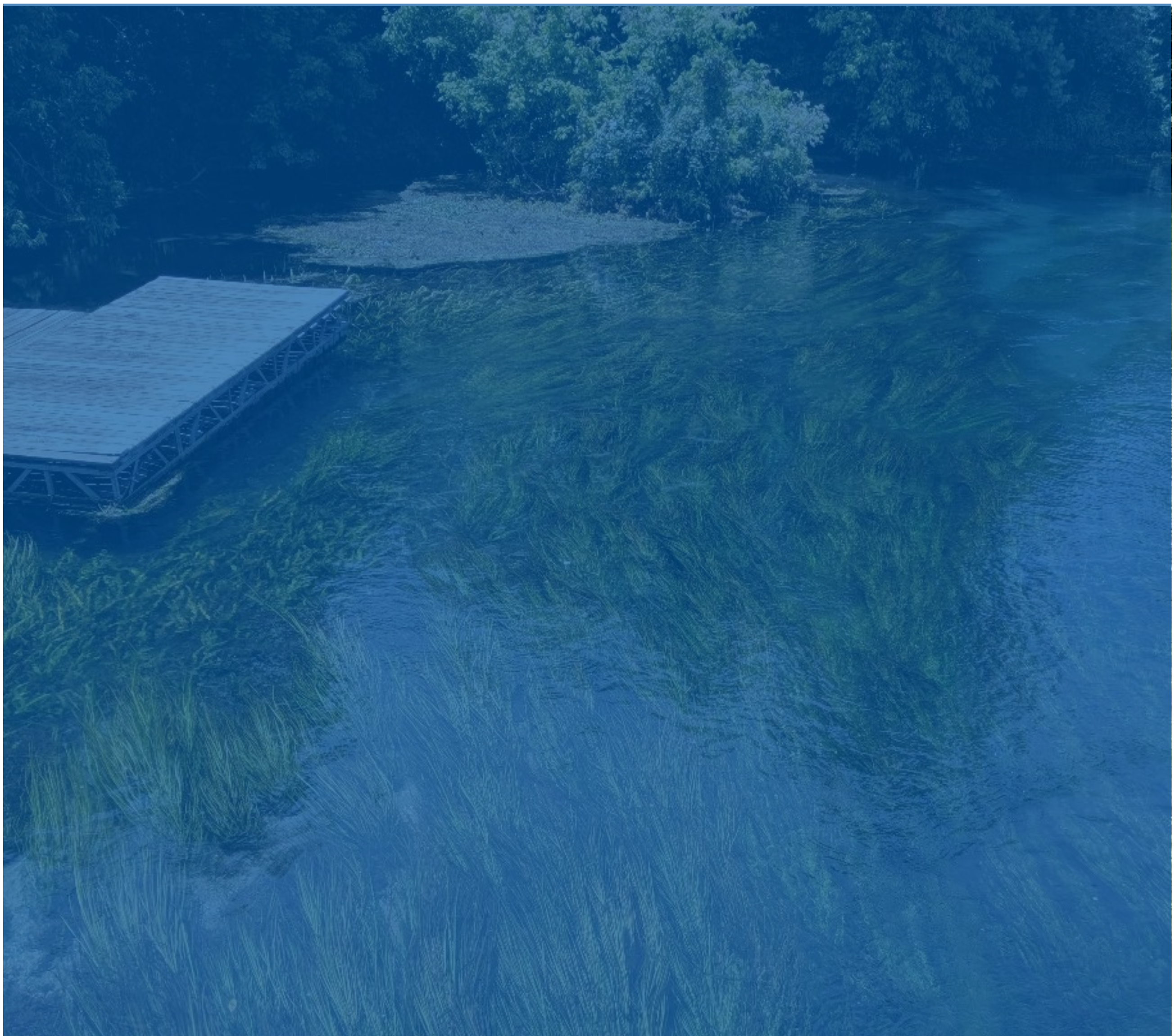

June 30, 2026

Protecting Minnesota's Headwaters of the Mississippi and Pineland Sands

Project Report in Fulfillment of the LLCMR Grant, ID Number: 2023-247

PROTECTING MINNESOTA'S HEADWATERS OF THE MISSISSIPPI AND PINELAND SANDS

Prepared for

White Earth Nation

June 30, 2026

Prepared by

With additional expertise provided by

Team Members and Report Contributors

Organization/Agency	Name
White Earth Nation	Jessica Johnson, Project Manager
	Andréa (Dre) Szabo, J.D., M.S.
	Michael Fairbanks, Chairman White Earth Nation
	Kellen McArthur
	Dustin Roy
University of Minnesota	Joe Magner, Ph.D. (Co-Project Investigator)
	John Nieber, Ph.D. (Co-Project Investigator)
	Kerry Holmberg, Research Scientist
	Joseph Anderson
	Brady Hartman
	Skylar Jensen
	Harprabhjot Kaur Dhaliwal
	Xiang Li, Ph.D.
	Leslie Ludtke
	Lillian Luensmann
	Phil Margarit, Ph.D.
	Bella Radler
	Lily Syzmanski
Bemidji State University	Trevor Wozniak
Young Environmental Consulting Group, LLC	Lauren Salvato, Project Manager
	Della Schall Young, Principal Scientist
	Emma Henefield
	Allegra Johnson McKee
	Rachel Kapsch
	Pooja Kanwar, Ph.D.
	Hannah LeClaire
	Suzy Lindberg
	Jenny Mocol-Johnson
	Benjamin Schiltz
Caddis Fish Consulting	Brenda DeZiel
Amiable Consulting and Mud Enviro	Amy Myrbo, Ph.D.
Red Lake Nation	Josh Jones
Watermark Numerical Computing	John Doherty, Ph.D.

Funding Acknowledgments

Funding for this project was provided by the Minnesota Environment and Natural Resources Trust as recommended by the Legislative-Citizen Commission on Minnesota Resources (LCCMR), LLCMR Grant, ID Number: 2023-247.

Some in-kind support was provided by White Earth Nation and the University of Minnesota.

ENVIRONMENT AND NATURAL RESOURCES TRUST FUND

On the Name: Ojibwa (Ojibwe, Ojibway), Chippewa (Chippeway), Anishinaabe (Anishinabe)

The people of White Earth Nation refer to themselves as Anishinaabe, which means “the original people” (Tribe, 1989). The term Ojibwe was given by the Europeans; however, it is not clear exactly where the name originated (Tribe, 1989). It is thought that some Europeans struggled to say Ojibway and pronounced it Chippewa (Tribe, 1989). Chippewa is the name officially used by the federal government in treaties with the Anishinaabe of Minnesota (Warren, 1885).

The term Anishinaabe or Anishinaabeg (the plural form) is used throughout the document for consistency and to follow how the White Earth Nation refers to themselves. The terms Ojibwe, Chippewa, or White Earth Nation are used only where legal or historical variants are cited in documents or statutes.

Use of Artificial Intelligence

The use of artificial intelligence (AI) was not strictly prohibited on this project, and guidelines were developed to use AI that maintains written originality and data reliability. The full policy and author-specific declarations can be found in Appendix 1: AI policy for Pineland Sands project.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

1 Project Purpose

Protecting Minnesota's Headwaters of the Mississippi Pineland Sands project is a multi-year, science-driven effort to assess the cumulative environmental impacts from land use changes over time within the Pineland Sands Aquifer (PSA) region in north central Minnesota. The term cumulative impacts refers to incremental activities that alter land use and can, over time, lead to more serious overall impacts in a watershed (Brooks, Ffolliott, & Magner, 2013). This region encompasses parts of the White Earth Reservation and the 1855 Treaty Ceded territory and is culturally significant to the Anishinaabeg for its abundant wild rice. It is also important ecologically, as it is a critical headwater area for the Mississippi River.

Minnesota's Pineland Sands region has experienced rapid growth in irrigated agriculture. Although agricultural growth plays an important role in the economy, its lasting effects on water quality, water quantity, and ecosystem are not fully understood. The region's complex hydrogeology—characterized by sandy soils overlying interconnected and semiconfined aquifers—has historically limited the availability of sufficient data and modeling to understand basin-wide impacts. These challenges became acute during the 2021 summer drought, when water levels declined dramatically, complaints about interference with private wells surged, and Indigenous and local communities faced immediate threats to water security and treaty-protected resources. This uncertainty has prompted a comprehensive evaluation to better understand the cumulative environmental impacts of the growth in irrigated agriculture.

The project's primary goals are as follows:

1) Assess aggregate water quality and quantity impacts from irrigated agriculture through enhanced groundwater and surface water monitoring and advanced predictive modeling;
2) Conduct a comprehensive analysis of existing state, federal, and tribal laws and policies related to water uses, including high-capacity wells; identifying strengths, weaknesses, and regulatory gaps; and
3) Identify and advance opportunities for an integrated, scientifically sound policy approach that balances water protection, treaty-protected rights, recreational uses, and agricultural production for current and future generations.

To achieve these goals, the project has multiple core activities:

- Baseline assessment and characterization of past and current hydrologic data to understand how land use changes impact hydrology over large time scales.

- Groundwater and surface water monitoring to characterize the current water quality and biological and ecological conditions.
- Groundwater and surface water scenario-based modeling with modular finite-difference flow model (MODFLOW) and soil and water assessment tool (SWAT+) to examine how land use changes, irrigation, and changes in climate affect the region's hydrologic processes.
- Engagement with tribal and local stakeholders to share project information as well as listen and learn from residents and users in the project area.
- Policy and legal review to assess how current regulations and governance frameworks influence water sustainability in the Pinelands Sands region in comparison with other regulatory schemes.
- Data dissemination through publications and conference presentations to promote transparency and facilitate productive conversations regarding the project's goals.

The primary project partners are the White Earth Nation (WEN) and the University of Minnesota (UMN). WEN is leading the effort to write the legal and policy aspects and incorporate critical historical knowledge of wild rice and the environment in the region. UMN is providing hydrological, biological, and geospatial expertise for field monitoring and model creation efforts. Consultants are providing additional specialized analyses.

2 Project Background

The project study area is located in north central Minnesota (MN) and encompasses the northern portion of the PSA region. It is approximately 592 square miles and spans parts of Wadena, Hubbard, and Becker counties. The northwestern section extends into the WEN Reservation. This landscape lies within the 1855 Ceded Territory, an area of importance for treaty-protected rights for the Anishinaabe people (Figure 1). The Pinelands Sands project area is part of the Crow Wing Watershed, which is part of the larger Upper Mississippi River basin that eventually drains to the Gulf near Louisiana.

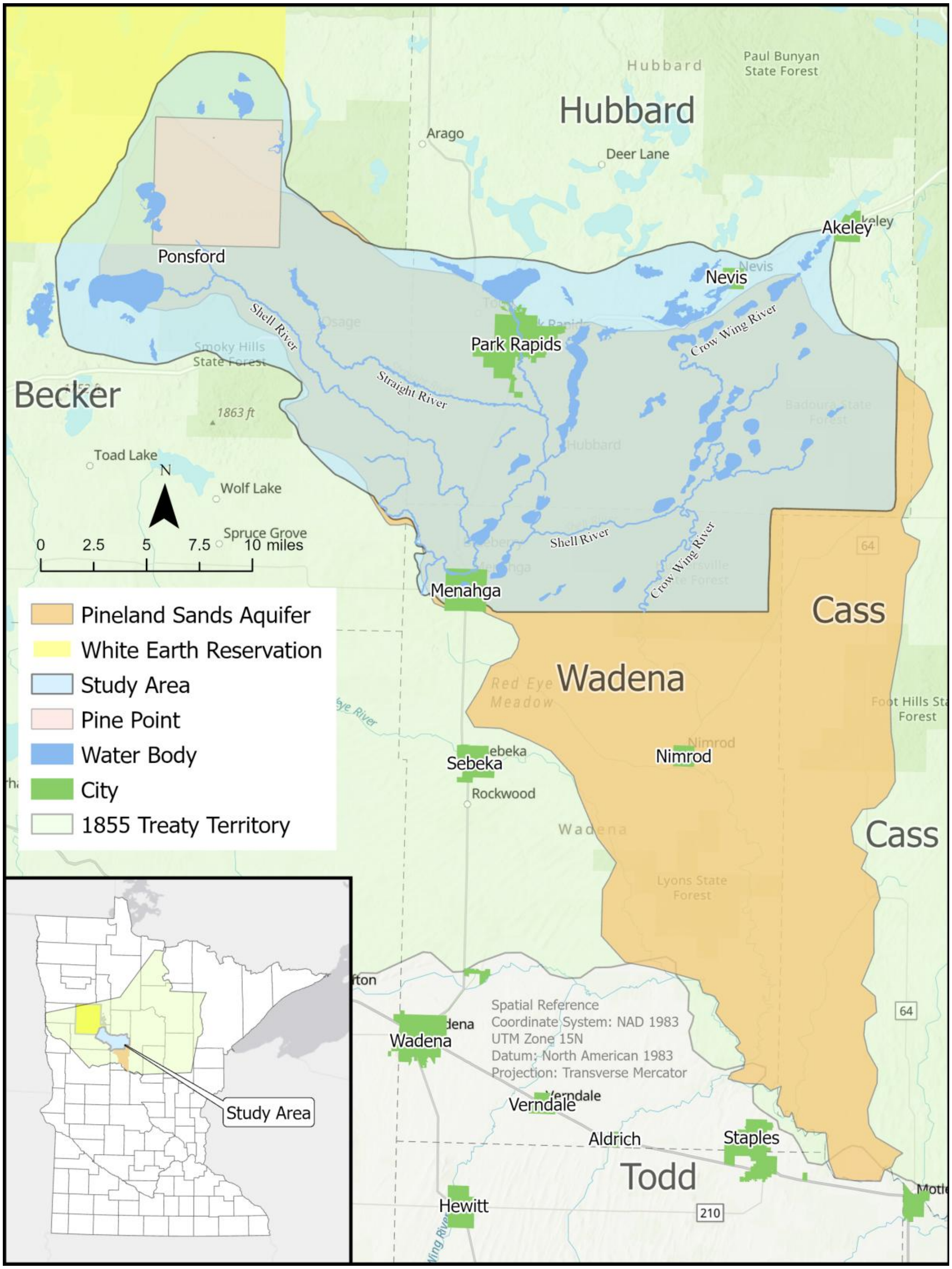

Figure 1: Project study area

Historical Context

The story of the Anishinaabeg[1] begins in time immemorial, but the geographical context for this project begins in the 1800s. In Minnesota, many Tribal nations ceded their lands to the United States (U.S.) government in the early 19th century, often in exchange for the repayment of debts typically imposed upon them or annual payments in the form of annuities, among other reasons (MHS, n.d.). A few treaties with the Minnesota Chippewa included specific language concerning rights retained in the lands ceded to hunt, fish, and gather (Kappler, 1838). The WEN asserts that the 1855 Treaty's Ceded Territory to the State of Minnesota (State) includes the continued rights to hunt, fish, and gather on those lands; however, others may contest this.

For the purposes of this project, the project area overlaps with the 1855 Ceded Territory as well as the footprint of the WEN reservation (Figure 2). The Treaty with the Chippewa of Mississippi of 1867 created the WEN reservation. The Treaty established approximately 829,443 acres of reservation consisting of 36 townships, a square of 36 miles on all sides, that contained "prime farmland in the Red River Valley, valuable stands of pine timber, and lakes and streams supporting seasonal resources" (Meyer M. L., 1994, p. 1) (Weil, 1989).

Wild Rice

Wild rice is an important cultural and nutritional staple for many Indigenous people in the western Great Lakes region and is the Minnesota State Grain. It is also important habitat for birds, including game species such as ducks. Known as Manoomin in Anishinaabemowin and Psiŋ in Dakota, the dominant species in Minnesota is northern wild rice (*Zizania palustris*), an annual aquatic grass. The plant grows in shallow (1-5 feet deep), clear, very fresh (low-salinity) waters. Today, Manoomin continues to be harvested and enjoyed by the Anishinaabeg and others alike throughout Minnesota.

The geographic range of wild rice has decreased markedly since the time before European settlement, when the plant was found as far west as the Rocky Mountains (Jenks, 1900). Land use change has been a major cause of the loss of wild rice water bodies, including ditching to drain land for agriculture by eliminating wetlands; shoreline development; wastewater discharge and runoff from mining and water treatment plants; agriculture; disturbance by animals including swans, carp, and beavers; and competing plant species, both native and introduced.

[1] The people of White Earth Nation refer to themselves as Anishinaabe, which means "the original people" (Tribe, 1989). Tribes are also called Ojibwe or Chippewa. The term Ojibwe was given by the Europeans, however it's not clear exactly where the name originated (Tribe, 1989). It is thought some Europeans struggled to say Ojibway and pronounced it Chippewa (Tribe, 1989).

Within the study area, there are numerous lakes with large wild rice populations that have been identified by the Minnesota Department of Natural Resources (MN DNR) (MN DNR, 2024a). WEN and other Anishinaabeg tribes have also identified their own lakes with wild rice populations that are often more expansive than the State's list.

As a treaty-protected resource, the health and distribution of wild rice, as well as other ecosystem inhabitants, are closely tied to water quality, hydrologic conditions, and management decisions. The scientific questions evaluated through monitoring, modeling, and policy analyses in this project are therefore closely connected to treaty-protected rights.

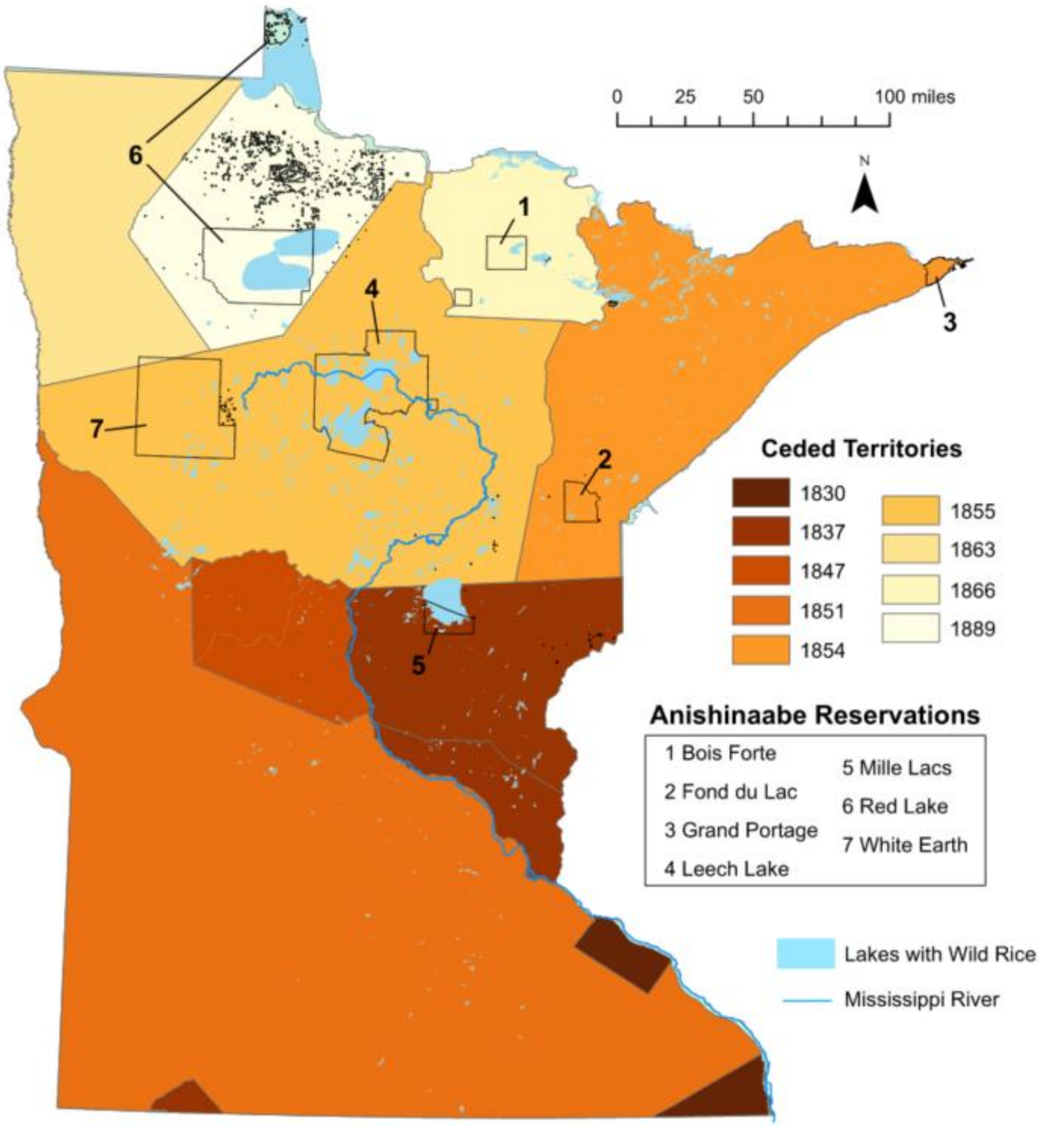

Figure 2: Map of Ceded Anishinaabe territory in Minnesota by treaty years

Characterization of the Study Area

Land Use and Population Growth

Before the settlement of American travelers heading west during the 1800s, Central Minnesota was covered in dense forest, bogs, wetlands, and native grasses (Winchell & Upham, 1888). The Anishinaabeg relied on the land and water for hunting and gathering (McClurken, et al., 2000). By 1888, commercial logging was underway in the region, paving the way for the agriculturally dominated landscape seen today (Winchell & Upham, 1888). Throughout the 1900s, more forested land was converted into farmland (McClurken, et al., 2000). Land conversion continues today (Stroom, 2024). From 2001 to 2019 alone, forest cover was reduced by about 9.4% in the project area, according to data derived from the National Land Cover Dataset (NLCD) (Dewitz, 2021). The 2019 NLCD characterizes the land use in the project area in Figure 3 (Dewitz, 2021). Some notable colors represent the prevalence of open water (blue), forested areas (light and dark green), and cultivated crops (brown). Currently, row-crop agricultural production in the area is primarily corn, dry beans, and potato fields. Irrigation with groundwater, nitrate fertilizers, and pesticides is needed to sustain the summer growth of these row crops, especially corn and potatoes.

Population growth has steadily increased since the 1970s within each of the three counties that the project area spans: Becker, Wadena, and Hubbard counties (U.S. Census Bureau, n.d.-a) (U.S. Census Bureau, n.d.-b) (U.S. Census Bureau, n.d.-c). Between 1971 and 2024, populations rose from about 25,000 to 35,400 in Becker County (+40%), from 11,000 to 22,100 in Hubbard County (+101%), and from 12,400 to 14,400 in Wadena County (+16%) (U.S. Census Bureau, n.d.-a, n.d.-b, n.d.-c). Combined, the population growth represents an approximately 49 percent increase. With changes in land conversion and population growth in the area, dams were built, along with roads to cross streams and culverts. Much of this growth is concentrated in larger cities like Park Rapids and Detroit Lakes, which experience an influx of tourism during the summer months (U.S. Census Bureau, n.d.-a, n.d.-b, n.d.-c). This increasing population corresponds to increased pressure and seasonal use of the local groundwater and surface water resources.

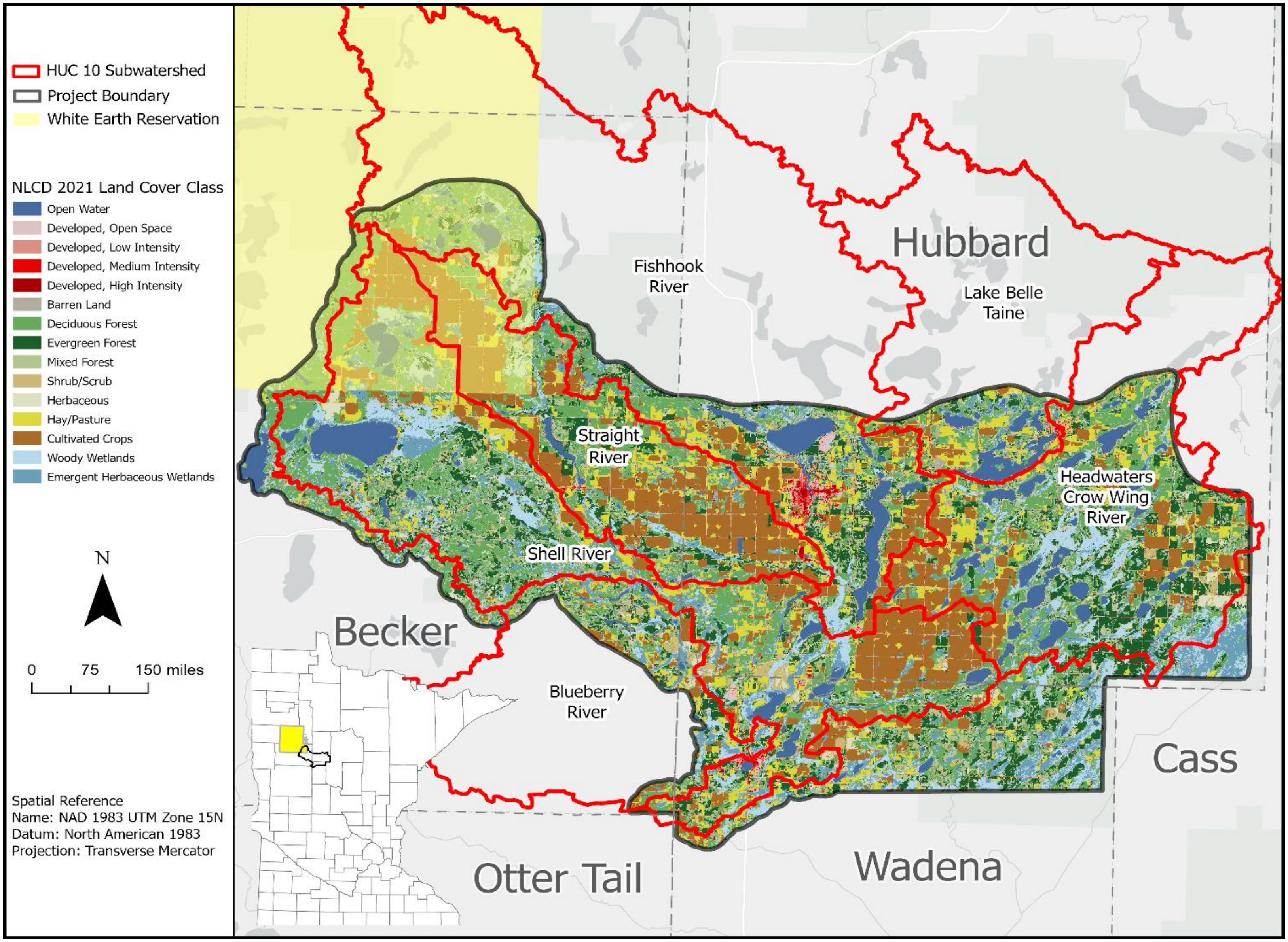

Figure 3: Land use in the project area

Geology and Hydrology

The PSA is a shallow groundwater system composed mostly of sand and gravel that was left behind by glaciers thousands of years ago. These sandy deposits sit atop occasional layers of clay and dense glacial material. In some places, the water-filled (saturated) sand can be more than 100 feet thick, and the total thickness of these glacial sediments can exceed 300 feet (Heglesen, 1977) (Lusardi, 2016) (Bauer, et al., 2016) (Lusardi, 2018) (Stark, Busch, & Deters, 1991). Beneath all of this is hard, ancient bedrock, Precambrian crystalline bedrock (Heglesen, 1977).

The landscape in this region is mostly flat with gentle hills. Glaciers shaped the area, leaving behind features such as small hills, lakes formed by meltwater, and broad sandy plains. These sandy landforms allow water to infiltrate into the groundwater very quickly and move easily through the aquifer (Lusardi, 2016) (Bauer, et al., 2016) (Lusardi, 2018). Some of the materials left behind by glacier movement were minerals and rocks consisting of gypsum and pyrite, respectively (MN DNR, 2024b) (Lindgren, 1995) (Meyer G. , 1986). Gypsum is a soft mineral composed of calcium sulfate dihydrate, and pyrite is a rock consisting of iron sulfide.

The study area's elevation slopes gradually toward the Crow Wing River in the southeast portion of the study area (Figure 1). The soils are dominated by coarse sandy soils, which have large pore space between particles, allowing water to move from the soil surface quickly to the subsurface. Sandy soils are also associated with a lower organic matter content and low retention of nutrients (Lusardi, 2016) (Bauer, et al., 2016) (Lusardi, 2018) (Yost & Hartemink, 2019) (Magdoff & van Es, 2021). These conditions make farming more difficult without irrigation, because crops do not retain water or nutrients easily. The same characteristics also make the area more vulnerable to pollution, especially from nitrates, sulfate, and pesticides, which can quickly move down into groundwater.

The region's sandy soils and glacial geology mean that its groundwater and surface water are closely connected. The three main rivers in the area are the Straight River, the Shell River, and the Crow Wing River. The Straight River is a cold-water trout stream that receives over 90% of its flow from groundwater. This type of river is called a "gaining stream" because groundwater flows into it. The Shell and Crow Wing Rivers are warmer streams that support wild rice. There are also hundreds of lakes and ponds throughout the area, many of which are important for recreation, such as fishing and boating, as well as for wild rice harvesting.

Ecology

The project area is classified as the Pine Moraines and Outwash Plains Section. Its sandy, well-drained soil shapes the ecology in the area (Hobbs & Goebel, 1982). Kettle lakes are common in this landscape, especially in areas formed by glacial deposits like outwash plains and moraines. Total precipitation varies within the region, from 23 to 27 inches annually (National Integrated Drought Information System, n.d.). Most precipitation occurs during the growing season (40%), with limited snowfall in the winter months (10–12%).

Before commercial logging, the most common vegetation was jack pine (Waters, 1977). Jack pine (*Pinus banksiana*) tends to occupy sandy soils unfit for white or red pine because of their low organic matter content. Jack pines are considered a pioneer species, usually the first pine to re-emerge following forest fires, which occur every 10 to 40 years (MN DNR, n.d.-a). Aspen, birch, and pine forests (mixed red and white pine) were also present in the landscape. Areas with well-saturated soils supported conifer bogs, swamps (wetlands), and wet prairies. The Public Land Survey in Minnesota, conducted between 1848 and 1907, indicated that the project area was predominantly woodlands[2]. The jack pine-dominated landscapes and openings category was observed across 84% of the Straight River subwatershed, 74% across the Shell River subwatershed, and 83.5% across the Headwaters Cross Wing River subwatershed (MN DNR, 2022).

The information in Table 1 compares disturbed land use in the 1848 survey with land use in 2019 specifically. Note that the survey techniques and categories of vegetation were not the same but did have overlap, which is why only disturbed land use is presented. Today, disturbed land use accounts for 56.5% of the Straight River subwatershed, 32.5% of the Shell River subwatershed, and 26.5% of the Headwaters Crow Wing subwatershed (Table 1).

[2] The surveys were not intended to be a complete record of all species, but the size and types of larger trees used as bearing trees were recorded, as well as descriptions of the landscape (MN DNR, 2022).

Table 1: Land distrurbance comparison between the 1850s and present day

Subwatersheds of the Crow Wing River watershed	Straight River	Shell River	Headwaters Crow Wing River	Fish Hook River	Blueberry River	Lake Belle Tain
Area (square miles)	82.5	175.2	148.3	285.7	95.7	112.5
Public Lands Survey 1848 to 1907 (MN DNR, 2022) (% disturbed)	0	0	0	0	0	0
Combined Disturbed Land Use Including: cultivated crops; pasture/hay; and developed/open/barren Land Cover/Land Use (Dewitz, 2021)) (% disturbed)	56.5	32.5	26.5	17.4	42.6	13.0

Rivers and Lakes

As previously mentioned, the three primary rivers flowing through the project area are the Straight, Shell, and Crow Wing Rivers. The Straight River is designated as a Class 2A cold-water trout stream (MN DNR, 2025a). It is a gaining stream with baseflow from groundwater that accounts for over 90% of its total annual streamflow (Stark, Busch, & Deters, 1994). The Shell and Crow Wing Rivers, which have wild rice stands, are designated as Class 2B streams, which are warmer waterbodies (MN DNR, 2025a).

The impairments for both rivers and lakes in the project area are highlighted in Figure 4 (MPCA, 2024). The Straight River and a segment of the Shell River both have dissolved oxygen impairments, while another segment of the Shell River has an aquatic life impairment based on fish bioassessments. The Crow Wing River has an impairment for mercury in fish tissue. The lakes in the project area have numerous impairments due to mercury in fish tissue. Five lakes in the project area also have nutrient impairments: First Crow Wing, Eighth Crow Wing, Portage, Lower Twin, and Blueberry Lakes. Nutrient impairments indicate that too much phosphorus or nitrogen (collectively referred to as nutrients) end up in the lake ecosystem. Too many nutrients can lead to algal growth, which can turn into toxic algal blooms. Elevated nutrient levels in a waterbody can also reduce oxygen levels, making conditions less ideal for aquatic species. This can, in turn, pose challenges for many species, especially trout, which rely on well-oxygenated waters.

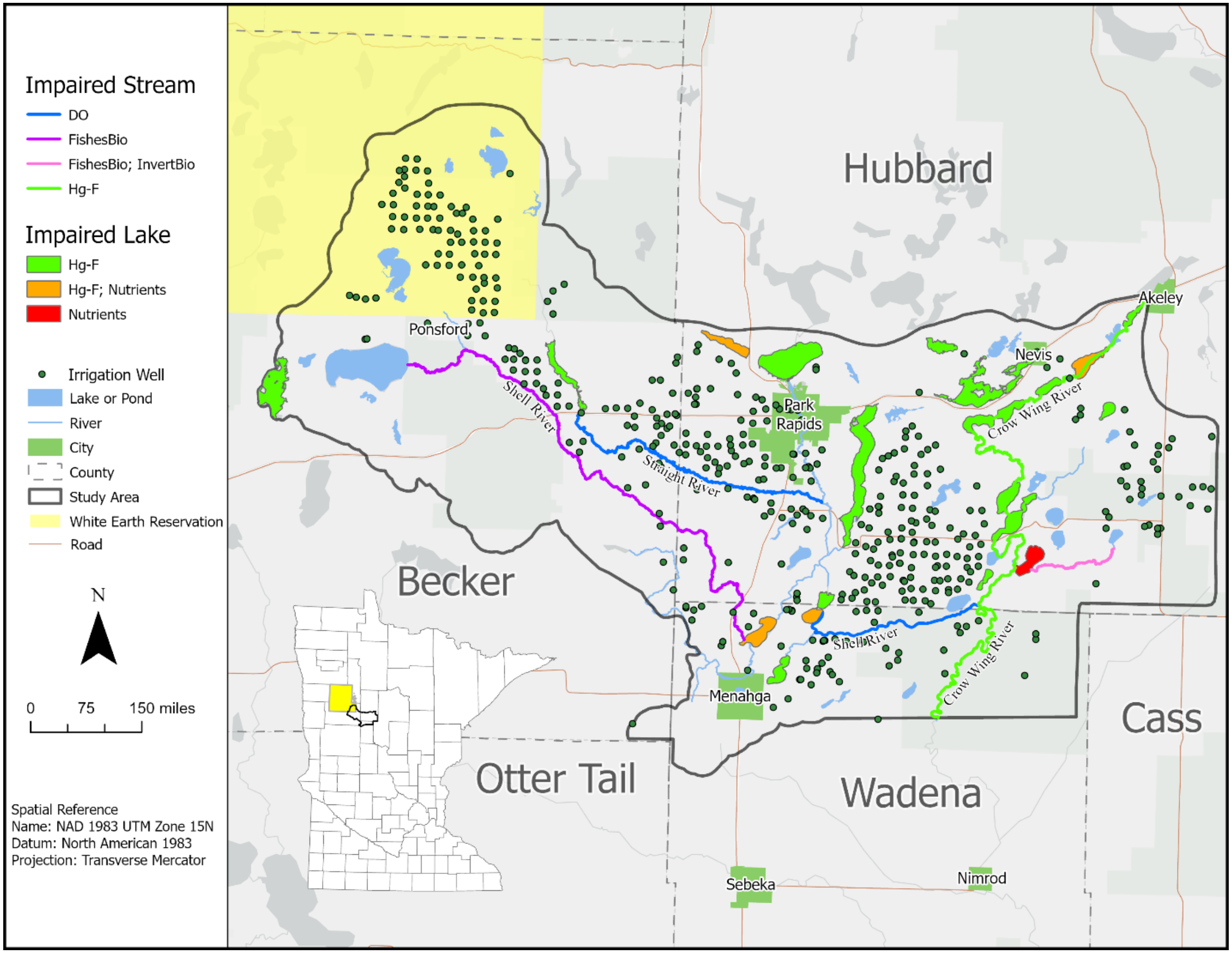

Figure 4: Impaired waterbodies in the project area

Groundwater Quantity and Use

In 2024, the MN DNR permit database registered 442 active high-capacity agricultural irrigation wells within the study area. Collectively, these wells were permitted to pump 19.3 billion gallons annually (MN DNR, 2024c). The increased allocation of groundwater pump overtime is substantial (Figure 5), and currently eighty four percent of the groundwater used in the Straight River GWMA is used for agricultural irrigation (MN DNR, 2017a). The growing concern about groundwater withdrawals, particularly those for agricultural irrigation, led to the creation of the Straight River Groundwater Management Area (GWMA) (MN DNR, 2017a). The GWMA plan was developed between 2016 and 2018 by the MN DNR and stakeholders in the GWMA (MN DNR, n.d.-b). The actions outlined in the plan are intended to help MN DNR better understand how water flows in the area and how to manage it sustainably while protecting the Straight River ecosystem. The Straight River GWMA is one of three areas in the State with this designation.

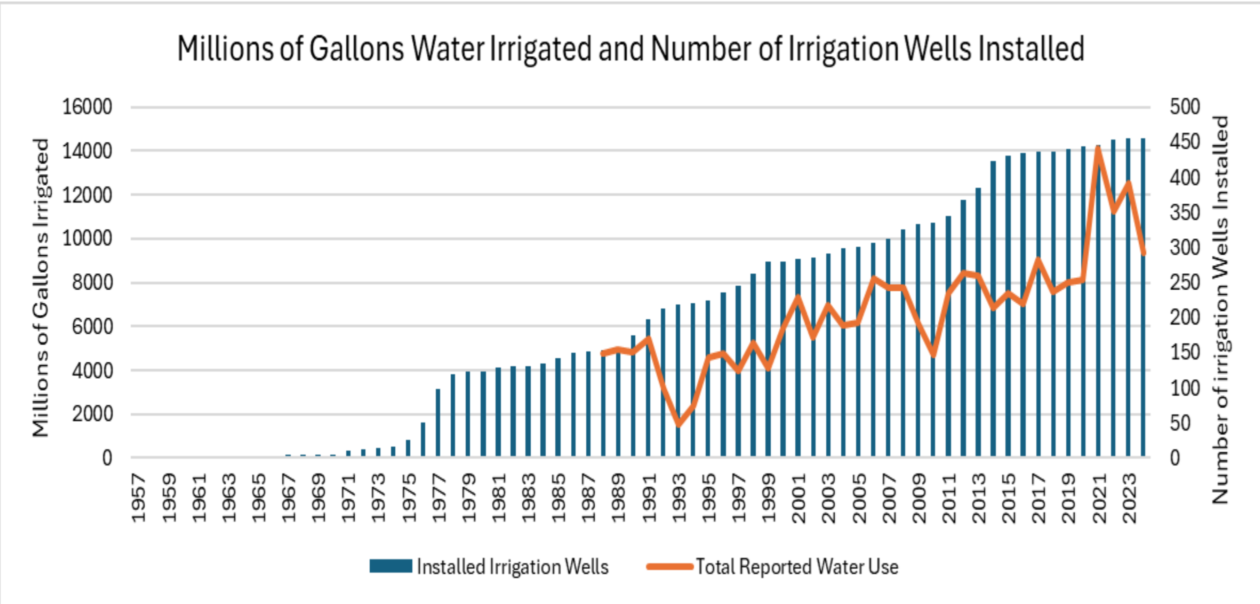

Figure 5: Water use by category in the Straight River GWMA

MN DNR led the development of the action plan for the Straight River GWMA, determining a set of actions to study the GWMA more closely and releasing the report in 2024. The report outlined the following key information and interpretations (MN DNR, 2024d):

- Between 93 and 97% of groundwater provides baseflow to the surface waters in the GWMA. The aquifer's confinement is thin, and groundwater flows freely between shallow and deeper aquifer systems in the GWMA.

