DISSERTATION

A CROSS SECTOR EVALUATION COMPARING NUTRIENT REMOVAL STRATEGIES IN URBAN WATER SYSTEMS

Submitted by

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ABSTRACT

A CROSS SECTOR EVALUATION COMPARING NUTRIENT REMOVAL STRATEGIES IN URBAN WATER SYSTEMS

Water supply management and reduction of nutrient pollution from urban water systems are two of the most important issues facing utility managers today. To better protect water supplies, many states have or are establishing total nitrogen (TN) and/or total phosphorous (TP) loading restrictions from urban water systems. Traditionally, these targets are met by wastewater treatment facility (WWTF) improvements, but stringent regulations can make this challenging and costly. As regulations increase it may be necessary or more cost effective to consider additional options for nutrient removal from urban water systems including water management practices or stormwater control measures (SCMs). There are a wide range treatment approaches that can be considered at a WWTF for improving nutrient removal but evaluating these scenarios can be challenging and is traditionally accomplished via mechanistic models specific to individual WWTFs requiring process expertise and a rigorous sampling and analysis program. Water management practices are traditionally considered for water supply improvement, however there is little research to characterize the impact on water quality. There is a need for additional research and tools that facilitate estimating effectiveness of various nutrient removal technologies and consider cross sector strategies and tradeoffs between adoption of practices.

To understand the impacts of water management practices, the impact of indoor conservation, source separation, and graywater and effluent reuse on WWTF influent and effluent and downstream water quality was characterized identifying which practices can potentially help meet nutrient reduction targets. For WWTF technologies, previously calibrated and validated mechanistic models were used to develop a simplified empirical model to more easily estimate and compare the effectiveness of various WWTF technologies as a function of influent wastewater quality. The findings from the water management practice evaluation and WWTF treatment comparison provided the framework for conducting an urban water systems evaluation by using the developed empirical models combined with the benefit of stormwater control measures (SCMs) characterized via the Simple Method to evaluate a multitude of

strategies for meeting nutrient removal targets in the urban water system. Lastly, this research considered the impacts on biosolids management with the increase of liquid stream removal at the WWTF.

The research identified source separation and effluent reuse as frequent part of effective nutrient removal strategies and part of an optimal nutrient removal strategy, and even necessary under stringent nutrient requirements. In terms of wastewater treatment, the benefit of adopting more advanced wastewater treatment processes will be most beneficial in carbon limited WWTFs, and negligible when there is adequate carbon for biological nitrogen and phosphorous removal. This includes sophisticated processes like nitrite shunt and 5-Stage Bardenpho and sidestream processes like struvite precipitation and ammonia stripping. While improvements to WWTF are likely with adoption of stringent nutrient regulations a multi objective optimization identified water management practices and SCMs to be part of all non-dominated nutrient removal strategies. As nutrient requirements become more stringent, the options for WWTFs in terms of processes are limited and frequently a combination of water management practices and SCMs is necessary. This was demonstrated via a systems analysis of cost-effective nutrient removal solutions in urban water systems that can be easily applied to other urban systems because of the empirical models developed with this research. These tools are necessary to help utility managers identify optimal nutrient removal strategies. As utilities invest in improvements to WWTF operations, there may also be notable impacts on biosolids management, primarily in terms of phosphorous, which may limit land application rates resulting in additional cost or disposal of biosolids that historically have been beneficially used in agriculture. These impacts must also be considered by utility managers when considering optimal nutrient removal strategies from urban water systems.

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1.0 INTRODUCTION

1.1 Background and Motivation

The US Environmental Protection Agency (USEPA) has recognized nutrient pollution, caused by excess nitrogen and phosphorous, as “one of America’s most widespread, costly and challenging environmental problems” (USEPA, 2016). Nutrient pollution originates from a variety of activities both point and non-point. Non-point source pollution includes stormwater runoff, fertilizer runoff, animal waste, soil erosion and atmospheric deposition. Point source pollution is primarily from urban environments with wastewater treatment facilities (WWTFs) being one of the principal sources and therefore a primary location for implementation of nutrient management practices (WERF, 2010).

Nitrogen pollution primarily occurs in the form of nitrate/nitrite (NO_3/NO_3), ammonia (NH_3) or organic nitrogen (O-N). Phosphorous pollution primarily occurs from orthophosphate (PO_4) while other forms of phosphorous are not typically as readily available (Sedlak, 1991). The presence of excess nitrogen and phosphorous, termed eutrophication, may accelerate the growth of photosynthetic algae and/or protozoans and result in a corresponding decrease in dissolved oxygen. The aquatic implications can have many adverse effects including mass mortalities (commonly referred to as fish kills), human illness or death, marine mammal illness or death, alterations to marine habitat, and alterations to the food web-dynamics (Anderson et al., 2002). While eutrophication is one of the primary indicators of the potential formation of harmful algal blooms (HAB), there are many other factors that will impact the magnitude and algal community including nitrogen to phosphorous ratios, temperature, geographic region, native aquatic species and water body type for example coastal, estuaries, lakes, streams (Anderson et al., 2002). There are many potential issues that arise from nutrient pollution that include environmental, economic and human health implications.

Given this issue, the USEPA is encouraging states to develop nutrient criteria management plans and many states are adopting nutrient regulations which are typically focused on WWTF. For example, Colorado has implemented nutrient regulations for annual median total inorganic nitrogen (TIN) of 15 mg N/L and median total phosphorous (TP) of 1 mg/L for existing facilities (CDPHE, 2012). Wisconsin has established statewide WWTF effluent TP limits of 1 mg/L (WDNR, 2011). North Carolina has implemented basin specific load based regulations requiring facilities to reduce annual TN and TP discharge loads by as much as 40% and 77% (NCDENQa, 2016) respectively for nutrient sensitive waters, resulting in the need for some WWTF to treat wastewater to 5.5 mg/L TN and 0.5 mg/L TP (NCDENQb, 2016) These stringent regulations could have substantial operational and capital cost implications to WWTF (Daigger et al., 2014), rendering even slight effluent concentration changes important.

Many WWTF provide some level of nitrogen removal but will range in terms of phosphorous removal. Conventional treatment plant operations may achieve effluent TN and TP concentrations between 5-30 mg/L and 4-10 mg/L respectively (Metcalf and Eddy, 2003). Nitrogen removal is traditionally achieved through a combination of nitrification and denitrification where the ammonia is oxidized to nitrate and then the nitrate is reduced to nitrogen gas (Grady et al., 1999). This is accomplished through a combination of aerobic and anoxic basins with internal recirculation lines to improve the nitrification and denitrification performance. Improving nitrogen removal can be challenging and require supplemental carbon, longer anoxic hydraulic retention times (HRT), higher return mix liquor (RML) rates, and/or longer solids retention time (SRT) to improve the treatment performance (Grady et al., 1999). Traditional treatment operations provide little phosphorous removal except for the phosphorous removal associated with cellular growth (Reynolds and Richards, 2008). Facilities that are required to achieve phosphorous removal often utilize biological phosphorous and/or chemical addition of ferric chloride to precipitate phosphate, but this can require substantial additional operational cost.

Given the challenges and technological limits in meeting effluent requirements at WWTF, there is growing interest in opportunities that reduce the influent WWTF contaminant loading, primarily source separation (urine diversion), where highly concentrated waste streams are either treated on-site or hauled off site and not discharged to the sanitary sewer. Urine can account for 75-80% of the nitrogen and 50-55% of the phosphorous mass loading in wastewater (Fewless et al., 2011), therefore source separation will reduce the flow associated with urine flushing and decrease nutrient loading to WWTF. Additionally, the diverted flows could provide a beneficial use as a fertilizer. While source separation reduces contaminant loading and indoor water use, there are notable social, economic, and infrastructure challenges to the wide spread adoption of source separation (Fewless et al., 2011).

In addition to nutrient pollution, many utilities also face water supply management issues to accommodate population growth, promote development, and/or to react to drought conditions. These water management practices emphasize reducing water use (conservation) or stretching a single supply to meet multiple uses (reuse). Conservation efforts may include both consumptive uses (irrigation) and non-consumptive (indoor), and reuse applications may consider adoption of graywater reuse or wastewater treatment plant effluent reuse. While these water management practices are considered for their impacts to water supply, little research has been done to quantify the corresponding impacts, good or bad, to downstream water quality. Conservation practices will reduce indoor water use but will not change the contaminant load resulting in more concentrated waste streams. Depending on application, reuse practices may impact both the downstream flow volume and contaminant load to the wastewater treatment facility. These relationships are often qualitatively discussed but are not frequently quantified in terms of impact on contaminant concentrations, loads to the WWTF, and impacts to the treatment plant performance.

With the widespread issue of nutrient pollution, increase in regulations and limitations in nutrient reductions from WWTF alone, there is a need for research that evaluate and identify effective nutrient removal strategies considering traditional and non-traditional approaches that are not limited to WWTF improvements. Traditionally, WWTFs are the first regulated and will utilize mechanistic models to evaluate process improvements for reducing effluent nutrient concentrations (WERF, 2003). While these models are effective, they require expertise and a significant amount of data inputs to develop and evaluate nutrient removal strategies. Additionally, in the urban sector water management practices or stormwater control measures may be necessary or more economical in reducing nutrient pollution. However, there is limited research currently available to evaluate and identify effective nutrient removal strategies considering cross sector adoption of practices.

1.2 Research Objective

This research aims to expand on the existing body of knowledge on strategies and approaches for nutrient management in evaluating and developing tools to more easily compare and identify effective nutrient removal strategies. The research was conducted with the following objectives:

Objective 1. Quantify the impacts of water management practices on influent wastewater quality, wastewater treatment efficiency and downstream nutrient loading to determine if these practices can be viable nutrient load reduction strategies.

Objective 2. Compare the nutrient removal effectiveness of various WWTF processes based on key characteristics including: influent wastewater quality, process configuration, and operational variables.

Objective 3. Investigate most effective nutrient removal strategies considering adoption of water management practices, WWTF process configurations, and stormwater control measures\

Objective 4. Explore tradeoffs between cost and efficiency for achieving nitrogen and phosphorous load reduction.

Objective 5. Assess the impacts of increasing nutrient removal at WWTFs on biosolids nutrient concentration and systems level impacts of land application of those biosolids.

1.3 Dissertation Structure

This dissertation is structured into four papers that evaluate different strategies and approaches in addressing nutrient pollution. Each paper has been prepared for peer reviewed journal publication or dissemination as a white paper as follows:

- Impact of Water Conservation and Reuse on Water Systems and Receiving Water Body Quality
- Development of Generalized Empirical Models for Comparing Effectiveness of Wastewater Nutrient Removal Technologies
- Assessing Cost-Effective Nutrient Removal Solutions in the Urban Water System
- Considering the Impacts of Nutrient Regulations to Biosolids Management – A Case Study on Potential Implications of Nutrient Regulations on Biosolids Management in Arid West

These chapters (Chapters 2-5) quantify and compare the impact of technologies and strategies for controlling nutrient pollution from the Urban Sector. Chapter 2 has been published as a research article in Environmental Engineering Science and considers the impact of water management practices as nutrient removal strategies quantifying the impact on nutrient load to the WWTF, nutrient removal at the WWTF, and downstream impacts. Chapter 3 has been drafted for publication as a research article and compares the effectiveness of common and innovative WWTF processes to characterize the estimate TN and TP percent removal as a function of influent wastewater quality. Chapter 4 has been drafted for publication and develops a systems approach framework for evaluating combinations of cross sector nutrient removal strategies (Water Management Strategies, WWWTF Processes and/or Stormwater Control Measures) for meeting target nutrient removal goals. Lastly, Chapter 5 has been distributed as a white paper with the Comprehensive, optimal and effective abatement of nutrients (CLEAN) center and considers the impact of increased liquid stream nutrient removal at WWTF on biosolids concentrations and solids disposal management.

In addition to the above topics, the research conducted with this dissertation has contributed to other research work including:

- *Modeled Response of Wastewater Nutrient Treatment to Indoor Water Conservation*, Anna McKenna, JoAnn Silverstein, Sybil Sharvelle, and Brock Hodgson, published in Environmental Engineering Science May, 2018
- *A Cyanobacterial Sidestream Nutrient Removal Process and its Life Cycle Implications*, Carlos Quiroz-Arita, John J. Sheehan, Nawa Raj Baral, Alexander Hughes, Graham Peers, Brock Hodgson, Sybil Sharvelle, and Thomas H. Bradley, Journal of BioEnergy Research, January, 2019

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2.0 IMPACT OF WATER CONSERVATION AND REUSE ON WATER SYSTEMS AND RECEIVING WATER BODY QUALITY[1]

Overview

Implementation of water demand reduction strategies may impact downstream water quality. These practices and impacts are particularly important for arid west regions with frequent water supply shortages. Downstream water quality impacts of indoor conservation, source separation, and graywater and effluent reuse were quantified for an arid water system where the receiving water stream is predominantly fed by snowmelt and experiences large fluctuations in flow. Estimates of indoor conservation indicate that, without wastewater treatment facility (WWTF) process modifications, conservation practices implemented during drought conditions to stretch a water supply will result in increased receiving water body nutrient concentrations (> 25% increase in total nitrogen (TN) and total phosphorous (TP)). Graywater reuse practices for toilet flushing or irrigation have negligible impacts on WWTF performance and downstream water quality even with wide adoption of the practice (100% population adoption). Conversely, adoption of source separation is estimated to notably improve influent nutrient loading which corresponded to an improvement in WWTF effluent loading. To meet potential stream standards city wide adoption of source separation would be necessary and there are likely more cost effective and feasible opportunities for improvements at the WWTF. The downstream impact of WWTF effluent reuse is largely dependent on the receiving water body and seasonality of flows, but the practice is beneficial during mid-range flow and dry conditions when effluent reuse is most valuable as an additional supply (> 25% decrease in TN and TP). However, water reuse does not provide a benefit to the WWTF under concentration based permits.

[1] A version of this chapter has been published as a research article by Environmental Engineering Science with the following citation:

Impact of Water Conservation and Reuse on Water Systems and Receiving Water Body Quality
Brock Hodgson, Sybil Sharvelle, JoAnn Silverstein, and Anna McKenna
Environmental Engineering Science 2018 35:6, 545-559

2.1 Introduction

The issue of water scarcity continues to be one of the most important issues facing water utilities. With limited additional water supplies, utilities must consider alternative management practices including water conservation, graywater reuse and WWTF effluent reuse. The implementation of these practices focuses on stretching a water supply to meet more demands, but the resulting impact to water quality is often neglected. To balance the pressures of water supply and the increasingly stringent nutrient discharge standards for WWTFs, there is a need to better understand the impacts of water conservation and reuse strategies on WWTF operations and receiving water body quality.

In arid western states, like California, water conservation and use restrictions are adopted to conserve water during drought conditions (Mini et al., 2015). Even areas not suffering from drought conditions will strive to achieve water efficiency to promote development and accommodate population growth. For example, New York has historically utilized water conservation to offset costly development of new water sources (Paulsen et al., 2007).

More stringent water quality regulations are closely connected with the issue of water scarcity to protect the water quality of existing supplies. Nutrients are one of the primary contaminants of concern which may lead to eutrophication, resulting in decreased dissolved oxygen, killing native species and producing compounds toxic to humans (Smith, 2003). Nutrient removal is one of the biggest challenges facing WWTF (Reardon et al., 2013) and many states are adopting nutrient regulations for WWTFs. For example, Colorado has implemented nutrient regulations for annual median total inorganic nitrogen (TIN) of 15 mg N/L and median TP of 1 mg/L for existing facilities (CDPHE, 2012). Wisconsin has established statewide TP limits of 1 mg/L (WDNR, 2011).

The Chesapeake Bay has implemented a total maximum daily load (TMDL) approach for limiting the annual dischargers from contributions across New York, Pennsylvania, Maryland, Delaware, District of Columbia, West Virginia, and Virginia (EPA, 2017). North Carolina has also implemented basin specific load based regulations requiring facilities to reduce annual total nitrogen (TN) and total phosphorous (TP) discharge loads by as much as 40% and 77% (NCDEQa, 2016) respectively for nutrient sensitive waters, resulting in the need for some WWTFs to treat wastewater to 5.5 mg/L TN and 0.5 mg/L TP (NCDEQb, 2016). These stringent regulations may have substantial operational and cost implications to WWTFs (Daigger et al., 2014), rendering even slight effluent concentration changes caused by water management practices important.

Common practices to reduce water demand include installation of water conserving fixtures, behavioral changes for reducing water use, graywater reuse for toilet flushing and/or irrigation, and WWTF effluent reuse (DeOreo et al., 2016; WSTB, 2016; Attari, 2014; Christova-boala et al., 1996; Rockaway et al., 2011).Indoor conservation can substantially decrease water use through simple retrofits, appliance replacement and behavioral changes (DeZellar and Maier, 1980; Attari, 2014). During times of drought, these practices may be adopted at a widespread scale and incentivized by utilities. Conservation practices reduce indoor water use, but increase pollutant concentrations as pollutant load per capita is independent of water use (Daigger et al., 2009).

Graywater has been estimated to account for as much as 50% of indoor water use (Sheikh, 2009) with nutrient concentrations averaging 14 mg/L TN and 4 mg/L TP (Jokerst et al., 2011). A study on graywater reuse for toilet flushing estimated a 27% reduction in residential water demand corresponds to an increase in influent BOD and ammonia concentrations of 41 and 43%, respectively (Parkinson et al., 2005). Graywater reuse for irrigation diverts hydraulic load and associated contaminant load. However, graywater is not as highly loaded with contaminants as wastewater and the result is lower hydraulic load and increased wastewater constituent concentration.

Utilizing graywater reuse does require some investment in additional infrastructure including dual plumbing, on-site storage, and some level of on-site treatment and is typically easier in new construction. Unlike conservation, adoption of graywater reuse may be highly heterogeneous and is more difficult to adopt at a widespread scale. WWTF effluent reuse diverts a portion of the treated effluent flow for non-potable water use, thus decreasing the associated pollutant load discharged to the stream. In North Carolina, where load based regulations are implemented, WWTF may receive a credit for the nutrients diverted with effluent reuse (NCDEQb, 2016). Additionally, the TN and TP load provides a beneficial nutrient source in irrigation reuse applications to improve plant growth (Toze, 2005).

The benefits of these water management practices on improving water supply has been widely studied, and conservation approaches like graywater reuse are recognized to potentially provide notable potable water savings particularly in arid regions (WSTB, 2016). Additionally, the impacts of water conservation on sanitary collection systems has been studied to correspond to reduced flushing velocities and increase in sediment deposition (Marleni et al., 2012).

However, little work has been done to understand the impacts that water management practices have on WWTF operations and downstream water quality. Increasing influent wastewater BOD concentrations (between 25% to 40%) have been noted as a result of water conservation, while little to no observed change in per capita BOD load was observed (DeZellar and Maier, 1980). There is limited current research on impacts of conservation on influent water quality. A statistical analysis performed on multiple New York WWTFs conclude that water conservation resulted in constant or increasing influent nitrogen concentrations but lower effluent TN loads, likely from longer hydraulic residence time and longer solids residence time (Paulsen et al., 2007).

While it is well understood that indoor water conservation and graywater reuse both will result in more concentrated wastewater (WSTB, 2016; Min and Yeats, 2011), the resulting impact to WWTF operations and performance and receiving water body quality is not well understood. The impact of water conservation and graywater reuse on WWTF performance is complex because while nutrient concentrations will increase, the resulting decrease in hydraulic load increases hydraulic and solids residence time.

Understanding the impact of effluent discharge to receiving water bodies when these strategies are adopted is further complicated due to potential reduction in wastewater discharge flow and variable impacts to receiving water body flow depending on whether conserved water remains in stream. Extensive modeling of water conservation and graywater reuse impacts based on receiving water body flow conditions is needed to better understand the impacts of these practices downstream water quality. This is particularly important in arid west regions where wastewater discharge can have a large impact on receiving water body quality during low flow and drought conditions.

One other water management strategy that is considered for the positive impact on nutrient load reductions is source separation, also known as urine diversion, where urine is source separated at the toilet (Wilsenach and van Loosdrecht, 2003). Urine accounts for 75-80% of the nitrogen and 50-55% of the phosphorous mass loading in wastewater (Fewless et al., 2011), therefore source separation will reduce the flow associated with urine flushing and decrease nutrient loading to a WWTF. Source separation has been hypothesized to potentially be a more effective way of nutrient pollution management than conventional centralized treatment (Ishii and Boyer, 2015).

The separated urine is collected at smaller scales (household, multi-residential, neighborhood) and treated separately resulting in a concentrated nutrient and phosphorous product. Additional infrastructure is required to allow for source separation, collection, and treatment via physiochemical and/or biological processes (Fewless et al., 2011), and like graywater reuse adoption will be heterogeneous and costly at large scale. It is well understood that source separation of urine could potentially lead to decreased nutrient loads in wastewater by up to 80% of TN and 45% of TP (Wilsenach and van Loosdrecht, 2003). However, the downstream impact of this reduction to WWTF influent and receiving water bodies has not been well studied.

The objective of this research is to quantify the water quality impact of several water management practices on nutrient discharge concentration and load and subsequent impact to receiving water body quality. This research applies a systems approach to evaluate the effects of indoor conservation, source separation, graywater reuse and WWTF effluent reuse using influent loading estimation, BioWin™ process modeling, and analysis of receiving water body nutrient loading. Several scenarios were developed at different levels of adoption and modeled at a Colorado WWTF with biological nutrient removal for various flow conditions (e.g. low, medium and high) in receiving water bodies.

2.2 Experimental Protocols

The studied area was selected because it is reflective of many arid west water systems with limited water supply, seasonal irrigation demands, and highly variable stream flow. Given these characteristics, water management practices and water quality standards are important in stretching and protecting the water supply. The water system is composed of inter-basin and imported supply, two water treatment plants (primary and seasonal), and one WWTF (Figure 2-1).

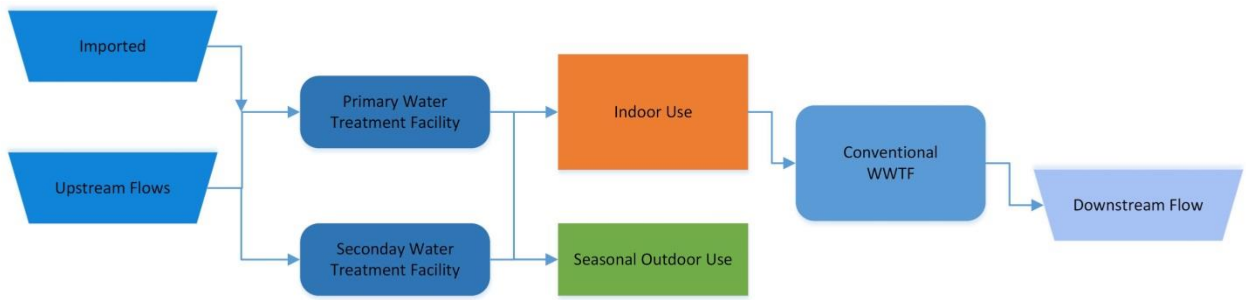

Figure 2-1. Water System Schematic

The system's dependence on imported supply, seasonal irrigation use, and utilization of a secondary water treatment facility for summer peak flows creates challenges in terms of water management and receiving water body quality impacts. There is a notable incentive in implementing water management practices that stretch the existing supply and minimize the dependency on imported flows while protecting the downstream water quality.

Scenarios were developed for each of the management practices to analyze the influent, effluent and downstream water quality. The influent scenarios included analysis of flow and concentration of 5 day biochemical oxygen demand (BOD_5), total suspended solids (TSS), TN, and TP to characterize variations in the influent water quality at the WWTF. All analysis was based on 2014 water quality data, provided by the studied WWTF, with a baseline population of 114,195 (P_S) people and an average wastewater production of 492.1 L/capita/d (d_B) (Base Scenario) representing wastewater flows from all sources (Residential, Inflow and Infiltration, and Commercial, Institutional and Industrial (CII)).

Indoor conservation was considered based on different levels of indoor water use on a per capita basis (d_{IN}). Source separation (P_{SS}) and graywater toilet and irrigation reuse (P_{GWT} and P_{GWI}) were considered at different levels of population adopting the technology. WWTF effluent reuse was considered at different levels of percent effluent reuse ($R_{\%}$). The water management scenarios are summarized in Table 2-1.

Table 2-1. Management Practice Scenarios

Scenario	Parameter Varied	# Scenarios	Range
Base (d_B)	N/A	1	130 gpcd
Indoor Conservation (d_{IN})	Per Capita Indoor Water Use	15	60-130 gpcd (0–54%)
Source Separation (P_{SS})	Adopting Population	5	0–114,195 ppl (0–100%)
Graywater Reuse for Toilet Flushing (P_{GWT})	Adopting Population	5	0–114,195 ppl (0–100%)
Graywater Reuse for Irrigation (P_{GWI})	Adopting Population	5	0–114,195 ppl (0–100%)
WWTF Effluent Reuse ($R_{\%}$)	Percent Effluent Reuse	5	0–40%

These scenarios were selected to develop a range of evenly distributed practice implementation up to extreme levels of adoption. The analysis was evaluated at a municipal scale where the influent wastewater is homogeneous and is not dependent on site-specific adoption. The baseline influent water quality is summarized in Table 2-2.

Table 2-2. Base Scenario Modeled Influent Water Quality

Influent	Value
Average Flow (m^3/s)	0.65 (14.8 MGD)
Total COD (mgCOD/L)	430
TKN (mgN/L)	33.9
Nitrate (mgN/L)	0.5
TP (mgP/L)	4.5
TSS (mgTSS/L)	176.2
Inorganic SS (mgISS/L)	12
pH	7.4
Alkalinity (mmol/L)	4.76
Calcium (mg/L)	192
Magnesium (mg/L)	30
DO (mg/L)	0

To characterize the impacts to influent flow and water quality, a mass balance approach was adopted similar to Wilsenach and van Loosdrecht, 2003. For graywater reuse, it was assumed that reuse is only from residential sources assuming collection from showers, clothes washer, bathtub, and 25% of faucet (excludes kitchen water) equating to 94.6 L/capita/d (d_{GI}) (DeOreo et al., 2016). The volume of water associated with toilet reuse was assumed to be 45.4 L/capita/d (d_T), within the typical range of 42.4 – 53.8 L/capita/d (DeOreo et al., 2016). For source separation, the reduction in influent wastewater flow was assumed to be 36.3 L/capita/d (d_{SS}) (Wilsenach and van Loosdrecht, 2003). The influent wastewater flow for each scenario was calculated based on equation 2-1.

$$\boldsymbol{Q_{IN}} = (\boldsymbol{d_B} \times \boldsymbol{P_S}) - (\Delta \boldsymbol{d_C} \times \boldsymbol{P_S}) - (\boldsymbol{d_T} \times \boldsymbol{P_{GWT}}) - (\boldsymbol{d_{GI}} \times \boldsymbol{P_{GWI}}) - (\boldsymbol{d_{SS}} \times \boldsymbol{P_{SS}}) \quad (2\text{-}1)$$

Where:

Q_{IN} = influent flow to treatment facility (volume/day)

d_B = base indoor water demand (volume/capita/day)

d_{IN} = indoor water demand (volume/capita/day)

$\Delta d_C = d_B - d_{IN}$ = change in average indoor water demand based on conservation (volume/capita/day)

d_T = toilet flush demand (volume/capita/day)

d_{GI} = graywater used for irrigation (volume/capita/day)

d_{SS} = flow reduction from source separation (volume/capita/day)

P_S = Service population (capita)

P_{GWT} = Population adopting graywater reuse for toilet flushing (capita)

P_{GWI} = Population adopting graywater reuse for irrigation (capita)

P_{SS} = Population adopting source separation (capita)

To evaluate the impact to influent water quality, source water quality characteristics were defined for influent wastewater load, graywater concentrations, and urine separation loading rates. A mass balance approach was utilized to evaluate the impacts a given scenario will have on the WWTF influent in terms of flow, BOD_5, TSS, TN and TP. The per capita wastewater load (L_{IN}) indicated in Table 2-3 was calculated based on the 2014 service area population and the influent flow and water quality (Table 2-2). Of note is that estimates of L_{IN} include load from CII flows. The calculated BOD_5 and TSS L_{IN} are comparable to design standards of 110 grams/capita/day and 100 grams/capita/day (10 State Standards, 2004).

Studies on graywater have reported highly variable water quality depending on the contributing sources with ranges of 76 – 200 mg/L BOD_5, 54 – 200 mg/L TSS, 5 – 17 mg/L TN, and 0.1 – 2 mg/L TP (Eriksson et al., 2002). In a study of the behavior of nutrients in a constructed wetland for graywater treatment, average graywater concentrations of 86.3 ± 40.3 mg/L BOD_5, 16.5 ± 7.2 mg/L TSS, 13.5 ± 8.7 mg/L TN and 4.0 ± 1.8 mg/L TP have been reported (Jokerst et al., 2011). Based on the large variability associated with graywater quality, representative concentrations of graywater (C_{GW}) were selected consistent with typical ranges and averages (Table 2-3).

Studies on the water quality of source separation report nitrogen and phosphorus loads of 6.8 – 12 grams N/cap/day and 0.63 – 1 grams P/cap/day (Vinneras, 2002; Von Munch and Winker, 2009; Wilsenach and van Loosdrecht, 2003), therefore a urine separation loading rate (L_{SS}) of 11 grams N/cap/day and 1 gram P/cap/day was selected (Table 2-3).

Table 2-3. Source Water Quality Characteristics

Source	BOD	TSS	TN	TP
Wastewater Influent Load (L_{IN}) (grams/capita/day)[1]	106.6	86.6	16.7	2.2
Graywater Concentration (C_{GW}) (mg/L)[2]	100.0	100.0	10.0	2.5
Urine Separation Loading Rate (L_{SS}) (grams/capita/day)[3]	0.0	0.0	11.0	1.0

[1]Calculated from the modeled influent concentration at the 75th Street Wastewater Treatment Facility and service population in 2014.
[2]Compiled from Eriksson et al., 2002 and Jokerst et al., 2011
[3]Compiled from Vinneras, 2002; Von Munch and Winker, 2009; Wilsenach and Loosdrecht, 2003

Based on the selected source water quality and calculated influent flow, the impact on influent wastewater quality was calculated for each scenario based on equation 2-2.

$$\boldsymbol{C_{IN}} = \frac{(\boldsymbol{L_{IN}} \times \boldsymbol{P_S}) - (\boldsymbol{C_{GW}} \times \boldsymbol{Q_{GWI}}) - (\boldsymbol{L_{SS}} \times \boldsymbol{P_{SS}})}{\boldsymbol{Q_{IN}}} \tag{2-2}$$

Where:

C_{IN} = influent concentration to treatment facility (mass/volume)

L_{IN} = base constituent load (mass/capita/day)

C_{GW} = graywater constituent concentration (mass/volume)

$Q_{GWI} = d_{GI} \times P_{GWI}$ = graywater irrigation demand (volume/day)

L_{SS} = source separation constituent load (mass/capita/day)

Q_{IN} = influent flow to treatment facility (volume/day)

P_{GWI} = Residential population adopting graywater reuse for irrigation (capita)

P_S = Service population (capita)

A BioWin^{TM} 5.0 (EnviroSim Associates Limited, 2016) calibrated and validated process model was used to evaluate the change in influent water quality for each water management scenario. The evaluated WWTF is a Modified Ludzack Ettinger (MLE) process with nitrification and denitrification, an average flow of 0.65 m^3/s and a permitted capacity of 1.10 m^3/s. The model was developed based on 2014 influent and effluent data and validated to five months of 2015 data (McKenna et al., 2017). The model calibration was developed by McKenna et al., 2017, and the base observed and calibrated modeled effluent water quality is provided in Table 2-4. Each scenario was modeled in BioWin^{TM} to determine the effluent flow, concentration (C_{EFF}), and load (L_{EFF}) and used to model the impacts to receiving water body.

