weighting of the criteria affected the resulting alternative scores.

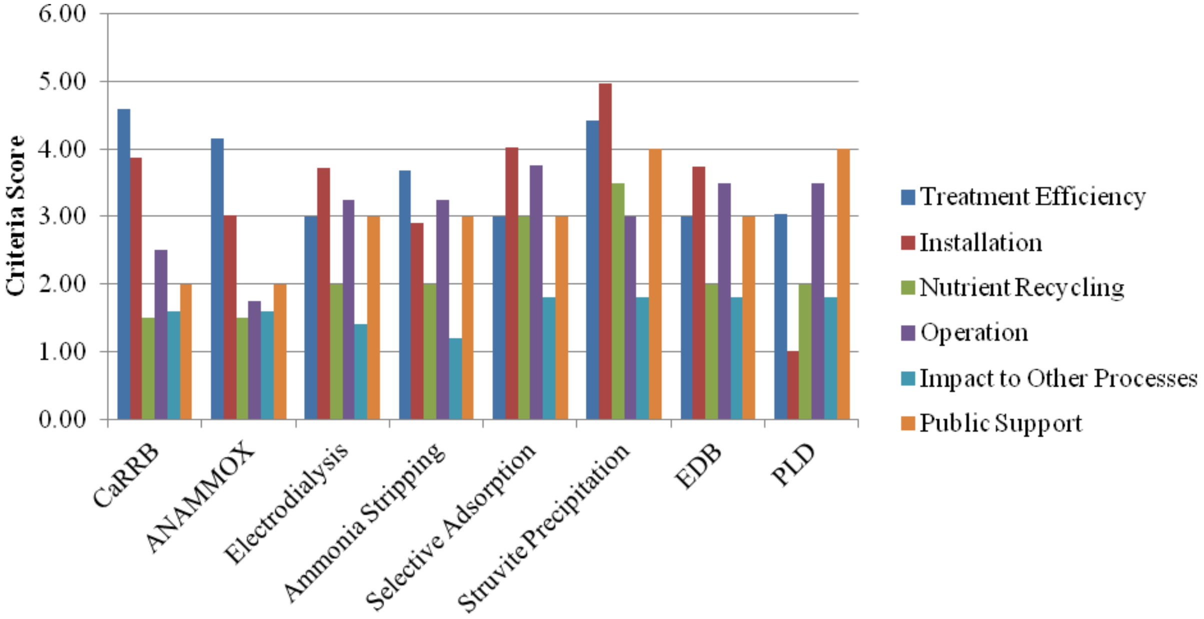

Figure 29: Criteria Scores (5 is best) for Nutrient Removal Alternatives

In the MCDA analysis, the alternatives are assigned scores dependent upon comparisons with the other alternatives, not on an absolute basis. Since only four of the alternatives allow nutrient recycling, for instance, those that do not still have a positive (and equal) score for this criterion. For all stakeholder weighting options, struvite precipitation was ranked as the best alternative in the majority of model runs and ANAMMOX and bioretention cells were ranked least favorable (Figure 30). The scores of other alternatives were, in general, more variable.

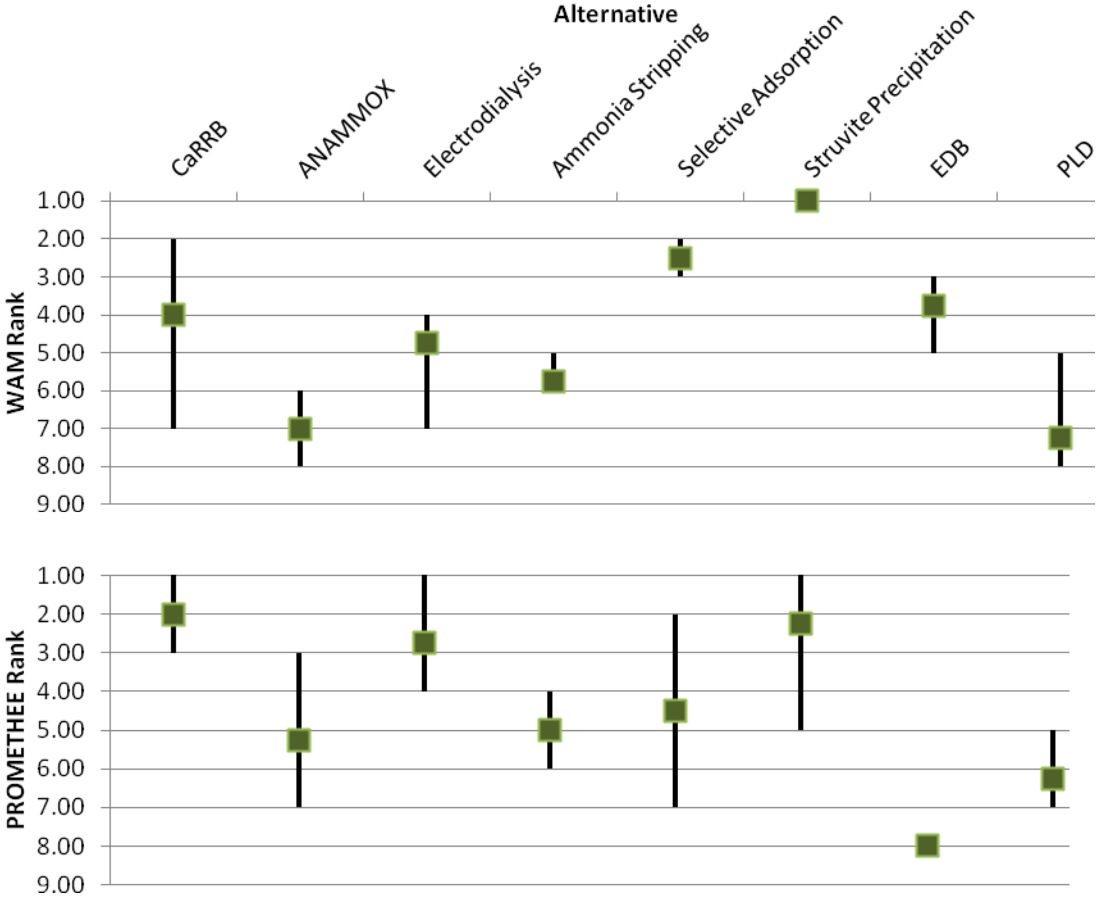

Figure 30: Summary of Scores for Alternatives from WAM and PROMETHEE MCDA Methods

6.0 DISCUSSION

Currently, the Modified Bardenpho process removes nitrogen and phosphorus at approximately the optimal point along the treatment curve (Figure 16), and achieves nitrogen removal to the highest degree practicable by a biological treatment process for DWRF. The modeled existing wastewater nutrient discharge is approximately 14.9 mg/L total nitrogen; these concentrations reasonably predict the reported values of 15.9 ± 2.4 mg/L total nitrogen. The modeled phosphorus value of 2.0 mg/L total phosphorus over-predicts the observed value of 1.1 mg/L, but was accepted as it was used for comparative purposes between other wastewater treatment scenarios. Modeled biological phosphorus removal predictions show a larger percent deviation from DWRF reported values, and are less responsive to biological sidestream treatment processes than nitrogen. This is likely due to the nature of the alternative biological processes, which target nitrogen as the removed pollutant; phosphorus concentrations from ammonia stripping/electrodialysis simulations similarly differ by less than 0.1 mg/L TP. Wastewater treatment for all modeled technologies show a potential nutrient removal effectiveness between 40% and 90% of influent wastewater concentrations; specific values are affected by the selected loading of chemical precipitants, recycle rates, reactor volumes, and other process variables. Comparing the existing effluent nutrient concentrations with the simulated wastewater effluent, additional removal efficiencies ranged between 7.5% and 30% for total nitrogen and between 45% and 78% for total phosphorus. Stormwater control measures all suggested potential removal efficiencies between 20% and 35%, measured as the decrease from existing annual nutrient loads.