- Water use increased approximately 137% from 1988 to 2021, largely due to irrigation. Many of those irrigation wells are located near streams.
- On the other hand, based on available monitoring data and specific to the scope of the Straight River GWMA, MN DNR stated, “There does not appear to be long-term sustainability concerns for the Ponsford submanagement area, but there are specific locations where pumping close to the stream and other groundwater streamflow areas limit the amount of baseflow reaching the stream.” It should be noted that there was a limitation in MN DNR’s period of record, and information on groundwater levels and streamflow began well after agriculture and irrigation had become dominant in the watershed.
- Nitrate contamination is a major issue, and the Straight River has higher nitrate levels than other rivers in the surrounding area.
- A Hubbard County private well testing effort collected pesticide samples to follow up on 357 wells with nitrate detections within the GWMA. At least one pesticide was detected in 204 of these 357 wells (57%).
- Because groundwater and surface water are highly connected, this interconnectivity poses ecological threats to species like trout, which rely on cold groundwater upwelling into surface water to survive. This leaves both groundwater and surface water vulnerable to contamination when nitrates or pesticides are applied to the landscape.

Groundwater Quality

The Environmental Protection Agency (EPA) establishes the maximum contaminant level (MCL) for nitrate in drinking water through the Safe Drinking Water Act. Nitrate levels higher than 10 mg per liter (mg L^{-1}) pose a risk to newborns, increasing their likelihood of contracting blue baby syndrome (MDH, 2025). The private well testing effort in Hubbard County townships in 2016 and 2017 assessed the number of private wells exceeding the federal drinking water standard (10 mg L^{-1}). The results of the private well study found that 15.2% of the 223 wells tested within Hubbard County had nitrate levels that exceeded the federal drinking water standard (MDA, 2018)(Table 2).

Table 2: Final well datasets for Hubbard County private well nitrate results, 2016-2017

Township	Total Wells	Percent of Wells greater than 10 mg L^{-1} Nitrate-Nitrogen
Badoura	38	13.2%
Crow Wing	202	7.4%
Henrietta	254	2.4%
Hubbard	223	15.2%
Straight River	128	7.0%
Todd	203	5.4%
Total	**1,048**	**7.6%**

Other water-quality contamination concerns in the area include pesticides. A different private well study looked at pesticide contamination in the area in 2017 and 2018 (Schaefer, Busch, Johnson, Meyer, & VanRyswyk, 2022). The study found that 46%, 57%, and 85% of all the wells tested in Becker, Hubbard, and Wadena, respectively, had detections of pesticides (Schaefer, Busch, Johnson, Meyer, & VanRyswyk, 2022).

The City of Park Rapids has also experienced elevated nitrate levels in its public drinking water supply. A deeper municipal well was installed for the City of Park Rapids due to high nitrate concentrations in the original shallower well (City of Park Rapids, 2018). The City's source water protection plan outlines strategies to keep drinking water safe. However, the plan acknowledges that the sand and gravel aquifer makes drinking water more vulnerable to contamination (City of Park Rapids, 2018). Although the deep well has clay and offers more protection, contaminants on the land surface can still reach groundwater through the interconnected layers of the PSA (City of Park Rapids, 2018) (Margarit, et al., 2025).

Climate Change Impacts

Climate data between 1985 and 2023 for north central Minnesota indicate that the average annual temperature has increased by 3.7 degrees Fahrenheit since 1985 (Coffman, et al., 2024). The climate model for the north central region revealed that the largest temperature increases are occurring during the winter months. Coffman et al. also reported an annual precipitation increase of 1.5 inches. According to their model predictions, by mid-century, the area is likely to experience an increase in temperature, a decrease in the number of days below freezing, a reduction in snow cover, and an increase in the amount of precipitation during big rain events (Coffman, et al., 2024). This climate analysis has many potential implications for the project area. Fewer days of snow cover could extend the agricultural growing season, spurring economic development in the area while keeping land as more intensive cropland for a longer period of the year. Higher temperatures increase evapotranspiration and could lead to crops requiring more irrigation and to increased growing-season groundwater withdrawals. The increase in more intense rain events could lead to more runoff and, depending on the land use type, more pollution discharged directly into surface waters.

Trout streams, wetlands, and native forests may be the most impacted by climate change, both by more intense storms and due to lower stream flows and groundwater levels. Summer droughts could further reduce critical streamflow levels already being lowered by irrigation and other groundwater uses. Lower groundwater levels can impact all habitats that rely on groundwater-fed systems. Additional stress to native species could weaken natural defenses against invasive species and other pests.

3 How to Use This Document

This report is intended for audiences that do not have expertise and background in law and policy, hydrology, geology, chemistry, modeling and engineering, and ecology. The findings from each scientific analysis, and law and policy review are synthesized in this document. The conclusions and recommendations section is based on the scientific and modeling analyses. The summaries are listed in order below:

Background (Chapter 0)

- A Brief History of White Earth Nation

Regulatory and Policy Background (Chapter 6)

- Water Laws in the United States
- Federal Indian Law
- Federal Environmental Regulations
- Minnesota Water Laws

Hydrology and Ecology Characterization (Chapter 0)

- A Review of Ecological Conditions
- Understanding Where Wild Rice Grows
- Field Study and Trend Analyses

Surface Water and Groundwater Modeling (Chapter 0)

- Assessing the Impacts of Land Use Conversion in the Straight River Watershed using SWAT+
- Groundwater Nitrate Source Apportionment in the Pineland Sand Region Using Bayesian Modeling
- Using a Groundwater Flow Model to Explore Objective-Based Scenarios in the Pineland Sands Aquifer Region

Conclusions and Recommendations (based on Chapters 7 and 8 are in Chapter 0)

References (Chapter 10)

Appendices (Chapter 11)

Appended to this main report are the full report versions of each of the summaries referenced above, which are written in more technical and discipline-specific language.

Report A: White Earth Nation History

Report B: Water Laws in the U.S.

Report C: Federal Indian Law Overview

Report D: Federal Environmental Regulations in Indian County

Report E: An Overview of Water Laws in Minnesota

Report F: Characterizing Rooting Zone Chemistry in Wild Rice Lakes

Report G: Pineland Sands: Field Study and Trend Analysis

Report H: From Forest to Farmland in the Straight River Watershed: What has changed since 1850?

Report I: Groundwater Nitrate Source Apportionment in the Pineland Sands Region Using Bayesian Modeling

Report J: Using a Groundwater Flow Model to Explore Objective-Based Scenarios in the Pineland Sands Aquifer Region

4 Study Hypotheses and Objectives

The following research hypotheses have guided the Pineland Sands project. The background and contextual information presented after each hypothesis provide justification and support, drawing on a review of relevant peer-reviewed literature. Objectives correlated to the hypotheses are presented below.

Hypotheses

Nitrate Contamination and Agriculture

The study team hypothesizes that elevated nitrate concentrations in both groundwater and surface water are primarily due to intensive agricultural practices and expanded irrigation.

In recent decades, land use in the Pinelands Sands region of north central Minnesota has changed dramatically, driven by the expansion of agriculture and irrigation. Increased irrigation accelerates nutrient leaching and runoff, which can push the nitrate levels in water beyond safe thresholds (Alva, Paramasivan, Fares, Sajwan, & Delgado, 2006). In fact, the City of Park Rapids was forced to drill a deeper municipal well due to high nitrate contamination (City of Park Rapids, 2018). Data sources and studies led by the Minnesota Department of Agriculture (MDA), the Minnesota Department of Health (MDH), MN DNR, and Minnesota Pollution Control Agency (MPCA) also confirm the presence of nitrate pollution in the area (Stroom, 2024) (MDA, 2018) (MN DNR, 2017a) (MDH, 2025).

Nutrient and Biogeochemical Changes

The study team anticipates that the excess nutrients in surface waters in the project area have resulted in a reduction in sensitive species like fish and macroinvertebrates; high nitrate levels show a connection to impaired biota (Camargo, Alonso, & Salamanca, 2005).

Biogeochemical changes have led to decreases in suitable habitat for microorganisms such as bacteria, fungi, protozoa, archaea, invertebrates, and even some fish (Mugani, Messana, & Di Lorenzo, 2015).

Excess nutrients, such as nitrate and phosphate, can lead to excessive algal growth (eutrophication). Decomposition of overabundant algae, in turn, can alter the chemical balance (oxygen availability) in the waters and sediments of aquatic environments. This balance—called “redox conditions”—affects how microbes function, how nutrients move through these systems (Patrick & DeLaune, 1972), and which animals and plants can live there. One place these changes are happening is in "hyporheic zones," which are shallow areas beneath and around rivers, wetlands, and lakes where surface water and groundwater mix (Smolders, Lucassen, Bobbink, Roelofs, & Lamers, 2010) (Cardenas, 2015). Hyporheic zones are vulnerable to pollution, agricultural impacts, dams, groundwater overuse, and climate change impacts (Mugani, Messana, & Di Lorenzo, 2015).

Impacts on Wild Rice

The study team also hypothesizes that wild rice will be absent in waterbodies with dissolved sulfide concentrations greater than 120 micrograms per liter ($\mu g\ L^{-1}$) in the rooting zone of wild rice (MPCA, 2017). This level was found to be harmful to wild rice in both field (Myrbo, et al., 2017) and experimental studies (Pastor, et al., 2017) in Minnesota.

Hydrogen sulfide, a gas that is toxic to most animals and can disrupt plant uptake of nutrients, forms in watery sediment when naturally occurring bacteria convert sulfate from the surface water to sulfide under oxygen-free ("low redox") conditions (Myrbo, et al., 2017) (Pastor, et al., 2017). Other factors such as water levels, shoreline development, and interactions with other species can also affect wild rice populations.

Changes in Hydrology from Agricultural Land Use

Agricultural land use has changed the hydrology and baseflow of streams and lakes in the project area. Expanding more intensive agricultural practices has the potential to reduce baseflow contributions to lakes and streams while increasing surface runoff. These baseflow changes may be contributing to higher water temperatures, lower dissolved oxygen levels, and other chemical changes. In the project study area, the groundwater and surface waters are interconnected (MN DNR, 2017a), and many of the streams rely on the inflow of groundwater through the streambed – called "gaining streams" (Winter, Harvey, Franke, & Alley, 1998). The Straight River is a designated GWMA and is under active management to ensure that long-term sustainable use of resources is balanced with the protection of ecosystems, water quality, and the rights of current and future users (MN DNR, 2017a).

Objectives

Evaluate Nitrate Sources

The study team evaluated nitrate trends in both surface water and groundwater using publicly available data from the MDH, MPCA, and MDA. To accomplish this, the project team applied a method outlined in Ramson et al. (2018), which uses nitrate concentration data along with other chemical indicators such as chloride, specific conductance, and land use mapping of the area to categorize the sources of contamination (Ransom, Bell, Barber, Kourakos, & Harter, 2018). This process used Bayesian analysis and Monte Carlo simulations to determine the most likely sources of nitrate, including row-crop agriculture, septic systems, and large-scale animal operations, such as feedlots.

Conduct Trend Analyses on Hydrologic and Chemical Data

The study team conducted trend analyses of both historical and newly collected hydrologic and chemical data to assess long-term changes in streamflow, temperature, nitrate concentration, and baseflow. The evidence from the MN DNR's Straight River GWMA study has suggested a decline in groundwater contributions to surface water, resulting in increased stream temperatures and increased nutrient loading (MN DNR, 2017a)

Develop and Apply Surface and Groundwater Models

The study team developed surface and groundwater models to evaluate how conditions from the pre-settlement period to the modern agricultural era have impacted surface runoff, evapotranspiration, nutrient runoff, and nutrient leaching. SWAT+, MODFLOW, and the Modular Three-Dimensional Multispecies Transport Model for Simulation (MT3DMS) were used to better understand how land use influences nitrate transport and hydrologic pathways.

The SWAT+ model was used to evaluate the impacts of land use change from the pre-settlement landscape to modern intensive agriculture on surface runoff, evapotranspiration, nutrient runoff, and nutrient leaching. SWAT+ outputs were used to drive the MODFLOW and MT3DMS models to analyze how land use affects nitrate transport and hydrologic pathways. MODFLOW and MT3DMS models were used to simulate groundwater flow and nutrient transport in the region's groundwater aquifers. Modeled scenarios tested current land-use conditions and alternative future scenarios reflecting changes in irrigation, agricultural intensity, and land cover.

Quantify Groundwater Chemistry in Hyporheic Zones

Using piezometer well nests along three stream sections, the study team determined the degree to which elevated nitrate and phosphate levels are altering the redox conditions and lowering dissolved oxygen in hyporheic zones.

Characterize the Chemistry of the Wild Rice Rooting Zone

Sulfate in surface water mixes and diffuses into sediment pore water (the rooting zone of wild rice) and can be converted into sulfide by naturally occurring bacteria, using organic matter in the reaction. High sulfide has been identified as harmful to wild rice in an extensive field survey in Minnesota (Myrbo, et al., 2017) and in experimental laboratory studies (Pastor, et al., 2017)). A level of 120 µg L^{-1} was shown by the Minnesota Pollution Control Agency (MPCA, 2017) to be the threshold above which harm to wild rice occurred. Pore water was collected to determine the levels of sulfide contamination in the rooting zone of lakes of interest. Surface water sulfate and sediment organic carbon and iron were also measured. These three factors were found to control the level of sulfide in pore waters (Pollman, et al., 2017). The results were categorized by the level of sulfide and sulfate pollution, sulfate sensitivity, and estimated population of wild rice. The purpose was to assess current sulfate and sulfide pollution, estimate the sensitivity of the lake to additional sulfate pollution, and determine whether there is a relationship between wild rice populations and pore-water sulfide concentrations.

5 Background

A Brief History of the White Earth Nation

Author: Dre Szabo, Land and Environmental Attorney, White Earth Nation

Introduction

The history of the WEN could easily fill several volumes of books. Below is a brief introduction to the traditional history of the people, a more modern review of federal and state policies' impacts on the WEN, and a brief recount of the resulting land use changes to the White Earth reservation.

Brief Notes on Historic Tradition

On the Name: Ojibwa (Ojibwe, Ojibway), Chippewa (Chippeway), Anishinaabe (Anishinabe)

The people of WEN refer to themselves as Anishinaabe, which means "the original people" (Tribe, 1989). The Anishinaabeg (plural form) are part of a larger group of people who speak languages that are similar and thought to be derived from one another called the Algic, or Algonquin, which is the largest and most widespread indigenous language families in North America (Warren, 1885). The term Ojibwe was given by the Europeans, however it is not clear exactly where the name originated (Tribe, 1989). It is thought that some Europeans struggled to say Ojibway and pronounced it Chippewa (Tribe, 1989). Chippewa is the name officially used by the federal government in treaties with the Anishinaabe of Minnesota (Warren, 1885).

Migration Story and Traditional Lifestyle

The Anishinaabeg migrated from the eastern shores of the Atlantic Ocean to "the place where food grows on the water" (referring to Manoomin or wild rice) by following a sacred cowrie shell, Megis (Tribe, 1989) (Warren, 1885). Below in Figure 6 is the migration route following the St. Lawrence River. The five original clans, A-waus-e, Bus-in-aus-e, Ah-ah-wauk, Noka, and Nonsone or Waub-ish-ash-e (Crane, Catfish, Loon, Bear, and Marten) were said to have made the journey (Meyer M. L., 1994) (Warren, 1885).

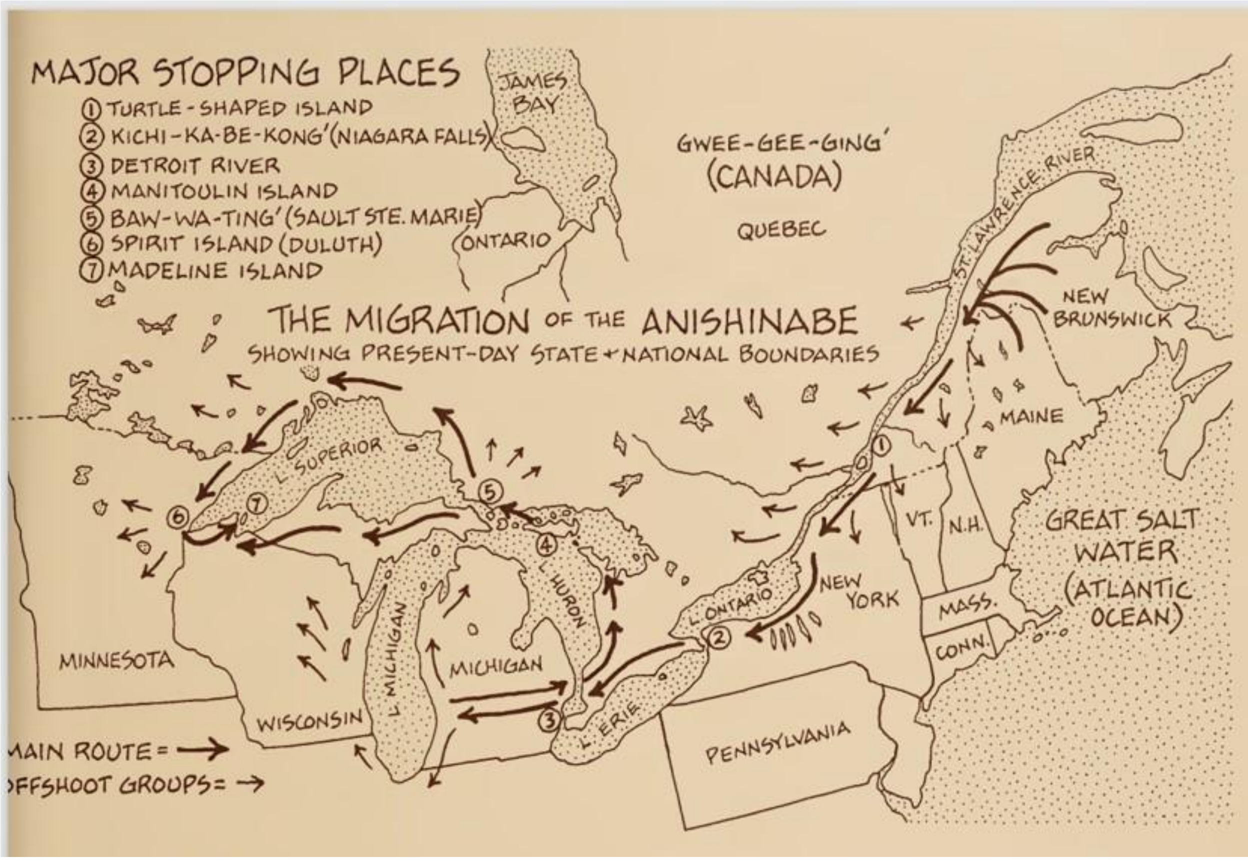

Figure 6: Migration of the Anishinaabe (Benton-Banai, 2010)

The Midéwiwin, Grand Medicine Society of Spiritual Advisors, recorded the migrations in birchbark scrolls (Meyer M. L., 1994). Modern Bands of Chippewa in Minnesota are thought to have derived from the area around what is now Sault Ste. Marie, at the outlet of Lake Superior between Michigan and Ontario (Meyer M. L., 1994). The Anishinaabe of Minnesota traditionally lived "exclusively in wooded country; their lands are covered with deep and interminable forests, abounding in beautiful lakes and murmuring streams, whose banks are edged with trees of the sweet maple, the useful birch, the tall pine, fir balsam, cedar, spruce, tamarac[k], poplar, oak, ash, elm, [and] basswood . . ." (Warren, 1885, pp. 39-40). They also traditionally traveled by lightweight birchbark canoes (Warren, 1885).

History Post European Contact

First Contact with Europeans: Fur Trading

Anishinaabe legends tell of a time when the Chippewa, Potawatomi, and Ottawa peoples were all one, but by the time Europeans first reached North America around 1500, the Chippewa were a separate group living north of Lake Huron and east end of Lake Superior (Tribe, 1989). By the 1600s, when the French fur traders came to the area, many of the Anishinaabe had moved south of Lake Superior and into upper Michigan, while others moved into the north woods of Wisconsin (Tribe, 1989). There are too many intricate details to capture in this brief history, but suffice it to say, the Anishinaabe initially interacted with the Europeans by trading furs and, to do so more effectively, warred with neighboring tribes, including the Sioux Tribes, the Fox Tribe, and the Iroquois from about the 1700s to the mid-1800s (Tribe, 1989) (Warren, 1885).

Governmental and Private Abuses, Changes in Land Use

Between the 1800s and the present day, many events have led to land-use changes, the extraction of trees, and other abuses directed at the WEN. The full reports of WEN history (Report A) and Federal Indian Law (Report C) describe the events and laws in more detail. Many Tribal nations were driven off their land during this time. For the WEN, this was driven by the timber industry's interests in logging and access to virgin timber (Weil, 1989). Many laws, such as the Clapp Rider of 1906, removed tribal status from individuals with "mixed blood." This paved the way for land to be easily sold to private interests. In fact, the Linnen Moorhead investigation of 1909 found in less than three years after the Clapp Act was passed 90% of land allotted to full bloods was mortgaged or sold and 80% of the reservation land passed into private ownership (Meyer M. L., 1994). The investigation concluded that the Indians were swindled out of the majority of their lands (Meyer M. L., 1994).

Conclusion

The ancestors of the people of the White Earth Nation migrated from the East Coast to the Midwest following a sacred journey intimately linked to the discovery of Manoomin. The WEN consists of several Anishinaabe groups who settled on the White Earth reservation during the late 19th and early 20th centuries. Much of the land base that was set aside for the Anishinaabeg of White Earth was fraudulently stolen through a variety of illegal schemes and abuses. An attempt to rectify past wrongs was made through the White Earth Lands Settlement Act, but vestiges of the past continue to influence land use on the reservation today. Many of the reservation's vast forests have now been cleared and converted to agricultural lands, which overlap with the 1855 Ceded Territory.

6 Regulatory and Policy Summaries

The summaries of policies and laws in this section provide background on waters laws in the U.S. and how they govern the use of surface and groundwater sources, the evolution of U.S. federal Indian law as a series of shifting policy eras, an overview of federal environmental laws that shape and support environmental programs led by Tribal nations, and an overview of water laws in Minnesota that are relevant to the project. The sections are organized as follows:

1. Water Laws in the United States
2. Federal Indian Law Overview
3. Federal Environmental Regulations in Relation to Indian Country
4. Minnesota Water Laws

Water Laws in the United States

Author: Dre Szabo, Land and Environmental Attorney, White Earth Nation

Water law in the U.S. developed largely in the 19th century (Klein, 2022), well before modern hydrologic science and technologies such as high-capacity groundwater pumping. As a result, U.S. water law is highly fragmented, primarily state-based, and often misaligned with scientific understanding of interconnected surface and groundwater systems. This section describes how groundwater and surface water laws vary around the U.S. and describes federal water laws and doctrines including the Public Trust Doctrine, Winters Doctrine, and the Constitution's Commerce Clause's federal navigability of waterways.

Surface Water Law

There are no federal laws that uniformly govern surface water. Instead, states rely on the historical doctrines: riparian, prior appropriation, and hybrid models. Each is described in more depth below.

Riparian Doctrine

The Riparian Doctrine dominates in the eastern U.S., the area that was first settled by Europeans. This doctrine evolved by tying water rights to landownership adjacent to a waterbody (Klein, 1995). Users may make "reasonable use" of surface water, so long as it does not unreasonably interfere with downstream users (NALC, n.d.) (Klein, 1995). Most riparian states now use regulated riparianism, requiring permits for significant or consumptive uses and allowing pro-rata reductions during shortages (Klein, Cheever, & Birdsong, 2013). Minnesota follows the riparian doctrine for surface waters.

Prior Appropriation Doctrine

The Prior Appropriation Doctrine dominates in the western U.S., a region characterized by arid conditions and a less abundant water supply than the eastern U.S. The principal is based on "first in time, first in right," which prioritizes water rights by seniority and the concept of beneficial use (e.g., agricultural, municipal, and industrial). In times of shortages, the most senior water rights holders are able to satisfy their water rights before more junior water rights holders. The doctrine is thought to have originated in the 19th century, when miners settled in the western U.S. and diverted water for mining-related needs. Because miners and other irrigators needed to divert water off tracts, often outside of watersheds, to their mines, the use of water in the arid West often resulted in no water left within streams (Klein, 1995). Environmental and instream uses have increasingly been recognized as beneficial uses since the late 20th century (Klein, 1995) (Wilkinson, 1985).

Hybrid Systems

Some states (e.g., California and Oklahoma) combine elements of riparian and prior appropriation systems (NALC, n.d.). Each state that uses a hybrid system is different, but they all share elements of each doctrine.

Groundwater Law

Like surface water, groundwater is regulated almost entirely by states, with no comprehensive federal framework. Groundwater law has lagged behind science and surface water regulation, despite groundwater supplying a substantial share of U.S. drinking water and irrigation (Owen, 2021). In Minnesota, groundwater supplies 75% of all the states' drinking water and 90% of all agricultural irrigation (MN DNR, n.d.-c).

There are five groundwater doctrines by which states govern groundwater: absolute ownership, correlative rights, prior appropriation, reasonable use, and the Restatement (Second) of Torts.

Absolute Ownership (Rule of Capture)

The absolute ownership is the oldest form of regulation derived from English Rule and common law, in which landowners may withdraw unlimited amounts of groundwater underlying the surface land (Reid, Kplinger, Miller, Eldstein, & Oiumet, 2021). Because the right to use groundwater is connected to the surface land, a landowner may withdraw an unlimited amount of ground water for any purpose (Mayer & Jennings, 2022). There is little protection for neighboring landowners. Texas largely retains this doctrine in traditional form (Reid, Kplinger, Miller, Eldstein, & Oiumet, 2021).

Correlative Rights

The right to use groundwater is based on land ownership above a basin or aquifer (Reid et al., 2021). With correlative rights each landowner is limited during times of shortage to a reasonable share, or fair and just proportion, of the underlying supply of the water (Mayer & Jennings, 2022) (Reid, Kplinger, Miller, Eldstein, & Oiumet, 2021). Minnesota follows the correlative rights doctrine.

Prior Appropriation

The Prior Appropriation Doctrine for groundwater mirrors that for surface water: senior users have priority. Prior appropriation creates a property right to use the water, but generally, the groundwater belongs to the state (Reid, Kplinger, Miller, Eldstein, & Oiumet, 2021).

Reasonable Use (American Rule)

The Reasonable Use is the most common form of groundwater doctrine in the United States, predominantly in eastern U.S. (Reid, Kplinger, Miller, Eldstein, & Oiumet, 2021). Groundwater must be used on the overlying land and put to a reasonable use (Mayer & Jennings, 2022). Most states modified the rule of capture by adding reasonable use to resolve conflicts between neighbors, hence it is referred to as the “American Rule” (Reid, Kplinger, Miller, Eldstein, & Oiumet, 2021).

Restatement (Second) of Torts

The Restatement (Second) of Torts rule combines the English rule and the American Rule with the aim of modernizing groundwater law. The Restatement of Torts is a balancing test focusing on unreasonable harm rather than land location. Liability arises if pumping causes unreasonable impacts on others or connected surface waters (Mayer & Jennings, 2022) (Reid, Kplinger, Miller, Eldstein, & Oiumet, 2021).

Key Problems in Groundwater Regulation

The key problems in groundwater regulation and governance in the U.S. are as follows:

- Legal doctrines predate scientific understanding of aquifers (Glennon, 2002) (Klein, 2022).
- States regulate groundwater inconsistently, even when sharing aquifers (Glennon, 2002).
- Law often fails to recognize hydrologic connections between groundwater and surface water (Glennon, 2002).
- Over-allocation, depletion, water quality decline, land subsidence, and economic harm are common risks (Reid, Kplinger, Miller, Eldstein, & Oiumet, 2021).
- Groundwater is often treated as a common-pool resource, raising concerns about which individuals/groups are maximizing the use of water, leaving less for others (Glennon, 2002).

Federal, Public, and Tribal Water Rights

Although states control most water regulation, federal law plays an important role in pollution regulation which predominantly relies on federal navigation as well as the public trust doctrine. To a certain extent, federal law also addresses water quantity and quality through reserved water rights and the public trust doctrine.

Navigation

The Commerce Clause of the U.S. Constitution has been interpreted to include Congress's authority to regulate navigable waters for trade and interstate commerce, such as the Mississippi River. The U.S. Army Corps of Engineers is responsible for maintaining navigable waterways on the Mississippi River through with lock and dam management, dredging, and sediment management. The EPA regulates navigable waters for pollution control. Navigable waters designated as the "Waters of the United States" (WOTUS) for pollution control and federal regulation are subject to numerous revisions, generally at least once with each change in federal administration.

Public Trust Doctrine

The concept of the public trust doctrine is a legal fiction where common and shared natural resources such as the air, water, and wildlife are public property held in trust by the state for the public's benefit (McGlothlin & Slater, 2013). The Illinois Central Railroad v. Illinois (1892) is considered a pivotal case in the interpretation of the public trust doctrine, in which the railroad company and the state had a conflict over title to a portion of the bed of Lake Michigan (Mayer & Jennings, 2022). The Court held that Illinois did not have the authority to sell the title to the bed of Lake Michigan because the Court held the title to such submerged lands under the navigable waterway is held in trust for the benefit of the public, so that they may fish, engage in commerce, or otherwise use the lake without interference from private parties (Mayer & Jennings, 2022).

Each state has had different successes using the public doctrine to protect ecological and recreational values. The public trust doctrine related to groundwater governance has one notable legal precedent, Clean Wisconsin Inc. v. Wisconsin Department of Natural Resources (DNR), 961 N.W. 2d 611, which held that the state's DNR did have the authority to impose conditions on high-capacity wells under the public trust doctrine (Mayer & Jennings, 2022).

Federally Reserved Water Rights (Winters Doctrine)

Reserved water rights is extremely important for Indian Country. When the federal government reserves land, it implicitly reserves sufficient water to fulfill the reservation's purpose (Klein, 2022). This conclusion was made in the seminal 1908 Supreme Court case Winters v US, 207 U.S. 564, 577.

Indian water rights are:

- Not lost by non-use (Goodrum, 2019).
- Not required to be permitted by states (Goodrum, 2019).
- Senior to later state-based water rights (Goodrum, 2019).
- Applies to surface water and, increasingly, to groundwater.

Conclusion

U.S. water law is a complex, historically rooted patchwork dominated by state doctrines that often fail to reflect modern science, environmental realities, and rising demand. Tensions persist between state authority, federal oversight, environmental protection, and Tribal water rights. As climate change, population growth, and groundwater depletion intensify, the disconnect between law and hydrology remains one of the central challenges facing U.S. water governance.

Federal Indian Law Overview

Author: Dre Szabo, Land and Environmental Attorney, White Earth Nation

Introduction

The document traces the evolution of U.S. federal Indian[3] law as a series of shifting policy eras, ranging from recognition of tribal sovereignty to forced assimilation and later self-determination. A central theme is the "pendulum swing" between treating tribes as sovereign nations and attempting to assimilate or eliminate them as a distinct political entity. This summary will cover the four main, generally accepted historical eras (from pre-1776 to the present).

- Doctrine of Discovery, Treaty Making (pre-1776 to 1826)
- Marshall Trilogy and Indian Removal Policy (1826 to 1887)
- Allotment, Assimilation (1887 to 1934)
- Reorganization, and Termination Eras (1934 to 1960)
- Self-Determination Era (1960 to present)

Doctrine of Discovery, Treaty Making (pre-1776 to 1826)

Tribal nations have self-governed since time immemorial and for centuries before European contact. In fact, during European "discovery," the conquerors discovered a vast network of trade and a variety of sophisticated governments (Strommer & Osborne, 2015). European powers and later the U.S. relied on the Doctrine of Discovery, which was used to justify and assert rights over lands (Getches, Wilskinson, Williams, & Fletcher, 2011). The British Crown and European settlers viewed relations with Indians differently, and ultimately, the Crown served as a buffer between Indians and European settlers. This dynamic changed following the Revolutionary War. The newly formed federal government set up a series of Acts for regulating the Indians and non-Indians by predominantly ensuring that the federal government, not individuals, interacted with the Tribal nations (Canby, 2015). These Acts became known as the Trade and Intercourse Acts (Canby, 2015)

Marshall Trilogy and Indian Removal Policy (1826 to 1887)

This period is associated with increasing tensions between Indian and non-Indians. Three

[3] Indian is a federally defined term generally referring to persons enrolled in federal recognized tribes (Hoss, 2025).

Supreme Court cases summarized the tone for this time period which had cascading effects for Indian and federal law for at least 150 years (Canby, 2015): Johnson v. McIntosh, 21 U.S. (8 Wheat.) 543 (1823); The Cherokee Nation v. Georgia, 30 U.S. (5 Pet.) 1 (1831); and Worcester v. Georgia, 31 U.S. 515 (1832). Johnson v. McIntosh concluded that Indians' right of occupancy could be extinguished only by the U.S., thereby giving the federal government ultimate authority (Canby, 2015).

Both Cherokee Nation v. Georgia and Worcester v. Georgia arose because the State of Georgia wanted to enact its own laws around Indian policy and take Cherokees' land for non-Indians (Getches, Wilskinson, Williams, & Fletcher, 2011) (Society, n.d.). Chief Justice Marshall determined that the Cherokee were a "state" with a "distinct political society separated from others capable of managing its own affairs and governing itself" but they could not be considered a "foreign state"; rather, they were a "domestic dependent nation" where the relationship between that of the U.S. and the tribe was like that of a "ward and his guardian" (Canby, 2015). Just one year after Cherokee Nation v. Georgia came Worcester v. Georgia case. The case was extremely influential for the proposition that the federal government, not state government, has plenary authority over the affairs of Indians and state law has no authority within Indian territory. This would later be modified with a series of federal cases, laws, and policies that paved the way for state jurisdiction within Indian Country (Canby, 2015).

Despite the recognition of tribal rights and sovereignty, there was still a pervasive push for Indian removal and their territories given to non-Indians to make better use of the land for "civilized activities." Ultimately, the federal government adopted a policy of removal whereby tribes out East were "voluntarily" removed, often through force and coercion, to relocate out West to reservations (Canby, 2015). To this end, Congress passed the Indian Removal Act in 1830, and it was clear if Tribes failed to relocate, they would lose federal protections and be subject to state laws (Strommer & Osborne, 2015).

Allotment, Assimilation (1887 to 1934)

The period between 1887 and 1934 can be characterized as an effort to dissolve tribal identity and integrate Indians into U.S. society. The results of this were massive land losses, cultural disruption, and fragmented land ownership. In 1892, Army Officer Richard Pratt promoted the idea of assimilation in an infamous exhortation that it was necessary to "kill the Indian" to "save the man." In fact, Pratt opened the first boarding school in 1879 in Pennsylvania to remove Native children from their culture and assimilate them into the dominant culture (Carlisle Indian School

Digital Resource Center, 1892).

The Dawes Act, passed in 1887, resulted in breaking up tribal parcels to individual Indian owners (Strommer & Osborne, 2015). The approach was intended to speed up the civilization of the Indians by breaking up the culture of communal tribal life and turning them into civilized, individualistic farmers (Strommer & Osborne, 2015). There was no provision in the Dawes Act for consent from either Tribes or individual tribal members (Canby, 2015). In the case of the WEN, much of the land surplus was sold to timber companies and non-Indian farmers (Tribe, 1989). The result of the Dawes Act was that 138 million acres of Indian lands in 1887 had shrunk to 48 million acres by 1934 (Strommer & Osborne, 2015).

Reorganization, and Termination Eras (1934 to 1960)

In 1928 a report documenting the shortcomings and failings of the allotment and assimilation policies of the federal government, called the Meriam Report, led to a dramatic change in federal policy known as the Indian Reorganization Act (IRA) of 1934 (Canby, 2015). The IRA represented a change from the previous era of Tribal policy and sought to protect Tribal land bases and design efficient modern systems of self-government (Canby, 2015). However, the IRA maintained federal government supervision and imposition on tribal governance. The WEN is a Band of the Minnesota Chippewa Tribe (MCT), which remains organized under an IRA constitution.

In the 1950s, federal Indian policy shifted again. The Termination Era benevolently[4] sought to "free" the Indians from federal oversight and make them regular American citizens, so it set out by authorizing specific statutes that terminated the special "ward" status of Tribes (Canby, 2015). Over one hundred Tribal Nations had their status terminated (Strommer & Osborne, 2015). The effects of this policy were destructive to Tribal Nations and included loss of land, relocation to larger cities, experiencing trauma from the relocation, and suffering from poverty (Canby, 2015) (Strommer & Osborne, 2015).

Arguably the most disastrous legislation of the Termination Era is what is known as Public Law 280, 67 Stat. 588 (1953), as amended 18 U.S.C.A. §§ 1162, 25 U.S.C.A. §§ 1321-26, 28 U.S.C.A. § 1360 (1953). Public Law 280 mandatorily gave jurisdiction to five states over the criminal affairs of Tribal Nations in California, Nebraska, Minnesota (except for Red Lake Nation), Oregon (except for Warm Springs Reservation), and Wisconsin (Canby, 2015). Tribal consent was not a

[4] The term was used to describe how the federal government perceived its own actions.

requirement, and often not sought by the states (Canby, 2015). Congress provided no funding to states to assume the responsibility of exercising jurisdiction over Indians, and states were still not allowed to tax Indian lands. Today, Minnesota is still a Public Law 280 mandatory state, but the Red Lake Nation remains exempt, and the Bois Forte Band of the MCT retroceded in 1973 (Mullen, 2016). WEN now has concurrent criminal jurisdiction thanks to the Tribal Law and Order Act of 2010 (Mullen, 2016).

Self-Determination Era (1960 to Present)

By the 1960s, a growing recognition of tribal rights and self-determination was forming. The American Indian Movement (AIM) was founded in 1968 in Minneapolis and focused on tribal sovereignty and self-governance (Greer, n.d.). AIM quickly gained national attention as a grassroots civil activist group and was successful at garnering the attention of the federal government with respect to certain tribal land rights and support for Indian education (Greer, n.d.).

In 1968, Congress passed the Indian Civil Rights Act which guaranteed many of the same protections in the Bill of Rights to Indian tribes and expressed support for tribal self-determination (Strommer & Osborne, 2015). President Nixon urged the abolition of Termination Era policies and sought to make self-determination the official federal Indian policy that eventually led to the passage of the Indian Self-Determination Education and Assistance Act (ISDEAA) of 1975 (Strommer & Osborne, 2015). ISDEAA authorized the Secretary of the Interior to negotiate contracts where tribes could assume the responsibilities of the federal government in key areas directly related to their well-being such as health care and education (Canby, 2015). Today, tribes can also negotiate for the assumption of responsibilities beyond federal programs designed specifically for tribes. The WEN has been considered a self-determination tribe since at least the late 1980s and is currently negotiating two separate contracts under ISDEAA to assume responsibilities for Department of the Interior programs.

In 1975, Congress established an American Indian Policy Review Commission, which included tribal representation to review federal Indian policy and consider methods to strengthen tribal government (Canby, 2015). Several important Acts affirming tribal self-governance and self-determination followed. In 1983, President Reagan reaffirmed the federal Indian policy of strengthening tribal governments with the additional goal of reducing their dependence on the federal government (Canby, 2015). Since President Reagan, every president except George H.W. Bush and Donald Trump has reaffirmed the federal Indian policy of self-determination and self-governance through executive order (Canby, 2015) (National Archives, n.d.). Today, virtually all federal agencies have internal policies affirming tribal self-determination and self-governance and processes for tribal consultation prior to taking federal action. State approaches are less uniform and range from support for tribal self-governance to tension with it. Minnesota has a statute affirming tribal self-governance and requiring all State agencies to engage in tribal consultation prior to taking actions that affect Tribal nations within the State, Minn. Stat. §10.65 (2025).

Conclusion

Federal Indian law is shaped by historical cycles of acknowledgment and suppression of tribal sovereignty. While modern policy favors self-determination, the enduring effects of earlier policies—especially land loss, jurisdictional complexity, and federal oversight—continue to influence Tribal nations today.

Federal Environmental Regulations in Relation to Indian Country

Authors: Benjamin Schiltz, Emma Henefield, and Lauren Salvato, Young Environmental Consulting Group

Introduction

This report section summarizes the environmental regulations that pertain to the WEN and federally recognized tribes across the U.S. The list of laws and topics below was chosen based on their relevance to the project.

- EPA Indian Policy
- Clean Air Act
- Safe Drinking Water Act
- Federal Insecticide, Fungicide, and Rodenticide Act
- National Environmental Policy Act
- Coordination and Consultation with White Earth Nation

EPA Indian Policy

The U.S. EPA was created in 1970 to protect human health and the environment (U.S. EPA, 2025a). Over time, federal policy evolved to recognize Tribal governments as sovereign entities. While there were some previous efforts to acknowledge tribes as sovereign nations, the EPA Indian Policy was one of the first policies to formalize government-to-government relations between tribes and the federal government (Reagan, 1983). The Indian Policy includes the following principles:

- Establishes a government-to-government relationship between EPA and federally recognized tribes.
- Affirms that tribes can
    - Set environmental priorities.
    - Administer programs similar to states ("Treatment as a State" or TAS).
- EPA provides funding, technical assistance, and program support as a way for tribes to build autonomy and develop their environmental programs.

