Exploring the Return on Investment Case for Drinking Water Protection in the Upper Mississippi River Basin

University of California, Santa Barbara

A Group Project submitted in partial satisfaction of the requirements for the degree of Master of Environmental Science and Management for the Bren School of Environmental Science & Management by

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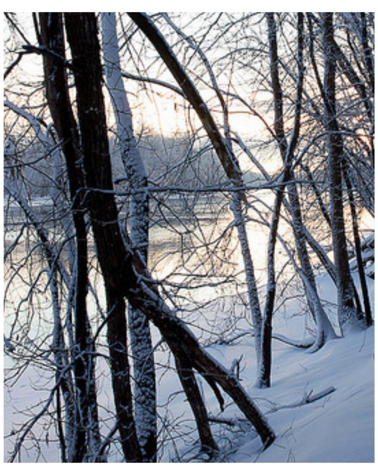

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Exploring the Return on Investment Case for Drinking Water Utility Engagement in Watershed Conservation

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The Group Project is required of all students in the Master of Environmental Science and Management (MESM) Program. The project is a year-long activity in which small groups of students conduct focused, interdisciplinary research on the scientific, management, and policy dimensions of a specific environmental issue. This Group Project Final Report is authored by MESM students and has been reviewed and approved by:

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Abstract

Watershed conservation efforts are at the nexus of three critical themes for environmental management: diffuse, nonpoint source pollution resulting in downstream, point-specific impacts; leveraging economics for positive environmental outcomes; and natural versus engineered grey infrastructure solutions. In Minnesota, wetland-to-cropland conversion and forest-to-cropland conversion are ranked amongst the highest in the nation, and state agencies report increasing nutrient and sediment concentrations in water bodies. As climate change makes agriculture more viable in northern reaches of the state, water quality impacts may become more pronounced. Source water protection, in the form of land and easement acquisition, is a potential strategy for preserving the integrity of watersheds in light of climate and land use changes. This project examined whether there is an economic incentive for water utilities in the Upper Mississippi River Basin to participate in source water protection efforts in exchange for decreased costs of treatment. Our analysis modeled changes to water quality based on future land use scenarios in the region, and compared the results with case studies of similar projects in New York, Ohio, and Brazil. Ultimately, a set of enabling conditions was developed to describe whether a return on investment would be attainable based on pollutant type and basin characteristics. In the absence of a positive return on investment for drinking water providers, a variety of additional benefits and beneficiaries to source water protection exist that could add significant value to conservation efforts.

Executive Summary

This report outlines research conducted to study the water quality impacts of increasing agricultural land cover in the Upper Mississippi River Basin and whether such impacts present a return on investment case for drinking water utilities to invest in watershed conservation. The work fills a noted research gap on the role of drinking water utilities in payment for watershed services programs. It broadly considers three key themes in environmental management:

- How can economics be leveraged for environmental outcomes?
- How do diffuse pollution sources contribute to targeted impacts?
- When should natural versus grey infrastructure solutions be implemented?

Over recent years, Minnesota has seen an increase in wetland-to-cropland and forest-to-cropland conversion. Minnesota state agencies are noting increasing nutrient and sediment concentrations. This poses potential water quality problems from a drinking water, aquatic life, and recreational standpoint. Narrowing on drinking water, this has the potential to prompt large capital investment in order for water utilities to properly treat water to drinking water standards. South of the study area, the Des Moines Water Works spent $1.2 million treating nitrate in 2015, and the City of Hastings, Minnesota spent $3.5 million in 2011 to construct its first water treatment plant to treat nitrates.

In 2015, The Nature Conservancy (TNC) created the Minnesota Headwaters Fund through a seed investment of $500,000 from Ecolab, Inc. Focused on proactively protecting water quality in the Upper Mississippi River Basin, the Minnesota Headwaters Fund enables downstream entities to invest in high-impact, upstream conservation projects. Drinking water utilities in Minneapolis, St. Cloud, and several smaller communities in the basin have participated in conversations with TNC about the Minnesota Headwaters Fund and its potential source water protection benefits. This project explores the numerous factors impacting that economic case and the potential for water utilities in the basin to achieve an ROI through investment in the fund in the form of reduced or averted treatment costs.

The study area of this project is the Upper Mississippi River Basin within the state of Minnesota. The basin sits in central Minnesota, covering an area of approximately 51,000 km^2. The northern region of the basin is predominantly forested land and forested wetlands. The southwest of the basin is predominantly agricultural. Dense urban areas in the watershed include the cities of Minneapolis, St. Paul, and St. Cloud. Given their size and their location along the Mississippi River, these three cities use surface water for drinking water. As such, they are of particular interest to this project as potential investors in land conservation to offset deteriorating water quality from increased agriculture.

The project was approached in four parts to analyze whether there is a return on investment case for water utilities to invest in land conservation. The first consisted of talking to water utilities in the study area to identify their needs and obtain relevant water treatment data. The second was to model water quality under current and future land use scenarios. The third synthesized acquired treatment costs to determine if an ROI was attainable for water utilities. The last interpreted the results as a set of enabling conditions for when water utilities may find it beneficial to invest in green infrastructure.

Utility conversations were conducted with St. Cloud, Minneapolis, and Hastings, Minnesota. St. Cloud and Minneapolis both use surface water as their drinking water source. Currently neither city is dealing with nitrate levels of concern. Hastings, serving a smaller population, uses groundwater as its drinking water source. Its nitrate levels were high enough that it prompted investment in a nitrate treatment facility in 2011. It is one of the few cities in Minnesota with a nitrate treatment facility.

Next, a model was implemented to calculate future nutrient and sediment levels in the basin. The Hydrologic and Water Quality System (HAWQS) was used to run three land use scenarios to calculate total nitrogen, total phosphorus, and sediment concentrations in the Mississippi River at two output locations: St. Cloud and Minneapolis. The three scenarios are: a baseline scenario to represent current conditions, a moderate agricultural expansion scenario that considers conservation work being done by TNC, and an aggressive agricultural expansion with the greatest increase in agriculture.

Multiyear averages for nine years of model results showed little differences in modeled outputs between the three scenarios. For St. Cloud, total nitrogen concentrations increased from a baseline of 1.8 mg/L to 2.12 mg/L for the moderate scenario and 2.13 mg/L for the aggressive scenario. Minneapolis total nitrogen concentrations increased from a baseline of 2.7 mg/L to 3.01 mg/L for the moderate scenario and 3.03 mg/L for the aggressive scenario. As a result of minimal changes in sediment and nutrient concentrations between scenarios, no ROI case for water utilities to invest in land conservation was found. While total phosphorus concentrations significantly predicted use of a treatment chemical, iron chloride ($FeCl_3$), chemical savings from a the modeled percent reduction in total phosphorus were minimal (approximately $10,200-$18,900 NPV in perpetuity) in comparison to the price of land acquisition for conservation (on the order of $10 million).

The analysis demonstrated that projected moderate and aggressive agricultural expansions are unlikely to result in sediment or nutrient concentrations of concern for water utilities. From this analysis and the literature review, the report synthesizes recommendations for future work in the Upper Mississippi Basin that fall under three areas: implications for climate resiliency, enabling conditions for a positive ROI, and targeting sub-basins.

Implications for Climate Resiliency

While land use change is a well-documented driver of water quality degradation, the scenarios modeled in this study were not significant drivers of water quality degradation. However, water quality parameters exhibited high inter-annual variability suggesting that year-to-year climatic drivers may have a greater influence on water quality. Climate change impacts on water quality in the Upper Mississippi River Basin are two-tiered. The first tier is the effect of warmer temperatures driving large-scale land use and land cover change, such as agricultural expansion into forested landscapes modeled in this study. The second tier is the increasingly variable climate trends associated with long-term climate change. One factor is precipitation variability. Low precipitation can result in higher pollutant concentrations while high precipitation increases pollutant loads.

While there is high uncertainty on what the impacts of climate change might be, public water supplies show high awareness of climate change as a possible challenge. As water utilities are faced with uncertainty associated with climate change, they may choose to explore holistic source water protection efforts, such as the one explored in this analysis.

Enabling Conditions for a Positive ROI

The findings of this study indicate that a purely economic ROI case for drinking water utility investment in Upper Mississippi River Basin conservation may be elusive. However, factors such as the nature of the pollutant of interest, the size of the basin, and the presence of regulatory drivers may set enabling conditions to achieve an ROI for water utilities.

As direct effect pollutants, such as sediment, increase or decrease in concentrations, they directly correspond to changes in treatment costs. Other contaminants, such as nitrate, require costly treatment to remove and could be referred to as threshold pollutants, as water utilities are not triggered to invest in expensive infrastructure and treatment technology until the contaminant exceeds a regulatory threshold. An ROI analysis for threshold pollutants would focus on the risk of exceedances of regulatory thresholds. Utilities with a low risk of exceeding regulatory thresholds are unlikely to realize an ROI through conservation investment.

The second condition, basin size, is likely due to smaller basins being more vulnerable to land use and land cover change than larger basins. For a large basin such as the Upper Mississippi River Basin to produce large water quality changes, the perturbations must be relatively large. Large basins also move larger volumes of water, leading to an in-stream dilution effect. However, neither basin size or type of pollutant affect an ROI case in isolation. Rather, water utilities and organizations invested in land conservation should consider basin size, type of pollutant, and regulatory drivers that will prompt changes in treatment technology and infrastructure.

Target Sub-basins

Since agriculture is predominant in the southwest portion of the basin, little to no increase in loading of water quality parameters were expected in this region. Rather, conversion of non-agricultural lands in the central and northern parts of the basin into agriculture were expected to be associated with the most significant changes in water quality parameters. HAWQS outputs were consistent with these expectations. Through this analysis, it is recommended that an effective management and conservation strategy is to target sub-basins with the largest increase in nitrogen, phosphorus, and sediment yields.

Project Objectives & Significance

Objectives

The objective of this project is to develop a return on investment (ROI) analysis for downstream water utilities to invest in upstream conservation via The Nature Conservancy's Minnesota Headwaters Fund. Throughout the project, the group:

- Interviewed representatives from at least one large, mid-size, and small water utility to gauge the interest and feasibility of investing in upstream conservation efforts at multiple scales;
- Modeled potential pollutant levels at utilities' source water intakes under a variety of land use scenarios, ranging from status quo to aggressive agricultural expansion into the Upper Mississippi River basin;
- Evaluated the potential for an investment in upstream conservation to generate an ROI for drinking water utilities in the form of reduced or avoided treatment costs.
- Developed a set of enabling conditions under which an ROI may be attainable for a drinking water utility.

Significance

A water fund is an institutional platform designed to cost-effectively harness nature's ability to capture, filter, store, and deliver clean and reliable water (The Nature Conservancy, 2017). In 2015, The Nature Conservancy (TNC) created the Minnesota Headwaters Fund through a seed investment of $500,000 from Ecolab, Inc. Focused on proactively protecting water quality in the Upper Mississippi River Basin, the Minnesota Headwaters Fund enables downstream entities to invest in high-impact, upstream conservation projects. In just over a year, the fund grew to over $3 million, well on the way to its three-year goal of $10 million, and has to date leveraged an additional $11 million in public funding (The Nature Conservancy, 2015). For communities in the basin, the fund has potential to deliver value in the form of green infrastructure, which refers to the use of natural and semi-natural areas like forests, wetlands, and grasslands to benefit human populations through the maintenance and enhancement of ecosystem services (Naumann *et al.*, 2011)

The Upper Mississippi River Basin is home to approximately 60 percent of Minnesota's population (Minnesota Pollution Control Agency, 2000). Over one million of those Minnesotans obtain drinking water directly from the river, specifically in the Twin Cities Metropolitan Area and upstream in St. Cloud (Minnesota Department of Natural Resources, 2010). The remainder receive drinking water from tributaries, lakes, or aquifers in the basin. Water utilities in the region have a financial interest in maintaining or improving the basin's surface and groundwater quality in order to manage treatment costs.

Current land management practices and land use trends threaten the health of the Upper Mississippi River and, consequently, drinking water quality. Recent research shows rates of cropland expansion throughout Minnesota on the order of 200,000 acres from 2008-2012, largely at the expense of forests, grasslands, and wetlands in the headwaters region (Lark *et al*., 2015). However, when targeted at nutrient sources and pathways, high-impact upstream conservation practices such as riparian buffers or land set-asides, have the potential to achieve measurable water quality improvements and help prevent future water quality deterioration (Tomer and Locke, 2011; Arabi *et al*., 2008). This project seeks to determine whether investments in such conservation practices can not only protect one of the world's most iconic rivers, but also result in an economic benefit for the cities that depend on it.

Drinking water utilities in Minneapolis, St. Cloud, and several smaller communities in the basin have participated in conversations with TNC about the Minnesota Headwaters Fund and its potential source water protection benefits. While these entities are investing their time, a solid economic case and implementation strategy could motivate drinking water utilities to invest money in the Fund. This project will explore the numerous factors impacting that economic case and the potential for various water utilities in the basin to achieve an ROI through investment in the fund. Moreover, the project's findings will serve as a framework by which other utilities may evaluate the potential for green infrastructure investments to meet their specific water quality challenges. The project will not only benefit drinking water utilities by providing a better understanding of enabling conditions that could result in an ROI on a water fund investment, but also fill a noted research gap on the role of utilities in conservation and payment for watershed services programs in the United States (Bennett *et al*., 2014).

Background

Water Quality and Upper Mississippi River Background

From its start at Lake Itasca through the northern Minnesota forests, the Mississippi River is nearly pristine; it is not until it reaches more agricultural and urban portions of central Minnesota that water quality deteriorates due to nutrient, sediment, and bacterial pollution (Minnesota Pollution Control Agency, 2017). As water quality declines downstream, water quality standards become more stringent, as the river must begin meeting drinking water requirements for over a million residents in the St. Cloud and Twin Cities Metropolitan Areas (Minnesota Pollution Control Agency, 2017; Minnesota Department of Natural Resources, 2010).

Despite a variety of contaminants in the Upper Mississippi River, this project focuses predominantly on nutrient and sediment pollution. High nutrient concentrations present an economic and public health concern for communities that rely on polluted water bodies as sources of drinking water and recreation (Dodds *et al*., 2008). Nitrate, a regulated drinking water contaminant, is linked to a variety of human health complications, including "blue baby syndrome" and thyroid dysfunction (USEPA, 2015). As a result, drinking water providers must meet a 10 mg/L nitrate (as nitrogen) drinking water standard, established under the federal Safe Drinking Water Act (Minnesota Pollution Control Agency and USGS, 2013). While still relatively low, nitrate concentrations in the Upper Mississippi River Basin have increased steadily in recent years, approximately 2-4 percent annually between St. Cloud and the Twin Cities (Minnesota Pollution Control Agency, 2013). Beyond drinking water concerns, nitrate and other nutrients in the Mississippi River also impact in-stream ecological health by contributing to eutrophication and reducing dissolved oxygen (Houser and Richardson, 2010), and contribute to a seasonally variable hypoxic dead zone in the Gulf of Mexico on the order of 20,000 km^2 each year (David *et al*., 2010; Turner *et al*., 2012).

While both point and nonpoint sources contribute to nutrient loading in the Upper Mississippi River Basin, nonpoint sources—especially those from agricultural activities—are particularly significant. The Minnesota Pollution Control Agency (MPCA) estimates that just 21 percent of total nitrogen loading in the basin is attributable to point sources, while agricultural groundwater, agricultural drainage, and cropland runoff account for a combined 49 percent (Minnesota Pollution Control Agency and USGS, 2013). The remaining sources of nitrogen in the Upper Mississippi River Basin include atmospheric deposition (13 percent), forestland (11 percent), and other nonpoint sources (6 percent) (Minnesota Pollution Control Agency and USGS, 2013). Though agricultural sources are significant in the Upper Mississippi River Basin, they contribute a smaller proportion of nitrogen to the river than in other, more agriculturally developed basins. For example, in both the Lower Mississippi River and Minnesota River basins, agricultural sources account for approximately 89 percent of total annual nitrogen loads in an average precipitation year (Minnesota Pollution Control Agency and USGS, 2013).

Among agricultural activities, corn and soybean cultivation is most closely associated with nitrogen pollution (Alexander *et al*., 2008). Given agriculture's impact on nutrient loading, changes in agricultural land use have significant implications for water quality in the Upper Mississippi River Basin. Donner *et al.* (2004) found that agricultural factors, including increasing fertilization rates and expansion of soybean production, greatly increased the magnitude and altered the timing of nitrogen export by the Mississippi River between 1960 and 1994. More recent expansions in corn production as a result of ethanol policies and higher commodity prices have the potential to further increase nutrient loadings (Secchi *et al*., 2010). In fact, one study modeling corn expansion onto only existing cropland in the Upper Mississippi River Basin estimated increases of up to 18.5 and 12 percent for nitrogen and phosphorus, respectively (Secchi *et al.*, 2010).

Studies linking projected agricultural land use conversions to modeled water quality impacts are scarce. Nevertheless, such land use changes and water quality impacts are increasingly being realized on the ground. Between 2008 and 2012, over 215,000 acres of cropland expansion occurred statewide in Minnesota, resulting in a loss of over 13,500 acres of forestland and over 25,600 acres of wetlands (Lark *et al*., 2015). These losses ranked the state first in the nation for wetland-to-cropland conversion, and second for forest-to-cropland conversion (Lark *et al*., 2015). This documented cropland expansion coincides with degrading water quality in the Upper Mississippi River, as MPCA announced plans in January 2017 to add 274 miles of the river to the state's impaired waters list (Bjorhus, 2017). This analysis explores how these ongoing changes in land use and water quality impact drinking water utility operations and long- term planning, and evaluates the feasibility of utility engagement in a basin-wide conservation initiative.

Linking Land Use and Water Quality

Global climate change and associated changes in temperature and precipitation have the potential to drive agriculture further north in upper reaches of the United States (Ramankutty *et al.*, 2002). Agricultural expansion into historically undeveloped areas in the Upper Mississippi River Basin threatens to jeopardize downstream water quality and hydrology and has been identified a source of concern by the MPCA.

Throughout the world, land use change can be linked to or driven by climate change, and is associated with alterations to water quality and nutrient loadings (Aichele, 2005; Legesse *et al.*, 2010; Tu, 2009). Prediction of future land use, including distribution and allocation of land uses in a given region can be derived from geographic and socioeconomic factors or generalized assumptions based on history of land use change in the region. The Annualized Agricultural Non-Point Source model (AnnAGNPS) is one example of a land use change model focused on agriculture (Yuan *et al.*, 2003). The projected land uses and conservation priorities selected have considerable effects on project outcomes; thus, it is important to use projections with high accuracy and derived from reputable sources.

Land uses directly impact groundwater and surface water quality within a given watershed. The effects of land use transition on increased nutrient loads and potential degradation of surface water quality have been quantified with a focus on both cropland expansion and urbanization (Schoonover *et al.*, 2005; Tu *et al.*, 2007; Woli *et al.*, 2008; Zampella *et al.*, 2007). The effects of land use transition on water quality and quantity are dependent on the nature, scale, and spatial distribution of the transitions themselves. Forested land, such as the land found in headwaters regions, is associated with improved water quality, while urbanized land is associated with increased nutrient loadings (Sliva and Williams, 2001). Urban sprawl, which can be characterized by per capita developed land use, has also been shown to significantly affect water quality through shifts toward developed land use and increase in population density (Tu *et al.*, 2007). Therefore, urban expansion into previously undeveloped land could result in significant impacts on water quality on a regional scale. Similarly, alterations in local hydrologic processes have been observed as a result of conversion from undeveloped or forested lands to urban or agricultural land uses (Aichele, 2005; Li *et al.*, 2009; White and Greer, 2006). Mixed agricultural use, such as livestock farming, cropping, and residence, was also found to significantly impact nitrate concentrations in groundwater (Choi *et al.*, 2007), and could also serve as a pathway for increased loadings of other pollutants.

The unique characteristics of different land uses, and the activities carried out on each, play a direct role in the amount and variety of nutrients and pollutants that may enter a watershed. In this project, the two land uses of primary concern are agricultural and undisturbed habitat/forest. Changes to vegetative coverage and soils as a result of cropland expansion will alter rates of evapotranspiration, infiltration, percolation, and absorption of water and chemicals applied to the land surface. Land use conversion may also affect the overall water balance, surface water temperature, and hydrologic cycle in the region (LeBlanc *et al.*, 1997). Physical alterations associated with agriculture may include flow modification, where water is diverted onto croplands for irrigation or implementation of tile drainage. Such alterations result in changes to the quantity of water available for runoff, streamflow and groundwater flow, in addition to changes in the chemical and biological processes of receiving water bodies (Dunne and Leopold, 1978). In the Mississippi River Basin, expansion of agricultural land has been linked directly to increased streamflow and baseflow as a result of conversion from perennial to seasonal crops (Zhang and Schilling, 2006). Overall, there is a strong relationship between land use types and the quantity and quality of water available downstream (Gburek and Folmar, 1999).

Spatial distribution of land uses may also play an important role for surface water quality, as it affects the processing and retention of nutrients traveling through different land uses. Land uses adjacent or in close proximity to streams make more significant contributions as compared to the aggregate, average land use over a larger area (Basnyat *et al.*, 1999; King *et al.*, 2005; O'Neill *et al.*, 1997). Various pollution control measures make use of this fact through filtration of polluted water from agricultural fields or urban development prior to its entrance into a water body. Riparian buffer zones may play an important role in sequestering sediments and nutrients in runoff prior to entering streams, as well as processing nutrients in groundwater (Basnyat *et al*., 1999; Schoonover *et al*., 2005). Combined, changes to the spatial distribution of land use within the Upper Mississippi River Basin could have significant impacts on water quality in the region, depending on the scale of perturbation relative to the entire basin. In addition to source water protection, widespread implementation of pollution control measures like buffer strips could provide an effective means of achieving water quality standards.

Headwaters and undeveloped, forested land are a land use of particular interest to this project. Headwaters will be targeted for conservation by TNC due to their positive impacts on water quality in the region. Headwaters are directly connected to downstream waters and significantly influence the supply, fate, and transport of nutrients in a given watershed. The hydrologic and biogeochemical processes associated with headwater streams affect the timing and distance of nutrient transport downstream, and also influence flow paths and residence times of solutes (Alexander *et al*., 2007).The role of headwaters in the Upper Mississippi River Basin may be significant in protecting water quality, and conversion of headwaters regions to agricultural land could result in significant water quality impacts.

Agricultural land use and croplands account for roughly 73 percent of nitrate contributions to the surface waters of Minnesota in an average precipitation year (Minnesota Pollution Control Agency and USGS, 2013). Within an agricultural parcel of land, there are a number of unique pathways which may deliver nitrate and other pollutants to receiving water bodies. These are surface runoff, tile-line transport, and leaching to groundwater which eventually reaches surface waters. Depending on the specific characteristics of a given parcel of agricultural land, as well as the specific agricultural practices being applied to it, the total amount of nitrate reaching surface waters may vary from less than 10 pounds per acre to over 30 pounds per acre (Minnesota Pollution Control Agency and USGS, 2013). The variables which may play a significant role in the transport of nitrate from agricultural land into surface waters fall into a few main categories: climate, land surface characteristics, soil characteristics, stream characteristics, and other flow-related variables (Brown *et al.*, 2011). In Minnesota, cropland tile drainage is the most significant contributor to nitrate in surface water, and intensive tiling is associated with the highest nitrate yields per watershed. Tiling, by nature, results in potentially permanent alterations to hydrology (Schilling *et al*., 2008). However, tile drainage may not be ubiquitous with cropland expansion in the Upper Mississippi River Basin due to its relatively well-drained soils, compared to other parts of the state or country (Minnesota Pollution Control Agency and USGS, 2013; Sugg, 2007).

This project models the effects of various land uses and land-surface characteristics on water quality and streamflow based on current conditions, expected changes to land use through USGS FORE-SCE projections, and potential conservation efforts through the Minnesota Headwaters Fund. The project utilizes the Hydrologic and Water Quality System (HAWQS) web-based water quality and quantity modeling tool, developed by Texas A&M University, the United States Department of Agriculture (USDA), and the United States

Environmental Protection Agency (US EPA). The HAWQS modeling tool is based on the Soil Water Assessment Tool (SWAT) developed by the USDA to model the effects of these changes and determine potential nutrient loadings in surface waters of the Mississippi River. SWAT has been used in previous studies to model the effects of land use and climate change on nutrient loadings in surface water, and can give detailed loadings of individual solutes (El-Khoury *et al*., 2015; Li *et al*., 2009; Schilling *et al*., 2008).