Table 2-4. Base Scenario Model Effluent Water Quality

Effluent	Observed Value	Calibrated Value
Flow (m^3/s)	0.65 (14.8 MGD)	0.65 (14.8 MGD)
Ammonia (mg N/L)	0.09	0.09
Nitrate (mg N/L)	13.9	12.7
Nitrite (mg N/L)	0.03	0.03
TKN (mg N/L)	1.8	1.9
TN (mg N/L)	15.8	15.2
TP (mg N/L)	2.7	3.1
TSS (mg/L)	6.5	6.7
TCBOD (mg/L)	3.3	3.1
pH	6.74	6.7

The WWTF discharges to the Boulder Creek (Boulder, CO) which is primarily fed by snowmelt and results in high peak flows in the spring and low flows in the winter and can experience flashy behavior resulting from heavy spring, summer or fall precipitation. A flow duration curve was prepared for 2000 to 2016 at USGS Site Number 06730200 located immediately upstream of the effluent discharge to characterize historical stream flows (Figure 2-2).

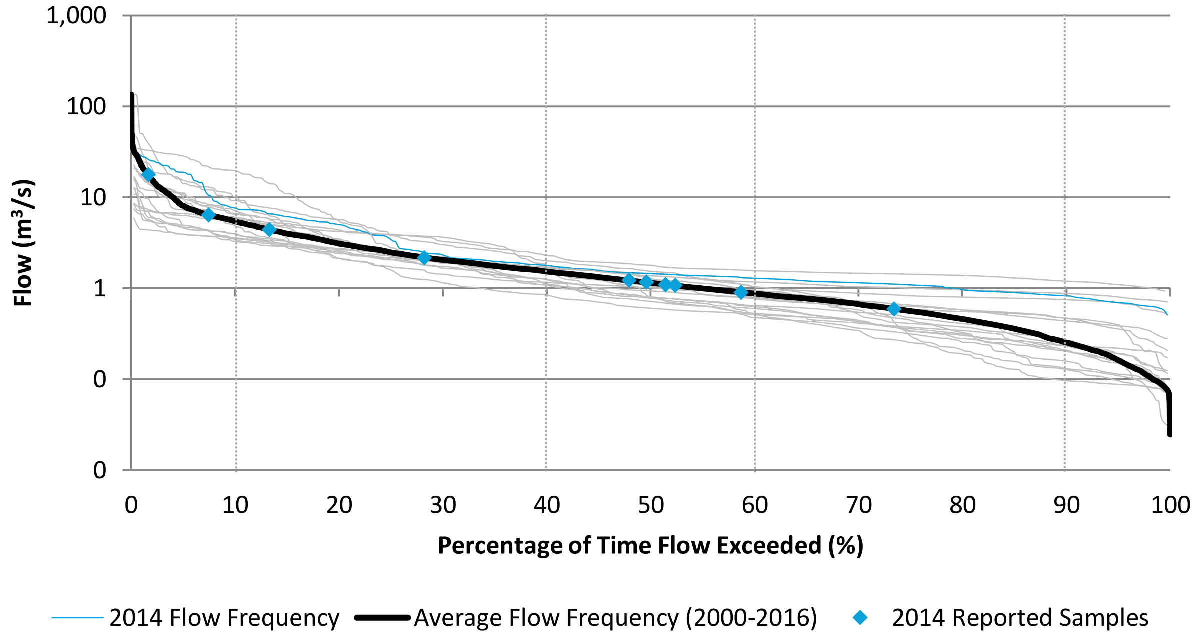

Figure 2-2. Average Boulder Creek flow duration curve at USGS station 06730200 based on daily stream flows from 2000-2016 indicating high flow, moist conditions, mid-range flow, dry conditions, and low flow and highlighting 2014 flow frequency. Data points indicate 2014 reported water quality samples and corresponding average flow frequency condition. The percent of effluent contributing to stream flow is indicated based on the average 2014 effluent discharge.

These stream characteristics are common in arid west regions where streams are primarily fed by snowmelt and the arid climate results in large variations between high and low flows. The flow duration curve is segmented into high flow, moist conditions, mid-range flow, dry conditions and low flow based on the frequency of flow occurrence corresponding to flow values of > 5.47, 5.47-1.53, 1.53-0.88, 0.88-0.25, < 0.25 m^3/s (<193, 193-54, 54-31, 31-9, <9 ft^3/s) respectively.

With this large variability in flow, the stream is generally more influenced by the effluent water quality in the fall and winter (low flow) and less influenced in the spring (peak flow). However, high flow and moist conditions can be observed at various times in the year during heavy rainfall events and mid-range flow, low flow and dry conditions can be common during periods of drought. The stream gauge is located immediately upstream of the WWTF effluent. During the study period (2014), the stream flows were above average ranging from 0.51-30.56 m^3/s.

The studied WWTF collected 12 monthly grab samples of flow, TN concentrations and TP concentrations in 2014 and reported to the CDPHE for nutrient regulation compliance (STORET, 2016). Samples are reported for the discharge as well as upstream and downstream of the discharge location. The scenario effluent flow, TN and TP were used to estimate downstream water quality on a monthly basis based on the upstream reporting data correlated to the historical flow frequency. In 2014, the monthly average discharge from the WWTF was 0.535-0.811 m^3/s (12.2 – 18.5 million gallons per day (MGD)) and the monthly average discharged flow accounted for 4 – 47% (average 30%) of the downstream flow with average monthly upstream flows ranging from 0.59-17.89 m^3/s (13.6 – 408.5 MGD).

The receiving water body will be more sensitive to changes in effluent concentration when the stream is effluent dominated and thus the percentage of stream flow comprised of effluent was provided for reference based on the average 2014 effluent WWTF flows and historical stream flows (Figure 2-2). The 2014 stream flow was high compared to the historical stream flow with one sample representative of average dry conditions, seven samples representative of average mid-range flow, two samples representative of moist conditions, and two samples representative of high flow conditions (Figure 2-2). Historically, Colorado frequently experiences drought conditions where in 2008 and the Boulder stream exhibited historical mid-range flow, low flow or dry conditions for 75% of the year.

The downstream flow (Q_{DS}) was determined based on the upstream flow (Q_{US}) and effluent flow (Q_{EFF}) discharged from the WWTF. For the indoor conservation scenarios, the downstream flow was calculated under two conditions, assuming either conserved flow (Q_{CONS}) stays in the stream (equation 2-3) or conserved flow is removed from the stream for consumptive use or not available (equation 2-4). For source separation, graywater reuse for toilet flushing, and graywater reuse for irrigation, it was assumed that conserved water does not remain in the stream (equation 2-4). For WWTF effluent reuse, the discharged effluent flow was calculated removing the percentage of water allocated for reuse ($R_{\%}$) (equation 2-4). Downstream concentrations (C_{DS}) were determined based on the effluent flow and concentration (C_{EFF}) and upstream flow and concentration (C_{US}) indicated in equation 2-5. The calculated downstream flow and concentrations were used to calculate the downstream load (L_{DS}).

$$\boldsymbol{Q_{DS}} = \boldsymbol{Q_{US}} + \boldsymbol{Q_{Eff}} + \boldsymbol{Q_{CONS}} \tag{2-3}$$

$$\boldsymbol{Q_{DS}} = \boldsymbol{Q_{US}} + \boldsymbol{Q_{EFF}} - (\boldsymbol{Q_{EFF}} \times \boldsymbol{R}_{\%}) \tag{2-4}$$

Where:

Q_{EFF} = effluent flow from treatment facility (volume/day)

Q_{IN} = influent flow to treatment facility (volume/day)

$Q_{CONS} = \Delta d_C \ x \ P_S$ = flow conserved (volume/day)

$R_{\%}$ = percent wastewater effluent reuse (percent)

$$\boldsymbol{C_{DS}} = \frac{(\boldsymbol{C_{US}} \times \boldsymbol{Q_{US}}) + (\boldsymbol{C_{EFF}} \times \boldsymbol{Q_{EFF}} \times (\boldsymbol{1} - \boldsymbol{R}_{\%}))}{\boldsymbol{Q_{DS}}} \tag{2-5}$$

Where:

C_{US} = upstream constituent concentration (mass/volume)

Q_{US} = upstream flow (volume/day)

C_{EFF} = effluent constituent concentration from treatment facility (mass/volume)

Q_{EFF} = effluent flow from treatment facility (volume/day)

C_{DS} = downstream constituent concentration (mass/volume)

$R_{\%}$ = percent wastewater effluent reuse (percent)

2.3 Results

In general, influent water quality was more notably impacted with indoor conservation and source separation adoption (Figure 2-3A and Figure 2-3B), and less impacted by graywater reuse adoption (Figure 2-3C and Figure 2-3D). Intuitively with adoption of indoor conservation, a constant influent load and decrease in water use equates to a notable increase in concentration (Figure 2-3A) where a 54% reduction in flow resulted in an increase in TN and TP concentration of 117% and 118% respectively (Figure 2-3A). Source separation was estimated to have a substantial decrease in influent TN and TP (Figure 2-3B) where full adoption of source separation would reduce influent flow by 7%, corresponding to a reduced influent TN and TP concentration of 63% and 40% respectively, and TN and TP load of 66% and 44% respectively (Figure 2-3B).

Like indoor conservation, graywater reuse for toilet flushing also assumed a constant influent nutrient load to the WWTF. However, the impacts on flow reduction were much less compared to indoor conservation, resulting in a less drastic increase on influent concentrations where even at full scale adoption a 9% reduction in flow equated to an increase in TN and TP concentration by 10% and 11% respectively (Figure 2-3C). Similarly, graywater reuse for irrigation had notable impacts on influent flow, but less drastic impacts on nutrient concentration as the reduction in flow is much greater than the reduction in load where at full scale adoption the flow is reduced by 19% and the influent TN and TP concentration increases by 17% and 11% respectively (Figure 2-3D).

It is important to note that to observe this level of adoption for graywater toilet or irrigation reuse would require significant amount of infrastructure investment including dual plumbing of buildings, on-site storage, and some level of on-site treatment. This level of adoption is not likely, and the model results overall suggest that these practices have negligible impacts on wastewater quality.

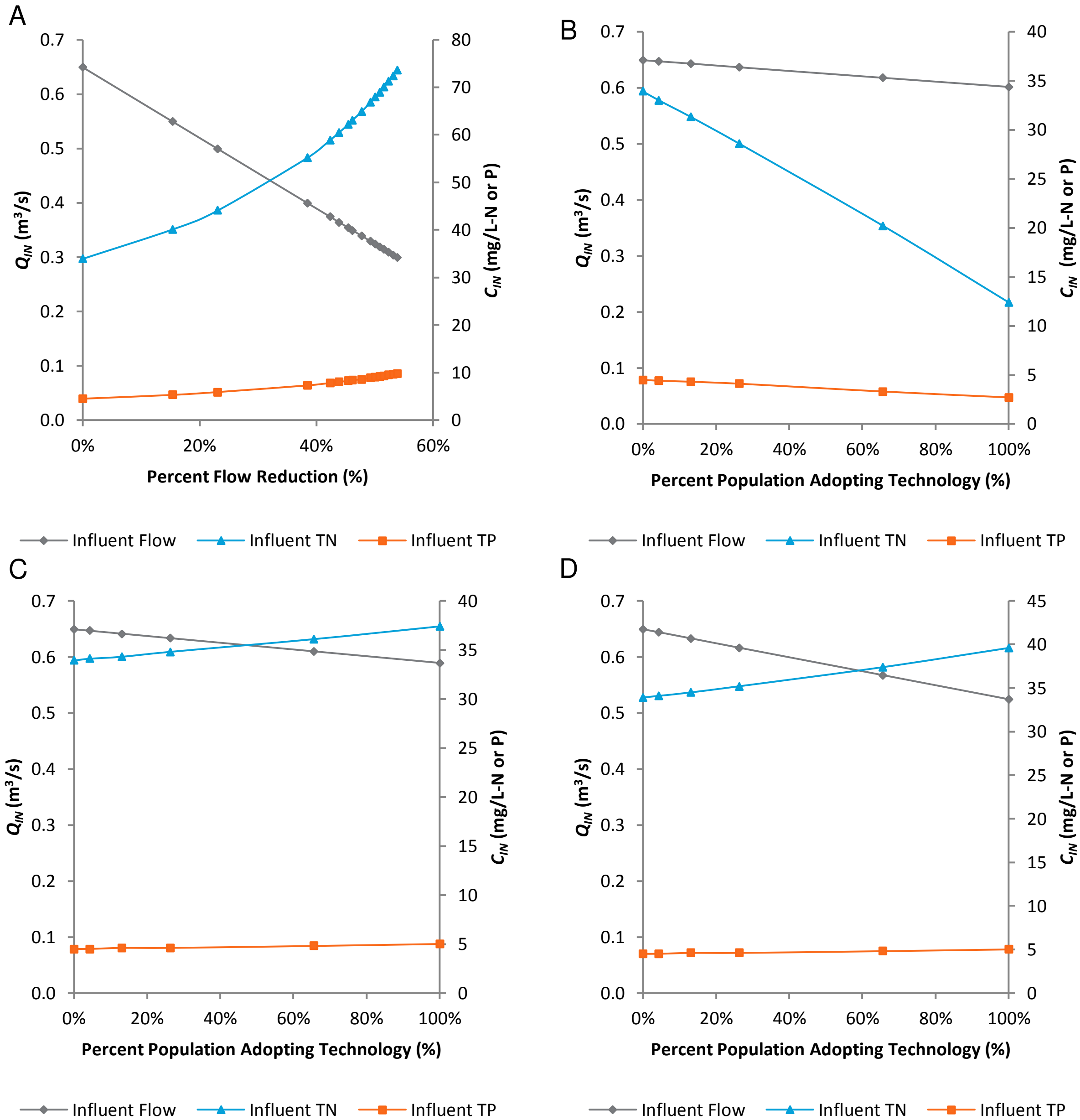

Figure 2-3. Impact on influent flow (QIN) and influent TN and TP concentration (CIN) for indoor conservation (A), source separation (B), graywater reuse for toilet flushing (C) and graywater reuse for irrigation (D).

Using the above influent water quality scenarios, effluent concentration of TN and TP were predicted based on BioWin™ modeling assuming no change in operational parameters. The resulting impacts at the highest levels of adoption evaluated were summarized in Table 2-5.

Table 2-5. Water Management Practices percent change to effluent water quality based on highest levels of adoption evaluated

Practice	Scenario	WWTF Effluent Percent Change				
		Q_{EFF}	TN L_{EFF}	TP L_{EFF}	TN C_{EFF}	TP C_{EFF}
Indoor Conservation	54% Flow Reduction (60 gpcd)	-54%	-14%	+1%	+87%	+118%
Source Separation	100% Population Adoption	-7%	-81%	-69%	-80%	-66%
Graywater Toilet Reuse	100% Population Adoption	-9%	+2%	+2%	+12%	+13%
Graywater Irrigation Reuse	100% Population Adoption	-19%	0%	-10%	+24%	+11%
WWTF Effluent Reuse	40% Reuse	-40%	-40%	-40%	0%	0%

The scenarios with more drastic influent impacts, conservation and source separation, showed the most notable impacts on effluent concentrations (Figure 2-4A and Figure 2-4B), and the scenarios that showed little to no impact on the influent concentrations, graywater reuse, had negligible impacts to effluent water quality (Figure 2-4C and Figure 2-4D). At high levels of indoor conservation, a reduction of effluent TN load is observed, similar to findings from Paulsen et al. (2007). This indicates that the TN load removal is improved with conservation; however, the wastewater treatment facility is not able to meet the same performance metrics in terms of concentration (Table 2-5; Figure 2-4A).

Similarly, an improved load removal is estimated with increasing adoption of source separation, indicating improvement in wastewater treatment facility efficiency where a reduction of influent TN and TP load by 66% and 44% respectively corresponded to a reduction in effluent TN and TP load of 81% and 69% respectively (Table 2-5), indicating that the percentage load removal increased with higher levels of adoption. While, the impacts of wastewater effluent reuse on concentration and load are clearly understood they are provided graphically for comparison where a 40% reduction in effluent flow equates to a 40% reduction in nutrient loads but no impact on effluent concentrations (Figure 2-4E; Table 2-5).

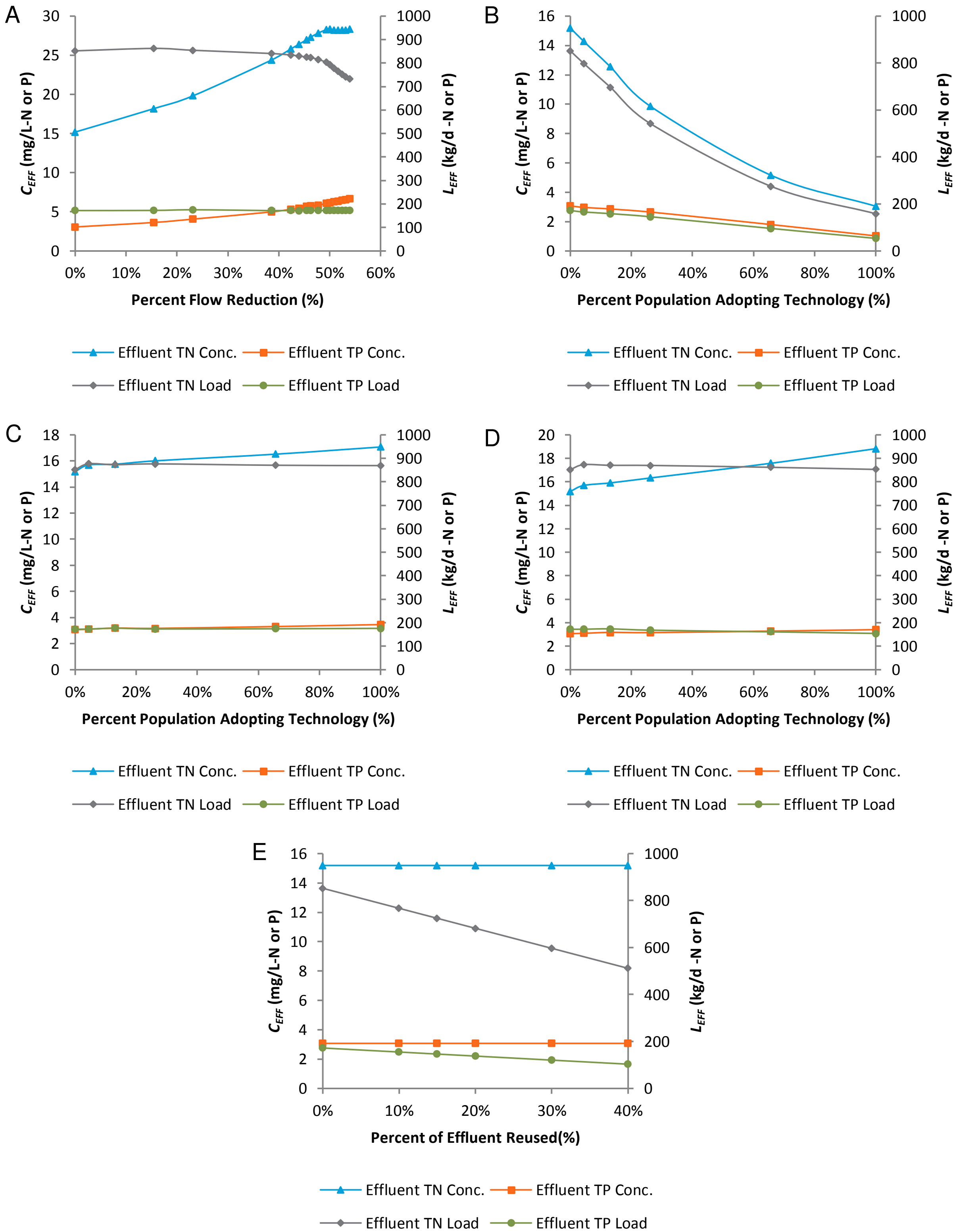

Figure 2-4. Impact on effluent TN and TP concentration (CEFF) and loading (LEFF) for indoor conservation (A), source separation (B), graywater reuse for toilet flushing (C), graywater reuse for irrigation (D), and WWTF effluent reuse (E).

The primary purpose of this work was to evaluate the impacts of water management practices on receiving water quality. For indoor conservation, while the effluent concentration increases with adoption, the effluent load is relatively constant, or even slightly improved (i.e. load reduction; Figure 2-4A) and there is little impact to downstream concentrations. If conserved flow stays in the stream, the result to the downstream concentrations is actually slightly improved where at 54% reduction in flow with indoor conservation reduces the TN concentration particularly in mid-range flow and dry conditions (Figure 2-5A). However, conservation practices are often implemented under drought conditions when water scarcity is an issue. If conserved flow is instead utilized for a consumptive use the downstream impacts are noteworthy evident with the mid-range flow and dry conditions considered (Figure 2-5B).

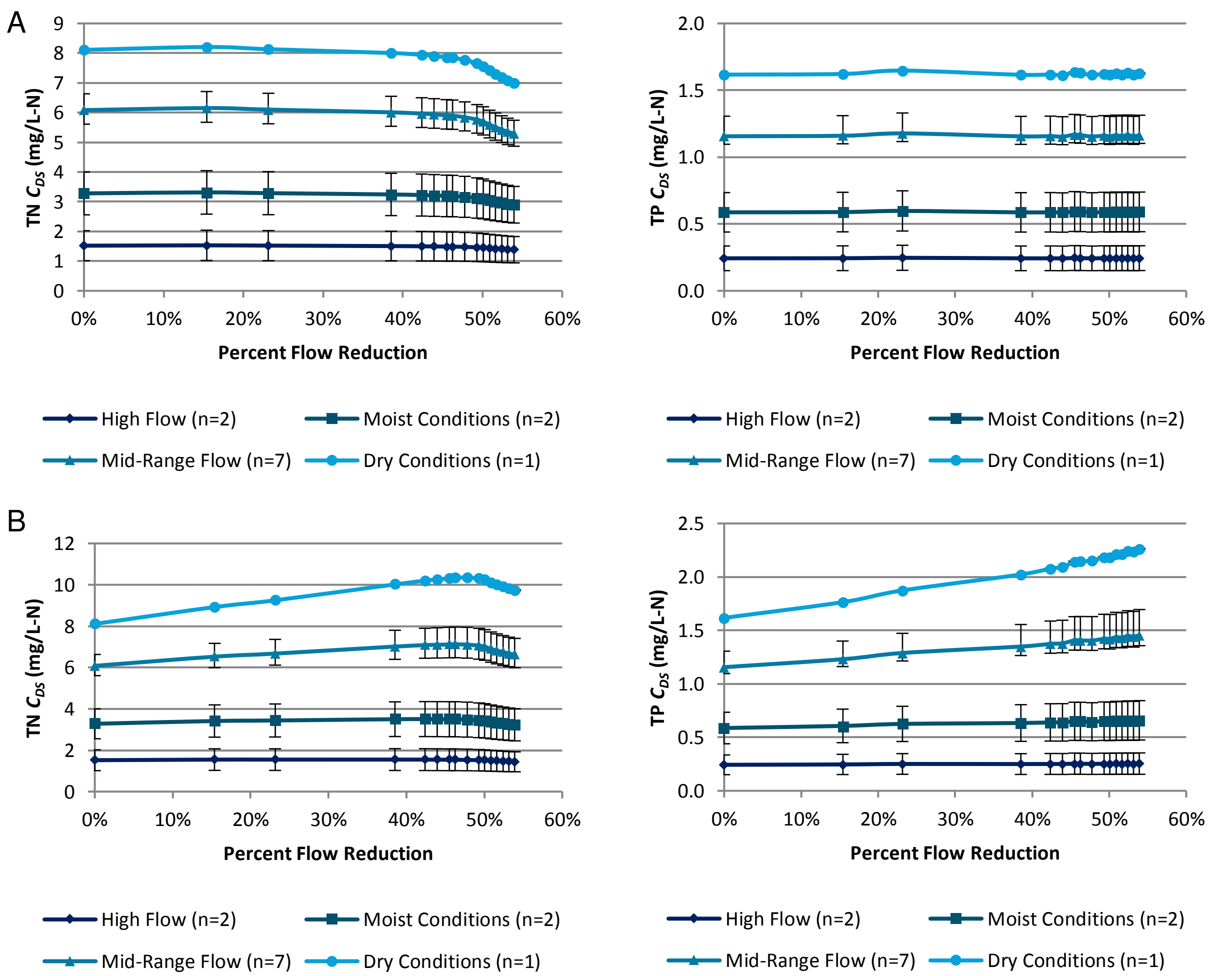

Figure 2-5. Indoor Conservation impact on downstream TN and TP concentration (CDS) with conserved flow returned to stream (A) and with conserved flow consumed (B) based on receiving water body flow condition with number of samples indicated in legend and max and minimum values indicated by bars.

For source separation, the positive effluent load reduction impacts to downstream water quality can be considerable particularly under dry and mid-range flow conditions which are common in the arid west where the effluent flow can dominate upstream flows (Figure 2-6). Both graywater reuse for toilet flushing or irrigation had negligible impacts on influent and effluent water quality, corresponding to negligible estimated changes in downstream TN and TP concentrations regardless of stream conditions (Figure 2-7). Lastly, the downstream concentration improvements associated with WWTF effluent reuse were considerable in mid-range flow and dry conditions where effluent reuse is most beneficial as an additional water source (Figure 2-8).

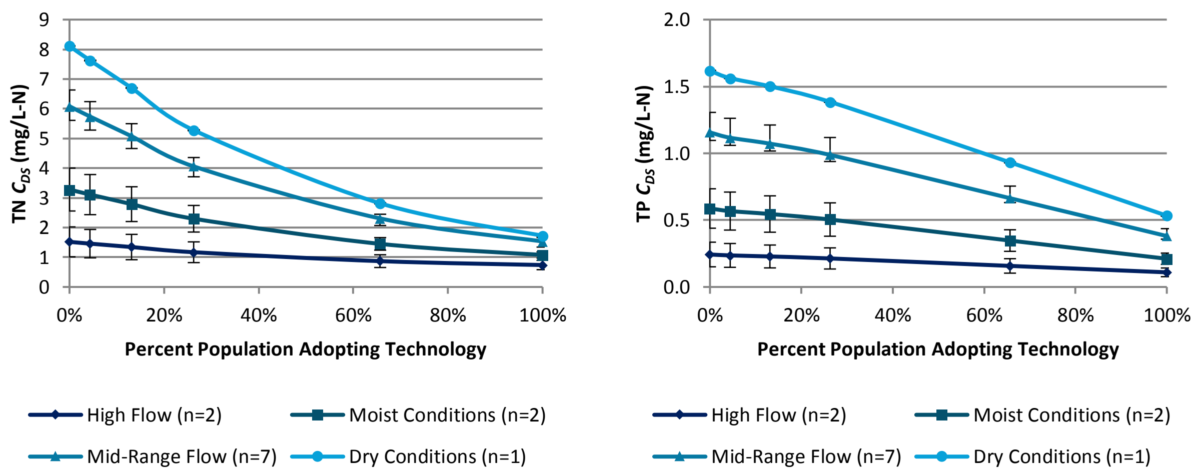

Figure 2-6. Source Separation impact on downstream TN and TP concentration (CDS) based on receiving water body flow condition with number of samples indicated in legend and max and minimum values indicated by bars.

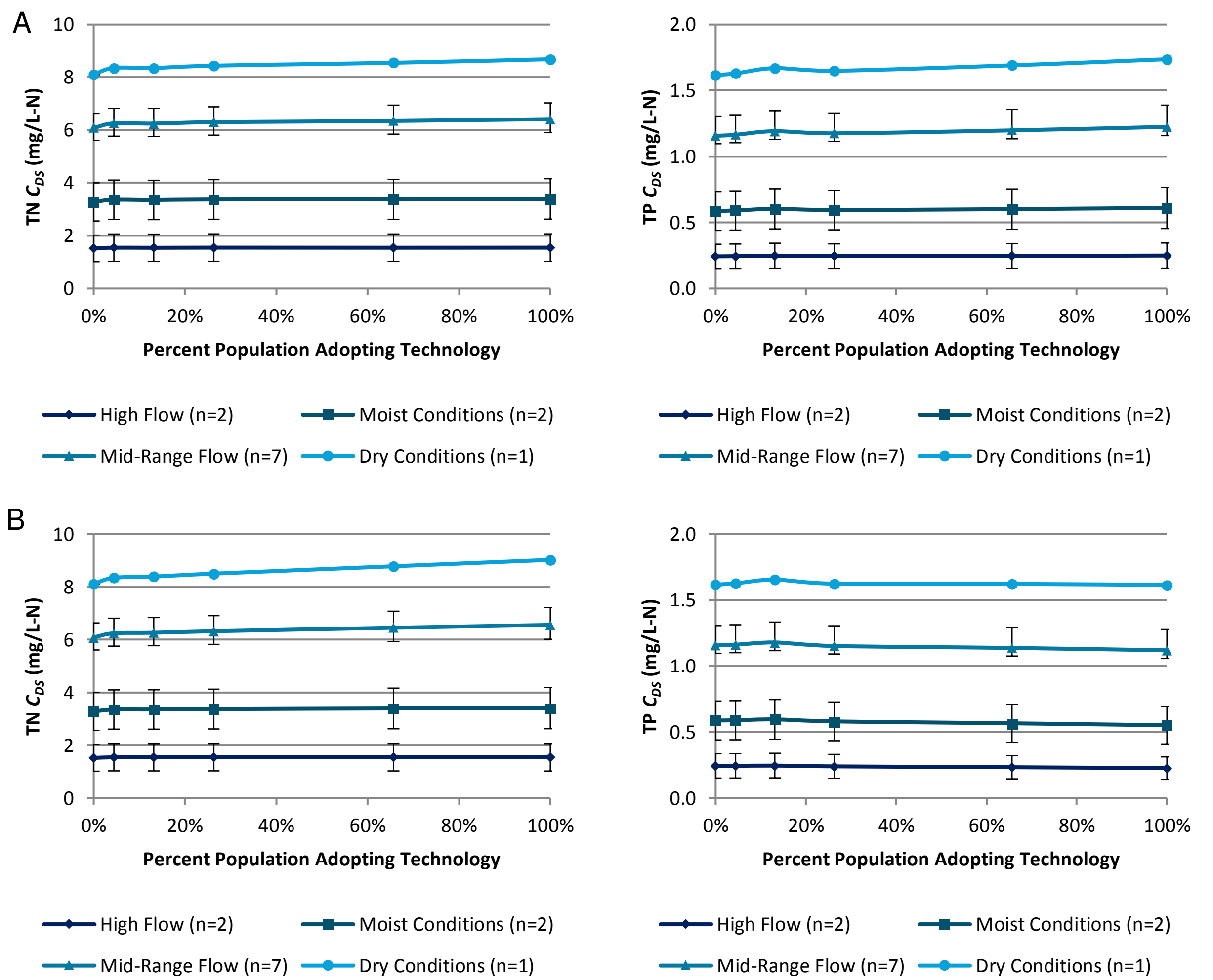

Figure 2-7. Impact on downstream TN and TP concentration (CDS) for graywater reuse for toilet flushing (A) and graywater reuse for irrigation (B) based on receiving water body flow condition with number of samples indicated in legend and max and minimum values indicated by bars.