Consultation and Coordination with Tribal Governments

The EPA Policy on Consultation with Indian Tribes establishes how the EPA conducts government-to-government consultation with federally recognized Tribal governments when EPA actions or decisions may affect Tribes. The policy implements Executive Order 13175 and the EPA Indian Policy and applies agency-wide as standard EPA practice (U.S. EPA, 2023). The Consultation Policy affirms three main statements:

- EPA commits to meaningful and timely consultation with Tribes before final decisions are made on actions that may affect them.
- Consultation is a two-way process, intended to ensure Tribal input is considered—not merely noticed—by EPA decision-makers.
- The policy provides national guidelines to promote consistency while allowing flexibility to tailor consultation to each Tribe and circumstances.

Tribal Partnership Groups

Supported by the Northern Arizona University Institute for Tribal Environmental Professionals, tribal partnership groups were developed to provide two-way communication between federal agencies and representatives of Tribal nations (NAU ITEP, n.d.). Additionally, these tribal partnership groups support one another in building capacity and sharing information on a variety of natural resource topics and concerns affecting those resources. There are many tribal partnership groups that serve different focuses. The Tribal Pesticide Program Council was formed to raise awareness about pesticide issues. The National Tribal Air Association advances air quality management, policies, and programs that serve the interests of American Indians and Alaska Natives (NAU ITEP, n.d.). The National Tribal Waters Council is a technical and scientific body that evaluates research and information for decision-making and raises water-related issues that represent the Indian and Alaska Native tribal members.

Clean Air Act

The Clean Air Act (CAA) was enacted in 1970 to outline EPA's responsibilities to protect and improve air quality and the stratospheric ozone layer (U.S. EPA, 2026a). However, it was not until 1990 that the tribal authority provision was added to the CAA amendments (U.S. EPA, 1998). The Tribal Authority Rule authorized eligible tribes to implement their own tribal air programs and outlined that tribes can achieve TAS. A tribe can opt to implement all or portions of an air quality program.

The largest air-quality threat to Tribal nations in the upper Midwest comes from wildfires across Minnesota and Canada. The short-term impacts of wildfire exposure are the onset of asthma, increased risk of asthmatic episodes and chronic obstructive pulmonary disease, and increased risk of respiratory infections. The long-term effects are less well known (Berman & Duffy, 2025).

Tribes are disproportionately impacted by air quality because of the U.S. government's systematic efforts to disrupt Indigenous communities (NTAA & MCAF, 2021). Tribal nations facing chronic poverty, poor health care because of reduced access, and substandard housing are at greater risk of air pollution and human health impacts.

Clean Water Act

The Clean Water Act (CWA) of 1972 was established as the primary objective of restoring and maintaining the chemical, physical, and biological integrity of the Nation's waters (U.S. EPA, 2026b). The EPA is the federal agency delegated to carry out the CWA by regulating point-source discharges into those waters and setting water quality standards for surface waters.

The 1987 amendments to the CWA authorized EPA to focus on nonpoint source (NPS) and provide leadership, guidance, and funding to eligible state and tribal entities (U.S. EPA, 2025b). States and tribes are allocated funding under Section 319(h) (U.S. EPA, 2025c).

Tribes were not included in the original 1972 statute but were later added when the CWA was amended in 1987 (U.S. EPA, 2016). In that rule, the EPA, for the first time, addressed Indian country implementation directly by retaining federal authority over Indian facilities rather than delegating it to states. The 1972 CWA amendments and other rules laid the path for eligible tribals to gain authority to administer their own CWA programs as a TAS.

Safe Drinking Water Act

The Safe Drinking Water Act (SDWA) was enacted by Congress in 1974 to protect public health by establishing national standards for drinking water (U.S. EPA, 2004). The SDWA sets legally enforceable limits for categories of drinking water (or MCLs) such as microorganisms, disinfectants, disinfection byproducts, inorganic chemicals, organic chemicals, and radionuclides (U.S. EPA, 2025d). In 1998, the EPA finalized a rule under the SDWA clarifying how federally recognized Tribal Nations may assume responsibility for certain drinking water programs in Indian Country (U.S. EPA, 1998). This rule supports Tribal participation in implementing SDWA requirements, subject to EPA approval, while the EPA retains implementation authority where Tribes have not received program approval. The term primacy refers to the delegated authority to implement the SDWA (U.S. EPA, 2026c). Tribes must have TAS before they can assume primacy under the SDWA.

Tribes may also have primacy over the Underground Injection Control (UIC) program, a permitting program that regulates five classes of UICs to protect groundwater drinking sources (U.S. EPA, 2026g). Having primacy status as a tribal entity, comes with the authority to issue enforcement for requirements that are not met under the SDWA UIC program (U.S. EPA, 2025e).

Federal Insecticide, Fungicide, and Rodenticide Act

The Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) was enacted in 1947 to establish requirements for pesticide labeling and registration within the U.S. Department of Agriculture (USDA). FIFRA has been amended many times to govern the complete life of pesticides in the U.S., including “registration, distribution, sale, and use of pesticides” and the disposal (U.S. EPA, 2026e). The 1972 amendments completely replaced the 1947 version and included the shift in oversight of FIFRA from USDA to EPA (Esworthy & Yen, 2012).

Under FIFRA, the EPA is responsible for implementing and enforcing pesticide regulations in Indian Country and works collaboratively with Tribal Nations to support the development of Tribal pesticide programs (U.S. EPA, 2026f). The EPA can work with Tribal nations as partners through cooperative agreements, training, technical assistance, funding opportunities, and partnership mechanisms to strengthen pesticide program capacity and compliance. Tribal governments may exercise regulatory and enforcement authority under federally recognized sovereignty, subject to EPA oversight.

National Environmental Policy Act

The National Environmental Policy Act (NEPA) was signed into law in 1970, alongside a wave of environmental laws enacted around the same time (CAA, CWA). NEPA is a policy framework for protecting the environment before federal actions that could affect it are undertaken (U.S. EPA, 2026d). The enactment of NEPA created the Council on Environmental Quality (CEQ) to serve as the caretaker of NEPA policy and implementation guidance and required the preparation of environmental impact statements and environmental assessments, as well as the consideration of project alternatives. NEPA applies to all federal agencies and guarantees public access to information, improves planning, and helps prevent environmental harm before it occurs.

The Bureau of Indian Affairs (BIA) is responsible for ensuring NEPA compliance for federal actions impacting tribal lands (U.S. BIA, n.d.). Tribal entities also have their own authorities to conduct

NEPA-style assessments (U.S. DOI, 2012). Often, activities affecting environmental lands require both BIA and tribal government approval, requiring coordination and cooperation among BIA and the tribal government.

NEPA is closely integrated with the National Historic Preservation Act (NHPA), particularly Section 106. Both processes are triggered by federal funding or approvals and are designed to ensure transparency and public involvement. Under Section 106, federal agencies must consider the effects of their actions on historic properties and provide the Advisory Council on Historic Preservation with an opportunity to comment (Advisory Council on Historic Preservation, n.d.). The process also requires consultation with State Historic Preservation Officers (SHPOs), Tribal Historic Preservation Officers, Indian Tribes, Native Hawaiian Organizations, Alaskan Native villages, local governments, and project applicants.

Conclusion

Federal environmental regulations in Indian Country form a complex but critically important framework for protecting human health, natural resources, and Tribal sovereignty. Tribes have progressively gained clearer pathways to exercise federal regulatory authority, participate in federal programs, and shape environmental decision-making in ways that reflect their unique cultural, ecological, and community priorities. While early federal statutes largely excluded Tribes, subsequent amendments, EPA policies, and executive actions have increasingly recognized Tribes as sovereign governments with roles comparable to states, provided they choose to assume those authorities. For the WEN, having its own environmental programs supports the Nation's regulatory responsibility and self-determination.

At the same time, federal funding mechanisms—such as the Indian General Assistance Program, Section 106 and 319 grants, Drinking Water State Revolving Loan Funds, and technical assistance programs—remain essential for building and sustaining Tribal capacity in environmental regulation. When presidential administrations change that do not prioritize water and natural resources, tribal governments often suffer from reduced funding and capacity building. The grants and capacity support are important for empowering Tribal nations to achieve TAS and grow their individual regulatory programs.

Minnesota Water Laws

Authors: Benjamin Schiltz and Emma Henefield, Allegra Johnson McKee, Rachel Kapsch, Young Environmental Consulting Group

Introduction and Purpose

This report summarizes the state and select local environmental regulations that pertain to the study area and to water protection in Minnesota generally. The laws and topics below were selected based on their strong relevance to this project.

History of Riparian Law in Minnesota

Minnesota follows a regulated riparian system, tying water use to land ownership with limits on unreasonable use (Klein, 1995). The state has modified traditional riparian doctrine through permitting and monitoring under Chapter 103G. Court decisions (e.g., White Bear Lake case) emphasize limits on cumulative groundwater pumping and protection of surface waters.

Minnesota Statute § 10.65

Minnesota Statute 10.65 recognizes the sovereign status of 11 federally recognized Tribal nations located in Minnesota and affirms their inherent rights to existence, self-governance, and self-determination (Minnesota Legislature, 2025a). This statute provides the basis for tribal liaisons within Minnesota state agencies, which seek to create meaningful coordination and consultation between state agency actions and Tribal nations.

Minnesota Statutes, Chapter 103G

Chapter 103G of Minnesota Statutes establishes the state's legal authority to regulate the "waters of the state," thus making it a crucial starting point for understanding water rights in the Pineland Sands region (Minnesota Legislature, 2025b).

Chapter 103G.221 through 103G.2375

The Wetland Conservation Act promotes a policy of no net loss of wetlands by requiring that drained or filled wetlands be replaced with wetlands of equal or greater public value (Minnesota Legislature, 2025b). It allows limited agricultural uses during dry periods, provided these uses do not permanently alter wetlands. The Act is implemented through Minnesota Rules Chapter 8420 and is administered locally by local government units (LGUs) with oversight from the Board of Water and Soil Resources (BWSR) (Minnesota Legislature, 2009).

Chapter 103G.271

Chapter 103G.271 requires permits for high-capacity wells where water use exceeds 10,000 gallons per day or 1,000,000 gallons per year (Minnesota Legislature, 2025c). Smaller withdrawals are generally not tracked, making it difficult to assess cumulative impacts on water resources. The statute includes protections for agricultural irrigation, which limit curtailment of water use during the growing season. It also requires that water appropriation permits be consistent with the Statewide Drought Plan.

Chapter 103G.287

This statute grants the MN DNR authority to approve or deny groundwater use permits and requires that appropriations avoid impacts to surface waters (Minnesota Legislature, 2025d). It allows the designation of GWMAs to address overuse concerns and includes a sustainability standard that requires the long-term protection of water resources and ecosystems. The Pineland Sands area overlaps with the Straight River GWMA.

Chapter 103G.291 through 103G.293

These statutes allow the Governor to declare a critical water deficiency during drought conditions and require public water suppliers to implement conservation measures that prioritize essential uses (Minnesota Legislature, 2025e). The statutes also direct the development of the Minnesota Statewide Drought Plan, which outlines staged response actions based on drought severity (MN DNR, 2009). Supporting tools such as the System-Wide Low-Flow Management Plan help manage river flows during drought by coordinating dam and hydropower operations (MN DNR, 2015). Water appropriation permits must be consistent with drought planning requirements.

Minnesota Statutes, Chapter 103H

Chapter 103H focuses on groundwater quality protection rather than regulating water quantity (Minnesota Legislature, 2025f). It emphasizes prevention through land-use planning and best management practices and coordinates efforts across the MN DNR, MPCA, and MDA. It is particularly relevant in areas such as the Pineland Sands where nitrate contamination is a concern.

Minnesota Statutes, Chapter 103I

This chapter regulates the construction, maintenance, and sealing of wells and establishes wellhead protection measures to prevent the contamination of drinking water sources

(Minnesota Legislature, 2025g). It is administered by the MDH and also allows for civil remedies in cases of well contamination.

Minnesota Statutes, Chapter 115

Chapter 115 gives the MPCA authority to establish water quality standards and prevent pollution.

Minnesota Administrative Rules, Chapter 7050 – Part 7050.0224

This rule establishes specific water quality standards to protect wild rice waters, identifying designated waters and setting limits such as sulfate concentrations (Minnesota Legislature, 2025h). These protections are important for maintaining the ecological integrity and cultural resources associated with wild rice.

Minnesota Statutes, Chapter 116D

Chapter 116D requires environmental review, such as Environmental Assessment Worksheets and Environmental Impact Statements, for projects that may have significant environmental impacts (Minnesota Legislature, 2025i). It is administered by the Environmental Quality Board, with implementation carried out by state or local agencies. The statute integrates environmental considerations into decision-making and has been applied in the Pineland Sands region to evaluate cumulative groundwater impacts.

Minnesota Statutes, Chapter 144

This chapter governs public water supply systems under the authority of the MDH (Minnesota Legislature, 2025j). It requires monitoring, compliance, and corrective actions to address contamination and ensures the protection of drinking water sources from pollutants.

Local Water Protection and Management

Local entities play an important role in implementing state water laws and planning efforts.

Soil and Water Conservation Districts

Soil and Water Conservation Districts provide technical assistance and promote voluntary conservation practices. They act as a bridge between state policy and local implementation by supporting landowners and encouraging conservation efforts.

Board of Water and Soil Resources

BWSR oversees local water management, wetland programs, and conservation funding across the state. It also supports watershed planning and restoration through a variety of statewide programs and initiatives, helping coordinate implementation of water resource protection at the local level. BWSR's One Watershed One Plan program addresses planning and restoration efforts at the local watershed level.

Conclusion

State and local regulations provide an important framework for protecting groundwater, surface water, wetlands, and drinking water resources. For the project area and study, these Rules and Statutes serve to conserve and sustain robust surface water and groundwater supplies and the region's overall ecological health. Together, these state and local regulations and the authorities granted therein work to address cumulative impacts, support restoration efforts, and underscore that protecting the Pineland Sands region will require coordinated, proactive management to ensure long-term sustainability of its natural and cultural resources.

7 Hydrology and Ecology Characterization

The section on hydrology and ecology characterization summarizes three distinct topics. The first is a review of existing analyses to understand the health of mussels, aquatic plants, and fish in the project area. The second was a study conducted to characterize the rooting zone of wild rice in 12 lakes within the study area. This was done by measuring water, porewater, and sediment parameters related to wild rice health. The final study evaluates isotopic and pesticide trends in the project area, as well as groundwater levels and their impacts from irrigation.

1. A Review of Ecological Conditions
2. Characterizing rooting zone chemistry in wild rice lakes
3. Field Study and Trend Analyses

A Review of Ecological Conditions

Author: Brenda DeZiel, Caddis Fish Consulting

This report summarizes previously completed studies in the project area to understand the condition of native mussels, fish, plant, and macroinvertebrate populations in the project area.

Background

The Pineland Sands project area is well known for high-quality lakes, rivers, trout streams and forests. These areas serve as a source of recreation, hunting, aesthetic, cultural, and spiritual enjoyment. In the project area and just outside is Itasca State Park, five State Forests, three Scientific and Natural Areas (SNAs), as well as multiple Aquatic Natural Areas and Wildlife Natural Areas (Figure 7). These protected areas are important for preserving the habitats of state- and federally listed species and species of rare or special concern, as well as for increasing habitat connectivity. The SNAs and other habitats have remnant stands of jack pine forests, second-growth hardwoods, and coniferous forests. Many breeding and migratory birds use these habitats (MN DNR, n.d.-d) (MN DNR, n.d.-e).

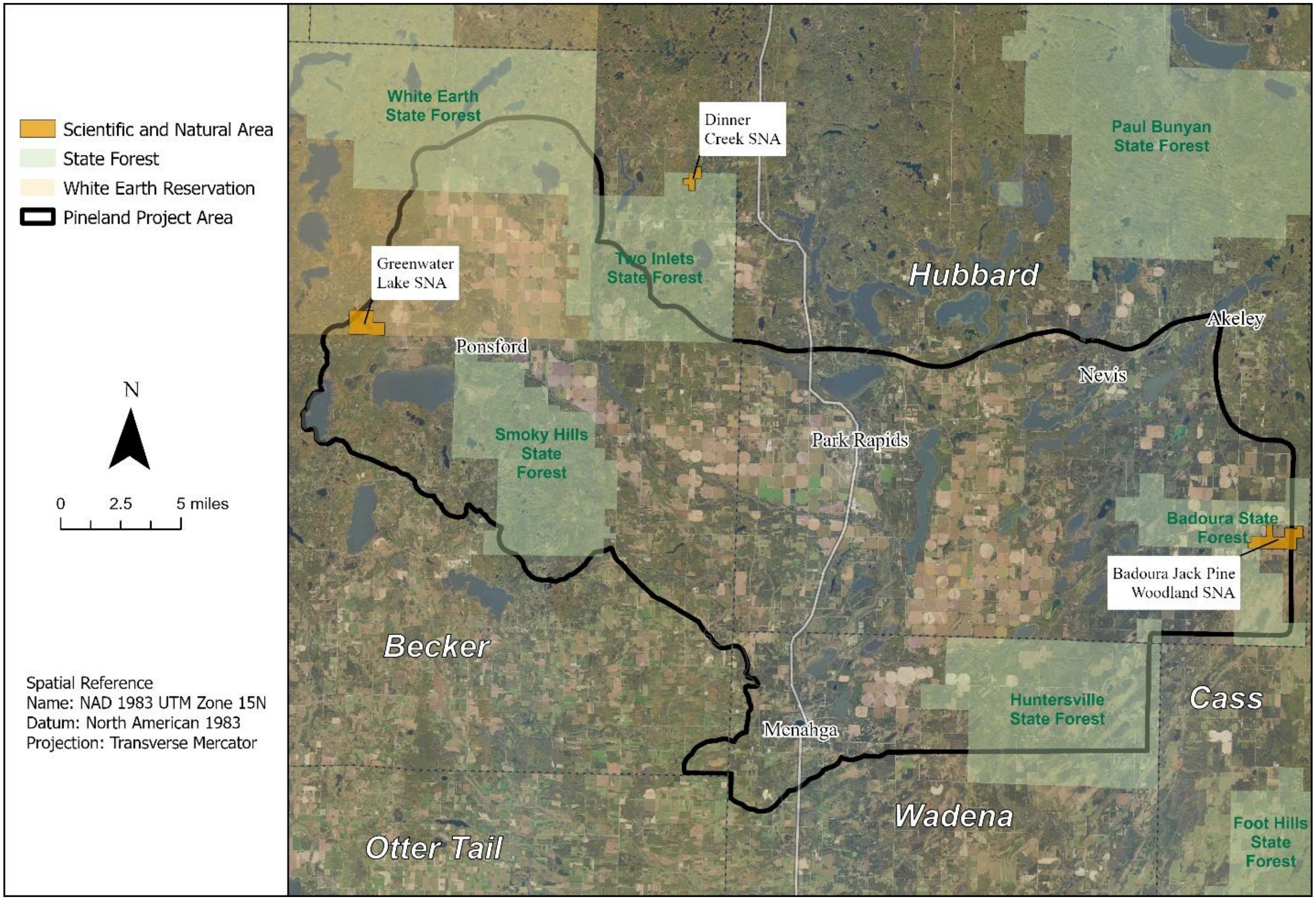

Figure 7: State protected areas

Biodiversity Significance

The Minnesota Biological Survey (MBS) collects and evaluates data on plants and animals across the state and evaluates the ecological condition of the landscape to support plant and animal communities. The MBS that overlaps with the project area identified and mapped areas of biodiversity significance based on the evaluation of rare species, native plants, and connectivity of the landscape (MN DNR, n.d.-f) (MN DNR, 2017b) (MN DNR, 2017b). The criteria are used to assess whether an area is of outstanding, high, moderate, or below significant levels. Figure 8 depicts a few outstanding areas in the Fish Hook River subwatershed in green. Some yellow or highly biodiverse sites were primarily located in the Fish Hook subwatershed but were also scattered across other subwatersheds. Most areas were rated as moderate or colored orange. Moderate sites may have some rare species within moderately disturbed native plant communities. Sites designated as “Below” or colored red could have rare species but are generally categorized as having fragmented landscapes that are developed and disturbed (MN DNR, 2017b). Areas dominated by cultivated crops were not surveyed for native plant communities, rare species, or connected landscapes and were assumed to be smaller and likely insignificant areas (MN DNR, n.d.-f).

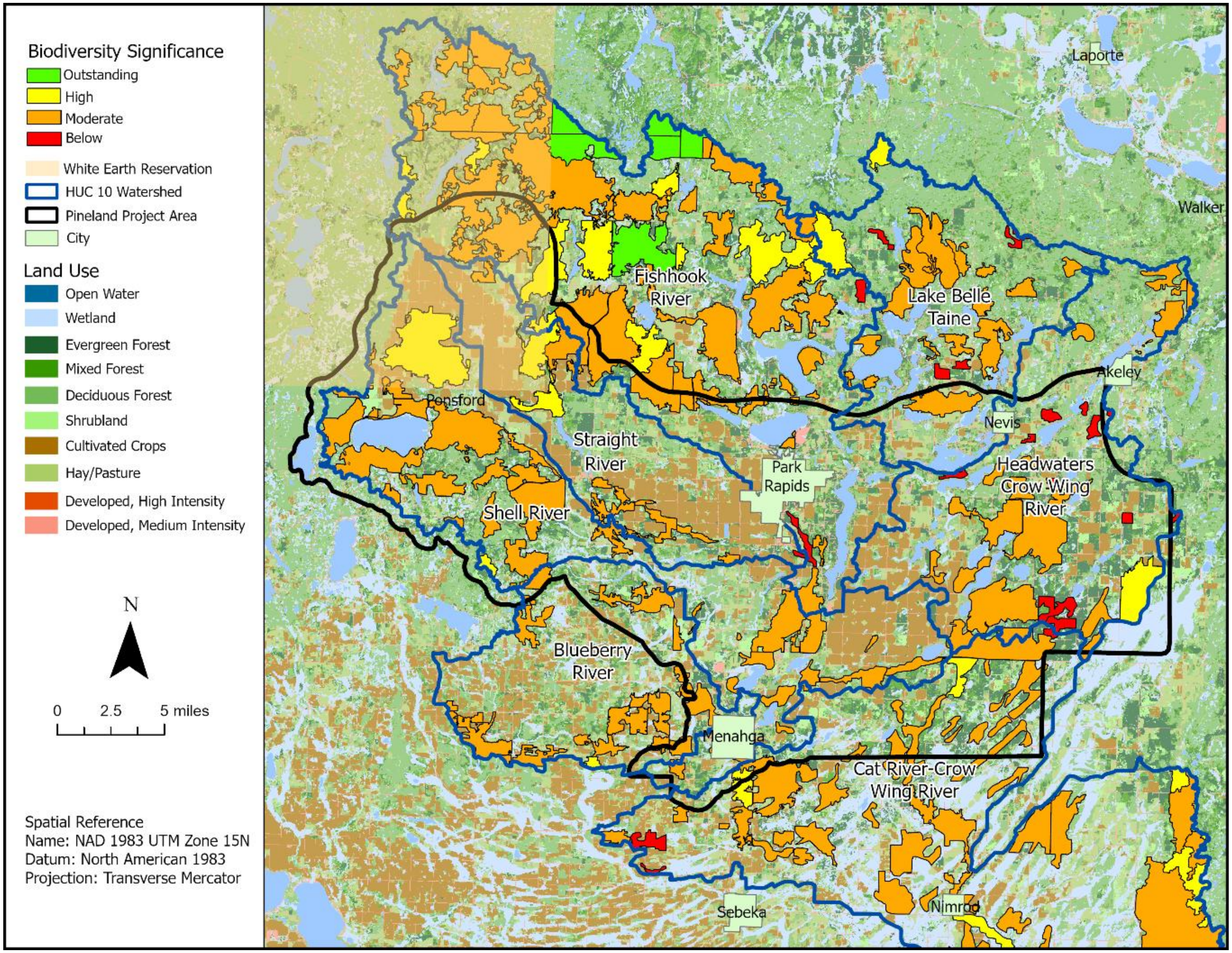

Figure 8: Areas of biodiversity and land use in the project area

Aquatic Life Assessments

MPCA conducts watershed-level assessments on a 10-year rotating basis. Part of those assessments include determining aquatic life use and whether it supports healthy habitats. The aquatic life use assessments include a survey of water chemistry, such as temperature, chloride, pH, dissolved oxygen, and assessments of the biology, macroinvertebrates, and fish communities, and overall habitat quality (MPCA, 2014a). The species surveyed in streams, rivers, and lakes are used to generate an index of biotic integrity (IBI) that accounts for species diversity and pollution tolerance in a waterbody. In other words, less tolerant species are more sensitive to pollutants, and more tolerant species can thrive in areas with pollution. MPCA will issue a “support” or “non-support” designation, which means that the waterbody does or does not support the beneficial use of that waterbody. Understanding the state of the waterbody and the stressors or drivers present can help land managers target solutions to improve aquatic life use conditions.

Mussel Surveys

Mussel surveys were conducted on four rivers (Shell, Straight, Blueberry, Crow Wing) and one lake (Shell Lake) in the project area in 2000 and between 2015 and 2016. A total of 10 mussel species were collected (Table 3**Error! Not a valid bookmark self-reference.**), some of which have special state designations as species of greatest conservation need (SGCN), species of concern, and threatened species (Sietman, 2026). The Shell River had the most mussel species (9), and the Shell River and Shell Lake had the most abundance (measured as Catch per Unit Effort, CPUE). The Straight River had only four species but a fair abundance. The Blueberry River had the fewest species and abundance.

Between the two sampling periods, 2000 and 2015–2016, juvenile recruitment (0-5 years old) was dramatically lower at 3 of the 4 stations on the Shell River in 2015–2016 (Sietman, 2026). A lack of recruitment means the age range of mussels is skewed toward older mussels, and there were likely stressors or conditions that prevented young mussels from surviving. The 2012 drought was a significant dry period that could have impacted the mussel population during the 2015-2016 survey.

Since the 2015-2016 survey, repeated droughts have occurred over the last five years (2021, 2023, 2024), with one considered “severe” (2021). Periods of very low water or dry stream beds can negatively impact mussel communities because mussels often live on rocky substrates in shallow water (between 1 and 3 feet).

Table 3: Mussel Species collected by the MN DNR in the Crow Wing River watershed in 2015-2016 (Sietman, 2026)

Mussel Common Name (*Scientific Name*)	Shell Lake	Shell River	Straight River	Crow Wing River	Fish Hook River	Blueberry
Cylindrical Papershell (*Anodontoides ferussacianus)*		9		10	0~	0
Pocket Book (*Lampsilis cardium)*		155	8	228	46	
Fat Mucket (*Lampsilis siliquoidea)*	96	1102	263	449	16	6
Heelsplitter* (*Lasmigona compressa)*		3	2	2		
Flutedshell# (*Lasmigona costata)*				1		
Black Sandshell*^ (*Ligumia recta)*		24		46	5	
Giant Floater (*Pyganodon grandis)*		1	4	63	0	
Lake Floater (*Pyganodon lacustris)*	29	10				
Creeper (*Strophitus undulatus)*		1		1		
Paper Pondshell (*Utterbackia imbecillis)*		2				
No. of Sample Sites	1	7	2	6	3	1
No. of Species (live/dead shell)	2/2	9/9	4/4	9/9	7/5	2/1
Total No. of Live Mussels	125	1307	277	800	67	6
Search time (minutes)	40	645	190	670	240	180
Catch per unite effort (CPUE) (mussels/minute)	3.13	2.03	1.46	1.19	0.28	0.03

*Species of Greatest Conservation Need (SGCN), ^MN Special Concern Species,

#MN Threatened Species

~0 means only shells were found, no live mussels.

Fish and Macroinvertebrate Surveys

The MN DNR conducted fish community surveys in the Crow Wing River watershed and calculated fish and macroinvertebrate IBIs (FIBI and MIBI, respectively). Fifty-two fish species and hundreds of aquatic insect taxa were collected within the subwatersheds (MPCA, 2014b). A few special-concern fish species were collected: the pugnose shiner and the least darter. Many other pollution-sensitive fish species were collected, such as brook trout, brown trout, mottled sculpin, longnose dace, hornyhead chub, and several redhorse species, as well as pollution-sensitive aquatic macroinvertebrates such as Hemorodromia, Isoperla, Atherix, Baetisa, and Acerpenna (MPCA, 2014b).

The results of the aquatic life assessment are displayed for the rivers in the project area (Table 4). Four rivers and streams are designated as non-supporting: Bender Creek, unnamed Creek, Straight River, and two segments of the Shell River. The Straight River has a dissolved oxygen impairment and is considered non-supporting, as a groundwater-dominated trout stream. The fish IBI is likely declining due to dissolved oxygen impairment. Recall that trout species are cold-water-dependent and are affected by changes in water oxygen levels. The Shell River segments are also considered non-supporting: the 681 segment has a dissolved oxygen impairment and reduced habitat quality for aquatic organisms. The 537 segment has an aquatic life impairment based on the fish survey results and, as a habitat, is considered degraded.

Table 4: Aquatic life use results from MPCA 2014 assessments

10-HUC Subwatershed	Stream Name Identification	AQL Assessment	AQL Analyses	AQL Impairments
Headwaters-Crow Wing River	Basswood Creek 07010106-568	FS	FIBI, MIBI	
	Indian Creek 07010106-569	FS	MIBI	
	Dinner Creek 07010106-690	FS	FIBI	
	Hay Creek 07010106-617	FS	FIBI, MIBI, T, Chl, pH, NH3	
	Crow Wing River 07010106-523	FS	FIBI, MIBI, T Chl, pH, NH3	
	Bender Creek 07010106-691	NS	FIBI, MIBI	Possibly Natural Background

10-HUC Subwatershed	Stream Name Identification	AQL Assessment	AQL Analyses	AQL Impairments
Fish Hook River	Fishhook River 07010106-543	FS	FIBI, MIBI, T Chl, pH, NH3	
	Fishhook River 07010106-542	FS	FIBI, MIBI, T Chl, NH3	
Shell River	Fish Creek 07010106-597	FS	FIBI, MIBI	
	unnamed ck 07010106-553	NS	MIBI	Natural Background
	Shell River 07010106-537	NS	FIBI, MIBI, T, DO, pH, NH3	FIBI, 4C+
	Shell River 07010106-681	NS	FIBI, MIBI, T, DO, pH, NH3	DO
Straight River	Straight River 7010106-517	FS	FIBI, MIBI	
	Straight River 07010106-558	NS	FIBI, MIBI, T, DO, pH, NH3	DO
Blueberry River	Kettle Ck 07010106-541	FS	FIBI, MIBI, T, NH3	
	Blueberry R 07010106-554	FS	FIBI, MIBI, T pH, NH3	

AQL = Aquatic Life, FS = fully supporting, NS = not-supporting, NA = not assessed, FIBI = Fish IBI, MIBI = Macroinvertebrate IBI, T = TSS, DO = dissolved oxygen, NH3 = ammonia nitrogen, 4C = changing habitat conditions might not improve FIBI.

Watershed Assessment

Alterations to the physical landscape, particularly the conversion of timberlands to agriculture, have affected local water and natural resources. The MN DNR conducted a watershed health assessment based on five ecological components (Biology, Connectivity, Geomorphology, Hydrology, Water Quality*)* and provided a condition score ranging from high to very low (MN DNR, n.d.-g). Some of the lowest scores are associated with terrestrial habitat and pollution sensitivity, whereas the hydrology components and many water storage areas are rated as high ecological ranking (

Table 5).

Table 5: Watershed health assessment framework ecological condition rankings

Assessment Categories	Components	Straight River	Shell River	Fish Hook River	Blueberry River	Headwaters - Crow Wing River	Lake Belle Tain
Hydrology	Water storage, wetland loss	H	H	H	H	H	H
	Water storage, altered water course	H	H (M)	H (L)	H-M	H	H (L)
	Water withdrawal	M (L)	H (L)	H	H	H (L)	H (M)
Geomorphology	Soil erosion susceptibility	H-M	H-M	H-M	H-M	H-M	H-M
	Pollution sensitivity of near surface materials	L-VL	M-L	M-L	M	M-VL	M-L
Connectivity	Terrestrial habitat connectivity	L-VL	L-VL	L-VL	L-VL	L-VL	L-VL
	Aquatic connectivity	H (M)	H-VL	H-M	H-M	H-M	H-M

H= high, H-M medium high, M = medium, L=low, VL = very low. The watershed assessments are conducted at the HUC-12 level, whereas the focus of this report is on HUC-10 watersheds. Where there was more than one rating of relatively equal coverage within a subwatershed both ratings are included with a hyphen (e.g., H-M). Score ratings in parentheses [e.g., H(L)] indicate that an isolated area was different from the majority of the 10-HUC subwatershed.

Aquatic Invasive Species

Aquatic invasive species (AIS) pose a significant threat to native ecosystems in Minnesota. The MN DNR monitors lakes and rivers for these species and maintains updated lists of infested waterbodies (MN DNR, 2025b). One of the most well-known AIS is the zebra mussel, which harms native mussels by attaching to their shells and eventually smothering them (USGS, 2024). Zebra mussels can also accumulate mercury in their bodies and increase mercury levels throughout the food chain (MDH, 2026). In addition to zebra mussels, other aquatic invasive species in the project area include certain snails, a crayfish, and invasive plant species (MN DNR, n.d.-h) (Figure 9).

The MN DNR also tracks terrestrial invasive species such as buckthorn, emerald ash borer, and spongy moth (MN DNR, n.d.7) (Figure 9). Trees that are already under stress—due to drought, pesticide exposure, or warmer climate conditions—have weaker natural defenses and are more vulnerable to pest invasions (Linnakoski, Kasanen, Dounavi, & Forbes, 2019).

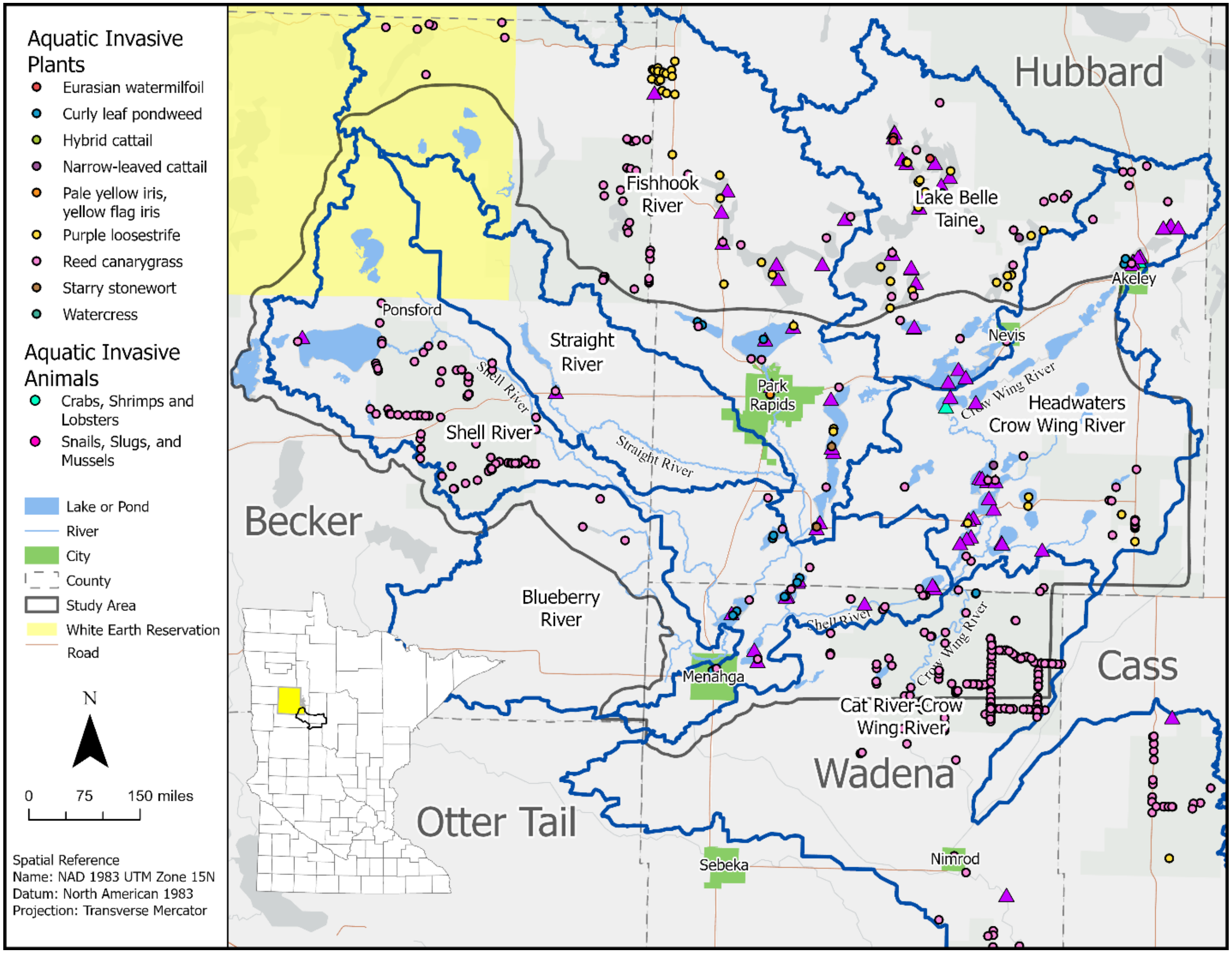

Figure 9: Aquatic invasive plants and animals present in the project area

Discussion

A companion publication to the watershed and monitoring assessment report published for the Crow Wing River watershed was an evaluation of the stressors causing ecological impairments. Consistent with local land use, primarily agriculture, forestry, and residential development, the primary stressors identified include (Stroom, 2014) (Johnson, 2023):

- Nutrients (nitrogen and phosphorus) from fertilizers, septic systems, animal agriculture, and runoff, which can lead to eutrophication, reduced dissolved oxygen, and direct toxicity to aquatic life.
- Hydrologic alterations, including both increased flows (from land conversion and wetland drainage) and reduced flows (from groundwater withdrawals for irrigation), which can degrade habitat through erosion, temperature changes, and reduced stream habitat.
- Fish passage barriers, particularly improperly designed culverts, can restrict movement, fragment habitat, and impair biological communities.
- Sediment was considered but determined to be a less significant stressor in this watershed due to flat terrain and highly permeable sandy soils, which limit runoff and erosion (Stroom, 2014).

The presence of dams, weirs, and culverts in the project area restricts fish movement and can constrain mussel reproduction and distribution, as many native mussel species rely on host fish to complete their life cycles. Although mussel populations are not formally evaluated using IBI metrics and may not be prioritized in aquatic life impairment decisions by the State, they are highly sensitive to connectivity disruptions and an important cultural species for the WEN as well as other Tribal nations. In addition, poorly designed culverts can alter stream hydraulics, creating widened, low-velocity segments that accumulate fine sediments and organic material, leading to reduced dissolved oxygen and degraded habitat conditions.

Efforts have been made in the watershed to address fish passage issues. MN DNR, in partnership with Carsonville Township and Becker County, received Lessard Sams Outdoor Heritage Funding to remove a low head dam on the outlet of Shell Lake and improve three other passages at road crossings (MPCA, 2023). The follow-on fish surveys suggested that the total number of fish and the types of species are increasing without the dam in place (MPCA, 2023).

Continued monitoring will be important to determine whether mussel, fish, and macroinvertebrate communities respond to any plans to remove infrastructure in the watershed, particularly given prior evidence of reduced recruitment and mussels' sensitivity to flow, habitat stability, and host fish availability. Together, these findings underscore that while habitat quality remains a limiting factor in some systems, improving connectivity can play a critical role in restoring both fish and mussel populations and should remain a key component of watershed management strategies.

Conclusions

This section compiled existing studies to evaluate the condition of aquatic and terrestrial biological communities, including mussels, fish, macroinvertebrates, and plants across the project area. The project area supports important and diverse biological communities, including sensitive and high-value species. However, declines in mussel recruitment, aquatic life impairments, habitat fragmentation, nutrient and oxygen stress, and invasive species indicate that ecological conditions are under increasing pressure. These stressors are closely linked to land use changes, hydrologic alterations, and climate variability.

Understanding Where Wild Rice Roots Grow

Author: Amy Myrbo Ph.D., Amiable Consulting/Mud Enviro

Background

Wild rice is an important cultural and nutritional staple for many Indigenous people in the western Great Lakes region. Known as Manoomin in Anishinaabemowin and Psiŋ in Dakota, the dominant species in Minnesota is northern wild rice (Zizania palustris), an annual aquatic grass. The plant grows in shallow (1-5 feet deep), clear, very fresh waters. Land use changes have caused the geographical range of wild rice to decrease (MN Tribal Wild Rice Task Force, 2018). Changes that have resulted in wild rice's reduced range include draining or ditching land for agriculture, eliminating wetlands, shoreline development, discharge/runoff from wastewater, mining, and water treatment plants, agriculture, and competing plant species, both native and introduced.

Wild rice populations go through a four-year cycle with variable growth based on nitrogen availability in the plant's rooting zone. First, there is a growing season of relatively high plant density, followed by a year of poor growth, and then two years of moderate growth. Each water body experiences its own cycle as nitrogen availability is a result of the abundance of "rice straw" or the dead stalks of the previous year's plant (Walker & Pastor, 2006) (Walker, Pastor, & Dewey, 2010).