Other models that were considered for this project include SPAtially Referenced Regression On Watershed attributes (SPARROW) and Better Assessment Science Integrating Point and Nonpoint Sources (BASINS) (Alexander *et al.*, 2007; Smith *et al*., 1997; Tong and Chen, 2002). Each model operates on different levels of spatial and temporal scales, but for the purposes of this project and given the basin's large spatial extent, HAWQS was determined to be the most practical (Baffaut *et al*., 2015).

Utility Engagement in Payment for Ecosystem & Watershed Services

In 2015, a total of 419 projects that conserve or rehabilitate green infrastructure were operational in watersheds in 62 countries around the world (Forest Trends, 2016). In North America, 107 of these projects were identified, with 8.9 million hectares under watershed management and transactions worth $3.8 billion between buyers and sellers of watershed services. A third of these transactions were directed at forest restoration and enhancement, while 38 percent of the money was used for sustainable agricultural and pastoral management interventions. A fifth of the transactions were diverted toward grassland conservation. Almost all of the money was used for conserving private lands, and very little was used for public lands (Forest Trends, 2016).

Water funds, like the Minnesota Headwaters Fund, are one of the several mechanisms used for watershed protection. Other mechanisms include public subsidies for watershed protection, water quality trading and offsets, and environmental water markets (Forest Trends, 2016). The value of public subsidies for watershed protection in 2015 was $23.7 billion worldwide while the value of transactions for water funds during the same period was around $565 million (Forest Trends, 2016). The value of transactions in water quality markets and environmental water markets was only in tens of millions of dollars worldwide. Although the public subsidies for watershed protection are significantly higher, the water fund model is growing in popularity with the number of programs increasing from 81 to 95 between 2013 and 2015 (Forest Trends, 2016). TNC has seven operating water funds in the United States (The Nature Conservancy, 2017). Most of the money for water funds in the United States comes from the public sector. In 2015, public sector/government contributed to 84 percent of the money invested in water funds, while the private sector and non-governmental organizations contributed 12 and 4 percent, respectively (Forest Trends, 2016).

One of the earliest and often cited examples of utility engagement with source water protection in the United States is of the Catskills watershed in New York. The city of New

York avoided building a multi-billion dollar water filtration facility and $300 million in annual operating costs by investing $1.5 billion in watershed conservation efforts starting in 1997 (Postel and Thompson Jr., 2005). For different parties, the reasons for investment in watershed protection are different. The drivers for investment in watershed protection include regulatory compliance requirements, by voluntary decisions of the customers and utilities to engage in such projects, or both. However, most watershed protection projects face similar barriers.

A survey of the 419 watershed investment projects globally indicates that future regulatory uncertainty, local stakeholders and partnership challenges, and lack of effective technical and financial capabilities for the projects are the key barriers to watershed investment (Forest Trends, 2016). Analysis of 19 source water protection programs in the United States found that gaining support from key utility representatives was a major challenge (Bennett *et al*., 2014). Another barrier was demonstrating and linking site-specific conservation actions with improvements in water quality and quantity downstream (Bennett *et al*., 2014). One study found that water utilities in the western United States raised issues of long-term watershed resiliency from implementing source water protection projects (Carpe Diem West, 2011). Although these studies provide an overview of the barriers to utility investment in source water protection, there can be several region-specific barriers to these projects. Hence, this project interviews utilities in the study area and obtains their inputs on potential barriers to investing in the Minnesota Headwaters Fund.

Return on Investment Analysis & Case Studies Source Water Quality Impacts on Drinking Water Treatment

Changes in water quality parameters at drinking water source intake locations could result in direct impacts on treatment costs. In an article submitted to the Water Resources Research journal, changes in water quality parameters such as turbidity and total organic carbon were related to treatment costs per 1,000 gallons, based on expenditures on chemicals, pumping, and granular activated carbon (Heberling *et al*., 2015). As water quality deteriorates, water treatment facilities incur greater costs by either having to increase treatment materials or by having to invest in long-term capital projects. Water treatment costs can then be weighed against the costs of investing in source water protection.

Within the Upper Mississippi River Basin, in 2011 the City of Hastings spent $3.5 million on its first water treatment plant to maintain nitrate levels below the maximum contaminant level of 10 parts per million (Minnesota Department of Health). In Iowa, Des Moines Water Works is planning on spending $15 million to double the size of its nitrate removal facility to handle growing nitrate levels (Des Moines Register). As these two cities demonstrate, capital investment to treat pollutants of concern can require millions of dollars of funding. Conservation efforts to protect source water can reduce treatment requirements. The link between treatment costs, water quality parameters, and nutrient abatement costs needs to be clearly understood to measure the incentives of conservation efforts.

In this section, several case studies are explored to understand how they approach studying and quantifying the importance of watershed conservation programs. Brazil has been successful in implementing various water fund projects in collaboration with TNC across different watersheds. The Camboriú, Brazil case study is provided as one which, similar to this project's scope, has engaged water utilities to reduce operation costs and considers the implications of downstream benefits. This project is in earlier stages in comparison to Camboriú and is taking a different approach toward funding, collecting private investments from individuals, businesses, corporations and foundations rather than a utilities surcharge (TNC Rio Grande Water Fund, 2014). Second, the Taos County, New Mexico case is provided as a domestic case study that highlights interaction with both private landowners and federal agencies and associated federal regulations. The Catskill Watershed case study looks at the success story of New York City in using source water protection to mitigate treatment costs. Last, the East Fork Watershed case study specifically looks at how upstream conservation benefits translate to actual treatment costs for water treatment facilities (Heberling *et al*., 2015).

Camboriú River, Santa Catarina State, Brazil

The recently created payment for watershed ecosystem services (PWS) program in the Camboriú River watershed in Santa Catarina State, Brazil was implemented to reduce concentrations of total suspended solids (TSS) at the municipal water intake and its associated water treatment costs and water losses. To evaluate the potential for TSS reduction, an extensive analytical framework was applied to analyze costs and benefits of scenarios with and without watershed conservation interventions.

The Soil and Water Assessment Tool (v. 2012 Rev. 637) was used to conduct a hydrological analysis of the ecosystem structure modeling land use and land cover (LULC) with and without the PWS program (Kroeger *et al.*, 2017, p. 3). High-resolution one-meter LULC data from 2004 – 2012 was used to predict near- to medium-term changes as well as a scenario in 2025 with the program in full implementation with critical lands conserved (Kroeger *et al.*, 2017, p.15).

The scenario with the PWS program includes riparian and headwater area restoration and degraded upland forest restoration. Landowners would receive compensation for the maintenance of interventions on their property and annual opportunity costs associated with not using the restored land (Kroeger et al. 2017, p. 11).

The program achieved an ROI (ROI > 1) at a 43-year horizon for the municipal water company, EMASA (Kroeger *et al.*, 2017, p. 3), strictly looking at the benefits of reducing TSS concentrations. However, the analysis also created a scenario including down the river co-benefits of reduced flood risk and increased water supply security during the tourist high season. Using a Brazil-specific household willingness to pay to increase water supply security and flood control, the average household willingness to pay was several orders of magnitude higher than what would be necessary to provide a positive ROI (Kroeger et al. 2017, p. 3).

Taos County, New Mexico, United States

The Rio Grande Water Fund (RGWF) identified four geographic focal areas, with an aim to restore forests, increase wildfire protection, and provide clean water security (Kruse *et al*., 2016, p. 2). Kruse *et al*. (2016) explored an ROI case for Taos County, New Mexico. In contrast to the Brazil case study, which includes co-benefits to support its positive ROI, the Taos County ROI focuses on avoided costs of wildfire associated with market goods and services in Taos County. It does not focus on downstream water supply benefits.

Scenarios were created for with and without a RGWF forests treatment. The treatment scenarios were generated using simulation runs, experiences in other locations, and generally accepted science and practice on fuel treatment effectiveness (Kruse *et al.*, 2016). Accordingly, there is some degree of inaccuracy in the treatment scenario that should be acknowledged.

The ROI analysis focused on market values for property, goods, and services impacted by RGWF (Kruse *et al.*, 2016). It ignores trying to monetize non-market benefits given its difficulty, and instead provides a conservative ROI that could only be further supported by ignored co-benefits.

Given the uncertainty of the fires and the associated benefits, benefits were distributed uniformly over a 20-year horizon, discounted at three percent. Initial treatment costs were also spread across a 20-year horizon discounted at three percent, and ignored maintenance costs (Kruse *et al.*, 2016). Benefits were found to greatly surpass conservation costs, by $32.9 million for a small fire event scenario and $68.2 million for a large fire event scenario (Kruse *et al.*, 2016). While benefits were calculated for the different land categories—federal, developed, and agricultural—the analysis did not include potential stakeholder treatment cost allocations (Kruse *et al.*, 2016).

The Rio Grande Water Fund is looking to leverage private investments from individuals, businesses, corporations, and foundations to scale up its restoration efforts, then use an executive committee representing various stakeholders and investors to determine which projects receive funding (TNC Rio Grande Water Fund, 2014).

Catskill/Delaware Watershed, New York, United States

The New York City drinking water supply system provides 1.1 billion gallons of water daily to nine million people (Watershed Agricultural Council, 2017). To date, this constitutes the largest unfiltered drinking water supply in the United States (Department of Environmental Conservation, 2018). All of this water is provided from surface water resources, with 90 percent received from the Catskill/Delaware Watershed and the remainder received from the Croton Watershed upstate (Watershed Agricultural Council, 2017). Agricultural land use is a relatively small fraction of both watersheds, accounting for just 5% and 6% of the Catskill/Delaware and Croton Watersheds, respectively (Pires, 2004).

In 1989, promulgation of the Surface Water Treatment Rule under the Safe Drinking Water Act altered the regulations around surface water filtration and disinfection. The City estimated the cost of a new filtration plant at $8-10 billion for construction and $1 million for daily operation (Department of Environmental Conservation, 2018).

To avert these costs, the City entered into a Memorandum of Agreement (MOA) with the U.S. Environmental Protection Agency in 1997. Under the MOA, the City receives a filtration avoidance determination in exchange for watershed-scale source water protection measures. Such measures include land acquisition, water quality regulations, and community partnerships (Pires, 2004).

From an ROI perspective, the Catskill/Delaware case study is widely considered a success, as the City avoided billions of dollars in infrastructure investment by committing hundreds of millions to watershed conservation (Pires, 2004). Today, New York City maintains reliable water quality, though the Department of Environmental Conservation notes persistent challenges related to sediment and turbidity in the Catskill Watershed, as well as eutrophication throughout the watershed systems (Department of Environmental Conservation, 2018).

East Fork Little Miami Watershed, Ohio, United States

While other case studies have looked at the overall costs and benefits of source water protection, Heberling et al. (2015) specifically looks at how drinking water treatment plants (DWTPs) can benefit from source water protection. The case study focuses on the Bob McEwen Water Treatment Plant (BMWTP), located in Batavia, Ohio. BMWTP has a capacity of 19 million gallons per day (MGD), and uses conventional clarification, filtration, and chlorine disinfection for its water treatment. Pollutants of concern are bulk total organic carbon (TOC), TOC as a source for disinfection byproducts, pesticides, dissolved manganese, algal toxins, and algal-derived taste and odor compounds (Heberling *et al*., 2015).

Costs calculated encompassed chemical costs, pumping costs, and costs related to treatment with granular activated carbon (GAC). BMWTP acquired GAC as a response to pesticide concerns, but also for algal toxins, algal-derived taste and odor compounds, and disinfection byproduct precursors. The water quality parameter of interest in the ROI study was total phosphorus. Its affinity for natural clay particles can increase water turbidity. Phosphorus is also linked to harmful algal blooms, which can be toxic but also increase turbidity.

Heberling *et al*. (2015) used two time series modeling approaches, error correction model (ECM) and polynomial distributed lag model (PDL), to translate treatment costs to source water protection efforts. The ECM develops a cost function for the treatment plant based on several water quality variables measured in the raw water. Variables include TOC, pH, turbidity, and water temperature. The PDL translates estimates of pollutant load reduction to significant water quality variables for the plant.

Overall, the study found that a 1% decrease in turbidity in the source water would decrease treatment costs by 0.02% immediately and by an additional 0.1% over subsequent days. For BMWTP, a 1% decrease in turbidity would result in savings of $1123 per year from decreased treatment costs (Heberling *et al*., 2015).

Translating treatment costs to source water protection, the two models show that a 1% decrease in total phosphorus load can decrease treatment costs by approximately $168 annually. A 1% decrease in total phosphorus load was shown to require 3,921 lb/year of phosphorus abatement upstream. Purchasing this abatement from upstream farms could range from $11,763-$105,867 depending on tillage and cover crops. In this case, BMWTP saw no economic incentive to pay for nutrient abatement upstream. However, the article notes the economic case might be different if a water treatment plant did not already possess advanced treatment processes or if pollutant loads were higher.

Methods

Study Area

The Upper Mississippi River Basin begins at Itasca State Park and drains 15 major watersheds within 21 individual counties in the state of Minnesota. The basin receives drainage from a variety of land cover and land uses, including forest, agricultural, prairie, and urban areas. Major rivers within the watershed include the Mississippi River, Rum River, Crow River (South Fork), Crow River (North Fork), Elk River, Clearwater River, Sauk River, Long Prairie River, Crow Wing River, Redeye River, Pine River, and Leech Lake River. Major lakes include Gull Lake, Whitefish Lake, Big Sandy Lake, Mille Lacs Lake, Leech Lake, Lake Winnibigoshish, Cass Lake, and Lake Itasca. The basin is subdivided by three distinct eco-regions as defined by the MPCA, including Northern Lakes and Forests in the northeast half of the basin, North Central Hardwoods in the southwest half of the basin, and Western Corn Belt Plains in the southwestern tip of the basin.

The modeled study area contains the entirety of the Upper Mississippi River Basin, minus three HUC 10 subunits. The three HUC 10 subunits that are excluded from analysis are 701010605, which is self-contained and does not convey flows into the rest of the basin, as well as 701020608 and 701020609, which drain to the river downstream of the source water intakes for the utilities of interest in this study. Since these HUC 10 subunits do not contribute flows into the rest of the watershed upstream of the drinking water intake locations within the project study area, they were excluded from the scope of analysis within HAWQS. Therefore, the modeled study area is approximately 51,052 km^2.

The northern and northeastern portions of the basin are where the largest expanses of forestland, forested wetlands, and lakes occur. These areas constitute the headwaters of the watershed. In the headwaters region, growth and development, including agricultural expansion, are the primary threats towards the existing landscape and water quality. Agricultural uses are the predominant land use within the southwestern half of the Upper Mississippi River Basin, as seen in the study area maps for the project (**Figure 1**). The most densely populated and urbanized areas are in the Minneapolis-St. Paul and St. Cloud metropolitan areas at the southeast and south central portions of the basin, respectively.

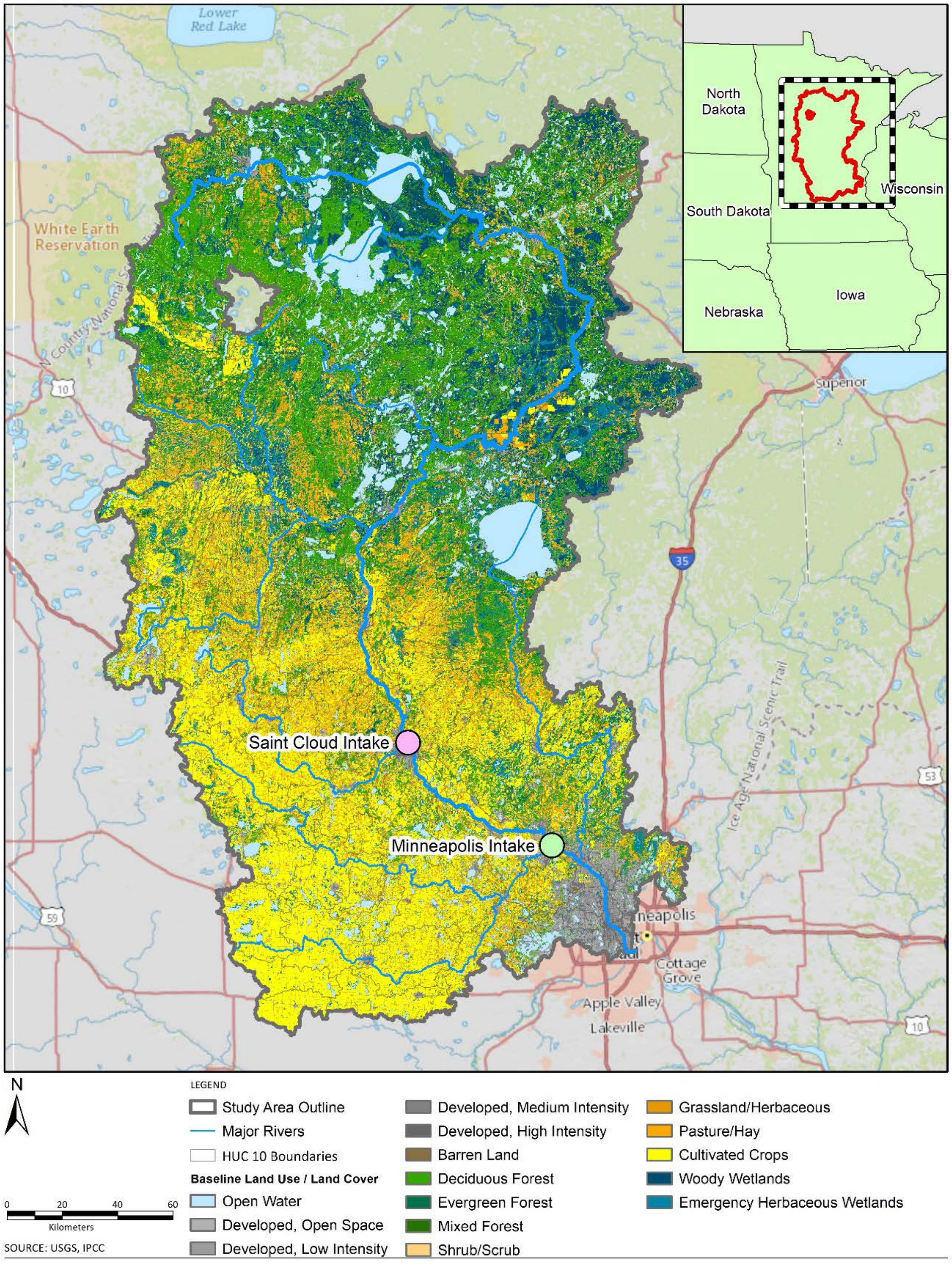

Figure 1. Baseline land use in the Upper Mississippi River Basin in Minnesota. Agricultural land is indicated in yellow and orange, forest and forested wetlands are indicated in green, and urban land is indicated in grey. City of Minneapolis and City of St. Cloud source water intakes are indicated in green and pink, respectively.

Climate within the Upper Mississippi River Basin is generally characterized by sub-humid continental conditions, with a range of weather patterns dependent on the season. Mean annual temperature across the basin is approximately 38 F for the northern portions, increasing to 44 F in the southern portions. Average monthly temperatures range from a low of 6 F in January to a high of 68 F in July in the northern portion of the basin, while in the southern portions average temperature ranges from a low of 11 F in January to 72 F in July. Upper level prevailing winds generally flow from west to east within the basin. Warm, humid air originating from the Gulf of Mexico prevails during the summer months, while cool, dry air dominates during the winter months. Precipitation varies throughout the basin, with average annual precipitation ranging from 24 inches in the northwest portions of the basin to 28 inches in the southeast portions (Gunard, 1985; Kuehnast *et al.*, 1982).

The Upper Mississippi River Basin lies within the Central Lowland geomorphic/ physiographic provinces, and largely consists of flat to rolling moraines and glacial outwash plains.

Hydrologic and Water Quality System (HAWQS) Model

Due to the large size of the study area and associated data and computing requirements necessary to perform water quality modeling in ArcSWAT, a cloud-based water quality modeling tool called the Hydrologic and Water Quality System (HAWQS) was used for the project. HAWQS is a national-scale decision-support tool developed by Texas A&M University and the US EPA to perform watershed modeling tasks (Yen *et al.*, 2016). HAWQS runs on the latest version of the Soil Water Assessment Tool (SWAT 2012 rev. 659) to simulate different water quality parameters like sediments, nutrients, and Biochemical Oxygen Demand (BOD). HAWQS has predetermined model parameters as default values and built-in national level datasets for all the SWAT inputs, including climate, land use, and soil data. For instance, users can choose climate data from datasets like Parameter-elevation Regressions on Independent Slopes Model (PRISM), National Oceanic and Atmospheric Administration's National Climate Data Center (NCDC), or Next Generation Weather Radar (NEXRAD) system data, all of which are readily available in the HAWQS interface. Similarly, soil data from the State Soil Geographic (STATSGO) dataset and reservoir data from the National Inventory of Dams was used in HAWQS (HAWQS Inputs, 2017). Land cover and land use data are taken from the National Land Cover Database (NLCD 2006) and the Cropland Data Layer (CDL). The model also includes elevation data from the National Elevation Dataset (NED), stream network data from the National Hydrography Dataset Plus (NHDPlus), and aerial deposition from the National Atmospheric Deposition Program (NADP). Several crop management related data are taken from datasets made available by the USDA (HAWQS Inputs, 2017). Tile drainage can be an important feature of an agricultural landscape affecting water quality parameters. HAWQS incorporates tile drainage on a national scale by assuming agricultural land use on soils categorized as "very poorly drained" will incorporate subsurface drainage. As a result of these built-in datasets, HAWQS users can create a SWAT project easily.

In any watershed model, SWAT divides the entire watershed study area into Hydrologic Response Units (HRUs), which are parcels of land in the study area with similar land use, soil type and slope. After this, using data such as precipitation, soil parameters and other inputs, SWAT calculates the surface runoff and sediment yield. The surface runoff in SWAT is calculated either by the Soil Conservation Service (SCS) curve number method or the Green & Ampt infiltration method (Arnold *et al.*, 2009). In most cases the SCS curve number method is employed. Once the surface runoff is calculated, other parameters like peak runoff are calculated. These two values, along with the soil erodibility factor, cover management factor and other such parameters, are used in the Modified Universal Soil Loss Equation (MUSLE) to calculate the sediment yield from different HRUs in the watershed. SWAT also calculates the overland and channel flows in the watershed and the channel processes modeled include the movement of water, sediments, nutrients, and pesticides (Arnold *et al.*, 2009).

A HAWQS analysis has four major steps, namely initialization, customization, output management, and output demonstration (Yen *et al*., 2016). HAWQS can perform watershed modeling at three different resolutions of the Hydrologic Unit Code (HUC 8, HUC 10 and HUC 12) and at three different time steps (daily, monthly, and annual). For this project, the HAWQS model was initialized at a HUC 10 resolution and the watershed with the HUC 10 number 0701020607 was selected as the downstream watershed, since this sub-basin contains the intake location for the Minneapolis drinking water utility. HAWQS automatically selects all the HUC 10 watersheds that drain into the selected downstream sub-basin. As a result, the entire study area covers about 51,000 sq.km with 109 HUC 10 sub-basins in North Central Minnesota. The model area, including model output locations relative to utility source water intake locations, is provided in **Figure 2**.

The model was run using the PRISM weather dataset with the simulation period starting January 1, 1990 and ending December 31, 2015. The simulation period includes with a 5-year warm-up period. The latest version of SWAT (SWAT 2012 rev. 659) was used to run the model in monthly time steps. As a result, the model gives outputs for every month from January 1, 1995 to December 31, 2015. Average annual water quality and quantity outputs from 2002-2010 were chosen as the default for this study, since NLCD 2006 is the default land use layer in HAWQS and NLCD 2006 is expected to be more representative of the land use in the chosen time period. Beyond the HAWQS baseline, two separate HAWQS scenarios were created by replacing NLCD 2006 with future land use scenarios for the year 2027. For all the three HAWQS scenarios, namely the baseline, moderate and aggressive agricultural expansion scenarios discussed below, no other model settings were altered aside from the land use layers, in order to isolate the effect of land use change on water quality by controlling for other factors.