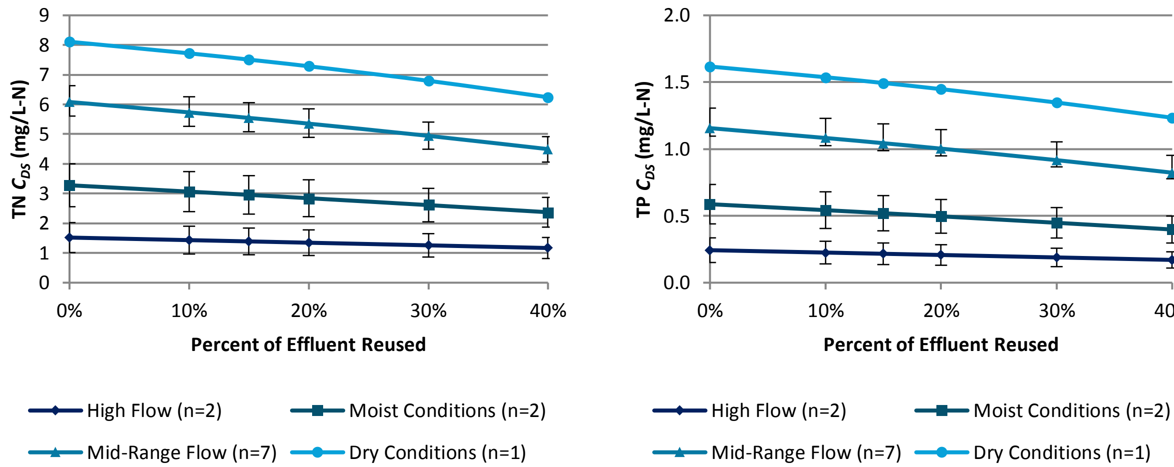

Figure 2-8. WWTF Effluent Reuse impact on downstream TN and TP concentration (C_{DS}) based on receiving water body flow condition with number of samples indicated in legend and max and minimum values indicated by bars.

To further illustrate the receiving body impacts, load duration curves were developed, assuming a stream standard of 2.01 mg/L TN and 0.17 mg/L TP (projected Colorado standards; CDPHE, 2017), for indoor conservation, source separation and effluent reuse where notable impacts to downstream concentrations were estimated (Figure 2-9 and Figure 2-10). These figures highlight the challenges for WWTF to sufficiently reduce effluent discharges to meet nutrient standards where even at full scale adoption of source separation, TP in the stream is still above the standard and TN load just meets the stream standard in dry conditions. However, practices like effluent reuse can be part of the solution to reduce nutrient loads (Figure 2-9 and Figure 2-10). Water conservation showed minimal impacts to stream loads and that is also reflected in the load duration curves (Figure 2-9 and Figure 2-10).

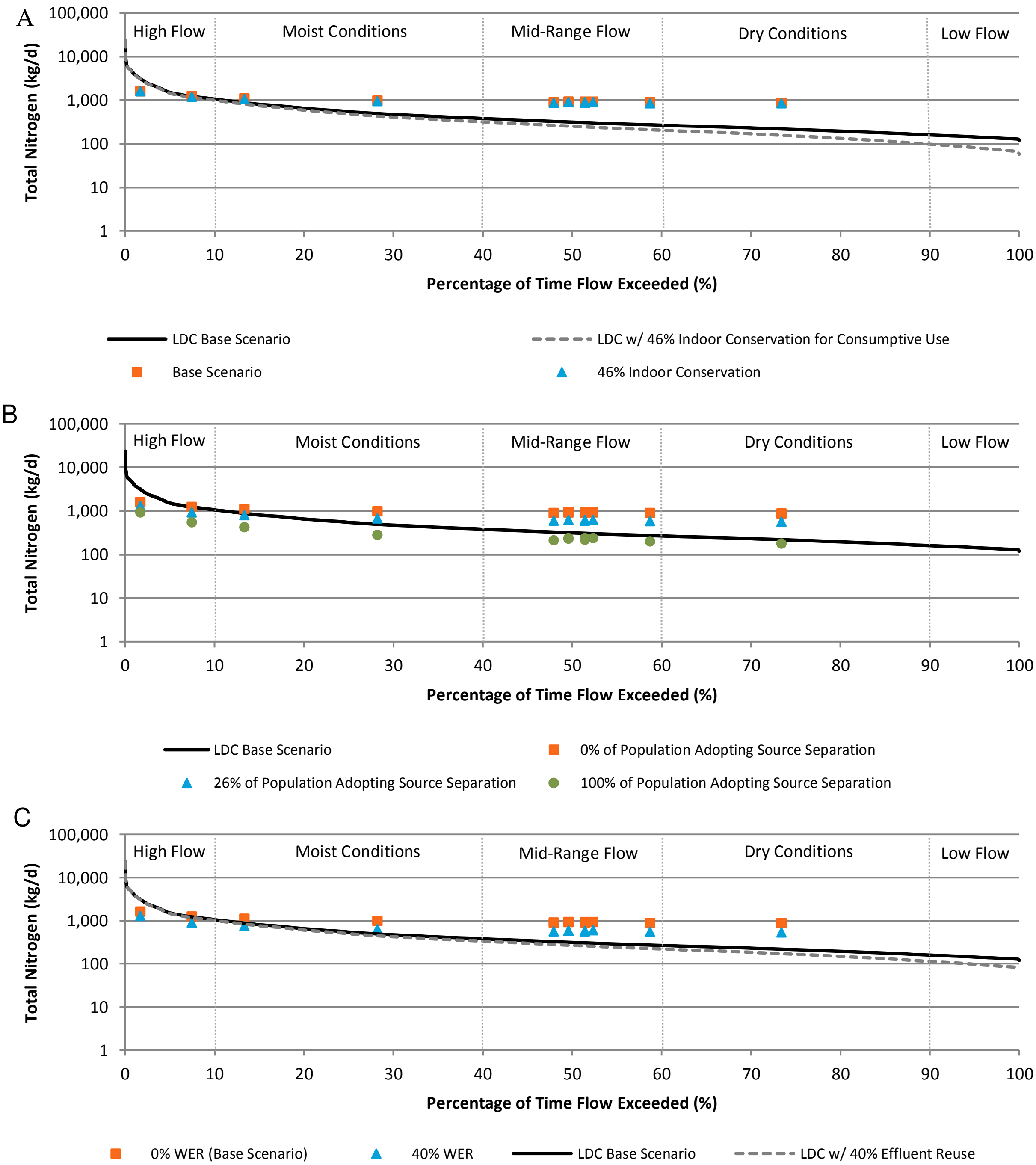

Figure 2-9. Downstream load duration curve based on a TN stream standard of 2.01 mg/L and adoption of indoor conservation (A), source separation (B), and WWTF effluent reuse (C) where the data points represent the TN stream loading correlated to the flow frequency. An adjusted load duration curve (LDC) was calculated for indoor conservation and effluent reuse where there is a notable reduction in effluent flow indicated with dashed lines.

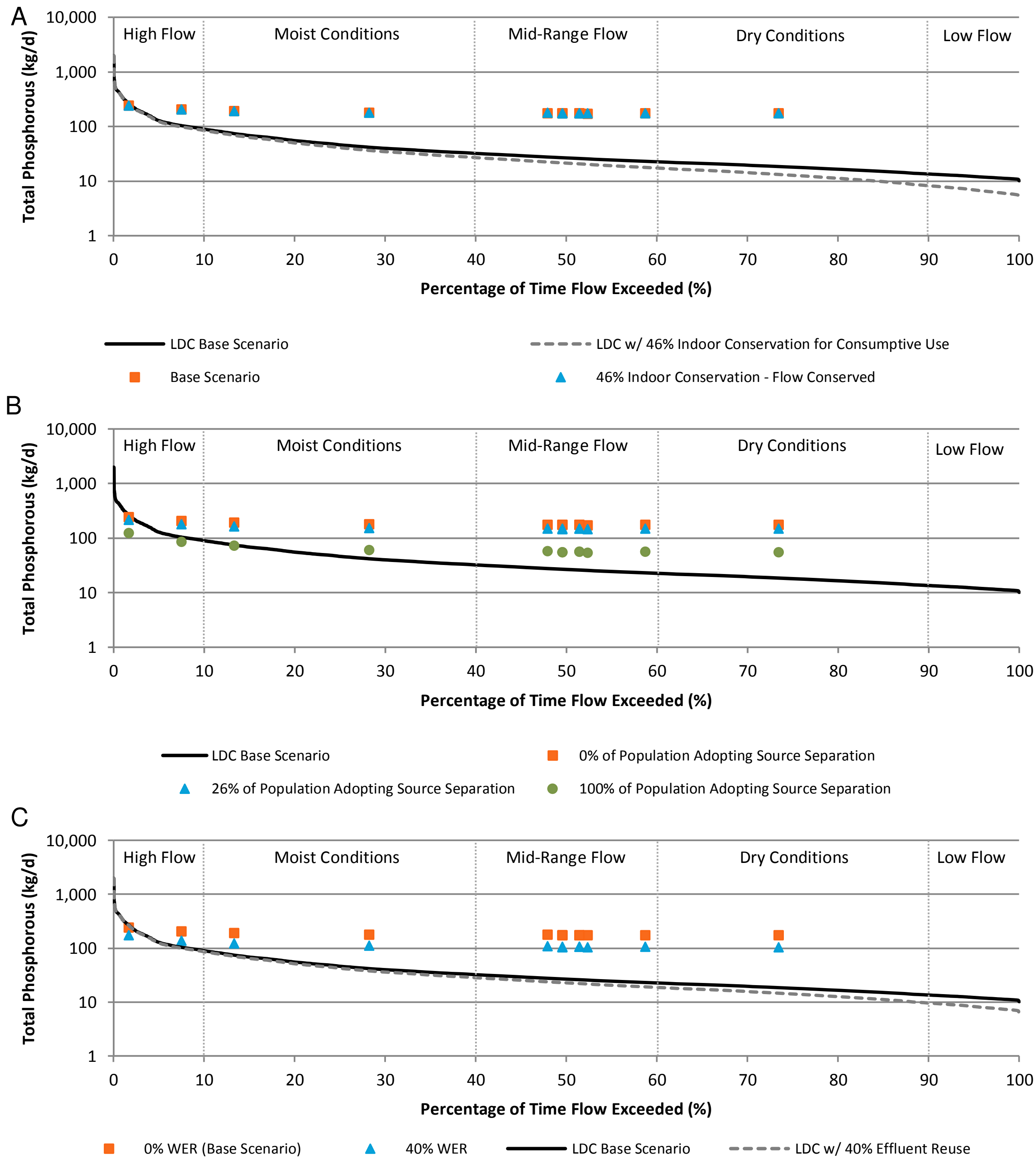

Figure 2-10. Downstream load duration curve based on a TP stream standard of 0.17 mg/L and adoption of indoor conservation (A), source separation (B), and WWTF effluent reuse (C) where the data points represent the TP stream loading correlated to the flow frequency. An adjusted LDC was calculated for indoor conservation and effluent reuse where there is a notable reduction in effluent flow indicated with dashed lines.

2.4 Discussion

The study area evaluated the impacts of water management practices in a typical arid west water system where water supplies are limited and receiving water bodies are predominantly snow melt dominated with large fluctuations in flow. Such water systems face great challenges for water supply management and preserving water quality. The results indicate that there are some noteworthy implications of water management practices on downstream water quality, particularly when the stream is experiencing mid-range flow, low flow or dry conditions. These conditions can be common during drought and effluent discharge can account for greater than 30% of the stream flow. These conditions are common in arid west systems where practices like source separation and effluent reuse could provide positive benefits to receiving water quality and high adoption of indoor water conservation could impact WWTF operations.

2.4.1 Water conservation

Between 1996 and 2016, it was estimated that there was a 15% residential indoor water use decrease per capita (DeOreo et al., 2016). However, notable improvements in indoor conservation are still viewed as possible with the potential for additional 35% reduction in indoor water use for residential uses with implementation of high efficiency devices (DeOreo et al., 2016) and further opportunities to reduce CII water use.

Increasing water conservation over a range of adoption levels was estimated to have negligible impacts in terms of WWTF load removal performance but notable increase in effluent concentrations. McKenna et al. 2017 evaluated increased adoption of indoor conservation and observed similar trends across the study area and 3 other WWTFs. As influent concentration increases, the WWTF load removal is limited with increasing influent concentrations as a result of pH inhibition, insufficient alkalinity and carbon deficiency limiting nitrification and denitrification (McKenna et al., 2017). Therefore; as conservation increases, WWTF may need to supply additional carbon or alkalinity to improve process efficiency (Lowe et al., 2009; McKenna et al., 2017).

Importantly, with many WWTFs regulated for effluent nutrient concentrations, increasing water conservation may require additional operational costs and treatment improvements and it is important to consider the downstream impacts. Two alternatives were evaluated where conserved flow stays in stream or is diverted for consumptive use representing two extremes. Under the first extreme where all conserved flow is maintained in streams the downstream impacts are negligible as there is little change in effluent load with conservation and the conserved stream flow provides additional dilution capacity (Figure 2-4A). However, under the other extreme where all conserved flow is diverted for consumptive use or limited based on supply, the downstream impacts on concentration are noteworthy particularly under mid-range flow and dry conditions where conservation practices are most commonly implemented (Figure 2-4B).

The impacts observed to WWTF performance noted in this study are likely broadly observed over WWTFs (McKenna et al., 2017). Thus, projected trends noted in this study in periods of varying flow conditions are likely to be observed in other similar receiving water bodies. The results from this study indicate that adopting conservation practices during drought conditions to stretch a water supply will have negligible impacts on influent and effluent loading but noteworthy impacts on downstream concentration. Increased indoor water conservation could require treatment modifications at the WWTF to improve system performance to meet permit requirements and preserve downstream water quality.

2.4.2 Source Separation

Source separation was evaluated for extreme levels of adoption including evaluation of the entire population adopting source separation which corresponded to a TN and TP load reduction of 66% and 44% respectively at the WWTF (Figure 2-3B). The large potential for wastewater nutrient reduction occurs because urine represents a substantial fraction of the nitrogen load (75-80%) and phosphorous load (50-55%) in wastewater (Fewless et al., 2011). Worth noting is that the reduction of influent load also resulted in improved percent load removal at the WWTF where the effluent TN and TP load was reduced by 81% and 69% respectively (Figure 2-4B; Table 2-5).

While source separation does improve the estimated percent load removal at the WWTF, this does not necessarily indicate a more effective way of reducing downstream nutrient load. The percent load removal improvement is largely a function of a smaller influent load skewing the calculation. However, looking at the mass balance at the WWTF influent (Figure 2-3B) and effluent (Figure 2-4B), the results generally indicate for every 1 kg of TP removed at the influent corresponded to an effluent reduction of

1.0 kg TP indicating a linear 1 to 1 mass decrease in influent and effluent TP load. Interestingly for TN, the effluent load removal return diminished as source separation adoption increases (Figure 2-4B).

At less than 26% population adoption, for every 1 kg TN removed at the influent (Figure 2-3B) corresponded to an effluent reduction of 0.94 kg TN (Figure 2-4B). However, if 100% of the population adopted source separation for every 1 kg of TN removed at the influent would only correspond to a reduction of 0.6 kg TN at the effluent (Figure 2-4B). One may hypothesize that reducing the influent load may notably improve the WWTF performance resulting in further reduction of effluent loads, but this does not appear to be the case for TN. The lack of improved performance is likely to be a function of dilute influent impacting nitrification/denitrification.

As expected, reduction of effluent load corresponds to notable improvements in downstream TN and TP loading and concentration (Figure 2-6). Based on the identified stream criteria, the TN standard is achievable with source separation, but would require full scale adoption (Figure 2-9). Improved treatment at the WWTF would still be necessary to meet the TP standard (Figure 2-10).

While the results for source separation show notable positive impacts on treatment operations, permit compliance, and downstream loading; there are significant social and economic barriers for adoption (Fewless et al., 2011). In addition, pharmaceuticals present in urine must be considered for appropriate management and treatment of the urine stream (Fewless et al., 2011). The cost to overcome these barriers may be more effectively spent to improve treatment practices at the WWTF, particularly for TP where source separation alone was not sufficient for achieving the desired stream standard. However, source separation may be an effective management practice in decentralized systems particularly in high density locations like a residence hall (Ishii and Boyer, 2015) or in rural areas served by on-site septic systems where treatment of nutrients is unreliable and difficult to achieve.

2.4.3 Graywater and Effluent Reuse

In general, graywater reuse for toilet flushing and irrigation showed minimal impact on influent and effluent water quality. At full scale adoption, where all graywater is collected for toilet or irrigation use, the impact in terms of effluent load are negligible (Table 2-5), and minimal impacts to downstream concentrations are projected (Figure 2-7). As previously discussed, full scale adoption would be infrastructure intensive and not practical showing that even at extreme scales the impact of graywater reuse to water quality is negligible. While graywater reuse has well studied benefits in terms of water conservation and supply, due to the relatively dilute nutrient concentrations, adoption of graywater reuse for irrigation or toilet reuse does not pose a notable concern or benefit for receiving water body quality.

Intuitively, when WWTF effluent reuse is adopted there is a corresponding reduction in effluent load and no change in effluent concentration (Figure 2-4E). However, given that many WWTF are regulated based on effluent concentrations, the downstream impacts of effluent reuse are often not credited for permit compliance. The impact of effluent reuse on downstream water quality is largely dependent on the percentage of effluent flows that comprise the total downstream flow. In effluent dominated streams, which can occur during low flow, dry conditions or mid-range flow, there will be more notable reduction to downstream nutrient concentrations (Figure 2-8).

These conditions are common in the arid region water systems. Typically, effluent reuse receives greater consideration for adoption in drought conditions when supplies are limited. Under these low flow stream conditions, effluent reuse could result in more notable improvements to downstream water quality accounting for the flow diverted for reuse. Of note is that impacts to water quantity to support ecosystem health should also be considered. Conversely, effluent reuse for irrigation in the spring, during non-drought periods when peak stream flow is observed, has little impact to stream nutrient concentrations (Figure 2-8). This suggests that observed benefits from effluent reuse on receiving water body quality may not be as notable in high flow receiving water bodies that are not effluent dominated.

Effluent reuse alone was not sufficient in meeting the desired in-stream TN and TP stream standards (Figure 2-9 and Figure 2-10), and while there is a clear beneficial load reduction with effluent reuse, the benefit in downstream concentration will be highly dependent on local conditions including the seasonality of stream flows and the timing of demands for effluent reuse. Depending on the watershed, seasonal variations may need to be considered when accounting for the beneficial load reduction recognizing a larger benefit in low flow and dry conditions and a reduced value during peak flow conditions. Importantly, effluent reuse does not help WWTFs meet concentration based regulations, whereas a load based regulation can account for the beneficial reduction in pollutant load.

Graywater and effluent reuse have the potential to decrease irrigation demand, which is particularly beneficial in reducing seasonal irrigation demands. In the case study area, stretching the existing supply can reduce the annual imported water and potentially negate the need for a secondary water treatment facility by reducing the seasonal potable water demands. The associated cost and energy savings may be significant, justifying adoption of water reuse practices. Many municipalities in the arid west region of the US have similar needs for additional imported water and secondary water treatment facilities to meet peak demand during the irrigation season. Because graywater and effluent reuse are likely to result in negligible or positive impacts to receiving water body concentration, these practices can offer water supply benefits without negatively impacting receiving water body quality.

2.4.4 Downstream Water Quality Impacts of Water Management Practices

The evaluated water demand reduction practices had a variety of water quality implications. As influent becomes more concentrated with increased adoption of indoor conservation, WWTFs may not be able to maintain existing effluent concentrations resulting in increased effluent TN and TP concentrations. As a result, WWTF may require process modifications including alkalinity and/or carbon addition (McKenna et al., 2017). While the water conservation can have notable benefits for stretching a water supply and downstream loading impacts may be negligible, downstream concentration impacts are noteworthy when conservation measures are implemented under drought conditions.

For source separation the results generally indicated that for every 1 kg of TN and TP removed at the influent a 1 kg reduction in the effluent was observed. However, this trend diminishes for TN at high levels of adoption. While notable load reductions of TN and TP can be achieved with source separation, the social barriers and implementation cost may limit wide spread adoption and it is likely more cost effective to invest in treatment improvements at the WWTF. While the benefits of source separation are largely a function of the high source concentrations, graywater is comparably dilute and adoption of graywater reuse for toilet flushing or irrigation is estimated to have negligible impacts on WWTF operations and downstream water quality even at high levels of adoption during dry conditions.

Conversely, the downstream impact of WWTF effluent reuse is largely dependent on the receiving water body and seasonality of flows but notably beneficial during mid-range flow and dry conditions. Graywater and effluent reuse in arid regions during peak months may reduce imported supply or possibly negate the need for a supplementary treatment facility with either positive or negligible impact to WWTF operations and receiving water body quality. Importantly, while downstream water quality improvements are projected to be noteworthy when effluent reuse is adopted in drought conditions, effluent concentrations remain constant which deters effluent reuse adoption under concentration based regulations.

This research focused on quantifying the impacts of individual practices, further studies should be done to quantify the combined impacts of water management practices because many of the practices are likely to be adopted in combination. Additionally, the research focused on water quality impacts at the WWTF where water quality is typically homogeneous; however, adoption of conservation practices is often heterogeneous and therefore there may be local, site specific impacts that are negative or beneficial and have not been quantified with this study.

2.5 Summary

The evaluated water demand reduction practices had a variety of water quality implications. As influent becomes more concentrated with increased adoption of indoor conservation, WWTFs may not be able to maintain existing effluent concentrations resulting in increased effluent TN and TP concentrations. As a result, WWTF may require process modifications including alkalinity and/or carbon addition (McKenna et al., 2017). While the water conservation can have notable benefits for stretching a water supply and downstream loading impacts may be negligible, downstream concentration impacts are noteworthy when conservation measures are implemented under drought conditions. For source separation the results generally indicated that for every 1 kg of TN and TP removed at the influent a 1 kg reduction in the effluent was observed. However, this trend diminishes for TN at high levels of adoption. While notable load reductions of TN and TP can be achieved with source separation, the social barriers and implementation cost may limit wide spread adoption and it is likely more cost effective to invest in treatment improvements at the WWTF. While the benefits of source separation are largely a function of the high source concentrations, graywater is comparably dilute and adoption of graywater reuse for toilet flushing or irrigation is estimated to have negligible impacts on WWTF operations and downstream water quality even at high levels of adoption during dry conditions. Conversely, the downstream impact of WWTF effluent reuse is largely dependent on the receiving water body and seasonality of flows but notably beneficial during mid-range flow and dry conditions. Graywater and effluent reuse in arid regions during peak months may reduce imported supply or possibly negate the need for a supplementary treatment facility with either positive or negligible impact to WWTF operations and receiving water body quality. Importantly, while downstream water quality improvements are projected to be noteworthy when effluent reuse is adopted in drought conditions, effluent concentrations remain constant which deters effluent reuse adoption under concentration based regulations. This research focused on quantifying the impacts of individual practices, further studies should be done to quantify the combined impacts of water management practices because many of the practices are likely to be adopted in combination. This research focused on water quality impacts at the WWTF where water quality is typically homogeneous; however, adoption of conservation practices is often heterogeneous and therefore there may be local, site specific impacts that are negative or beneficial and have not been quantified with this study.

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3.0 DEVELOPMENT OF GENERALIZED EMPIRICAL MODELS FOR COMPARING EFFECTIVENESS OF WASTEWATER NUTRIENT REMOVAL TECHNOLOGIES[2]

Overview

The effectiveness of common approaches was quantified at four facilities using mechanistic models. Generalized empirical models were developed by applying statistical methods on the predicted values characterizing nutrient removal as a function of influent wastewater quality. The resulting empirical models provide a framework to estimate and evaluate nutrient removal effectiveness of various approaches and inform systems level decisions on technology adoption. When carbon limited, more sophisticated approaches like 5-Stage Bardenpho and nitrite shunt provide the most notable benefit in nutrient removal efficiency (67% ± 3.3% and 89% ± 2.8% respectively for TN), but little benefit is estimated under non-carbon limited conditions between traditional solutions like A2O and advanced process configurations like 5-Stage Bardenpho (82% ± 2.8% and 85% ± 3.3% respectively for TN). Sidestream physical/chemical processes can provide improvement in nutrient removal efficiency particularly at carbon limited WWTFs, but negligible benefit is estimated with adoption of sidestream biological processes.

3.1 Introduction

Reduction of nutrient pollutants from WWTFs is one of the largest issues faced by most water utility managers (Reardon, et al., 2013). The US Environmental Protection Agency (US EPA) has recognized nutrient pollution, caused by excess nitrogen and phosphorous, as “one of America’s most widespread, costly and challenging environmental problems” (US EPA, 2016). Nutrient pollution originates from a variety of activities both point and non-point. Point source pollution is primarily from urban environments with WWTFs often representing one of the principal sources (WERF, 2010) and, therefore, a primary

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[2] A version of this chapter has been drafted for publication as a research article and will be submitted after successful completion of defense.

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location for implementation of nutrient management practices. For this reason, utility managers require a variety of tools for evaluating and identifying effective nutrient removal strategies.

The issue of nutrient pollution has been widely recognized as a significant issue in coastal estuaries across the US including the Long Island Sound, Chesapeake Bay, Gulf of Mexico, and San Francisco Bay to name a few (Howarth et al., 2002). The extent and severity of nutrient pollution in United States water bodies requires widespread participation from both point and non-point sources to achieve the necessary water quality goals. Currently, nutrient pollution is not regulated at the federal level and many states (coastal and non-coastal) do not have or are still in the process of implementing nitrogen and phosphorous effluent requirements for WWTFs. Presently twenty-three states have established some level of nitrogen and phosphorous regulations, and many states are adopting or increasing the extent of watersheds that are regulated for nutrient pollution (US EPA, 2018). For example, Colorado has implemented nutrient regulations for new WWTFs for annual median total inorganic nitrogen (TIN) of 7 mg N/L and total phosphorous (TP) of 0.7 mg/L (CDPHE, 2012). Wisconsin has established statewide TP limits of 1 mg/L (WDNR, 2011). North Carolina has implemented basin specific load based regulations requiring WWTFs to reduce annual TN and TP discharge loads by as much as 40% and 77% (NCDEQ, 2016) respectively for nutrient sensitive waters, resulting in the need for some WWTFs to treat wastewater to 5.5 mg/L TN and 0.5 mg/L TP (NCDEQ, 2016).

Traditional WWTFs can achieve effluent TN and TP concentrations between 5-30 mg/L and 4-10 mg/L respectively (Metcalf and Eddy, 2003). To achieve low effluent concentrations, WWTFs may need to supplement with chemical addition. It may be necessary to utilize an external carbon source to enhance denitrification (US EPA, 2013) and/or supply sufficient Volatile Fatty Acids (VFAs) necessary to sustain biological phosphorous removal (Ekama, 1986). However, our understanding of biological phosphorous removal is improving on how to foster and promote Phosphorous Accumulating Organisms (PAOs) and ways for WWTFs to develop a source of VFAs through sidestream fermentation and select more efficient PAOs, like *Tetrasphaera*, that can compete in anoxic and even aerobic environments making reliable effluent TP concentrations of < 1 mg/L achievable (Barnard et al., 2017). Alternatively, effluent TP concentrations of < 0.5 mg/L can be achieved with chemical addition of divalent and trivalent metals, like ferric chloride, to precipitate phosphorous (Morse et al., 1998). There are also operational schemes that may yield more effective nitrogen removal like nitrite shunt controls, which requires 25% less aeration and 40% less carbon, compared to traditional nitrification-denitrification (Jimenez et al., 2014). Lastly, WWTFs may adopt sidestream treatment technologies that may provide a more efficient treatment operation and cost-effective way of reducing nutrient loads, decrease the mainstream return load, and potentially recover a valuable nutrient resource for fertilizer. Sidestream technologies include struvite precipitation, ammonia stripping, Centrate and RAS Re-aeration Basin (CaRRB), ANNAMOX, and post-aerobic digestion (PAD) to name a few.

To meet lower effluent concentrations, WWTFs have a variety of treatment approaches that can be considered including mainstream and sidestream nutrient removal technologies. Traditionally, WWTFs will utilize mechanistic models or pilot studies to evaluate the potential effectiveness of these technologies. Mechanistic models of WWTFs use a series of state variables, kinetic parameters, and water quality and process specific inputs to evaluate biological, chemical, pH, gas-liquid and mass transfer reactions. The most common model is the activated sludge model (ASM2), which has been adopted and modified by a variety of software platforms including BioWin™, GPS-X, WEST, STOAT and Simba (WERF, 2003). Utilizing these mechanistic models to evaluate the effectiveness of various nutrient removal technologies is well understood and widely practiced and has been proven to be effective in identifying preferred approaches that warrant adoption or further evaluation through pilot studies (WERF, 2003). However, these mechanistic models are based on complex biological and chemical relationships and can be costly, labor intensive and technically difficult to develop, calibrate and evaluate.

As an alternative to mechanistic models, statistical models have been developed to generalize WWTF processes to identify commonalities and trends in performance. Gori et al. (2011) applied a simple rational procedure for evaluating the impacts of COD and solids fractions on a WWTFs carbon and energy footprint. Suchetana et al. (2016) utilized a hierarchical modeling approach with generalized linear models to estimate the likelihood of permit compliance for biochemical oxygen demand (BOD), total solids suspension (TSS) and ammonia (NH_3) based on various WWTF characteristics. Rivas et al. (2008) developed a mathematical model for optimization of design parameters for WWTF. Despite the need to assess efficacy of WWTF technologies for nutrient removal across multiple facility sizes and unit process configurations, statistical models have not been developed generalizing anticipated effluent TN and TP of WWTFs,

Existing mechanistic and statistical models do not allow for evaluation of a range of treatment technologies and/or widespread adoption at various WWTFs within a watershed. One of the most significant considerations in nutrient removal, particularly denitrification, is the influent water quality and carbon availability (US EPA, 2013), but there is limited research comparing the impacts of influent carbon and nutrient removal across treatment approaches and how that may impact process selection. There is a need for tools that enable assessment of different treatment technologies at a single WWTF and at the systems level considering adoption at multiple WWTFs.

The objective of this research is to investigate trends in nutrient removal efficacy based on process configuration and key explanatory variables. This was accomplished through development of generalized empirical models built from outputs from mechanistic models. Empirical models can be used to readily quantify and compare the effectiveness of common nutrient removal processes at WWTFs. Generalized empirical models can allow for a unique systems analysis approach for watershed scale evaluation of implementing nutrient removal approaches that would be computationally and resource intensive to accomplish through use of traditional mechanistic models. Additionally, generalized empirical models allow for an easy way to evaluate the efficiency of different process configurations accounting for the variations in influent water quality conditions and carbon availability at different WWTFs. Empirical models are not intended to replace mechanistic models or pilot studies, but rather to develop an approach that facilitates widespread evaluation of different WWTFs technology approaches and provide a computationally inexpensive (Rivas et al., 2008) way of comparing common nutrient removal strategies.