For phosphorus, physicochemical processes are predicted by BioWin to achieve the most removal and provide the greater cost-to- mass removal benefits. However, biological processes remove more nitrogen and have a greater cost efficiency than either nitrogen-targeting physicochemical process considered. Among technologies that target phosphorus, struvite precipitation is the most efficient. Higher pollutant concentrations in wastewater are required by all of the alternative processes analyzed, for which reason they are modeled as sidestream treatments. The cost-effectiveness of these technologies is benefitted from such concentrated waste streams. All sidestream wastewater treatment technologies that were analyzed include considerable construction and maintenance costs compared with the no-change scenario of operating Modified Bardenpho. The current process, including all wastewater treatment, is calculated to incur a 20-year cost of about $140 million (*Fort Collins Spending Transparency, 2015*). Adding a sidestream treatment will cost between $2 and $105 million, not including mainstream treatment, over a 20-year planning horizon depending on the technology used. Physicochemical sidestream processes allow for nitrogen and/or phosphorus recovery, revenue that can partially off-set operational costs. Struvite precipitation allows for the largest percent return, with a potential revenue of between 28% and 67% operating costs; pellets produced by this process are ideally suited for fertilizer as they contain all the major nutrients required for plant growth. A return of between 0.2% and 10% operating costs were predicted to be possible from ammonia stripping, electrodialysis, and selective adsorption.

Both stormwater treatments considered in this analysis achieve a lower total mass removal and cost efficiency of both nitrogen and phosphorus than the sidestream technologies, particularly those employing physicochemical processes. However, the cost-efficiencies of the two alternatives, as compared with each other and per-nutrient, vary greatly. Differences between the two scenarios become more apparent when comparing the treatment of phosphorus; retrofitting existing flood control basins achieves approximately 30 percent less phosphorus removal than implementing thousands of new bioretention basins. However, retrofitting EDBs incurs a unit treatment cost approximately a quarter of that of implementing PLDs. The retrofitting EDBs is most favorable in comparison with wastewater sidestream treatments, and despite removing considerably less nutrient has a cost-efficiency for nitrogen removal similar to that of electrodialysis and a cost-efficiency for phosphorus removal much higher than those of all wastewater scenarios. Therefore, retrofitting BMPs may be a feasible option to reduce pollutants while maintaining a minimal economic impact compared to the construction of new BMPs, but it is not practical when compared to the efficiency of wastewater sidestream technologies.

Stormwater controls remove both nitrogen and phosphorus, whereas wastewater sidestream technologies generally target only one of these nutrients. The simultaneous removal is not clear from nutrient-to-nutrient comparisons, so the total benefit of stormwater pollutant removal is higher than implied by Figures 24 and 25. This was, however, accounted for in the MCDA analysis. Similarly, the determination of per-capita utility increases was performed such that a direct comparison could be made between stormwater and wastewater costs, but which result in over-emphasizing stormwater rates felt by individual property owners. Stormwater rates vary according to lot size and its land use. To simplify consumer-rate calculations, a per-person charge was determined without distinguishing between classes of home-ownership. The result for scenario 2 is a base-charge 75 times greater than those for the wastewater scenarios. In reality, the capital costs of building bioretention basin in residential or commercial areas would likely be included in the property values on which they are located. A bioretention basin near a residential lot, for instance, will cause the price of the unit to increase by a percentage which is small compared to the entirety.

In addition, the effectiveness of nutrient removal by implementing PLDs may be increased in upcoming years. The infiltration and reduction of stormwater runoff is a major mechanism by which bioretention basins remove all pollutants, including nutrients, from urban discharge (Liu, Sample, Bell, & Guan, 2014). In this study, an average volume reduction of 30 percent was assumed and obtained from BMP studies within Fort Collins by the CSU Stormwater Center the previous summer. However, a new bioretention design has been implemented which raises the underdrain outlet, allowing stormwater increased time to infiltrate. This design reduces runoff 85 percent and removes a further 60 percent of nutrients. Moreover, the improvement does not rely on additional equipment or labor that will increase the cost of construction or maintenance. Although implementing bioretention was not an attractive alternative in this analysis, more efficient designs and construction methods may eventually make it an ideal alternative.

In 2012, the City of Fort Collins passed a law requiring the implementation of BMPs in all new developments. These must treat, at a minimum, the water quality capture volume (WQCV). This requirement was not included in this analysis. Such considerations would assume in the second stormwater scenario that a portion of the currently undeveloped areas would treat the WQCV independent of the City's efforts. While this would have the same total effluent effect, it would decrease the nutrient removal amounts responsible by utility efforts. In other words, the assumption affects expense calculations (costs would decrease) and potential for effective intra-municipality nutrient trading (less credit would be gained). Despite this possibility of changed outcomes, the statue only effects new developments while the stormwater scenario considered here places BMPs in *all* areas not currently treated. Open areas account for less than five percent of Fort Collins land use, so the discrepancy is not estimated to be major.

Regulation 85 includes an allowance for nutrient trading to achieve a community-based coordination to reduce overall pollutant loads to receiving waters. Such trading would take place within a bilateral-negotiation market structure, in which the participating parties must agree on the amounts to be traded, the actions by which nutrient reductions will be achieved, and the monitoring mechanisms to verify removal without coordinating structures by outside agencies. The proposed trading arrangement must then be approved by the CDPHE Water Quality Control Commission. Point-nonpoint trades are to be issued with a minimum 1:2 pollutant removal ratio; thus, every 2 units of phosphorus or nitrogen removed by stormwater would reduce wastewater effluent requirements by 1 unit of the applicable nutrient. As can be seen by Figures 24 and 25, comparing the nutrient removal potential for each alternative, the stormwater scenarios remove only a fraction of the nitrogen and phosphorus that wastewater treatments can remove. Phosphorus, which is the nutrient for which DWRF would likely benefit from trading, is removed by extended detention basins and bioretention ponds in scenarios 1 and 2 at a rate of approximately 4,000 and 6,000 lbs. per year, respectively. Taking into account the 1:2 trading ratio and the average flow rate from DWRF, the resulting increase to regulated phosphorus concentrations would be 0.23 and 0.33mg/L. Assuming a BNR-optimized effluent water quality of 1.1 mg/L TP, trading with stormwater would potentially bring discharges into compliance with Regulation 85. Although additional removal would be required by the WWTP, it could potentially be achieved by increasing phosphorus treatment through BNR. Despite the possibility, nutrient trading with stormwater would not be a cost-effective way to meet discharge standards. All sidestream technologies considered removed more nutrient and were lower in total cost than either stormwater scenario. Furthermore, the DWRF effluent phosphorus can fluctuate throughout the year and as a result of changes to the treatment process and/or influent quality. It is not uncommon to observe concentrations exceeding 2mg/L. In this case, stormwater nutrient removal would not be able to bring the treatment plant into regulatory compliance.