A multi-year average (2002-2010) was chosen for comparison of model outputs since model outputs from either a single year or just a few years that have extreme precipitation events can skew the interpretation of results. One other aspect to note about these scenarios is that, although the moderate and aggressive agricultural expansion scenarios

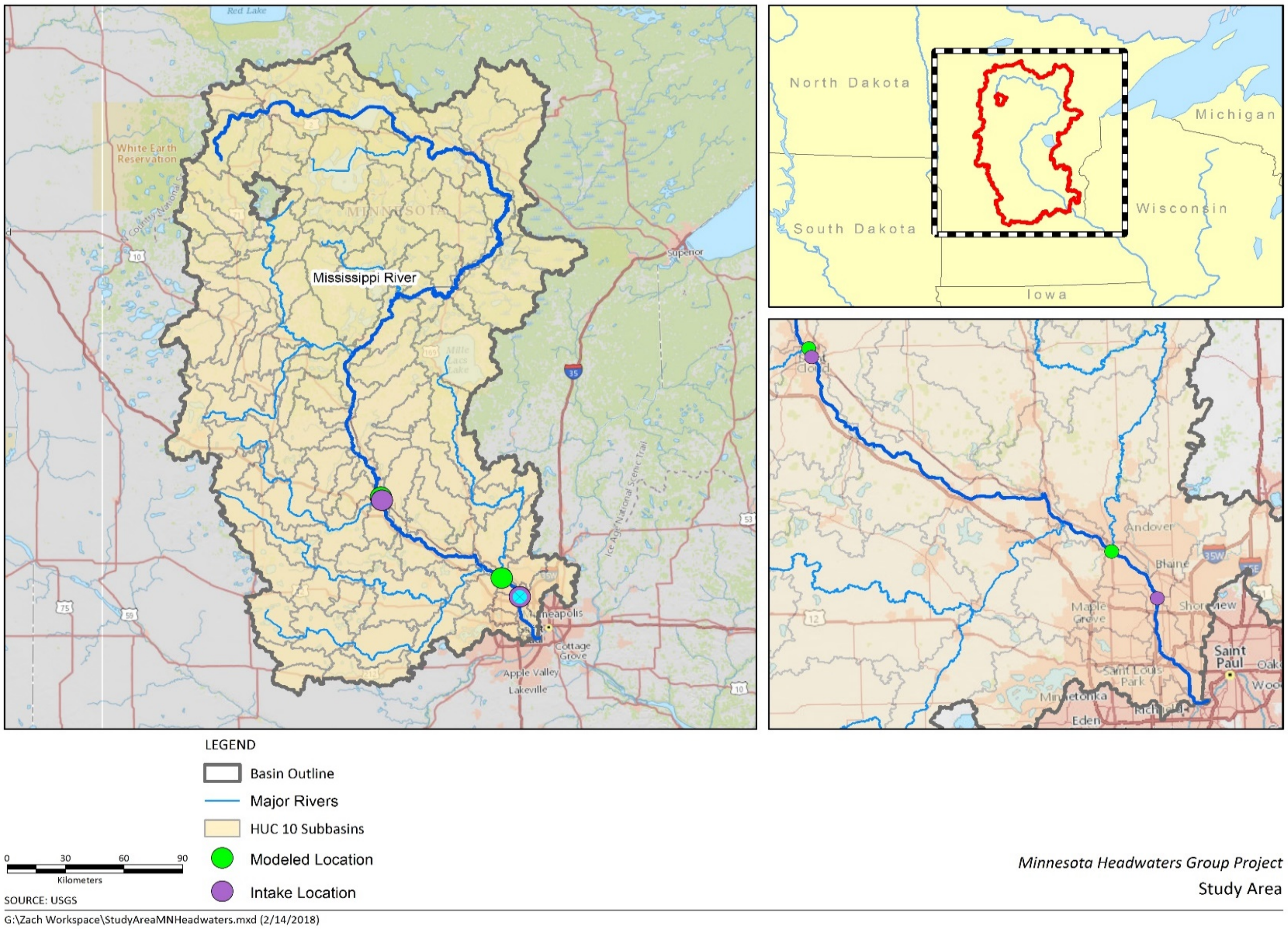

Figure 2. The Upper Mississippi River Basin, as delineated by the state of Minnesota, from headwaters to Minneapolis. Source water intakes for the cities of Minneapolis and St. Cloud, Minnesota are indicated in purple. Model output locations for the Hydrologic and Water Quality System (HAWQS) are indicated in green.

represent land use in 2027, the model outputs are compared for the years 2002-2010 in all the scenarios. This is because HAWQS has precipitation data only through 2015 and, as a result, the model simulation date cannot exceed 2015. One way to interpret the 2002-2010 model outputs under 2027 land use scenarios is to assume these represent the 2022-2030 model outputs.

In a traditional SWAT analysis done using ArcSWAT, after obtaining initial model outputs from using default weather and land use data, the model is calibrated by changing input parameters to make sure that model outputs like flow, sediments or total suspended solids (TSS), Total Nitrogen (TN) and Total Phosphorous (TP) loads in the river reach at the study area outlet are in close agreement with the observed values in the real world. Usually calibration follows a step-by-step process where the stream flow is calibrated first, followed by TSS, and then by TN and TP (HAWQS Calibration Process, 2017). The calibration process involved modifications to model input parameters like phosphorous/nitrogen percolation co-efficients, Universal Soil Loss Equation (USLE) soil erodibility factor, and many others. The model is then validated by comparing the model results from the post-calibration time period with real world water quality and quantity values from the same time period. For example, Ahiablame et al., modeled the effects of land use change on water quality for a 1,605 km^2 watershed in South Dakota for which they performed a SWAT analysis using ArcSWAT. Their analysis used NLCD 2011 for their default land use scenario, with 2005-2014 as the model simulation period and 2005 was used as a warm-up year. The model outputs from years 2005-2013 were used for model calibration, and the model outputs from 2014 were used for validation.

HAWQS has been calibrated for different water quantity and quality parameters at 79 watersheds at the HUC 8 level in the United States. Within these HUC 8 level watersheds, HUC 10 and HUC 12 level watersheds are assigned the same calibrated parameters that were assigned to the HUC 8 watersheds (HAWQS Calibration Process, 2017). For the study area of this report, HAWQS has been calibrated for HUC 8 level watersheds draining into three different locations, namely the Mississippi River at St. Paul (a few miles downstream of Minneapolis), Royalton (upstream of St. Cloud), and Winona in Minnesota (HAWQS Calibration Locations, 2017). The model has been calibrated for flow based on the data from United States Geological Survey (USGS) stream gage stations at these three locations and it has been calibrated for sediments at the Winona location. Ideally, the HAWQS model of the study area would be calibrated and validated for model outputs at the outlet points of interest, specifically at Minneapolis and St. Cloud drinking water intakes. However, due to data and time constraints, no calibration was performed for the model in this study beyond the built-in calibration provided by HAWQS. Hence, the absolute values of model outputs may not be in close agreement with the observed values in the real world. Nevertheless, the percentage changes between water quantity and water quality outputs between different model scenarios, which is of the greatest interest to this study, do not differ between non-calibrated and calibrated models (HAWQS Webcast, 2017).

Land Use Change Scenarios

In order to assess the changes to water quality parameters associated with land use change within the HAWQS model, a number of additional land use scenarios must be obtained for comparison with the baseline scenario. The baseline scenario in the HAWQS model is based on NLCD 2006 data, and is supplemented by the Crop Data Layer (CDL) from 2011-2012.

In this analysis, land use predictions and forecasted agricultural expansion are based on previous work by The Nature Conservancy in the Upper Mississippi River Basin for identifying priority conservation parcels, as well as by the United States Geological Survey (USGS) Forecasting Scenarios of Land-use Change (FORE-SCE) modeling framework supplemented by the International Panel on Climate Change (IPCC) Special Report on Emissions Scenarios (SRES). The FORE-SCE framework utilizes global, national, and local driver variables to provide a variety of spatially-explicit future land use and land cover change scenarios (*USGS EROS :: Land-Cover Modeling*). FORE-SCE land use projects are based upon the IPCC SRES climate scenarios, which account for changes in population, income, and agricultural productivity data to develop statistical models and project future land use for a variety of future climate scenarios. The goal of choosing future land use scenarios was to incorporate enough individual scenarios with varying degrees of agricultural expansion in order to assess the sensitivity of the basin to such changes, and to determine the relative changes to water quality parameters that would result from the changes. Two future land use scenarios have been modeled in the project: a moderate agricultural expansion scenario and an aggressive agricultural expansion scenario.

The moderate agricultural expansion scenario was chosen to represent a conservative level of agricultural expansion and urban expansion within the basin, in addition to other relevant land use changes. For this scenario, a land use scenario developed by the USGS EROS FORE-SCE land use modeling tool, and based on the Intergovernmental Panel on Climate Change (IPCC) SRES A2 Emissions Scenario, was used. The A2 Emissions Scenario was developed to account for economic and technological development, population change, energy consumption, and land use change. The A2 emissions scenario assumes that world population will increase continuously, and surpass 10 billion by 2050, coupled with regional economic development and relatively slow technological advancement (Nakicenovic *et al*., 2000). Land use and land cover prediction maps generated by the FORE-SCE modeling framework showed a modest increase in agricultural expansion within the study area, and were also generally consistent with results from previous studies focusing on cropland conversions (Lark *et al*., 2015). Thus, they were deemed suitable for the modeling purposes of the project.

Although FORE-SCE land use projections extend to the year 2100, the data from year 2027 was chosen to represent land use and land cover change ten years from the time of modeling and allow for a reasonable decision-making timeframe. This moderate agricultural expansion scenario may be thought of as a scenario where TNC's conservation interventions are incorporated, because there is relatively minimal cropland expansion into areas identified for conservation by TNC in previous assessments (approximately 15,000 out of 87,000 total acres identified at risk of agricultural conversion were in agricultural land use by 2027 under this scenario).

The aggressive agricultural expansion scenario was generated by combining the FORE-SCE IPCC A2 (2027) land use-land cover (LULC) map used in the minimal agricultural expansion scenario with priority parcels identified by TNC for conservation through the Minnesota Headwaters Fund, which were subsequently assumed to be converted into agriculture. Although these parcels are identified by TNC as conservation priorities, their conversion into agriculture is representative of a “worst case” scenario for the headwaters region, where no conservation has been achieved and agriculture has expanded into the priority areas. In this scenario, approximately 42,000 additional acres transition from forested land to agriculture by 2027.

Applying Land Use Changes to HAWQS

Similar to SWAT, the HAWQS model operates by dividing the modeled basin into individual HRUs. HAWQS identifies HRUs based on specific land use, slope, and soil classifications within a sub-basin (US EPA, 2017). This project employed HAWQS modeling at the HUC 10 watershed resolution. Based on land use, slope, and soil classifications across the 109 HUC 10 watersheds in the project area, HAWQS identified 11,168 unique HRUs.

HAWQS input data includes an area for each HRU (in km^2) as well as a fraction, which indicates the given HRU’s proportion of the total HUC 10 area. Land use changes are applied in HAWQS by updating this fraction value for each HRU in relation to the expansion or reduction of a certain land use class in a HUC 10.

Updated fraction values for each of the 11,168 HRUs were applied using a Microsoft Excel-based tool that generated new areas for each of the HRUs based on a given land use change scenario and recalculated the proportion of each HUC 10 accordingly. First, the area of each NLCD land use classification in every HUC 10 in 2006 (baseline) and 2027 (future) was determined for land use change scenarios using ArcGIS. This step was used to determine the change in the relative proportion of each NLCD land use class for each HUC 10 between the baseline and future years.

Once these areas were input into the Excel tool, they were reclassified from NLCD land use classes to HAWQS land use classes. **Figure 3** illustrates how NLCD land use classifications corresponded to HAWQS land use classifications. When one NLCD land use classification

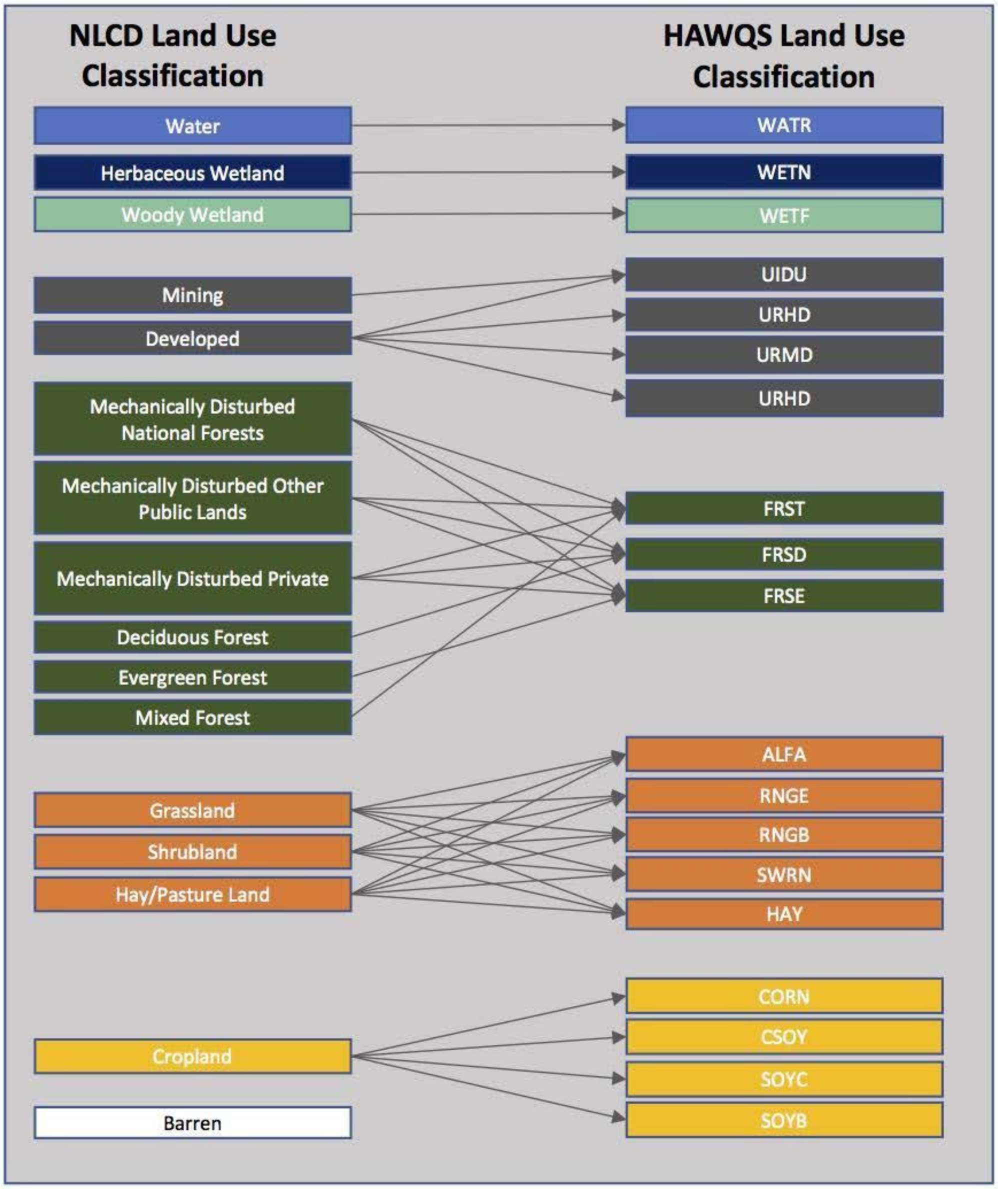

Figure 3. Chart explains how NLCD land use classifications were recategorized to fit HAWQS land use classification categories. Where broader NLCD classifications were broken into more specific HAWQS classifications, the baseline percent cover of HAWQS classifications in the HUC 10 sub-basin were applied to the NLCD class. For example, NLCD "Cropland" was broken into HAWQS categories based on the relative proportions of corn, corn-soy rotation, soy-corn rotation, and soybeans in the baseline scenario for a given HUC 10.

was divided among multiple HAWQS land use classifications, it was done so according to the original breakdown of the land use classifications in the HAWQS baseline. For example, CORN, CSOY, SOYC, and SOYB constitute the four primary cropland categories in HAWQS. If CORN comprised 30 percent of the of the total of those categories in a given HUC 10 according to the HAWQS baseline, then 30 percent of the NLCD Cropland classification would be allocated to the CORN land use class in HAWQS for that sub-basin.

Once NLCD land use classes from the 2006 and 2027 years were converted to HAWQS land use classifications, the relative proportions for each land use class in each HUC 10 were calculated. This change in relative proportions between the 2006 and 2027 years was then applied to the HAWQS baseline scenario using a basic cross-multiplication formula:

$$P^* = \frac{H_b \times S_f}{S_b}$$

where P^* is the new relative proportion of the land use class in a given HUC 10, H_b is the relative proportion of that land use class under the HAWQS baseline, S_f is the relative proportion of the land use class in the future year of the land use scenario (2027), and S_b is the relative proportion of the land use class in the baseline year of the land use scenario (2006).

Applying the changes in relative proportion from the land use change scenarios to the HAWQS data did not conserve total area in each sub-basin, resulting in new relative proportions that did not sum exactly to one. This was due to differences between the underlying land use datasets used for HAWQS and the land use change scenarios. To normalize these values and ensure total area was conserved between the HAWQS baseline scenario and the new scenario, the newly calculated relative proportions for each land use class were divided by the sum total of all the land use class proportions in the basin.

Next, the new, normalized relative proportions for each land use class were multiplied by the total area of the HUC 10 sub-basin to obtain new areas for each land use class. The area for each HRU was then updated relative to the change in area for the land use class in the HUC 10 using a basic cross-multiplication formula:

$$\frac{Baseline\ HRU\ Area}{Baseline\ Area\ of\ Land\ Use\ Class\ in\ HUC\ 10} = \frac{New\ HRU\ Area}{New\ Area\ of\ Land\ Use\ Class\ in\ HUC\ 10}$$

Finally, after all HRU areas were updated, new fraction values were recalculated by dividing the new HRU area by the total HUC 10 area. This process was completed for all HUC 10 in the study basin and repeated for each land use change scenario.

This methodology is subject to limitations and assumptions. First, while it did not dramatically alter the new land use relative proportion values, normalizing values to ensure relative proportions at the HUC 10 level summed to exactly one did introduce some additional error. Ultimately, this was determined to be necessary in order to conserve area at the study basin and sub-basin scale between the HAWQS baseline and the future land use scenarios.

Second, this study updated HRU areas, but did not alter the other components of HRUs—namely soil type and slope class. Effectively, this assumes that as areas of a certain HRU expand or retract based on the overall land use trend in a sub-basin, the areas of a given soil type and slope class expand or retract similarly. This is unrealistic, as land use conversions would likely occur across slope and soil classes. This limitation occurs largely because HAWQS does not spatially define HRUs for users, making it impossible to tell whether land use changes are occurring across slope and soil types on the ground.

Finally, this methodology does not allow for more rapid expansion or retraction of specific land cover classes within broader land use types. Different sub-classes of forest, urban, and cropland all account for the same fraction of total forest, urban, and cropland in the future as they do in the baseline scenario. For example, if urban industrial land use (UIDU) constitutes 2.5 percent of all urban land in a HUC 10 in 2006, it was assumed to constitute 2.5 percent of all urban land in the HUC 10 in 2027. This assumption results from the reclassification of NLCD land use classes into HAWQS land use classes, as the land use change scenarios conducted on NLCD data only captured trends across the broader land use categories.

Utility Conversations

Phone conversations were conducted with three water utilities in the Upper Mississippi River Basin to gain an understanding of how water utilities in the study area differ in terms of various parameters including water quality problems, populations served, and treatment technologies. Each phone conversation lasted 45-60 minutes and was followed up by e-mail correspondence to gather requested water utility data where available. Conversations followed a standardized script in order to collect similar information across all three utilities. The water utilities contacted are summarized in **Table 1** below:

Table 1. Utility characteristics for drinking water providers in Minneapolis, St. Cloud, and Hastings, Minnesota. Information was obtained through phone conversations and e-mail correspondence.

Water Utility	Location	Population Size	Facility Capacity	Water Source
City of St. Cloud Public Utilities	St. Cloud, MN	Over 70,000	24 MGD	Surface Water
Minneapolis Water	Minneapolis, MN	500,000	160 MGD	Surface Water
City of Hastings Public Works	Hastings, MN	23,000	--	Groundwater

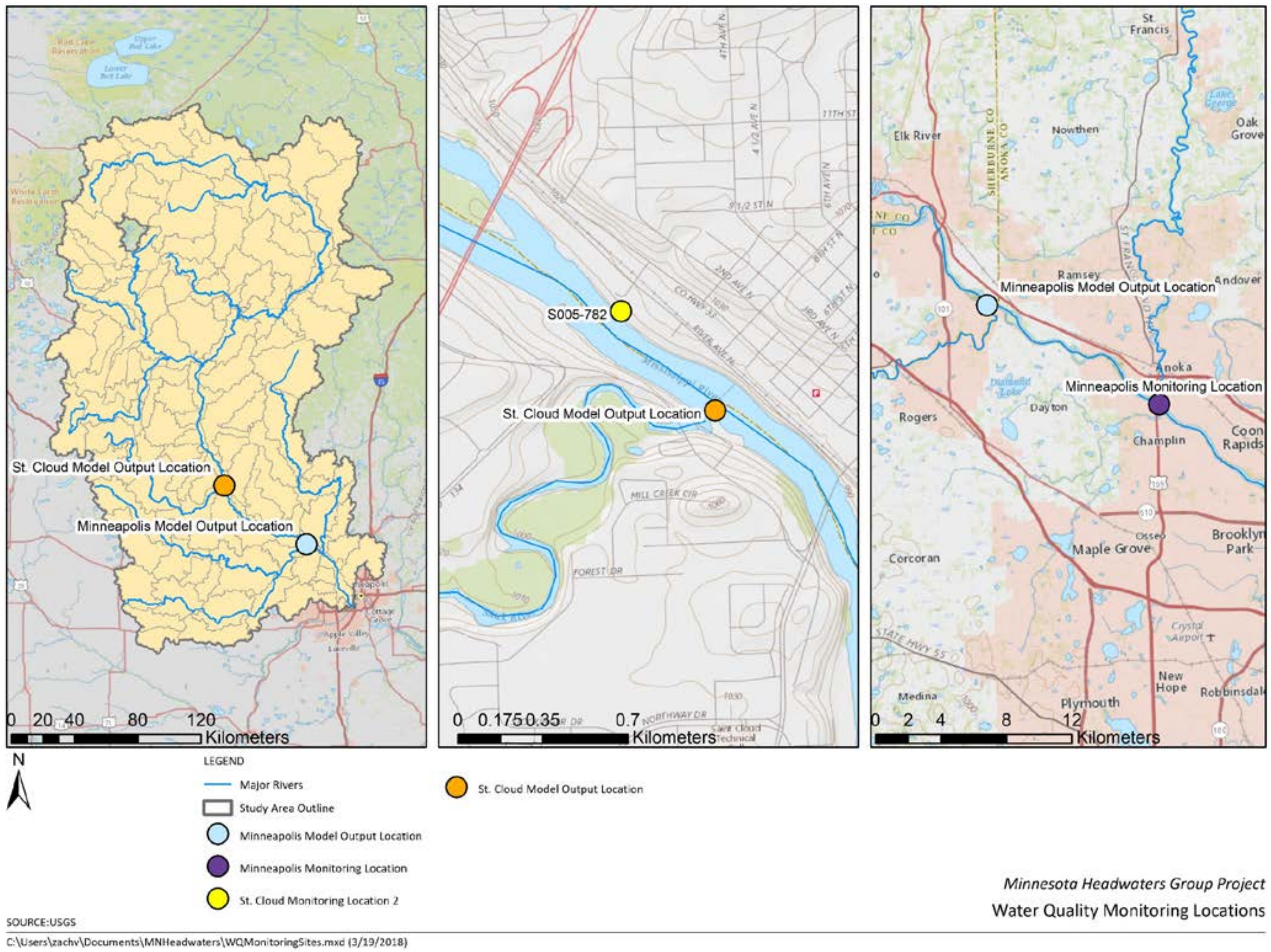

Figure 4. Left: St. Cloud and Minneapolis model output locations within the larger Upper Mississippi River Basin. Middle: HAWQS model output used to represent the St. Cloud drinking water intake and two water quality monitoring stations with observed data used to for comparison with model outputs. Right: HAWQS model output used to represent the Minneapolis drinking water intake and one water quality monitoring station with observed data used to for comparison with model outputs.