3.2 Methods

The generalized empirical models were developed by running many mechanistic model scenarios under different process configurations (Figure 3-1). This exercise was performed using existing models from four WWTFs to characterize the variability associated with adopting these technologies at different locations. The process models were reconfigured to include various nutrient removal processes and evaluated against scenarios based on randomizing key explanatory characteristics that were identified to be significant. Using the model results, a statistical evaluation was performed to generalize the model results to a linear regression empirical model that can be easily applied to other WWTFs. Lastly, the developed empirical models were testing using a fifth WWTF mechanistic model to verify the empirical model provides a reasonable estimate of the mechanistic model.

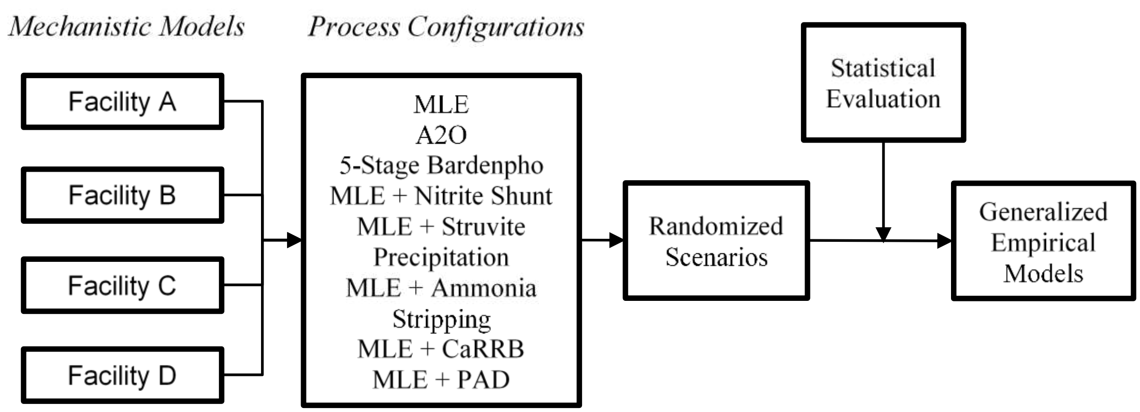


Figure 3-1. Approach for developing generalized empirical models using mechanistic model results

3.2.1 Mechanistic Model

The generalized empirical model development was accomplished using the results of traditional mechanistic models evaluated in BioWin^{TM} 5.0 (EnviroSim Associates Ltd, 2017) using previously calibrated and validated biological process models of four WWTFs tested against a fifth process model. BioWin^{TM} is a broadly recognized tool utilized by utility managers and operators, consulting engineers and researchers for design and evaluation of various wastewater treatment approaches (WERF, 2003; Foley et al., 2010). Developing a BioWin^{TM} model requires a sophisticated sampling and analysis program to obtain or estimate model inputs including but not limited to influent wastewater quality, wastewater fractions, and process kinetics. BioWin^{TM} is used by many wastewater utility managers and in peer reviewed publications to evaluate and compare alternative treatment configurations or identify opportunities for treatment improvements that would be costly and difficult if not impossible to conduct at a WWTF. Five models were obtained from local utilities in Colorado each having unique process conditions (Table 3-1). These process models were developed by the utility or their consultants and have been calibrated and validated against existing process data. The models have been used to evaluate the existing treatment process configuration and various process improvements.

Utilizing multiple process models from different WWTFs provides a range of performance under varying permitted capacity, influent wastewater quality, treatment configuration, kinetic parameters, process controls and effluent water quality. The provided WWTF mechanistic models are configured as common biological nutrient removal treatment arrangements classified as Modified Ludzack Ettinger (MLE), Anaerobic/Anoxic/Oxic (A2O), or 5-Stage Bardenpho (5SBAR). The MLE process configuration achieves nitrification-denitrification, A2O achieves biological phosphorous removal in addition to nitrification-denitrification, and 5-Stage Bardenpho achieves enhanced nitrification-denitrification and biological phosphorous removal.

The key characteristics of the evaluated WWTFs are shown in Table 3-1 including permitted capacity, average flow, mixed liquor return (MLR) rate, return activated sludge (RAS) rate, solids retention time (SRT), hydraulic retention time (HRT), key influent wastewater characteristics, wastewater fractions and biological growth rates. These five WWTFs represent a range of common WWTFs characteristics and treatment approaches. Facilities A-D were used for empirical model development, and Facility E was used to test the empirical models.

Table 3-1. Summary of evaluated facilities influent wastewater quality, process conditions and characteristics, wastewater fractions and biological growth rates

	Facility A	Facility B	Facility C	Facility D	Facility E
Permitted Capacity (MGD)	25.0	23.0	17.0	65.0	160.0
Average Flow (MGD)	14.8	11.8	7.5	30.0	86.0
Influent COD (mg/L)	430.0	448.0	868.0	649.8	663.2
Influent TN (mg/L)	33.9	31.8	47.4	50.7	46.5
Influent TP (mg/L)	4.5	4.0	6.9	8.0	10.0
Influent COD:TN	12.5	14.1	18.3	12.8	14.3
Influent COD:TP	95.6	112.0	126.5	66.3	12.8
Process Configuration	MLE	A2O	5SBAR	A2O	MLE
MLR (%Influent Flow)	111%	200%	200%	213%	51%
RAS (% Influent Flow)	148%	39%	75%	48%	70%
SRT (d)	10	12.5	8	11.8	11.3
Anaerobic HRT (h)	0.0	0.9	1.6	0.3	0.0
Anoxic HRT (h)	2.0	0.9	3.3	1.6	1.4
Aerobic HRT (h)	10.0	6.9	6.5	4.3	5.5
Fraction of non-colloidal slowly biodegradable COD	0.73	0.75	0.75	0.76	0.78
NH3-N:TKN	0.60	0.66	0.61	0.57	0.66
PO4-P:TP	0.47	0.56	0.60	0.50	0.50
AOB Growth Rate (1/d)	0.70	0.90	0.90	0.85	0.90
NOB Growth Rate (1/d)	0.50	0.70	0.70	0.70	0.70
AAO Growth Rate (1/d)	0.10	0.10	0.10	0.20	0.10
OHO Growth Rate (1/d)	3.20	3.20	3.20	3.20	3.20
PAO Growth Rate (1/d)	0.95	0.95	0.95	0.95	0.95

COD = chemical oxygen demand; TN = total nitrogen; TP = total phosphorous; MLR = mixed liquor return; RAS = return activated sludge; SRT = solids retention time; HRT = hydraulic retention time; NH3-N = ammonia as nitrogen; TKN = total kjeldahl nitrogen; PO4-P = phosphate as phosphorous; AOB = ammonia oxidizing bacteria; NOB = nitrite oxidizing bacteria; AAO = anaerobic ammonia oxidizing organisms; OHO = ordinary heterotrophic organisms; PAO = phosphorous accumulating organisms.

Starting in 2013 Colorado has required reporting of effluent TN and TP for WWTFs as monthly grab samples. The mechanistic model effluent was reviewed against the average reported effluent data from these WWTFs (Table 3-2). The modeled effluent TN for Facility A, B and D, was within the standard deviation of the average reported effluent TN, while Facility E the model estimated TN is slightly outside of the reported TN. The mechanistic model for Facility C estimates an effluent TN and TP lower than the average and standard deviation of the reported data. Facility C recently went through significant improvements. The mechanistic model reflects these improvements, but the reported data is prior to these improvements. In terms of TP, the models for Facility A and E provide a higher estimate than the reported average, while Facility B and D estimate TP to be higher than the reported average. This review identifies that the mechanistic models estimated TN are generally consistent with the reported effluent TN and identifies more variability in terms of TP which can be challenging to reliably model due to the sensitivity of the PAOs in practice. It's also important to note that the mechanistic models are developed from a rigorous sampling and analysis plan which provides a better representation of the influent and effluent water quality than monthly grab sample values.

Table 3-2. Reported effluent TN and TP (average, standard deviation, and sample size) compared to BioWin™ estimated effluent

Facility	Reported Effluent TN (mg/L ± STD)	n	Modeled Effluent TN (mg/L)	Reported Effluent TP (mg/L ± STD)	n	Modeled Effluent TP (mg/L)
A	14.9 ± 3.0	12	15.8	2.6 ± 0.5	12	3.1
B	13.9 ± 1.8	20	14.9	2.1 ± 1.2	20	2.0
C	15.8 ± 2.2	19	6.5	2.2 ± 1.2	19	0.2
D	10.3 ± 1.9	22	10.4	0.8 ± 0.6	22	2.0
E	22.2 ± 5.5	43	15.9	2.0 ± 0.8	42	6.8

TN = total nitrogen; TP = total phosphorous; STD = standard deviation; n = sample size

3.2.2 Mainstream Process Configurations

At each WWTF, a series of mainstream nutrient removal approaches including MLE, A2O, and 5-Stage Bardenpho were evaluated. The calibrated process models were reconfigured to represent these three mainstream process configurations while maintaining the calibrated process conditions of the model; primarily influent water quality, process kinetics, RAS, SRT, and solids handling (summarized in Table 3-1). MLE was also evaluated under a nitrite shunt process control which is a variation of the traditional dissolved oxygen (DO) setpoint control. This control strategy is intended to only partially nitrify which reduces the aeration requirements for nitrification and the carbon requirements for denitrification.

3.2.3 Sidestream Process Configurations

The WWTFs were also evaluated considering common sidestream nutrient removal technologies. The evaluated sidestream configurations include struvite precipitation, ammonia stripping, CaRRB, and PAD. To accomplish this the MLE process configuration was revised in BioWin™ as described below.

Struvite Precipitation was modeled by adding a reactor in the model where N and P are removed in the centrate return line. The model components were used to replicate the chemical formation and physical removal of struvite. The model was configured so that the reaction was not limited by pH, magnesium, or reactor size to model an ideal struvite precipitation process.

Ammonia Stripping was modeled comparable to struvite precipitation via the addition of a model reactor in the centrate line to remove N. The model was configured so that the reaction was not limited by pH, airflow, chemical addition or reactor size to model an ideal ammonia stripping process.

CaRRB was modeled by adding a sidestream biological reactor in the centrate return line to model the biological nitrification of ammonia to nitrate. A portion of the RAS was diverted to serve as the mixed liquor supply for the CaRRB process. The RAS rate, reactor size, and aeration rate were adjusted for each WWTF to maximize the nitrification rate, but the process could be limited by pH inhibition. The CaRRB process was evaluated both with and without pH controls to prevent inhibition. The findings indicated negligible improvement to effluent TN or TP with pH control.

PAD was modeled by adding an aerobic digestor downstream of the existing anaerobic digesters. The aerobic digester than models the nitrification of the high ammonia stream discharge from the aerobic digesters. Like CaRRB, the reactor size and aeration rate were adjusted for each WWTF to maximize the nitrification rate, but the nitrification process could be limited by pH inhibition. Similar to CaRRB, PAD was initially evaluated with and without pH controls and negligible benefit in terms of effluent TN or TP was found with the prevention of pH inhibition.

3.2.4 Scenario Development

Using the above described process configurations, randomized scenarios were developed using Microsoft Visual Basic random number generator to randomize key process variables within typical ranges in wastewater treatment to identify sensitive model parameters in determining effluent TN and TP concentrations and nutrient removal efficiency. Preliminary simulations randomized influent COD, TN, TP, alkalinity, temperature, flow (and therefore HRT), SRT, RAS, and MLR. The randomized model parameters were constrained within typical expected ranges from literature (Metcalf and Eddy, 2003; Reynolds and Richards, 2008) as indicated in Table 3-3. Randomized scenarios that resulted in COD:TN ratios outside of 12 to 26 were discarded as extreme. The goal of this work was to develop a simplified empirical model minimizing unnecessary explanatory and insignificant variables. Therefore, preliminary results were reviewed to exclude unnecessary variables that did not significantly impact the estimated percent removal of TN and TP ($p < 0.1$). After this exercise, randomized scenarios were limited based on significant parameters when tested in combination which is discussed more in the following section which included influent COD ($p > 0.05$), TN ($p > 0.05$), TP ($p > 0.05$) for all the modeled facilities and MLR for two of the four modeled facilities ($p > 0.1$). Using these variables, scenarios were developed by randomly varying key parameters using a random number generator within the constrained ranges (Table 3-3) until 20 unique scenarios were randomly created for simulation in BioWin™ at each of the WWTFs for each process configuration. This equates to 80 BioWin™ simulations per process configuration for the empirical model development and a total of 800 BioWin™ simulations to complete the study.

Table 3-3. Randomized parameters for developing BioWin™ simulation scenarios

Randomized Variable	Range
Influent COD	300-1,000 mg/L
Influent TN	25-50 mg/L
Influent TP	4-10 mg/L
Alkalinity	60-120 mg/L as $CaCO_3$
Temperature	3-27 °C
Flow	25-125% Permitted Capacity
SRT	5-20 days
RAS	25%-100% of Influent Flow
MLR	100%-600% of Influent Flow

COD = chemical oxygen demand; TN = total nitrogen; TP = total phosphorous; MLR = mixed liquor return; RAS = return activated sludge; SRT = solids retention time; $CaCO_3$ = calcium carbonate

3.2.5 Generalized Empirical Model Development Considering Uncertainty

The statistical evaluation implemented a regression evaluation using the randomized scenario variables using R statistical software (R Core Team, 2017). Initially, a multivariate linear regression analysis (equation 3-1) was considered using multiples of the preliminary evaluated variables tested independently, in combination ($x_1 + x_2$), for interactive dependencies ($x_1 * x_2$), and as log transformed ($\ln x_1$). As previously discussed, the multivariate linear regression did not identify a consistent statistical combination of multiple explanatory variables for characterizing nutrient removal effectiveness. This is understandable given the complexity of biological interactions and process variables and is hypothesized to be primarily because (1) the combination of influent water quality, process variables, and wastewater kinetics are too complex to be explained in a generalized multiple regression model requiring complex mechanistic models like BioWin™, and (2) the influence of influent COD, TN and TP overshadows the impact of other process variables, e.g. denitrification limitations due to carbon availability as previously discussed (US EPA, 2013). Preliminary modeling identified that MLR tested in conjunction with COD, TN and TP was significant for two of the four evaluated facilities ($p > 0.1$), and for all the evaluated facilities including MLR as explanatory variable as part of a linear regression model did not improve the overall model fit. Therefore, MLR was included as a variable in the process modeling to account for the variability associated with different MLR rates, but MLR was not included as a variable in the empirical model (equations 3-2 and 3-3) because it was not consistently statistically significant when combined with influent COD, TN and TP. Using the simulation results, a single variable linear regression analysis was performed to correlate the estimated TN and TP percent removal as a function of the influent COD:TN and COD:TP respectively. Given the tailing effect of the TN results, a polynomial regression equation was considered, and the statistical evaluation was performed based on the equation 3-2 for TN and equation 3-3 for TP.

$$y = \beta_0 + \beta_1 x_1 + \beta_2 x_2 + \beta_3 x_3 + \cdots \beta_k x_k + \varepsilon \quad (3\text{-}1)$$

$$TN_{(\%)} = \beta_0 + \beta_1 \times (COD{:}TN) + \beta_2 \times (\mathrm{COD{:}TN})^2 + \beta_3 \times (\mathrm{COD{:}TN})^3 + \varepsilon \quad (3\text{-}2$$

$$TP_{(\%)} = \beta_0 + \beta_1 \times (COD{:}TP) + \varepsilon \quad (3\text{-}3)$$

The regression analysis was first performed individually for each WWTF, and then the model results for all WWTFs were treated as a single data set and fit to a generalized model. This approach was done to capture the uncertainty based on the model fit, excluded variables, and variations across WWTFs. The generalized model reflects the average response across the four WWTFs for each of the process configurations.

3.2.6 Model Verification

To verify the generalized models, the empirical models for MLE, A2O and 5-Stage Bardenpho were tested against the mechanistic modeling results for Facility E. This approach was selected to verify that the empirical models provide a reasonable estimate consistent with mechanistic models. Utilizing mechanistic models like BioWin^{TM} is a widely accepted approach to evaluate and compare WWTF process modifications and improvements (WERF, 2003; Foley et al., 2010). To characterize the model performance, two error statistics were calculated; mean relative error (MRE; equation 3-4) and bias fraction (BIAS; equation 3-5) where $\hat{v}_i$ is the validation data and v_i is the estimated data. The MRE represents the average ratio of the empirical model error to the mechanistic model, and the BIAS represents the sum of the empirical model errors divided by the sum of the mechanistic models.

$$MRE = \frac{1}{n}\sum_{i=1}^{n}\begin{cases}\frac{\hat{v}_i - v_i}{\hat{v}_i} \; if \; \hat{v}_i \neq 0 \\ \frac{v_i}{|v_i|} \; if \; |v_i| > 0.001 \times STD(v_i) \\ 0 \quad otherwise\end{cases} \tag{3-4}$$

$$\text{BIAS} = \frac{\sum_{i=1}^{n}(\hat{v}_i - v_i)}{\sum_{i=1}^{n}\hat{v}_i} \tag{3-5}$$

Additional model testing was considered to test the empirical models against observed water quality data. However, there are limitations in data availability to perform this analysis. Influent COD, TN and TP water quality data is not required for regulatory reporting and therefore many WWTFs do not routinely collect this data. Data that is collected can have quality control issues that can include sampling location, sampling method and sample timing resulting in highly variable data that is difficult to statistically correlate. In developing a mechanistic model, a rigorous sampling and analysis plan must be followed which includes diurnal water quality sampling along the process to obtain the data necessary to accurately correlate influent and effluent parameters. This rigorous sampling plan is not routinely performed by WWTFs. Additionally, the empirical modeling considered some process configurations that are not common in practice and evaluated a wide range of influent COD to nutrient ratios to characterize the impacts of carbon limitations and carbon addition. Quality data for this wide range of processes and COD to nutrient ratios were not available and historically carbon addition is not widely implemented as WWTFs have not faced as stringent effluent TN discharge requirements. Mechanistic models provide a tool for evaluating multiple configurations and influent concentration ranges, but there are a limited number of WWTFs that operate under these configurations or ranges that can be referenced for observed data. Because BioWin^{TM} is so widely accepted, testing of empirical models using a calibrated BioWin^{TM} model from an additional facility was considered an acceptable approach. The estimated BioWin^{TM} results were also compared to similar treatment configurations reported effluent concentrations for regulatory compliance to check the relative performance of the empirical models against observed data.

3.3 Results

3.3.1 MLE Empirical Model

The statistical evaluation for the MLE process configuration was performed for each WWTF and the TN and TP removal efficiency were characterized as a function of influent COD:TN (Figure 3-2) and influent COD:TP (Figure 3-3). An analysis of variance identified that the model results of the four WWTFs were statistically different based on a 95% confidence interval which is expected due to WWTF variations; primarily influent wastewater, wastewater fractions and process configuration. The difference in the individual model results helps to account for the variability associated with different process configuration that cannot be captured using a multivariate statistical analysis and requires complex mechanistic modeling. To characterize the average response accounting for the variability and uncertainty across WWTFs, the statistical evaluation was performed for the combined model results (Figure 3-2 and Figure 3-3). The efficiency of TN removal demonstrates a tailing effect as a function of influent COD:TN, reflecting that the effectiveness of carbon addition will decrease as the COD:TN increases (Figure 3-2). Comparing the different WWTFs, the models indicate more variability at low COD:TN ratios and less variability at high COD:TN, suggesting that under non-carbon limited conditions the process is less sensitive to other process parameters. Conversely, the TP removal demonstrates a linear response with the addition to COD:TN (Figure 3-3). For MLE process configurations, the TP removal largely reflects an increase in biological assimilation as growth biological growth increases with additional available carbon. In general, comparing the different processes it can be observed that there is a notable difference in the different technologies under carbon limited conditions and less variability between the different technologies under non-carbon limited conditions.

a

b

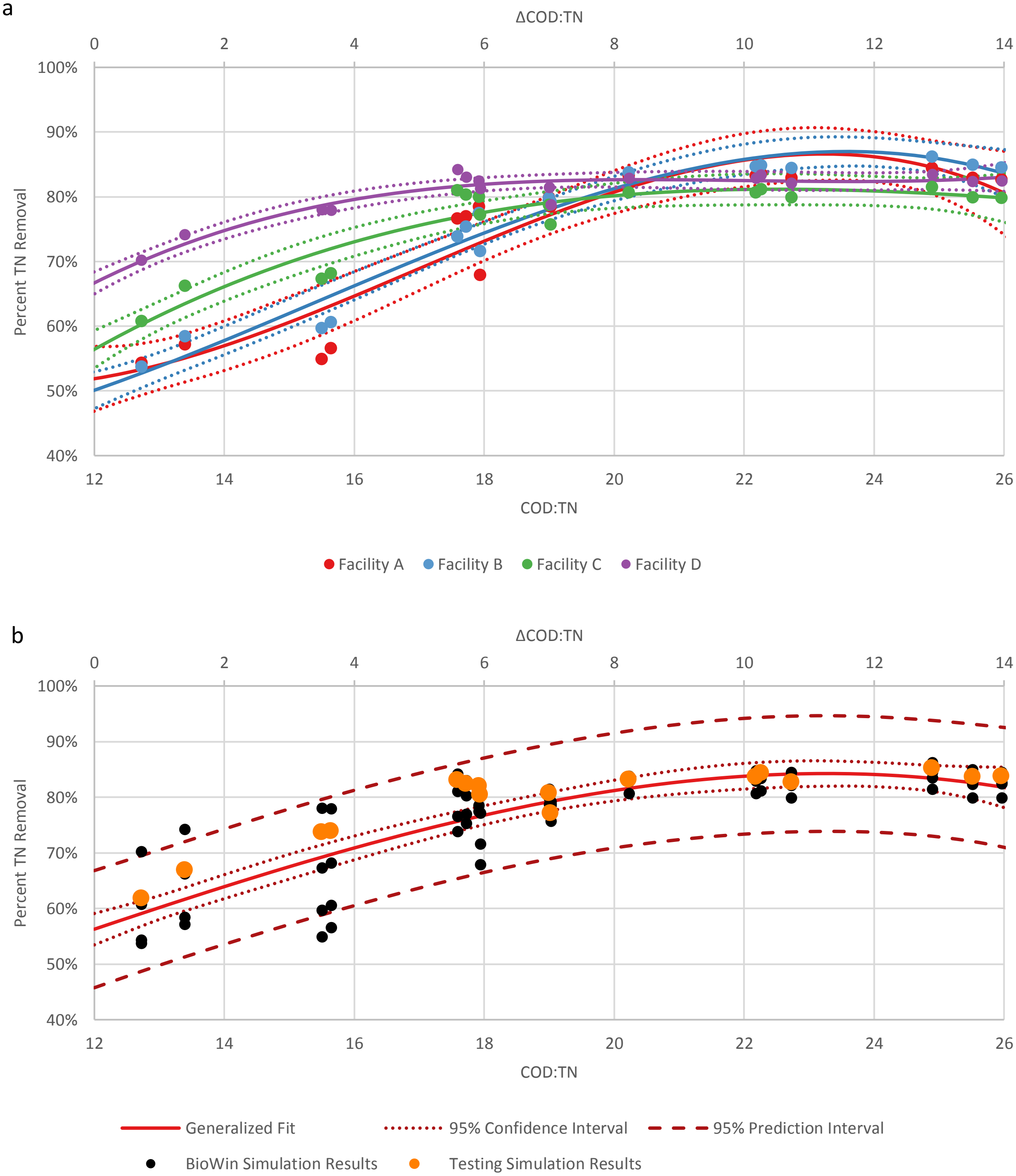
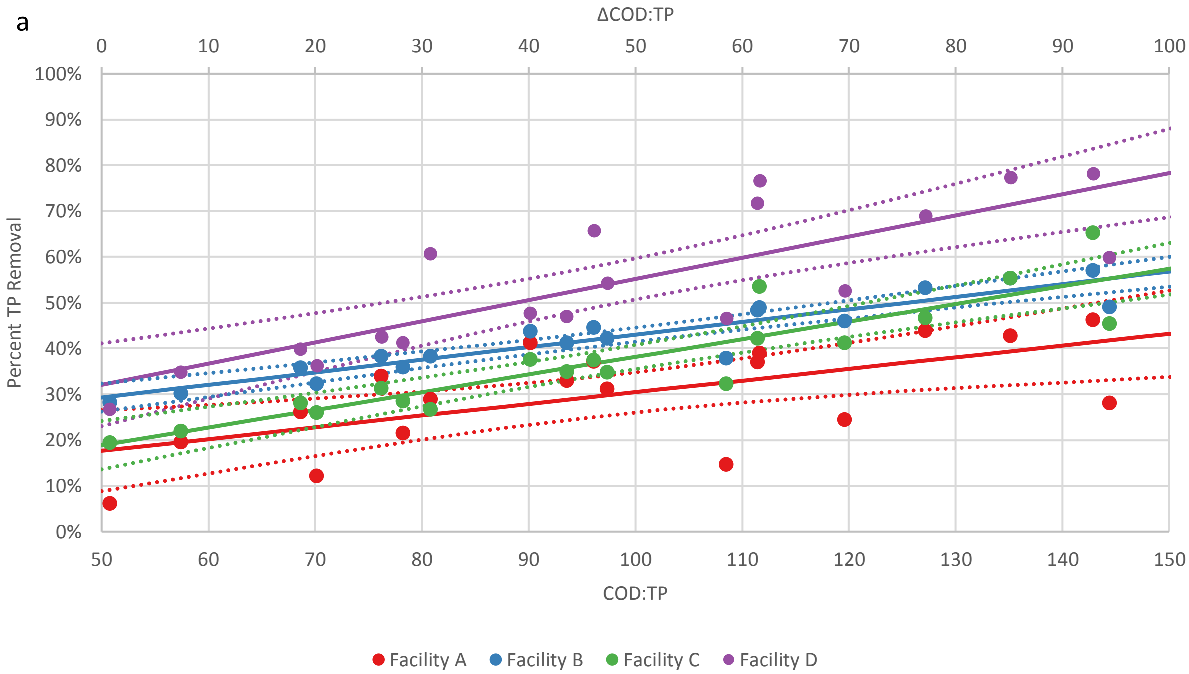


Figure 3-2. a) Individual WWTFs linear regression fit of MLE TN removal efficiency as a function of influent COD:TN with 95% confidence interval (dotted lines) where the points represent the randomized scenario BioWin^{TM} simulation results for the Individual WWTFs, and b) generalized linear model of MLE TN removal efficiency as a function of influent COD:TN with 95% confidence interval and 95% prediction interval where the points represent the combined BioWin^{TM} simulation results



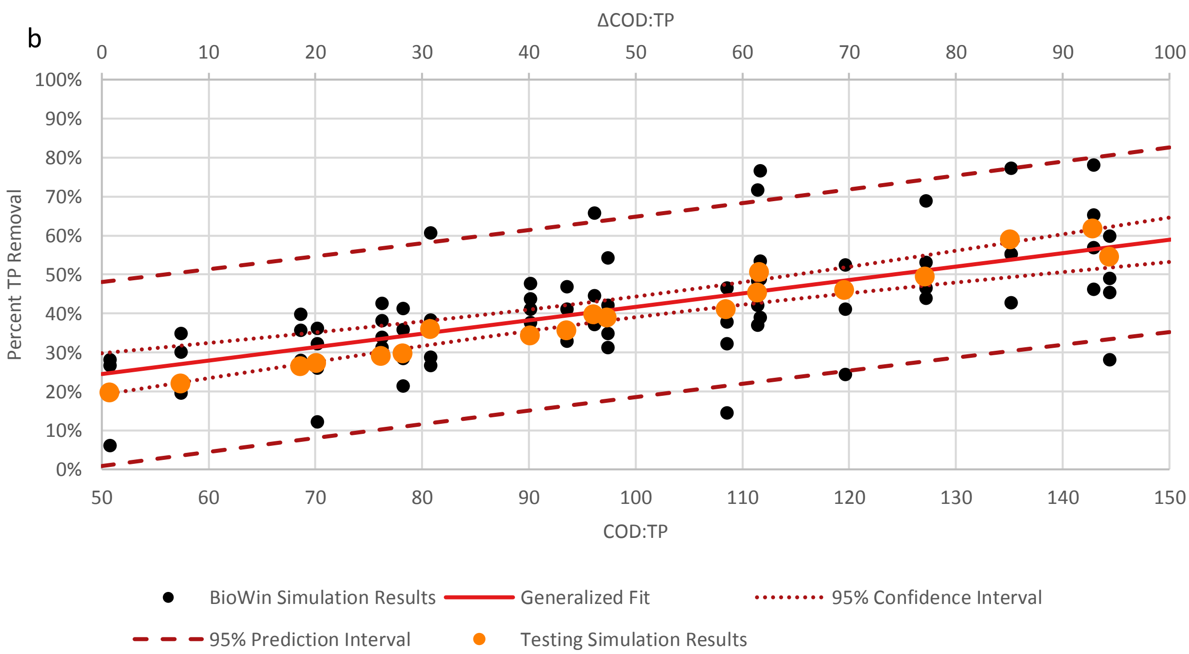


Figure 3-3. a) Individual WWTFs linear regression fit of MLE TP removal efficiency as a function of influent COD:TP with 95% confidence interval (dotted lines) where the points represent the randomized scenario BioWin^{TM} simulation results for the Individual WWTFs, and b) generalized linear model of MLE TP removal efficiency as a function of influent COD:TP with 95% confidence interval and 95% prediction interval where the points represent the combined BioWin^{TM} simulation results

3.3.2 Process Configuration Generalized Empirical Model

The generalized model development approach was repeated for the remaining treatment processes analyzed via mechanistic models to compare TN and TP removal efficiency (Figure 3-4 and Figure 3-5), and generalized equations were developed to represent each process accounting for the uncertainty based on model fit (Table 3-4). These models represent the average response of nutrient removal across the four modeled WWTFs based on the process implemented and influent COD:TN or COD:TP ratio. It is apparent that nitrogen removal efficiency achieved by treatment process is sensitive to COD:TN (Figure 3-4). In addition, in more carbon limited conditions, the relative performance of processes is very different while when COD:TN increases above 20, there performance of the processes becomes more similar (Figure 3-4). The generalized model also demonstrates a clear impact of TP removal as a function of COD:TP (Figure 3-5). As COD increases, there is proportional increase in VFAs which promotes PAO growth.