Other variables and processes also play a role in wild rice abundance from year to year. These factors mean that a survey of wild rice populations in a water body in a given year may not reflect the highest or lowest levels of that population over multiple years (e.g., through one or more cycles). Factors that impact wild rice growth include, but are not limited to:

- Abrupt high-water levels during the early "floating leaf stage," typically in June, which pull out or drown young plants by reducing sunlight penetration and thus photosynthesis.
- Pests (caterpillars, swans, carp, and others) and fungal infections.

In the mid-1900s, as part of his Ph.D. thesis, John Moyle studied how the chemistry of water in Minnesota's lakes influenced plant type and growth. He observed that wild rice generally did not grow well in sulfate-rich waters (Moyle, 1945) (Moyle, 1956). This research and Moyle's expertise served as the framework for later legislation on sulfate discharge into Minnesota waters. Since 1975 the state of Minnesota has limited the discharge of water with sulfate (SO_4^{2-}) levels over 10 milligrams per liter (mg L^{-1}) to waters used to grow wild rice. Extensive experiments and field studies were conducted by UMN to confirm Moyle's work. The experiments and studies unequivocally showed that sulfate, when converted to sulfide in the rooting zone of wild rice, is one of three controls on wild rice presence/absence, along with water clarity and temperature (Myrbo, et al., 2017) (Pastor, et al., 2017).

Introduction

To convert sulfate into sulfide, the reaction requires an electron acceptor (sulfate) and an electron donor such as organic matter, reduced iron, ammonia, etc. Organic matter is commonly abundant in lake and river sediments, especially in water bodies with slow flow or stagnation. Sulfate-reducing bacteria (SRB) use this organic matter to convert sulfate to sulfide. When SRB oxidizes organic matter to convert sulfate into sulfide, there are multiple compounding effects beyond increased sulfide levels, including:

- Chemical components of the organic matter are released to the environment, leading to what is known as "internal loading," (i.e., addition of chemicals from *within* the water body [in this case the sediments] rather than from external sources such as streams, groundwater, and runoff).
    - Sedimentary organic matter contains nitrogen and phosphorus which, when released from the sediments, can lead to algal blooms and eutrophication.
    - Inorganic mercury (Hg) deposited from the atmosphere adheres to sedimentary organic matter and is released to the water column when organic matter is oxidized. SRB then methylates (adds one carbon and three hydrogen atoms) to Hg to form methylmercury (meHg), which is the only form of Hg that enters the food chain and causes neurological problems in animals that eat fish, including humans, eagles, ospreys, otters, and others.
    - The carbon in the sediment organic matter is converted to dissolved organic carbon, which can decrease lake transparency by coloring the water, and dissolved inorganic carbon, which can change the alkalinity and buffering of the water (Myrbo, et al., 2017b).

Research, particularly from the Netherlands, has shown how increased sulfate loading to water bodies and resulting increases in sulfide in sediments have led to substantial changes in aquatic plant communities (Lamers, et al., 2013) (Simkin, Bedford, & Weathers, 2013). Sulfide levels above 120 micrograms per liter (120 $\mu g\ L^{-1}$) harm wild rice (MPCA, 2017). LaFond and colleagues have determined through experimental research (LaFond-Hudson, Johnson, Pastor, & Dewey, 2020) that sulfide can react with iron in sediment to form a coating on wild rice roots that disrupts the uptake of nitrogen when the plant is setting seeds. This finding is congruent with earlier work that showed that wild rice plants exposed to sulfide in the sediment pore waters (the water between particles of sediment in the rooting zone) have fewer seeds and that those seeds are smaller (Pastor, et al., 2017). The coating of wild rice roots is readily accepted as how sulfide exposure destroys wild rice populations over time.

Individually and together, the effects of sulfate conversion to sulfide – toxic sulfide, nutrients, mercury, methylmercury, and carbon compounds – can cause major changes in aquatic ecosystems, disruption of habitats, and health consequences for people and animals, beyond the harm to wild rice. Sulfate can be converted to sulfide and back to sulfate an essentially unlimited number of times (Hansel, et al., 2015); each time sulfide is produced by the reaction with organic matter, additional amounts of nutrients, Hg and meHg, and DOC and DIC are released, making sulfate a persistent and powerful pollutant in freshwater aquatic ecosystems.

Project Objectives and Goals

The objective of this study was to characterize the rooting zone in wild rice lakes by measuring water, porewater, and sediment parameters related to wild rice health. The goal was to collect these parameters and analyze whether they were correlated with wild rice abundance at each site. As part of the larger research study, the objective is to analyze the data to determine whether the parameters and wild rice health are related to groundwater and surface water drawdown and agricultural activities in the region.

Hypotheses and Predictions

Based on extensive field data collected by UMN and the MPCA in 2011-2013 (Myrbo, et al., 2017), three factors were determined to control the amount of toxic sulfide in sediment pore waters (Pollman, et al., 2017). With these controls in mind, water bodies with high organic carbon and low sediment extractable iron should be more sensitive to sulfate pollution than those with low organic carbon and high iron:

- Surface water sulfate (more sulfate leads to more sulfide through conversion)
- Sediment organic carbon (more organic carbon leads to more sulfide as microbes use organic carbon to convert sulfate to sulfide)
- Sediment extractable iron (more iron leads to less sulfide as iron binds with sulfide to form minerals).

Study Area and Access

In April 2025, White Earth Division of Natural Resources (WEDNR) staff developed a list of 17 wild rice lakes for sampling (Table 6). This list partially overlaps with the lakes sampled by the UMN team, with four lakes in common. Four of the Crow Wing chain of lakes were added to the list because field workers could efficiently travel to multiple adjoining lakes with a single launch of the boat. The WEDNR provided expertise by characterizing each lake by the quality of its wild rice population.

Wild rice population quality among the lakes sampled varied considerably. Sampling occurred at 25 sites in 19 unique lakes in summer-fall 2025 (open water) and March 2026 (through the frozen lake surface). Table 6 and Table 7 show the sampling locations, sulfate and sulfide measures, and wild rice abundance. Lakes were accessed using a canoe, motorized boat, or by driving onto the frozen lake surface. The field team located wild rice beds in the field. Where wild rice beds were absent, the field team would sample near water lily beds. Some water lilies are invasive and outcompeting with wild rice. Qualitative notes on wild rice/water lily abundance, sediment characteristics, and other notable features were collected.

Methods

Water Characterization and Sampling

Water samples were collected for major ions and ^{18}O/D isotopes by holding a syringe about 0.5 meters below the water surface and then filtering water into sample vials. Equipment was used to measure water clarity, pH, and conductivity. Additional details on equipment and sampling protocols can be found in Report F: Characterizing Rooting Zone Chemistry in Wild Rice Lakes.

Collection of Pore Water Sulfide, Sediment Iron < TOC (Total Organic Carbon), and aeDNA

The porewater is known as the area between the lake bottom and the overlying water. This sample is collected using a gravity corer (50 centimeters long, with 70-millimeter-diameter polycarbonate tubes). The gravity corer was deployed from a boat/through the frozen lake surface to collect an intact portion of the lake bottom and overlying water. A Rhizon™ sampler with 0.2 micrometer (μm) pores was used to filter and collect porewater anaerobically from a core from each site. Anaerobic sampling is crucial for accurate measurement of chemically reduced compounds, including sulfide, which convert to other compounds (in this case, sulfate) when exposed to oxygen.

Other components of data collected included samples for ancient environmental DNA (aeDNA) of wild rice, and sedimentary environmental DNA (sedaDNA) was also employed to determine whether wild rice had been present at the sampling site in the recent past (approximately 5-20 years). More details on sampling methods can be found in Report F: Characterizing Rooting Zone Chemistry in Wild Rice Lakes.

Results and Discussion

Data and summary statistics are shown in Table 6 and Table 7. Sedimentary environmental DNA (sedaDNA), employed to determine whether wild rice had been at the sampling site in the recent past (5-20 years). Data processing was limited to calculating “sulfate sensitivity,” a new parameter representing the ratio of organic carbon to extractable iron. This parameter is intended to indicate the vulnerability of study water bodies to increasing sulfide based on their sediment chemistry.

Table 6: Pore water sulfide, surface water sulfate, sulfate sensitivity, total organic carbon, and extractable iron measurements from study lakes, with wild rice abundance field observations, White Earth Division of Natural Resources personnel's notes about wild rice abundance, and results from the sedimentary wild rice DNA. Sorted by wild rice abundance.

Site Name	Sulfide ($\mu g\ L^{-1}$)	Sulfate ($mg\ L^{-1}$)	Sulfate Sensitivity	TOC (%)	Extr. Fe ($\mu g\ g^{-1}$)	Wild Rice Abundance	Notes from WEDNR	sed. wr DNA (P=present / A=absent)
First Crow Wing Lake (dense bed)	41	0.83	26	17.9	6840	High		P
Little Basswood Lake (bay)	47	0.04	233	13.3	570	High	Excellent	P
Little Basswood Lake (landing)	35	0.04	60	20.6	3440	High	Excellent	A
Kurttis Corner Pond	35	0.12	37	30.4	8230	High	Excellent	A
Shell Lake	68	0.04	44	23.2	5260	High	Excellent – important ricing lake	P
First Crow Wing Lake (mixed bed)	67	0.83	22	19.8	9100	Medium		A
Second Crow Wing Lake (mixed bed)	41	Nd	50	11	2200	Medium		P
Sixth Crow Wing Lake (mixed bed)	86	1.27	27	10.1	3680	Medium	Crow Wings, just south of Nevis, does get riced.	A
Eight Crow Wing Lake (middle bed)	44	1.78	31	9.56	3080	Medium		A
Ninth Crow Wing Lake (mixed bed)	60	1.69	52	11.8	2280	Medium		A
Big Basswood Lake	50	0.54	24	26.7	11300	Medium	Excellent	A
Bass Lake	141	0.35	13	28.6	22200	Medium	Not sure it does, but it should based on what it is connected to	A
Radar Lake	72	0.50	47	37	7930	Medium	(Closed off now) inactive rearing pond but still	A

Site Name	Sulfide (µg L^{-1})	Sulfate (mg L^{-1})	Sulfate Sensitivity	TOC (%)	Extr. Fe (µg g^{-1})	Wild Rice Abundance	Notes from WEDNR	sed. wr DNA (P=present / A=absent)
							active. Near monitoring well. Has some rice but not across the entire lake	
First Crow Wing Lake (sparse bed)	22	0.83	27	10.4	3900	Low		A
Second Crow Wing Lake (sparse bed)	20	5.27	45	10.5	2320	Low		P
Wahbegon Lake	74	0.10	69	28.8	4190	Low	Poor. Everything around it has rice	A
Hinds Lake	88	4.38	115	35.9	3120	Low	Off reservation	A
Big Rush Lake (site 1)	18	0.69	17	15.9	9380	Low	Poor. Seeded bed since 1980s	A
Big Rush Lake (site 2)	55	0.69	20	16.3	8260	Low	Poor. Seeded bed since 1980s	A
Straight Lake (side bay)	362	11.95	76	19.6	2580	Absent	Off reservation	A
Straight Lake (by resort)	56	11.95	30	9.83	3280	Absent	Off reservation	A
Straight Lake (by inlet)	25	11.95	3	15.6	59000	Absent	Off reservation	A
South Roberts Lake	112	0.15	62	28.8	4680	Absent	Poor	A
Bass south of Shell	133	0.21	76	27.9	3680	Absent	Nothing harvestable	A
Mission Lake	20	0.13	1	18.9	131000	Absent	Poor. The flow out of it is orange, high iron content but no rice	A
Bad Medicine Lake	72	0.11	21	7.36	3480	Absent	There and not very good. Good clarity; lots of springs	A

Table 7: Summary statistics for data collected from wild rice lakes in this study.

Summary Statistics Category	Sulfide ($\mu g\ L^{-1}$)	Sulfate ($mg\ L^{-1}$)	Sulfate Sensitivity	TOC (%)	Extra Fe ($\mu g\ g^{-1}$)
High	362	11.95	233	37	131000
Low	18	0.04	1	7.36	570
Average	71	2	47	19	12499
Standard Deviation	68	4	46	9	26685

Results for parameters of interest are provided in Table 6. The sites selected for sampling are variable in their geochemistry. Porewater sulfide and TOC range over an order of magnitude, while sulfate and extractable iron range over two and three orders of magnitude, respectively. The study area has naturally very low-salinity, sulfate-poor waters. Sulfate values at sampled sites were generally low to very low compared with the state standard of 10 mg L^{-1} (Figure 10).

Three sites (Bass Lake [north of Shell River], Bass Lake [south of Shell River], and the side bay at Straight Lake) had porewater sulfide levels over 120 µg L^{-1}, the level determined to harm wild rice (Figure 11) (MPCA, 2017). One additional lake (South Roberts Lake) was found to have high porewater sulfide (112 µg L^{-1}), slightly below the 120 µgL-1 threshold. The remaining lakes in the study set have porewater sulfide levels below 120 µg L^{-1}. Both of the sites with >120 µg L^{-1} pore water sulfide, as well as the site with 112 µg L^{-1}sulfide, had only <0.5 mg L^{-1} sulfate in surface waters. The highest sulfide was found in a bay of Straight Lake, which has about 12 mg L^{-1} sulfate in surface waters. Of note, in the MPCA-UMN field study (Myrbo, et al., 2017), five sites had sulfate <2 mg L^{-1} and sulfide >120 µg L^{-1}. None are geographically close to the Pineland Sands area, but four of five are situated on glacial surficial material from the Des Moines Lobe (which covers a large portion of the state, including the Pineland Sands area), which may contain sulfur derived from ancient marine sediments.

Wild rice was absent in two out of three lakes with high porewater sulfide, as expected based on the 120 µgL^{-1} threshold. However, the lake with the third-highest sulfide levels had wild rice in medium abundance. Any additional sulfate discharged to the lakes in this study is expected to increase porewater sulfide because the lakes contain large amounts of organic sediment. Because sulfide can react with sediment iron to form a coating on wild rice roots that disrupts uptake of nitrogen at the point in the season when the plant is setting seeds, some sulfide will react with dissolved iron in lake sediments. This may also cause wild rice to be harmed by the buildup of harmful iron sulfide plaques on plant roots (LaFond-Hudson, Johnson, Pastor, & Dewey, 2020).

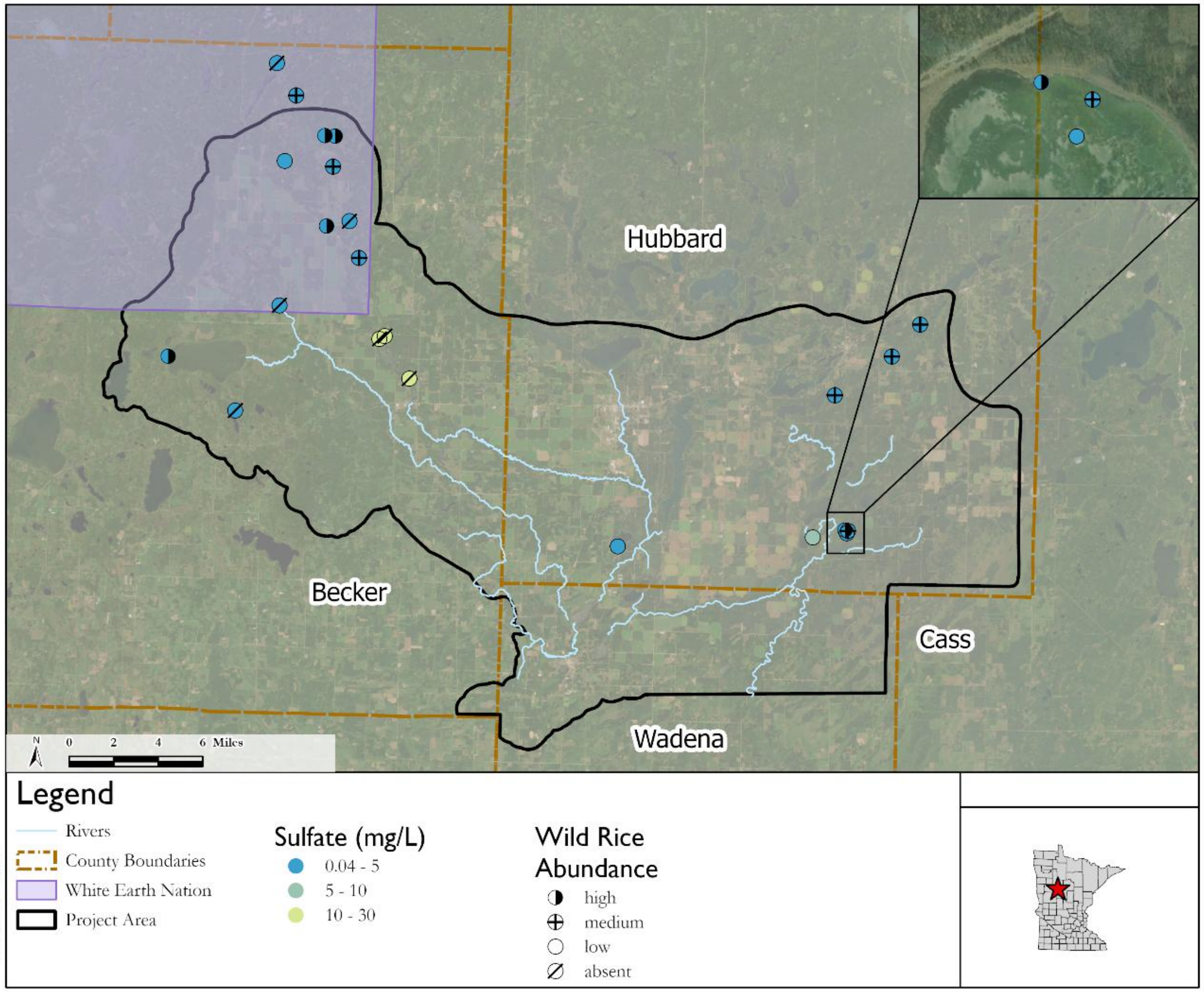

Figure 10: Sulfate measurements and wild rice abundance across study area

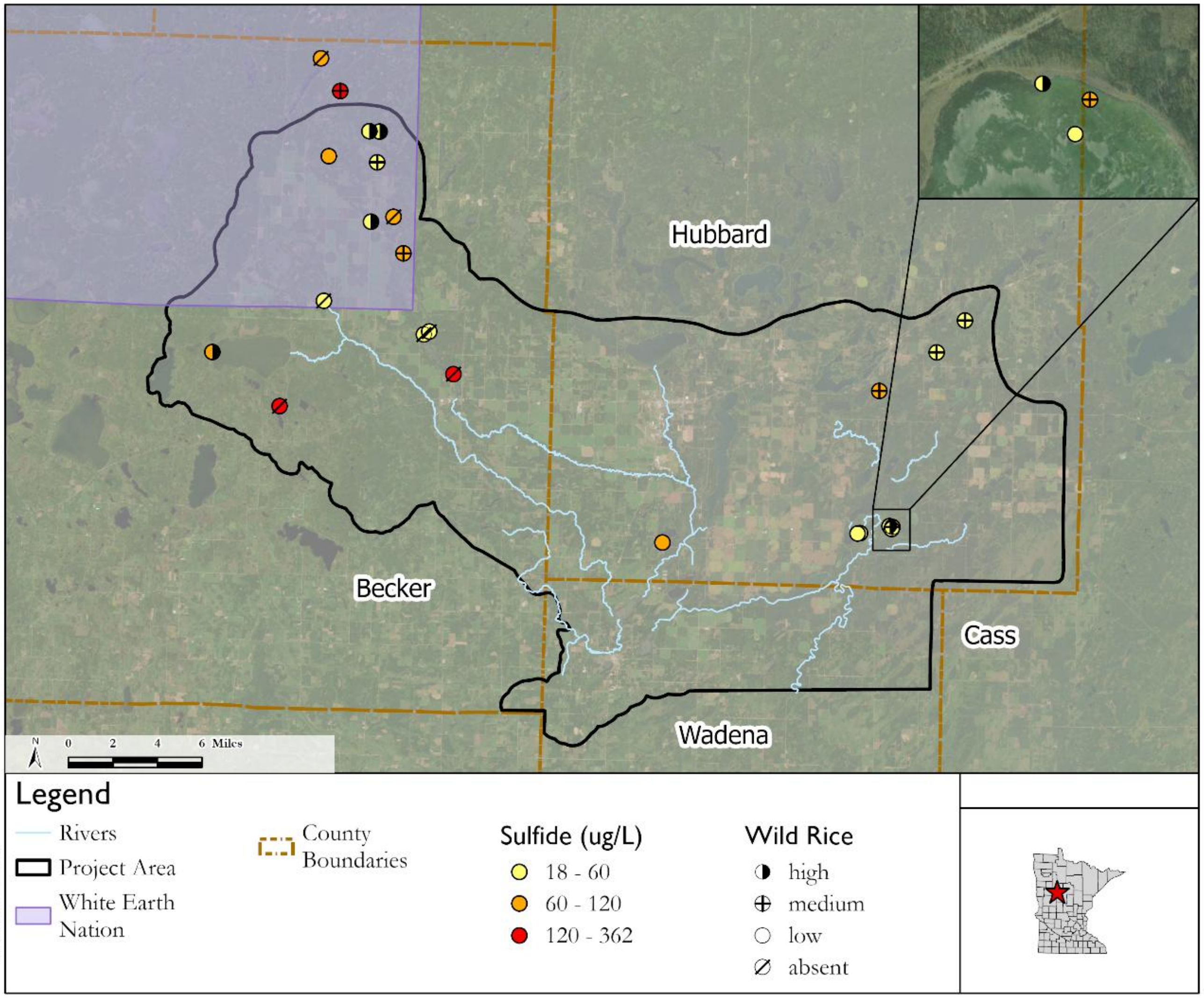

Figure 11: Sulfide measurements and wild rice abundance across study area

The new parameter, sulfate sensitivity varies widely, from a value of 1 (Mission Lake, which has extremely high iron abundance) to a value of 233 (one site in Little Basswood Lake that has extremely low iron). This metric is highly influenced by the iron value (the denominator), which varies much more than the organic carbon value, and could be modified or further studied and refined. As a preliminary index, this number simply indicates the degree to which any additional sulfate discharged to the lakes in this study is expected to lead to increases in pore water sulfide. However, all of the lakes in the study are sensitive to added sulfate, because they largely have high organic matter in their sediments. Some sulfide will be mitigated by dissolved iron in those lakes with abundant iron in the sediments, but wild rice may also be harmed by the buildup of harmful iron-sulfide plaques on plant roots (LaFond-Hudson, Johnson, Pastor, & Dewey, 2020).

Wild rice aeDNA results in the study were detected in only five samples. Three of these were classified as having "high" abundance of wild rice, and the other two were "medium" and "poor," respectively. No other correlations were determined based on sediment type, sample holding time, or other factors. Overall DNA was detected in all samples. These results are in keeping with results from the ongoing research described above: even in sites with current stands of wild rice, and recent histories of abundant wild rice populations, detections are relatively rare. Researchers are working with international aeDNA experts to refine and further test the extraction and analysis techniques, with hopes of improving the reliability of this method so that it can be used by and for Tribal conservation and restoration.

Controls on Wild Rice Abundance

Sulfide is clearly not the only factor controlling whether wild rice is present in a lake: of five lakes (seven total sites) where wild rice was absent, only two had>120 $\mu g\ L^{-1}$ porewater sulfide; the others ranged from 20-25 $\mu g\ L^{-1}$ to 112 $\mu g\ L^{-1}$. Water transparency, another important control on wild rice presence/absence (Myrbo et al., 2017a) was high (over one meter) in all lakes in the study, indicating that transparency is not the controlling factor. As previously described, iron toxicity could also be a limiting factor.

In Bad Medicine Lake, wild rice was absent despite a sulfide level of 72 $\mu g\ L^{-1}$, well below the threshold. This lake was characterized by WEDNR as having many springs, which could influence sediment and porewater chemistry in locations where springs enter the lake. Other possibilities, such as sediment nutrient availability, a small seed bank, competition from other plants, or disturbance by animals, could be other causes of the absence of wild rice in this lake.

Field observations agreed well with WEDNR personnel's categorization of each lake, indicating that the lakes are relatively stable in their levels of wild rice abundance over the longer term.

Conclusion

Study results indicated that a variety of factors influence the health and abundance of wild rice in lakes. Sulfide clearly plays an important role: all but one lake with wild rice had pore water sulfide levels below 120 µg L^{-1}; two of five lakes where wild rice was absent had levels above 120 µg L^{-1}; and one had 112 µg L^{-}1 pore water sulfide. Most of the lakes sampled are vulnerable to additional sulfate inputs from multiple sources, which are expected to further harm wild rice populations. Care should be taken to monitor lakes with high sulfate sensitivity, especially those with healthy wild rice populations. Sulfide levels varied between sites with different sediment compositions. Because sulfide typically builds up over time, additional monitoring is recommended. Additional observations and measurements could be productive, such as sediment nutrient levels, aquatic plant surveys to understand any pressures from competing species, characterization of the seed bank in lakes where wild rice is currently absent, and monitoring for pesticides that may disrupt the wild rice life cycle. The wild rice aeDNA technique was inconclusive in this study. Additional observations and measurements could be productive, such as sediment nutrient levels, aquatic plant surveys to understand any pressures from competing species, characterization of the seed bank in lakes where wild rice is currently absent, and monitoring for pesticides that may disrupt the wild rice life cycle.

Field Methods and Trend Analyses

Authors: Joseph Anderson, Brady Hartman, Kerry Holmberg, Lillan Luensmann, Bella Radler, Lily Syzmanski, University of Minnesota, and Trevor Wozniak, Bemidji State University

Introduction and Background

In support of the hypotheses stated above that 1) elevated nitrate concentrations in groundwater and surface water are primarily from intensive agricultural practices and the use of irrigation, and 2) agricultural land use has changed the hydrology and baseflow of streams and lakes in the project area, the study team collected a wide range of data to support the analyses. The data analyses also incorporated both historical sources and newly collected hydrologic and chemical data to understand the long-term trends in streamflow, temperature, nitrate concentration, and baseflow.

This report describes the methods used to collect the field data and then presents four analyses conducted on this data. Those individual analyses include methods and additional information specific to the study's goals and objectives. Those analyses are as follows:

- An evaluation of historic and current pesticide trends compared to federal aquatic life benchmarks.
- Water chemistry and land use interactions to understand water chemistry and pesticide presence related to land use.
- Groundwater level and irrigation analyses to understand regional groundwater conditions, temporal groundwater trends, and potential relationships between increasing irrigation groundwater use and groundwater level responses within the study area.

Methods

The site descriptions and data collection methods presented here apply to all fieldwork conducted for the four trend-related studies. The data collected were used to support multiple analyses, including water chemistry and land use interactions, pesticide trends, and groundwater levels and irrigation impacts. While this section provides an overview of the shared field methods, each individual analysis includes its own detailed methods section describing the specific datasets, calculations, or equations used.

Sites and Data Collection

Surface water and groundwater monitoring were conducted in the Pinelands Study Area from Fall

2023 to Spring 2026 to support trend analysis of hydrologic and chemical data. Twenty-two sites were chosen to represent subwater basins with different land uses and levels of irrigation (**Error! Reference source not found.**, Figure 14, and Table 8). This allowed for characterization and trend comparison of basins and the impacts of different land uses. Chosen sites also included upper, mid and lower sections of each river segment as well as near-surface water and groundwater sites to allow a fuller hydrogeochemical picture of the project area. Collected data were scrutinized for missing data and errors. Checks were made for outliers and inconsistencies via graphing and summary statistics.

Table 8: List of sites,locations and site types

Site	Location	Site Type
1	Shell Lake Outlet	Stream
1A	Shell Lake Outlet Riparian1	Near surface
1B	Shell Lake Outlet Riparian2	Near surface
1C	Shell Lake Outlet Streambed	Near surface
1D	Shell Lake Stream Duplicate	Stream
3	County Road (CR) 125 Shell River	Stream
3A	Shell River CR125 Riparian1	Near surface
3B	Shell River CR125 Riparian2	Near surface
3C	Shell River CR125 Streambed	Near surface
4	Shell River Blueberry Bridge	Stream
4A	Shell River Blueberry Bridge Riparian1	Near surface
4B	Shell River Blueberry Bridge Riparian2	Near surface
4C	Shell River Blueberry Bridge Streambed	Near surface
5	Blueberry Spring	Spring
6	Blueberry Lake Outlet	Stream
7	Shell River Arbor	Stream
8	Fish Hook River Arbor	Stream
9	Shell River Highway (Hwy) 24	Stream
10	Shell Camp	Groundwater
11	Shell/Crow Hwy 13	Stream
12	Crow River Hwy 109	Stream
13	Crow River Crown pt	Stream
14	Crow River Hwy 13	Stream
15	Fish Hook River	Stream
16	Straight Hwy 149	Stream
17	Straight River CR 115 Stream	Stream

Site	Location	Site Type
17A	Straight River CR115 Riparian1	Near surface
17B	Straight River CR115 Riparian2	Near surface
17C	Straight River CR115 Streambed	Near surface
17D	Straight River CR 115 Stream Duplicate	Stream
18	Straight CR115 Spring	Spring
19	Straight Lake Outlet	Stream
20	Hwy 44 Concrete Structure	Stream
21	Shell Smokey Hills	Stream
100	Bass Bay Flow Station	Stream
GW1a	Agricultural Field Well Nest Deep	Groundwater
GW1b	Agricultural Field Well Nest Shallow	Groundwater
GW2	CRP Land Well	Groundwater
GW3	Pine Point Well	Groundwater
GW4	Greenwater SNA	Groundwater Well
L1	Big Bass Lake	Lake
L2	Big Rush Lake	Lake
L3	Fish Hook Lake	Lake
L4	Lower Twin Lake	Lake
L5	First Crow Wing Lake	Lake
L6	Straight Lake	Lake
L7	Shell Lake	Lake

A variety of measurements were collected at each of the lake, groundwater, spring, stream and near surface sites, including elevation, water levels, water chemistry, field parameters, nutrients, pesticides, and isotopes. The purpose of each measurement is described in Table 9 and the frequency of sampling at each of the site types is described in Table 10. As noted in Table 10, there was a one time sampling of nitrate in groundwater wells in October 2025. The sites sampled are displayed in Figure 14.

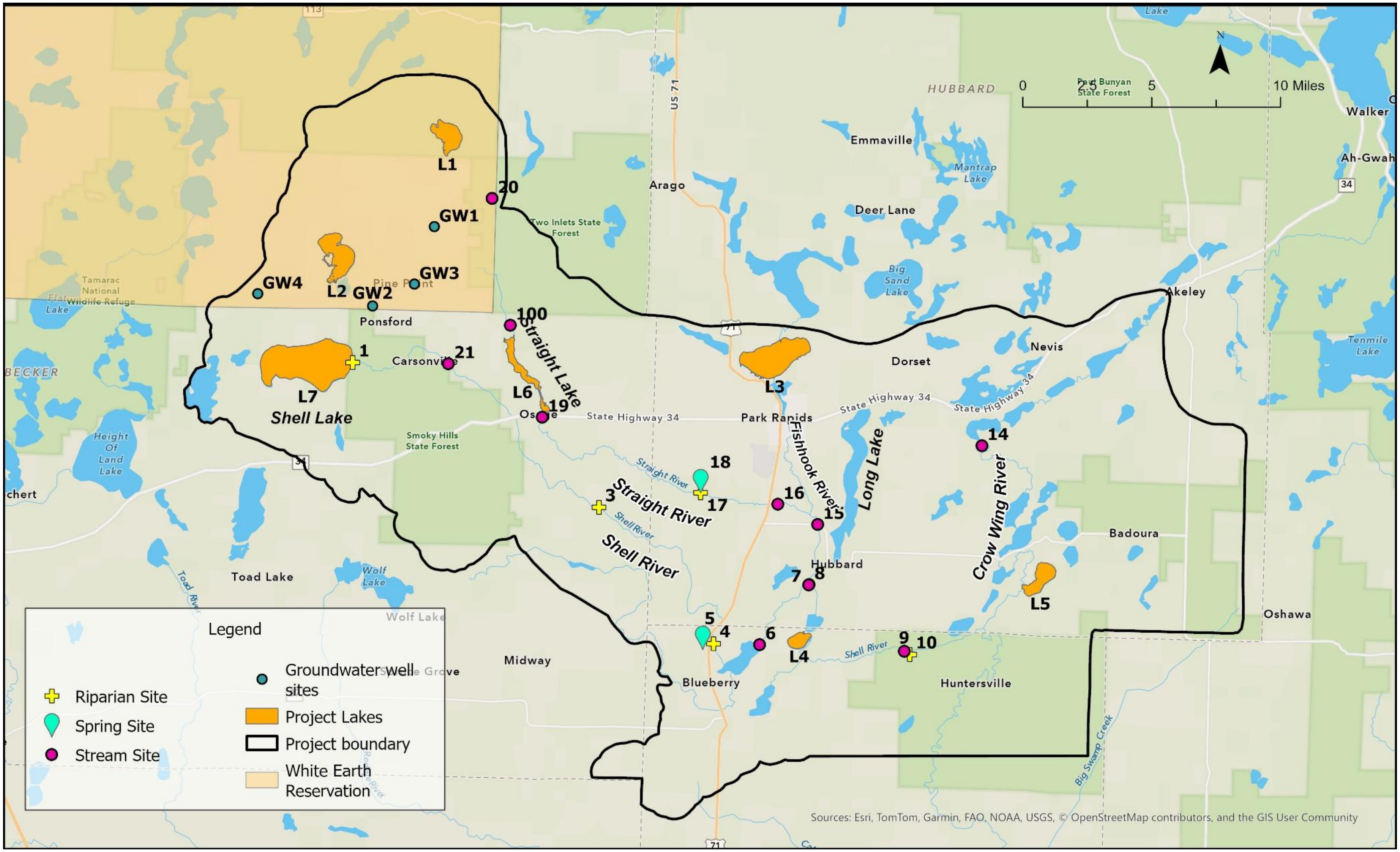

Figure 12: Project sampling sites

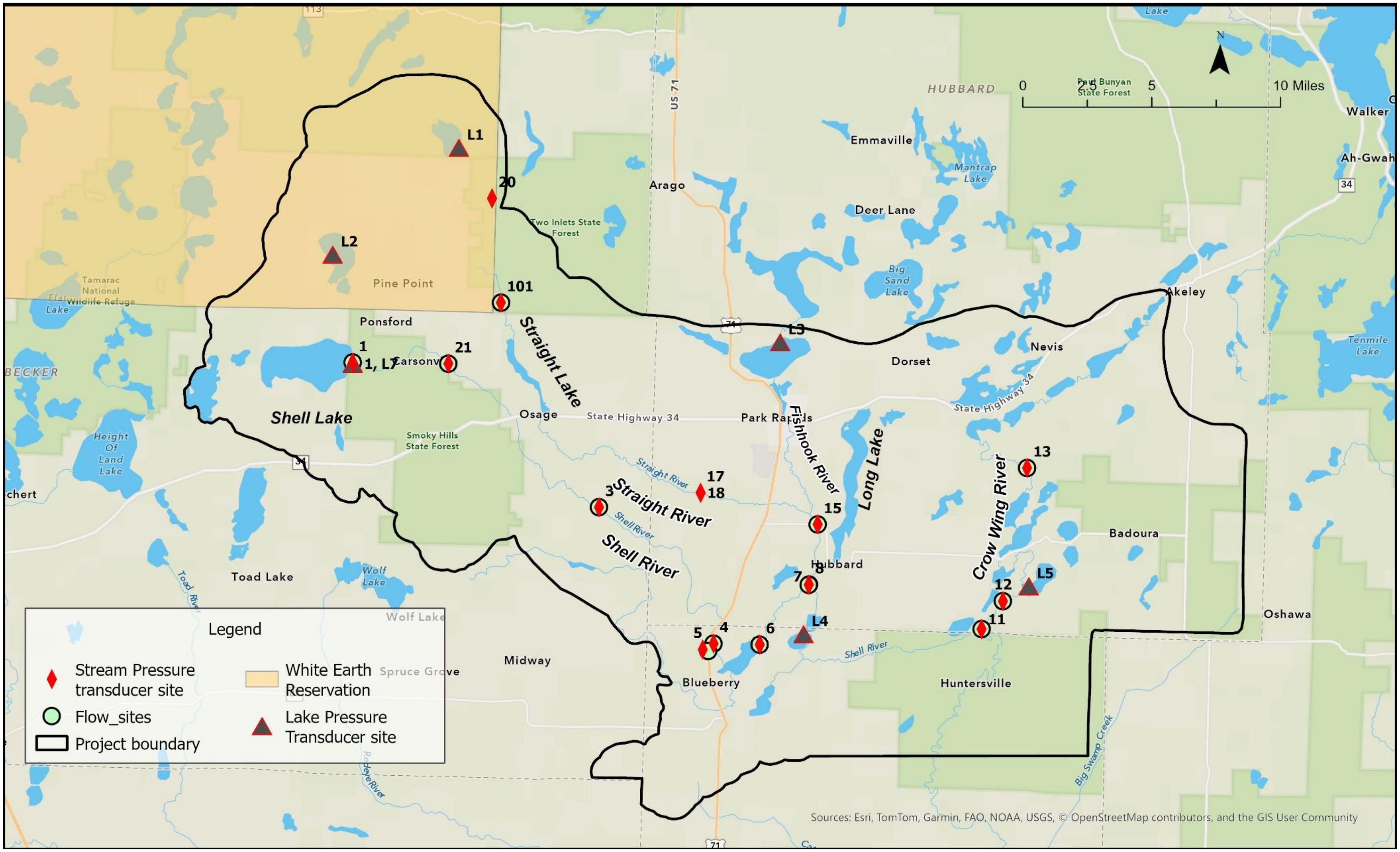

Figure 13: Stream flow measurement and pressure transducer sites

Table 9: Overview of measurements collected and purpose

Measurements	Purpose	Device(s)/Analysis Method
Elevation, feet (ft)	Site and well elevations were used to calculate pressure transducer readings	TrimbleDA2
Streamflow, cubic feet per second ($ft^3 s^{-1}$)	To differentiate between baseflow and runoff conditions, characterize stream chemistry, quantify contaminant loads, and be used as modeling inputs	StreamPro acoustic Doppler current profilers or FlowTracker2
Stream water levels, feet (ft)	Used to estimate discharge using rating curves and look at trends with precipitation and over time. Stream measurements can be used to assess precipitation responses and determine whether streams are gaining or losing.	Solinst Levelogger 4 and 5 Pressure Transducers (PTs). The water pressure is converted to water levels within the Solinst software. Measurements were collected every 15 minutes.
• Field parameters • pH • Specific conductivity, microsemens per centimeter, $\mu S\ cm^{-1}$ • Dissolved oxygen (DO), $mg\ L^{-1}$ • Temperature, Celsius (C)	pH is a measurement of how acidic or basic a waterbody is; specific conductance measures the ability of water to conduct an electrical current. It can also be used to understand the amount of dissolved material in water; DO measures the amount of available oxygen in a waterbody; Temperature is a foundational measurement that impacts the other field parameters and is a measurement of aquatic ecosystem health.	YSI multiparameter Sonde

Measurements	Purpose	Device(s)/Analysis Method
Water Chemistry (mg L^{-1}) Fluoride, Lactate, Acetate, Formate, Chloride, Nitrite - N, Bromide, Nitrate - N, Sulfate, Thiosulfate, Phosphate, Lithium, Sodium, Ammonium, Potassium, Magnesium and Calcium. Nitrate, sulfate, and total phosphorus.	The concentrations of these nutrients can indicate the level of anthropogenic contamination and the water's natural source.	RMB Laboratories analyzed samples collected
Isotopes oxygen-18 (^{18}O) deuterium (^{2}H) and nitrogen-15 (^{15}N)	Serve as indicators of potential water sources or contamination.	Water ^{18}O and ^{2}H were analyzed by GasBench isotope-ratio mass spectrometry. Analysis (15N, 18O) of NO3- in water was done by bacteria denitrification assay
Pesticides, micrograms per liter (μgL^{-1}) Nitrogen- and phosphorus-containing herbicides, chloroacetanilide metabolites (NP/CAAM), and neonicotinoid insecticides	Pesticides for a wide range of uses, but more specifically in agriculture.	Samples were analyzed by LC-ESI/MS/MS

Table 10: Samples collected by site type

Site type	What was sampled and frequency
Stream and near-surface water sampling	• Monthly measurements of streamflow between May and November 2024 and April through November 2025. • Nested piezometer water depths were recorded beginning in the fall of 2023. • Stream water depths were recorded by PTs at 14 of the stream sites beginning in the fall of 2023. • Pesticide samples were collected in October 2025. • Monthly ion samples, field parameters, and basic water chemistry measurements were taken from August to November 2023, February to November 2024, and April through November 2025. • Isotope samples were taken quarterly and after a few 1-inch rain events to catch the isotopic signatures during these times as well.
Groundwater	• Pesticide samples were collected in October 2025. • Basic field parameter measurements and ion and isotope samples were taken monthly from April 2025 to October 2025 in all but the Greenwater Lake well. The Greenwater well was sampled in October 2025 and February and April 2026 to capture any seasonal changes. • Water levels were measured with PTs from April 2025 to November 2025.
Nitrate Well[5]	A one-time sampling in October 2025 of 23 MN DNR quaternary water table aquifer observation wells included basic water chemistry measurements, major ions, ^{18}O,^{2}H of water, ^{15}N and ^{18}O of water NO_3 isotopes, inductively coupled plasma optical emission spectrometry elements, and total phosphorus.