Water Quality Data

The HAWQS model output data was compared with observed water quality data. One water quality station was used for comparison with the HAWQS model output point used to represent the Minneapolis intake location. The site is UM8716. Data for this site was collected from the EPA Storage and Retrieval (STORET) database. Data was obtained for the years 2002-2010.

Data was also gathered to compare with the St. Cloud HAWQS intake. One water quality station—S005-782—was used for comparison. Data for the site was collected from STORET. No data was available for years 2002 and 2003, therefore, data used is for 2004-2010. Data for the site was sparser than for site UM8716. Approximately half the amount of data entry points for the St. Cloud site were available compared to 230 data entry points for the Minneapolis site.

Data gathered to compare to the HAWQS data was collected to be as close as possible to the HAWQS model intake. **Figure 4** shows the location of the HAWQS model intakes in St. Cloud and Minneapolis relative to the location of water quality monitoring stations. While the locations vary, environmental conditions should be similar enough to present a similar range in water quality values. Another challenge was the sparsity of water quality data points. Neither water quality station contains continuous data points. Rather, few points were collected for most months in a year. More water quality data would give a greater direct comparison to HAWQS data.

Results & Discussion

Insights from Utility Conversations

Conversations with water utility representatives provided a greater understanding of their treatment capacity and the different contaminants of concern. Salient points from each of the conversations are summarized below.

St. Cloud

While St. Cloud does not obtain water from the Sauk River directly, it is a tributary to the Mississippi River. Several feedlots are located along the Sauk River, creating a concern for nitrogen and ammonium. Another concern is sediments from erosion and runoff. St. Cloud does not express particular concern around nitrate pollution, with raw water concentrations around 1.5 mg/L and the nitrate drinking water standard set at 10 mg/L. Additionally, St. Cloud has conducted pilot studies with UV and ozone to stop using chlorine in their treatment process. As of now, chemicals are the largest line item in their treatment costs. Chemicals are used to remove color from water for aesthetic preferences.

St. Cloud has a source water protection plan, though it has not been updated in 10 years. Both regulatory compliance and reducing economic risk are drivers for the water utility's interest in a Headwaters Fund. St. Cloud realizes that anything that impacts water quality upstream of their intake has an effect on them.

Minneapolis

The Mississippi River is the only source of drinking water for the City of Minneapolis, which serves Minneapolis residents and functions as a wholesale distributor to other nearby municipalities. Generally, the river is a reliable source for the City, with minimal pollutant spikes near the intake. The City's drinking water treatment process employs lime softening techniques, chloramine disinfection, and filtration. Specific pollutants of interest to the City include organics—measured by UV-254—as well as occasional increases in ammonia during snowmelt. The City also monitors taste and odor, which can be influenced by in-stream eutrophication and algal blooms.

Like most drinking water utilities, regulatory drivers dictate planning and decision-making for the City of Minneapolis. The City is currently monitoring a shifting regulatory landscape with respect to various contaminants of emerging concern, including disinfection byproducts like N-nitrosodimethylamine (NDMA) or micropollutants (i.e., microplastics). However, the City also acknowledged that while regulatory drivers and economics guide a considerable amount of the utility's decision-making, municipal entities make investments for a variety of reasons. Specifically, the region's ethic of water resources conservation and a general interest in long-term source water protection were cited as potential motivators of a watershed conservation strategy.

Hastings

Hastings, unlike the other two water utilities interviewed, is on a groundwater system. The City of Hastings has a water treatment plant with the primary purpose of reducing nitrate levels. The treatment plant was put into service 10 years ago at a cost of $3.5 million, and it is one of the few nitrate treatment plants in Minnesota. Water is treated to nitrate levels of around 4-5 ppm using an ion exchange system. Near Hastings, agricultural practices in the community in combination with more permeable soils have made nitrate contamination increase at a quicker rate.

The City of Hastings is becoming more aware of the importance of source water protection. However, current regulatory efforts are confined within city limits, and Hastings does not contain many agricultural properties. Therefore, minimal work has been taken by Hastings to work with the agricultural industry to reduce nitrate contamination.

Analysis of Land Use Change Scenarios

When comparing land uses in the baseline scenario (2006) to those in the moderate and aggressive scenarios in 2027, there were a number of significant changes that occurred. **Figure 5** shows a comparison of land uses by category within the study area for the three modeled land use scenarios. Individual land uses were combined into larger categories for a more direct comparison between functional land use groupings and to filter out land use transitions between similar land uses, such as newly developed wetland to open water or forested wetland. These categories are based on the HAWQS Land Use Classification categories shown in **Figure 3** (Applying Land Use Changes to HAWQS Section). The cropland category contains the CORN, CSOY, SOYB, and SOYC land uses, the forested land category contains the FRSD, FRSE, and FRST land uses, the urban land category contains the UIDU, URHD, URLD, and URMD categories, the wetlands category contains the WATR, WETF, and WETN categories, and the rangeland category contains the ALFA, HAY, RNGB, RNGE, and SWRN land uses.

The moderate and aggressive future land use scenarios experience similar trends when compared to the baseline scenario. The largest increase by category is urban land uses, with an additional 650 km^2 of area added for the moderate scenario and 655 km^2 for the

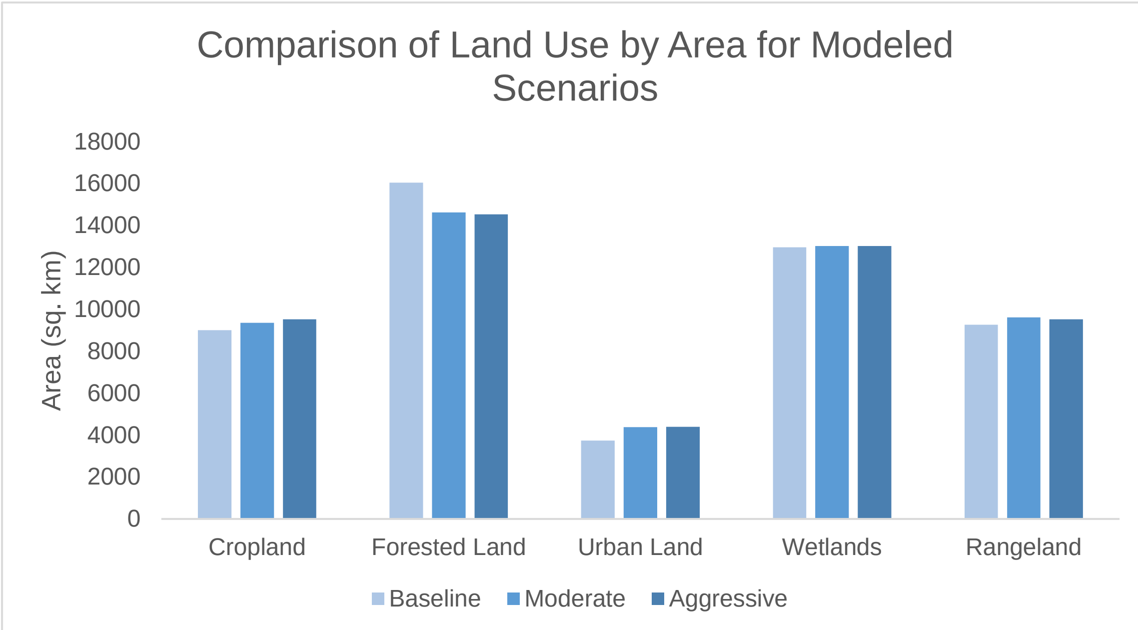

Figure 5. Comparison of land use by category for the three modeled land use scenarios within the study area. Land uses were divided into five categories for comparison, including cropland, forested land, urban land, wetlands, and rangeland.

aggressive scenario. Similarly, there are increases in the total area of cropland by approximately 350 km^2 for the moderate scenario, and approximately 525 km^2 for the aggressive scenario. Rangelands experience comparable increases, of approximately 350 km^2 for the moderate scenario and 265 km^2 for the aggressive scenario. Wetlands experience slight increases in total area of approximately 65 km^2 and 58 km^2 for the moderate and aggressive scenarios, respectively. The only land use category that experienced a decrease in total area between the two future scenarios was forested land, with approximately 1410 km^2 lost in the moderate scenario and 1500 km^2 lost in the aggressive scenario. Since the net area of all land use change must equal zero for the study area, it is evident that the gains in cropland, urban land, rangeland, and to a lesser degree wetlands, are a result of direct losses to forested land.

Model Fit

As previously mentioned, water quality stations contain sparse data with few points per month. In contrast, HAWQS produced daily and monthly data. Given the limitations in observed data, a data point to data point comparison between HAWQS daily data and observed water quality data, such as with a time series analysis, would not give an accurate assessment of the model fit. Rather, boxplots were used to verify that the HAWQS daily output fell within the same range as observed water quality data.

HAWQS daily data occasionally produced spikes in total nitrogen (TN) and total phosphorus (TP) concentrations that were not validated patterns by any of the collected water quality data. These spikes accounted for 2 percent of the Minneapolis HAWQS daily data and less than 1 percent of the St. Cloud HAWQS daily data and were not included in the analysis.

Observed TN data for Minneapolis falls within the modeled TN data (**Figure 6**). For most months, the median for HAWQS and the water quality site data are within 0.5 mg/L of each other. However, HAWQS' daily data produced a broader range in TN concentrations skewed to the right, as is evidenced by all HAWQS months having several outliers on the upper limit. Data for TP at the Minneapolis model output shows that HAWQS tends to overestimate TP during the winter months, November through March (**Figure 7**). For the remainder of the year, the HAWQS boxplots and the nearest monitoring station (UM8716) boxplots are better aligned. Similar to TN, HAWQS data for TP is skewed to the right with several upper limit outliers.

HAWQS validation at the St. Cloud intake used one water quality site (Site S005-782), which had fewer data points than the Minneapolis water quality site. HAWQS daily data falls within a similar range as Site S005-782 (**Figure 8**). Similar to the Minneapolis location, HAWQS overestimates TP for the winter months, but is better aligned during the remainder of the months (**Figure 9**). HAWQS data is skewed to the right with several outliers for TN and TP at both the Minneapolis and St. Cloud model output locations. This is not observed in the data for the water quality sites; however, water quality site data is also more limited. Overall, HAWQS produces TN and TP concentrations that are in line with observed water quality data. Of important note is that both HAWQS and water quality site TN concentrations are well below levels that would raise concerns for drinking water treatment (i.e., the nitrate maximum contaminant level of 10 mg/L (as nitrogen)).

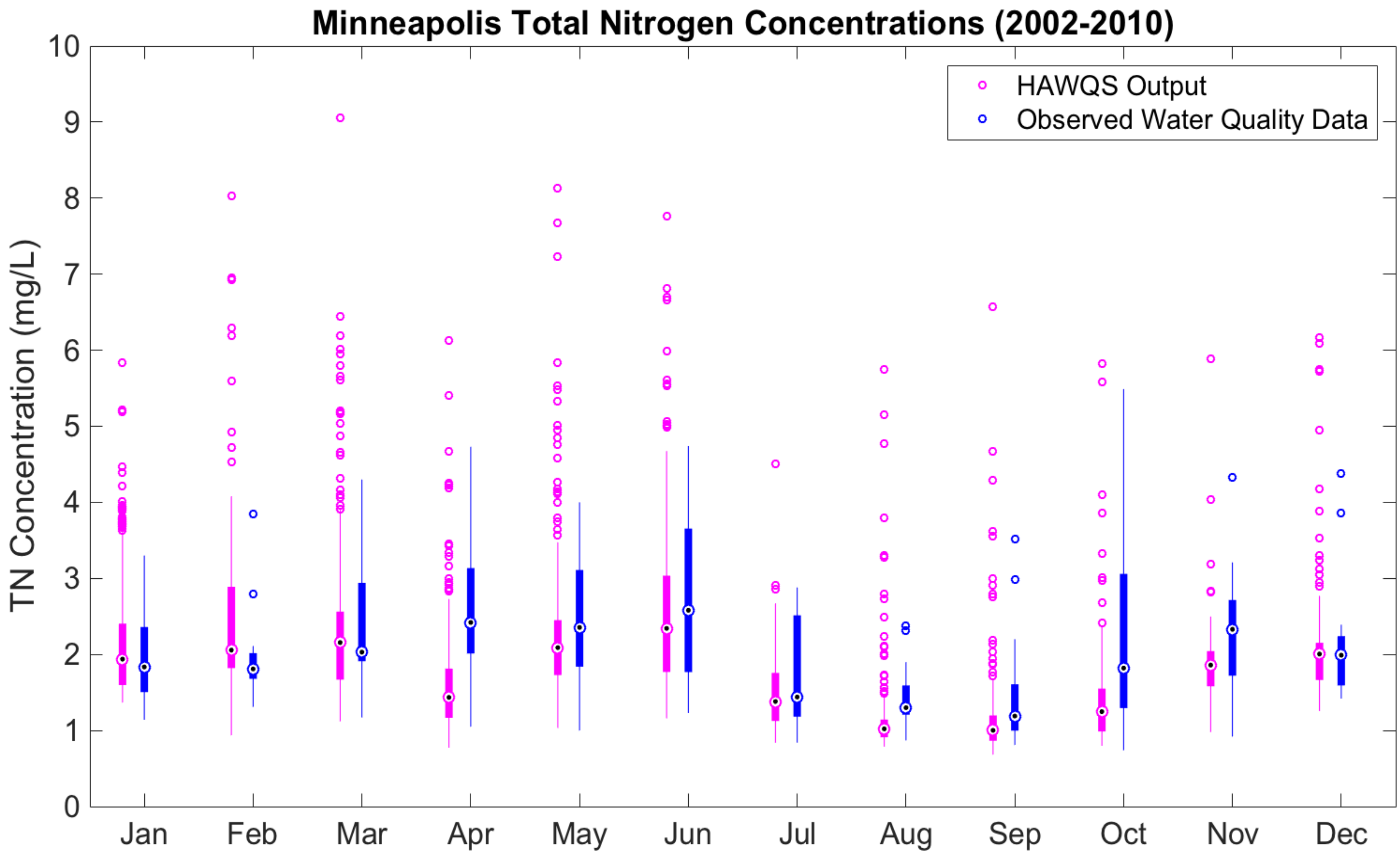

Figure 6. Minneapolis total nitrogen concentrations by month for the years 2002-2010. HAWQS data is in pink, water quality site UM8716 is in blue. The box indicates endpoints of the interquartile range (IQR, 25th – 75th percentile), with white circle with black dot at sample median. Whiskers extend to final observation within 1.5 × IQR. Outliers are shown as open circles.

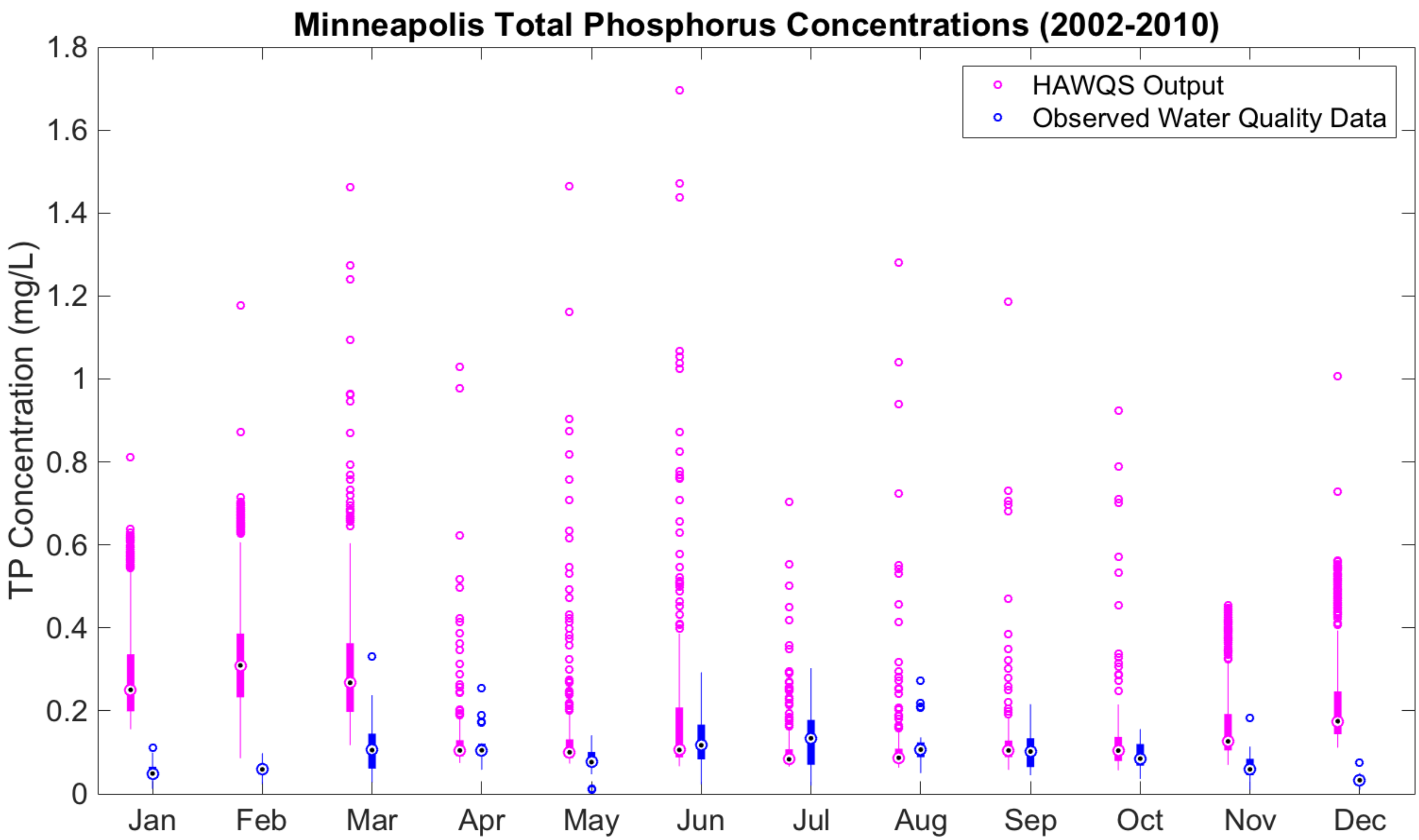

Figure 7. Minneapolis total phosphorus concentrations by month for the years 2002-2010. HAWQS data is in pink, water quality site UM8716 is in blue. The box indicates endpoints of the interquartile range (IQR, 25th – 75th percentile), with white circle with black dot at sample median. Whiskers extend to final observation within 1.5 × IQR. Outliers are shown as open circles.

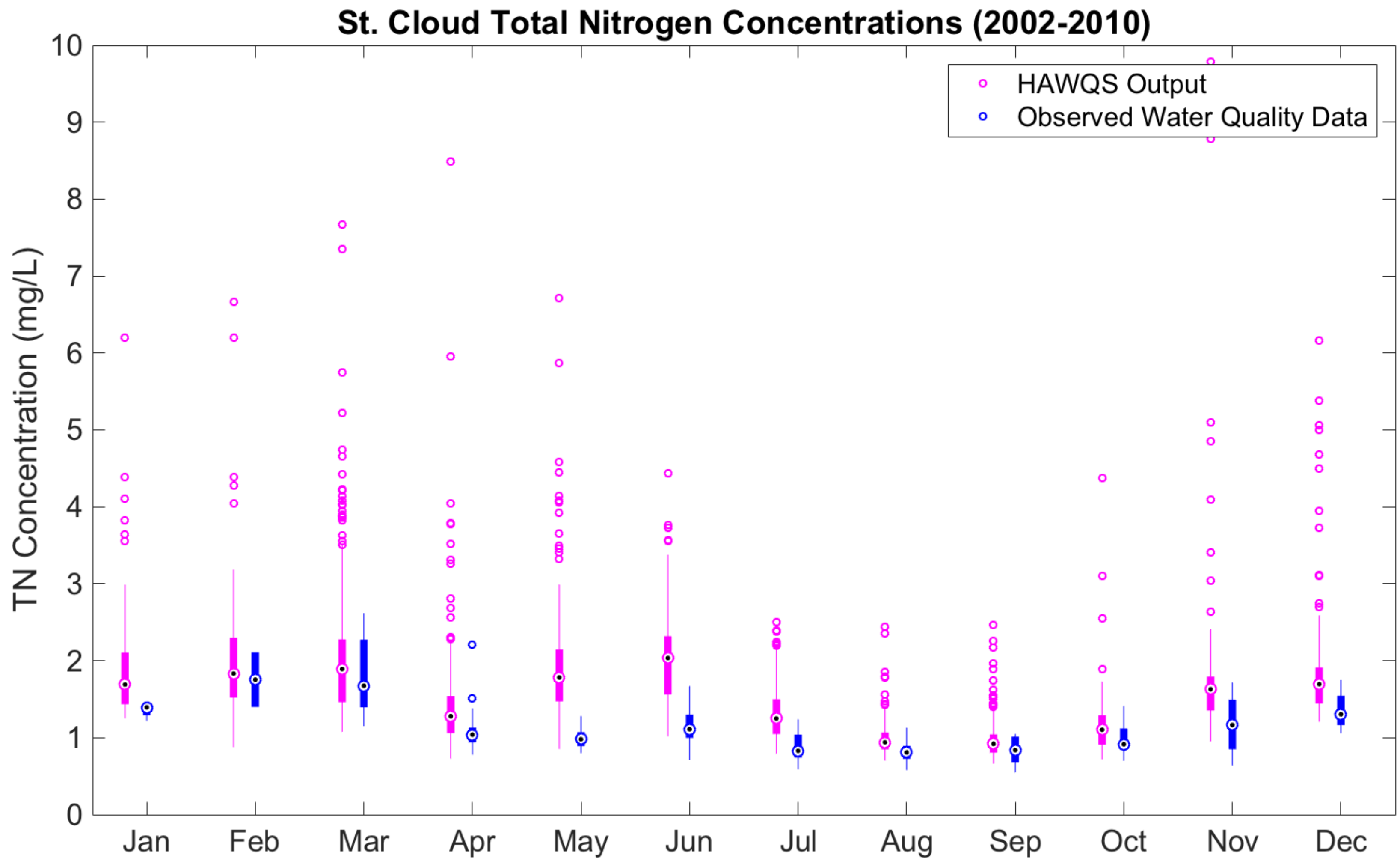

Figure 8. St. Cloud total nitrogen concentrations per month for the years 2002-2010. HAWQS data is in pink, observed water quality from site S005-782 is in blue. The box indicates endpoints of the interquartile range (IQR, 25th – 75th percentile), with white circle with black dot at sample median. Whiskers extend to final observation within 1.5 × IQR. Outliers are shown as open circles.