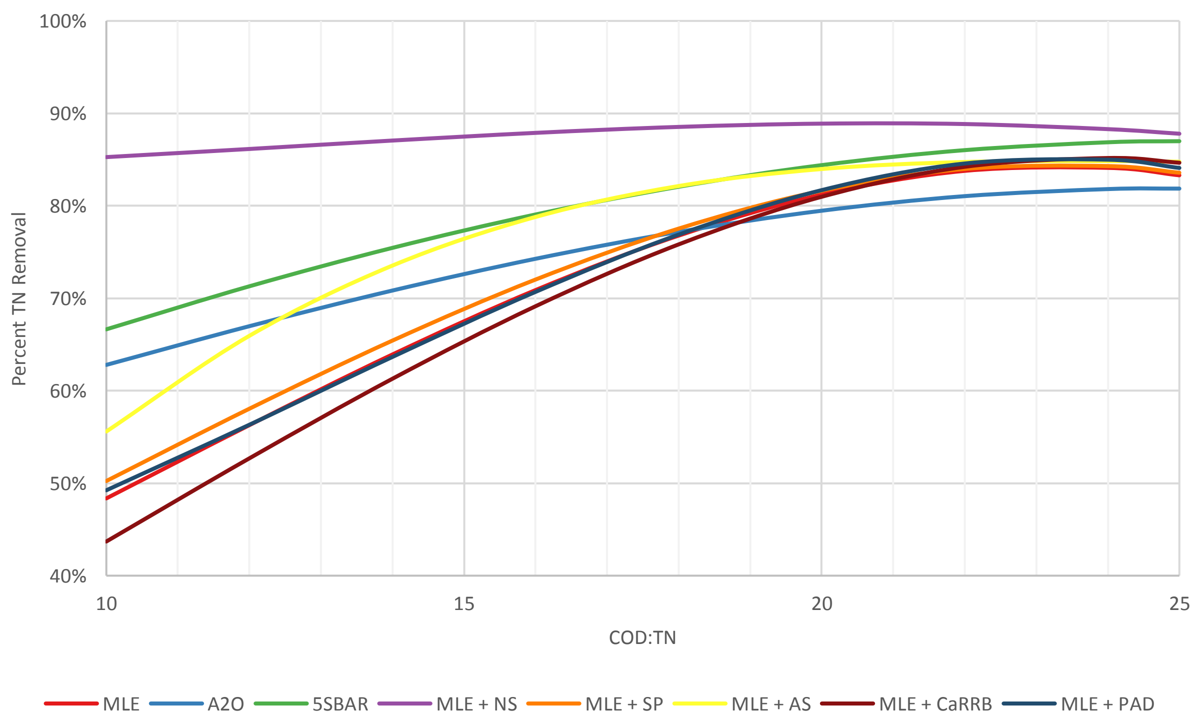


Figure 3-4. Generalized empirical models for comparing TN removal efficiency based on WWTF process configuration

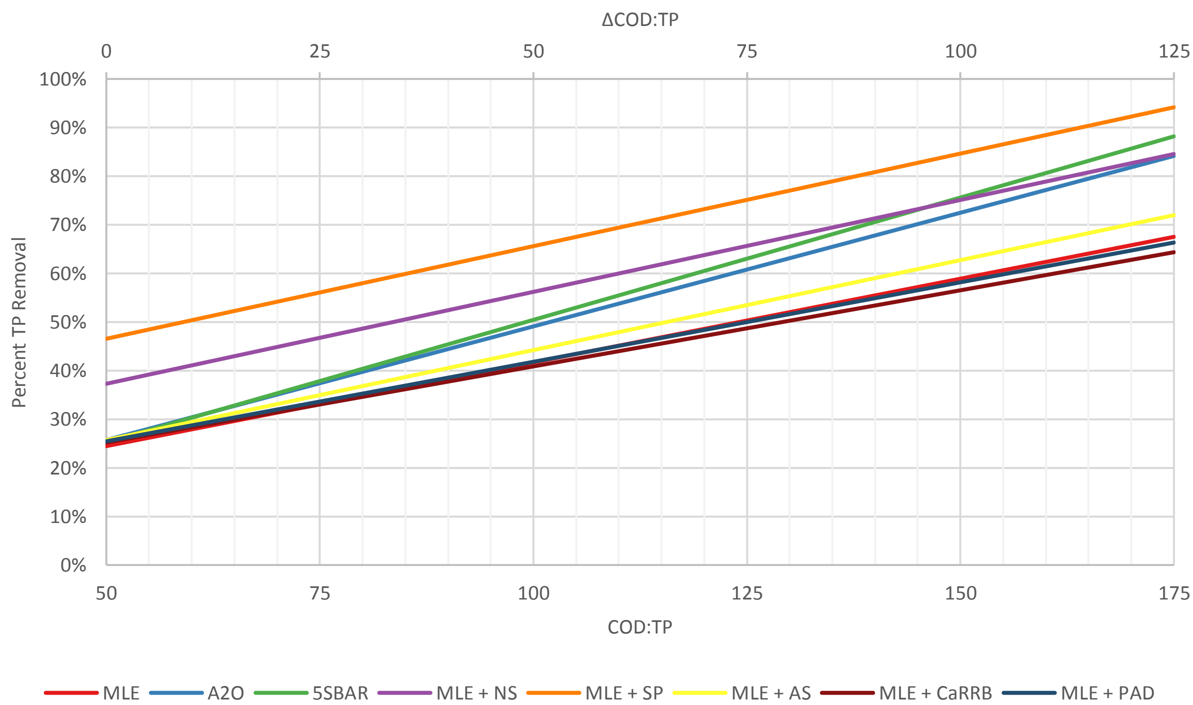


Figure 3-5. Generalized empirical models for comparing TP removal efficiency based on WWTF process configuration

Table 3-4. Generalized empirical models for estimating TN and TP removal based on WWTF process configuration where ε is the empirical model error term and R^2 is the coefficient of determination indicating the model fit performance

WWTF Process	Percent TN Removal β_0	β_1	β_2	β_3	ε	R^2	Percent TP Removal β_0	β_1	ε	R^2
MLE	18.2%	1.24%	0.259%	-0.0082%	5.1%	0.78	7.2%	0.34%	12%	0.40
A2O	40.2%	2.15%	0.030%	-0.0020%	2.8%	0.78	2.3%	0.47%	15%	0.41
5SBAR	37.9%	3.20%	-0.021%	-0.0011%	3.3%	0.72	0.1%	0.50%	16%	0.43
MLE + NS	85.0%	-0.587%	0.083%	-0.0022%	2.8%	0.07	18.4%	0.38%	20%	0.20
MLE + SP	15.6%	2.28%	0.185%	-0.0067%	4.9%	0.77	27.5%	0.38%	6%	0.76
MLE + AS	-52.1%	16.48%	-0.658%	0.0087%	3.9%	0.75	7.2%	0.37%	12%	0.41
MLE + CaRRB	6.2%	2.17%	0.238%	-0.0079%	6.0%	0.78	9.5%	0.31%	10%	0.43
MLE + PAD	36.4%	-1.96%	0.437%	-0.0113%	5.1%	0.79	9.1%	0.33%	11%	0.39

MLE = modified Ludzack Ettinger; A2O = anaerobic, anoxic, oxic; NS = nitrite shunt; SP = struvite precipitation; AS = ammonia stripping; CaRRB = centrate and RAS reaeration basin; RAS = return activated sludge; PAD = post aerobic digestion.

3.3.3 Model Verification

The developed empirical models were tested against a fifth BioWin^{TM} model to verify that the generalized model results provide a reasonable estimate of traditional mechanistic modelling approaches. Using Facility E, the randomized process scenarios were evaluated and the BioWin^{TM} model results were compared against the empirical model estimate (Table 3-5). The testing results indicates that the developed empirical models reasonably reflect the mechanistic modeling results where the MRE for TN was within 4% and the MRE for TP was within 8%.

Table 3-5. Empirical models (n = 20 for each process) tested against Facility E mechanistic model

WWTF Process	**TN**		**TP**	
	MRE	**BIAS**	**MRE**	**BIAS**
MLE	3.8%	3.5%	-6.2%	-2.8%
A2O	-3.7%	-3.7%	-6.3%	-3.8%
5SBAR	-1.3%	-1.2%	-7.3%	-5.0%

MLE = modified Ludzack Ettinger; A2O = anaerobic, anoxic, oxic; 5SBAR = 5-Stage Bardenpho

The general trends between MLE, A2O and 5-Stage Bardenpho identified in the empirical models were also reviewed against the observed trend from the reported effluent concentrations that were categorized into 4 tiers based on treatment type where MLE would be comparable to Tier 2, A2O would be comparable to Tier 3 and 5-Stage Bardenpho would be comparable to Tier 4/5 (Figure 3-6). The percent removal is not known from the data because the influent wastewater quality is not reported or available. The general trend is the same as demonstrated by the empirical model where the effluent TN and TP decreases consistent based on sophistication of treatment type.

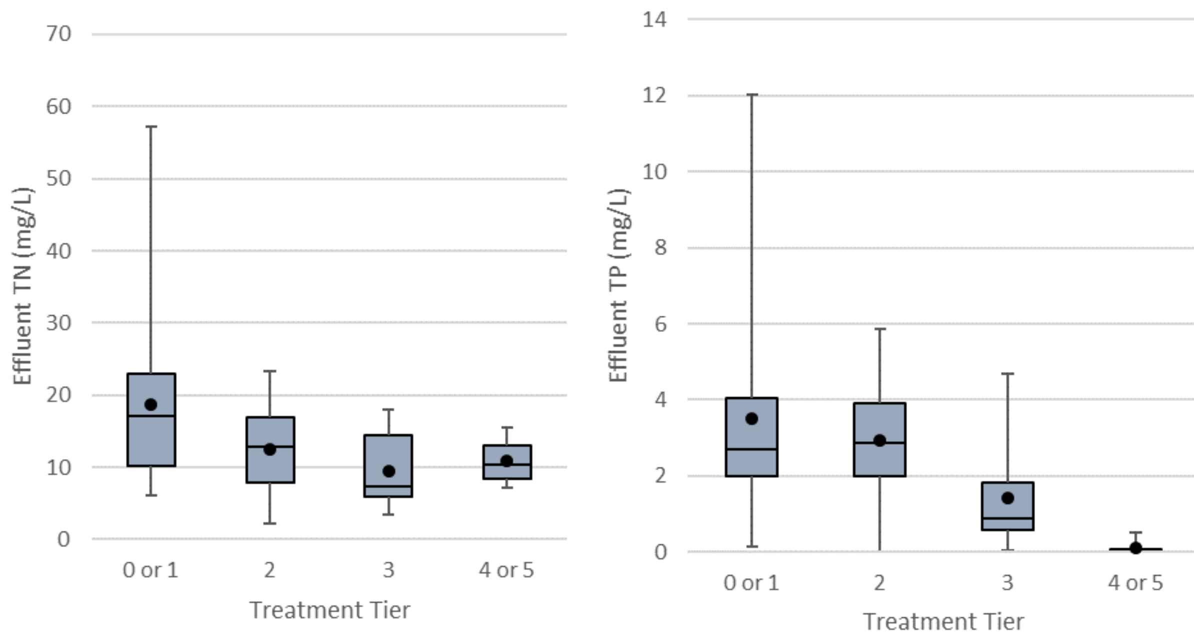


Figure 3-6. Effluent TN and TP based on treatment type reported by Colorado where Tier 0 or 1 represents mechanical treatment with little or no nutrient removal, Tier 2 represents biological nitrogen removal processes like MLE, Tier 3 represents BNR like A2O, and Tier 4 or 5 represents eBNR like 5-Stage Bardenpho

Performance of WWTFs is inherently highly variable over time making consistent estimate of effluent concentrations with mechanistic models challenging (Table 3-2). For example, the observed effluent TN versus the calibrated BioWin™ model effluent TN resulted in an MRE for the 5 WWTFs from -10% to 58% (data not shown). There are a variety of factors that impact the empirical model predicted nutrient removal efficiency depending on variations in process configurations or influent water quality; primarily, the readily biodegradable fraction of COD (rbCOD) that will vary depending on factors like industrial sources, sewer length, and temperature (Gori et al., 2011). However, the shape and slope of the empirical model trend provides a basis for comparison against other treatment process. The developed empirical are effective to identify relative comparisons of efficacy of treatment processes (Figure 3-4 and Figure 3-5), as well as relative performance across varying influent concentrations. Empirical models developed here should be used to provide relative comparisons of process performance across different technologies and ratio of COD to nutrient and are not intended to provide exact estimates of effluent concentration. It can be concluded that the empirical models provide a reasonable estimate of TN and TP percent removal consistent with mechanistic models, and therefore provide a reasonable base for comparing various process configurations while considering the impacts of carbon limitations. While these models provide a good comparison to mechanistic models and a foundation for process comparison, a robust and reliable influent and effluent water quality data set would be necessary to test the model performance against observed data.

3.4 Discussion

Anderson et al. (2002) describes the wastewater treatment process as being "complex, non-linear, with a range of time constants and never being in steady-state." All WWTFs are governed by unique constraints including: collection system characteristics, influent water quality, wastewater fractions, kinetics and nuances in process configuration. These differences make it difficult to readily predict the efficiency of various treatment practices at any given WWTF. The generalized empirical models developed here help bridge this gap providing an estimate of technology efficiency for TN and TP removal based on COD:TN and COD:TP ratios in influent or achieved via carbon addition (Figure 3-4 and Figure 3-5). These results indicate a wide range of process efficiency based on influent wastewater quality and treatment approach (summarized in Table 3-6). The difference in efficiency for mainstream processes is more pronounced under carbon limited conditions as illustrated by the left side of the axis and less pronounced on the right where carbon is no longer the limiting process constraint (Figure 3-4; Table 3-6). In terms of alternative process modifications, the percent improvement was estimated to provide a comparison to the traditional MLE process configuration (Table 3-7) where processes like Nitrite Shunt, Struvite Precipitation, and Ammonia Stripping provide noticeable benefit in terms of TN and/or TP removal efficiency but CaRRB and PAD provide negligible improvement. The importance of carbon to enhance nutrient removal was clearly demonstrated. To avoid chemical carbon addition, WWTFs can assess ways to operate to increase carbon concentrations in the mainstream process. Some considerations include limiting settling of solids in primary clarifiers to allow more carbon to pass through or primary sludge fermentation to recover the carbon removed in the primary clarifiers resulting in readily available VFAs.

Table 3-6. Comparison of predicted nutrient removal range based on mainstream treatment process

Mainstream Process	TN	TP
MLE	48% – 84%	24% – 68%
A2O	63% – 82%	26%- – 84%
5SBAR	67% – 87%	25% – 88%

MLE = modified Ludzack Ettinger; A2O = anaerobic, anoxic, oxic; 5SBAR = 5-Stage Bardenpho

Table 3-7. Comparison of the predicted improvement to nutrient removal MLE efficiency based process modifications

Process Modification	TN	TP
NS	4% – 37%	13% – 17%
SP	0% – 2%	22% – 27%
AS	1% – 10%	1% – 4%
CaRRB	-5% – 1%	-3% – 1%
PAD	0% – 1%	-1% – 1%

MLE = modified Ludzack Ettinger; NS = nitrite shunt; SP = struvite precipitation; AS = ammonia stripping; CaRRB = centrate and RAS reaeration basin; RAS = return activated sludge; PAD = post aerobic digestion.

3.4.1 Nitrogen Removal Efficacy

The generalized empirical models enable comparison of the response of adopting various nitrogen removal approaches or supplementing with additional carbon. The empirical models (Figure 3-4) represent the average response at the four model WWTFs. Traditionally, nitrogen removal is obtained via nitrification-denitrification as demonstrated in the MLE process which had an estimated removal efficiency between 48%-84% (Figure 3-4). The denitrification process can often be carbon limited (US EPA, 2013); therefore, a WWTF may consider carbon addition to improve the treatment efficiency which would shift the estimated TN removal along the x-axis (Figure 3-4) by as much as 36%. At some point, the traditional MLE process will not be carbon limited but limited based on controls and process configuration where additional carbon would provide little benefit. That point of inflection from this study based on the empirical model was estimated around a COD:TN of 20. If the WWTF is carbon limited, literature suggests that methanol can be added at a dose of 3-3.5 mg/L per mg/L of NO3-N removal for denitrification filters (WERF, 2010). There are 1.5g COD/g methanol (WERF, 2010); therefore 4.5-5.25 mg COD addition are necessary per mg of NO3-N removed. However, the generalized MLE model results demonstrate the necessary COD dose will not be linear and will actually require higher carbon doses when targeting TN removal > 85% (25 mg COD per mg of TN removed).

To obtain higher levels of TN removal and/or avoid carbon addition, a WWTF may consider a more advanced multi-stage process like 5-Stage Bardenpho which was modeled to have a TN removal efficiency between 67%-87% (Figure 3-4). Comparing the empirical models for the MLE with the 5-Stage Bardenpho, there is a more notable benefit in adopting a more sophisticated process configuration at a WWTF with low COD:TN ratios (19% increase, left side of x-axis) and is less beneficial as the COD:TN ratio increase (3% increase, right side of x-axis). Again, the actual COD:TN ratio for a given WWTF will be dependent on a variety of factors including influent rbCOD fraction, which would decrease the benefit of adoption of multi-stage processes like 5-Stage Bardenpho at WWTFs with high influent concentrations of rbCOD. Conversely, for carbon limited WWTFs, investing in a 5-Stage Bardenpho process could provide a notable increase in TN removal and potentially avoid or reduce the need for carbon addition (Figure 3-4).

Alternatively, WWTFs may consider process control improvements like nitrite shunt or ABAC for improving nitrogen removal efficiency. Under ideal operation, nitrite shunt can provide an aeration reduction of 25% and potential reduction in COD required for denitrification by 40% (Jimenez et al., 2014). For this study, the nitrite shunt empirical model indicates a generally flat response as a function of COD:TN over the evaluated influent range with a relatively constant TN removal efficiency of approximately 87% (Figure 3-4). This is because the denitration (NO_2 to N_2) process is not carbon limited even at low influent COD:TN ratios. As a result, nitrite shunt provides the most efficient nitrogen removal (85%-89%; Figure 3-4) of the evaluated treatment approaches, even when comparing the benefits of carbon addition and more advanced process configuration like 5-Stage Bardenpho.

To date, there are few installations of full-scale nitrite shunt due to the difficulty in controlling nitrite oxidizing bacteria (NOB) selection. Nitrite shunt has been demonstrated successful at the St. Petersburg Southwest Water Reclamation Facility in Florida achieving effluent TIN of approximately 2.0 mg-N/L (Jimenez et al., 2014). Additionally, research is improving our understanding of the biological mechanisms and process design for the out-selection of NOB, which may make nitrite shunt more viable in the future (Jimenez et al., 2014). Many studies have demonstrated energy savings and improved nitrogen removal with adoption of other process controls like ABAC (Amand et al., 2013). While there is a clear benefit to advanced control strategies like nitrite shunt and ABAC on aeration and nutrient removal, these low DO control strategies may be more susceptible to process upsets, higher emissions of nitrous oxide, and issues with sludge settling (Amand et al., 2013).

Alternatively, WWTFs may adopt sidestream treatment processes for nutrient removal. For nitrogen removal, this may include biological processes (CaRRB or PAD) or physical/chemical processes (ammonia stripping) that were evaluated in combination with a mainstream MLE process configuration. Results indicate that sidestream ammonia stripping, which was modeled as an ideal reaction, could provide a notable improvement on TN removal efficiency achieving as much as 10% additional TN removal with ammonia stripping at low influent COD:TN ratios (Figure 3-4). Like nitrite shunt, the benefit of a sidestream physical/chemical process like ammonia stripping will be more beneficial in low COD:TN wastewater where denitrification is more susceptible to carbon limitations, and negligible at high COD:TN WWTFs where the mainstream nutrient removal is not carbon limited. At low COD:TN ratios the empirical model indicates that sidestream physical/chemical process may provide comparable TN removal efficiency as multi-stage process configurations like 5-Stage Bardenpho (Figure 3-4). Additionally, sidestream physical/chemical processes provide the opportunity for nutrient recovery of ammonia sulfate that can be used as commercial fertilizers (Gustin and Mrinsek-Logar, 2011). While there is a clear benefit with ammonia stripping, it has yet to be successfully demonstrated at WWTFs, but research has shown successful implementation in similar concentrated waste streams (Gustin and Mrinsek-Logar, 2011).

There are many instances of successful installations of struvite precipitation (Lackey, 2018). While struvite precipitation is typically considered for phosphorous removal (discussed more below), a common question is what benefit struvite precipitation has on nitrogen removal given that there is a one to one molar ratio of nitrogen removed and struvite formed. While this is true from a molar perspective, the empirical model results indicate negligible benefit with adoption of struvite precipitation for nitrogen removal (Figure 3-4). This is due to the mass difference between ammonia and phosphate in struvite and the fact that centrate streams are typically more concentrated in nitrogen then phosphorous therefore removing all of the phosphorous load would still result in a return of nitrogen load to the plant.

There are also demonstrated installations of sidestream biological treatment processes. Based on results here (Figure 3-4), inclusion of sidestream biological processes like CaRRB and PAD show little benefit in terms of improvements to effluent nitrogen removal. This is largely because these processes nitrify but do not denitrify and utilize carbon in the sidestream that would be used for denitrification in the mainstream. Since most WWTFs are not hydraulically limited, the modeling results indicate negligible (or even negative) benefit in adopting these practices for purposes of improving nutrient removal efficiency. These processes also require aeration and will suffer from pH inhibition if pH controls are not implemented resulting in additional treatment cost of implementing the sidestream process. However, there could be other benefits in terms of process controls or management.

3.4.2 Phosphorous Removal Efficacy

With the onset of more strict phosphorous discharge limits, many WWTFs historically have implemented chemical phosphorous precipitation (Morse et al., 1998). This can achieve reliable effluent phosphorous concentrations <0.5 mg/L (WERF, 2010). This is a proven approach in wastewater treatment through the addition of divalent and trivalent metals, most commonly iron or aluminum, at the primary clarifier, secondary treatment, or secondary clarifier (Morse et al., 1998). The main benefit of chemical phosphorous precipitation is reliability compared to biological phosphorous removal in achieving target effluent TP concentration. However, chemical phosphorous removal is costly and recent advances in the understanding of biological phosphorous removal and the PAO community pave way for improved success in implementation of biological phosphorus removal. Analyzing the results from the developed empirical models, the MLE process configuration highlights the expected TP removal as a function of influent COD:TP that varies from 24%-68% over the range evaluated (Figure 3-5). This represents little to no enhanced biological phosphorous removal but does show some increase in phosphorous removal as COD increases indicating increased removal as a function of cellular growth for nitrogen removal and some PAO growth. To enhance PAO growth and biological phosphorus removal, traditionally WWTFs will include an anaerobic basin upfront as represented in the A2O and 5-Stage Bardenpho empirical models that results in a steeper response in total phosphorous removal as the COD:TP increases ranging from 25%-88% (Figure 3-5). This is because as the COD increases in the influent, the fraction of VFAs is assumed constant therefore the VFAs also increase and are available to promote PAO growth (Figure 3-5). The model results reflect limitations in adopting an anaerobic basin to promote biological phosphorous removal when insufficient VFAs are present which is more likely in low COD:TP wastewater than in higher COD:TP wastewater.

Results from the nitrite shunt empirical model show that carbon reduction that improves nitrogen removal also improves the TP removal by 13%-17% over the evaluated range (Figure 3-5). As previously discussed, nitrite shunt provides a 40% COD savings which also improved the PAO growth and therefore the biological nutrient removal in the MLE. This was observed by Jimenez et al. (2014) at the St. Petersburg Southwest Water Reclamation Facility in Florida which achieved effluent TP of 0.5 mg/L with nitrite shunt and biological phosphorous removal at DO set point of 0.5 mg/L. This has also been identified in other studies where operating under lower DO concentrations of <1.5 mg/L which can favor PAOs where a DO concentration of >2.0 mg/L can favor glycogen accumulating organisms (GAO) reducing the effectiveness of biological phosphorous removal (Law et al., 2016).

It is important to note that while developed showed a linear response in P removal with increasing COD, and the associated fraction of VFAs, successful implementation of biological phosphorous removal at many WWTFs has indicated that reliable concentrations of <1.0 mg/L-TP are achievable (Barnard et al., 2017). In reality, successful implementation of biological phosphorous removal will act more as a switching function when proper PAO conditions are promoted. Historically, this was thought best through inclusion of an anaerobic basin on the front end of the process (A2O or 5-Stage Bardenpho) promoting the growth of organisms like *Accumulibacter* which function well when adequate VFAs are present but may be less efficient compared to other PAO organisms (Barnard et al., 2017). Many WWTFs may not have sufficient influent VFA concentrations to sustain biological phosphorous removal (Ekama, 1986), potentially requiring VFA addition or process considerations to sustain biological phosphorous removal. In recent years, significant advances have been made in understanding the PAO community, and Barnard et al. (2017) have identified treatment considerations which promote growth of multiple PAO organisms like *Tetrasphaera* fostered through deep anaerobic conditions achieved via sidestream fermentation or possibly even by turning off a mixer in the mainstream. Modeling the efficiency of these treatment processes often under predicts the level of phosphorous removal that is achievable and there are many WWTFs that have demonstrated effluent TP < 1 mg/L with biological phosphorous removal (Barnard et al., 2017). Even still, biological phosphorous removal can be more susceptible to process upsets and requires careful promotion of the PAO community. The ability to perform chemical phosphorous addition in “not to exceed” regulatory structures.

Many WWTFs are considering sidesteream treatment to achieve P removal and recovery. Struvite precipitation can reduce nuisance struvite formation, recover phosphorous, and potentially improve effluent TP concentrations. Based on the model results, comparing the baseline MLE configuration with the addition of sidestream struvite precipitation estimates a 22-27% increase on percent TP removal at the WWTFs (Figure 3-5). Unlike chemical phosphorous, implementing sidestream struvite precipitation allows for a way to recover the removed phosphorous for beneficial use. Of note is that increased biological phosphorous removal will result in elevated phosphorous concentrations in the biosolids. This can result in higher levels of nuisance struvite in the sidestream and may impact land application of biosolids due to high levels of phosphorous. Therefore, as biological phosphorous removal increases, sidestream struvite precipitation would provide a greater benefit in improving the mainstream phosphorous removal, reducing the phosphorous concentration in the biosolids, and recovering a valuable phosphorous product. Many WWTFs in Europe have successfully adopted phosphorous recovery practices, and some countries even require WWTFs to include phosphorous recovery to reduce the dependency on industrial fertilizers (Lackey, 2018).

The other evaluated sidestream technologies are typically considered for nitrogen removal, however there is often the question of what impact removing nitrogen in the sidestream may have on mainstream phosphorous removal. In general, sidestream ammonia stripping, CaRRB or PAD showed little impacts, positive or negative, on phosphorous removal (Figure 3-5). As previously discussed, the success of biological phosphorous removal will be more dependent on sidestream technologies that develop VFAs and promote a diverse PAO community.

3.5 Conclusion

Empirical models were developed through this study that enabled broad comparisons of nutrient removal technologies at WWTFs. The developed empirical models for MLE, A2O and 5-Stage Bardenpho were tested against a mechanistic model and demonstrated an MRE for TN and TP within 5% and 8%, respectively. This demonstrates the developed empirical models provide a reasonable estimate of TN and TP percent removal consistent with mechanistic modeling. The importance of COD:TN ratio to achieving nutrient removal was clearly demonstrated via empirical modes.

To achieve low levels of nitrogen removal, traditional nitrification-denitrification can achieve target effluent requirements >4 mg/L but can likely require carbon addition. Infrastructure improvements, like multi-stage processes, may provide some improvement in nutrient removal (as much as 19% TN removal improvement) but will likely still require carbon addition to achieve low target effluent concentrations. Optimizing this process through nitrite shunt, ABAC, and/or low DO control can reduce the air flow requirements and also minimize or negate the need carbon addition (as much as 37% TN removal improvement). In both cases, the benefit is most notable in low COD:TN wastewaters where denitrification is carbon limited. Successful biological phosphorous removal will be primarily dependent on the influent VFA fraction and most importantly on the promoting a diverse PAO community.

WWTFs may also consider sidestream practices that can provide beneficial improvements on effluent nutrient concentrations and potentially provide a recovered, marketable nutrient product. Most promising, sidestream struvite precipitation can improve TP removal efficiency 22%-27% as well as benefit maintenance, enhance biological phosphorous removal efficiency, improve effluent water quality, reduce the phosphorous concentration in the biosolids, and provide a beneficial fertilizer source reducing the use and import of synthetic fertilizers. Other technologies like ammonia stripping, can improve the TN removal efficiency as much as 10% while currently technologically limited, could provide similar benefits in terms of nitrogen recovery as process technology improves. However, sidestream biological processes that convert ammonia to nitrate, and do not remove nitrogen, showed little to no benefit for effluent TN concentrations.

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4.0 ASSESSING COST-EFFECTIVE NUTRIENT REMOVAL SOLUTIONS IN THE URBAN WATER SYSTEM[3]

Overview

Many states are adopting more stringent nutrient load requirements resulting in the need for utilities to invest in costly infrastructure improvements. Much work has been done to assess the efficacy of wastewater treatment technologies and stormwater control measures for nutrient reduction potential. The analysis presented here provides a unique assessment of combinations of nutrient load reduction strategies across the water supply, wastewater and stormwater sectors. A demonstration study was conducted evaluating 7,812 cross sector nutrient removal strategies in the urban water system using empirical models to quantify efficacy of common wastewater treatment, water management and stormwater control measures (SCMs). To meet stringent nutrient concentration or load requirements, wastewater treatment facilities will likely require advanced biological nutrient removal with carbon and ferric addition. Even with these technologies implements, wastewater treatment facilities (WWTFs) may still be unable to obtain satisfactory nutrient removal to meet downstream nutrient targets. In addition, municipalities can consider water management practices and SCMs to further reduce nutrient loading or provide a more cost-effective nutrient removal strategy. For water management practices, source separation and effluent reuse were frequently identified as part of the most effective nutrient strategies, but face engineering, political and social adoption barriers. Similarly, SCMs were frequently part of effective nutrient removal strategies compared to only adopting nutrient removal practices at the wastewater treatment facility. This research provides the framework and demonstrates the value in utilizing an urban water system approach to identify optimal nutrient removal strategies that can be easily applied to other urban areas.

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[3] A version of this chapter has been drafted for publication as a research article and will be submitted after successful completion of defense.

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4.1 Introduction

The issue of nutrient pollution is recognized as one of the most significant environmental issues today (US EPA, 2016). Many states have recently adopted or are in the process of adopting regulations to reduce nutrient loading into watersheds (US EPA, 2018). Nutrient pollution originates from a variety of activities including point sources, urban nonpoint sources, rural nonpoint sources, and atmospheric deposition (Daigger et al., 2014). In an urban water system, nutrient pollution is primarily from wastewater treatment facilities (WWTFs) and stormwater runoff. Improving nutrient removal at WWTF may have substantial capital and operational cost implications (Daigger et al., 2014), therefore it is important to identify the most cost-effective strategies for meeting nutrient reduction targets considering WWTF process improvements as well as other solutions in the urban water system, like water management practices and stormwater control measures (SCMs).

Traditionally, reduction of nutrient from the urban water system is primarily achieved by implementing process improvements at WWTFs. However, it may be necessary or more cost effective to consider implementing a combination of water management practices, wastewater technologies and SCMs throughout an urban water system. The most effective nutrient removal strategies will vary based on a variety of factors including: density and size of the urban environment, the type and degree of commercial and industrial applications, impervious area, and natural background nutrient loads originating from upstream areas in the watershed. In some areas, the contributions from nonpoint sources are so significant that even complete elimination from point sources would still not meet the target nutrient requirements (WERF, 2010) and adopting nutrient removal strategies only at WWTF may not meet target stream concentrations (Son and Carlson, 2012). Therefore, it is important that a nutrient load reduction strategy considers application of practices across the urban water system.