[5] In support of the Groundwater Nitrate Source Apportionment modeling conducted by Xiang Li.

Site type	What was sampled and frequency
Lake	• Basic water chemistry measurements, field parameters, and ion and isotope samples were taken monthly from seven lakes from June to October 2024 and from June to October 2025. • Pesticide samples were collected in October 2025 (for two lakes). • Lake depths were recorded with PTs in all lakes from June to October 2024 and from June to October 2025.
Spring	The springs were sampled for the same parameters and at the same frequency as the stream sites.

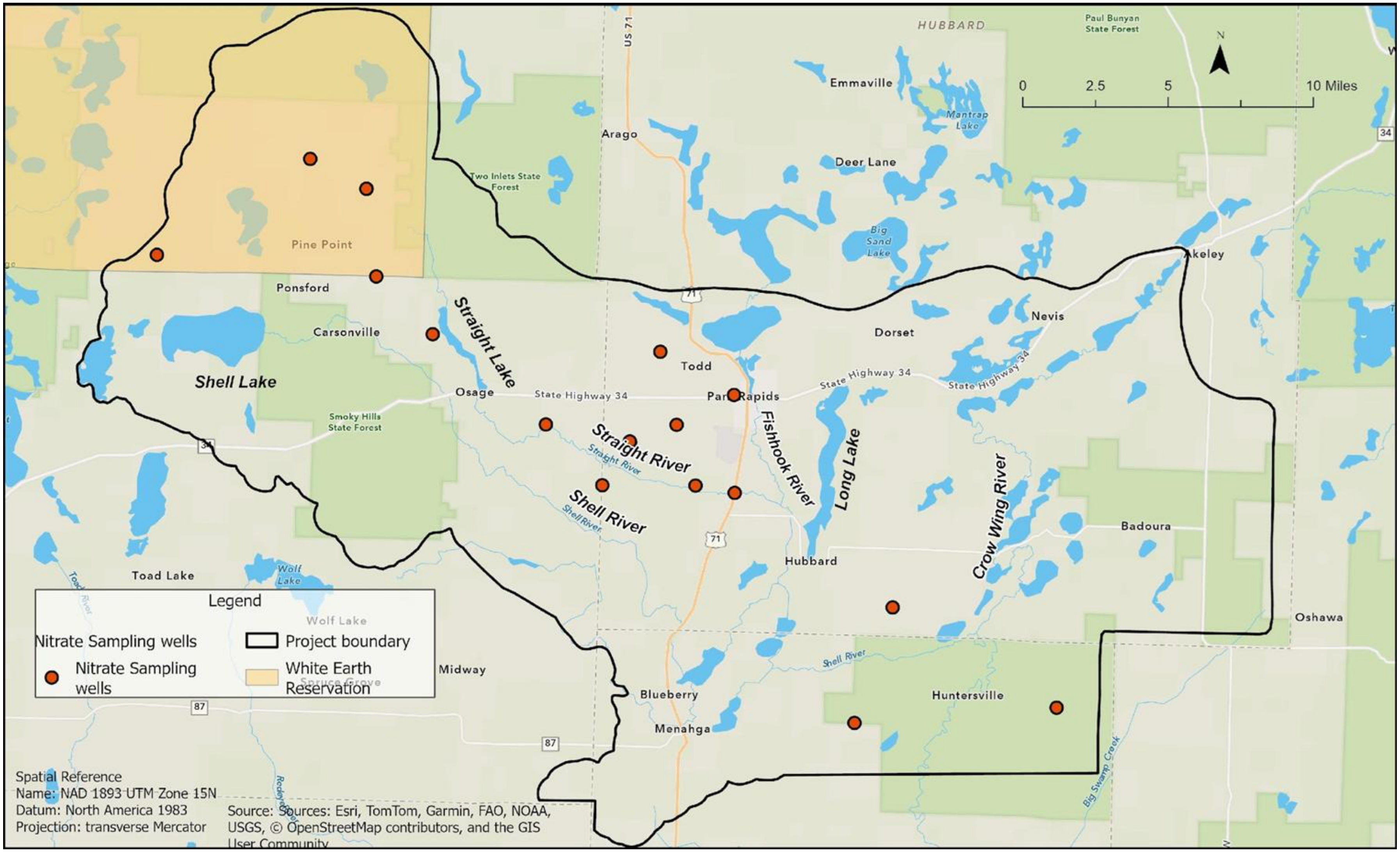

Figure 14: Nitrate groundwater wells sampled in October 2025

The following sections will describe study-specific methods, results, and discussion for the four trend analyses conducted using these data.

Pesticide Benchmarks

Pesticides detected in groundwater and surface water can serve as useful indicators of agricultural chemical transport through shallow groundwater systems, streambed exchange zones, springs, and connected surface water features. In the project study area, sandy soils, irrigation, shallow groundwater, and groundwater-fed streams/springs create conditions where pesticides may move from the land surface to groundwater and, in some settings, to surface water. This section evaluates whether detected pesticide concentrations from current sampling and historical data exceed federal EPA aquatic life benchmarks, Human Health Benchmarks for Pesticides, federal MCL's, and MDH human health-based guidance values. These benchmark values were developed as toxicity and human health criteria-based screening values used to estimate pesticide risk to freshwater organisms and human health, and they can be used to help interpret monitoring data.

Project Context

Historical pesticide data were extracted from the Water Quality Portal (WQP) using the Tools for Automated Data Analysis. The historic pesticide results were used to place the pesticide sampling results collected in 2025 (called "Current results") in a broader temporal context (Figure 15: Pesticide sampling sites). The analysis focused on selected herbicides, neonicotinoid insecticides, and pesticide degradates. Current and historical datasets were compared with selected ecological and human health benchmark values to determine whether any detected concentrations were above screening values. Because aquatic life benchmark values are intended for surface water ecological screening, this comparison supports the objective of evaluating whether pesticide detections may indicate potential groundwater-to-surface-water transport concerns. This is most relevant for springs, streambed wells, riparian wells, shallow groundwater locations, and other areas where groundwater may be hydrologically connected to streams, wetlands, or lakes.

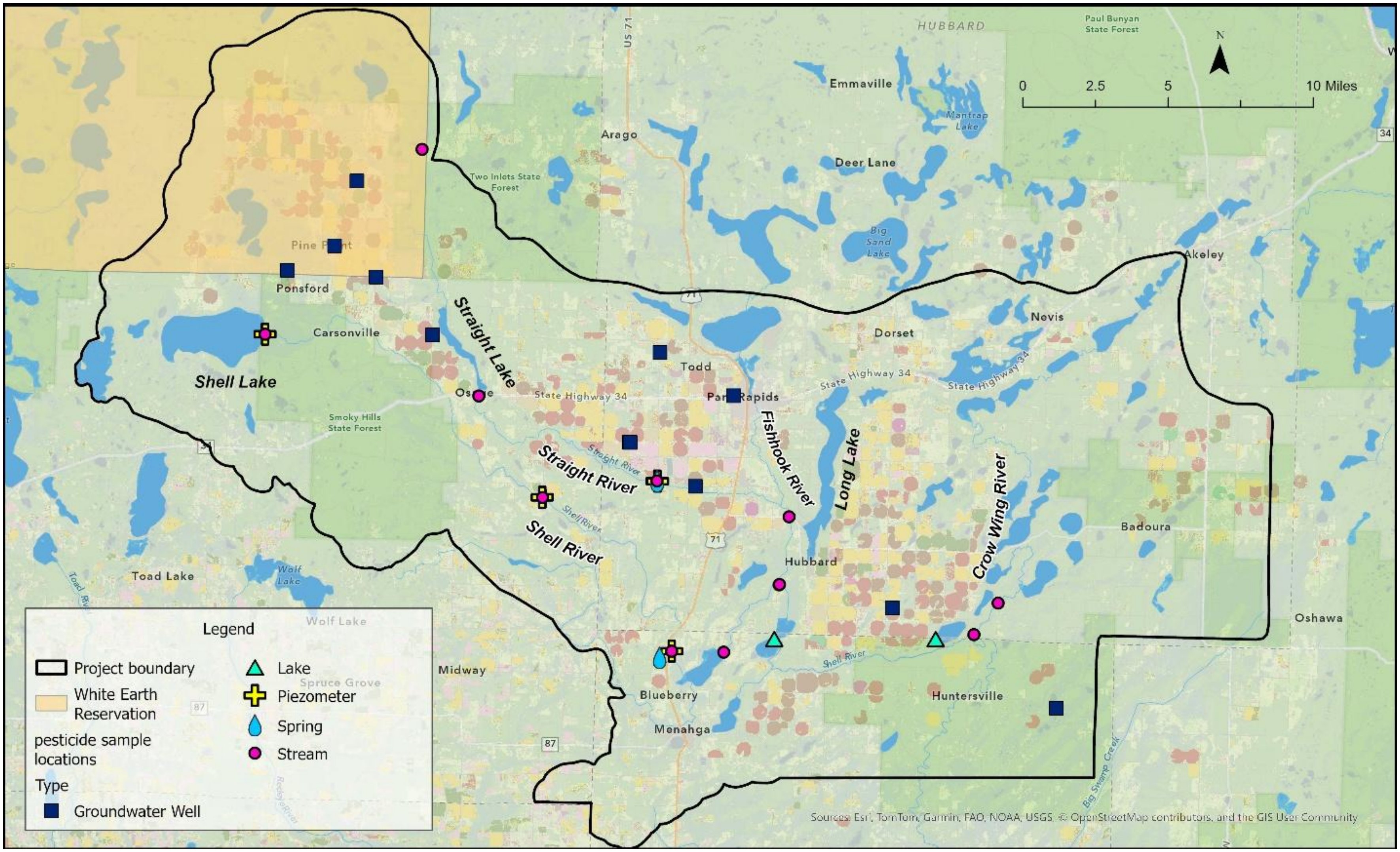

Figure 15: Pesticide sampling sites

Methods

Pesticide Sampling and Data Sources

Pesticide samples were collected to determine the presence or absence and concentration of pesticides in the project area waters. Additional information on pesticide sampling can be found in the methods section above. The benchmark comparison was evaluated using two main data sources. The current dataset was the October 2025 pesticide sampling results. The month of October was selected based on seasonal timing because pesticide movement can be affected by pesticide application periods, precipitation timing, and recharge. The historical pesticide records included analytes collected between 1980 to as late as 2022. They were sorted and reviewed in Microsoft Excel by analyte, sample result, media type, date, site, and county. Historical sample results were then compared individually against available federal ecological benchmark values and available state and federal human health and drinking water benchmark values for pesticides with applicable benchmarks.

All concentrations used in the benchmark comparison were standardized to micrograms per liter ($\mu g\ L^{-1}$). Non-detect results were not treated as exceedances. Results were flagged only when a detected numeric concentration exceeded an applicable numeric benchmark or guidance value.

Pesticides Selected for Aquatic Life Benchmarks

U.S. EPA aquatic life benchmark categories were reviewed for exposure impacts on freshwater vertebrates, freshwater invertebrates, nonvascular plants, and vascular plants. U.S. EPA updated the aquatic life benchmark table in September 2025, and the table now includes 782 chemicals (U.S. EPA, 2025f). The specific values can be found in Appendix 2: EPA aquatic life benchmark values.

At the state level, MDH guidance values provide human health context for contaminants in groundwater or drinking water sources. The guidance values also identify exposure duration and health endpoints, which are important for understanding whether a value is based on acute, short-term, subchronic, chronic, or cancer exposure assumptions (MDA, 2026) (MDH, 2016a) (MDH, 2016b) (MDH, 2020) (MDH, 2016c). Those specific values can be found in Appendix 3: Minnesota Department of Health, human health based guidance.

A refined list of pesticides was analyzed based on land use and other factors such as past data results (Table 11). Table 11 also includes a summary of pesticide use and the potential human health effects from prolonged exposure.

Table 11: Description of pesticides evaluated, use, and potential human health effects

Pesticide Name	Use*	Potential Human Health Effect(s)
Alachlor	Chloroacetanilide herbicides are used to control grasses and broadleaf weeds, including use on crops such as corn and soybeans (MDH, 2016a)	Eye, liver, kidney or spleen problems; anemia; increased risk of cancer (MDH, 2016a)
Atrazine	Chlorotriazine herbicide used to selectively control annual grass and broadleaf weeds, with major agricultural use on field corn, sweet corn, sorghum, and sugarcane (U.S. EPA, 2025g)	Cardiovascular system or reproductive problems (U.S. EPA, 2025h)
Metolachlor	Chloroacetanilide herbicides are used on agricultural crops, including corn and soybeans, to control broadleaf weeds and grasses (MDH, 2018)	Decrease in body weight, decrease in reproductive success, potential human carcinogen (EPA, 1995)
Metribuzin	Triazinone herbicides are used to control broadleaf weeds and some grasses (MDA, 2026)	Developmental, reproductive, neurotoxicity, and endocrine effects (MDH, 2013)
Metribuzin DADK	Desamino-Diketo-Metribuzin is a degradate, or daughter product of metribuzin that occurs when metribuzin, the parent product, breaks down in the environment (MDH, n.d.)	Developmental, reproductive, neurotoxicity, and endocrine effects (MDH, 2013)
Clothianidin	Neonicotinoid insecticides are used on food and non-food crops and is commonly used as a seed treatment, including corn and soybean seed. Clothianidin can also occur as a breakdown product of thiamethoxam (MDH, 2016b)	Reproductive effects (MDH, 2016b)
Imidacloprid	Neonicotinoid insecticides are used to control insects on agricultural crops, ornamental plants, nursery plants, and residential yards and gardens. In agricultural settings, it is used mainly as a seed treatment and foliar spray (MDH, 2020)	Decreased immune system effectiveness, tremors, and thyroid lesions (MDH, 2020)

Pesticide Name	Use*	Potential Human Health Effect(s)
Thiamethoxam	Neonicotinoid insecticides are used to control sucking and chewing insects such as aphids, thrips, and beetles. It is used on crops, ornamental plants, turf, and other settings, and it can break down into clothianidin (MDH, 2016c)	Adverse developmental, reproductive effects, and liver effects (MDH, 2016c)

*Chloroacetanilide and triazinone herbicides are classifications based on chemical structures

Results

A sample result was flagged when the measured concentration exceeded one or more available benchmark values for the same analyte. Because the benchmark exceedances identified in this comparison primarily occurred in groundwater or groundwater-connected samples, the results were interpreted as potential concerns for groundwater pathways rather than direct surface water aquatic life exceedances. One historical lacustrine wetland result was evaluated for use in a surface-water ecological screening comparison. The results below are organized by current pesticide results (sampled in 2025) and historical results (sampled between 2000 and 2022).

Current Pesticide Sampling Results

The current pesticide dataset two benchmark exceedances. It should also be noted this dataset was much smaller and not fully representative of pesticide concentrations in a year. The groundwater sample result for clothianidin was 0.0785 μg L^{-1}, in exceedance of the 0.05 μg L^{-1} freshwater invertebrate chronic benchmark (Table 12). The result was located in the Straight River watershed at a site named 10 MN DNR. The second benchmark exceedance was for metribuzin, at the 18/ CR115 Straight River Spring. Also located in the Straight River watershed, the result was 3.14 μg L^{-1} compared to the nonvascular plant NOAEC value of 2.3 μg L^{-1}. Site 18/CR115 was collected from a shallow piezometer installed in a spring. This site may warrant attention because springs can link groundwater chemistry to surface water receiving environments.

Table 12: Current pesticide sample results above EPA aquatic life benchmark values

Site	Media	Sample date	Analyte	Result (μg L^{-1})	Benchmark exceeded	Benchmark value (μg L^{-1})
10 MN DNR	Ground Water Well	10/06/2025	Clothianidin	0.0785	Freshwater invertebrate chronic (ChronicD)*	0.05

Site	Media	Sample date	Analyte	Result ($\mu g\ L^{-1}$)	Benchmark exceeded	Benchmark value ($\mu g\ L^{-1}$)
18/ CR115 Straight River Spring	Ground Water Piezometer	10/25/2025	Metribuzin	3.14	Nonvascular plant NOAEC^	2.3

*ChronicD: Refers to EPA's chronic benchmark for freshwater invertebrates. It is a screening value for long term exposure risk to aquatic invertebrates.

^NOAEC: Means no observed adverse effect concentration (NOAEC). This is the highest concentration tested at which harmful effects were not observed in the toxicity study.

Historical Pesticide Results

The historical pesticide dataset had 250 groundwater sample results above at least one aquatic life benchmark for six different pesticides. Table 13 includes the number of historical samples collected that exceeded the aquatic life benchmark values and the general location of the exceedance. The locations are visually represented in the Figure 16. The majority of exceedances occurred in the Straight River and Shell River watersheds. Some sample results were above more than one benchmark endpoint. Herbicide-related historical exceedances were less frequent but still present. Note that no HHBPs were exceeded.

Table 13 Historical pesticide results above benchmark values

Analyte	Number of results above the benchmark	Sample date range	Site/county distribution	Maximum result ($\mu g\ L^{-1}$)
Alachlor	3	11/13/2000 to 05/13/2002	1 site in Becker County	0.72
Benchmarks crossed: U.S. EPA Aquatic Life Nonvascular plant NOAEC^ = 0.035 $\mu g\ L^{-1}$; vascular plant NOAEC = 0.339 $\mu g\ L^{-1}$, National Drinking Water Regulation MCL goal = 0 $\mu g\ L^{-1}$				
Atrazine	1*	08/02/2017	1 site in Becker County	1.133
Benchmarks crossed: U.S. EPA Aquatic Life - nonvascular plant (IC50E) = <1 $\mu g\ L^{-1}$				
Clothianidin	106	10/24/2011 to 10/12/2022	3 sites in Becker County; 8 sites in Hubbard County; 3 sites in Wadena County	1.46
Benchmark exceeded: U.S. EPA Aquatic Life Freshwater invertebrate chronic (ChronicD)* = 0.05 $\mu g\ L^{-1}$				

Analyte	Number of results above the benchmark	Sample date range	Site/county distribution	Maximum result (µg L^{-1})
Imidacloprid	106	05/12/2010 to 10/12/2022	2 sites in Becker County; 6 sites in Hubbard County; 3 sites in Wadena County	2.26
Benchmark exceeded: U.S. EPA Aquatic Life Freshwater invertebrate chronic (ChronicD) = 0.01 µg L^{-1}; freshwater invertebrate acute (AcuteC)# = 0.385 µg L^{-1}. MDH (Short-term, Subchronic, Chronic) = 2 µg L^{-1}				
Metolachlor	7	03/28/2000 to 05/18/2005	1 site in Becker County; 1 site in Wadena County	31.2
Benchmark exceeded: U.S. EPA Aquatic Life Nonvascular plant NOAEC = 1.5 µg L^{-1} and IC50E = 8 µg L^{-1}; vascular plant NOAEC = 4.4 µg L^{-1} and IC50F = 14 µg L^{-1}. The highest result also crossed plant IC50= endpoints and the freshwater vertebrate chronic (ChronicB+) benchmark = 30 µg L^{-1}				
Metribuzin	4	03/28/2000 to 05/13/2015	1 site in Becker County; 2 sites in Wadena County	7.84
Benchmark exceeded: Nonvascular plant NOAEC = 2.3 µg L^{-1}				
Metribuzin DADK	1	05/15/2000	1 site in Becker County	38
Minnesota state guidance: MDH(Acute) = 30 µg L^{-1}; (Short-term, Subchronic,Chronic) = 10 µg L^{-1}				
Thiamethoxam	24	10/29/2014 to 10/19/2021	4 sites in Hubbard County; 3 sites in Wadena County	6.34
Benchmark exceeded: U.S. EPA aquatic life freshwater invertebrate chronic (ChronicD) = 0.74 µg L^{-1}				

*ChronicD: Refers to U.S. EPA's chronic benchmark for freshwater invertebrates. It is a screening value for long-term exposure risk to aquatic invertebrates.

^NOAEC: Means no observed adverse effect concentration (NOAEC). This is the highest tested concentration at which harmful effects occur.

#AcuteC: Refers to U.S. EPA's acute benchmark for freshwater invertebrates. It is usually based on short-term toxicity tests, which were not observed in the toxicity study.

" < ": Means the benchmark is reported as less than the listed value; the exact benchmark value is not specified yet and may be lower.

= IC50: The inhibition concentration where a measured function, such as algal or plant growth, is reduced by 50 percent.

“ * “: the asterisk next to the atrazine result indicates that one detected result was above the IC50E < 1 μg L^{-1}benchmark

+ChronicB: U.S. EPA chronic benchmark for freshwater vertebrates.

- Desaminodiketometribuzin is a degradate, or daughter product of metribuzin that occurs when metribuzin, the parent product, breaks down in the environment.

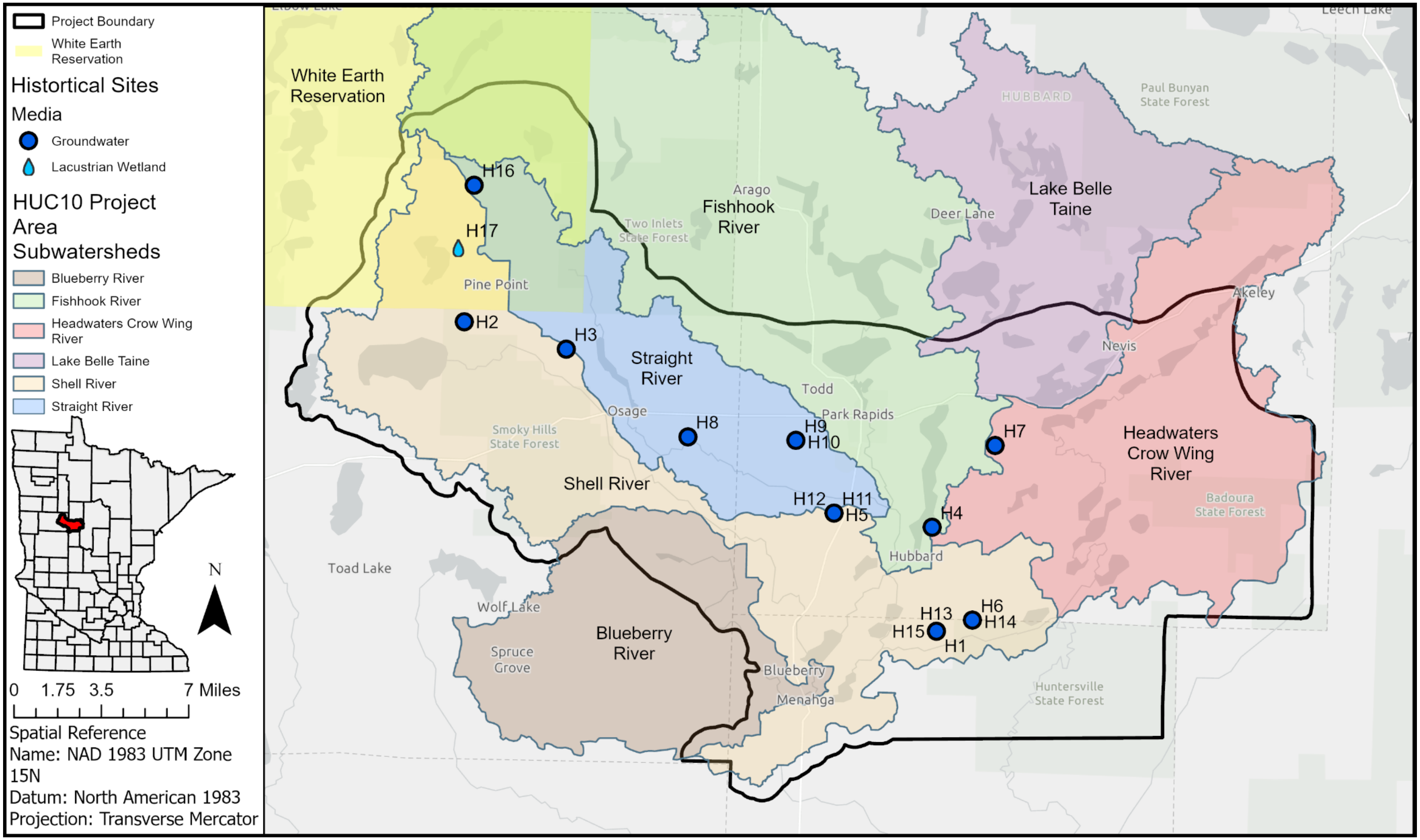

Figure 16: Location of historical pesticide exceedances for U.S. EPA aquatic life benchmarks

Discussion

Neonicotinoid insecticides were the most frequently represented pesticide group among historical results above reviewed benchmark values. Among the historical results, clothianidin had 102 of 146 detections above the freshwater invertebrate chronic benchmark, imidacloprid had 101 of 117 detections above the freshwater invertebrate chronic benchmark, and thiamethoxam had 22 of 113 detections above the freshwater invertebrate chronic benchmark, representing approximately 70%, 86%, and 19% of positive historical results, respectively.

MDA monitoring conducted in 2024 provides broader context for neonicotinoid results in the State rather than a direct comparison to the October 2025 project sites. In MDA's statewide 2024 ambient groundwater monitoring, clothianidin was one of the most frequently detected parent pesticides, with a reported detection frequency of 49% (MDA, 2025). In MDA's 2024 river and stream monitoring, clothianidin and imidacloprid accounted for most pesticide detections above aquatic life reference values, further supporting their relevance to benchmark screening in groundwater-connected aquatic settings (MDA, 2025). Neonicotinoid results above the benchmark screening values are important because they are tied to chronic freshwater invertebrate endpoints, meaning the comparisons represent screening-level concern for longer-term exposure to aquatic insects and other freshwater invertebrates rather than observed biological effects in the field (U.S. EPA, 2026h). This is particularly relevant where mobile compounds may move through sandy groundwater settings, because chemical sorption, persistence, and soil-profile conditions influence leaching potential (Lewis, Tzilivakis, Warner, & Green, 2016) (Perez-Lucas, Vela, El Aatik, & Navarro, 2019).

The benchmark endpoints also helped identify the ecological receptor groups most relevant to the screening results. For clothianidin, imidacloprid, and thiamethoxam, the most sensitive aquatic life comparisons in this study were associated with freshwater invertebrate endpoints (U.S. EPA, 2026h). In groundwater connected streams, springs, wetlands, or lakes, those receptors may include aquatic insects and other small invertebrates that serve as food sources for fish, amphibians, and wetland birds. For alachlor, atrazine, metolachlor, and metribuzin, the most relevant benchmark endpoints were associated with aquatic plants and algae, including nonvascular plants, aquatic vascular plants, duckweed, and algal growth endpoints (U.S. EPA, 2026h). These organisms should be viewed as potentially sensitive ecological receptors represented by the screening values, not as evidence of observed biological effects.

Herbicide-related benchmark-above concentrations were less frequent but still important. Alachlor, metolachlor, and metribuzin crossed plant based aquatic life benchmark values in the historical dataset, and the current metribuzin result at 18/CR115 Straight River Spring shows that herbicide concentrations above aquatic life screening values can also occur in groundwater-discharge settings. Atrazine was represented by one historical NARS lacustrine wetland above the aquatic plant benchmark screening used.

Metribuzin DADK was evaluated separately from parent metribuzin because the DADK result was compared with Minnesota human-health guidance, while parent metribuzin was compared with U.S. EPA aquatic life plant benchmarks. MDA's 2024 groundwater results showed that metribuzin DADK was detected, but at a lower frequency than major chloroacetanilide degradates such as metolachlor ethane sulfonic acid (ESA), metolachlor oxanilic acid, and alachlor ESA (MDA, 2025). Metribuzin DADK remained in the screening because one historical detected concentration was above Minnesota guidance. However, broader degradate-pattern interpretation should focus more on the degradates detected more frequently across the monitoring record.

Human-health benchmark findings were more limited than aquatic life findings. U.S. EPA HHBP values, federal MCLs, and Minnesota state health-based water guidance values were reviewed for the relevant pesticides, but no current detections were above an U.S. EPA HHBP, federal MCL, or Minnesota state guidance value. In the historic dataset, two imidacloprid results and one metribuzin DADK result were above Minnesota state health-based guidance values. These human-health findings are presented separately from aquatic life benchmark findings because they reflect different receptors and exposure assumptions.

Conclusion

The benchmark values exceeded were measured in groundwater, and these results should be interpreted as screening level indicators of potential concern rather than direct evidence of surface water aquatic life impairment, human health effects, or regulatory noncompliance. Future work should prioritize paired groundwater and surface water sampling at springs, streambed wells, riparian wells, and gaining stream reaches; incorporate non-detect/censored-data methods for trend analyses; compare older and current land cover conditions; and evaluate soil organic carbon and depth to water to better understand pesticide mobility in sandy groundwater settings.

Water Chemistry and Land Use Interactions

Overview

Recent landcover conversion from forest and native perennials to irrigated row crops have raised concerns about changes to surface and groundwater chemistry, specifically nitrate, chloride, sulfate, and pesticide loading. Building on historical data extracted from the WQP, this report summarizes the water chemistry collected during 2023 to 2025 and the October 2025 MN DNR well chromatography.

This section aims to answer two questions:

- Which dissolved ions co-vary across waters in the project area, and what does that structure imply about their source?
- How do ion chemistry and pesticide occurrence relate to surrounding land use, and is agricultural cover the dominant cause of elevated concentrations?

Methods

Inputs for this analysis came from laboratory analyses of water samples from project sites, including ions and pesticides, as well as Sonde water chemistry and land-use cover. All analyses use samples or data collected between 2023 and 2026. All processing was performed in R 4.5.3 (R Core Team, 2025). Scripts were authored in Visual Studio Code.

Spearman Correlation for Ion Comparisons

To understand how the different chemicals in the water were related to each other, this study evaluated thirteen common substances (ammonium, bromide, calcium, chloride, fluoride, lithium, magnesium, nitrate, nitrite, phosphate, potassium, sodium, and sulfate) in lakes, streams, and groundwater wells, both individually and all together. Because the data included a wide range of values—some very high, some very low, and some near detection limits—a method that examines general patterns and rankings rather than exact numerical relationships was used. This approach helps show whether certain chemicals tend to increase or decrease together. For each pair of chemicals, only the samples where both were measured were compared to make use of as much data as possible. The project team then calculated the strength of the relationships between the chemicals and assessed whether they were meaningful. Spearman's correlation was used to assess the strength of the relationship between two sets of water chemistry variables. The results of Spearman's correlation range from -1 to 1. The higher the value, the greater the correlation.

Redundancy Analysis

Water chemistry data from all site types were averaged. The data were then adjusted so that different chemicals could be fairly compared, even when measured on different scales. Next, land use around each site was collected and grouped into a few broad categories, such as agriculture, forest, developed land, wetlands, open water, and barren. The second land-use grouping covered specific types of agricultural uses (corn, soybeans, alfalfa, potatoes, total wheat, fallow cropland) and different types of forests and wetlands. Spearman rank correlations were used to identify patterns between land use and water chemistry. To better understand these patterns, the results also indicated how strongly each type of land use was linked to the results and confirm the relationships using a consistent, pattern-based approach.

Spearman Rank Correlation for Pesticide versus Land Use

The pesticide data were simplified for each site in two ways:

- whether any pesticide was detected at all, and
- how many different pesticides were found

If no pesticides were detected, that counted as zero. Field blanks and duplicates were removed from the dataset.

Land use data from 2020-2024 were obtained for the project area and grouped by similar land types. For example, different kinds of corn were combined into one category, and different types of forests were grouped together. After combining the pesticide and land use data, a few sites with insufficient land use information were removed, leaving 37 sites for analysis. Next, land use was compared with pesticide results to look for patterns. The project team examined whether certain types of land use were associated with pesticide detections, and whether sites with higher pesticide levels had different land use than those without. The most common land use types associated with agriculture were used.

Results

Dissolved Ion Relationships

Across all sites (lakes, rivers, groundwater), nitrate covaries strongly and positively with sulfate, chloride, and bromide, and these four anions are tightly intercorrelated among themselves

Figure 17). Fluoride is the consistent outlier, appearing as a band of blue cells in the heat map. Fluoride is negatively correlated with sulfate, chloride, and nitrate.

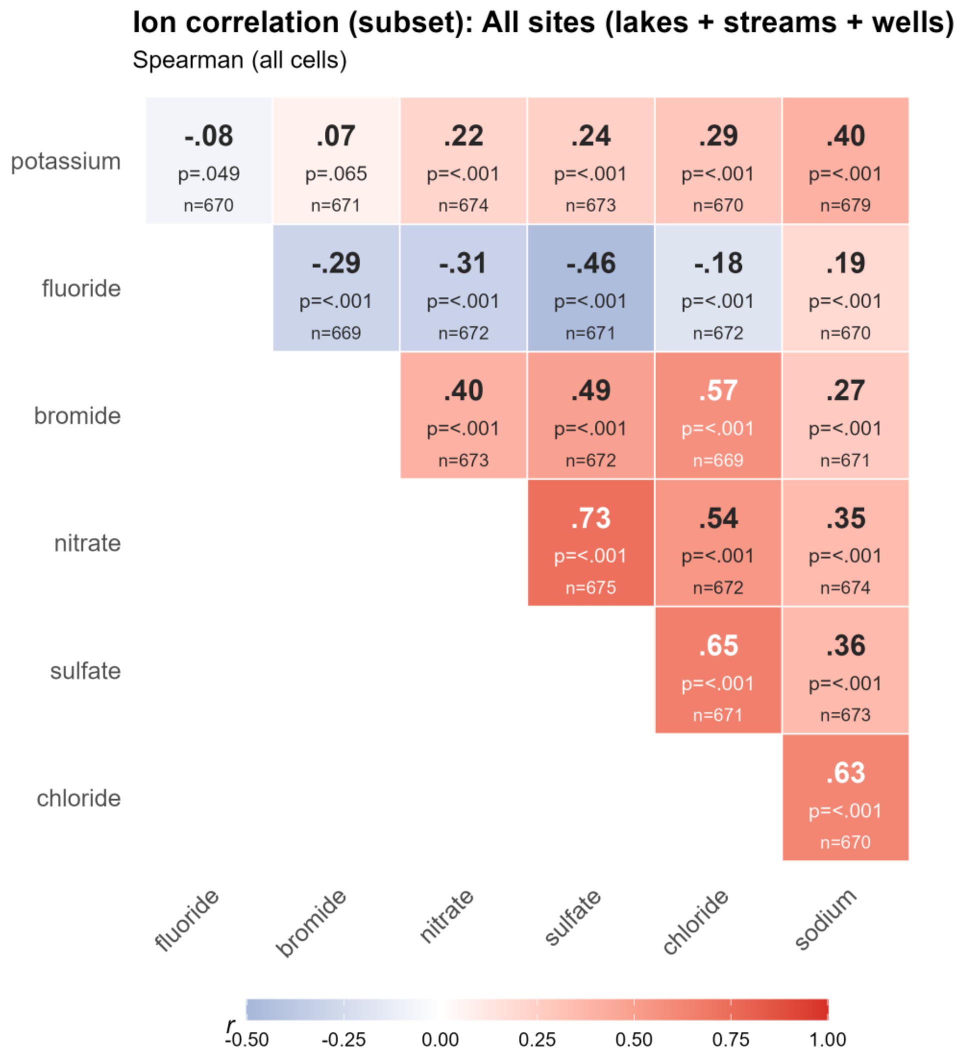

Figure 17: Combined ion vs. land-use Spearman correlation matrix

Redundancy Analysis – Ions to Land Use Comparison

Across the combined dataset, one of the strongest patterns observed was the difference between areas dominated by agriculture and those with more wetlands and open water. Sites surrounded by farmland had a very different mix of chemicals than sites surrounded by wetlands and lakes (Figure 18). A second analysis was conducted to determine whether the results varied by specific crop type. The results in Figure 19 were consistent with Figure 18. The signature of agricultural use in water chemistry results is strong.

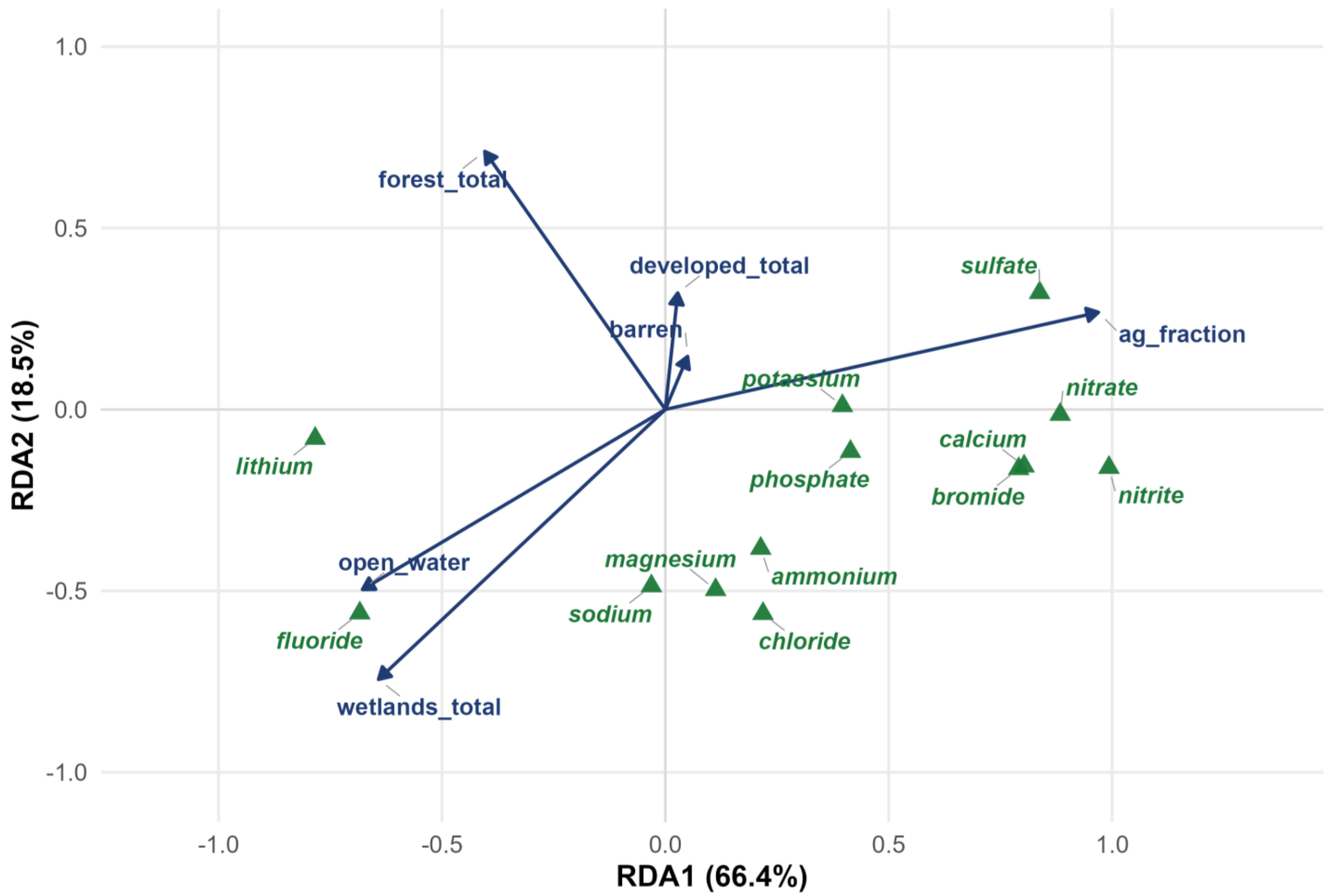

Figure 18: Ions to land use comparison, combined, Redundancy Analysis Triplot

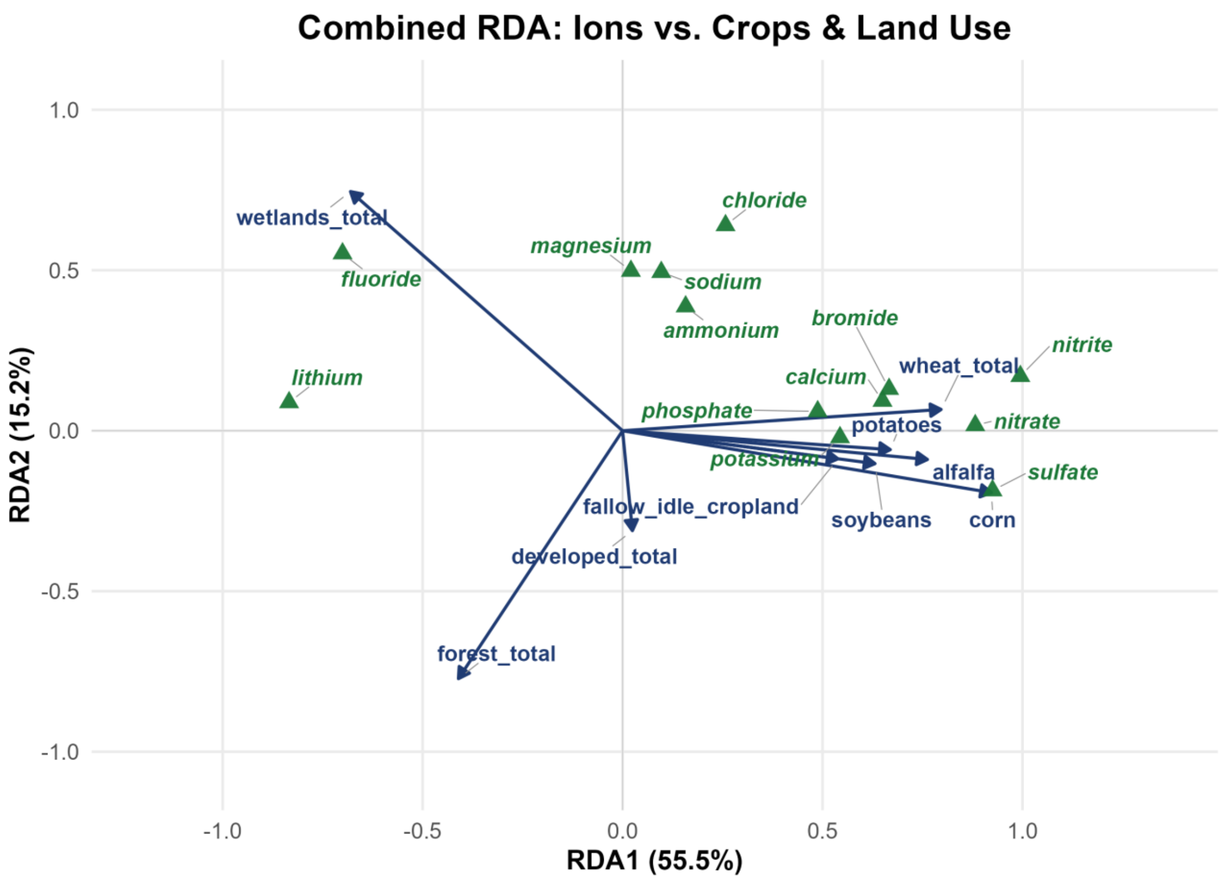

Figure 19: Ions to land use comparison, selected specific crops and land uses, Redundancy Analysis Triplot

Pesticides Compared to Land Use

In the pooled dataset, the results comparing pesticide values to land use indicated that row crops classes were positively associated with pesticide detection (Figure 20). Crops such as potatoes, corn, dry beans, sugar beets, wheat, hay, and soybeans were associated with a higher likelihood of pesticide detection. In simple terms, the more agriculture around a site, the more likely pesticides were detected. These patterns are consistent with agricultural land use being the dominant driver of pesticide occurrence. These patterns are consistent with agricultural land use being the dominant driver of pesticide occurrence. On the other hand, places with more wetlands or open water were less likely to have pesticides. Forested areas did not show a strong pattern either way. This could be because open herbaceous wetlands account for most of the buffering effect rather than wooded wetlands or forests.