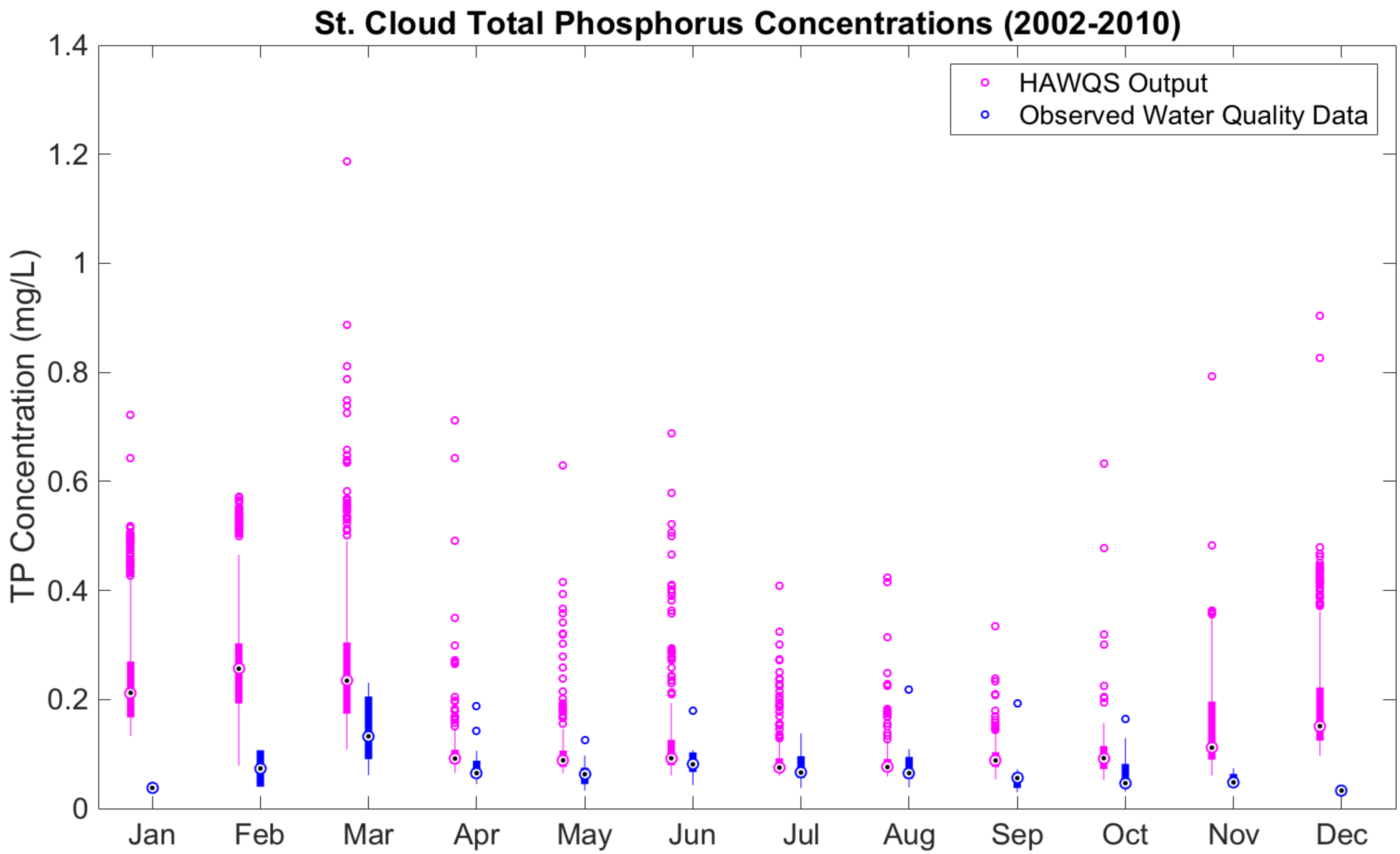

Figure 9. St. Cloud total phosphorus concentrations per month for the years 2002-2010. HAWQS data is in pink, observed water quality from site S005-782 is in blue. The box indicates endpoints of the interquartile range (IQR, 25th – 75th percentile), with white circle with black dot at sample median. Whiskers extend to final observation within 1.5 × IQR. Outliers are shown as open circles.

Model Results

Basin-Level Model Results

The HAWQS model was simulated for the baseline scenario with default land use and land cover and also for the moderate and aggressive agricultural expansion scenarios with their respective land use and land cover layers for the year 2027. Conservation interventions identified by TNC would avert the aggressive agricultural expansion scenario and would result in the moderate agricultural expansion in the study area. Outputs like streamflow, sediment concentration, total nitrogen (TN), and total phosphorus (TP) concentrations were compared across scenarios. Additionally, sediment loads and TP loads were also compared between scenarios. For the HAWQS model simulations, the water quantity and quality outputs are given at the HUC 10 level. As a result, there exists a discrepancy between the model output locations the source water intake locations of St. Cloud and Minneapolis water utilities (see **Figure 2**). The water quantity (streamflow) and pollutant load outputs from the HAWQS model correspond to points along the Mississippi River located a couple of miles upstream of the water intake locations.

The sediment, TN, and TP concentration values from the HAWQS model are average monthly values for the entire river reach in the HUC 10s containing the St. Cloud and Minneapolis drinking water utilities. Despite this discrepancy, it is assumed that the changes in the water quantity and quality outputs from HAWQS will be representative of changes in water quality parameters at the water intake locations. While the HAWQS model was not calibrated for the locations used in this study, it has been calibrated at a couple of locations in the entire study area. Hence, the absolute values of water quantity and quality outputs from the model may not be in close agreement with the actual observed values. Nevertheless, the percentage changes between scenarios will be similar for non-calibrated and calibrated models.

Figure 10A shows the average (2002-2010) stream flow near St. Cloud decreasing from around 303.4 m^3/sec for the baseline scenario to 297.6 m^3/sec for the moderate agricultural expansion scenario and 297.2 m^3/sec for the aggressive agricultural expansion scenario. TNC's interventions would result in a 0.15% increase in streamflow (between the moderate and aggressive agricultural expansion scenarios). Similarly, the average streamflow near Minneapolis decreased from 353.37 m^3/sec for the baseline scenario to 347.3 m^3/sec for the moderate agricultural expansion and 346.7 m^3/sec for the aggressive agricultural expansion scenario. Similar to the results at St. Cloud, TNC's interventions would increase the flow by 0.15%.

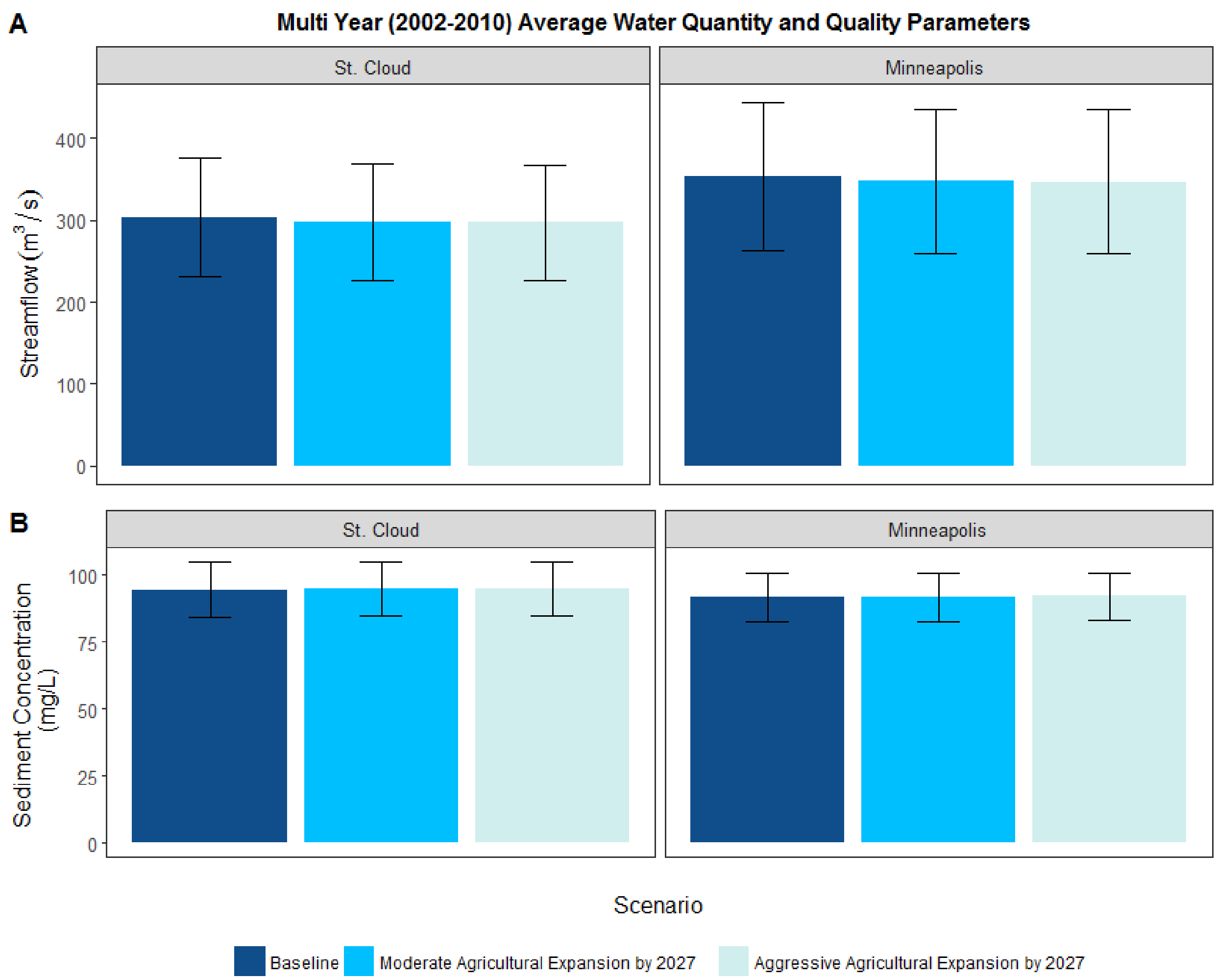

Figure 10. Multi-year average streamflow and sediment concentration values at locations near St. Cloud and Minneapolis drinking water utility intakes. The outputs for the baseline, moderate, and aggressive agricultural expansion scenarios are shown. The error bars represent the standard deviation of the outputs for the years 2002-2010.

Figure 10B shows the average (2002-2010) sediment concentration near St. Cloud and Minneapolis. The sediment concentrations increased from about 94.56 mg/L for the baseline scenario to around 94.67 mg/L for the moderate agricultural expansion scenario. The sediment concentration was around 94.79 mg/L for the aggressive agricultural expansion scenario. TNC's conservation interventions would result in a 0.13% decrease in sediment concentration near St. Cloud. Similarly for Minneapolis, the sediment concentration is around 91 mg/L, with TNC's interventions reducing the concentration from 91.91 mg/L in the aggressive agricultural expansion scenario to 91.75 mg/L in the moderate expansion scenario, a 0.18% decrease. For both Minneapolis and St. Cloud, the modeled changes in sediment concentrations are minimal, and changes fall within the margins of error.

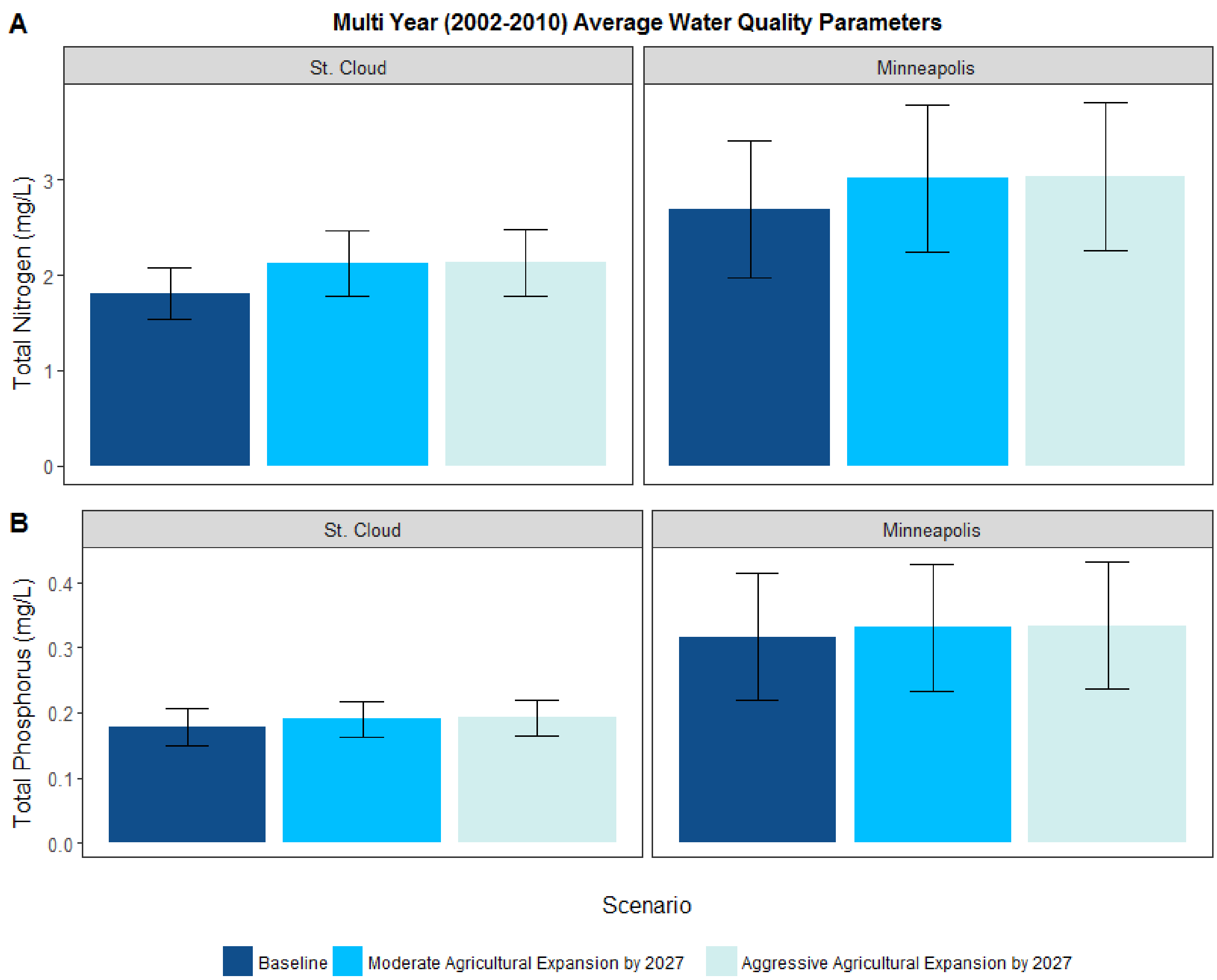

Figure 11. Multi-year average TN and TP concentration values for the HUC 10 level Mississippi River reaches containing St. Cloud and Minneapolis drinking water utilities. The outputs for the baseline, moderate, and aggressive agricultural expansion scenarios are shown. The error bars represent the standard deviation of the outputs for the years 2002-2010.

In addition to sediment concentrations, TN and TP concentrations were also compared between scenarios. One of the key questions the study sought to answer was whether future agricultural expansion would increase the nitrate concentrations to a level of concern for drinking water providers (i.e., the Maximum Contaminant Level (MCL) of 10 mg/L (as nitrogen)) at the Minneapolis and St. Cloud utility water intake locations. Since, HAWQS gives outputs of nitrate loadings instead of concentrations, TN was used as a proxy for nitrate concentrations. TN is the summation of nitrogen present in ammonium, nitrate, nitrite, and organic nitrogen. Generally, modeled TN outputs were comprised of 80-90 percent nitrate. TP values were also considered because of the affinity of phosphorus for natural clay particles, which can affect turbidity and hence drinking water treatment costs (Heberling *et al.*, 2015). TP is the summation of phosphorus present in phosphate and organic phosphorus.

Figure 11A shows the TN concentration at St. Cloud increasing from 1.8 mg/L for the baseline scenario to 2.12 mg/L for the moderate agricultural expansion scenario and 2.13 mg/L for the aggressive agricultural expansion. Similarly for Minneapolis, the TN concentrations increased from 2.7 mg/L for the baseline scenario to 3.01 mg/L for the moderate agricultural expansion scenario and 3.03 mg/L for the aggressive agricultural expansion. Nitrate concentrations, which are a fraction of TN concentrations, would be lower than the values shown above for both the St. Cloud and Minneapolis locations. TNC's conservation interventions would decrease TN concentrations by roughly 0.6% at both the St. Cloud and Minneapolis locations.

Figure 11B shows TP concentration of 0.17 mg/L for the baseline scenario at St. Cloud. This increased to 0.190 mg/L in the moderate agricultural expansion scenario and 0.192 mg/L in the aggressive agricultural expansion scenario. Similarly, the TP concentrations at Minneapolis increased from 0.317 mg/L in the baseline scenario to 0.331 mg/L in the moderate agricultural expansion scenario and 0.334 mg/L in the aggressive agricultural

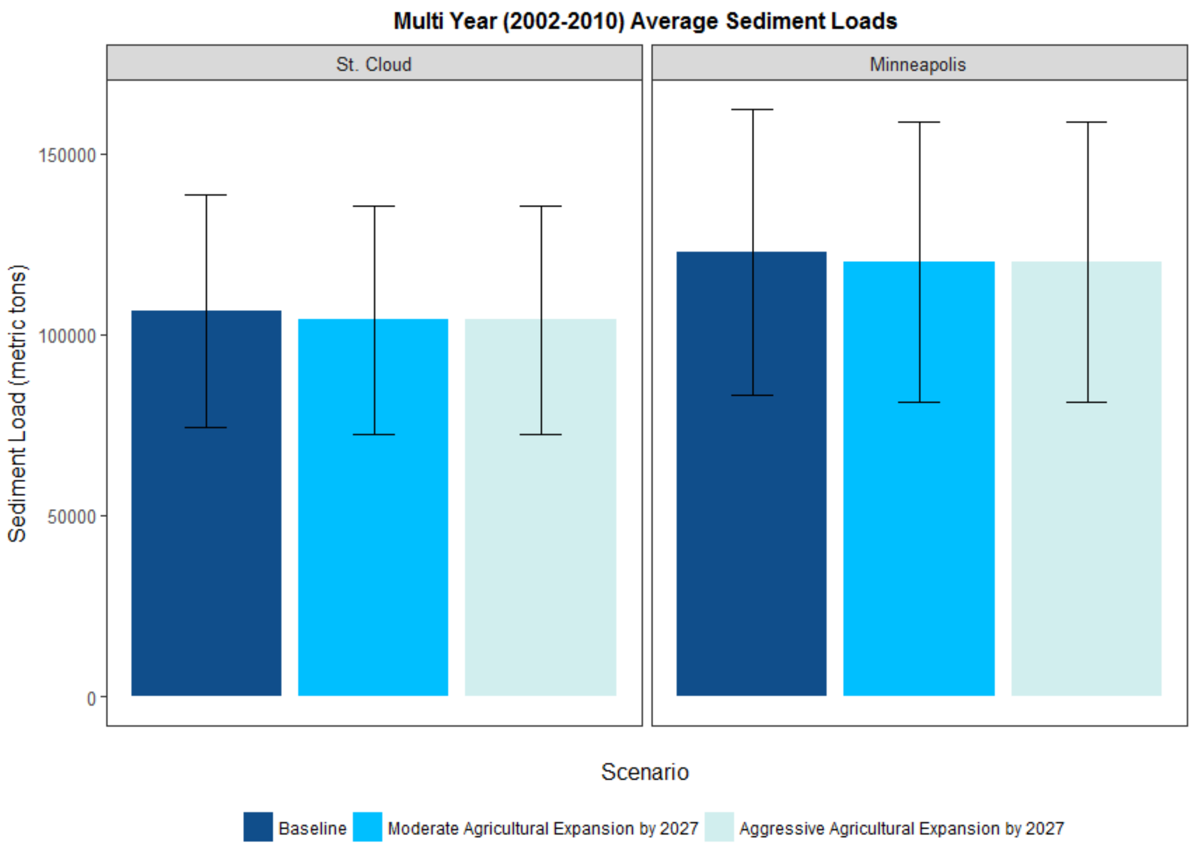

Figure 12. Multi-year average sediment loads at locations near St. Cloud and Minneapolis drinking water utilities. The outputs for the baseline, moderate and aggressive agricultural expansion scenarios are shown. The error bars represent the standard deviation of the outputs for the years 2002-2010.

expansion scenario. At both St. Cloud and Minneapolis, the changes in TP concentrations from TNC's interventions are minimal.

In addition to nutrient and sediment concentrations, sediment and phosphorus loads were also compared across scenarios. TN loads were not compared, since nitrate concentrations (as a fraction of TN concentrations) were well below the MCL and would not affect the treatment costs. **Figure 12** shows sediment loads of 106,492 metric tons at St. Cloud for the baseline scenario. The sediment loads decreased to 104,031 metric tons for the moderate agricultural expansion scenario and then increased to 104,051 metric tons for the aggressive agricultural expansion scenario. TNC's interventions would decrease the sediment loads by 0.02% near St. Cloud. The sediment load at Minneapolis decreased from 122,769 metric tons for the baseline scenario to 120,062 metric tons for the moderate agricultural expansion scenario and increased to 120,171 metric tons for the aggressive agricultural expansion scenario. TNC's interventions would decrease the sediment loads by 0.09% near Minneapolis. Since the model is not calibrated, the absolute values of these loads may not be representative of the actual loads. Nevertheless, the percentage changes between scenarios would not change between non-calibrated and calibrated models.

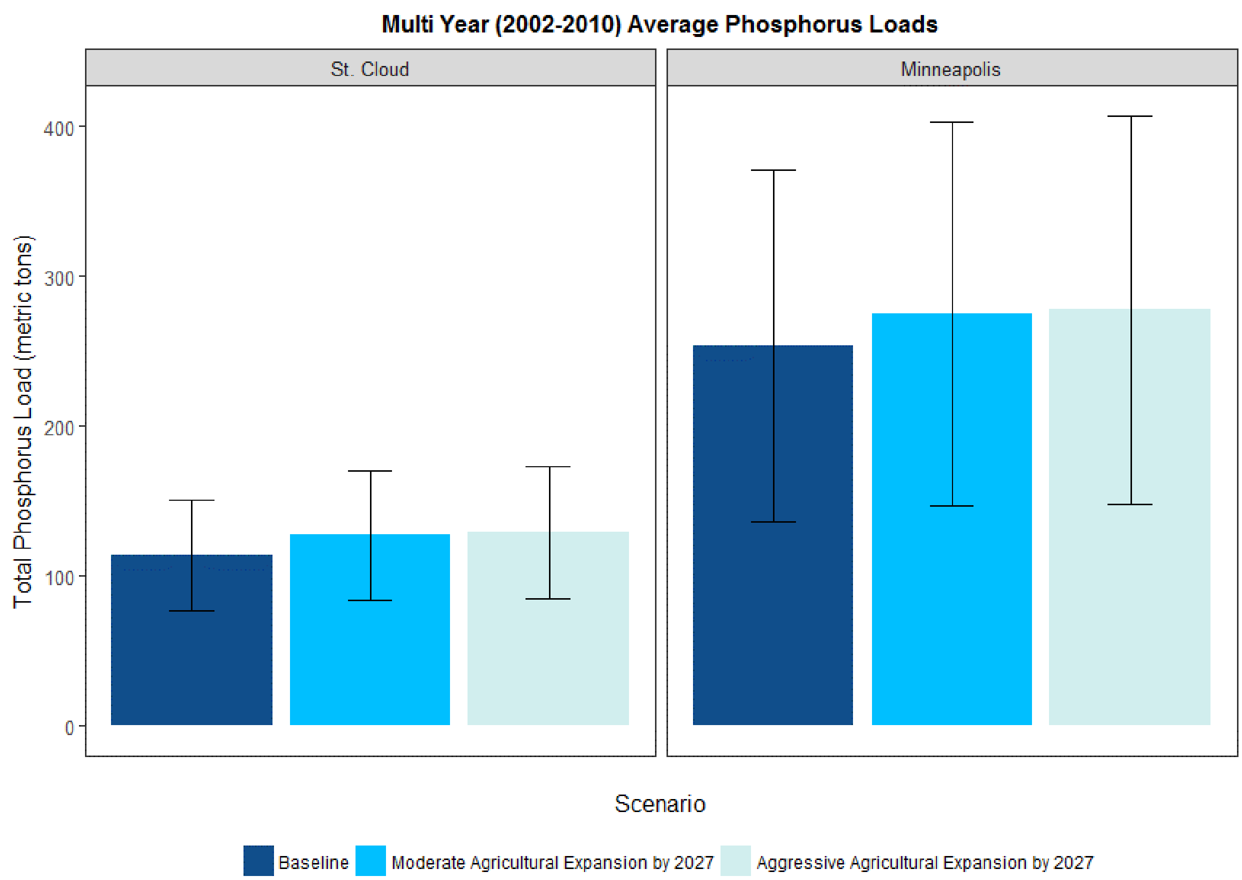

Figure 13. Multi-year average Total phosphorus loads at locations near St. Cloud and Minneapolis drinking water utilities. The outputs for the baseline, moderate and aggressive agricultural expansion scenarios are shown. The error bars represent the standard deviation of the outputs for the years 2002-2010.