7.0 CONCLUSION

It was found that Fort Collins stormwater is accountable for 20,000 lbs. of phosphorus and 200,000 lbs. of nitrogen. Wastewater point sources are responsible for 30,000 lbs. and 500,000 lbs. of phosphorus and nitrogen, respectively. Although wastewater discharges are larger for both nutrients, the total annual nutrient discharges into regional surface waters from both sectors are within the same order of magnitude. Despite this, concentrated nutrient loading permits greater total and cost-effective removal within the wastewater treatment sector; this efficiency is increased through the use of sidestream technologies. These results hold whether nutrient removal is compared to the existing level of removal (so that only additional nutrient removal is considered in the analysis) or to baseline (influent) concentrations.

Physicochemical wastewater process were found to be the most efficient, in terms of both total removal and cost, for phosphorus removal. Biological processes were more efficient for nitrogen removal. All wastewater processes achieved greater nutrient removal and cost-efficiency compared to the implementation of bioretention basins. Retrofitting EDBs throughout Fort Collins would be as cost effective as electrodialysis is for nitrogen removal, but achieve much less overall mass removal. Struvite precipitation in particular was ranked highly by MCDA. Unlike many of the wastewater technologies considered, struvite precipitation removes both nitrogen and phosphorus. It is one of the two treatments associated with very low treatment variation, and also allows for considerable reduction in recycling and/or aeration requirements for the mainstream BNR.

It was demonstrated that DWRF is currently carbon-limited, reducing the efficiency of nitrification in its mainstream treatment process. A noteworthy area for additional inquiry is the cost of carbon addition as it compares with the costs predicted for the wastewater sidestream alternatives. It is probable that the cost-effectiveness of carbon addition would make it a viable and attractive alternative for DWRF. However, this would not be the case for all wastewater treatment plants, as not all are carbon limited.

Despite the relatively small potential for total nutrient removal from stormwater treatment, it may be advantageous for municipalities to consider upgrading existing flood control SCMs to ones that capture and treat the water quality control volume. This stormwater alternative was the more cost-effective of the two considered. Costs of retrofitting and constructing SCMs were high in this analysis, but in reality are flexible, as they can be directly controlled by the number and type implemented. Retrofitting extended detention basins was also shown to be the scenario that is least detrimental for consumer rates. Additional research into the value of the combined treatment, as well as market incentives for nutrient recycling from wastewater is needed to further characterize cost-effective urban nutrient removal possibilities. Any alternative to nutrient removal will pose a notable increase to consumer rates and operating budgets. However, due to the cost-efficiency and overall low cost of retrofitting EDBs, further research on the appropriateness of nutrient removal is recommended.

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APPENDIX A

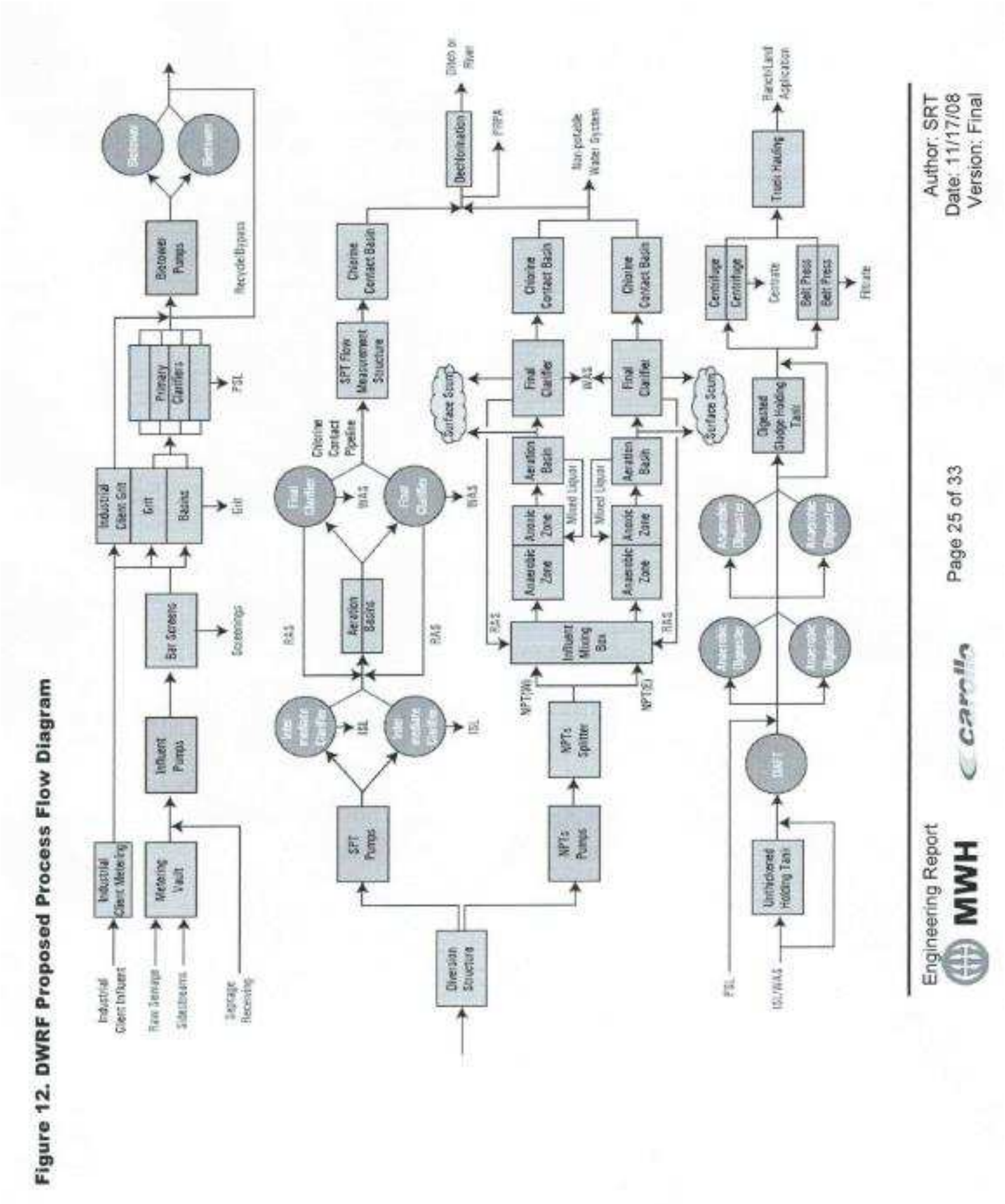

Figure A-1: Current Layout of Drake Water Reclamation Facility (Obtained from City of Fort Collins)