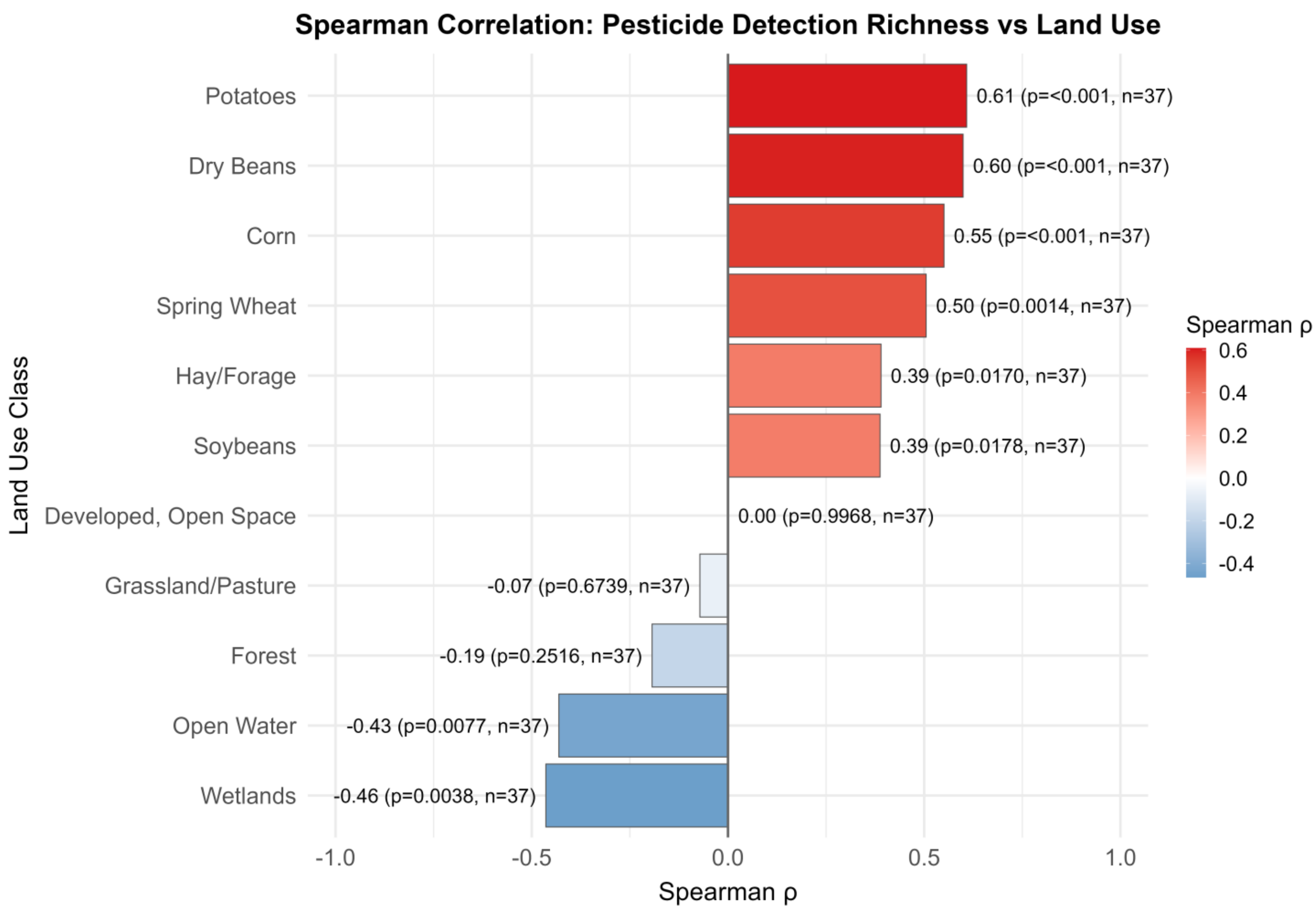

Figure 20: Spearman correlation: pesticide detection richness vs. land use

Conclusions

This water chemistry and land use interactions section offers the following conclusions:

- Agricultural land use was the dominant landscape driver of both ion chemistry and pesticide occurrence across lakes, streams, and wells in the project area.
- Wetlands and open water were least associated with pesticide occurrence, and if they were adjacent to agricultural land use, their role of filtering nutrients and chemicals is critical.
- Row-crop classes (notably dry beans, potatoes, corn, sugar beets, and soybeans) were most strongly associated with pesticide detection.

Water Level and Irrigation Analyses

Groundwater levels are influenced by a variety of interacting factors, including precipitation, recharge, evapotranspiration, aquifer characteristics, land use, and groundwater withdrawals. Evaluating long-term groundwater monitoring records can provide insight into how groundwater conditions have changed historically, while irrigation withdrawal records can help identify temporal patterns in agricultural water use.

The analyses presented in this section evaluate long-term and recent trends in groundwater elevations and irrigation groundwater withdrawals across the study area. In addition, relationships between irrigation groundwater use and nearby groundwater monitoring wells were examined to identify areas where increasing groundwater withdrawals coincide with declining groundwater elevations. Together, these analyses provide a regional assessment of groundwater conditions and potential indicators of groundwater stress within the Pineland Sands region.

Data Analysis

Groundwater Elevations

Groundwater elevation data from 81 MN DNR observation wells were analyzed in R to evaluate long-term and recent trends in groundwater elevation across the study area. Daily groundwater elevation measurements were aggregated into annual mean groundwater elevations for each monitoring well to reduce short-term variability and emphasize long-term temporal behavior.

Groundwater monitoring records extended from 1970 to 2026, allowing evaluation of groundwater conditions both prior to and during the modern irrigation period. Linear regression analyses were performed for each monitoring well, with groundwater elevation as the dependent variable and year as the independent variable. Two separate analyses were conducted: (1) a long-term analysis using the full available monitoring record for each well, and (2) a recent-period analysis using groundwater elevations from 2015 to present. Monitoring wells with insufficient temporal coverage were excluded from trend analyses.

Groundwater trend classifications were based on the slopes of linear regressions. Groundwater trends were classified as strong decline (≤ -0.10), moderate decline (-0.10 to -0.03), stable (-0.03 to 0.03), moderate increase (0.03 to 0.10), and strong increase (≥ 0.10). Maps and interactive visualizations (using Shiny and Plotly in R) were developed to evaluate temporal behavior and assess potential influences of data outliers on trend classifications.

Irrigation Water Use Trend Analysis

Annual irrigation groundwater withdrawal data were analyzed to evaluate long-term and recent changes in irrigation groundwater use across the study area. Irrigation analyses were based on reported groundwater withdrawal records from permitted irrigation wells available between 1988 and 2024 and were summarized annually for each irrigation well using reported annual groundwater withdrawal volumes. Because the dataset reflects available reported withdrawals rather than all historical irrigation activity within the study area, observed trends should be interpreted within the context of the available record.

Linear regression analyses were performed for each irrigation well to evaluate temporal trends in irrigation groundwater use. Similar to the groundwater analyses, both long-term and recent-period trend analyses were conducted. Irrigation groundwater withdrawal trends were classified using linear regression slope values, including strong decrease (≤ -1), moderate decrease (-1 to -0.2), stable (-0.2 to 0.2), moderate increase (0.2 to 1), and strong increase (≥ 1).

Spatial distributions of irrigation groundwater withdrawal trends were visualized using static mapping products generated in R. Aggregate irrigation groundwater withdrawals were also calculated by summing annual withdrawals across all irrigation wells in the study area.

Irrigation–Groundwater Relationship Analysis

Relationships between groundwater used for irrigation and nearby groundwater monitoring wells were evaluated using a custom workflow developed in R and an interactive dashboard. Irrigation wells were spatially paired with MN DNR observation wells within a 3 kilometer (km) buffer using spatial analyses in the simple features package in R.

Data for groundwater extracted annually for irrigation were paired with annual groundwater elevation summaries for each irrigation-monitoring well combination. To emphasize long-term patterns and reduce the influence of short-term year-to-year variability, locally weighted regression smoothing (LOESS) was applied to both groundwater elevation and irrigation groundwater withdrawal time series. LOESS generates a smoothed trendline by fitting localized regressions across subsets of the data, allowing underlying temporal patterns to be evaluated without assuming a strictly linear response. Inverse relationships between irrigation groundwater use and groundwater elevations were then evaluated using correlations between the smoothed annual trendlines. Negative correlations indicate periods where increased irrigation groundwater use corresponded with declining groundwater elevations.

Irrigation wells were subsequently classified according to the strength of inverse relationships identified with nearby groundwater monitoring wells. Correlation classifications included no inverse relationship (correlation ≥ -0.3), moderate inverse relationship (-0.5 to -0.3), strong inverse relationship (-0.7 to -0.5), and very strong inverse relationship (< -0.7). Irrigation wells lacking nearby groundwater monitoring wells within the 3 km buffer or lacking sufficient overlapping temporal data were classified as having no usable nearby MN DNR wells. Spatial distributions of this relationship (between groundwater volume used for irrigation and groundwater elevation) were visualized using static and interactive mapping products.

Results

Groundwater Elevation Trends

Long-term (1970 through 2024) groundwater elevation trends indicate that a majority of monitoring wells exhibited declining groundwater elevations across the study area (Figure 21). Quantitative summaries indicated that 77.4% of monitoring wells exhibited declining groundwater elevation trends over the full monitoring period.

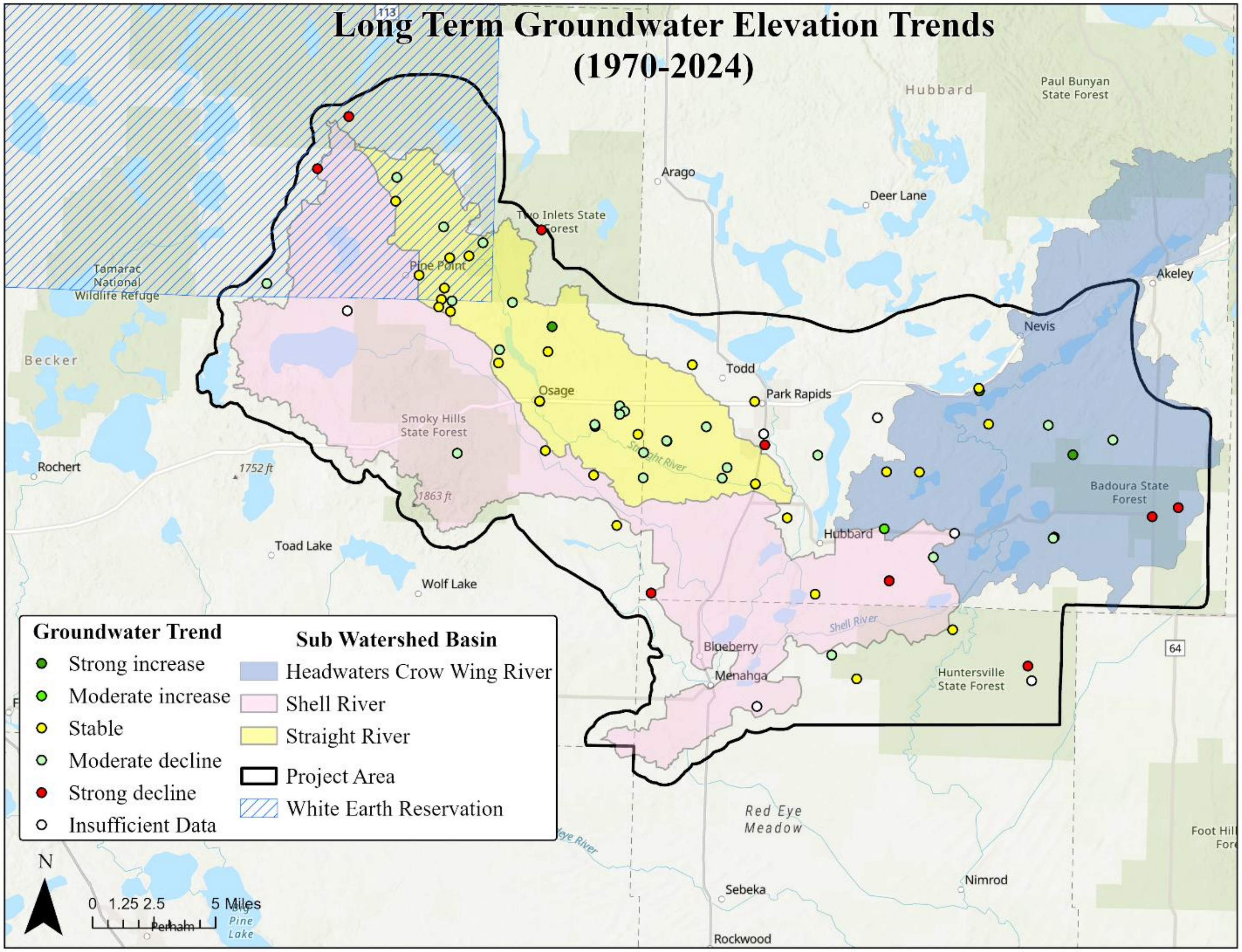

Figure 21: Long-term groundwater elevation trends (1970-2024)

Recent (2015 through 2024) groundwater elevation analyses demonstrated more widespread and pronounced declines (Figure 22). Recent-period analyses indicated that 91.4% of monitoring wells exhibited declining groundwater elevation trends.

Transition analyses comparing long-term and recent groundwater classifications demonstrated substantial shifts toward declining groundwater conditions during the recent analysis period. A total of 27 wells previously classified as stable transitioned to moderate or strong groundwater declines in the recent analysis period. Overall, 43 of 81 monitoring wells exhibited moderate or strong groundwater declines during the recent-period analysis.

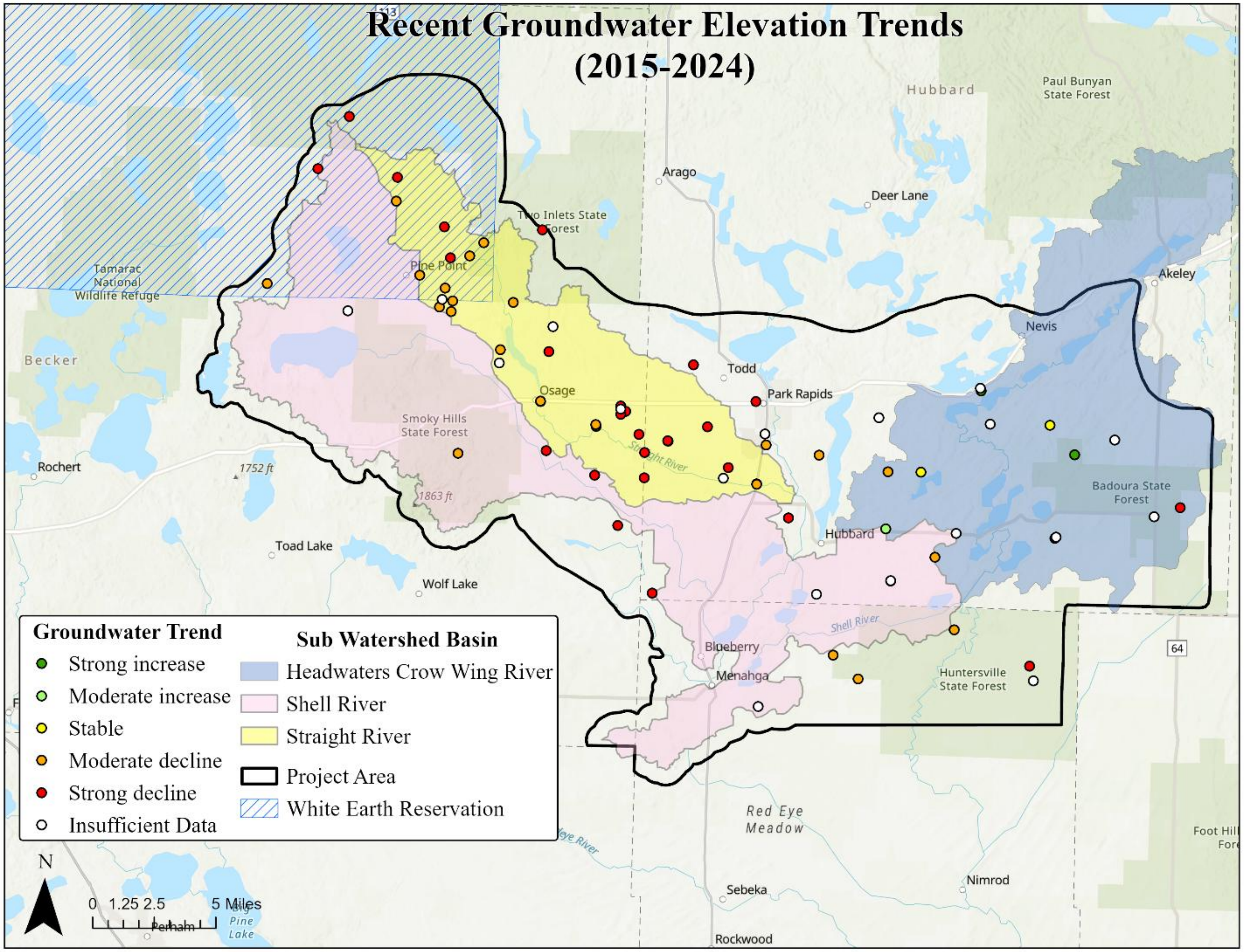

Figure 22: Recent groundwater elevation trends (2015–2024)

Irrigation Water Use Trends

Trends in long-term groundwater withdrawal for irrigation from 1988 through 2024 indicate increasing irrigation groundwater use across much of the study area (Figure 23). Approximately 50.1% of irrigation wells were classified as exhibiting moderate or strong long-term increases in groundwater withdrawals. Recent period irrigation from 2015 through 2024 analyses similarly demonstrated widespread increases in irrigation groundwater withdrawals (Figure 24), with approximately 78.7% of irrigation wells exhibiting moderate or strong recent increases.

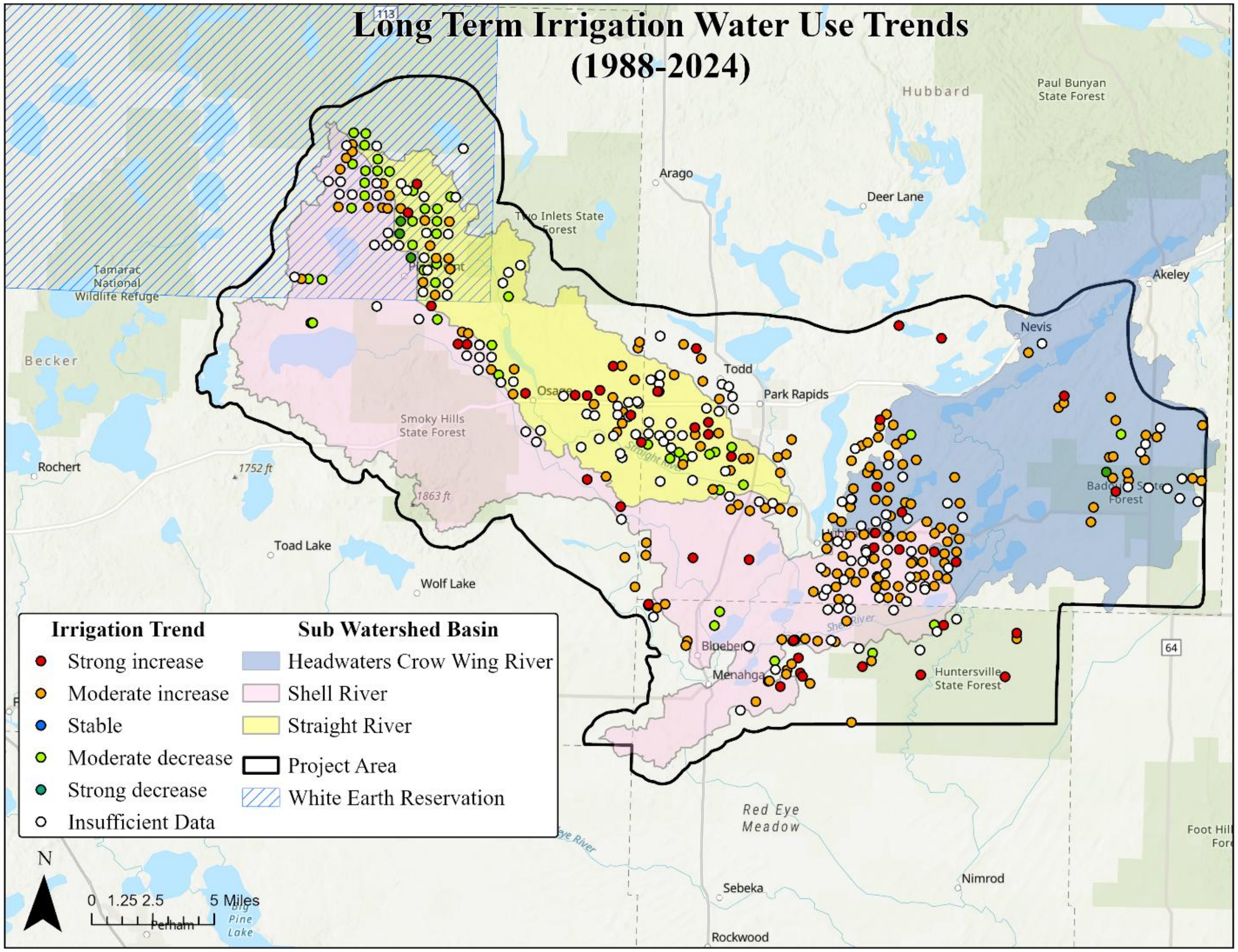

Figure 23: Long-term irrigation water use trends (1988-2024)

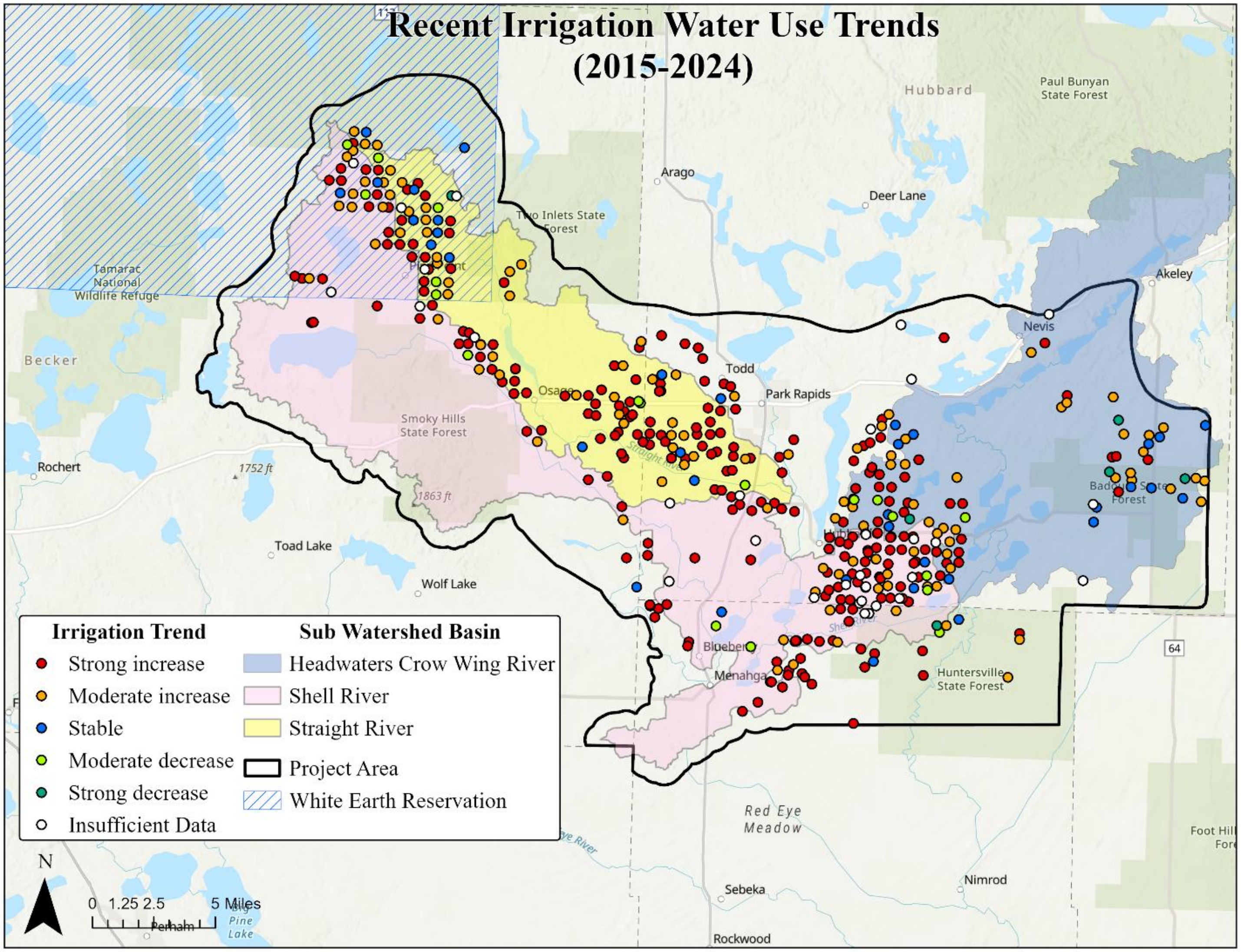

Figure 24: Recent irrigation use trends (2015-2024)

Aggregate irrigation groundwater withdrawals from 1988 to 2024 exhibited a strong increasing trend through time (

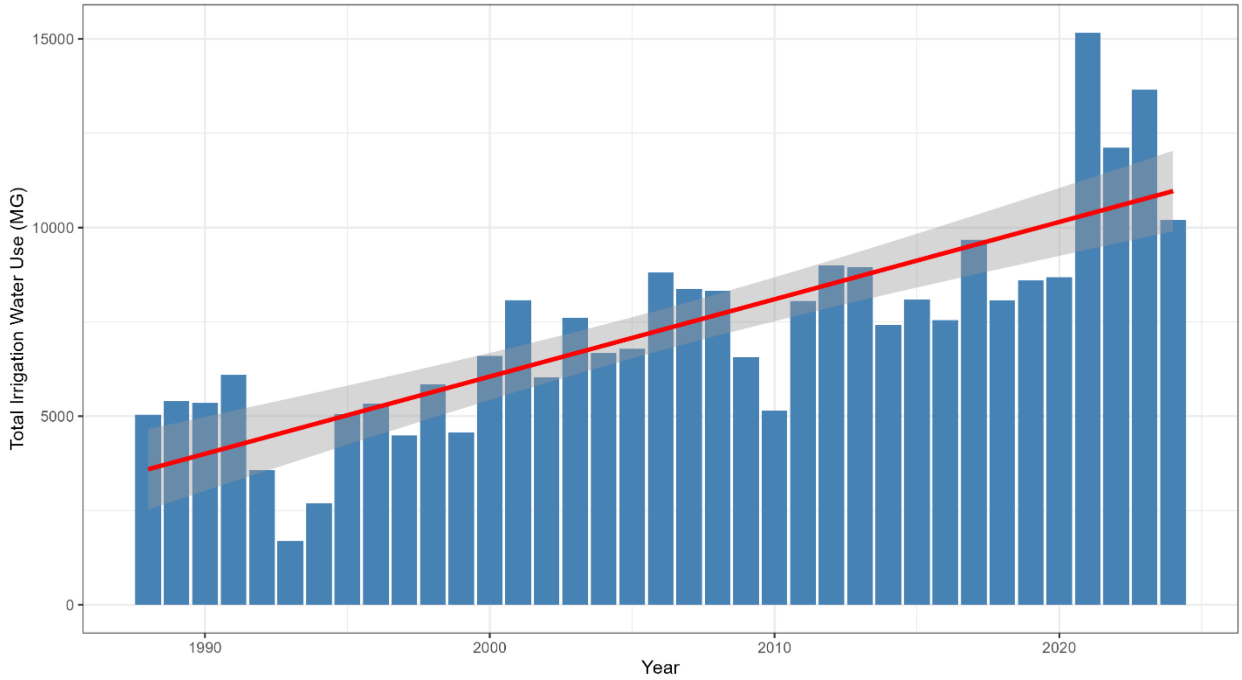

Figure 25), increasing by approximately 205 million gallons per year over the monitoring period. Linear regression indicated that total reported annual irrigation groundwater withdrawals increased by approximately 205 million gallons per year between 1988 and 2024 ($R^2 = 0.655$, $p < 0.001$). This corresponds to an overall increase of approximately 102% across the monitoring period.

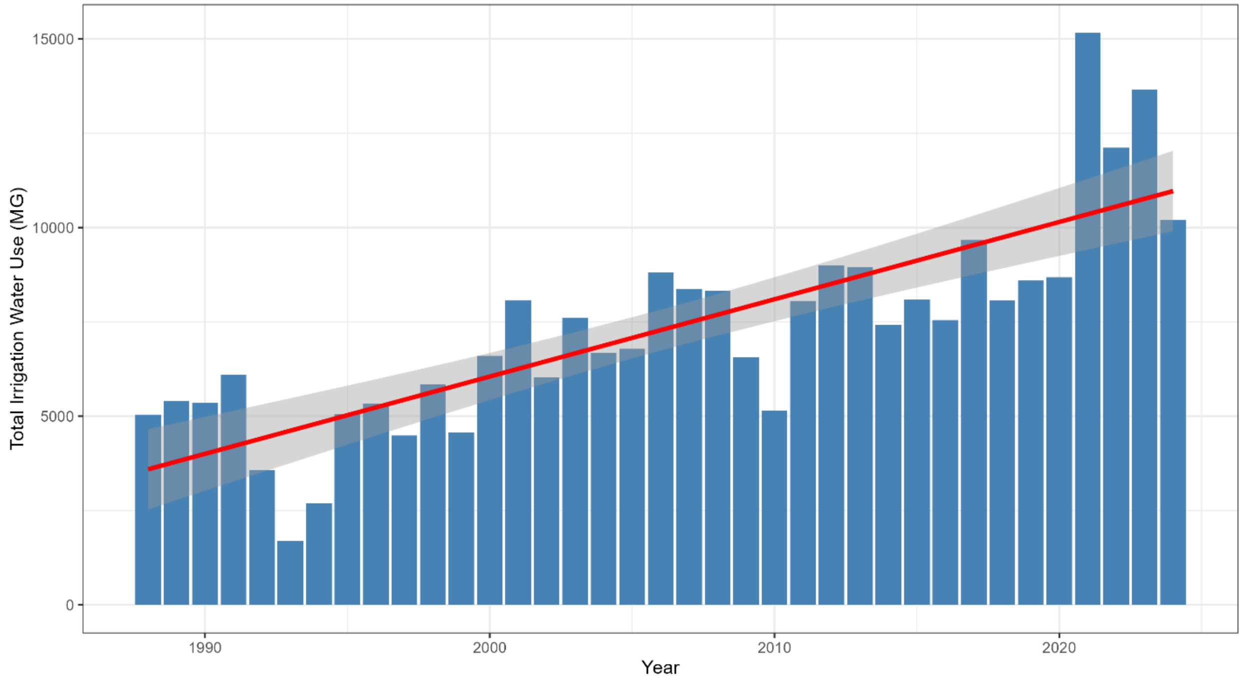

Figure 25: Aggregate annual irrigation water use (1988-2024)

Irrigation–Groundwater Relationship

Spatial analyses evaluating relationships between irrigation groundwater withdrawals and nearby groundwater monitoring wells identified multiple areas exhibiting moderate to very strong inverse relationships (Figure 26). Negative relationships were identified in which increases in irrigation groundwater withdrawals corresponded to declines in groundwater elevations at nearby monitoring wells.

Approximately 35.4% of irrigation wells exhibited moderate to very strong inverse relationships with nearby groundwater monitoring wells, including 43 wells classified as strong inverse relationships and 79 wells classified as very strong inverse relationships. In contrast, 25.6% of irrigation wells exhibited no apparent inverse relationship, while 39.0% of irrigation wells lacked usable nearby groundwater monitoring wells within the analysis criteria.

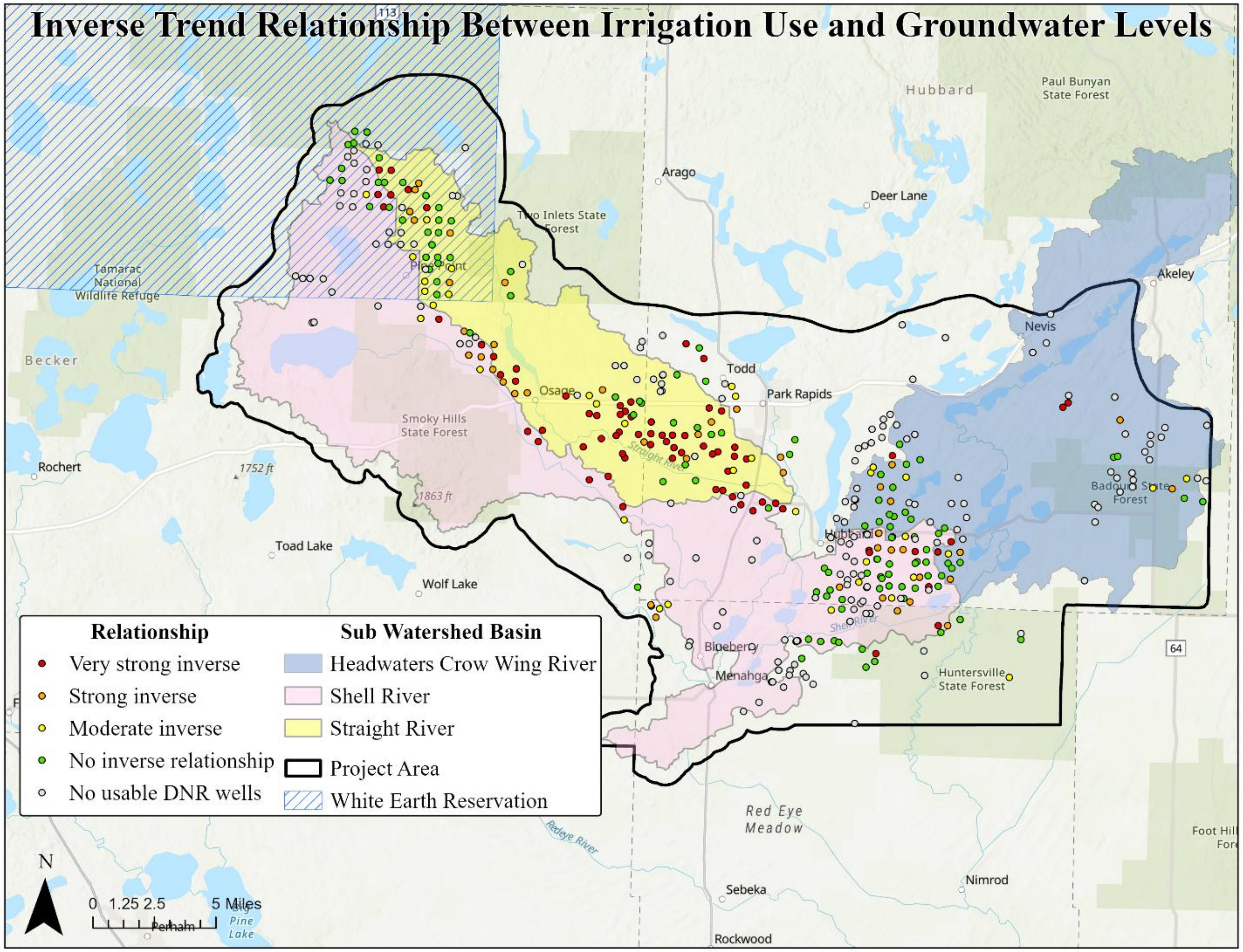

Figure 26: Inverse trend relationship between irrigation use and groundwater levels

Discussion

Groundwater monitoring records within the study area began substantially earlier than available irrigation groundwater withdrawal records, with many groundwater monitoring wells having records dating back to the 1970s. In contrast, records of irrigation groundwater withdrawal primarily begin in the late 1980s and continue to the present day. Consequently, long-term groundwater trend analyses evaluate groundwater behavior over periods longer than those available for direct comparison with irrigation withdrawal records.

The substantial decline in groundwater elevation observed during the recent analysis period (2015 to 2024) coincided with strong increases in aggregate irrigation groundwater withdrawals across the study area. Aggregate irrigation groundwater withdrawals approximately doubled over the monitoring period, while the proportion of groundwater monitoring wells showing declining groundwater elevations increased substantially between long-term and recent analyses. These temporal patterns are consistent with increasing regional groundwater stress during the modern irrigation period.

However, groundwater behavior within the study area is influenced by multiple interacting factors, including precipitation variability, recharge conditions, aquifer properties, crop rotations, and irrigation demand. Agriculture in the region commonly uses rotational cropping, and irrigation demand varies substantially from year to year depending on climatic conditions and crop water requirements. Consequently, inverse relationships identified between irrigation groundwater withdrawals and groundwater elevations should be interpreted as indicators of potential regional groundwater stress rather than definitive evidence of direct causal relationships at individual wells.

Additionally, although linear regression slopes provided useful regional indicators of groundwater behavior, some individual wells exhibited nonlinear temporal responses or sensitivity to outlier observations. Consequently, groundwater trend classifications should be interpreted as broad regional indicators rather than definitive representations of groundwater behavior at individual wells.