Figure 13 shows a multi-year average TP load of around 113.93 metric tons for the baseline scenario at St. Cloud. The TP loads increase to around 127.15 metric tons for the moderate agricultural expansion scenario and to 128.86 metric tons for the aggressive agricultural expansion scenario. The TP loads at Minneapolis increased from 253.44 metric tons for the baseline scenario to about 275 metric tons for the moderate agricultural expansion scenario and to 277.6 metric tons for the aggressive agricultural expansion scenario. TNC's conservation interventions, therefore, would decrease the TP loads by 1.33% at St. Cloud and by 0.95% at Minneapolis.

Although all the above figures represent multi-year averages of water quantity and quality values over the 2002-2010 period, there is considerable inter-annual variation in these values. This is evident from the error bars that represent the standard deviation in all these figures. For instance, the average stream flow (from the model outputs) at St. Cloud for the default scenario was 303.5 m^3/sec, but the annual stream flow (from the model outputs) varied from 183.9 m^3/sec in 2006 to 424.7 m^3/sec in 2010. The streamflow at Minneapolis varied from 210.5 m^3/sec in 2006 to 498 m^3/sec in 2010. Similarly the TN concentration (from the model outputs) at Minneapolis for the default scenario varied from 1.84 mg/L in 2009 to 4.3 mg/L in 2003. These inter-annual variations are observed in all the water quality and quantity values shown above.

From **Figure 5**, it is clear that between 2006 and 2027 in the study area, forested land will be replaced by increasing cropland, urban land, and to some extent by rangeland expansion. This pattern is supportive of land use changes that have been observed in Minnesota in the recent past (Lark *et al.*, 2015). Past studies have shown that cropland expansion in the Midwest can result in increased surface runoff, because intensive agricultural practices can lead to more soil compaction, decreasing soil permeability resulting in more surface flow (DeJong-Hughes *et al*., 2001). At Minneapolis and St. Cloud locations, the streamflow decreases by about 1.8% from the baseline to the moderate scenario, and by about 0.15% from the moderate to the aggressive expansion scenario. This observed decrease in streamflow, despite the expansion of cropland upstream, could be due to the fact that rangeland is also expanding in the study area at the same rate as cropland, and increased rangeland along with increased hay and pastureland can decrease the streamflow in a watershed (Ahiablame *et al*., 2016). Urban land expansion in the study area (by 17.4% from the baseline) does not significantly augment the streamflow at Minneapolis or St. Cloud because urban land accounts for only about 7% of the total land use in the study area.

The increases in sediment concentrations at Minneapolis and St. Cloud, although of small magnitude (in the range of 0.01-0.18% increase between baseline, moderate and aggressive agricultural expansion scenarios) could be directly attributed to the decrease in forest land and increase in cropland and urban land between these scenarios. The increases in TN and TP concentrations at St. Cloud and at Minneapolis for the three land use scenarios can also be attributed to increase in cropland and the associated increase in fertilizer use in these scenarios. Finally, very minimal changes were observed between the moderate and aggressive agricultural expansion scenarios because the magnitude of cropland expansion between those two scenarios (approximately 175 km^2) is minor compared to the size of the basin (approximately 51,000 km^2).

Variability and Potential Drivers of Water Quality

With respect to flow, nutrient, and sediment pollution, the results of the HAWQS analysis show that variability over time exceeds any modeled changes in water quality between the land use change scenarios. This observation of higher inter-annual variability suggests that factors with greater temporal variation may be stronger drivers of water quality in the basin than the relatively static land use scenarios created for this study. One such factor is climate, with precipitation specifically playing a key role in the transport of pollutants throughout the basin. The relationship between precipitation and pollution concentrations is generally distinct and counters that of precipitation and pollutant loads, suggesting both high and low precipitation events may have unique implications for water quality in the basin.

The National Climatic Data Center, National Weather Service/National Oceanic and Atmospheric Administration (NCDC NWS/NOAA), Parameter-elevation Regressions on Independent Slopes Model (PRISM), and both PRISM bias-corrected and original NOAA Next Generation Weather Radar (NEXRAD) supply climate data to HAWQS (US EPA, 2017). Average annual precipitation in the basin ranges generally from 600-800 mm (23-31 inches) and shows considerable seasonality. Precipitation is lowest in January, then rises throughout the spring before peaking between June and September. **Table 2** shows monthly average precipitation and standard deviation for the basin from 2002-2010.

Table 2. Monthly mean precipitation (in mm) for the Upper Mississippi River Basin between 2002-2010, based on HAWQS input data. Values following the mean indicate one standard deviation. Source: HAWQS, 2017.

	January	February	March	April	May	June
Average Precipitation (mm) ($\bar{x} \pm \sigma$)	12.67 ± 11.92	18.10 ± 7.57	37.20 ± 22.35	59.67 ± 21.93	76.25 ± 27.44	111.71 ± 41.57
	July	**August**	**September**	**October**	**November**	**December**
Average Precipitation (mm) ($\bar{x} \pm \sigma$)	90.10 ± 38.90	85.21 ± 39.91	97.25 ± 41.68	85.24 ± 37.63	22.72 ± 16.45	28.99 ± 13.77

Flow near the City of Minneapolis source water intake responds to precipitation fluctuations relatively quickly. **Figure 14** shows both mean monthly flow (in m^3/sec) and precipitation (in mm) from 2002-2010. Like precipitation, flow shows considerable seasonality, reaching its lowest flows in the winter and peaking in the summer months.

Conversely, pollutant concentrations show a different relationship to precipitation in the basin. **Figure 15** and **Figure 16** show TN and TP concentrations near the City of Minneapolis' source water intake, respectively. For both pollutants, concentrations generally show an inverse relationship to precipitation, with peaks in concentration occurring at times of relatively low precipitation. However, closer analysis of peak pollutant concentrations indicates that this inverse relationship with precipitation is more complex than a simple dilution effect. For both pollutants, peak concentrations do not always correspond with minimum mean precipitation or flow values, but rather they commonly occur one to two months following minimum precipitation and flow in the basin. This lag may indicate a flushing effect, where pollutant concentrations increase following an initial uptick in precipitation or snowmelt. Alternatively, other seasonal or otherwise temporally variable factors (e.g., land management practices) could be driving pollutant concentration spikes, largely independent of precipitation trends.

While TN and TP concentrations show a quasi-inverse relationship with precipitation, sediment concentrations near the City of Minneapolis' source water intake correlate more directly with basin-wide precipitation. **Figure 17** shows average monthly sediment concentrations and precipitation from 2002-2010. Unlike TN and TP concentrations, trends in sediment concentrations generally follow trends in precipitation, with peak sediment concentrations usually occurring between April and July.

Fluctuations in pollutant concentrations may not exhibit the same trend as pollutant loading in the basin. While TN and TP concentrations generally declined with increasing precipitation, pollutant loading for TN, TP, and sediment were all positively correlated with precipitation. Regression analyses were conducted to assess the relationship between monthly precipitation and average monthly flow, TN load, TP load, and sediment load near the City of Minneapolis' source water intake. In all cases, precipitation significantly predicted flow or loading and explained a significant proportion of the variance. Unsurprisingly, precipitation most strongly predicted flow ($R^2 = 0.63$, $p < 0.001$), followed by sediment loading ($R^2 = 0.60$, $p < 0.001$).

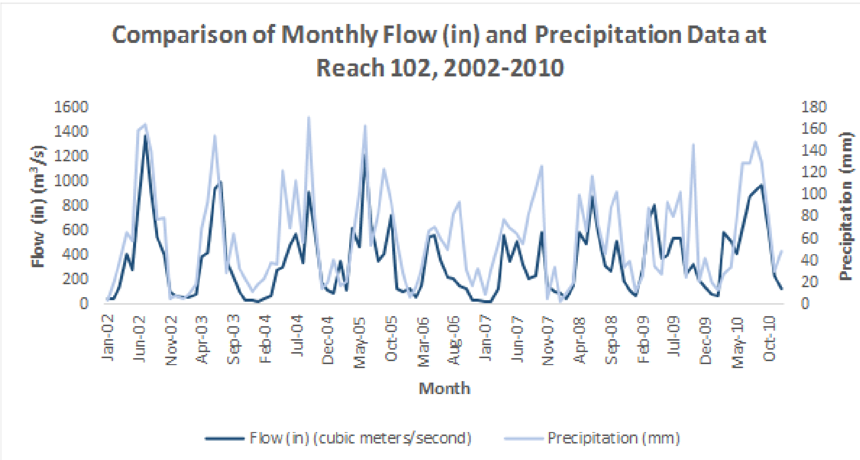

Figure 14. Average monthly flow (left y-axis)(in m3/sec) and monthly precipitation (right y-axis)(in mm) from 2002-2010. Reach 102 includes the City of Minneapolis source water intake. Source: HAWQS.

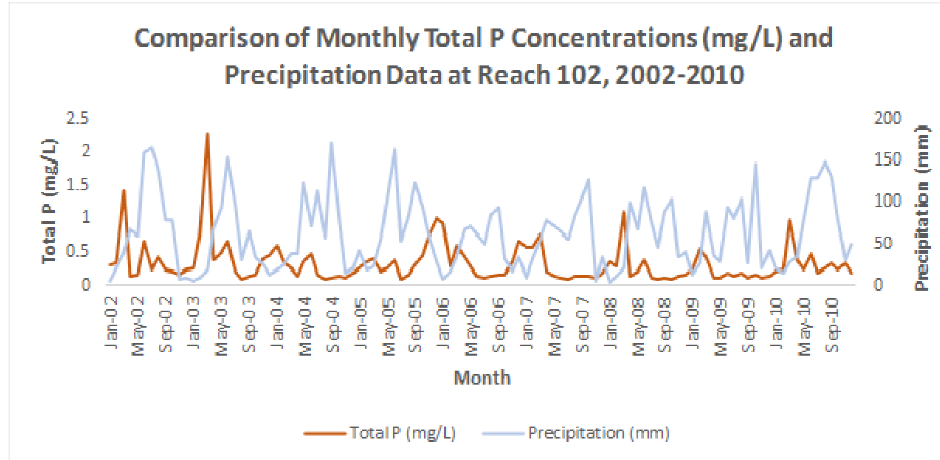

Figure 16. Average monthly TP concentrations (left y-axis)(in mg/L) and monthly precipitation (right y-axis)(in mm) from 2002-2010. Reach 102 includes the City of Minneapolis source water intake. Source: HAWQS.

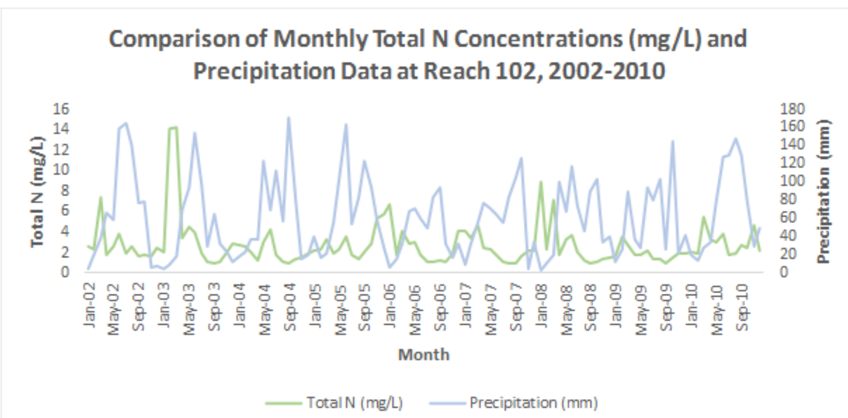

Figure 15. Average monthly TN concentrations (left y-axis)(in mg/L) and monthly precipitation (right y-axis)(in mm) from 2002-2010. Reach 102 includes the City of Minneapolis source water intake. Source: HAWQS.

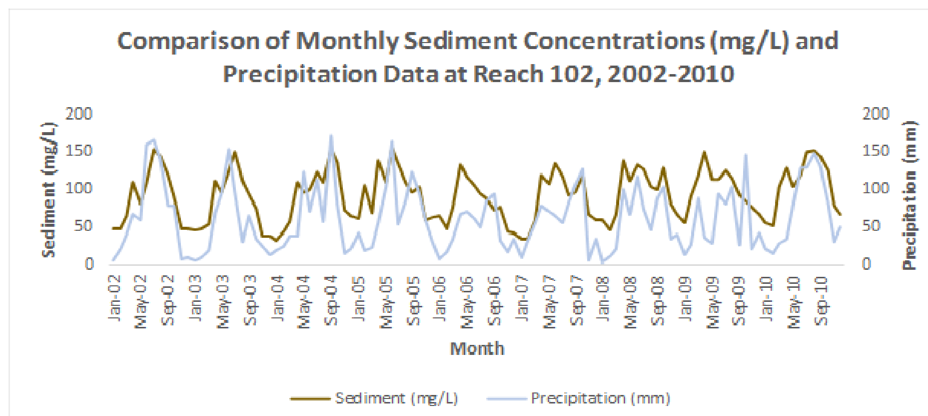

Figure 17. Average monthly sediment concentrations (left y-axis)(in mg/L) and monthly precipitation (right y-axis)(in mm) from 2002-2010. Reach 102 includes the City of Minneapolis' source water intake. Source: HAWQS.

Sub-Basin Analysis of Model Results

The purpose of this section is to analyze the impacts of land use change from the moderate and aggressive scenarios on water quality throughout the study area, and determine both the spatial distribution and magnitude of these changes. Although results over the entire study area show modest increases in TN, TP, and sediment, understanding the spatial variability of these changes is important for a number of management and planning purposes. Outputs from HAWQS, including yields for TN, TP, and sediment were included as part of this analysis, and results are represented as the relative changes to yield in tons per acre occurring within each HUC 10 sub-basin, averaged over the ten year model simulation.

Total Nitrogen

Figure 18 shows the percent change in TN contributions to the study area of the ten sub-basins with the highest nitrogen yield. Of these ten sub-basins for both the moderate and aggressive scenarios, the average change is a 3% increase in TN loading, with a maximum of approximately 24% and a minimum of -1%. Among the sub-basins, only two of the top ten TN contributors experience an increase of TN yield of over 5%, while the remaining six experience an increase 2% or less.

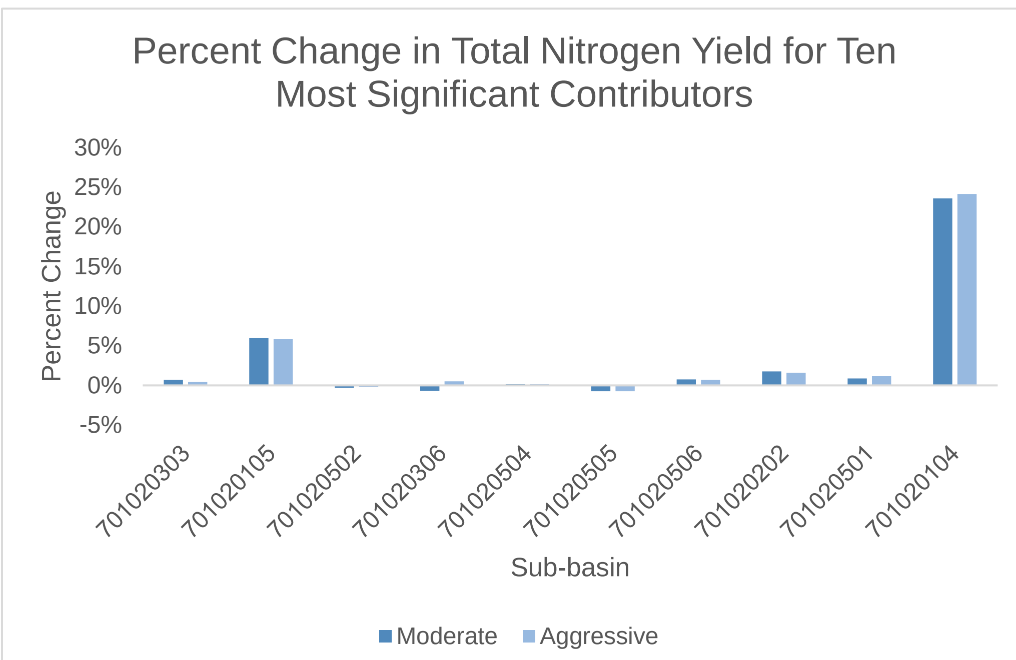

Figure 18. Changes in TN yield by HUC 10 sub-basin for the ten most significant contributors of nitrogen in the Upper Mississippi River Basin. The moderate and aggressive agricultural expansion scenarios (shown in dark blue and light blue) are displayed relative to the baseline scenario. Changes in yield on the order of 0 - 1% are the most common and indicative of relatively low changes to the overall loading of nitrogen in the watershed resulting from land use change in the sub-basins currently contributing the most TN.

Figure 20 shows two side-by-side maps related to TN within the study area. The left side shows the ten-year average for magnitude of nitrogen yield by HUC 10 sub-basin. Sub-basins which contribute the largest proportion of total nitrogen in the study area, displayed in darker shades of color, are located primarily in the southwest and western portion of the basin. The location of the largest contributors is consistent with the predominantly agricultural land uses in the corresponding areas. The map on the right side shows the percent change in yield of TN from the baseline scenario to the moderate agricultural expansion scenario, and is intended to identify the sub-basins where nitrogen loading is expected to increase or decrease. For TN, larger yields by sub-basin are expected to occur in the central portion of the basin, expanding northwest from existing hotspots of nitrogen loading. In areas where increases are expected to occur, changes in nitrogen yield could range from 50 – 150%, which are significant increases from the baseline scenario. Aside from these hotspots, increases in yield are less drastic, ranging from 0-25% or no increase.

Total Phosphorus

Figure 19 shows the percent change in TP yield by sub-basin of the ten most significant phosphorous contributors. Of the ten sub-basins in both the moderate and aggressive scenarios, there is an average increase of 2%, a minimum of -1%, and a maximum increase

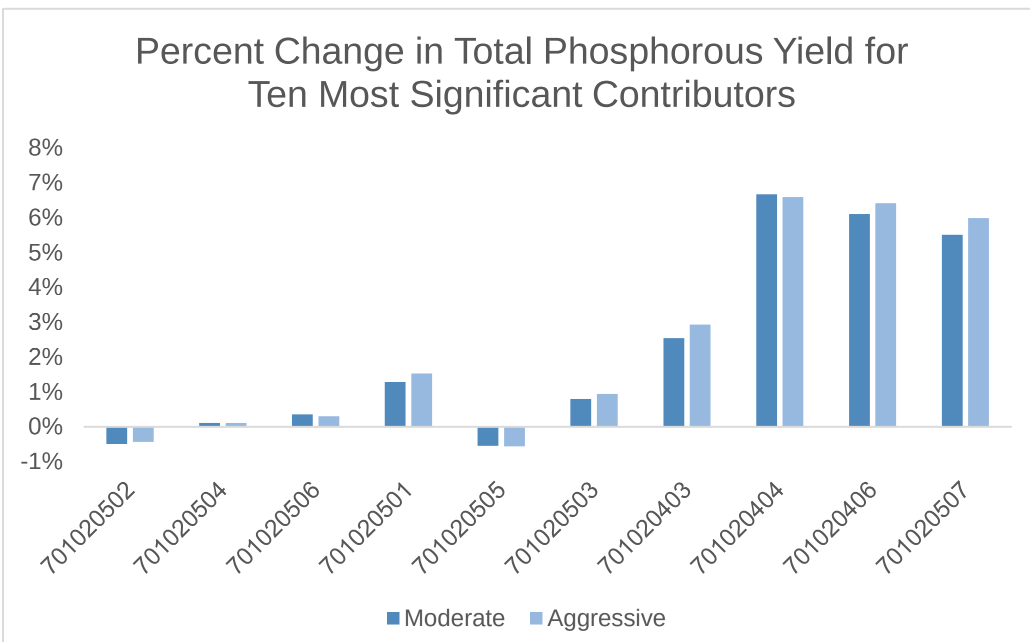

Figure 19. Changes in TP yield by HUC 10 sub-basin for ten most significant contributors of phosphorous in the Upper Mississippi River Basin. The moderate and aggressive agricultural expansion scenarios (shown in dark blue and light blue) are compared to the baseline scenario. Changes in yield on the order of 0 - 2% are the most common and indicative of relatively low changes to the overall loading of phosphorous in the watershed resulting from land use change in the sub-basins currently contributing the most TP.

of 7%. Differences between the moderate and aggressive land use scenarios are minimal, often on the order of 0% to 2%.

Figure 21 shows two side-by-side maps related to TP within the study area. The left side shows the magnitude of total phosphorous yield in the study area by HUC 10 sub-basin. Sub-basins which contribute the largest yields of TP in the study area, displayed in darker shades of color, are located primarily in the southwest portion of the basin. The location of the largest contributors is consistent with the predominantly agricultural land uses in the corresponding areas. The right side shows the percent change in yield of TP from the baseline scenario to the moderate scenario, and is intended to identify the sub-basins where phosphorous loading is expected to increase or decrease relative to the entire basin. For phosphorous, sub-basins with the largest increases of yield are expected to occur in the central portion of the basin as land use changes, expanding north and northeast from existing hotspots of phosphorous loading. Sub-basins in the northeast corner of the study area also experience significant increases in total phosphorous yield. A majority of sub-basins in the study area experience increases in phosphorous yield of 0–25%; however, increases may range as high as 40-70% for some sub-basins.

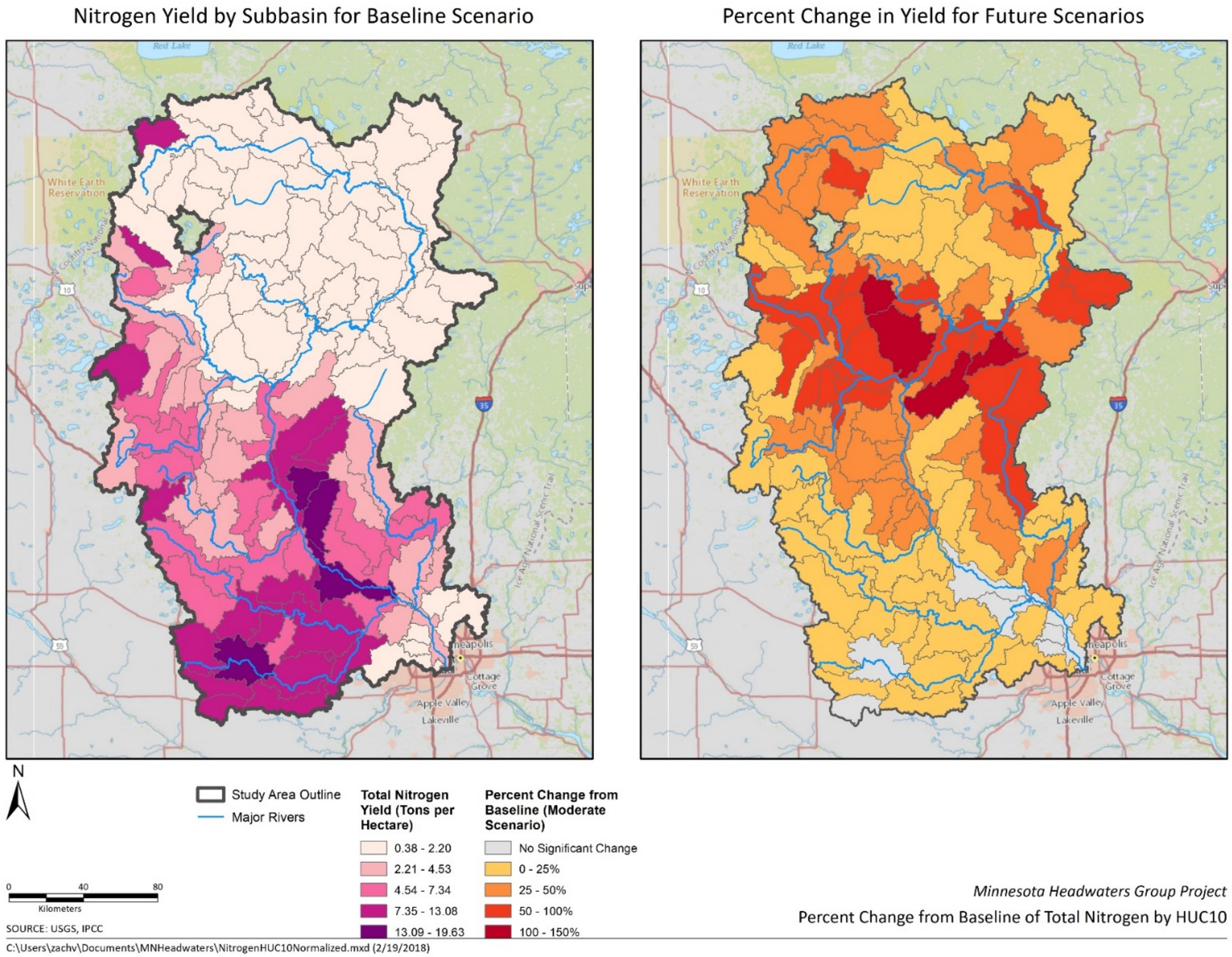

Figure 20. Left map shows nitrogen yield (tons/hectare) by sub-basin under baseline conditions. Highest nitrogen yields occur in largely agricultural sub-basins in the southern and western portions of the study area. Right map shows each sub-basin's percent change in nitrogen yield. Sub-basins with the greatest percentage increase in nitrogen yield are located in the central and eastern portions of the study area.