Sources of nutrient pollution and strategies for reducing nutrient loading have been evaluated in a variety of watersheds, most notably in coastal estuaries, across the US including the Long Island Sound, Chesapeake Bay, Gulf of Mexico, and San Francisco Bay to name a few (Howarth et al., 2002; Falk et al., 2015). Various approaches have been applied to evaluate the individual effectiveness of wastewater technologies and SCMs. For WWTFs, there are a variety of treatment approaches and process configurations that can be considered. These configurations are typically evaluated with mechanistic models that use a series of state variables, kinetic parameters, and water quality and process specific inputs to evaluate biological, chemical, pH, gas-liquid and mass transfer reactions (WERF, 2003). To reduce the level of treatment necessary at WWTFs, water management practices can be adopted like source separation of urine or graywater reuse to reduce influent nutrient load to a WWTF, or WWTF effluent reuse to reduce the effluent flow and associated nutrient load directly discharged to the receiving water body. McKenna et al. (2018) and Hodgson et al. (2018) utilized a mass balance analysis combined with mechanistic modeling to quantify the impacts of common water management strategies on WWTF performance and downstream water quality identifying a benefit to downstream water quality when adopting source separation, graywater irrigation reuse or source separation. For stormwater, pollutant load and SCM effectiveness is typically quantified using the Simple Method or computer-based models like EPAs Stormwater Management Model (SWMM) or the Hydrologic Simulation Programming-FORTRAN (HSPF) model (Schueler, 1987; Ohrel, 2000). Both models have been implemented effectively where the simple method provides an estimate with fewer inputs while the computer-based models provide a higher level of accuracy but require more data for inputs and model calibration.

The existing tools and modeling approaches for evaluating the effectiveness of a nutrient removal practice are data intensive and time consuming (WERF, 2010). Additionally, the modeling efforts focus on evaluating nutrient removal from a single sector, i.e. either stormwater or wastewater. Limited research has assessed efficacy of nutrient removal across sectors within the urban water system. An urban water system evaluation framework is needed to enable cross-sector analysis to identify optimal nutrient removal strategies considering water management practices, wastewater treatment technologies and SCMs.

The objectives of this research are to investigate the most effective nutrient removal strategies considering adoption of water management practices, WWTF process configurations and SCMs and to explore tradeoffs between cost and efficiency for removing both nitrogen and phosphorous. To accomplish this, an urban water system analysis framework was developed to identify most effective nutrient removal strategies for meeting target loading conditions by implementing empirical models and estimating unit cost to quantify and compare the effectiveness of individual and combinations of nutrient removal strategies. The approach developed here provides a framework to assess and compare many nutrient removal practices and combination of strategies that would be computationally intensive using traditional modeling approaches.

4.2 Methods

4.2.1 Urban Water System Analysis

The urban water system analysis approach was accomplished by utilizing empirical models to estimate nutrient removal effectiveness of a given technology and assuming a unit costs as a function of percent adoption to determine the total cost and cost per pound of nutrient load reduced. Previously developed empirical models (described below) were used such that inputs could be readily obtained for a study area and scenarios developed to evaluate combinations of nutrient removal strategies across the urban water system (Figure 4-1).

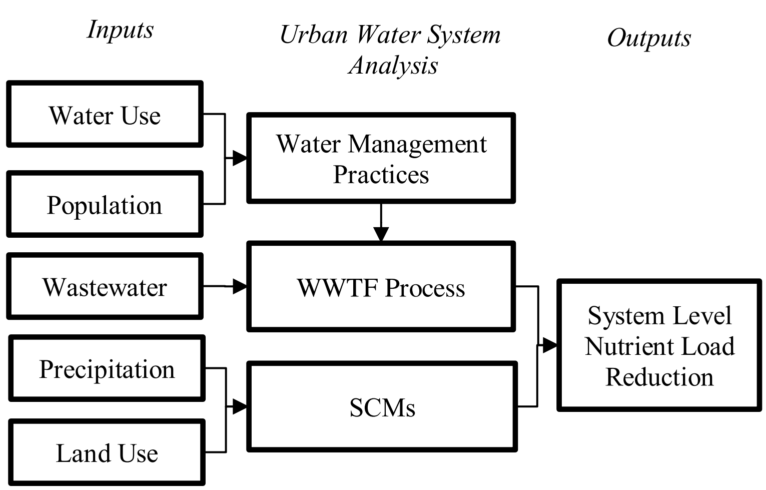

Figure 4-1. Overview of Urban Water System Cross-Sector Analysis Framework for Evaluating and Identifying Effective Nutrient Removal Strategies (WWTF = wastewater treatment facility).

Practices included in the analysis were:

- **Water Management Practices** - Urine Source Separation (SS), Graywater Irrigation Reuse (GIR), WWTF Effluent Reuse (WR)
- **WWTF Process Configurations** - Existing Process, Modified Ludzack Ettinger (MLE), MLE + Nitrite Shunt (NS), MLE + Carbon Addition (CA), MLE + Ferric Addition (FA), MLE + CA + FA, MLE + Struvite Precipitation (SP), MLE + Ammonia Stripping (AS), MLE + Centrate and RAS Reaeration Basin (CaRRB), MLE + Post Aerobic Digestion (PAD), Anaerobic, Anoxic, Oxic (A2O), and 5-Stage Bardenpho (5SBAR)
- **SCMs** - Extended Detention Basin (EDB), Bioretention (BR)

Scenarios were developed combining the practices which included one of twelve WWTF process arrangements, none or one of three water management practices, and none or one of two SCMs. Water management practices were evaluated at 5% adoption intervals from 0%-50%, and SCMs were evaluated at 10% adoption intervals from 0%-100%, therefore a total of 7,812 scenarios were evaluated for achieving nutrient reduction targets.

Evaluating combinations of these practices was accomplished by synthesizing previously developed modeling approaches from each urban water system. The modeling approach for each urban water system was developed using means and methods consistent with standard practice and calibrated and validated to measured data where possible (Hodgson et al., 2018; Hodgson and Sharvelle, 2019; Schueler 1987).

4.2.2 Water Management Practices

Previous studies completed by McKenna et al. (2018) and Hodgson et al. (2018) quantified the impact of indoor conservation, source separation, graywater toilet reuse, graywater irrigation reuse and WWTF effluent reuse on wastewater treatment efficiency, effluent water quality and downstream nutrient loading using a calibrated wastewater modeling (BioWin^{TM}) tested against measured data. BioWin^{TM} is widely used to evaluate WWTF performance based on various process simulations (WERF, 2003; Foley et al., 2010) The impact of indoor conservation and graywater toilet reuse had negligible impacts on WWTF performance and effluent water quality, so those management practices were not evaluated here. Linear regression relationships were obtained (Hodgson et al., 2018) for source separation and graywater irrigation as a function of population adoption (equation 4-1; Table S-1 in Appendix A) and effluent reuse as a function of percent flow reused (equation 4-2; Table S-1 in Appendix A

$$\Delta L_{EFF} = \beta_0 + \beta_1 \times (\%POP) + \beta_2 \times (\%POP)^2 \tag{4-1}$$

$$\Delta L_{EFF} = \beta_0 + \beta_1 \times (\%REU) \tag{4-2}$$

4.2.3 WWTF Processes

To enable comparative adoption of WWTF processes, a study was completed to quantify to develop generalized regression relationships for readily evaluating and comparing technology effectiveness using four calibrated wastewater models (BioWin^{TM}). The models were obtained from different utilities and have been developed and calibrated to reasonably represent the WWTF performance based on measured data. The model results were used to develop simplified empirical models to facilitate systems level evaluations of wastewater treatment performance while accounting for uncertainty across WWTF based on variations in process characteristics, influent wastewater characteristics and operational controls. The empirical models were tested against a fifth wastewater model and verified against measured effluent data to characterize the model BIAS and MRE error (Hodgson and Sharvelle, 2019). The 95% prediction interval was used to verify that the empirical models reasonably estimate nutrient removal under various process configurations while accounting for the uncertainty due to excluded variables, variations across WWTFs and model fit. The generalized models estimate the percent TN and TP removal is characterized as a linear regression relationship (equation 4-3 and 4-4 respectively).

$$TN_{(\%)} = \beta_0 + \beta_1 \times (COD{:}TN) + \beta_2 \times (\mathrm{COD{:}TN})^2 + \beta_3 \times (\mathrm{COD{:}TN})^3 \tag{4-3}$$

$$TP_{(\%)} = \beta_0 + \beta_1 \times (COD{:}TP) \tag{4-4}$$

The parameters for Equation 3 and 4 are presented in Table S-2 in Appendix A. For TN and TP removal with CA, biological phosphorous removal (A2O and 5SBAR), and FA the percent removal was assumed based on literature reported achievable removal efficiencies due to limitations in process modeling. The achievable TN removal was assumed 95% for CA and the achievable TP removal was assumed 90% for

FA (USEPA, 2010). For biological phosphorous removal, 80%-90% TP removal is achievable (USEPA, 2010) therefore TP removal was assumed to be 85% for A2O and 90% for 5SBAR.

4.2.4 Stormwater Control Measures

The impacts of SCMs on nutrient loading were estimated based on the Simple Method (Schueler, 1987) which is a widely accepted method for estimating pollutant load based on land to calculate the urban stormwater loads with and without implementation of various SCMs where nutrient loading is calculated based on equation 4-5.

$$L = 0.1 * P * Pr * Rv * A * C \qquad (4\text{-}5)$$

Where: *L* is pollutant load in kg, *P* is precipitation in cm, *Pr* is fraction of precipitation that produces runoff, *Rv* is runoff volume coefficient, *A* is drainage area in hectares, *C* is event mean pollutant concentration as mg/L and 0.1 represents the unit conversion.

A *Pr* value of 0.9 was assumed (Schueler, 1987), and *Rv* was calculated as a function of percent impervious to account for the amount of precipitation that results in runoff as indicated in equation 4-6 (Schueler, 1987):

$$Rv = 0.05 + 0.009 * I \qquad (4\text{-}6)$$

Where: *I* is the percent imperviousness.

To conduct the analysis, current land use areas are obtained for the study area from the local municipality(s). The percent imperviousness for each land use polygon is calculated using the 2011 USGS National Land Cover Database (USGS, 2014), and the land use types are generalized to open space, residential, commercial, industrial, institutional or highway. Lastly, median runoff concentrations for TN and TP are assumed from literature reported measured runoff concentrations based on land use classification (Wright Water Engineers, Inc. et al., 2013; Table S-3 in Appendix A).

To account for the change in pollutant load with the implementation of SCMs, Equation 5 was adapted to apply a percent removal of nutrient loads by SCMs to percentages of impervious area treated. SCMs were not considered to reduce the load for pervious areas. equation 4-7 shows the adaption which accounts for the load from the impervious area treated by the SCM and the load from the area not treated by the SCM where the treated area is calculated based on equation 4-8, the not treated area is calculated based on equation 4-9, and the percent impervious area for the not treated area is adjusted based on equation 4-10.

$$L = 0.1 * P * Pr * C * (A_{NT} * (0.05 + 0.009 * I_{NT}) + A_T * (0.05 + 0.009 * 100\%) * R_{\%}) \tag{4-7}$$

$$A_T = T_{\%} * A * I \tag{4-8}$$

$$A_{NT} = A - A_T \tag{4-9}$$

$$I_{NT} = \frac{A*I - A_T*100\%}{A_{NT}} \tag{4-10}$$

Where, A_T represents the area impervious area that is treated by the SCM, $T_{\%}$ represents the percent of impervious area treated by the SCM, A_{NT} represents the remaining area not treated by the SCM, I_{NT} represents the percent impervious of the non-treated area and the $R_{\%}$ is the assumed percent removal from a SCM based on values from literature (CWP, 2007).

For BR the percent removal for TN and TP was assumed to be 46% and 5% respectively, and for EDB the TN and TP percent removal was assumed to be 24% and 20% respectively (CWP, 2007). The two SCMs evaluated were bioretention basins (BR) extended detention basins (EDB). The percent of impervious area implementing SCMs ($T_{\%}$) was evaluated from 0% to 100% adoption at 10% adoption intervals. For this analysis, the baseline condition assumes no SCMs are present.

4.2.5 Cost Analysis

For each urban water management solution, annualized unit costs were developed to assess cost effectiveness of a scenario. The annual unit cost analysis approach was adapted based on the USEPA Guidelines for Preparing Economic Analysis (US EPA, 2014). A Present Value Cost (PVC) was calculated accounting for fixed cost (FC), annual operating and maintenance cost (MC), opportunity cost (OC), irregular cost (RC), and reduced cost (S) over the life of the project discounted to account for the time value of money (equation 4-11). The PVC represents the total cost to install and maintain the various practices for the practice life which was assumed to be 20 years.

$$PVC = FC + \sum_{t=1}^{n} \frac{1}{(1+r)^t} [MC_t + OC_t + RC_t - S_t] \tag{4-11}$$

Where: PVC is present value cost, FC is fixed cost incurred with initial practice adoption, t is the annual period, n is the final period (assumed 20 years), r is the interest rate (assumed 4.3%), MC is the yearly maintenance costs, OC is the opportunity costs, RC is the irregular costs, and S is the reduced costs.

The PVC were primarily developed referencing literature cost data (USEPA, 2015; USEPA, 2010; CDM, 2007; Ishii and Boyer, 2015; WSTB, 2015). When cost information from literature was limited, engineering costs estimates were developed for practice adoption. The development of the unit costs is documented in Appendix A. The PVC costs were developed as a unit cost (per gpd, per capita, or per acre) so the PVC estimate could be applied at different levels of adoption and treatment scales. All costs were adjusted to 2018 dollars based on the Consumer Price Index (CPI) by multiplying by the CPI ratio as indicated in equation 4-12. When necessary future costs were adjusted to account for inflation at an assumed rate of 2.5%.

$$CPI\ Ratio = \frac{CPI_{2018}}{CPI_t} \tag{4-12}$$

4.2.6 Demonstration Area

The developed systems approach was demonstrated using Fort Collins, CO as the study area. Fort Collins is an urban area with a population of 143,986 (2010 US Census), a mix of residential, commercial and open space areas, and is served by two WWTF's (Table 4-1). Per capita water use was estimated based on (Hodgson et al., 2018). The effluent nutrient concentration and flow from the WWTFs and the upstream nutrient load was obtained from 2014 effluent water quality reported for regulatory compliance and obtained from USEPA Water Quality Portal. Additional influent WWTF water quality data was obtained from the wastewater utility. The average annual precipitation data was obtained from NOAA as an annual average precipitation and assumed evenly distributed throughout. This assumption is acceptable at smaller scales, but the spatial distribution of precipitation will be more significant as the evaluation scale is increased. Land use boundaries were obtained from the City and generalized to open space, residential, commercial, industrial, institutional and highway.

Table 4-1. Model Inputs and Assumptions

Water Management Practices	Value	Units
Population	143,986	people
Graywater Generation	94.6	L $Capita^{-1}$ d^{-1}
WWTF	**Value**	**Units**
Average Flow (Total)	0.54	m^3 s^{-1}
Permitted Flow (Total)	1.26	m^3 s^{-1}
Influent TN (Average)	32.0	mg L^{-1}
Influent TP (Average)	4.0	mg L^{-1}
Influent COD:TN	14.0	Ratio
Influent COD:TP	112.0	Ratio
Effluent TN (Average)	11.9	mg L^{-1}
Effluent TP (Average)	2.0	mg L^{-1}
Current LCC Treatment Cost	0.79	\$ / L d^{-1}
SCMs	**Value**	**Units**
Average Annual Rainfall	40.8	cm yr^{-1}
Open Space	2,891	hectares
Residential	8,825	hectares
Commercial	2,374	hectares
Industrial	620	hectares
Institutional	308	hectares
Highway	122	hectares

4.2.7 Target Conditions

Historic water quality data and receiving water body flows were reviewed for the demonstration area to identify existing conditions and acceptable stream loading conditions and is included as part of Appendix A. The state of Colorado regulates annual TN and TP loading based on the annual median of the daily average flows with an allowable 1-in-5 year exceedance interval. Based on the last 10 years of data (USGS Gauge No. 06752280), this would be the 2009 annual median flow of 0.31 m^3/s, which represents the second driest year in the period. Stream concentrations were obtained upstream of the demonstration area, between the two WWTFs, and downstream of the demonstration area from data reported to the state of Colorado in 2014 which is a recent requirement with the adoption of statewide nutrient regulatory requirements.

4.2.8 Multi-objective Optimization

Each strategy includes three outputs; PVC, TN load, and TP load. Using these three outputs, a multi-objective optimization was performed to minimize these outputs identifying the most effective strategy exploring tradeoffs in reduction TN versus reduction in TP versus PVC. This was accomplished by determining the Pareto optimal front for all strategies which identifies non-dominated solutions where no one variable can be improved without degrading one or both of the other variables. The multi-objective optimization was performed using the Pareto ranking method developed by Zitzler and Thiele (1999). This analysis provides a way to focus in on the optimal solutions to allow for a more focused evaluation of effective nutrient removal strategies.

4.3 Results

Based on the 2009 median streamflow, the downstream loading capacity for TN and TP is 45,292 and 3,831 kg per year respectively (Table 4-2). The 1-in-5 year exceedance median represents a much lower loading capacity than the average streamflow due to the highly variable receiving stream flows. Due to this large variability, four different target removal levels were considered representing four tiers of nutrient removal required (Table 4-2).

Table 4-2. Target Nutrient Loading for Total Nitrogen and Total Phosphorous

Tier	Streamflow Condition	TN Load kg yr^{-1}	TP Load kg yr^{-1}
1	30% Improvement from Existing Nutrient Load	209,991	30,470
2	2009 Average Flow, 2.50 m^3/s	184,158	15,576
3	Average of 10-yr Annual Median Flows, 1.88 m^3/s	119,230	10,084
4	2009 Median Flow, 0.31 m^3/s	45,292	3,831

A total of 7,812 nutrient strategies were evaluated to characterize the effluent TN loading, TP loading and PVC (Figure 4-2). The multi-objective optimization was performed to identify optimal scenarios considering tradeoffs in minimizing PVC, TN discharge and TP discharge. Of the 7,812 evaluated strategies, a total of 176 optimal non-dominated solutions were identified in the Pareto front (indicated in color on Figure 4-2). Of the non-dominated solutions, viable strategies were identified that meet the defined nutrient targets developed based on the historical flow analysis and baseline loading conditions (Table 4-2). Since the evaluation defines both TN and TP loading conditions, there are limited optimal nutrient removal strategies capable of meeting the defined nutrient targets.

A 30% improvement in the existing loading conditions requires the least amount of nutrient removal improvement, and therefore results in the largest number of viable strategies that meet the target load (Figure 4-2). Based on the 2009 average flow, there are still a variety of strategies capable of meeting the target TN and TP requirements. However, as the target load is decreased there are no viable strategies that meet the target based on average 10-year median flow target or the 2009 median flow target. To meet these targets, background nutrient loading would need to be addressed in addition to the removal strategies evaluated in the urban water system.

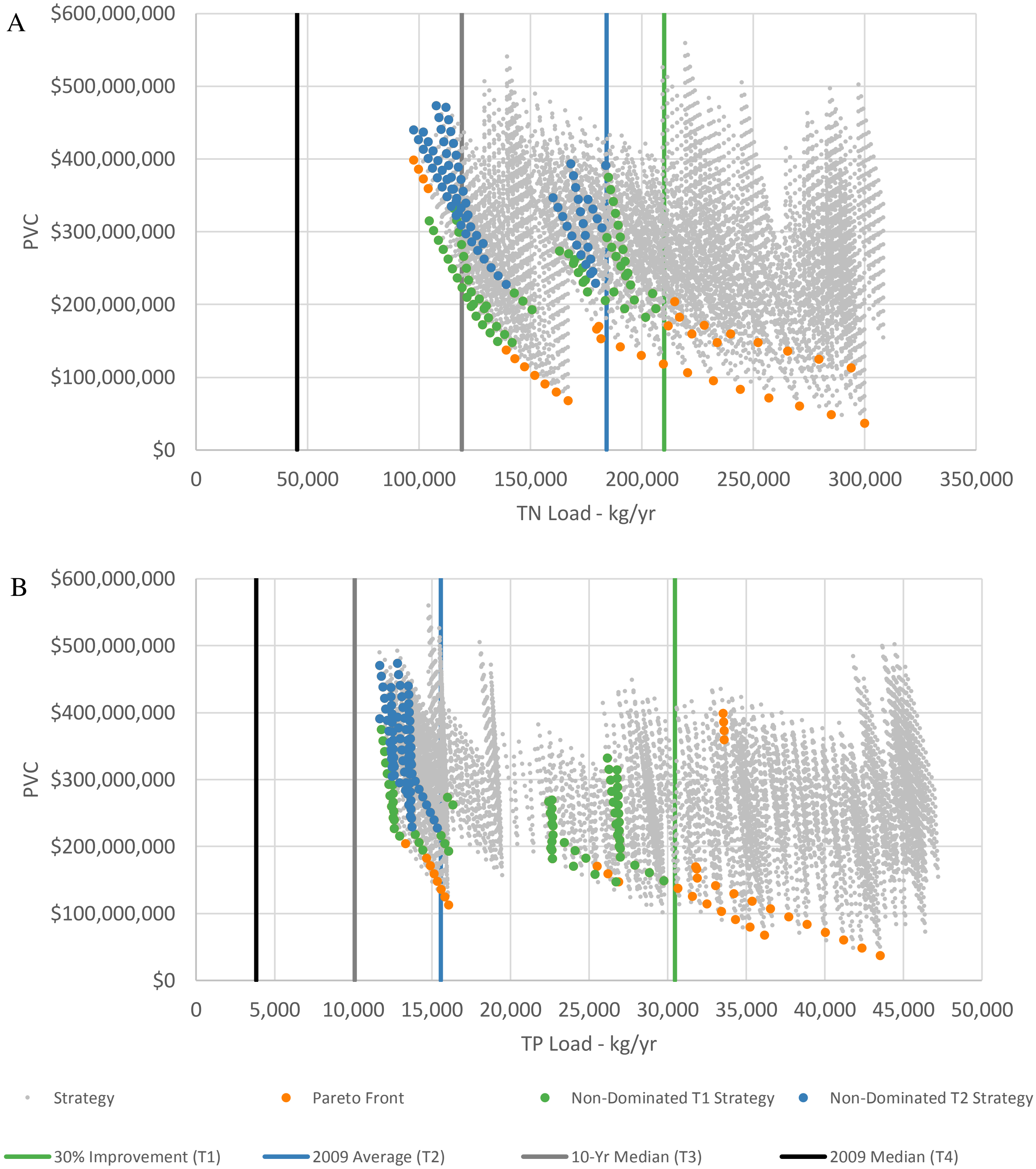

Figure 4-2. Present Value Cost (PVC) versus the annual Total Nitrogen (TN) load with target TN load indicated based on 2009 average flow, average 10-year median flow, and 2009 median flow (A), and PVC versus the annual Total Phosphorous (TP) Load with target TP load indicated based on 2009 average flow, average 10-year median flow, and 2009 median flow (B). The colored points identify non-dominated strategies that meet both TN and TP load requirements corresponding to the color of the stream target line and the pareto front considering cost, TN and TP.

From the non-dominated solutions determined by the Pareto Front, the optimal solutions were highlighted based on PVC, TN unit cost and TP unit cost for unconstrained, T1 and T2 targets (Figure 4-3). The optimal solution from the Pareto Front was identified as MLE w/ CA + FA and adoption of 15% SS. This strategy was also the optimal PVC for meeting the T2 target, while MLE + NS with 35% WR was the optimal PVC for meeting the T1 target.

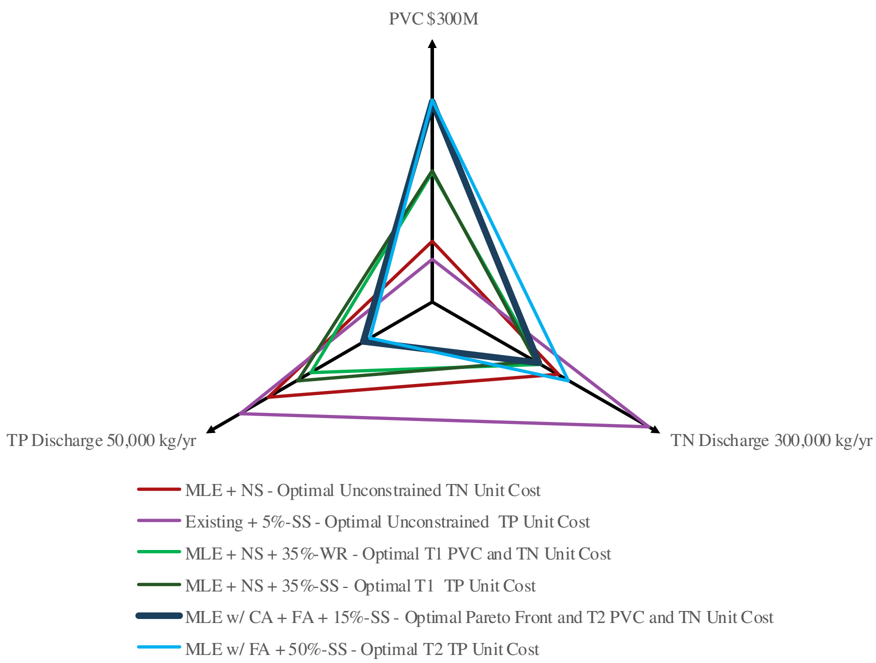

Figure 4-3. Pareto Front analysis non-dominated optimal Present Value Cost (PVC), Total Nitrogen (TN) unit cost and Total Phosphorous (TP) unit cost for unconstrained, Tier 1 (T1) and Tier 2 (T2) targets. Optimal non-dominated solution indicated with thicker line type.

The optimal nutrient strategies in the urban water system will be dependent on the defined nutrient target as well as characteristics like population, water use, local climate considerations and land use. While the optimal T1 and T2 strategies were highlighted that minimize PVC, there are other options that are also viable and effective in balancing tradeoffs between PVC, TN and TP as identified in the Pareto Front (Figure 4-2). To highlight effective practices, frequency plots were developed to quantify the frequency a given nutrient removal practice was part of an optimal nutrient removal strategy. This was first done for all non-dominated solutions, and then by constraining the optimal solutions to strategies that meet defined nutrient targets (Figure 4-4), and grouped based on PVC as low, medium or high. There are 176 non-dominated strategies which include a variety of combinations of WWTF improvements, water management practices and SCMs (Figure 4-4A). If the non-dominated solutions are constrained to those that meet the T1 nutrient load target, the number of viable strategies is reduced to 141 (Figure 4-4B), and 76 strategies that meet T2 nutrient loading target (Figure 4-4C). When constrained, there are no strategies that maintain the existing wastewater treatment process and the WWTFs would need to adopt A2O, MLE +NS, MLE+SP, MLE+FA or MLE+CA+FA. Worth noting is both water management practices and/or SCM were frequently necessary in effectively meeting the nutrient targets. As strategies are constrained, there is also a shift away from low cost strategies.

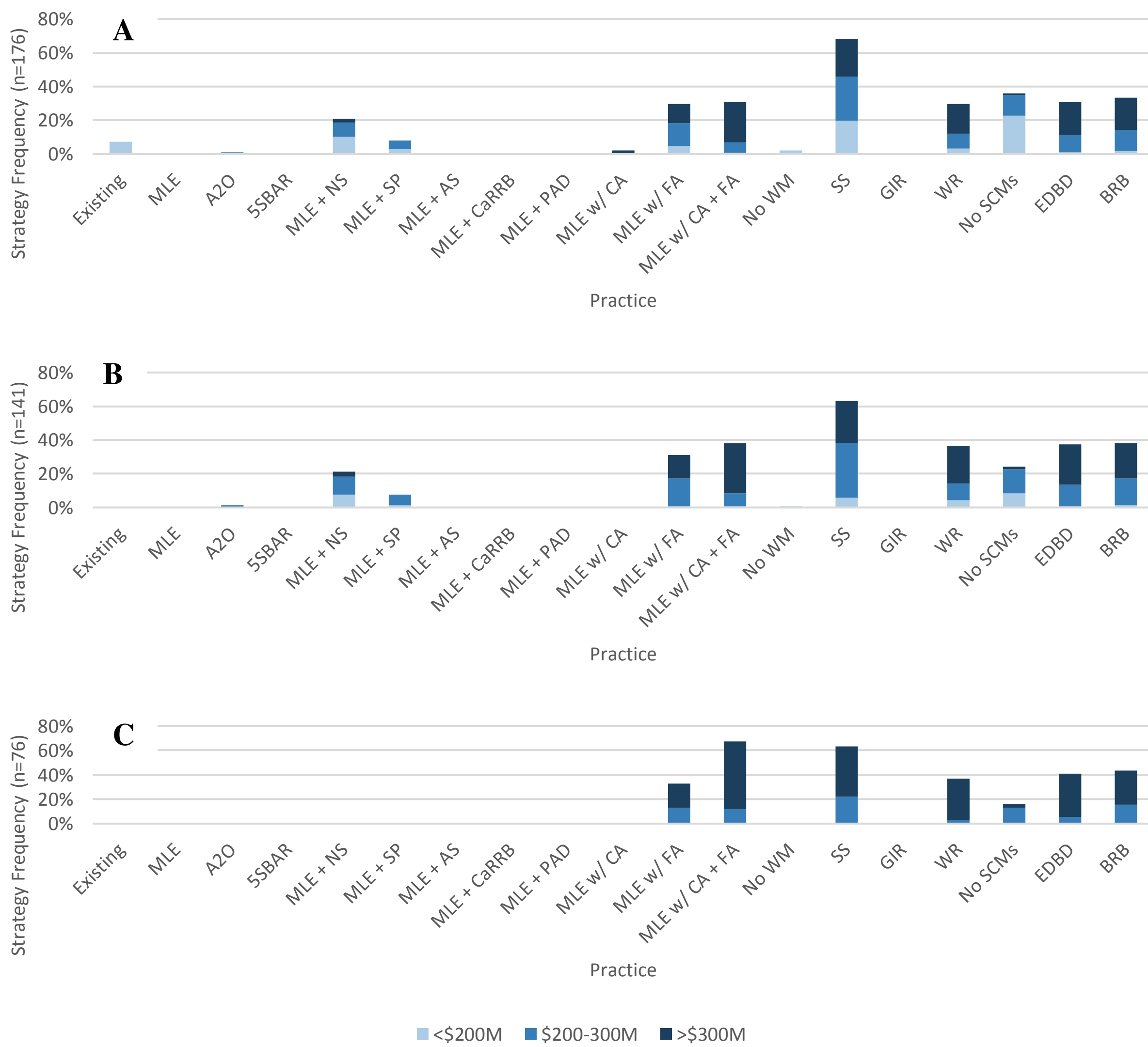

Figure 4. Frequency practice was part of an effective strategy as identified by the pareto front without loading constraints (A), that meet the Tier 1 (T1)loading constraints (B), that meet the Tier 2 (T2) loading constraints (C)

4.4 Discussion

The non-dominated strategies identify a variety of combinations of WWTF technologies, water management practices and SCMs can be part of effective nutrient removal scenarios. While individual strategies with adoption of a single practice in one sector was considered, all of the non-dominated solutions were based on a combination of practices across sectors. Interestingly, implementing technologies like 5SBAR, MLE + AS, MLE + CaRRB and MLE + PAD were not part of any non-dominated solutions. This indicates that the cost for implementing these technologies does not result in a reciprocal benefit to both TN and TP. Of the water management practices assessed, wastewater effluent reuse and source separation were identified in many of the non-dominated scenarios (Figure 4-4). While there may be social and political barriers for these practices, these practices result in an equal tradeoff in TN and TP load reduction while minimizing PVC. Similarly, both SCMs were frequently part of non-dominated scenarios, but adopting SCMs alone could not meet the identified nutrient targets. Of the non-dominated strategies, the optimal solutions were identified as the strategy that provided the overall most efficient minimization of TN and TP load, and PVC (Figure 4-3).