8 Surface Water and Groundwater Modeling

The modeling efforts summarized in this section span both surface water and groundwater modeling. The first modeling effort using SWAT+ was built to understand how hydrologic processes have changed from the 1850s to the present day. The second section describes the nitrate source apportionment conducted using Bayesian modeling to understand nitrate sources in the project area. Finally, the regional groundwater and nitrate transport modeling framework builds on the nitrate modeling work done by Margarit et al., (2025) and uses that information, combined with an uncertainty and multi-objective analysis, to assess trade-offs among farmer social equity, protection of nitrate concentrations in waterbodies, and agricultural prosperity. The sections are presented as follows:

1. Assessing the Impacts of Land Use Conversion in the Straight River Watershed using SWAT+.
2. Groundwater Nitrate Source Apportionment in the Pineland Sands Region Using Bayesian Modeling.
3. Regional Groundwater and Nitrate Transport Modeling Framework.

Assessing the Impacts of Land Use Conversion in the Straight River Watershed using SWAT+

Author: Harprabhjot Kaur Dahliwal, Ph.D. Candidate, University of Minnesota

Introduction and Study Overview

Changes in land use, especially when forests are converted to farmland, have a major impact on how water moves through an area and on water quality. Over the past 300 years, large areas of forests and wetlands around the world have been turned into agricultural land. This shift has significantly changed how natural ecosystems function, mainly because farming and grazing areas have expanded so rapidly since the 1970s (Yohannes, Cotter, Kelboro, & Dessalegn, 2018) (Onstad & Jamieson, 1970) (Lacher, et al., 2019) (Foley, et al., 2005) (Ramankutty, et al., 2018). These land changes have disrupted the natural water cycle. As a result, more nutrients (like nitrogen from fertilizers and animal manure) are lost from the land into nearby rivers, lakes, and groundwater, leading to poorer water quality (Wan, et al., 2014) (Namugize, Jewitt, & Graham, 2018).

Vegetation also plays an important role in how water moves and in water quality (Wan, et al., 2014) (Namugize, Jewitt, & Graham, 2018). Forests and other natural landscapes enable water to percolate into the ground and have deep-rooted plants, whereas crops typically have shallow roots (Li, Si, Zhang, & Miao, 2022) (Zhang, et al., 2022) (Scanlon, Jolly, Sophocleous, & Zhang, 2007). Increased farming—especially when combined with irrigation—can change how groundwater is replenished and how water flows below the soil surface (González Perea, Daccache, Rodríguez Díaz, Camacho Poyato, & Knox, 2018). Farming activities like tillage break up the soil structure create larger channels and spaces that allow water (and anything dissolved in it, like nutrients) to move more quickly through the ground. This speeds up the movement of water and nutrients from the surface into deeper soil layers and groundwater (González Perea, Daccache, Rodríguez Díaz, Camacho Poyato, & Knox, 2018).

Another important effect of agriculture is on nitrogen cycling. Modern farming relies heavily on fertilizers, which add large amounts of nitrogen to the soil. However, crops may not take up all the nitrogen, especially if over applied, and any leftover nitrogen can escape into the environment by washing into rivers, seeping into groundwater, or being released into the air (Ali, et al., 2025) (Bouwman, et al., 2013). More nutrients on land that move through water columns more quickly end up in groundwater and surface water sources, increasing the amount of nutrients in water bodies (Van Meter K. , Basu, Veenstra, & Burras, 2016) (Jasechko, et al., 2017). High nitrogen levels in water bodies can lead to problems like algal blooms and low oxygen levels, which harm aquatic life (Van Meter K. , Basu, Veenstra, & Burras, 2016) (Diaz & Rosenberg, 2008).

The Straight River watershed in Minnesota is an example of a vulnerable agricultural area in which increased agricultural intensification has impacted the hydrology and water quality. Over the past three decades, farmers have begun growing irrigated row crops such as corn, soybeans, and potatoes. These changes have raised concerns about groundwater quality, nitrate leaching, and nutrient runoff in surface water in the watershed (Stroom, 2024). The watershed is particularly important because it supports one of Minnesota's premier cold-water trout fisheries, which is strongly dependent on groundwater discharge. In addition to its ecological importance, the watershed contains aquatic resources of long-standing cultural significance for Tribal Nations and local communities. Concerns have been raised regarding how historical land use changes may have altered the environmental conditions that support resources associated with treaty-reserved hunting, fishing, and gathering rights.

Most research done in this area has focused only on recent decades, without considering what conditions were like before large-scale human changes. Without a historical baseline, it is harder to fully understand how much human activity has altered watersheds over time. To address this, this study compares three different time periods: before large-scale settlement (1850s), mid-period agricultural development (2006)[6], and intensive agricultural development (2022). The goal is to understand how land-use changes over time have affected water movement and nitrogen pollution in the Straight River watershed. The SWAT+ modeling tool was used to evaluate factors such as how much water enters the soil, how groundwater is recharged, how much water flows in streams, and how nitrates move through both soil and water systems. By combining historical land-use information with scientific modeling, the study provides a more complete picture of how increasing agriculture has changed the watershed over time.

The study aims to

1. Quantify changes in how water moves into the soil and replenishes groundwater due to land-use changes,
2. Evaluate where and how much nitrate is moving through the soil and contaminating groundwater, and
3. Understand how changes in water movement and farming practices together affect nitrate pollution in groundwater and surface water.

Study Area

The Straight River watershed measures 21,532.41 hectares and is located in in Hubbard County, Minnesota near Park Rapids. The elevation is the watershed ranges from 503.23 to 421.23 meters (Figure 27)**Error! Reference source not found.**. The Straight River watershed is part of the Upper Mississippi River basin. The underlying geology is the area is characterized by sand and gravel (see Geology and Hydrology for additional details). Because cultivated crops are a significant land use in the study area (Figure 3), this is an ideal setting for evaluating the long-term impacts of land use change on hydrology and the transport of nitrogen.

[6] The CDL 2006 land use dataset was selected based on the first available date of use.

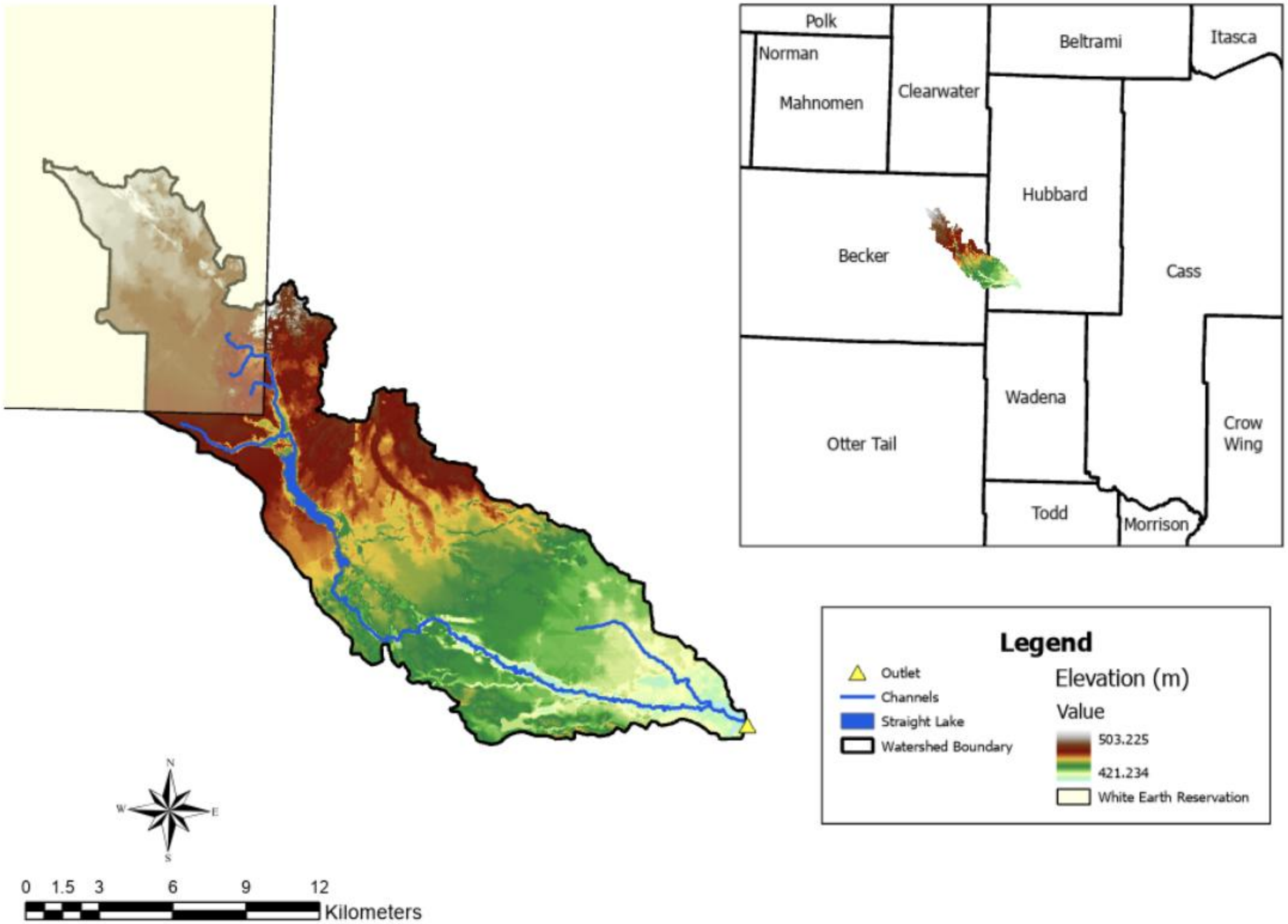

Figure 27: Straight River watershed

Methodology

SWAT+ is a widely used modeling software that can simulate how water and nutrients move through a watershed (Arnold, et al., 2012). The model breaks the watershed into smaller subcatchments based on land use, soil type, and slope to better represent how different areas behave. Inputs to this model require high-resolution data on land use (such as soil type, land use, and elevation) and weather (such as rainfall, temperature, and wind).

To ensure the model's accuracy, it was calibrated and tested using observed data, including meteorological records and streamflow and water quality (e.g., nitrate concentration) data collected within the watershed. The model performed reasonably well in predicting streamflow but was less accurate in predicting nitrate because long-term nitrate data were limited.

Three land-use scenarios were developed to evaluate the long-term effects of watershed development on hydrologic and nitrogen dynamics: one scenario modeling 1850s conditions, one for 2006 conditions, and finally one for 2022 conditions. Once the model was calibrated using recent conditions (2022), the same settings for soil type, topography, and river morphology were applied to two past land-use scenarios (2006 and the 1850s). This allowed the research team to isolate the effects of land-use change over time, without other factors interfering. Overall, the model was considered sufficiently reliable to examine how changes in land use have affected water movement and nitrate contamination in the watershed over time. More specific details on methods and model calibration can be found in Report H.

Results

Land use within the Straight River watershed has changed drastically from the pre-settlement period (1855) to modern conditions (2006 and 2022), with a shift from natural vegetation to agricultural and developed land uses (Figure 28). The watershed was dominated by forests and wetlands during pre-settlement times. Between the 1850s and 2006, the area was converted from natural vegetation agricultural land. Between 2006 and 2022, agricultural development and conversion of forest to agricultural production intensified. Forest cover continued to decline, with only 38.6% of the original forest remaining in the cultivated systems, and a further transition to pastureland (around 13.5%). Approximately 33.4% of the wetland area transitioned to agricultural land. The results support a shift from natural landscapes in 1855 to a mixed agricultural, land-dominated system by 2022. The significant conversion of forests and wetlands to agricultural land uses provides the context for significant changes in water fluxes, groundwater nitrogen contamination, and nitrogen exportIng from the watershed.

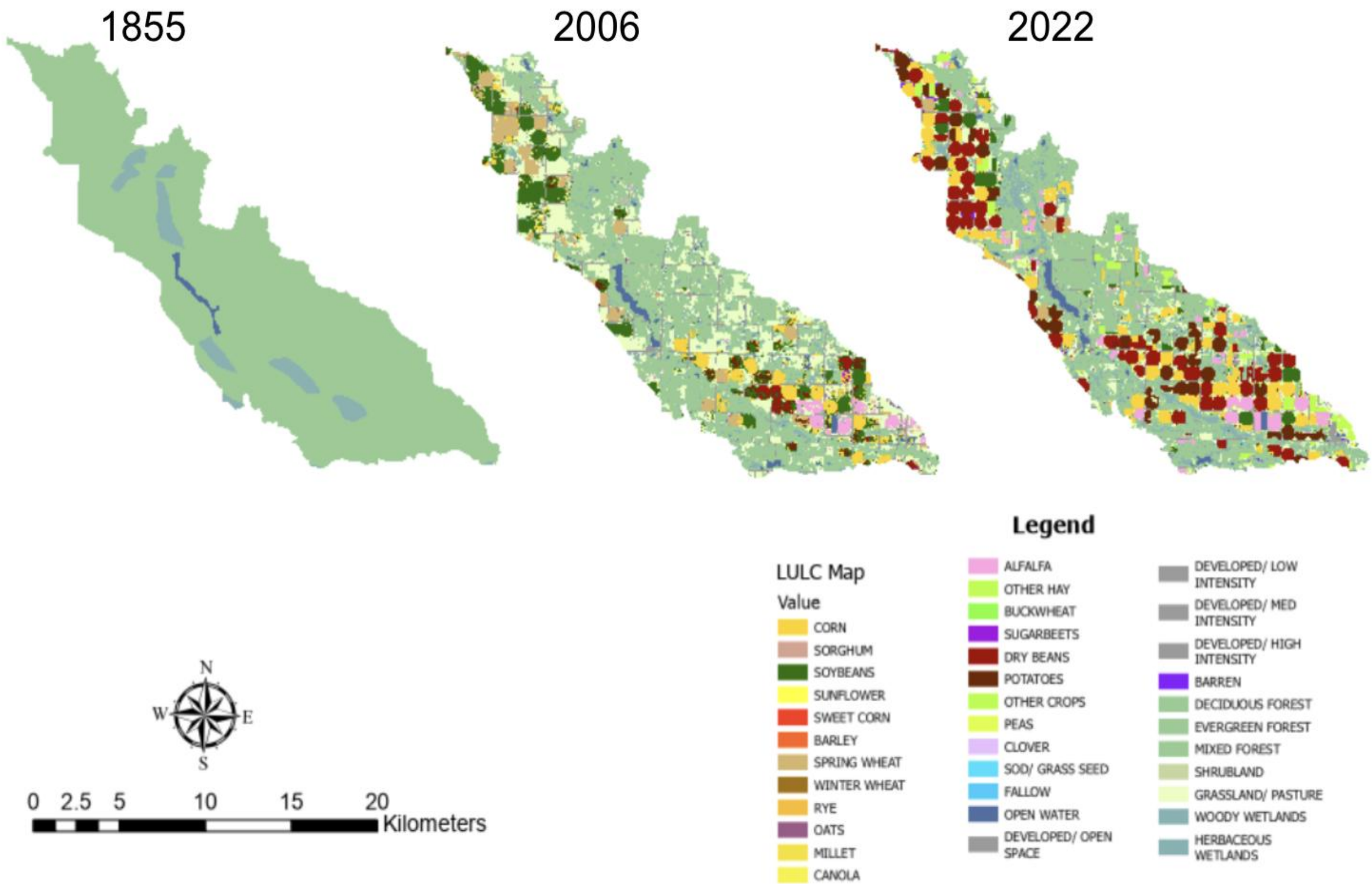

Figure 28: Spatial distribution of land use and land cover (LULC) across three scenarios: 1850s (pre colonial settlement), 2006 and 2022 (modern agricultural use)

Hydrologic Response to Land Use Changes

The SWAT+ model generated percolation rates, or how fast water moves through the soil profile and vadose zone down to the underlying shallow aquifer. It accounts for infiltration, evapotranspiration, soil hydraulic properties, and water movement between soil layers. Under pre-settlement conditions, the mean annual percolation was calibrated to be 117.9 millimeters (mm) (Figure 29). By 2006, this had increased to 206.22 mm, and by 2022, annual percolation had increased to 256.9 mm. The increase in annual percolation rates from 1850 (pre-colonial settlement) to 2022 exceeded 118%.

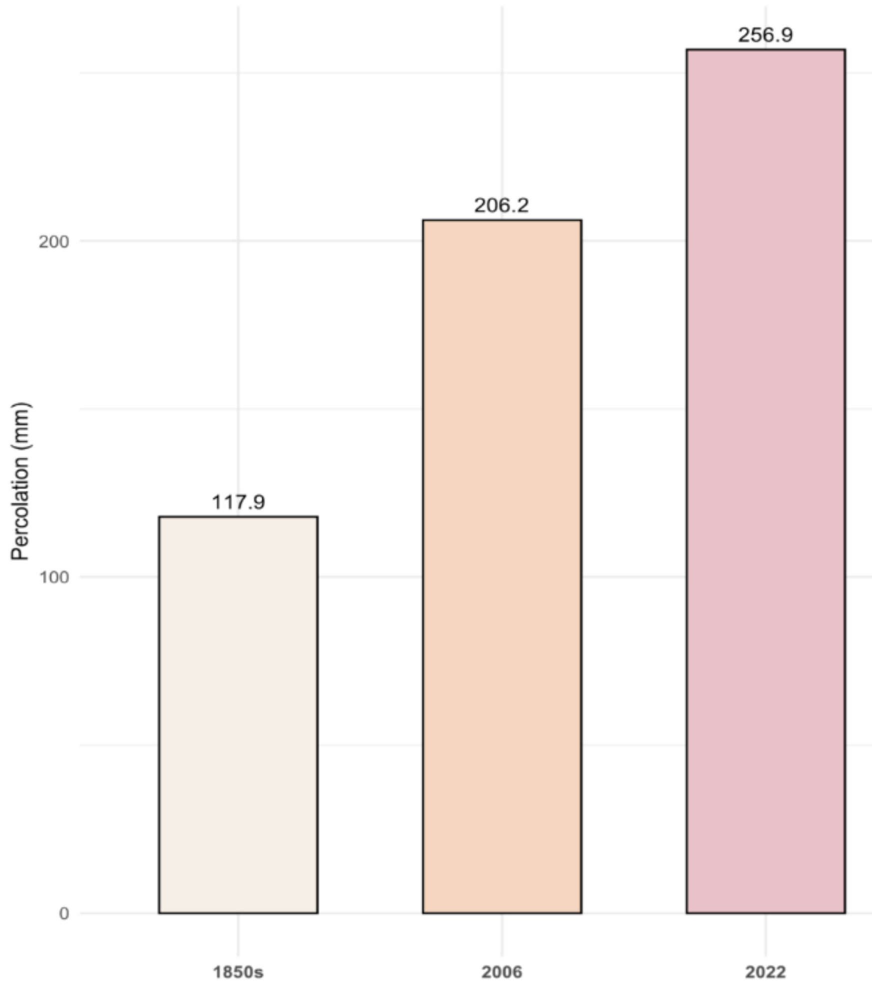

Figure 29: Mean water percolation across land use scenarios

Nitrate Leaching

Nitrate leaching is a measure of how nitrate is lost from the soil column into the underlying groundwater. This value is measured based on kilograms of nitrate leached per hectare (kg N/ha). The SWAT+ model outputs for the 1850s (pre-settlement) were calculated to be 1.14 kg N/ha per year (Figure 30). With settlement and the development of intensive agriculture, the values increased in 2006 and 2022 to 23.5 kg N/ha per year and 32.4 kg N/ha per year, respectively. The results represent a significant increase in nitrate losses from the soil profile: 1,960% increase between the 1850s and 2006 and more than 2,740% between the 1850s and 2022.

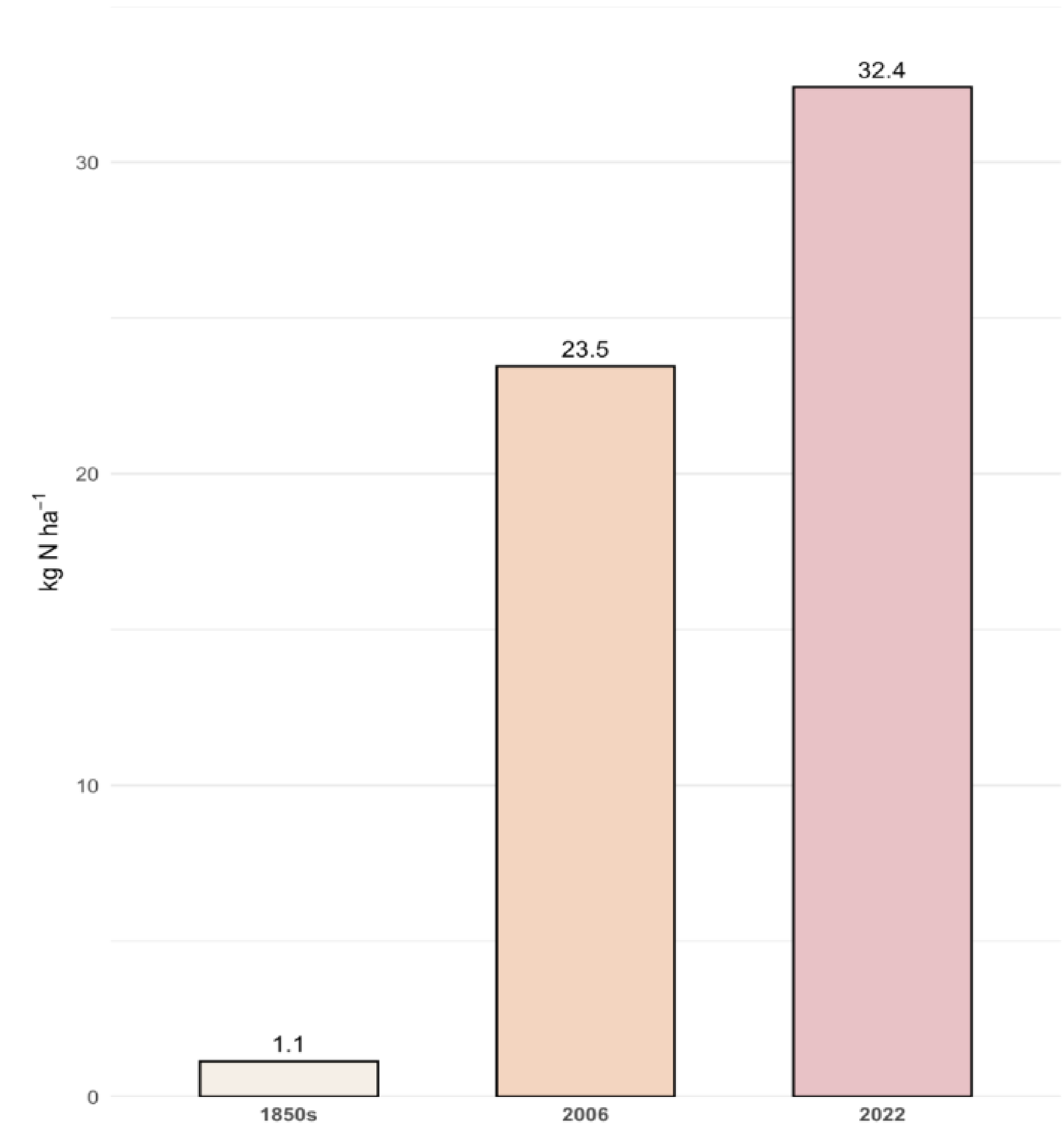

Figure 30: Mean nitrate leaching across land use scenarios, measured in kg of nitrate per hectare per year

Groundwater Recharge and Nitrate Transport

The SWAT+ model outputs included the total estimated nitrate input to groundwater measured in tons of nitrate. According to the SWAT+ simulations the annual nitrate load to groundwater at the watershed scale increased from 24.4 tons in the 1850s to 503 tons in 2006, with a further increase to 694.5 tons in 2022 at the watershed scale, as shown in Figure 31.

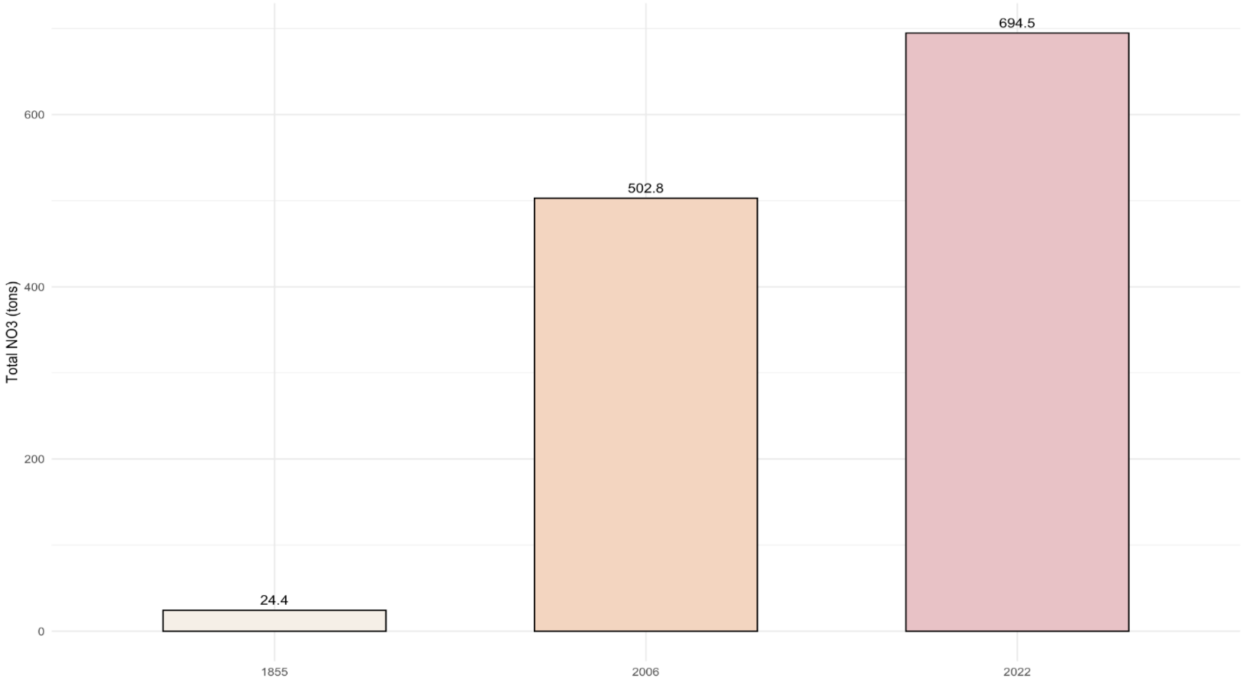

Figure 31: Total nitrate input to the groundwater

Streamflow and Nitrate Transport

Mean annual streamflow, or amount of water passing through a river, is measured as a volumetric rate per unit time. From the 1850s to 2006, the measured streamflow rate increased from 0.8 meters cubed per second ($m^3 s^{-1}$) to 1.4 $m^3 s^{-1}$and further increased to 2.1 $m^3 s^{-1}$ in 2022, as shown in Figure 32. The increase equates to a 68% increase in streamflow between the 1850s and 2006, and a 147% increase between the 1850s and 2022.

In parallel, nitrate export via the flow in the Straight River, quantified in tons per year, followed a similar trend (Figure 32). In the 1850s, the annual nitrate load at the watershed outlet was 5.3 tons, increasing to 20.1 tons in 2006 and then to 119 tons in 2022. This is an increase of over 264% between the 1850s and 2006 and more than 2,045% by 2022, highlighting a significant increase in nitrate export from the watershed. These results highlight the strong relationship between land-use change and increased nitrogen transport and the effects on downstream water quality. The amount of nitrate exported through the Straight River is found to be much smaller than the calculated input to the underlying groundwater. This indicates that significant amounts of legacy nitrate are present in the aquifer and will migrate over time. Over time, the amount of nitrate exported is expected to continue increasing as long as nitrate continues reach the PSA. Issues related to legacy nitrate have been analyzed and presented by a number of publications, including (Puckett, Tesoriero, & Dubrovsky, 2011) (Van Meter K. , Basu, Veenstra, & Burras, 2016).

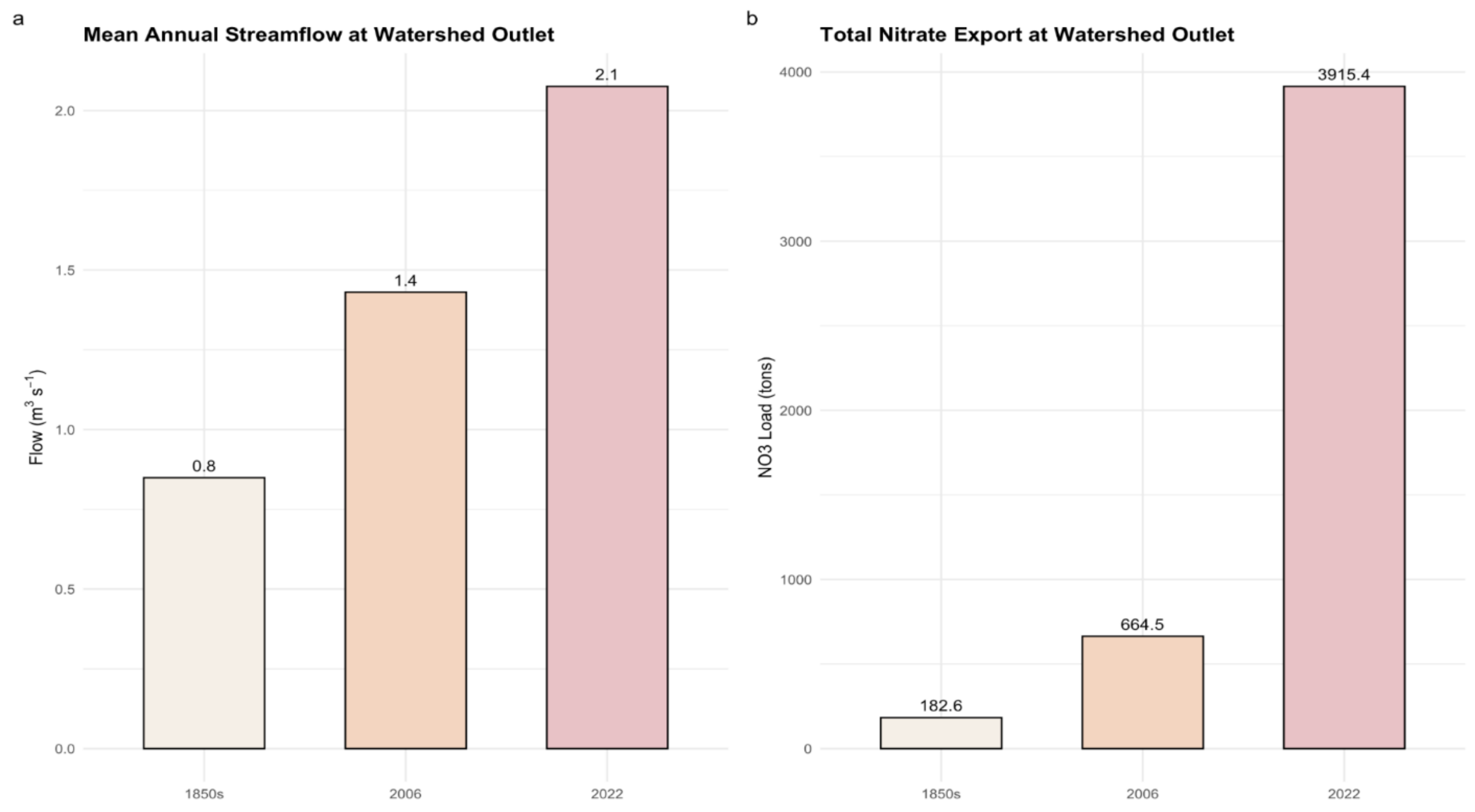

Figure 32: Mean annual stream flow at the outlet of the Straight River watershed (Left side). Total nitrate export at the Straight River watershed (Right side).

Discussion

By comparing past and present land-use conditions and utilizing SWAT+ models, this study showed that land-use changes, increased water movement, and declining water quality are closely connected. Consistent with the studies about water movement and nutrient transport in agriculturally dominated watersheds, this analysis showed that historical land use and conversion to more agriculturally dominated land use have significantly altered how water and nutrients move across the landscape (Foley, et al., 2005) (Allan, 2004). One of the most significant hydrologic changes observed was the more than twofold increase in water percolation rates between the 1850s and 2022, from 118 mm to 256.9 mm, respectively (Figure 29). This increase indicated greater downward movement of water through the soil profile under land use dominated by agriculture.

Groundwater recharge rates also increased, which supports that replacing the deep-rooted perennial vegetation with annual crops reduces the evapotranspiration and results in more water available for downward movement to percolate and ultimately increases the groundwater recharge (Scanlon, Jolly, Sophocleous, & Zhang, 2007). However, increased recharge does not necessarily indicate increasing groundwater storage or rising groundwater levels. In the Straight River watershed, groundwater levels have been declining in some areas because of increased groundwater withdrawals for irrigation. Although recharge increased under agricultural land use, groundwater level response depends on the balance between recharge and groundwater withdrawals. Therefore, increased recharge can occur simultaneously with declining groundwater levels in areas of intensive irrigation.

The rise in nitrate leaching is consistent with studies showing that nitrate leaching globally is strongly linked to intensive agriculture (Di & Cameron, 2002). In this analysis, an increase of 2,740% was observed between the 1850s and 2022 (Figure 30). These findings are also consistent with the removal of forests, which serve as nitrogen sinks and ultimately lower nitrogen losses (Yousaf, et al., 2021).

Long-term analysis in Midwestern U.S. basins has shown that the transition to agriculture results in decreased evapotranspiration, increasing the period of streamflow and groundwater recharge (Scanlon, Jolly, Sophocleous, & Zhang, 2007) (Schilling, Jha, Zhang, Gassman, & Wolter, 2008). Under native vegetation conditions, groundwater recharge in northern Midwestern regions primarily occurs during snowmelt and early spring (March–April), before vegetation is fully active. The increased water uptake by plants results in a significant decrease in groundwater recharge once perennial plants develop leaves and transpiration increases in late spring. Row crops such as corn and soybeans are in the early growth phases during this period and do not reach their peak transpiration until later in the season. Agricultural fields remain hydrologically active for longer, allowing recharge to continue into spring and early summer.

Longer periods of groundwater recharge and nitrate leaching increase groundwater vulnerability to contamination. Consistent with other studies, our study supports the vulnerability of groundwater quality to nitrate contamination in agricultural areas (Nolan, Ruddy, Hitt, & Helsel, 1997) (Spalding & Exner, 1993). Elevated nitrogen inputs linked with agricultural intensification are drivers of increased nitrate loss, resulting in river systems and lakes receiving increased nitrate (Turner, Rabalais, & Justic, 2006).

This study's findings highlight a strong connection between land use change and changes in the movement of water and the transport of nutrients. The capacity of watersheds to retain water and nutrients has decreased due to the conversion of natural vegetation to agricultural land. This conversion to agricultural use increases the likelihood of contamination of surface water, as more nitrate leaves the field and enters downstream waters; contamination of groundwater, as land-use changes lengthen groundwater recharge periods; and more rapid percolation into groundwater. Integrated land management practices and the installation of conservation practices are needed to address some of the consequences of modern agricultural production. Applying the right amount of fertilizer at the right time of year, planting cover crops to keep roots in the soil year-round and adding buffer strips to reduce nitrogen losses are a few ways to minimize nutrient loss.

Groundwater Nitrate Source Apportionment in the Pineland Sands Region Using Bayesian Modeling

Author: Xiang Li, Ph.D., University of Minnesota

Introduction and Purpose

The Pineland Sands region of Minnesota has experienced rapid growth in irrigated agriculture. This expansion in agricultural production is a key driver for regional economic development. However, it has long-term implications for water quality, particularly regarding nitrate contamination. Assessing these impacts is critical for protecting environmental resources.

The purpose of this study is to evaluate whether agriculture practices could be responsible for elevated nitrate levels in the Pineland Sands region using nitrate apportionment. The methods for this analysis were adapted from a regional study conducted in San Joaquin Valley, California, where nitrate contamination in groundwater is severe. Ransom, et al. (2016) evaluated nitrate contamination by allocating groundwater pollution to specific sources with constrained uncertainties and measured how much each source added to the total nitrate pollution. The same approach was applied in the Pineland Sands project area.

One challenge for apportioning nitrate sources is accounting for uncertainty in data quality and quantity and understanding the complexity of the local hydrology. In an agricultural system, the schedule of applying fertilizer impacts the local environment through complex hydrological processes. Any unused fertilizer could leach into the groundwater via variable travel pathways and with different travel time across the project area. Therefore, it is essential to estimate nitrate source apportionment while accounting for uncertainties associated with the available environmental data.

Modeling Approach

Both historical pesticide records and newly collected field data from the Pineland Sands region were reviewed. This analysis was adapted from a regional study conducted in the San Joaquin Valley, California—an area with severe groundwater nitrate contamination—that estimated nitrate source apportionment while accounting for uncertainty (Ransom, et al., 2016). Following a similar framework, the project team applied a Bayesian Markov Chain Monte Carlo (MCMC) approach to estimate nitrate sources. The model is based on a mass balance of analytes, where nitrate observed in groundwater wells is assumed to originate from a mixture of hypothesized sources, each contributing in different proportions.

Data Used for Model

The MCMC model data inputs included historical data sourced from the National Water Quality Monitoring Portal (National Water Quality Monitoring Council, n.d.), which compiles water quality data from federal and states agencies. The historical data was downloaded using the R package called "Tools for Automated Data Analysis" (TADA), which was developed by the U.S. EPA to assist with water quality data download and assessment (U.S. EPA, n.d.). Field data collected between 2024 and 2025 were merged with historical data (Figure 33). To ensure the highest data quality, the project team validated the TADA dataset against other state-sourced water quality data (MPCA, MDA, MN DNR) for version and quality control. As shown in Figure 33, there are 488 sites (in lakes, streams, and groundwater wells) from historical data and 70 field sampling sites. The analyte data from these sites span from the 1970s to 2025.

Data collected during the project period were sampled between 2024 and 2025 and used to enrich historical data records for this study. Field sites were selected based on the stability of the land use over time.

Land use information was also used to characterize land use patterns and define pollution sources' analytes. Land use information was derived from the National Agricultural Statistics Service (NASS) CDL and grouped into eight simplified categories: water, wetlands, nature, pasture, row crops, legumes, small grains, and urban areas (USDA, 2022) (Figure 34). Figure 34 displays general land use patterns in the project area for 2024. Agricultural activities are spatially concentrated along the Straight River, Shell River, and Crow Wing River. These land uses are primarily row crops with smaller patches of pasture, legumes, and small grains (labeled in brown, yellow, gray, and light green, respectively). Outside these regions are major rivers and lakes (open water, labeled in dark blue), nature (mainly forests, labeled in green) and wetlands (labeled in light blue) and additional widely distributed land use groups. Urbanized regions (labeled in red) are scattered in the north and south of the Pineland Sands region, including the cities of Park Rapids and Menahga.

Based on these eight categories, three groupings were developed:

- **Nature** represents primarily forested areas, with more than 50% natural land cover.
- **Agriculture** includes row crops, legumes, pasture, and small grains, with more than 60% of the area dominated by agricultural use.
- **Urban** represents developed land with varying intensity, with less than 20% natural areas.