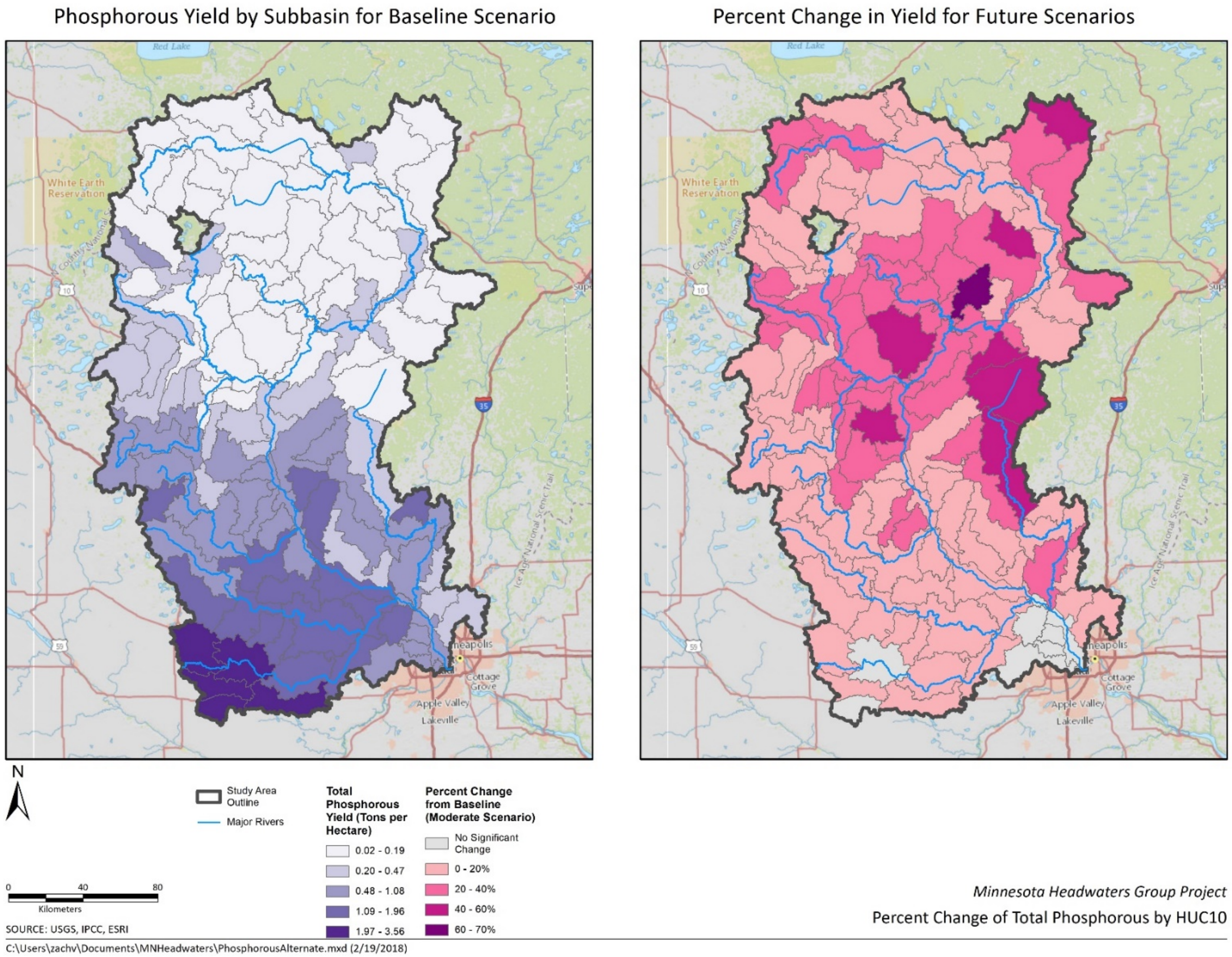

Figure 21. Left map shows total phosphorus yield (tons/hectare) by sub-basin under baseline conditions. Highest phosphorus yields occur in largely agricultural sub-basins in the southern and western portions of the study area. Right map shows each sub-basin's percent change in phosphorus yield. Sub-basins with the greatest percentage increase in phosphorus yield are located in the central and northeastern eastern portions of the study area.

Sediment

Figure 22 shows the percent change in sediment contributions to the study area of the ten most significant sediment contributors. In comparison to the TN and TP data, there are similar differences between the moderate and aggressive land use scenarios in terms of total sediment yield by sub-basin. Of the ten largest contributors for the moderate scenario, there is an average increase of 1%, a minimum of -10%, and a maximum increase of 5% in contribution to total sediment loading. For the ten largest contributors in the aggressive scenario, there is an average increase of 1%, a minimum of -9%, and a maximum of 7%. Differences between the moderate and aggressive land use scenarios are minimal and range between 0 to 2%.

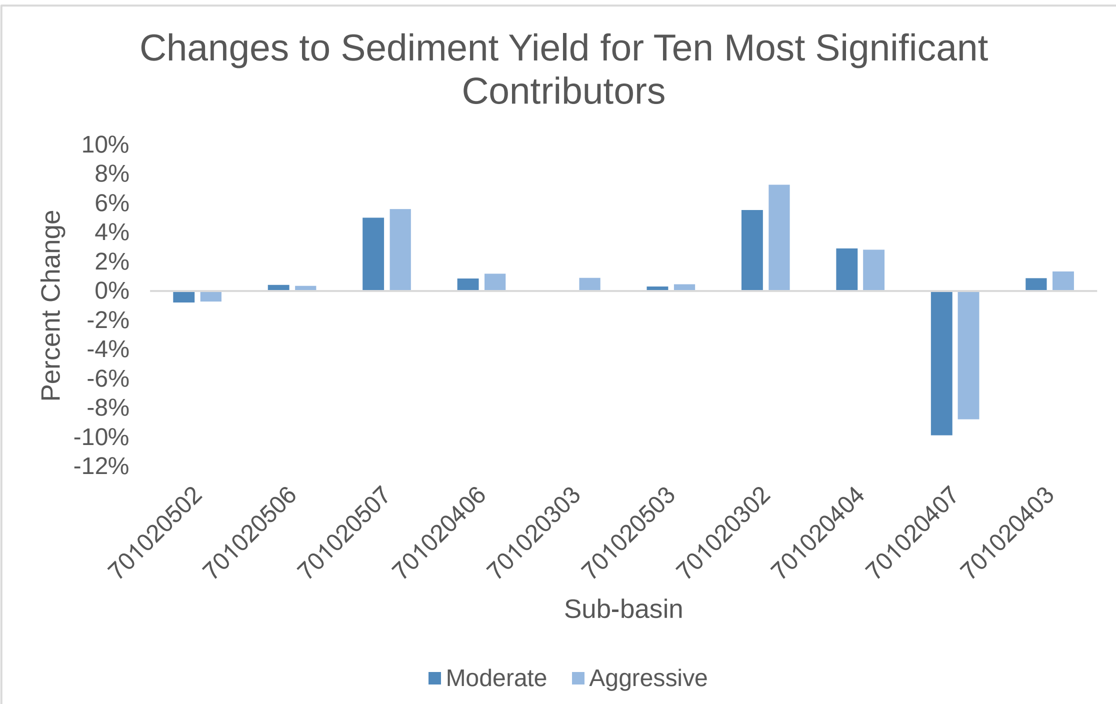

Figure 22. Changes in sediment yield by HUC 10 sub-basin for ten most significant contributors of sediment in the Upper Mississippi River Basin. The moderate and aggressive agricultural expansion scenarios (shown in dark blue and light blue) are compared to the baseline scenario. Changes in yield on the order of 0 - 2% are the most common and indicative of relatively low changes to the overall loading of sediment in the watershed resulting from land use change in sub-basins currently contributing the most to sediment loading.

Figure 23 shows two side-by-side maps related to sediment within the study area. The left side shows the magnitude of sediment yield in the study area by HUC 10 sub-basin. Sub-basins which contribute the largest yields of sediment in the study area, displayed in darker shades of color, are located primarily in the southwest portion of the basin.

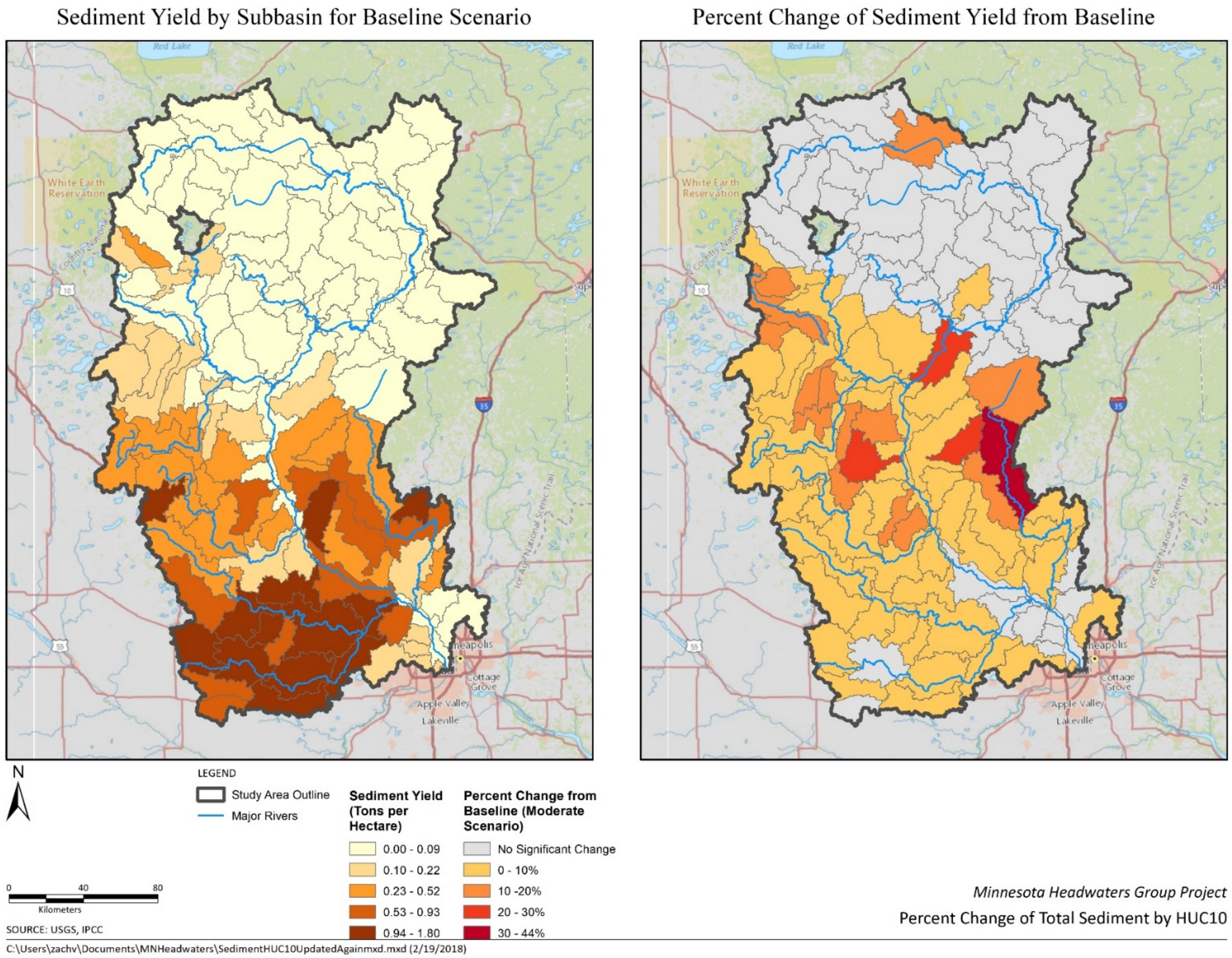

Figure 23. Left map shows sediment yield (tons/hectare) by sub-basin under baseline conditions. Highest sediment yields occur in largely agricultural sub-basins in the southern and western portions of the study area. Right map shows each sub-basin's percent change in sediment yield. Sub-basins with the greatest percentage increase in sediment yield are located in the central and eastern eastern portions of the study area.

The location of the largest contributors is consistent with the predominantly agricultural land uses in the corresponding areas. The right side shows the percent change in yield of sediment from the baseline scenario to the moderate scenario, and is intended to identify the sub-basins where sediment loading is expected to increase or decrease relative to the entire basin. For sediment, sub-basins with the largest increases of yield is expected to occur in the central portion of the basin as land use changes, expanding north and northeast from existing hotspots of sediment loading. A majority of sub-basins in the study area experience increases in phosphorous yield of 0–10%; however, increases may range as high as 30-44% for some sub-basins.

Discussion of Sub-basin Results

Results for the analysis of water quality parameters by sub-basin were, for the most part, consistent with expectations based on predictions of future land uses and current understanding of its link with water quality. A major premise of this study is that climate change and associated changes in local climate variables such as temperature and precipitation could drive agricultural activities further northward in the Upper Mississippi River Basin, amongst other concurrent land use changes. This premise is captured in the range of future land use scenarios generated for input into the HAWQS model. The moderate and aggressive land use change scenarios included an additional 350 km^2 and 525 km^2 acres of cropland, respectively, as well as an additional 650 km^2 and 655 km^2 of urban areas. Therefore, it was expected that there would be an overall increase of the selected water quality parameters as compared to the baseline, and that the aggressive scenario would experience larger increases.

As described in the Background section, the scientific literature describes a clear link between various land uses, such as agriculture or urban areas, and water quality parameters like TN, TP, and sediment. As such, it was expected that changes in these parameters would be concurrent with changes to land use from the moderate and aggressive scenarios and scale with the total area of such changes. Analysis of linear regression for changes to cropland, urban land, wetlands, and forested land and their impacts on the three water quality parameters showed that both cropland and forested land were significant predictor variables. Increased cropland was positively associated with increases in TN, TP, and sediment loadings, while forested land was negatively associated with these variables.

Since agriculture dominates in the southwest portion of the basin and the majority of arable land has already been converted to agriculture, little to no increase in loading of water quality parameters was expected in these regions. Instead, conversion of non-agricultural lands in the central and northern parts of the basin into agriculture were expected to be associated with the most significant changes to the chosen parameters. Results of HAWQS outputs were consistent with these expectations, in particular with nitrogen and phosphorous. In the central portion of the basin where agricultural uses are not currently maximized, there are significant increases in TN and TP yields by sub-basin.

The drivers of these changes are tied to specific land use transitions, most notably decreases in forested land and wetlands, or increases in urban and agricultural lands.

Results from the HAWQS output show relatively small differences between the percent increase in TN and TP loadings by sub-basin for both the moderate and aggressive scenarios. This case is especially true for sub-basins which contribute the largest proportion of TN and TP to begin with. Since land uses in these sub-basins are already saturated with primarily agricultural land, small increases in agricultural land make relatively small increases in total loading. However, for sub-basins that contribute smaller relative proportions of total loading, the same increase in agricultural or urban land can lead to significantly larger increase in TN and TP contributions. Although these changes may small relative to basin-wide loading, 40-150% increases in pollutant loading within individual sub-basins could occur, which at the very least could result in localized water quality impacts.

Although an additional 42,000 acres of cropland expansion are captured within the aggressive scenario, the transition is a relatively insignificant increase relative to the area of the entire study area, particularly when only accounting for water quality parameters such as TN, TP, and sediment. This aspect is compounded by the fact that the additional 42,000 acres of agriculture are distributed relatively evenly throughout the entire basin, resulting in a dilution of its impact on water quality on the sub-basin level, as opposed to if all of the conversions occurred within a more condensed area.

From a management and conservation perspective, these results show that actions taken to affect water quality on a basin-wide scale must conducted relative to the size of the basin itself. However, if mitigation of impacts to local water quality parameters are considered a worthwhile investment for TNC, then results at the sub-basin scale show where conservation practices could have the largest effect. Targeting sub-basins where the largest increases in N, P, and sediment yields are expected to occur could buffer the effect of land use change and reduce the magnitude of expected increases to associated water quality parameters.

Additional Considerations & Recommendations

Economic Case for Drinking Water Utility Investment

TNC's land purchases and conservation easements in the study area would decrease sediment, TN and TP concentrations near the St. Cloud and Minneapolis water intake locations. Since, the reduction in sediment concentrations at both the locations is very minimal (around 0.1% – 0.2%), cost savings from reduced sediment concentrations were not estimated. Additionally, the cost savings from TN reductions (around 0.4% - 0.8%) were not estimated, because even under the aggressive agricultural expansion scenario, the TN concentrations did not approach the maximum contaminant level of 10 mg/L for nitrates in drinking water.

Therefore, cost savings from reductions in TP concentrations (around 0.6%-1.2%) alone were estimated. Based on the treatment cost and pollutant data provided by the drinking water utility at Minneapolis, a 1% reduction in TP concentrations near the Minneapolis water intake location reduces the treatment cost by around $340 per year. For comparison, Heberling *et al.* (2015) report that, for a drinking water treatment facility in Ohio, a 1% decrease in TP load near the intake location of the plant decreased the treatment costs by $168 on average. Heberling and Price (2017) report that from a literature review of drinking water treatment studies, a 1% decrease in turbidity results in savings within the range $121-$13,060. The estimated yearly savings at Minneapolis are in a similar range of values reported in the literature.

Assuming, conservatively, that these benefits, accrue in perpetuity once TNC's interventions are fully implemented, the net present value (NPV) of the benefits is around $10,200 - $18,900 (in 2015 USD)[1]. The benefits from TP reductions at St. Cloud water utility were not calculated because there was no statistically significant relationship between chemical costs and TP concentrations at St. Cloud (see Appendix 1). These benefits are minimal compared to the proposed scale of investments (of the order of millions of dollars) in source water protection in the study area by TNC.

Despite these relatively low direct benefits to drinking water utilities, there are several co-benefits of source water protection other than drinking water treatment cost savings that may add value to source water protection efforts. For example, there can be increased carbon sequestration from lands that have been purchased and conserved as part of the Headwaters Fund. The amount of additional carbon dioxide (CO_2) stored (or the CO_2 prevented from escaping into the atmosphere by preventing agricultural conversion) can be quantified, and these CO_2 emissions (in tons of CO_2) could be traded in carbon offset

[1] The NPV was calculated using a 2% discount rate. Although there was no specific discount rate for watershed protection projects in Minnesota, we used the discount rate close to the value recommended by the Bureau of Reclamation. A discount rate of 2.875% is recommended by the Bureau of Reclamation for federal water infrastructure projects.

markets. This could increase the total yearly benefits from source water protection. Source water protection can also reduce the amount of sediments entering the rivers. There could be benefits from reduced sedimentation and siltation near dams in the area, which could be quantified by estimating the cost savings from reduced siltation. There can also be other benefits from biodiversity conservation and improved water quality for fish habitat. All of these have the potential to increase the monetary benefits from TNC's source water protection initiatives.

In addition to drinking water utilities dependent on surface water, other groups may have a vested interest in source water protection programs. For instance, groundwater utilities can be potential partners in the Headwaters Fund because they might experience decreases in groundwater pollutant levels by investing in land conservation and conservation easements. Industrial scale water consumers, like Coca-Cola or other local institutions like hospitals, who benefit from improved water quality could also be potential stakeholders in source water protection. Other beneficiaries could be homeowners, recreationists, or environmental groups in specific sub-basins in the study area that are willing to pay a premium on improved water quality in rivers and lakes in those specific sub-basins. Including such diverse groups in source water protection initiatives can also increase the economic case for the program overall.

Recommendations:

- Improve the economic case for source water protection by quantifying co-benefits of TNC's interventions in the basin.
- Identify other groups that have incentive to invest in source water protection, such as groundwater utilities or large water users within a utility's service area.

Implications for Climate Resiliency

Land use change, particularly change to agricultural and urban land use, is a well-documented driver of water quality degradation. However, the land use changes modeled in this study do not appear to occur at a great enough magnitude to be significant drivers of source water quality for drinking water utilities in the near future. Water quality parameters in the study area did exhibit high inter-annual variability, though, suggesting that year-to-year climatic drivers may influence water quality to a greater degree. In this regard, the basin and the drinking water utilities within it may be more susceptible to long-term climate change impacts.

Climate change functions as an unknown, with implications for water supply reliability. In the Upper Mississippi River Basin, climate change impacts to water quality may be two-tiered. The first tier, explored extensively in this study, is the effect of warmer temperatures driving large-scale land use and land cover change, such as agricultural expansion into forested landscapes. The second tier is the increasingly variable climate trends associated with long-term climate change. For example, global climate change is linked to more extreme local precipitation events (Min *et al*., 2011).

As explored in the study results, precipitation variability could contribute to water quality impacts in the basin. Periods of low precipitation were associated with higher pollutant concentrations, which may affect drinking water treatment for contaminants of concern. Conversely, higher precipitation events can increase pollutant loadings. While such events were generally associated with lower pollutant concentrations due to dilution, higher pollutant loading could result in impacts to aquatic ecosystem health or public health associated with eutrophication (i.e., harmful algal blooms). Other research indicates the potential for such precipitation-induced challenges for public water suppliers, but notes the importance of utility-specific assessment of climate change vulnerability (Arnell and Delaney, 2006).

While literature is sparse, researchers have explored implications of climate change on drinking water reliability and source water protection (Emelko *et al*., 2011; Arnell and Delaney, 2006). Emelko *et al*. (2011) discuss potential drinking water treatment considerations associated with more frequent and intense wildfires in forested source watersheds, finding higher levels of dissolved organic carbon (DOC), nutrients, metals, and other water quality parameters (Emelko *et al*., 2011). Though the impacts of such land disturbance is less of a concern in a basin as large as the study area, the findings reiterate the value of holistic source water protection strategies that build in adaptive management capacity in the face of uncertainty.

Arnell and Delaney (2006) explored climate change adaptation in the context of utility planning and decision-making. The researchers observed high awareness of climate change as a challenge among public water suppliers, but also somewhat limited ability to employ adaptation strategies (Arnell and Delaney, 2006). The authors further note the significance of regulatory drivers in guiding utility decision-making, which suggests that climate resiliency considerations should be framed around regulatory compliance. As utilities look to hedge against uncertainty associated with climate change and bolster more holistic source water protection efforts, large-scale watershed conservation strategies may become more attractive, even absent a short-term economic ROI case.

Recommendation:

- Consider engaging utilities and municipalities from a climate resiliency perspective, as drinking water providers may be more inclined to use source water protection strategies to hedge against uncertainty associated with climate change.

Enabling Conditions for a Positive Return on Investment

The findings of this study indicate that a purely economic ROI case for drinking water utility investment in an Upper Mississippi River Basin conservation initiative may be elusive. For the utilities of interest and the pollutants assessed, the land use perturbations applied were not sufficient to alter water quality at a level meaningful enough to affect drinking water treatment costs. This does not mean that drinking water utilities have no role to play in basin-wide conservation programs. Numerous case studies have assessed the potential for public water suppliers to attain an ROI via an investment in watershed conservation efforts. The findings of this study suggest that factors such as the nature of the pollutant of interest, the size of the basin, and the presence of regulatory drivers may set the enabling conditions under which a drinking water utility investment in a watershed conservation fund would achieve an ROI.