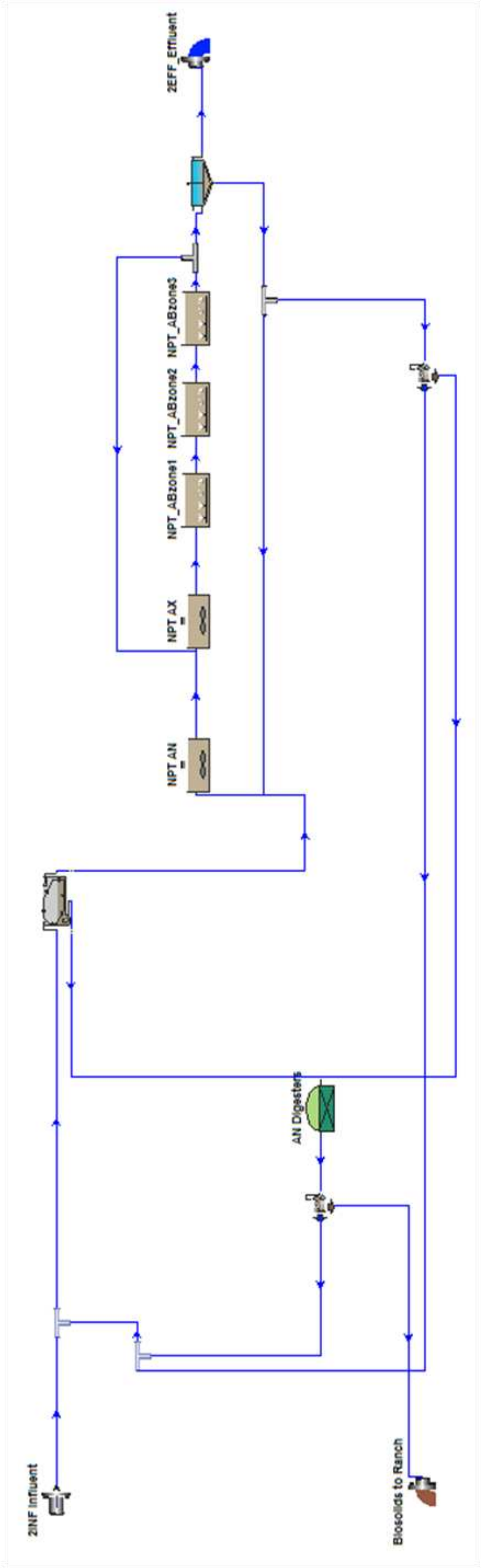

Figure A-2: BioWin Layout of Modified Bardenpho (Current Treatment)