For meeting the defined T1 nutrient target, a combination of WWTF improvements, water management practices and/or SCMs were necessary, and more cost effective than implementing only WWTF improvements (Figure 4-3; Figure 4-4B). Considering wastewater treatment technologies, the most cost-effective solutions adopted MLE + SP or MLE +NS (Figure 4-4B). These technologies were combined with some level of water management practices SS or ER and/or implementing EDB or BR SCMs (Figure 4-4B). The results suggest that investing in water management and stormwater practices can potentially provide a more cost-effective approach in meeting nutrient targets versus investing only in WWTF improvements. To facilitate this, there is opportunity that water management practices and SCMs could be considered for nutrient trading depending on the local regulatory structure. As the nutrient load is reduced with the T2 target, water management practices and/or implementing SCMs are still part of optimal solutions, but WWTF would need to also adopt MLE w/ CA and/or FA (Figure 4-3; Figure 4-4C).

As mentioned previously, there were no strategies identified for meeting the T3 or T4 target. In certain instances, stringent nutrient requirements may result in target TMDLs that are not technologically achievable with wastewater treatment and stormwater treatment (WERF, 2010). To meet these targets, strategies would need to be implemented for reducing the background nutrient loading in additional to the evaluated urban water system strategies. The ability to meet low target nutrient loads will depend on a variety of factors unique to the urban area and the receiving water body. Looking at the study area, the streamflow is highly variable and can experience very low flow conditions which are heavily influenced by climate conditions, precipitation and snow pack (see Figure S-2 in Appendix A).

4.5 Conclusion

The developed urban water system approach provides an innovative framework for evaluating urban nutrient removal strategies across sectors. While there a variety of WWTF practices that can be considered, a nutrient control strategy that includes advanced biological nutrient removal systems with carbon addition and supplemental ferric addition will likely be necessary to meet stringent nutrient requirements. Even with adoption of this technology, a wholistic urban water system approach will likely be needed that includes water management and SCMs to meet target requirements.

Of the water management practices, source separation was most frequently identified as part of effective nutrient strategies. While source separation was consistently part of the most effective nutrient removal strategies, there are significant barriers to adoption both in terms of engineering, installation, maintenance and public perception (Fewless et al., 2011). Conversely, many utilities already adopt some level of effluent reuse and should also consider this as part of a viable nutrient removal scenario. However, in western states with prior appropriation the quantity of effluent reuse may be limited. Additionally, utilizing effluent reuse to meet nutrient regulatory requirements may have a negative impact by diverting flows to streams impacting aquatic species (Hodgson et al., 2018; WERF, 2010).

Similarly, SCMs proved to be an important part of effective nutrient scenarios. However, given that the cost for SCMs is primarily a function of area treated the cost effectiveness of SCMs will be highly variable depending on the percent contribution of stormwater runoff to nutrient pollution which will vary regionally based on precipitation (equation 4-5). For example, in wet climates with high levels of annual precipitation, the nutrient load from stormwater will reflect a larger portion of the overall nutrient loading, therefore investing in SCMs that reduce this load will be more cost effective.

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5.0 CONSIDERING THE IMPACTS OF NUTRIENT REGULATIONS TO BIOSOLIDS MANAGEMENT – A CASE STUDY ON POTENTIAL IMPLICATIONS OF NUTRIENT REGULATIONS ON BIOSOLIDS MANAGEMENT IN ARID WEST[4]

5.1 Introduction

The purpose of this study was to consider the impact that nutrient regulations may have on biosolids management particularly in the arid west. The Colorado Department of Public Health and Environment (CDPHE) has recently adopted/revised regulations to reduce nutrient pollution. Regulation 85 (Reg 85) limits existing wastewater treatment facilities (WWTF) to effluent total inorganic nitrogen (TIN) and total phosphorous (TP) annual median discharge concentrations of 15 mg/L-N and 1 mg/L-P respectively (CDPHE, 2012). Additionally, Regulation 31 was revised to limit stream concentrations of total nitrogen (TN) and TP to as low as 1.25 mg/L-N and 0.11 mg/L-P (CDPHE, 2016) which will require additional nutrient removal at WWTF's. The adoption of these regulations requires improvements to many existing WWTF operations. The majority of TN removal is traditionally achieved through nitrification and denitrification resulting in the formation of nitrogen gas; therefore, additional TN removal has minimal impacts to the nitrogen fraction in biosolids. However, TP removal is primarily achieved through either biological or chemical phosphorous removal and will result in the additional accumulation of phosphorous in biosolids proportional to the mainstream load removal. The proper management of biosolids is governed under Regulation 64 and EPA 40 CFR Part 503 (CDPHE, 2014; EPA, 1999) which dictates acceptable application sites and loading rates. With improved phosphorous treatment there may be a notable elevation in phosphorous content in biosolids that could impact existing management practices. Currently, biosolids is valuable resource utilized by many farmers for the beneficial nutrient content and typically applied to meet the nitrogen demands of crops.

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[4] A version of this chapter has been distributed as a white paper through the Center for Comprehensive, optimal and Effective Abatement of Nutrients (CLEAN) a research group of the Colorado State University Civil and Environmental Engineering Department

Brock Hodgson, Sybil Sharvelle, Troy Bauder, and Jim Ippolito
CLEAN Center, June 2017

5.2 Study Area and Approach

To identify the potential impact of elevated phosphorous content in biosolids, a case study was performed for WWTF in the Colorado segment of the South Platte River Basin to discuss the challenges associated with biosolids management in the arid west. The study basin includes 131 permitted discharges, 38 of which have a permitted capacity > 1 MGD and are required to comply with Reg 85 (CDPHE, 2012). Many WWTF are still in the planning/upgrading phase of meeting the adopted nutrient regulations, therefore the 2013 and 2014 reported Reg 85 data was used as the baseline conditions to evaluate the potential impacts of improved removal on biosolids concentrations.

Each facility was evaluated individually to estimate the impacts of additional phosphorous treatment to biosolids concentrations. Reg 85 requires monthly reporting of effluent flow and TN and TP concentrations. The calculations were performed for each reported sample and a facility average was calculated. The following steps were performed as part of this evaluation:

1. Calculate the additional load removal based on the minimum additional treatment necessary to meet Reg 85.
2. Calculate the change in biosolids phosphorous content based on the additional load removal.
3. Evaluate the impacts to biosolids management based on the Colorado Phosphorous Index (COPI) and phosphorous pollution concerns.

The additional load removal was calculated assuming that in order to ensure compliance, all reported samples above the Reg 85 requirement must be reduced to an effluent TN and TP concentration of 15 mg/L-N and 1 mg/L-P respectively. To evaluate the potential impacts to biosolids, assumptions were made on the biosolids formation rate and nutrient fractions. The Environmental Protection Agency (EPA) conducted a study of biosolids generation at multiple WWTF and developed a biosolids generation factor (BGF) of 205.7 dry tons per year / MGD Treated (USEPA, 1999). The nutrient components of biosolids can vary between 3-8 %TN and 1.5-3.5 %TP (Sullivan et al., 2007). A portion of the nitrogen in biosolids will be quickly lost as ammonia gas or not immediately available and therefore only a fraction, approximately 30% in the arid west, of the TN, is classified as plant available nitrogen (PAN; Barbarick and Ippolito, 2007). These values are comparable to 2013 observed values at Metro Wastewater

Reclamation District (MWRD; Denver, CO) which observed a BGF of 181.7 DTPY/ MGD Treated (MWRD, 2013b) and nutrient fractions of 6.95% TN (30% PAN) and 2.55% TP (MWRD, 2013a). Based on this data, Table 5-1 indicates the assumed biosolids generation rates and nutrient fractions used in the biosolids analysis.

Table 5-1. Baseline Biosolids Assumptions

	Value
BGF (DTPY/MGD)	205.7
TN	5.5%
PAN	30%
TP	2.5%
TP / P_2O_5	2.29
TP (P_2O_5)	5.73%

5.3 Results and Discussion

5.3.1 Improvements to Mainstream Treatment

Reg 85 will require improved TN and TP removal reducing the effluent nutrient loading. Many of the existing facilities do not currently meet the TP effluent concentration requirements based on the average reported 2013 and 2014 data (Figure 5-1). Therefore, these facilities will be required to improve existing treatment operations (Figure 5-1). As performance improves to meet the lower discharge requirements, the corresponding total reduction to TN and TP effluent loading was calculated (Figure 5-2). The regulations would result in an approximate reduction of 14% TN and 54% TP loading in the South Platte Basin (Figure 5-2).

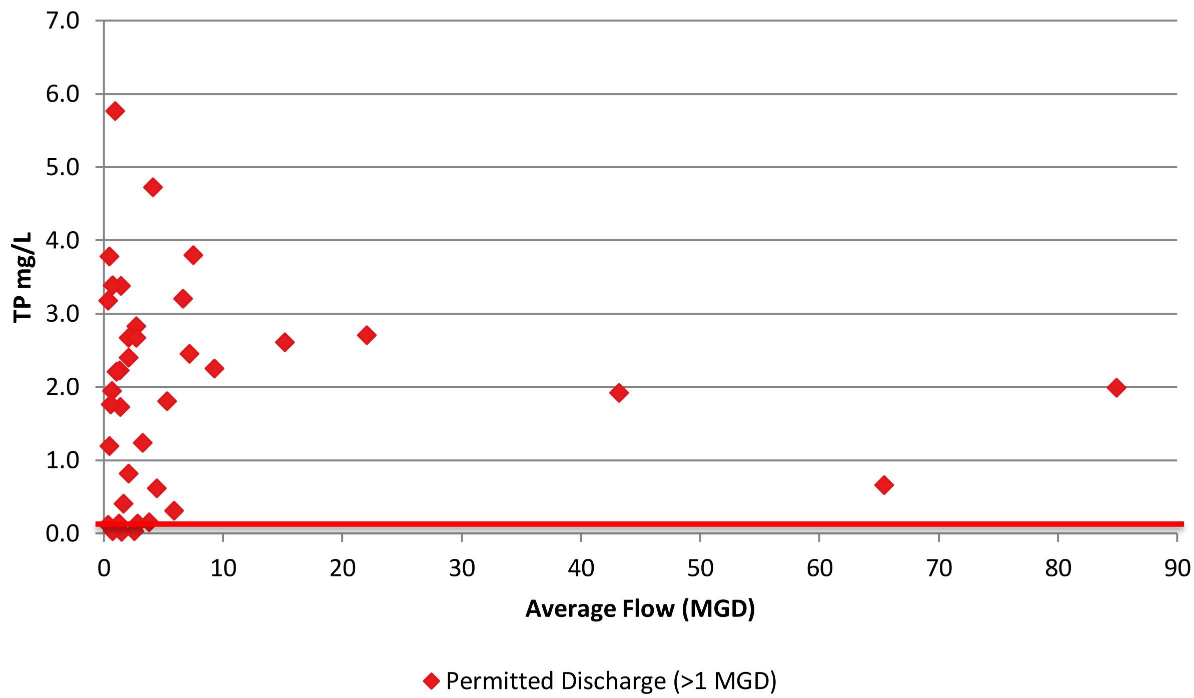

Figure 5-1. Average Effluent TP Concentration (Permitted Capacity > 1 MGD)

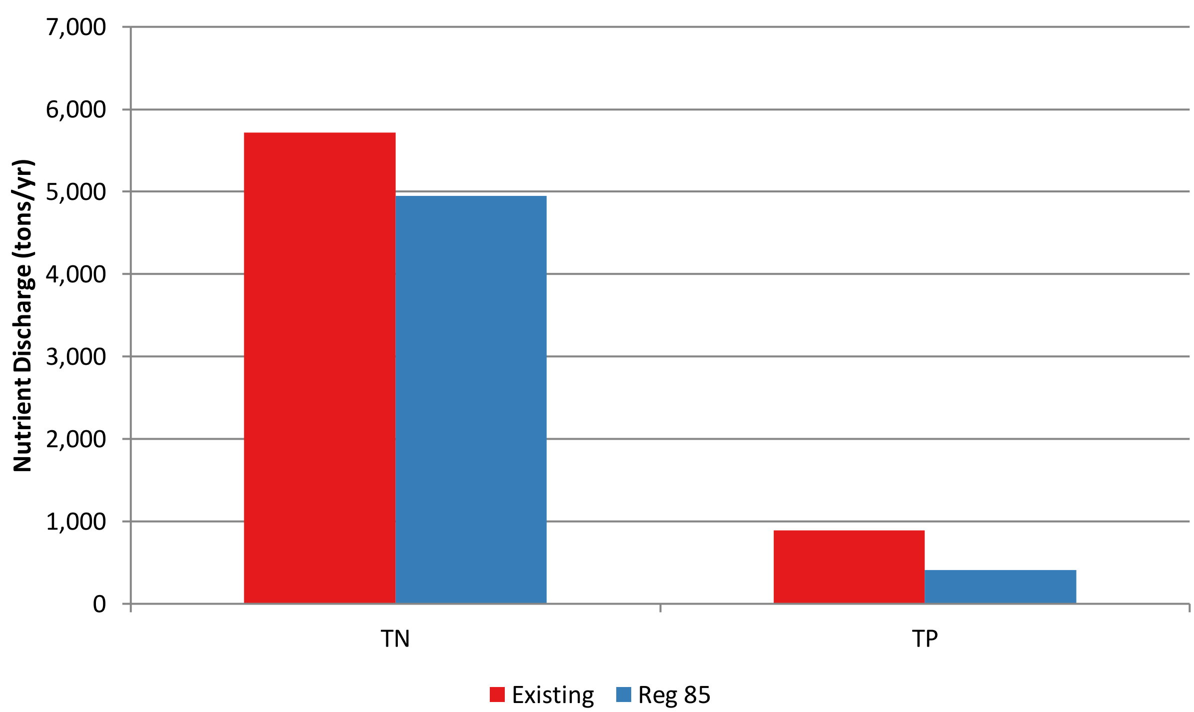

Figure 5-2. South Platte WWTF Total Nutrient Load (Permitted Capacity > 1 MGD)

5.3.2 Impact to Biosolids Phosphorous Content

As previously discussed, the TP removed will ultimately end up in the biosolids. If Bio-P is utilized, under anaerobic and aerobic conditions the additional phosphorous will be stored as polyphosphate by phosphorous accumulating organisms (PAOs) and wasted with the sludge (Leslie et al., 1999). If chemical phosphorous is utilized alum or ferric chloride is added to precipitate the orthophosphate with the sludge (Reynolds and Richards, 1996). To achieve high levels of phosphorous removal, facilities may utilize polishing filters to remove suspended biological treated phosphorous and/or chemically treated phosphorous not easily settled. Filtered phosphorous is backwashed with the filters and the waste streams are typically returned to the sludge.

Therefore, as TP removal is increased the corresponding impact to biosolids concentration was considered for each facility. The phosphorous content for all facilities was initially assumed to be 2.5% as TP, and the increase in biosolids phosphorous content (Figure 5-3) was calculated based on the additional TP load removed.

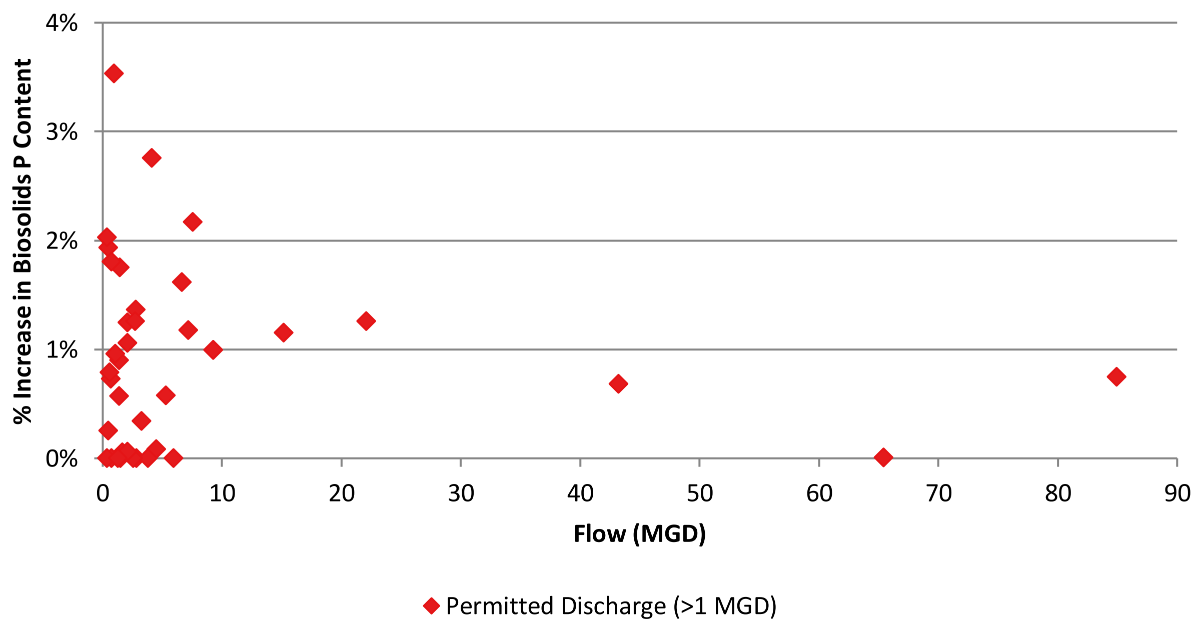

Figure 5-3. Increase in Biosolids Phosphorous Content

Agricultural management is primarily concerned with available phosphate (P_2O_5) content which is calculated as 2.29 times %TP content assuming 100% availability. Initially it was assumed that all biosolids have an average TP content of 5.73% as P_2O_5 (2.5% as P). The estimated TP content of the biosolids is reported in Figure 5-4. Based on the additional load of phosphorous removed as required by Reg 85, biosolids phosphorous content was estimated as high as 14% as P_2O_5 (Figure 5-4).

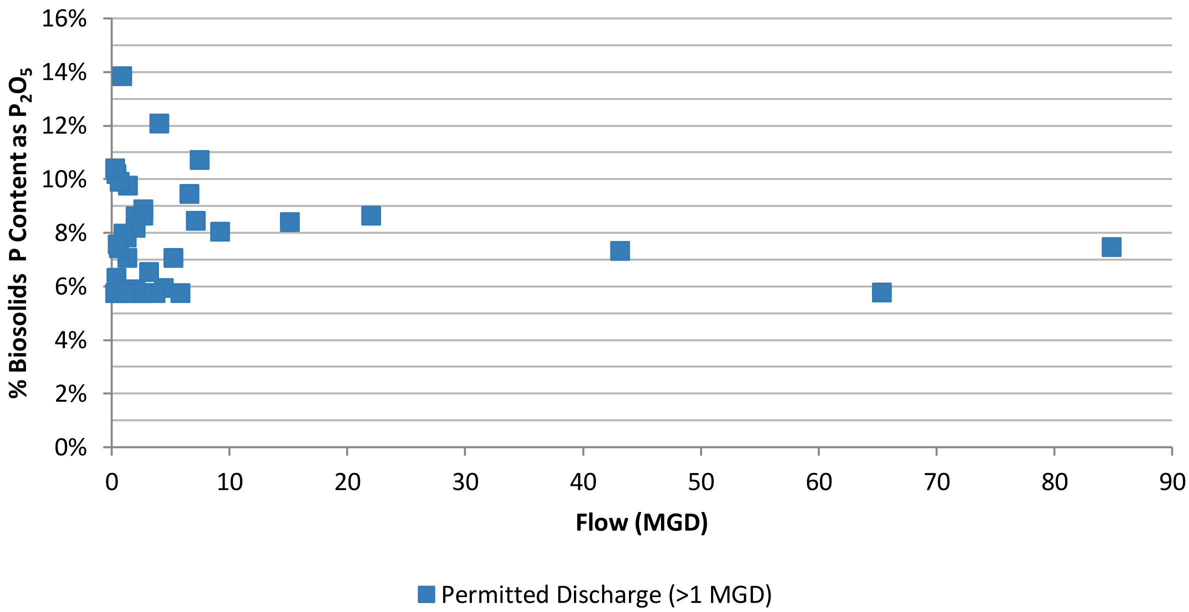

Figure 5-4. Estimated Biosolids Phosphorous Content as P_2O_5

5.3.3 Impact to Biosolids Management

There are potential biosolids management implications as the phosphorous content of biosolids increases. Biosolids are traditionally land applied to crops because of the beneficial nitrogen and phosphorous content. However, crop demands for nitrogen are much larger than phosphorous. Therefore, biosolids application is typically applied based on the plant available nitrogen in the biosolids and phosphorous is often over applied. This is common with other organic fertilizers such as manure or compost. The over application of phosphorous is a potential concern as the excess phosphorous not utilized by the crops could potentially transport downstream and pollute water bodies. While the potential of transport is a concern, phosphorous is typically significantly less mobile than nitrogen. This will be largely a function of the form of phosphorous in the biosolids and the soil types. The semi-arid climate, such as Colorado, the neutral to basic soil types, often high in $CaCO_3$, and the opportunity to properly manage irrigation water make the offsite pollution of phosphorous less likely, but this has not been well studied to date.

The USDA/NRCS typically develop P and N indices at the state level consulting with local land grand universities to develop management tools to help guide the potential over application of nutrients from organic nutrient sources like biosolids. These indices provide a scoring system to identify the phosphorous runoff potential based on variables like site conditions, application rates, management practices, etc. If the potential for phosphorous runoff is significant, the biosolids applications must be limited to the crop phosphorous requirements. This limitation is notable from a biosolids management perspective because biosolids are more valued for the nitrogen content than the phosphorous content, and because crop demands for nitrogen are much higher, the application rates per area are much lower when limited based on phosphorous making disposal more difficult and costly.

For these reasons, it is important to consider what impacts additional mainstream phosphorous treatment may have on biosolids content. It is estimated many facilities will observe notable increases in biosolids phosphorous content. The estimated biosolids phosphorous content as presented in Figure 5-4 were ranked based on the estimated TP content of biosolids to visualize the potential range of concentrations and the number of facilities that may be impacted (Figure 5-5).

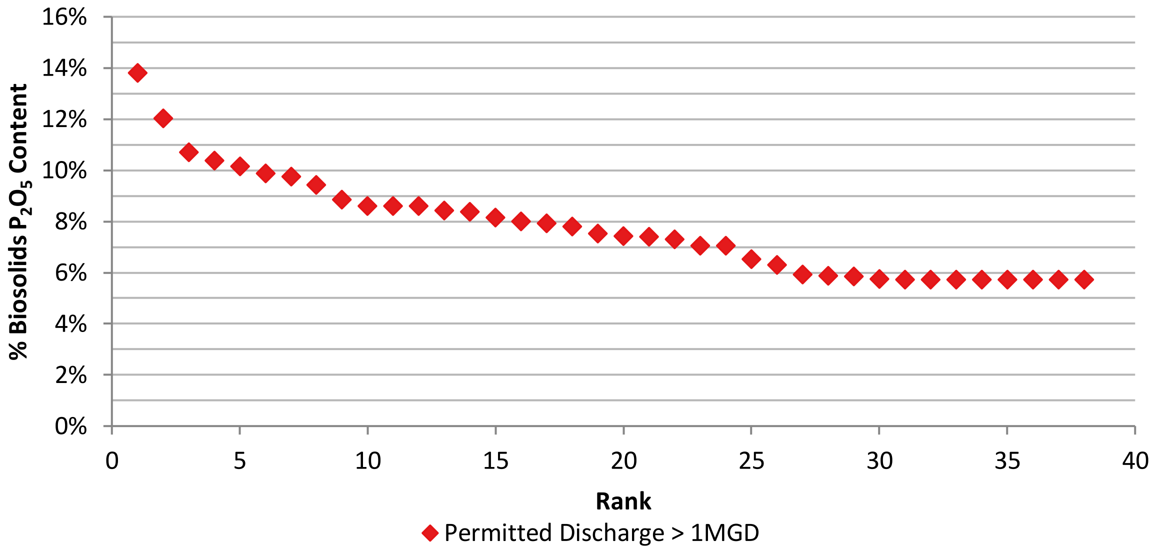

Figure 5-5. Facilities Ranked based on Estimated Biosolids Phosphorous Content as P_2O_5

The elevated P content will result in additional over application of phosphorous and may ultimately limit the application of biosolids based on the crop phosphorous requirements. The current COPI does not differentiate the form of phosphorous in biosolids or implemented treatment approaches with regards to the P application limitations. For instances where chemical treatment is utilized, like alum addition, the phosphorous is strongly bond and therefore highly unavailable in terms of valuable nutrient to crops or a pollution concern. Conversely, organic phosphorous has a potentially greater availability.

Consistent with the existing COPI requirements, the estimated application of phosphorous based on the percent phosphorous in biosolids was calculated at different nitrogen application rates (Figure 5-6) assuming a biosolids nitrogen content of 5.5% TN and 30% PAN. Typically, dryland crops, like wheat, will have lower nitrogen application rates (~50 lbs. PAN/acre) while irrigated crops like corn will have higher nitrogen application rates (~185 lbs. PAN/acre). Figure 5-6 provides an understanding of the increase in phosphorous application associated with the increase in biosolids phosphorous content.

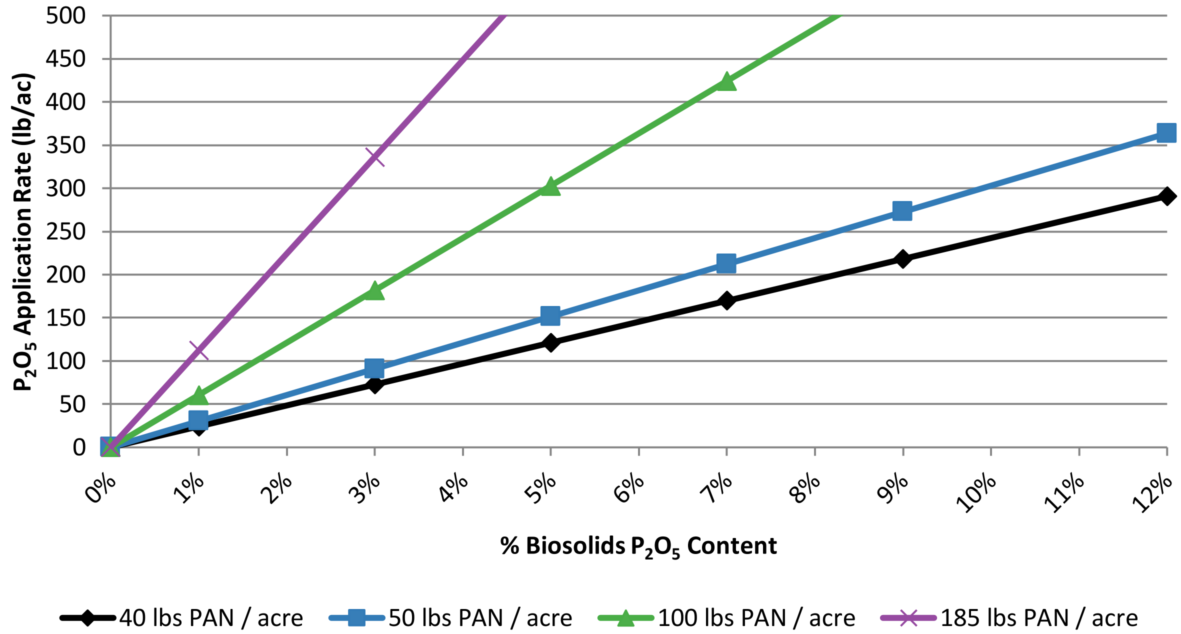

Figure 5-6. P_2O_5 Application Rate based on Biosolids Phosphorous Content as P_2O_5

5.3.4 Phosphorous Indices and Regulations

The Colorado Phosphorus Index (COPI), Version 5 (Sharkoff et al., 2012) was developed by the Natural Resource Conservation Service (NRCS) to qualitatively identify the potential risk of over application of phosphorous associated with organic nutrients (e.g. Biosolids). The index is based on the transport risk, the measured phosphorous content of the soil, the application rate of the organic nutrients, the application method and timing, and any mitigation or best management practices (BMP) that are implemented at the application location. Each application site is scored based on these criteria to identify the risk of excess phosphorous application. The resulting total score indicates if application based on nitrogen requirements is satisfactory or if application must be phosphorous limited (Table 5-2).

Table 5-2. COPI Score Interpretation

Assessment Interpretation	Risk	Total Score
Organic P application rate may exceed the crop P requirements if the application rate does not exceed the N requirement for the next crop	Low	4-11
P application rate is restricted to the crop P requirement for the next crop	Medium	12-13
P application rate is restricted to crop P removal for next crop if a P draw-down strategy is implemented for the crop rotation	High	14-15
Do not apply P to this field until the risk of P movement off-site is decreased	Very High	16

To understand the management implications, an example COPI was performed for a field where biosolids may be applied for a dryland crop requiring 50 lbs. PAN / acre and a biosolids phosphorous content of 5.5% TN and 7% TP as P_2O_5. It is estimated that 23 of the 38 facilities could have biosolids phosphorous content greater than 7% TP as P_2O_5 (Figure 5-5). Table 5-3 indicates the estimate COPI score for the example consideration.

Table 5-3. COPI Example Calculation

	Factor	Basis	Score
1	P Transport		
A	Index Surface Runoff (Irrigated and Non-Irrigated)	Slope% 1-5; Saturated Hydraulic Conductivity <1.0-0,1 um/s	2
B	Rill and Interrill Erosion (Non-Irrigated)	Erosion Rate >5-10 tn/ac	2
C	Wind Erosion (Irrigated)	-	-
2	Soil Test P	61-120 ppm Bray P1	2
3	P Application Rate	P_2O_5 Application >200 lb./ac	4
4	P Application Method and Timing	Fall/Winter Applied and Incorporated	3
5	BMP/Mitigation Credits	Liability/Management Limits Reliability	0
		TOTAL	13

The resulting score of 13 has notable implications in terms of biosolids management and requires application rates to be P limited (Table 5-2). This score was calculated assuming modest site conditions in terms of P transport and Soil Test P. The assumed values relative to index surface runoff, rill and interrill erosion, and soil test P are very conceivable based on observed field measured data, and these factors could score higher further limiting phosphorous application. As discussed above, the over application of P could easily be at rates >200 lb./ac as the phosphorous content in biosolids increases resulting in p application score of 4 (Figure 5-6). The utilities have little control over the P application timing as biosolids are produced year-round and therefore the application timing cannot be limited to the spring. Lastly, BMP or mitigation credits present management and liability concerns making it difficult for utilities to depend on these credits. This suggests that a high COPI rating is possible even maintaining the same application rates as the phosphorous content in biosolids increases.

The COPI serves as a guidance document but is not a regulatory document. Biosolids management is governed by Regulation 64 which requires WWTF to submit information on metals content, pathogen destruction, site conditions, groundwater depth, application rates, etc. for the land application of biosolids. Similar to the COPI criteria; Regulation 64 considers site conditions (ex. slope) and nitrogen loading rates when determining the appropriate application of biosolids. With regards to phosphorous, Regulation 64 will require compliance with guidance documents, like the COPI, if application on fields with soil test P values of 120 ppm – Bray P1 extraction or 200 ppm – Mehlich 3 extraction for soil pH < 6.5 or 80 ppm – sodium bicarbonate or 40 ppm – AB-DTPA for soil pH > 6.5 (CDPHE, 2014). Therefore, facilities may be limited based on the COPI depending on field measured soil phosphorous values. Additionally, the COPI also must be followed with in the event that a landowner or crop producer receives funding from the USDA.