Types of Pollutants

Pollutants have unique effects on drinking water treatment processes. The potential for conservation programs to generate meaningful cost savings for drinking water providers depends largely on the nature of the contaminant. Some contaminants, such as sediment, are common in raw water and existing infrastructure at a drinking water treatment plant is generally intended to provide removal. Filtration systems, settling basins, and the addition of flocculants are common practices among drinking water systems, particularly those dependent on surface water supplies. For these contaminants, the cost of increasing pollution is linked to an increase in variable costs—such as longer settling times or higher chemical costs. These contaminants may be called *direct effect* pollutants, since any increase or decrease in pollutant concentration would correspond to a change in costs.

Other contaminants, such as nitrate, are less commonly present at levels exceeding human health standards and require costly treatment to remove (e.g., ion exchange, microbial denitrification). Drinking water treatment providers are unlikely to have infrastructure in place to remove such contaminants. Consequently, when the concentration of these pollutants exceeds human health standards (i.e., a Maximum Contaminant Level (MCL) under the Safe Drinking Water Act), treatment requires a significant upfront investment on the part of the utility. Once this fixed cost is borne, the cost of increasing pollution is once again linked to variable costs, such as the cost of chemicals or energy to run the contaminant removal system. These contaminants may be called *threshold* pollutants, as increases in pollutant concentrations only correspond to a change in treatment costs when concentrations approach or exceed a regulatory threshold.

Figure 24 shows how changes in the concentration of direct effect and threshold pollutants affect drinking water treatment costs.

In theory, direct effect and threshold pollutants exist on the same gradient; a direct effect pollutant is simply a pollutant with a low enough threshold that occurs at high enough levels such that treatment in a drinking water system is commonplace. Functionally, though, these pollutants have vastly different implications for an ROI analysis of payment for ecosystem services programs. Direct effect pollutants may be more likely to generate

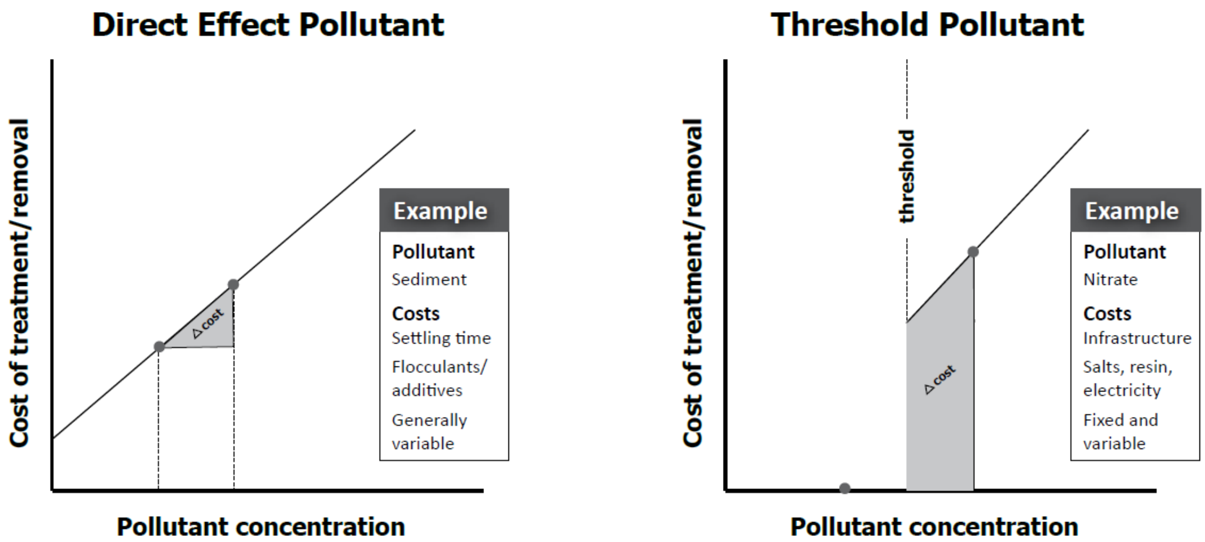

Figure 24. Visualization of the effect of changes in pollutant concentrations on drinking water treatment costs for direct effect and threshold pollutants.

some level of cost savings, as any reduction can be linked to a reduction in treatment costs. However, the reduction in direct effect pollutant concentration would likely need to be more meaningful in order for a utility to recoup the costs of an investment, as providers will likely still need to continue some, albeit reduced, level of treatment once conservation measures are in place. Alternatively, investment in upstream conservation and source water protection efforts may be worthwhile for a utility facing increasing concentrations of a threshold pollutant, as the drinking water provider may be able to avert significant fixed infrastructure costs. An ROI analysis in this case must focus on the risk of exceedances of regulatory thresholds. Utilities with a low risk of exceeding such thresholds are unlikely to realize an ROI through a conservation investment, as nominal changes in pollutant concentrations well below the regulatory threshold are of minimal consequence to a utility. Furthermore, an investment in conservation efforts must also prove sufficient to keep concentrations below regulatory thresholds, otherwise a utility would continue to face high fixed infrastructure costs and an ROI will not be achieved.

Basin Characteristics

The role of public utilities in payment for ecosystem/watershed services programs is a noted research gap, and while more utilities are expressing interest in such programs, demonstrating a solid business case for participation is a consistent barrier (Bennett *et al.*, 2014). Nevertheless, three case studies discussed in this report provide useful insights regarding the potential role of utilities in conservation and source water protection efforts and the conditions under which an ROI is attainable.

Case studies in New York, Ohio, and Brazil show how the nature of pollutants and regulatory drivers affect ROI attainability for drinking water utilities. However, these case studies also allow for a comparison across watersheds to better understand specific basin characteristics that may influence the feasibility of achieving an ROI. While a multitude of factors influence the economic case for drinking water utility investment in conservation initiatives, few influence it as greatly as the area of the basin from which drinking water is derived. **Figure 25** is a conceptual graph, where each case study is a point plotted on two axes. The horizontal axis indicates the scale of a pollution problem—ranging from specific pollutant concerns (e.g., exceedance of a regulatory threshold, costly chronic pollution challenges) to more general, less urgent water quality problems. An ROI attainability zone encompasses points near the origin, with a narrower pollution concern and a smaller basin size. In this graph, the Upper Mississippi River Basin lies on the far ends of both axes.

At nearly 52,000 km^2, the Upper Mississippi River Basin is over 12 times larger than the next largest basin among the selected case studies (Catskill/Delaware Watershed, New York, United States). The scale of the basin is the clearest difference between the watersheds where an ROI was attained and the Upper Mississippi River Basin, and it is likely a key factor that makes or breaks ROI for drinking water utility. This is because smaller basins are likely more vulnerable to land use and land cover change than larger basins. For a basin as large as the Upper Mississippi River Basin, the scale of land use perturbations—either to trigger or solve a water quality problem for a drinking water utility—must be similarly large. Large basins also move larger volumes of water, which can contribute to an in-stream dilution effect, making drinking water utilities more resilient to potential land use changes and buffering them from associated water quality problems. For smaller basins, small land use perturbations can generate greater water quality impacts. In these basins, drinking water utilities are more vulnerable to landscape-level land use changes, but conservation solutions may also prove more effective.

The vulnerability of small basins to water quality impacts may be one reason large drinking water utilities rarely rely on surface water resources that drain small watersheds. The Catskill/Delaware Watershed case study is a bit of an anomaly, in that the basin size is relatively small for supplying a city as large as New York. In this case study, the regulatory driver of the Surface Water Treatment Rule served as a key regulatory threshold that bolstered the ROI case for the drinking water provider. Even in a watershed of relatively small size, the necessary land use perturbations were significant, as the City of New York committed hundreds of millions of dollars to land acquisition efforts in an already heavily forested watershed.

Finally, the ROI attainability zone delineated in **Figure 24** is not static, but rather it may change with respect to utility interest, shifting regulations, climate change, or other factors. It is not hard to envision how factors such as lax enforcement of regulatory standards or unanticipated and unpredictable climate variability associated with climate change may alter the boundaries of the ROI zone for a given utility.

Recommendations:

- Consider the nature of pollution when conducting an ROI analysis for drinking water providers. Target source water protection efforts in basins or sub-basins approaching a regulatory threshold, or a basin with considerable direct effect pollution that significant affects overall treatment costs.

- Where possible, target source water protection efforts in smaller basins, where conservation interventions can generate a more meaningful impact on water quality at utility intake locations.

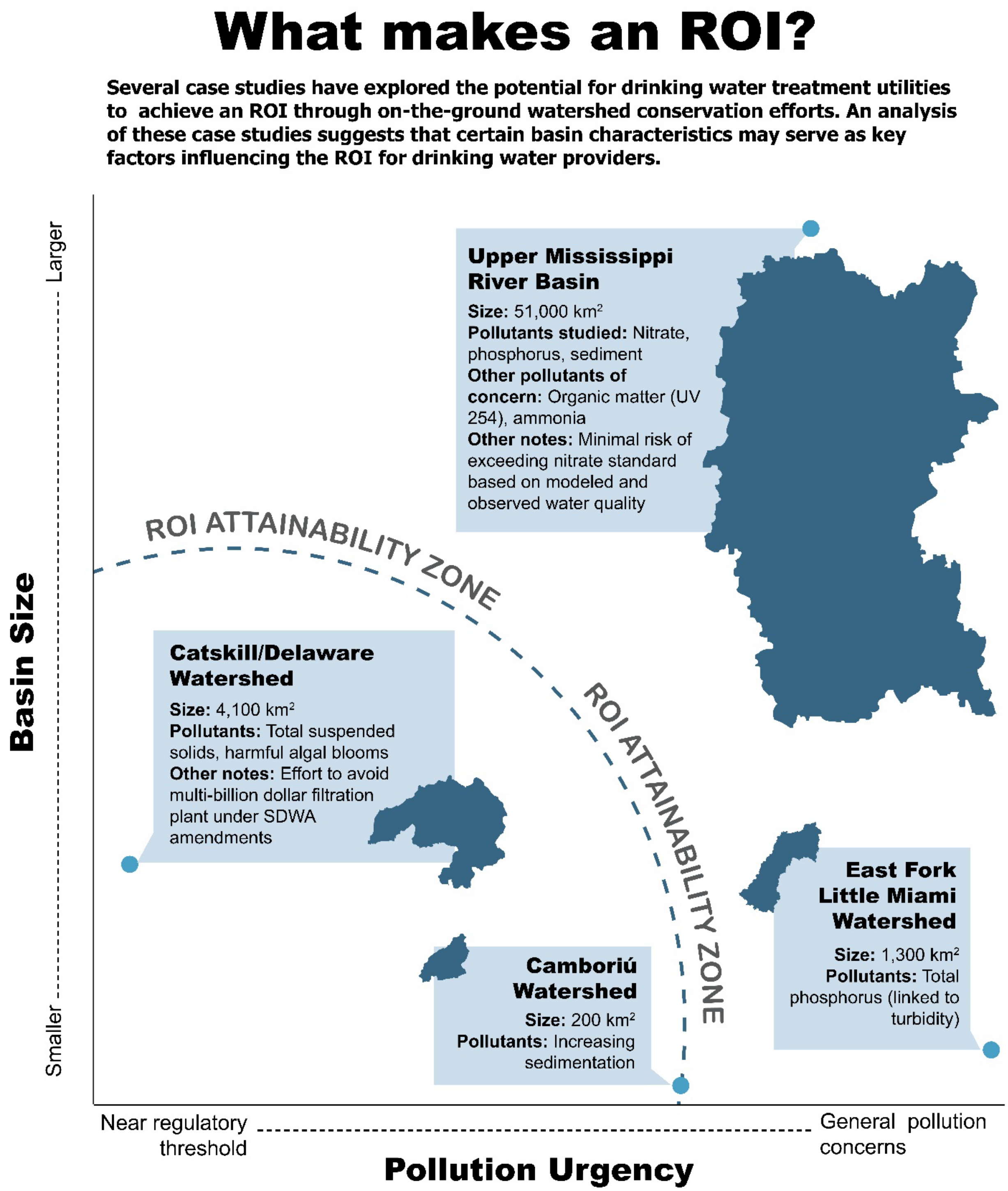

Figure 25. A conceptual graph of various ROI analyses for drinking water utility participation in payment for ecosystem/watershed services programs, informed by case studies and this report's analysis on the Upper Mississippi River Basin. Watershed sizes generally reflect the relative size of each basin.

Conclusion

The Nature Conservancy's Minnesota Headwaters Fund was created in order to proactively address the negative consequences of land use change and forest-to-cropland conversions within the Upper Mississippi River Basin. In recent years, Minnesota has experienced significant conversions of this type, followed by subsequent increases in nutrient and sediment concentrations within water bodies. As climate change threatens to push agriculture northward into predominantly forested areas in the region, there is a growing need for research to quantify the effects of land use change and resulting impacts to water quality.

In this study, the feasibility of water utility engagement and contribution into the Headwaters Fund was assessed in order to determine whether a return on investment (ROI) case could be made in the form of reduced treatment costs for drinking water. Interviews with large, medium, and small water utilities throughout the region provided valuable insight and context regarding this issue. Two major water utilities in the basin, located in the cities of St. Cloud and Minneapolis, noted that land use changes upstream of their intake locations would inherently affect their own operations. However, pollutants of concern varied by utility, with sediment and ammonium being a concern for some and organics, ammonia, and odor being a concern for others. Furthermore, utility conversations highlighted the fact that regulatory drivers dictate management decisions and that conservation initiatives would have to follow a clear path within existing regulatory frameworks.

The project modeled water quality in the basin across a spectrum of land use scenarios, including a baseline scenario, a moderate agricultural expansion scenario, and an aggressive agricultural expansion scenario. Land use changes in the moderate and aggressive agricultural scenarios were consistent with previous studies, showing an increase in cropland and urban land at the expense of forested land. Despite these changes in land use, water quality parameters such as total nitrogen and total phosphorous experienced only marginal increases (<1%), while sediment experienced marginal decreases. Further analysis of the model outputs showed that inter-annual variability of flow and precipitation in the basin may be a stronger driver for water quality at the basin-wide level than land use changes. However, on the sub- basin level, land use change may account for more significant changes in water quality. Analysis of model outputs at the sub-basin scale showed the largest relative increases in total nitrogen and total phosphorous yields in sub- basins located in the central portion of the study area, where cropland conversions are expected to increase.

Given that changes to total nitrogen, total phosphorous, and total sediment were marginal on the basin- wide level, it was unlikely that there would be a positive ROI for water utility engagement in the Headwaters Fund. Indeed, regression analysis of treatment costs in response to a 1% reduction in total phosphorous showed a net savings of $10,000 –

$18,900 for treatment costs in perpetuity. Since the total cost of land acquisition through the Headwaters Fund would be on the order of $10 million, there was not a positive ROI.

Despite these results, a number of key takeaways and enabling conditions were generated from the analysis. Comparisons between other case studies showed that ROI attainability varies with basin size and pollutant urgency. Larger basins with general, less imminent, pollution concerns are unlikely to generate an ROI under a similar scheme. However, smaller basins with pollutants approaching regulatory thresholds are likely to receive greater benefits from conservation efforts and avoidance of costs associated with developing treatment infrastructure for new pollutants. Furthermore, quantifying the benefits of source water protection beyond drinking water treatment costs could greatly increase the economic viability of such programs. Overall, a low ROI for water utility engagement should not be a barrier for implementing source water protection and headwaters conservation. Organizations such as The Nature Conservancy would be best suited incorporating these results as a small piece within their broader strategy, and continuing to identify other beneficiaries in order to maximize their efforts and fully capture the true range of benefits generated by headwaters conservation.

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Appendices

Appendix 1 – Water Quality and Treatment Costs Analysis

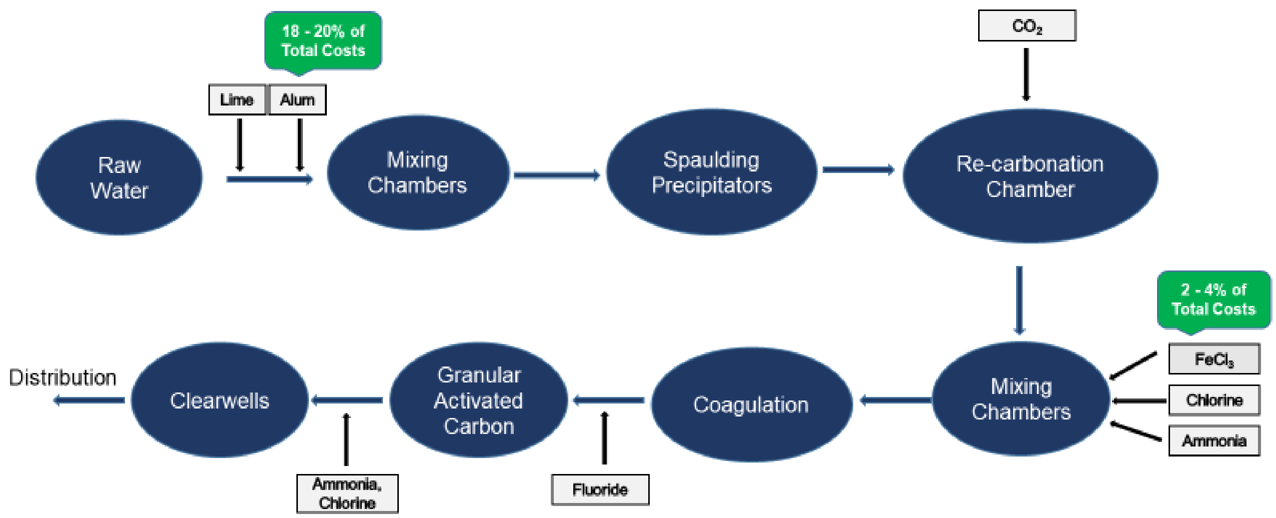

Figure A1. Schematic of the drinking water treatment process at Minneapolis.

Figure A1 shows all the stages that are involved in the drinking water treatment process at the Minneapolis drinking water utility at Fridley. The top four stages take place in the softening plant and the bottom four stages are in the filtration plant. A number of chemicals are added throughout the treatment process, starting with lime for softening, alum for coagulation, ferric chloride for coagulation/flocculation, chlorine and ammonia for disinfection, and fluoride to prevent tooth decay. Only alum and ferric chloride are important to estimate cost savings because these chemicals are added to remove the sediments and suspended particles from the water. Total Phosphorous (TP) concentrations are a good proxy for the amount of suspended particles in the source water because TP has affinity for clay particles (Heberling *et al.*, 2015). Hence, TP concentration reductions could result in reduced chemical usage to coagulate the suspended particles, which would result in cost savings. The average cost of all the chemicals used at the utility is around $4 – $4.5 million annually. Alum usage accounts for 18-20% of the costs, and ferric chloride usage accounts for 2-4% of the costs.

To estimate the cost savings from 1% reduction in TP concentrations, single variable linear regression models comparing log TP concentrations with pounds of alum and ferric chloride per million gallons (MG) were used. There is a significant negative relationship between log TP concentrations and alum usage (Alum (lbs per MG) = -61.325 log TP +

2.379, R^2=0.16, p<0.001) which means that as TP concentrations decrease, alum usage increases. This is counter intuitive to our understanding of how TP concentrations affect alum usage. Hence, we omit this result. Figure A2 shows the significant positive relationship between log TP concentrations and ferric chloride usage. The linear regression shows that a 1% decrease in TP concentration reduces the amount of ferric chloride required by 0.115 pounds per MG. Multiplying this value by the inflation adjusted cost of ferric chloride and the amount of water treated annually at the Minneapolis drinking water treatment utility, we get $340 of annual cost savings for the Minneapolis water utility from a 1% reduction in TP concentrations.

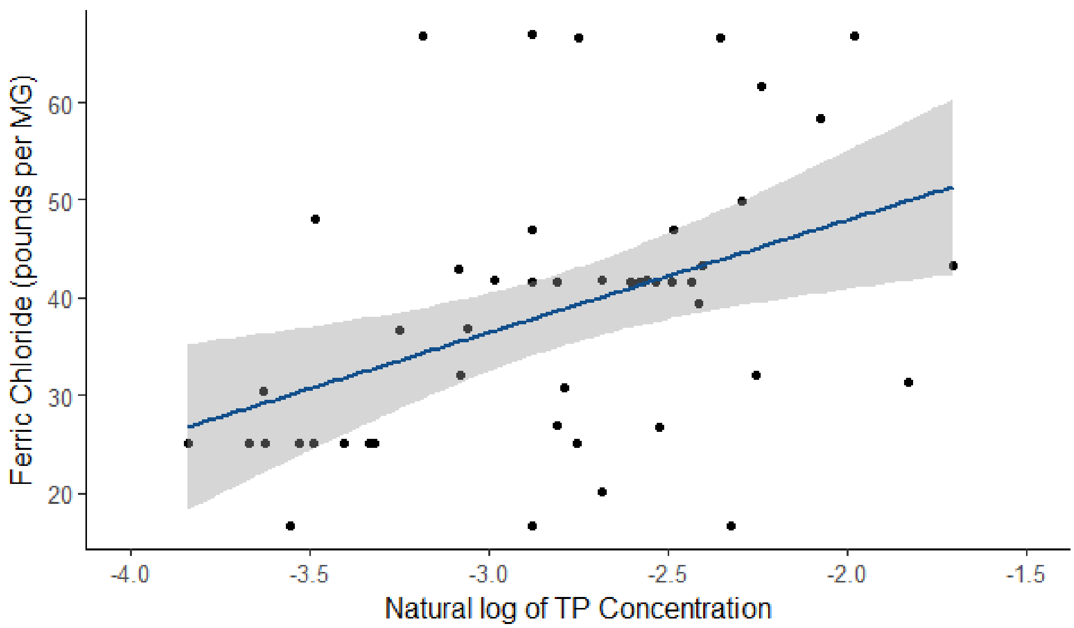

Figure A2. Linear Log Regression of Ferric Chloride vs TP Concentration. The dark blue line represents the significant positive relationship between TP concentration and ferric chloride usage (y = 11.51 x - 70.98, R^2=0.19, p=0.0016). The blue shaded portion represents the 95% confidence interval region.

The drinking water treatment process at St. Cloud is similar to the one shown in Figure A1 except that it does not involve the use of ferric chloride for coagulation. It just uses alum. There was no statistically significant relationship between log TP concentration and ferric chloride usage (R^2=0.02, p=0.236). All the regressions were performed based on the data provided by the Minneapolis and St. Cloud water utilities.

Appendix 2 – Utility Interview Questions

Questions for Utility Personnel

General Questions

1. Tell us a bit about your role with the utility and your community.
2. What is the population served by the utility and what is your service area?

Treatment Process

3. Can you give us a few technical details of plant operation, like what different treatment techniques are used? (Prompt: Can you walk us through the drinking water treatment process - from intake to distribution)
4. What are the main pollutants (if any) in the Mississippi River water that you are concerned about? Currently and in the future. (If not initially mentioned:) Are Nitrates and Phosphorus on your radar? (Prompt: More broadly, what challenges do you foresee on the horizon for your utility in terms of drinking water quality?)
5. What is the capacity of your utility to address these challenges, and what does the planning and decision-making process look like?
6. What treatment techniques do you use (if any) for treating nitrates? If you don't currently treat, would you have to upgrade your facility in the future?

Green Infrastructure and Engagement with TNC

7. To what extent has your utility engaged in source water protection efforts to date?
8. What is the key motivation/key drivers for your interest in Minnesota Headwaters fund in collaboration with TNC? (Prompt: Is it to reduce future economic risk for your utility or Regulatory compliance? Other?)
9. What are some obstacles that you foresee for engagement with the headwaters fund, or watershed conservation initiatives more broadly? (regulatory uncertainty, scientific/technical challenges, financial challenges etc.)
10. Are there any grey infrastructure alternatives that the water utility is looking into? What would you view as the advantages and disadvantages of this approach?

Data Availability

11. Where can we find data on the location of your water intake locations? Data on water quality at intake locations (preferably nitrate/phosphorus concentrations)?
12. Do you know how we might be able to access data on costs of operation of the water treatment plant and costs of the treatment processes? (for eg: USD/kWh of pumping, kWh/m^3 of pumping, USD/kg of coagulate, kg of coagulate/ m^3 of water etc.) or do you know who we might be able to contact to obtain such data?

Other Questions

13. To what extent is collaboration with other utilities (either other municipalities, or other utilities within your community) feasible when considering an investment in the Minnesota Headwaters Fund?

14. High-level question: what would it take for your city to seriously consider engaging in a watershed conservation initiative like the Minnesota Headwaters Fund? What do you see as essential, enabling conditions for this kind of investment to occur?