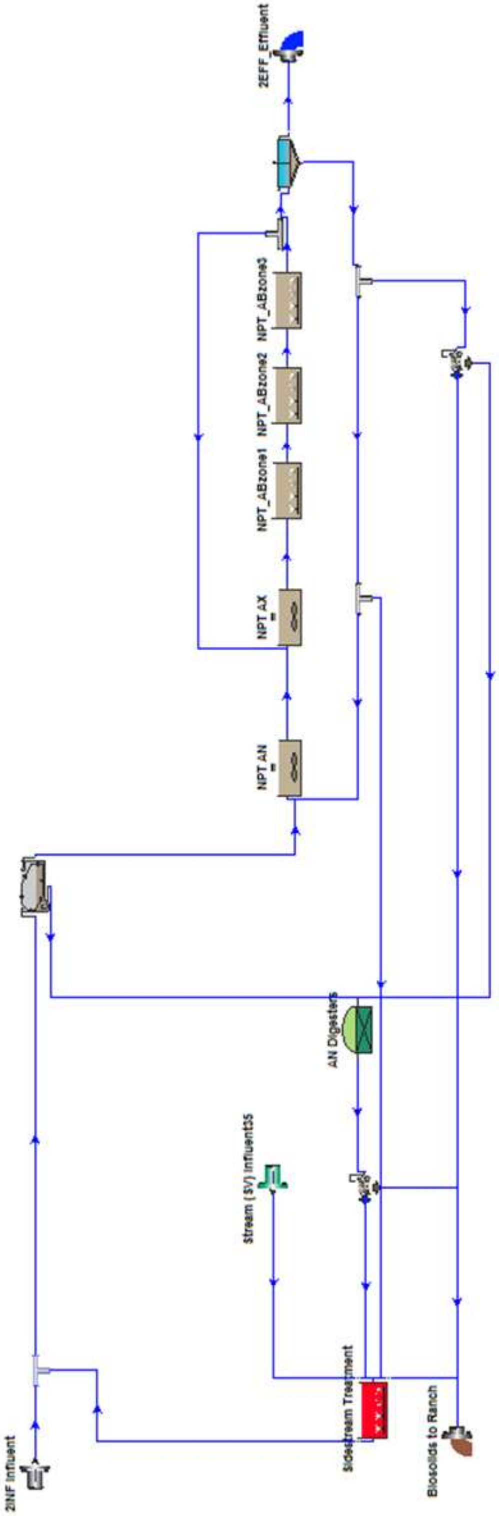

Figure A-3: BioWin Model of CaRRB Treatment Process

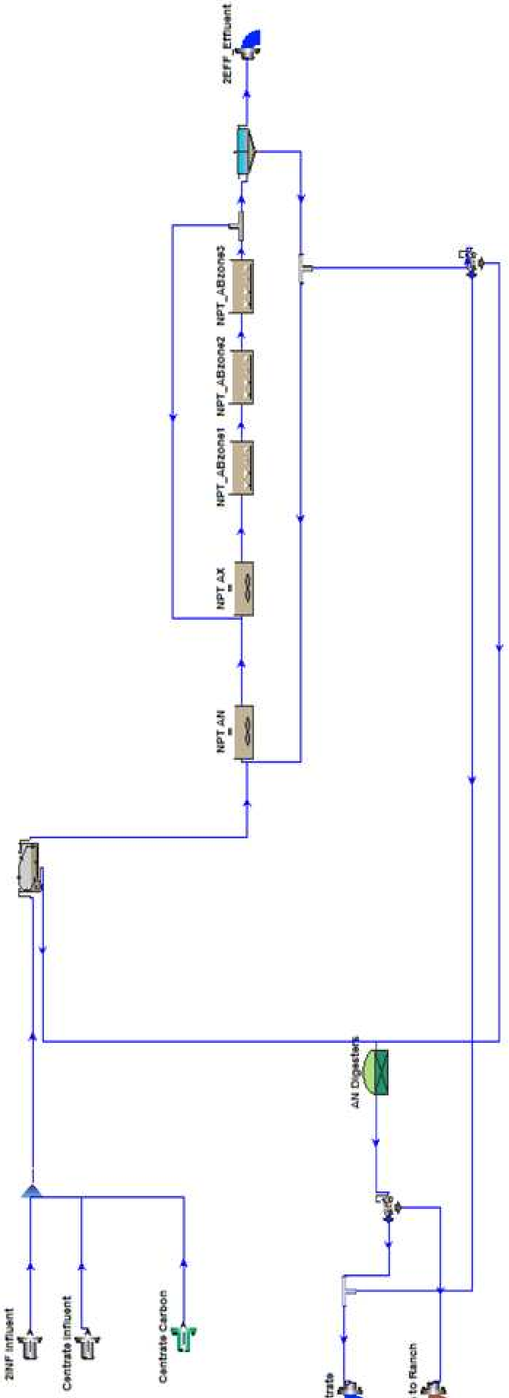

Figure A-4: BioWin Model of ANAMMOX Treatment Process

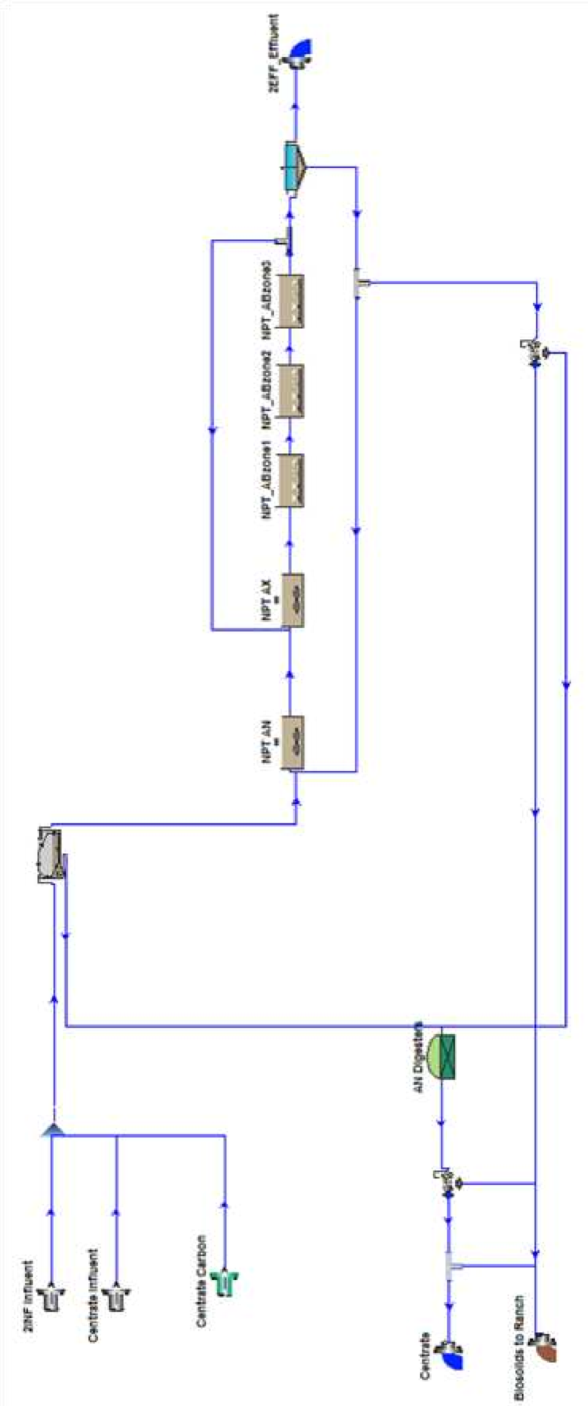

Figure A-5: BioWin Model of Struvite Precipitation Treatment Processes

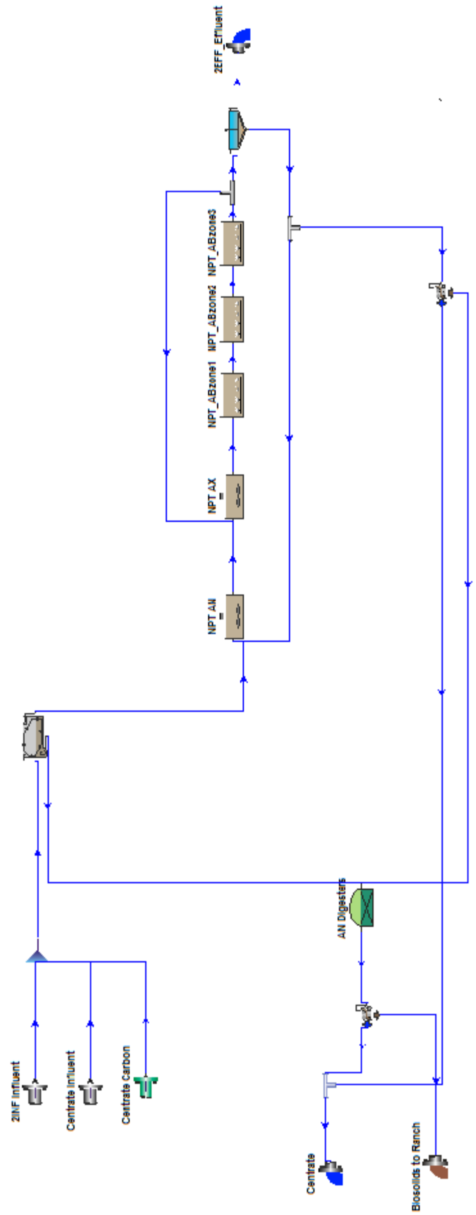

Figure A-6: BioWin Model of Selective Adsorption Treatment Processes

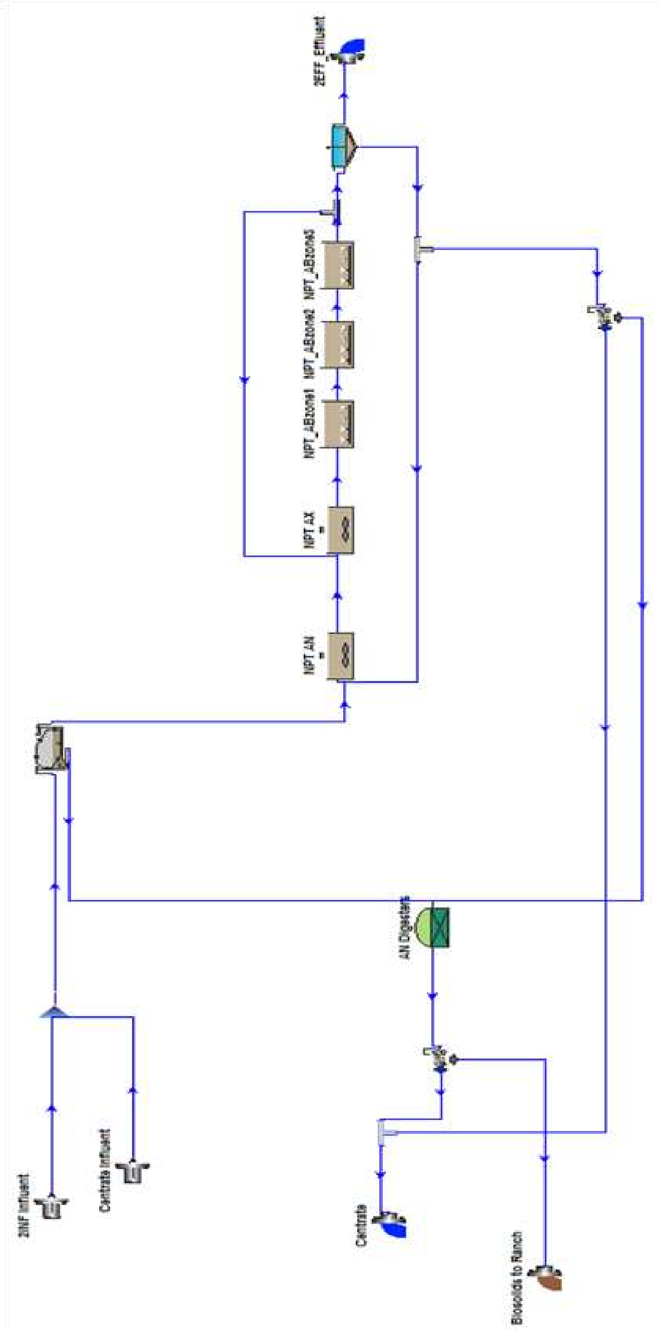

Figure A-7: BioWin Model of Ammonia Stripping and Electrodialysis Treatment Processes