5.3.5 Implication of Nutrient Regulations and Biosolids Management

Evaluating the WWTF in the South Platte River Basin, it is estimated that nutrient regulations presents notable implications on biosolids management primarily associated with the increase in biosolids phosphorous content. The primary concern is as the phosphorous content in biosolids increases, the additional over application of phosphorous based on nitrogen application rates poses a threat to off-site loss of phosphorous and the possibility of nutrient pollution. The potential shift to land application of biosolids based on phosphorous requirements may have three notable management implications.

Potential Implications:

- Significant additional land required for biosolids application
- Nitrogen fertilizers would need to be supplemented to meet crop demands
- Facility Managers are concerned that biosolids have limited value as a fertilizer product which may increase the amount of biosolids landfilled

The first implication is the potential increase in area necessary for biosolids application. To evaluate this implication, an estimate was made to consider the potential extent assuming all biosolids application is limited based on phosphorous requirements and the average annual crop nutrient requirements are 75 lbs.-N/acre and 40 lbs.-P/acre. Figure 5-7 indicates the land required for biosolids application based on the nitrogen requirements (existing) and in the event that application is limited based on phosphorous (existing and Reg 85 compliance). The increase in acreage required is primarily the result of 3 factors: (1)

lower crop demands for phosphorous compared to nitrogen, (2) only ~30% of the TN in biosolids is PAN allowing for application rates ~3 times higher than the nitrogen content of the biosolids, and (3) the increase in biosolids phosphorous content as discussed in the previous sections. For this reason, limiting biosolids application based on phosphorous would reduce the phosphorous application rate by 85% and increase the area required by 833% (Figure 5-7). This figure represents an extreme case if all biosolids were limited to phosphorous agronomic loading rates. However, local conclusions can be drawn that if a given field is P limited ~8 times the acreage would be required compared to N limited application given the estimated biosolids nutrient content. The potential implication on nutrient regulations and application rates is not unique to Colorado. Jones, 2012 documented an increase in land required associated with phosphorous limited applications in Florida and the implications in terms of shortages of application sites, increase in disposal cost, or movement to Class A Biosolids (Jones, 2012). These costs primarily fall on the utilities and should be considered as part of the cost for nutrient removal at WWTF.

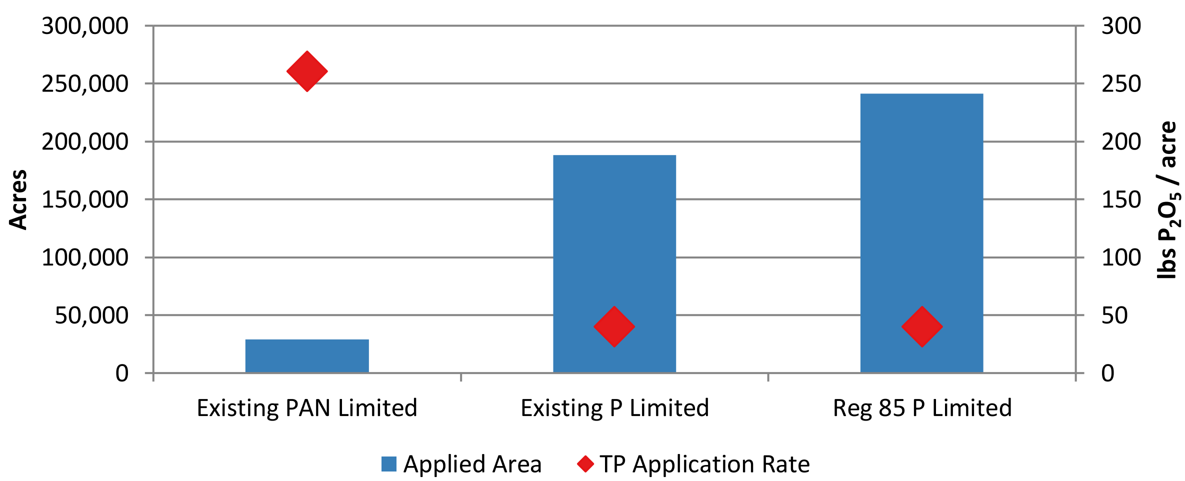

Figure 5-7. Biosolids Application Area and Corresponding P Application Rate

The second potential implication is the additional nitrogen that would need to be supplemented to meet the crop TN requirements. The supplemental nitrogen required was estimated based on the same assumed crop demands of 75 lbs.-N/acre and 40 lbs.-P/acre. In order to satisfy the nutrient demands in the event that biosolids are applied based on crop phosphorous requirements, it was assumed that fertilizer would be supplemented as necessary. Figure 5-8 presents the resulting nutrient application rates based on the existing biosolids PAN limited, biosolids after Reg 85 compliance PAN Limited, and biosolids after Reg 85 compliance P limited. Figure 5-8 indicates that a significant mass of nitrogen would need to be imported into the basin as supplemental fertilizer. Since nitrogen fertilizer application is not regulated and limited to agronomic rate like biosolids, it could pose an additional non-point pollutant concern associated with potential over application, leaching or site run-off.

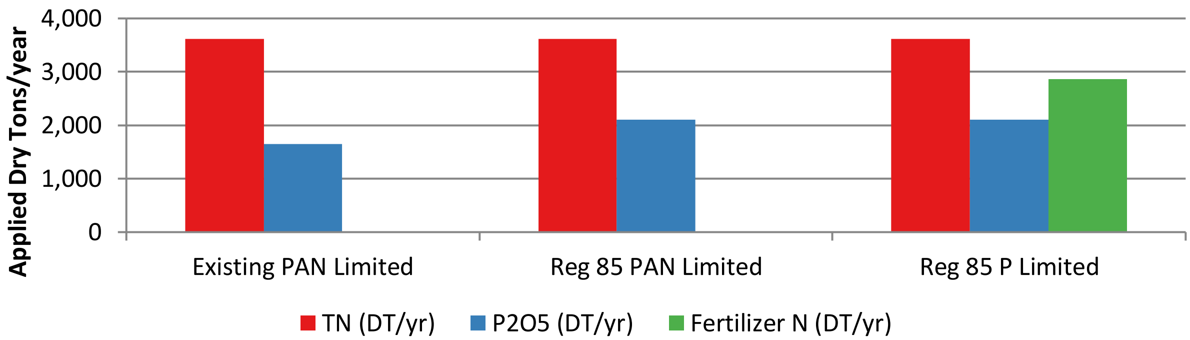

Figure 5-8. Biosolids Application Area and Corresponding P Application Rate

The last potential implication is the devaluation of biosolids for beneficial reuse given the P limitations as a result of the elevated phosphorous content. Biosolids contain a variety of macro nutrients valuable in agriculture (Table 5-4; Sullivan et al., 2007). While the other macro and micro nutrients are utilized by crops, it is traditionally at much lower rates making the nitrogen in biosolids most valuable. Additionally, nitrogen application rates account for the availability of the nitrogen in biosolids with limiting applications based on the PAN. Therefore, traditionally biosolids applications result in the over application of some nutrients like phosphorous and the under application of other nutrients like potassium (Sullivan et al., 2007).

Table 5-4. Biosolids Macronutrients (PNW, 2007)

Nutrient	Range
Organic Matter	45-70%
Nitrogen (N)	3-8%
Phosphorus (P)	1.5-3.5%
Sulfur (S)	0.6-1.3%
Calcium (Ca)	1-4%
Magnesium (Mg)	0.4-0.8%
Sulfur (S)	0.6-1.3%

In the event that biosolids application becomes P limited, there may be a notable reduction of the market value of the biosolids. Based on crop demands, the primary value of biosolids is in meeting the nitrogen agronomic rates as discussed above. Additionally, application sites would require supplemental nitrogen fertilizers. Supplemental fertilizers would come at an additional cost to the farmer or the utility. Lastly, like nitrogen, not all of the phosphorous in biosolids is readily available for crops. Therefore, limiting applications based on total phosphorous would likely under supply plant available phosphorous. These limitations could negatively affect yields and the value of biosolids to the farmer.

The existing COPI does not account for the available phosphorous fraction in evaluating the P loss potential, or in calculating the phosphorous loading rates. In contrast, the Florida Phosphorous Index includes a factor of only 1.5% of biosolids application to account for the availability of phosphorous (Hurt et al., 2013). Other states, like Pennsylvania, have also adopted P source coefficients to account for variations in mobility based on the type of phosphorous in biosolids. A study calculating the fertilizer replacement value of biosolids estimate the plant available fraction of biosolids phosphorous to be 40% (Sullivan et al., 2007). The variance in phosphorous availability in biosolids reflects an unknown relative to biosolids management. There is a need to better quantify the runoff potential and transport risks associated with biosolids phosphorous and account for the fraction of phosphorous that is available for use to crops or as a pollutant concern. This may include considerations based on the treatment technologies utilized by the WWTF to remove phosphorous. Chemical phosphorous treatment may exhibit very low transport risk based on the recalcitrant nature of phosphate product. WWTF that utilize Bio-P may result in a higher portion available based on the weaker bonds associated with the polyphosphate. However, in high pH soils common in the arid west available phosphorous can quickly bond with anions to form poorly soluble compounds making them less of a pollutant risk. In either case, a better understanding of the plant availability of phosphorous and the P loss potential relative to the treatment types may have important implications on biosolids management.

In addition to the availability of phosphorous, one of the most important considerations is the proximity and runoff conditions to receiving water bodies. Regulation 64 includes restrictions that no biosolids can be applied within defined locations (e.g. one linear mile) from a state water body (CDPHE, 2014). The

COPI includes a screening tool exempting facilities where runoff does not pose a risk to receiving waters but does not provide criteria for assessing this risk. Given the importance and implications of limiting phosphorous application, there is an opportunity to better define the risk of phosphorous pollution given the proximity to water bodies.

5.4 Conclusion

The adoption of nutrient regulations may have notable impacts to biosolids management primarily as a result of elevated phosphorous content in biosolids. As phosphorous content in biosolids is increased, traditional application based on the nitrogen agronomic rates may result in the over application of phosphorous at rates that pose a risk of runoff and site loss. This may result in applications to be phosphorous limited which would result in a large increase in application area, require supplemental nitrogen fertilizer applications, and result in a decrease in the market value of biosolids. Understanding the availability of phosphorous will be an important consideration in calculating the risk of P loss and the limit at which phosphorous can be applied. The type of treatment utilized may have noteworthy implications on the fraction of phosphorous available. In either case, nutrient regulations will impact existing biosolids management. At the least, utilities will face the challenge of ensuring and improving biosolids applications to prevent P loss to surface streams. There is the opportunity for improved biosolids guidance and regulatory framework to aid utilities in responsible application of biosolids accounting for the additional expected phosphorous content, while maintaining the beneficial use of biosolids as a nutrient source. To accomplish this additional research is necessary to provide a better understanding of the pollutant concerns associated with biosolids phosphorous and ensure that phosphorous removed in WWTF does not pose a threat as non-point pollutant.

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6.0 CONCLUSION

Utility managers face difficult decisions in identifying the cost-effective and efficient solutions in meeting nutrient reduction goals. These stringent nutrient regulations may result in limits that are not technologically achievable and/or very costly to achieve at WWTFs. This research aimed to develop a framework for easily evaluating and comparing effective nutrient removal technologies at WWTF and quantify the potential benefits of adopting water management practices and SCMs as part of cross sector nutrient removal strategy.

There has been little research to date characterize the potential positive or negative water quality impacts with the adoption of water management practices. This research quantified the water quality impacts of common water management practices at the WWTF influent, WWTF effluent and downstream receiving water body. The findings suggest that indoor conservation, resulting in a higher concentration influent, can have a negative impact in WWTF effluent nutrient concentrations and downstream receiving water body quality. With adoption of indoor conservation, process improvements may be necessary to avoid process limitations that occur with increase in influent concentrations. In terms of graywater reuse for irrigation or toilet reuse, there were negligible impacts observed at the WWTF or to downstream water quality. Conversely, the reduction in nutrient loading from source separation provided a notable impact on WWTF effluent and downstream water quality. While there are many social and technological barriers to the widespread adoption of this practice, this research highlights that there can be a notable benefit in terms of water quality if feasible locations are identified. Similarly, WWTF effluent reuse is already accepted in many areas as a viable strategy for reducing downstream nutrient loading. However, the benefit of effluent reuse as a nutrient removal strategy will be largely dependent on the receiving water body dynamics, including seasonality of flows corresponding to demand for reused water. The potential for effluent reuse as a nutrient removal strategy will also be dependent on regulatory structure (concentration versus load) and local water policy (riparian versus prior appropriation).

There is a depth of research with regards to WWTF process configurations for reducing TN or TP. This depth of knowledge can make it difficult for utility managers to understand how these various process configurations compare and what technologies should be considered for adoption in meeting more stringent nutrient requirements. These processes are traditionally evaluated using well understood mechanistic models which require a rigorous sampling and analysis plan and modeling expertise. This research built off these widely accepted mechanistic models to provide a simplified empirical model for estimating and comparing the effectiveness of common and innovative WWTF process configurations. Using statistical methods, empirical models were developed estimating the percent TN and TP removal as a function of influent COD:TN and COD:TP respectively. Using these influent parameters allows for utility managers to easily apply and understand the benefits of technologies at other WWTF. The results indicate that under carbon limited conditions, sophisticated treatment process configurations like 5-Stage Bardenpho and MLE operated as Nitrite Shunt can be most effective in removing both TN and TP. However, under non-carbon limited conditions there is little difference in terms of nutrient removal performance between MLE, MLE operated as Nitrite Shunt, A2O and 5-Stage Bardenpho. Conversely, utility managers are often interested in the potential benefit of adopting sidestream nutrient removal technologies for meeting effluent nutrient removal targets. Sidestream processes that recover nutrients, like struvite precipitation and ammonia stripping, were shown to provide a noticeable benefit on overall nutrient removal. Like the more sophisticated mainstream technologies, the benefit will be more pronounced under carbon limited conditions and less pronounced under non-carbon limited conditions. However, the research identified that there is negligible or even negative benefit in terms of overall nutrient removal in implementing sidestream processes that only partially nitrify and/or consume carbon that would be returned to the mainstream like CaRRB and PAD.

One of the primary objectives of this research was to develop a framework for evaluating nutrient removal strategies across the urban water system sector that can be used to identify the most cost-effective strategy for meeting nutrient target goals. To perform this analysis, it is necessary to have simplified models from each sector to facilitate a synthesized evaluation. This was accomplished using the results from the water management study and WWTF comparison study to apply an empirical modeling approach for estimating nutrient removal effectiveness combined with the Simple Method to estimate the benefit of SCMs. By implementing an urban water system evaluation, it is possible to identify the most cost-effective nutrient removal strategies not limited to WWTF improvements. Using a multi-objective optimization, the non-dominated solutions considering tradeoffs in reducing TN, TP and costs were identified. The results indicate that water management practices, primarily source separation or effluent reuse, and SCMs were frequently part of non-dominated nutrient removal strategies. This highlights that the most effective nutrient removal strategies from an urban water system should not be limited to WWTFs. Considering the impacts of nutrient regulations, the non-dominated solutions were constrained to identify the most cost-effective solutions that meet stream-based nutrient limits. With adoption of more stringent nutrient regulations, the number of viable WWTF technologies is limited but water management practices and/or stormwater practices are still a frequent part of non-dominated strategies. Based on the most stringent achievable nutrient standard evaluated, WWTFs would have to implement MLE with supplement carbon addition and ferric addition as well as source separation and SCMs. This would be very technically difficult if not practically unfeasible and highlights the challenges that utility managers face in meeting stringent proposed nutrient regulations. In this case, additional measures would need to be implemented outside of the urban water system to reduce background nutrient concentrations and contributions from non-point sources.

Implementation of additional nutrient removal technologies at WWTF may result in negative cost and impacts to biosolids management. Improvements in phosphorous removal at the WWTF will result in elevated concentrations of phosphorous in biosolids which is often beneficially land applied to crops based on agronomic nitrogen loading rates. As the phosphorous concentration is elevated, the application rates may be limited to phosphorous which would require a much larger application area and supplemental nitrogen via imported fertilizer. This shift would be very costly in terms of WWTF biosolids operations and may require landfill of wastewater solids that have been traditionally used as a valuable nutrient resource. It is important that utility managers consider these impacts as part of an overall nutrient removal strategy. This also identifies the need for regulations to be developed through cross sector coordination where surface water regulations like CDPHE Regulation 31 and the intended and unintended impacts are coordinated with agronomic regulations and policies like the Colorado Phosphorous Index in meeting an overall nutrient reduction goals in a watershed.

APPENDIX A

Supplemental Material for Chapter 4: Assessing Cost Effective Nutrient Removal Solutions in Urban Water Systems

Table S-1. Empirical Model Fit Values for Water Management Practice Impact to Total Nitrogen (TN) Load and Total Phosphorous (TP) Load

Water Management Practice	TN Load Impacts			TP Load Impacts	
	β_0	β_1	β_2	β_0	β_1
Source Separation	1.0	-1.51	0.70	1.0	-0.68
Graywater Irrigation Reuse		NS		1.0	-0.10
Effluent Reuse	1.0	-1	-	1.0	-1.0

Table S-2. Generalized Empirical Model Inputs for Estimating Percent Total Nitrogen (TN) and Total Phosphorous (TP) Removal based on wastewater treatment facility (WWTF) Process (MLE = Modified Ludzack Ettinger, A2O = Anaerobic, Anoxic, Oxic, 5SBAR = 5-Stage Bardenpho, NS = Nitrite Shunt, SP = Struvite Precipitation, AS = Ammonia Stripping, CaRRB = Centrate and RAS Reaeration Basin, PAD = Post Aerobic Digestion, CA = Carbon Addition, FA = Ferric Addition).

WWTF Process	Percent TN Removal				Percent TP Removal	
	β_0	β_1	β_2	β_3	β_0	β_1
MLE	18.2	1.24	0.259	-0.0082	7.2	0.34
A2O	40.2	2.15	0.03	-0.002	$TP_{(\%)} = 85$	
5SBAR	37.9	3.20	-0.021	-0.0011	$TP_{(\%)} = 90$	
MLE + NS	85	-0.59	0.083	-0.0022	18.4	0.47
MLE + SP	15.6	2.28	0.185	-0.0067	27.5	0.47
MLE + AS	-52.1	16.48	-0.658	-0.0087	7.2	0.47
MLE + CaRRB	6.2	2.17	0.238	-0.0079	9.5	0.47
MLE + PAD	36.4	-1.96	0.437	-0.0113	9.1	0.47
MLE + CA	$TN_{(\%)} = 90$				7.2	0.34
MLE + FA	18.2	1.24	0.259	-0.0082	$TP_{(\%)} = 90$	
MLE + CA + FA	$TN_{(\%)} = 90$				$TP_{(\%)} = 90$	

Table S-3. Median runoff concentrations for Total Nitrogen (TN) and Total Phosphorous (TP) based on Land Use

Land Use	Median TN (mg/L)	Median TP (mg/L)
Open Space	3.76	0.41
Residential	4.19	0.45
Commercial	2.79	0.22
Industrial	3.60	0.25
Institutional†	4.19	0.45
Highway	3.60	0.28

†Assumed residential runoff concentrations for institutional land use

Wastewater Treatment Unit Cost Development

The fixed and recurring unit cost for MLE, A2O and 5SBAR were adapted from a compilation of nutrient pollution cost data for similar process configurations (USEPA, 2015). The costs provided are FC and MC as a function of gpd treatment capacity. The costs were reviewed based on treatment type and the average FC and MC was calculated. The costs for operating MLE + NS was assumed to have the same FC, but the operational costs were reduced by 12% because the aeration requirements were assumed to be reduced by 25% (Jimenez et al., 2014) and aeration represents between 45%-75% of a facilities energy cost (Rosso et al., 2008).

For CA and FA, in addition to the MLE cost a FC of $0.19 per gpd and MC of $0.07 per gpd per year was assumed (USEPA, 2008; CDM, 2007). For CA, a methanol dose of 3.8 g of methanol per g of nitrate (NO_3) as nitrogen removed based on observed application rates at WWTFs (USEPA, 2010), and assuming a cost of $2.36 per gallon of methanol (CDM, 2007). For FA, a dose of 1.5 moles of ferric per mole of TP removed was assumed accounting for effects of pH, alkalinity and competing reactions which equates to a dose of 2.7 lb. of ferric per lb. of TP removed with FA, and assuming a cost of $483 per ton of ferric chloride (USEPA, 2010).

There is limited published cost data for sidestream treatment which includes SP, AS, CaRRB, and PAD. In addition to the MLE unit cost discussed above, engineering costs were developed for the additional cost of sidestream treatment for a conceptual 10 MGD WWTF with assumed influent nutrient concentrations, return activated sludge flow, waste rate, primary underflow, centrate flow, and centrate load. The FC included considerations for land acquisition, installation, tanks, piping, pumps, and other ancillary equipment. The MC included considerations for recycling/pumping, mixing, aeration, process specific chemical addition, chemical addition for pH adjustment, and other energy cost. RC where included for irregular costs for rehab or replacement of equipment. Lastly, AS and SP included recovered cost (S) with the sale of fertilizer, a product of the sidestream treatment process. The equipment was sized in accordance with WWTF industry standards and/or comparable to existing installation design criteria (Metcalf and Eddy 2003; Huang and Shang 2006; Reynolds and Richards 1995). Equipment, material, labor and other unit costs were referenced from CapdetWorks Version 3.0 (Hydromantis, 2014) and RSMeans 2018.

Water Management Unit Cost Development

The FC and MC for water management practices were developed referencing data published by municipalities or case studies evaluating unit cost of practice adoption. For SS, FC and MC were adapted from Ishii and Boyer 2015 which evaluated the adoption of source separation at a multi-residential dormitory at the University of Florida (Gainesville, FL). The cost for adoption of source separation will be highly driven by the cost to dual plumb the building therefore additional cost for dual plumbing was added to the FC assuming $1,250 per resident adapted based on dual plumbing cost of a multi-residential facility at University of California, Santa Barbara. The evaluation included total FC and MC which were adapted into per capita unit costs. For GIR, FC were obtained from a report documenting the installation of irrigation reuse systems at 38 residences (WSTB, 2016). The cost included FC per thousand gallons and MC were assumed to be 10% of FC (WSTB, 2016). For ER, published billing rate unit costs were obtained from five utilities in Colorado that offer ER and/or non-potable water services (Aurora, CO; Brighton, CO; Broomfield, CO; Colorado Springs, CO; Denver, CO). These values were obtained in 2017 and were adjusted to 2018 dollars and are assumed to represent PVC and account for all costs to install, own and maintain the non-potable water service system.

Stormwater Control Measures Unit Cost Development

SCM costs were developed as a function of acres treated by a given SCM. For EDB and BR, costs were adapted referencing the compilation of nutrient control cost developed by (USEPA 2015). The FC was obtained from the compilation of nutrient control cost based on SCM or comparable SCM types and a MC was assumed to be 10% of FC and adjusted to 2018 dollars.

Developed Unit Cost

Table S-4. Unit Present Value Cost (*PVC*) for Treatment / Practice Adoption (MLE = Modified Ludzack Ettinger, A2O = Anaerobic, Anoxic, Oxic, 5SBAR = 5-Stage Bardenpho, NS = Nitrite Shunt, SP = Struvite Precipitation, AS = Ammonia Stripping, CaRRB = Centrate and RAS Reaeration Basin, PAD = Post Aerobic Digestion, CA = Carbon Addition, FA = Ferric Addition, SS = Source Separation, GIR = Graywater Irrigation Reuse, ER = Effluent Reuse, EDB = Extended Detention Basin, BR = Bioretention)

Practice	*PVC*
MLE[1]	$ 2.50 / gpd
A2O[1]	$ 5.43 / gpd
5SBAR[1]	$ 7.31 / gpd
MLE + NS[1,2]	$ 2.35 / gpd
MLE+CA[1,3]	$ 3.82 / gpd + 6.58 / lb.-additional NO3-N removed
MLE+FA[1,3]	$ 3.82 / gpd + 1.90 / lb.-additional TP removed
MLE+CA+FA[1,3]	$ 5.14 / gpd + 6.58 / lb.-additional NO3-N removed + 1.90 / lb.-additional TP removed
MLE + SP[1,4]	$ 3.50 / gpd
MLE + AS[1,4]	$ 4.71 / gpd
MLE + CaRRB[1,4]	$ 5.33 / gpd
MLE + PAD[1,4]	$ 5.16 / gpd
SS[5]	$ 366.35 / capita
GIR[6]	$ 13.97 / 1,000 gal
ER[7]	$ 2.51 / 1,000 gal
EDB[1]	$ 19,255 / acre
BR[1]	$ 15,305 / acre

1Unit Cost adapted from USEPA 2015-A Compilation of Nutrient Control Cost Data

2Aeration Savings assumed based on Jimenez et al., 2014 estimated aeration savings and fraction of facilities energy cost due to aeration Rosso et al., 2018

3Additional Cost for chemical addition calculated based on USEPA, 2010 and CDM, 2007

4Additional Cost for sidestream implementation developed based on Engineering Cost Estimate for 10 MGD WWTF

5Unit Cost adapted from Ishii and Boyer, 2015

6Unit Cost adapted from WSTB, 2015

7Unit Cost adapted from published non-potable and reuse utility rates from Colorado utilities

Existing and Target Conditions

Historic water quality data and receiving water body flows were reviewed to identify existing conditions and acceptable stream loading conditions. The historical stream flows from 2008-2017were analyzed at USGS Gauge No. 06752280 which is immediately upstream of the second WWTF effluent (Figure S-1). The state of Colorado regulates annual TN and TP loading based on the annual median of the daily average flows with an allowable 1-in-5 year exceedance interval. Based on the last 10 years of data, this would be the 2009 annual median flow of 10.9 cfs, which represents the second driest year in the period (Figure S-1). Stream concentrations were obtained upstream of the study area, between the two WWTFs, and downstream of the study area from data reported to the state of Colorado in 2014 which is a recent requirement with the adoption of statewide nutrient regulatory requirements (Figure S-1).

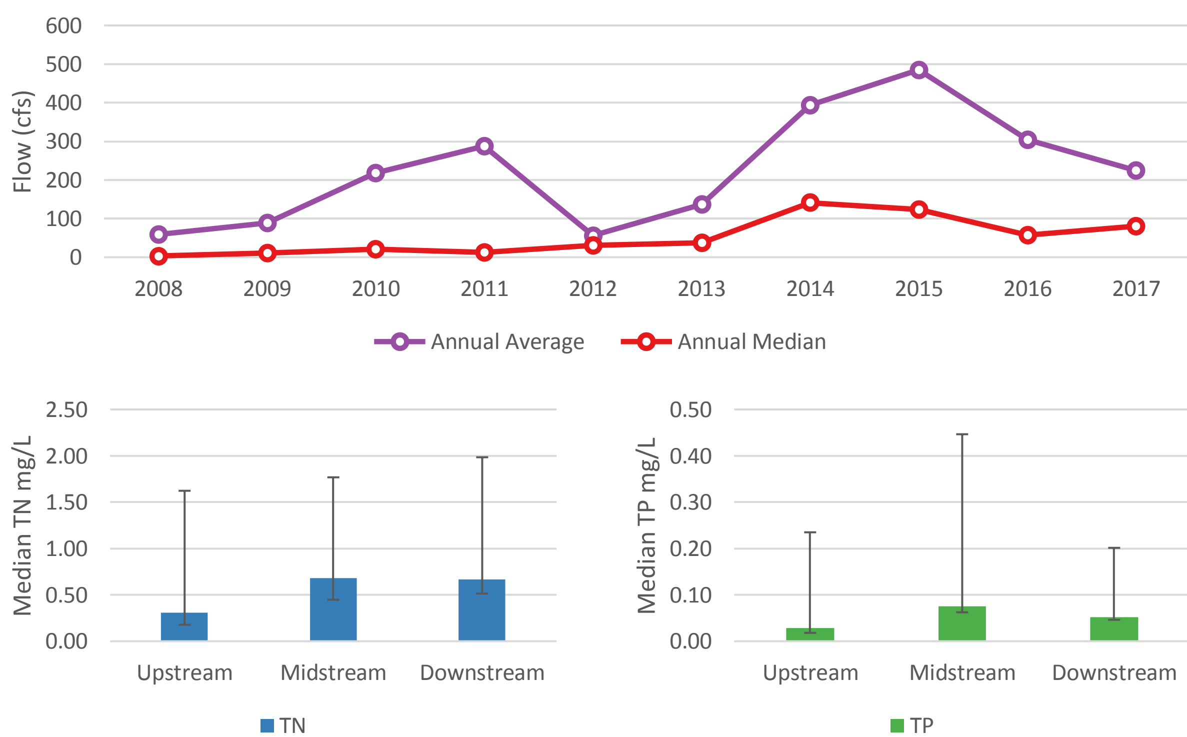

Figure S-1. USGS Gauge No. 06752280 Annual Median Streamflow and Average Annual Median with a 1-in-5 year recurrence interval (A), and Measured Total Nitrogen (TN) and Total Phosphorous (TP) Stream Concentrations (B)

Using this information, the current loading conditions were determined based current WWTF contributions using the 2014 reported nutrient data and the stormwater contribution estimated based on the simple method (Figure S-2). The baseline condition reflects annual average nutrient loading conditions. The existing baseline discharged load for TN and TP is 661,000 and 88,00 lb./yr respectively.

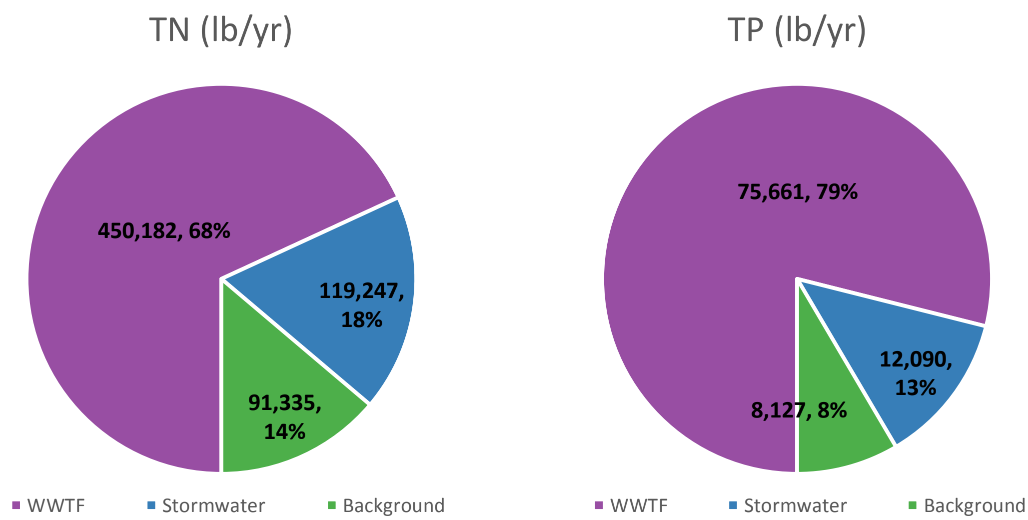

Figure S-2. Baseline Total Nitrogen (TN) and Total Phosphorous (TP) Load Conditions