APPENDIX B

Table B-1: Unadjusted unit costs for wastewater sidestream processes

Item	***Unit Cost***		***Source***
Basin Addition			
Land costs	$0.86	sqft	(Olson, Urbonas, et al., 2013)
Installation costs	45%	Equipment $	(EPA, 2009)
Tank Costs	$1	$/gal	(Hydromantis Environmental Software Solutions, n.d.)
WW Piping Costs	$0.30	$/gal	*30% of building costs*
Stripping Tank	$35,000	$/ft.	(Hydromantis Environmental Software Solutions, n.d.)
ED Membranes	$2,000,000	$/MGD	(Leitz & Boegli, 2001)
Chemical Storage and Pumps			
Storage Tank	$1,000	$/1K tank	(Colorado University, 2002)
Pumps	$7,000	$/pump	(Colorado University, 2002)
Aeration Equipment			
Blower	$448,000	$/ea.	(Hydromantis Environmental Software Solutions, n.d.)
Recycling (per year)			
Internal	$1,332	$/MGD	(Hydromantis Environmental Software Solutions, n.d.)
Activated Sludge	$1,340	$/MGD	(Hydromantis Environmental Software Solutions, n.d.)
Sidestream Recycle	$1,332	$/MGD	(Hydromantis Environmental Software Solutions, n.d.)
Aeration (per year)			
Mainstream Power	$125,560	$/mgO2/L	(Hydromantis Environmental Software Solutions, n.d.)
Sidestream Aeration	$125,560	$/mgO2/L	(Hydromantis Environmental Software Solutions, n.d.)
Chemical Addition			
Hydrated Lime (Ca(OH)2)	$0.18	$/lb.	(Hydromantis Environmental Software Solutions, n.d.)
Aluminum (Al2(SO4)3)	$0.27	$/lb.	(Hydromantis Environmental Software Solutions, n.d.)
Ferric Chloride	$0.36	$/lb.	(Hydromantis Environmental Software Solutions, n.d.)
Magnesium Chloride	$250	$/ton	(Seymour, 2009)
Anit-scalant	$11	$/gal	(Leitz & Boegli, 2001)
NiSO4 (ED)	$1	$/lb.	(Pruyn et al., 1970)
Energy			
Additional Electricity Cost	$0.04	$/kWh	(Citry of Fort Collins, 2014)
Labor Costs			
Annual Labor	$35.00	per hour	Engineering Judgment
Sidestream Mixing and Pumps (per year)			
Mixing	$23,000	$/MGD	(Colorado University, 2002)
Pumping	$20,000	$/MGD	(Colorado University, 2002)
Membrane Pumping	$1,424	$/kgal	(Pruyn et al., 1970)
Fertilizer			
Good Quality	$500	$/ton	(Seymour, 2009)
Fair Quality	$300	$/ton	Estimated from above
Poor Quality	$100	$/ton	Estimated from above
Rehabilitation			
Tanks and Piping	40%	Equipment costs	(Sewer Infrastructure Advisory Group, 2013)
Blowers and Pumps	80%	Equipment costs	(Sewer Infrastructure Advisory Group, 2013)
ED Membranes	100%	Equipment costs	(Pruyn et al., 1970)
Stripping Tank	50%	Equipment costs	Engineering Judgment

APPENDIX C

C.1 Ammonia Stripping Calculation Example

The following calculation set provides an example of the methods and equations used to determine the tank height requirement for a sidestream ammonia stripping unit set up to remove 85% of influent phosphorus.

Givens and Assumptions:

$NH^3_{water,in}$	Influent aqueous ammonia conc.	225 mg/L (2.38E-04 mol/mol H_2O)
$NH^3_{water,out}$	Effluent aqueous ammonia conc.	33.75 mg/L (3.57E-05 mol/mol H_2O)
$NH^3_{air,in}$	Influent gaseous ammonia conc.	0 mol/mol air
$NH^3_{air,out}$	Effluent aqueous ammonia conc.	1.51E-04 mol/mol air
Q	Average wastewater flow	0.4 MGD
Lq	Liquid loading rate	500 lb. H_2O/h-ft^2
G	Gas loading rate	1000 lb. air/h-ft^2
W	Stripping tower width	60 ft.
A	Area of stripping tower	139 ft^2
f	Excess factor	1.2
T_{in}	Temperature of influent	85 degrees Fahrenheit
AR	Ammonia removal efficiency	85%

Effluent Nitrogen Concentration:

$$H_c = 0.1117 * e^{0.02615\, T_{in}} \qquad 1.029 = 0.1117 * e^{0.02615\, T_{in}}$$

$$_m = L_q/\text{mmH}_2\text{O} \qquad 27.8 = 500\text{lb H}_2\text{O/h-ft}^2/29$$

$$G_m = G/\text{mmAir} \qquad 34.5 = 1000\text{lb air/h-ft}^2/18$$

$$H_C G_M/L_M = \frac{H_C G_M}{L_M} \qquad 1.227 = \frac{1.029 * 34.5}{27.8}$$

$$HTU = 9.7 \quad (From\ Figure\ 1)$$

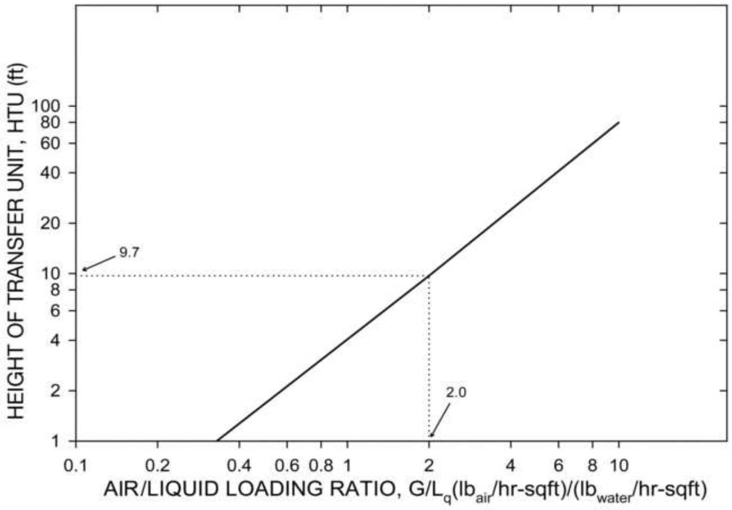

Figure C-1: Typical relationship between the height of transfer unit and the gas/liquid ratio (Huang and Shang, 2006)

$$W/HTU = \frac{W}{2} * HTU \qquad 3.09 = \frac{60ft}{2} * 9.7$$

From Figure 2:

$$\left(\frac{X}{X_1}\right)^{L/HG} = 0.702$$

$$\frac{X}{X_1} = 0.636\ at\ \ Z/HTU = 1\ and\ T = 85°F$$

$$\frac{X}{X_1} = 0.636\ at\ \ Z/HTU = 1\ and\ T = 75°F$$

$$\frac{X}{X_1} = 0.15\ when\ AR = 85\%\ and\ T = 85°F$$

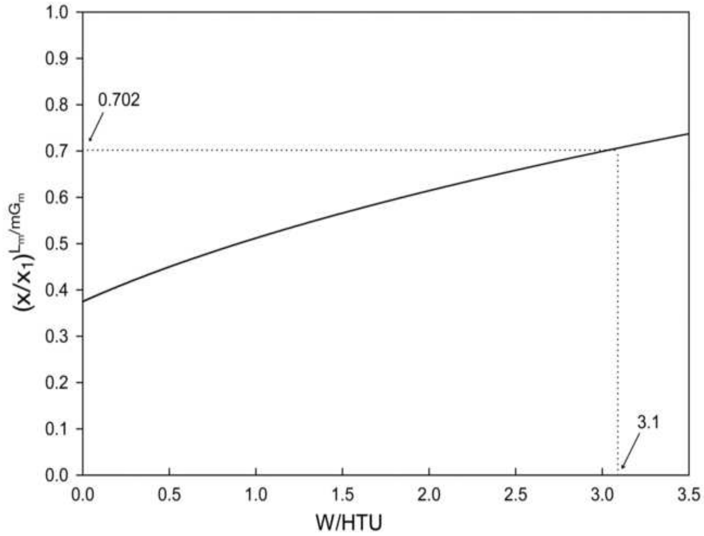

Figure C-2: Typical design relationship between W/HTU vs. $(X/X_1)^{L/mG}$ at Z/HTU=1 (Huang and Shang, 2006)

From Figure 3:

Z/HTU at (T = 85°F) = 4.5

Z/HTU at (T = 75°F) = 6

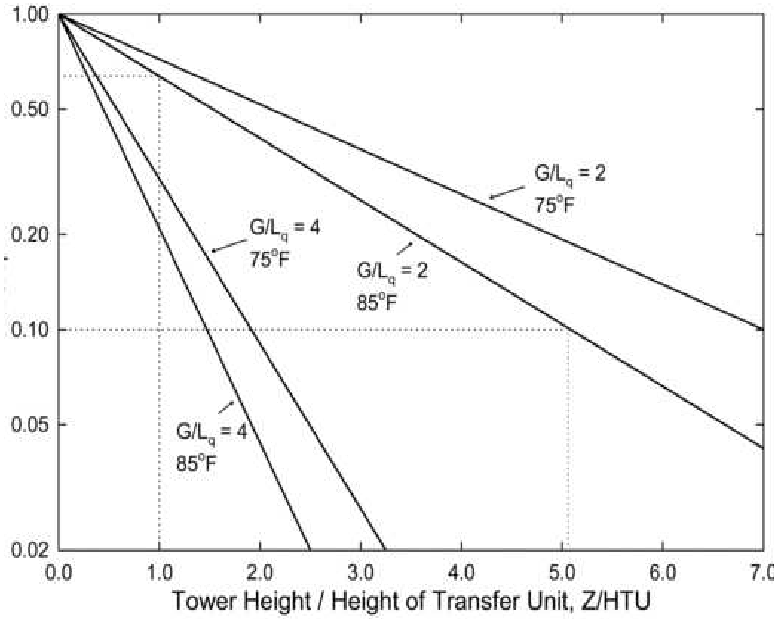

Figure 31: Example of design relationship between Z/HTU and X/X1 for ammonia stripping tower (Huang and Shang, 2006)

Tower Height Assuming T = 85°F

$Z = HTU * (Z/HTU) \quad using\ the\ largest\ of\ the\ two\ \ Z/HTU\ s$

$58.2\ ft = 9.7 * 6 \quad using\ the\ largest\ of\ the\ two\ \ Z/HTU\ s$

C.2 Electrodialysis Calculation Example

The following calculation set provides an example of the methods and equations used to determine the power required for a sidestream electrodialysis unit set up to remove 85% of influent phosphorus.

Givens and Assumptions:

Q_f	Influent flow rate	400,000 gal/day (17.5 L/s)
C^d_{inlet}	Influent nitrogen concentration	0.00363 mol/L
z	Charge of ion	(-)1
F	Faraday constant	96485 Amp/mol
N	Number of cell pairs	2
ε	Current utilization efficiency	0.9
V	Voltage	170 V

Effluent Nitrogen Concentration:

$$C^d_{outlet} = C^d_{inlet} - (C^d_{inlet} * .85)$$

$$0.00054\frac{mol}{L} = 0.00363\,mol/L - (0.00363\,mol/L * .85)$$

Current Required:

$$I = \frac{zFQ_f(C^d_{inlet} - C^d_{outlet})}{N * \varepsilon}$$

$$2{,}898\,Amps = \frac{(1) * 96{,}485\,Amp - s/mol * 17.5\ L/s * (0.00363\ mol/L - 0.00054\ mol/L)}{2 * 0.9}$$

Power Required:

$$P = I * V$$

$$492{,}612\,kWh = 2{,}898\,Amps * 170\,V$$

C.3 Struvite Precipitation

The following calculation set provides an example of the methods and equations used to determine the magnesium chloride required and sludge produced for a sidestream struvite precipitation unit set up to remove 85% of influent phosphorus.

Givens and Assumptions:

Q_{in}	Influent flow rate	400,000 gal/day
%TP	Percent TP removal:	.85
R	Number of struvite reactors	1 reactor
Mg:P	Ideal Mg:P ratio:	1.5
MW_{MgCl}	Molar weight MgCl:	59.7 g/mol
MW_{P}	Molar weight phosphorus	30.97 g/mol
P_{in}	Influent phosphorus conc.	220 mg/L (10,763 mol/day)
P_{out}	Effluent phosphorus conc.	33 mg/L (1614 mol/day)
P_{rxn}	Phosphorus removed	187 mg/L (9437 mol/day)
SP	Struvite produced per phosphorus removed	245.4 g struvite/mol P

Magnesium Chloride Required:

$$MgCl_{required} = \frac{P_{rxn} * Mg:P * MW_{MgCl}}{907184.74 \frac{g}{ton} * 365\ days}$$

$$340\ ton MgCl/year = \frac{187\ g/mol * 1.5 * 59.7 g/mol}{907184.74\ g/mol * 365\ days}$$

Struvite Produced:

$$Struvite\ Produced = \frac{P_{rxn} * SP}{907184.74\ g/mol * 365\ days}$$

$$932\ ton/year = \frac{9,437\ \ mol/day * 245.4\ g\ stuvite/mol\ P}{907184.74\ g/mol * 365\ days}$$

C.4 Selective Adsorption

The following calculation set provides an example of the methods and equations used to determine the aluminum required and sludge produced for a sidestream selective adsorption unit set up to remove 85% of influent phosphorus.

Givens and Assumptions:

Q_{in}	Influent flow rate	400,000 gal/day
%TP	Percent TP removal:	.85
R	Number of struvite reactors	1 reactor

Mg:P	Ideal Al:P ratio:	1.75 mol/mol
MW_{Al}	Molar weight Al:	26.98 g/mol
MW_P	Molar weight phosphorus	30.97 g/mol
P_{in}	Influent phosphorus conc.	220 mg/L (10,763 mol/day)
P_{out}	Effluent phosphorus conc.	33 mg/L (1614 mol/day)
P_{rxn}	Phosphorus removed	187 mg/L (9437 mol/day; 625 lb./day)
SP	Sludge produced per phosphorus removed	6.5 lb. sludge/lb. P

Magnesium Chloride Required:

$$Al_{required} = \frac{P_{rxn} * Al{:}P * MW_{Al}}{907184.74\,\frac{g}{ton} * 365\ days}$$

$$174\ ton\,Al/year = \frac{187\ g/mol * 1.75 * 26.98 g/mol}{907184.74\ \ g/mol * 365\ days}$$

Struvite Produced:

$$Sludge\ Produced = \frac{P_{rxn} * SP}{907184.74\,g/mol * 365\ days}$$

$$1{,}208{,}378\ lb/year = \frac{625\ \ lb/day * 6.5\ \ lb\ sludge/mol\ P}{907184.74\ g/mol * 365\ days}$$

APPENDIX D

D.1 Data for Social Criteria

Table D-1: Raw data for alternatives for social criteria (1)

Alternative	**Nutrient Recycling**		**Operation**	**Impact within Treatment Setting**	
Criteria Description	**Perception of progressive practices** by local communities.	Local **recycled fertilizer demand** that may be met.	**Maintenance Requirements**	**Odors and nuisances** produced during operation	**Aesthetic appeal** of treatment unit
CaRRB	Poor	None	High: the same requirements are necessary as in BNR, but are slightly more flexible due to process repetition	Low; Earthy smell	None
ANAMMOX	None, or little	None	Very High: ANAMMOX is a relatively new technology and problems may arise due to numerous process factors (nitrate toxicity, hydraulic changes, methanol toxicity, etc.) requiring knowledgeable operators	Low; Earthy smell	None
Selective Adsorption	Little to moderate, as it is a resource-intensive practice that yields usable by-product	By-product can be used as fertilizer, but is not as universal or pure quality as struvite	Low	Low; Ferric chloride is an odorless compound	None
Struvite Precipitation	Moderate, as the by-product (struvite) can be used (thus reducing waste)	Struvite is valuable for raw material in the phosphate industry or as a binding material in cements	Average: optimal Mg:P ratios must be maintained, but other wastewater characteristics do not need to stay in narrow ranges	Low; Magnesium chloride is odorless	None

Table D-2: Raw data of alternatives for social criteria (2)

Alternative	Nutrient Recycling		Operation	Impact within Treatment Setting	
Electrodialysis	Moderate, as it allows nutrient recycling	By-product can be used as fertilizer, but is not as pure quality as struvite	Low: process requires little process intervention, but membrane replacement is necessary after every 3-6 years	Anti-scalant has a characteristic odor, but is not used in large amounts in the side-stream process	None
Ammonia Stripping	Moderate to high, as nutrients may be recycled and there is relatively little chemical demand	By-product can be used as fertilizer, but is not as pure quality as struvite	Average: biological fouling and encrusting of packing material may occur due to organics and iron in the influent - these lower efficiency and require occasional replacement of packing material	Sulfuric acid, a foul-odor compound, is used. Also, fouling may occur in the packed towers and produce foul odors.	None
Retrofitted EDBs	Low to moderate	None	Average: Sediment removal (every 2 years), inlet and outlet cleaning (3 times per year), and nuisance control (4 times per year)…includes all BMPs, spread across the City	Odor, insects, and overgrowth associated with stagnant water (minimized by regular maintenance)	May develop an unaesthetic muddy layer (mitigated by a forebay and micro pool - assumed).
New PLDs	Moderate to high, as bioretention can serve multiple goals, including as scenic habitat area, a runoff reducer, and water treatment unit.	None	Average: vegetation must be replanted and debris must be removed once a year…includes all BMPs throughout the City	Odor, insects, and overgrowth are issues associated with stagnant water; these are minimized by regular maintenance.	Insect activity and decaying vegetation, or capture of trash carried by runoff; however, often implemented as an aesthetic amenity to the community

D.2 Data for Environmental Criteria

Table D-3: Raw data of alternatives for environmental criteria (1)

CRITERIA	**Treatment Efficiency**			**Nutrient Recycling**	**Operation**		**Impact within Treatment Setting**
Criteria Description	Additional **total nitrogen removal** from the existing discharge loads.	Additional **total phosphorus removal** from the existing discharge loads.	**Treatment variation** or the **probability of process upset**	Volume of **sludge reduction**	**Start-up period** to obtain nutrient removal	**Waste to Landfill**, due to increased sludge production	**Additional pollutant removal** beyond nitrogen and phosphorus
CaRRB	146,425 lbs./year	0 lbs./year	Medium; efficiency is lowered by cold and wet weather	None	Moderate; however, shortened due to the use of activated sludge from mainstream treatment	Low	None
ANAMMOX	105,546 lbs./year	0 lbs./year	Medium; ANAMMOX is a finicky process, but has been reported to be reliable once it is started	None	Long, taking months to years	Low	None
Selective Adsorption	0 lbs./year	39,873 lbs./year	Very reliable, and its efficiency is dependent only on the dosage of alum or ferric	High	Very short	High	None

Table D-4: Raw data of alternatives for environmental criteria (2)

CRITERIA	Treatment Efficiency			Nutrient Recycling	Operation		Impact within Treatment Setting
Struvite Precipitation	114,928 lbs./year	52,941 lbs./year	Very reliable as long as proper influent water quality is maintained as it is a physicochemical process	Very High	Very short	High	None
Electrodialysis	95,494 lbs./year	0 lbs./year	Membranes are easily clogged by residual organic matter	Medium	Short	Medium	None
Ammonia Stripping	95,494 lbs./year	0 lbs./year	The process is not efficient in cold weather	High	Short	Medium	None
Retrofitted EDBs	33200 lbs./year	3990 lbs./year	Nutrient removal is dependent on and influent concentrations.	None	Very little	None	Moderate removal of suspended solids, metals, oil, and grease
New PLDs	53500 lbs./year	5875 lbs./year	Nutrient removal is dependent on plant growth rate and influent concentrations.	None	Very little	None	Organics, pathogens, and pollutants absorbed to suspended particles, such as oil and metals

D.3 Data for Economic Criteria

Table D-5: Raw data of alternatives for economic criteria (1)

CRITERIA	**Treatment Efficiency**		**Costs**				**Operation**
Criteria Description	**Cost per pound nitrogen removed**	Cost per **pound phosphorus removed**	**Capital cost** incurred during construction	**Rehabilitation costs** incurred over a 20-year lifecycle	**Potential revenue** from fertilizer	**Maintenance costs** incurred during operation	**Monitoring requirement**s due to influent water sensitivity
CaRRB	$6	$N/A	$2,058,084	$929,474	$0	$420,868	Influent water quality must be kept within a narrow pH and DO range
ANAMMOX	$6	$N/A	$4,550,351	$2,062,375	$0	$170,968	The process is sensitive to influent nitrite, pH, DO (0-0.5 mg/L), and temperature (30-38 C). All must be maintained within narrow ranges.
Selective Adsorption	N/A	$12	$828,419	$282,804	$7,233	$268,393	Selective adsorption is not highly reliant on influent water quality; however, monitoring would be required to ensure adequate treatment efficiency

Table D-6: Raw data of alternatives for economic criteria (2)

CRITERIA	Treatment Efficiency		Installation				Operation
Struvite Precipitation	$49	$2	$2,062,197	$931,095	$220,421	$177,877	A molar ratio of (Mg:P:NH4 of 1:1:1), requiring stringent influent monitoring of this ratio, the pH, and DO concentration
Electrodialysis	$54	N/A	$1,921,193	$1,761,280	$68,887	$3,010,771	Electrodialysis is not highly reliant on influent water quality; however, monitoring would be required to ensure adequate treatment efficiency
Ammonia Stripping	$13	N/A	$6,202,457	$4,016,420	$68,882	$462,165	Influent water must have a higher pH (between 10.5 and 11.5) and temperature
Retrofitted EDBs	$49	$412	$3,540,729	$0	$0	$28,199,589	No monitoring requirements would be necessary, except to identify maintenance issues
New PLDs	$162	$1,566	$261,887,831	$66,707,950	$0	$114,242,107	No monitoring requirements would be necessary, except to identify maintenance issues