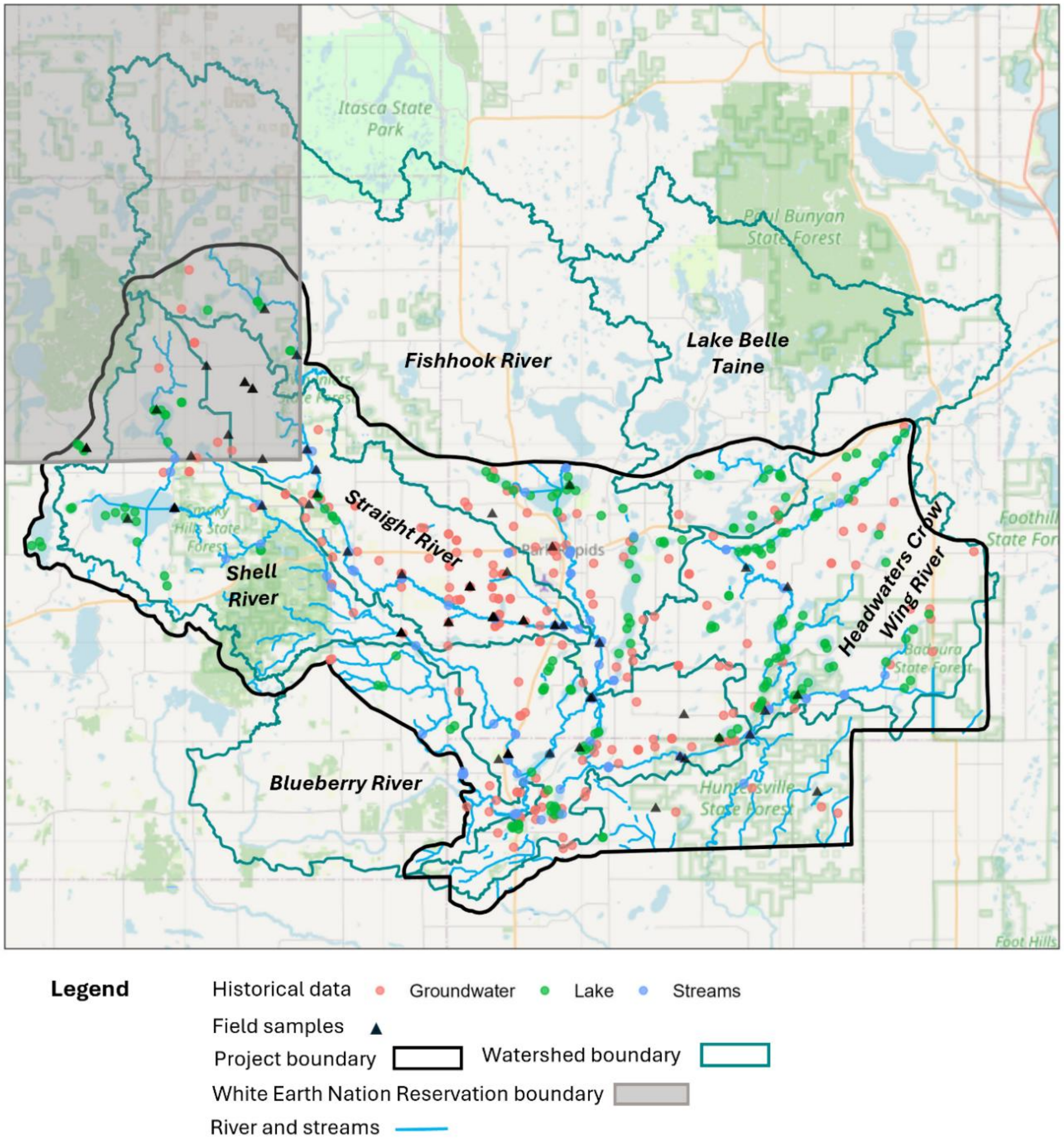

Figure 33: Historical data and recent field sampling locations

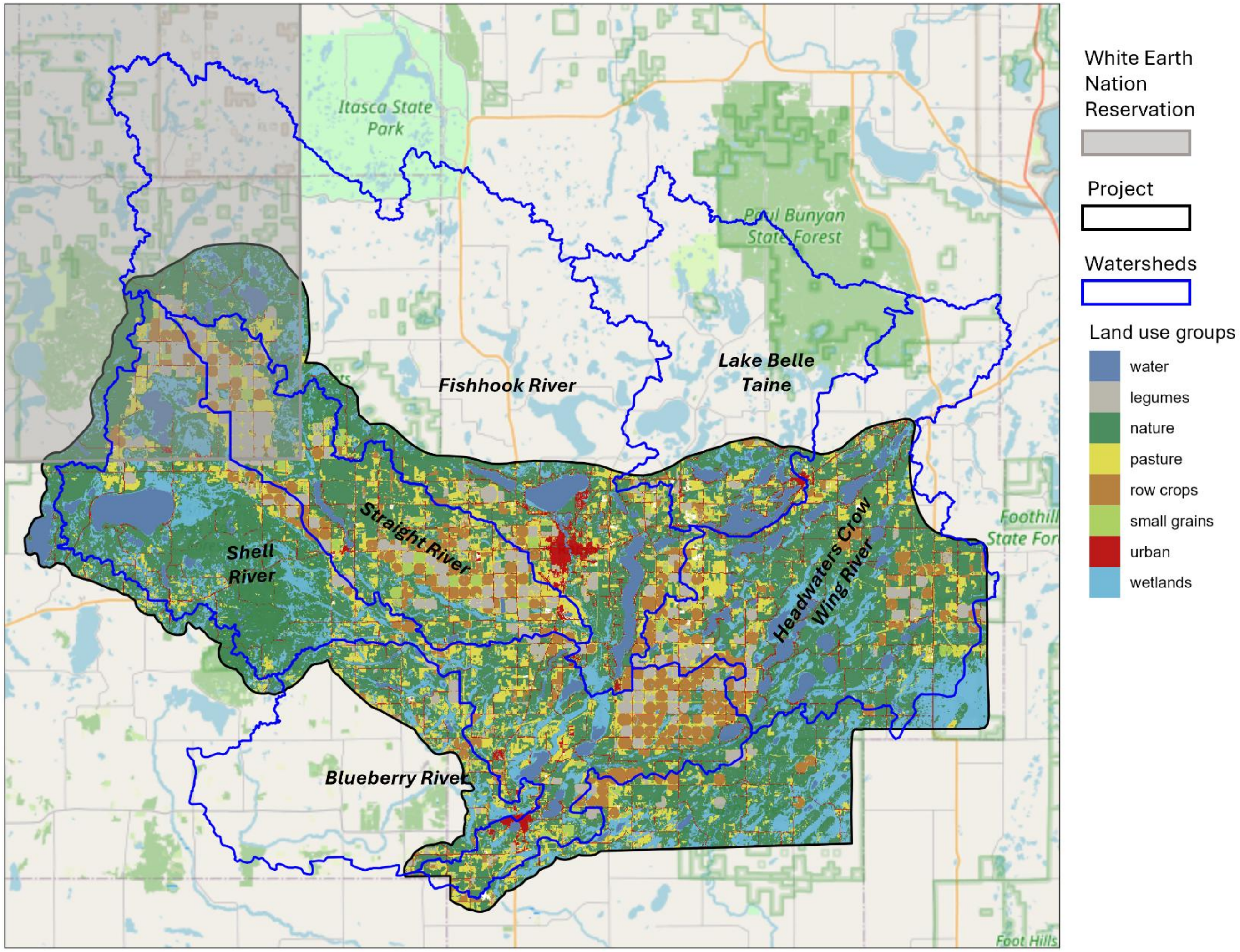

Figure 34: Land use categories

In this study, land use was used as a proxy to represent the dominant contamination sources surrounding each groundwater well. Wells were grouped according to the proportion of nearby land use to identify areas most likely influenced by specific sources.

For background sites, wells were classified based on dominant land cover. Wells with more than 50% natural land cover were assigned to the “nature” group, those with more than 40% wetlands to the “wetland” group, and those with more than 60% agricultural land to the “agriculture” group. These thresholds were selected based on the best available understanding of site conditions while ensuring enough wells for meaningful analyses.

Water Quality Data

The different land use types represent potential pollution sources that may negatively impact groundwater quality. To conduct a preliminary assessment of groundwater nitrate in the study area, water quality data were analyzed alongside pesticide occurrence to identify patterns of co-occurrence. This data informed the modeling scenarios.

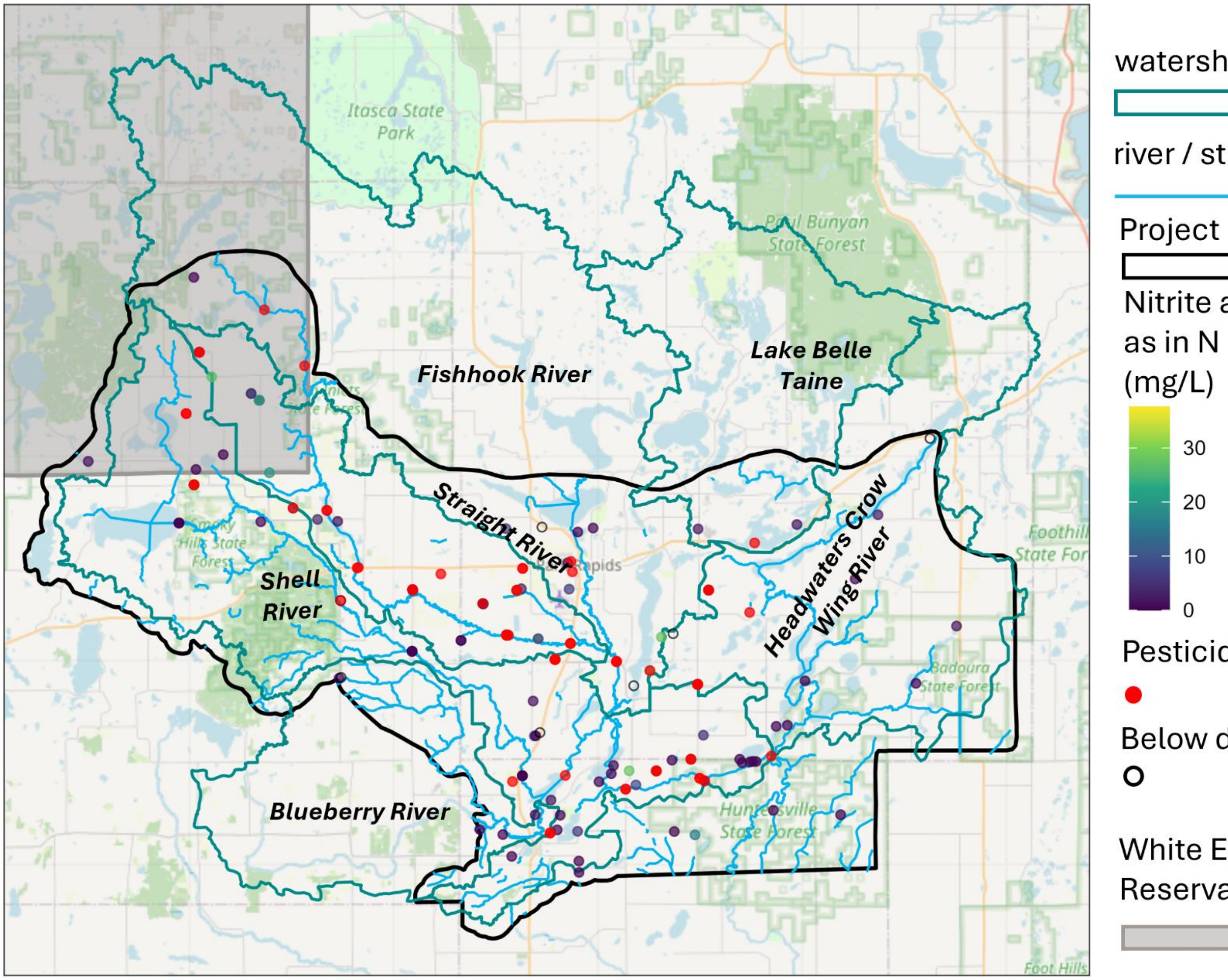

Figure 35 highlights nitrate and nitrite concentrations in groundwater wells alongside sites where pesticides have been detected. Locations with repeated pesticide detections (more than one confirmed measurement above detection limits through 2025) are identified as pesticide sites and displayed as red points. Nitrate and nitrite concentrations shown on the map represent temporal averages for each well between 1970 and 2025. Wells with concentrations consistently below detection limits are shown separately with open symbols; there are five wells with values consistently below detection limits.

Modeling Scenarios

As illustrated in

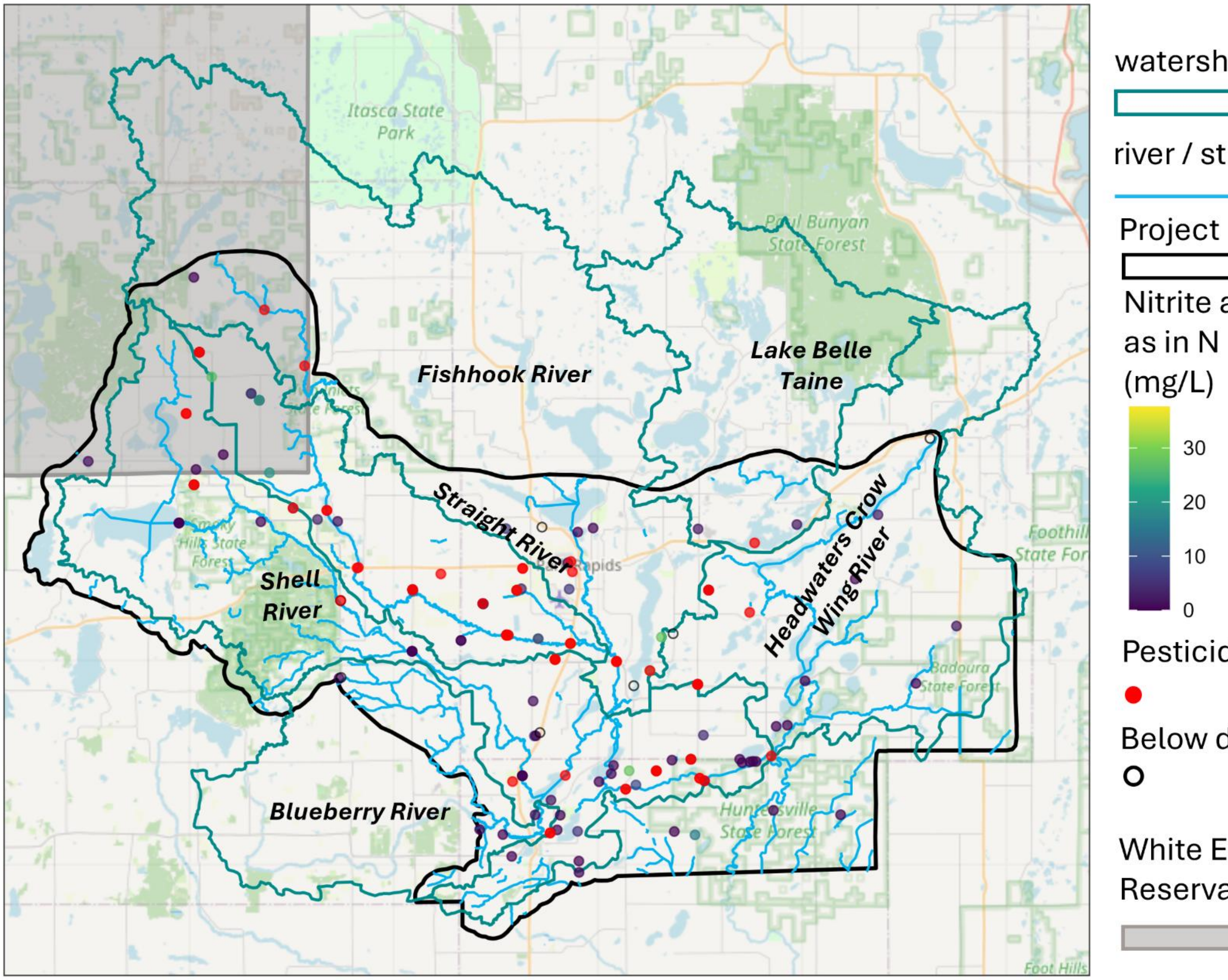

Figure 35, elevated nitrate and nitrite levels were generally co-located with the pesticide presence in the project area. Nitrate and nitrite levels tended to decrease with increasing distance from those locations having detected pesticide concentrates. The pattern between nitrate levels and pesticide presence suggest that contaminant sources and transport processes may be different in scenarios if pesticides are detected or not. Based on this observation, two distinct modeling scenarios were developed in this study, divided based on "impaired" and "background" sites. Groundwater wells located near areas with detected pesticides were defined as impaired sites (shown in blue) (Figure 36**Error! Reference source not found.**). These sites are primarily associated with agricultural activities where pesticides are applied and can be released into the environment, leading to soil and groundwater contamination. The impaired sites are concentrated along the diagonal spanning northwest to southeast across the project area. Wells that are far from pesticide presence are referred to as background sites (shown in black), indicating that their nitrogen sources are predominantly associated with natural background conditions, with minimal influence from agricultural practices.

As shown in Figure 36, impaired sites are groundwater wells that are located inside the 2.4-kilometer (km) radius to represent a buffer (grey circle) surrounding the sites having a detected pesticide presence. The circular buffer was selected to approximate the source of the well contamination and represents an estimate that accounts for pollutant transport including hydraulic conductivity and well characteristics (Horn & Harter, 2009). This buffer was applied uniformly across all sites and therefore does not account for site-to-site variability.

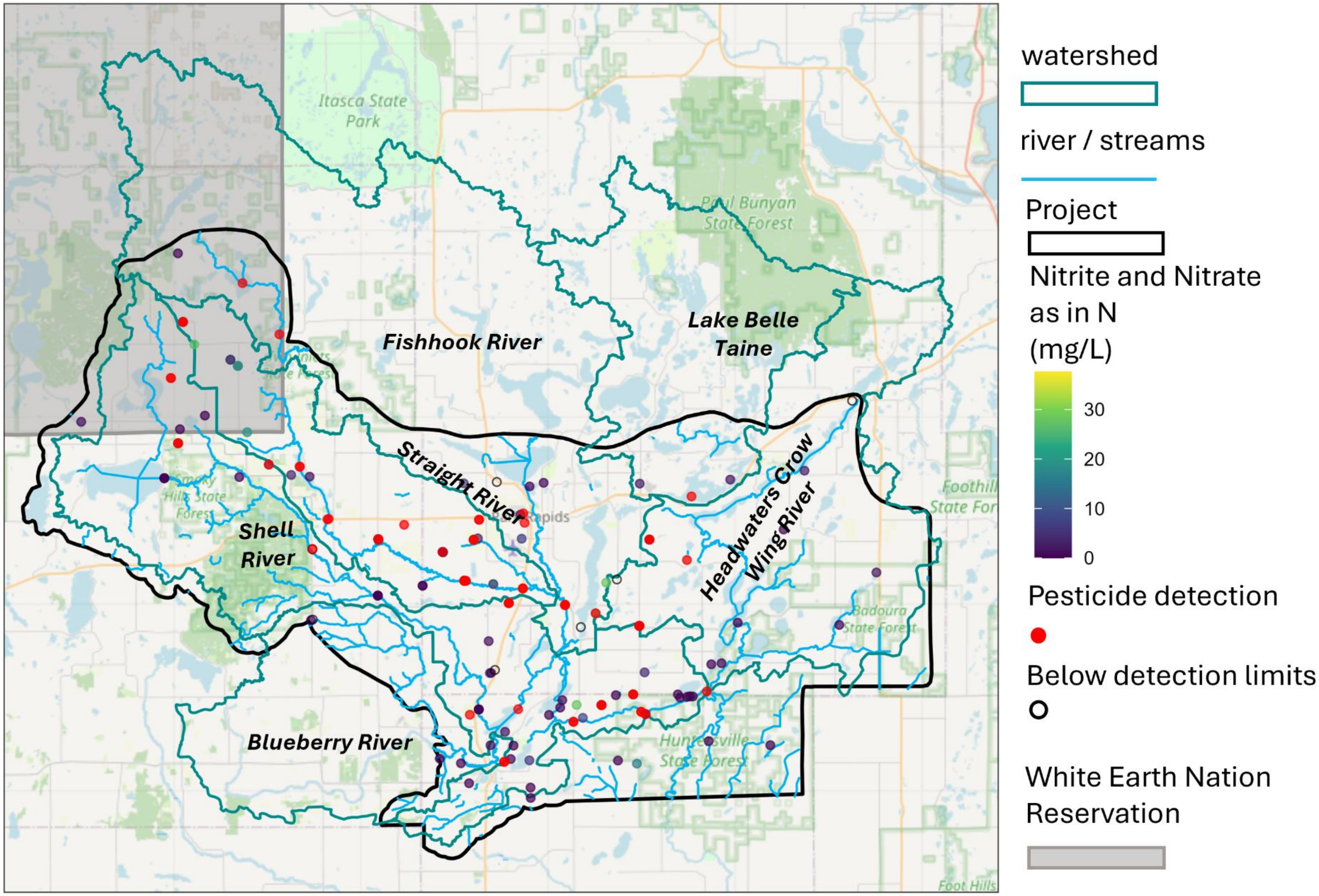

Figure 35: Nitrate and nitrite levels in wells, shown near pesticide source

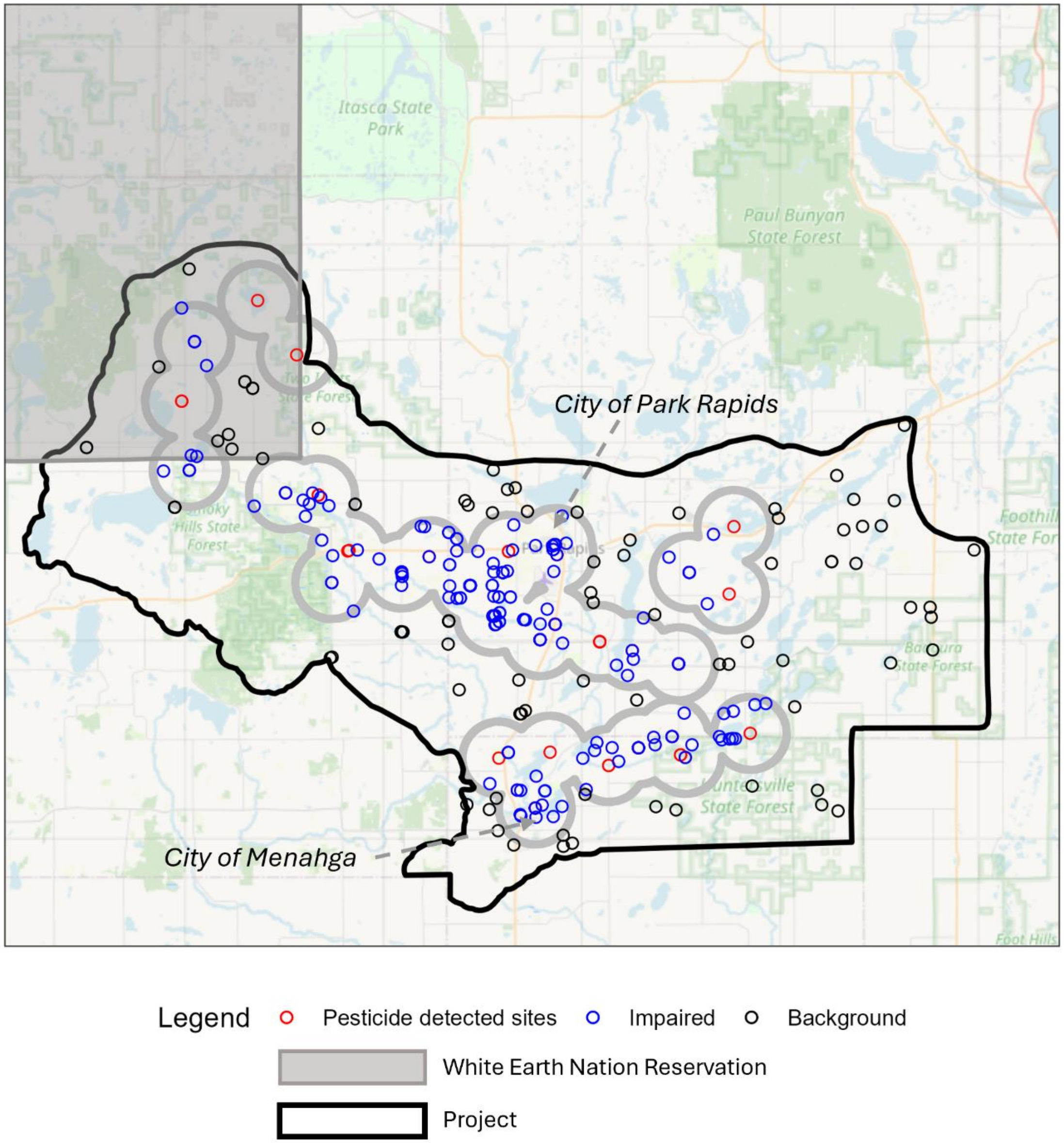

Figure 36: Impaired sites, background sites, and pesticide source locations. The gray lines (circles) indicate the boundaries of buffer zones.

Although nitrate and nitrite are the primary contaminants of concern, they were not used directly in the modeling analyses due to incomplete datasets across wells. Instead, the model relies on more widely available water quality parameters—such as chloride, sulfate, calcium, and sodium—as proxies of contamination sources. Patterns of these parameters are commonly associated with different land uses and activities and tend to occur alongside nitrate. By analyzing patterns across multiple parameters to characterize unique signatures, the model can infer the likely source apportionment of nitrate contamination even when nitrate data are limited.

Modeling Methods

Model Background and Development

As previously described, the methods for analyses were adapted from a regional study conducted in the San Joaquin Valley, California, where nitrate contamination in groundwater is severe (Ransom, et al., 2016). This regional study evaluated nitrate contamination by attributing groundwater pollution to specific sources with constrained uncertainties. It quantified the extent to which each source contributed to nitrate pollution. The project team adapted this statistical approach for the Pineland Sands Region study.

Water chemistry from wells was analyzed for multiple chemical indicators and sources such as fertilizer, manure, septic systems, and/or natural background. Those four signatures were defined using iodine and isotopes in nitrogen, oxygen, and boron. In this project, constrained by available data and land use information, pollutant source definitions were discarded and, instead, sources were characterized to fit the project scope. A Bayesian modeling approach was then used to estimate the contribution of each source to nitrate levels in each well. The results from Ransom et al. (2016) showed that groundwater contamination typically reflects multiple sources rather than a single dominant one and aligns well with known land-use patterns.

In general, the methods in this section are very brief and do not illustrate the full extent of model development. For more details on the Bayesian mass balance model, data requirements for the model, and source characterization for background and impaired sites, refer to the full technical Report I.

The Contribution of Septic

To determine the extent of the impact of septic leaching and potential impacts on the concentration of nutrients found in the surface and groundwater of the project area, nutrient loading to the watershed and to a 3000-ft buffer around the Straight River were calculated. Information on estimated nutrient loading in sandy soils was gathered from published literature

(Margarit, et al., 2025)(U.S. EPA, 2002) (Randall & Mulla, 2001) (MDA, 2015). Private well locations from the County Well Index (UMN, 2020) were used as a proxy for the presence of a septic system. While not all wells correspond to individual onsite wastewater treatment systems, the approach provides a reasonable estimate of the spatial distribution of rural residences and potential septic sources within the study area. This approach may overestimate or underestimate the actual number of septic systems in some locations. However, it is sufficient for comparing the relative magnitude of potential septic and agricultural nitrogen loading at the scale of a river riparian zone.

The following assumptions were made during data analyses:

- A loading factor of 10 – 15 kilograms nitrogen/septic system/year was adopted based on values commonly used in watershed nutrient loading studies and supported by nitrogen production rates reported by the U.S. EPA (2002).
- Agricultural nitrogen loading was estimated using a screening range of 40–60 kilograms nitrogen/hectare/year. This range is consistent with nitrate leaching rates reported for row-crop agriculture on sandy soils in Minnesota and the Upper Midwest by (MDA, 2015).

The project team's estimates indicated that agriculture was the dominant potential source of nitrogen loading in both the river corridor and watershed. Estimated septic system contributions account for approximately 3–8% of total nitrogen loading within the 3000-ft river buffer and 1–3% at the watershed scale. While septic systems appear to be a relatively minor source compared to agriculture, they may contribute localized nitrate inputs due to direct discharge to shallow groundwater. For the purposes of this report, septic systems were not included in this modeling effort.

Results

Results of the MCMC modeling are presented for two groups of wells: background and impaired. For each well, the model estimates the proportion of different sources (e.g., wetlands, agriculture, urban) with all source contributions summing to one (100%). Rather than producing a single value, the model generates a range of possible values for each source, reflecting uncertainty in the data and system.

For the background wells, the model estimates contributions from wetlands, agriculture, and natural sources. Results are summarized using the median of the modeled distribution, with uncertainty represented by the 25th and 75th percentiles. For the impaired wells, the model estimates contributions from row crops, pasture, urban sources, and natural background. In this case, results are summarized using the mean, with uncertainty similarly described using percentile ranges.

Background Sites

As shown in Figure 37, three panels depict the median distributions of estimated source apportionment for wetlands, agriculture, and nature. These results reveal that the majority of the background wells in the study area are from either natural or wetland sources with negligible agricultural impacts. Overall, this estimate suggests that wetlands (0.46) and natural sources (0.45) contribute about the same amount of nitrate and nitrite. For these wells, the estimated contributions from wetlands and natural areas are somewhat uncertain—usually by about ±10% to ±20%, and sometimes up to ±30%. This uncertainty likely happens because the chemical characteristics of natural areas and wetlands are very similar, which makes it harder to tell them apart. Collecting more or more focused samples could help better distinguish these sources and reduce uncertainty, leading to more confident estimates of where the nitrate is coming from.

Impaired Sites

The results for the impaired wells—located near pesticide detections and likely influenced by agricultural activity—show a different pattern than the background sites. In these wells, nitrate contributions from row crops and pasture are relatively high, while contributions from natural and urban sources are much lower. Figure 38 summarizes the median estimates of the four categories (pasture, row crops, urban, and nature) across all impaired wells. Overall, the results indicate that pasture and row crops contribute equally, with median apportionment values of 46% each.

As with the background wells, there is some uncertainty in these estimates. It can be hard to differentiate contributions from row crops and pasture because they often involve similar inputs, which leads to wider ranges in the estimates. While natural and urban sources are generally small for most wells, a few wells still show noticeable amounts from these sources. Urban contributions are usually less than 10% and were mainly found near the City of Park Rapids, where there is more development than in the surrounding areas.

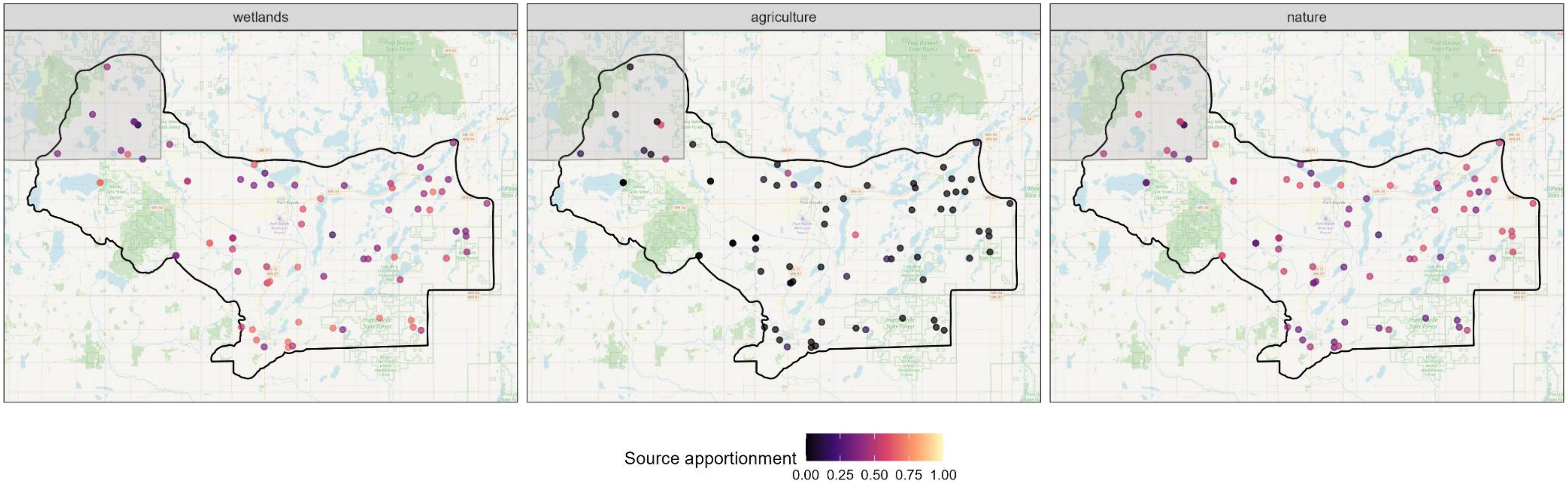

Figure 37: Median of source apportionment distributions for background sites

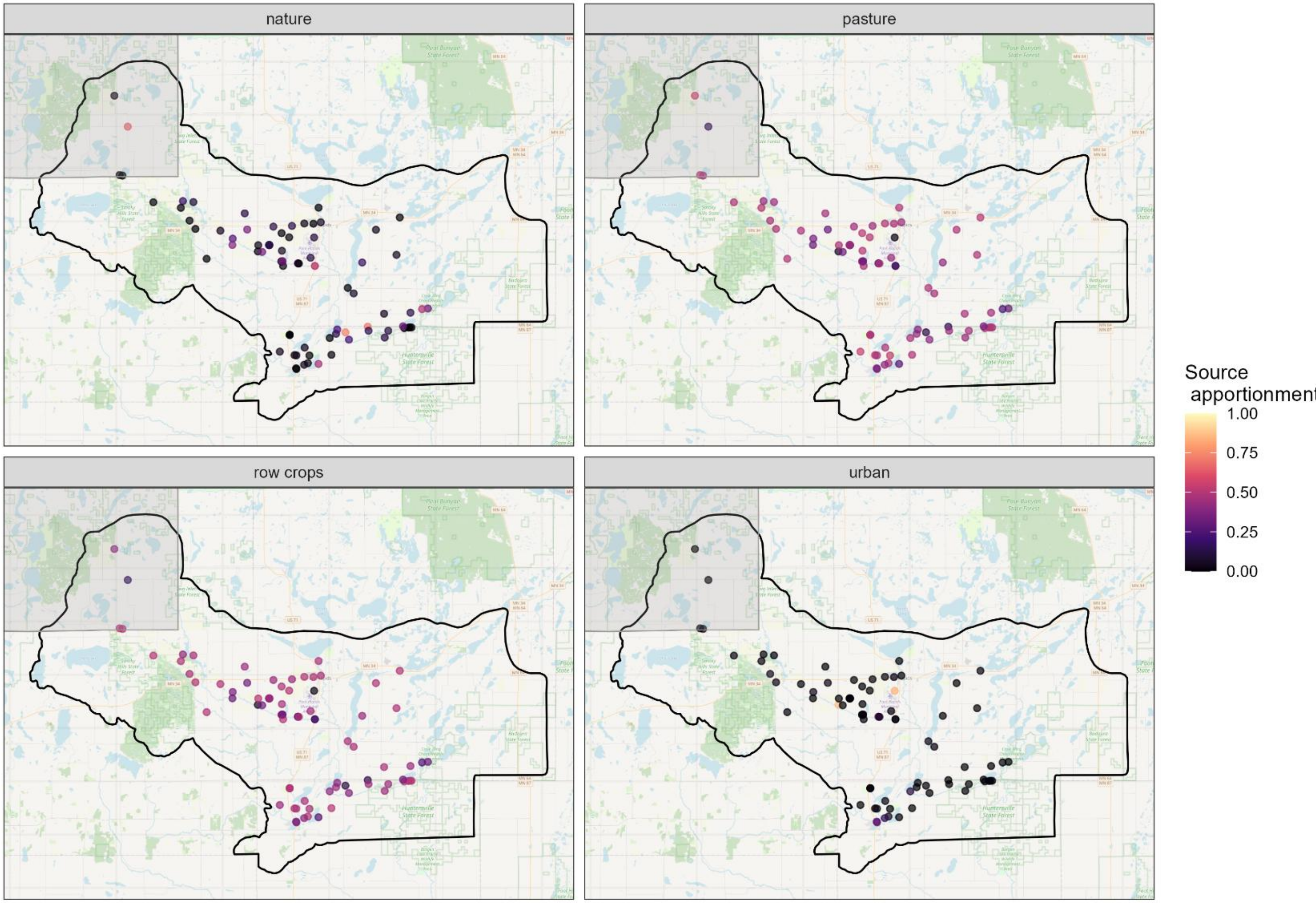

Figure 38: Median of source apportionment distributions for impaired sites

Discussion

The modeling results indicate that groundwater in the Pineland Sands region shows clear differences in nitrate sources with proximity to agricultural activity. As shown in Figure 39, background wells (red) are generally dominated by natural and wetland sources, whereas impaired wells (blue/turquoise) are primarily influenced by agricultural sources, especially row crops and pasture. While this overall pattern aligns with expectations, the results also highlight important limitations and nuances. Uncertainty is highest for sources with similar chemical characteristics, such as natural areas and wetlands or different types of agricultural land. This makes it more difficult to clearly distinguish between these sources and indicates that the current data may not fully capture their differences. Additional data collection (e.g., additional chemical analytes such as antibiotics) and improved characterization of source signatures would help refine these estimates in future analyses.

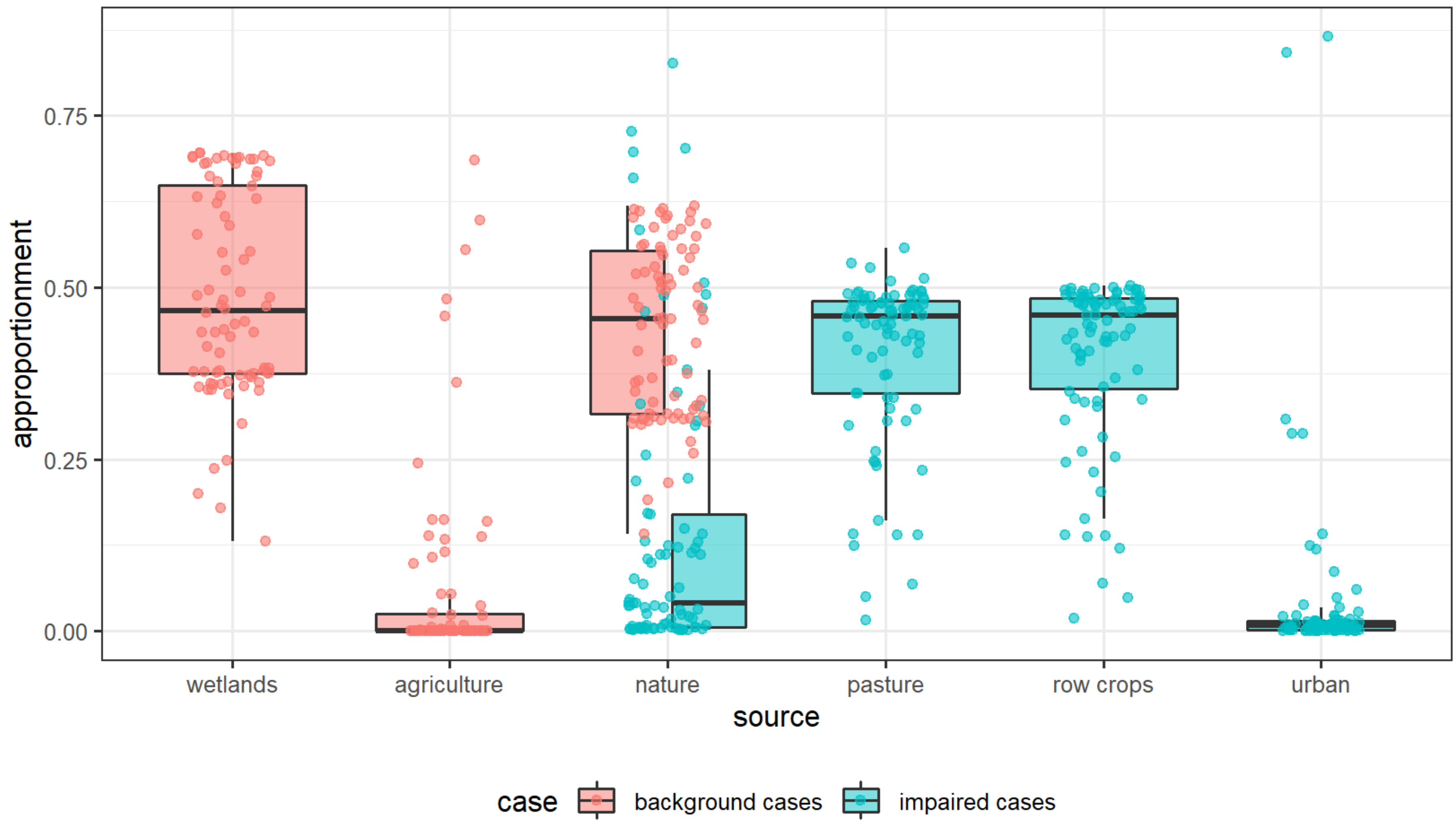

Figure 39: Medain apportionment values for background and impaired cases

Conclusion

The MCMC analysis on mass balance models were developed to estimate nitrate source apportionment for pollutants monitored in groundwater wells. The Pineland Sands study area was divided into background sites (least impacted by pesticide detections) and impaired sites (with detected pesticide presence). Overall, model results aligned with the project team's assumptions:

- In background sites, wetlands and natural sources contributed a larger share of nitrate compared to agricultural sources.
- In impaired sites, agricultural sources, particularly pasture and row crops, contributed substantially more than natural and urban sources.

Although most wells follow the expected pattern based on how "background" and "impaired" sites were defined (using pesticide detections), a subset of wells shows contrasting behavior. For example, some background wells exhibit relatively high contributions from agricultural sources, while some impaired wells retain substantial contributions from natural sources. These mismatches suggest that the current classification based on pesticide detections may not fully capture the full range of agricultural influences. Several factors may explain these discrepancies. Some wells classified as background may still be affected by agricultural inputs that were not detected or recorded in available pesticide data. Alternatively, nitrate contamination may arise from agricultural practices in which the pesticides used as a marker in this study were not applied. These findings suggest that pesticide detections alone may not be sufficient to fully represent agricultural impacts on groundwater.

Finally, these unexpected results underscore the complexity of the region's agricultural system. Interactions among land use, hydrology, and crop management practices create variable transport pathways and lag times, making source attribution inherently challenging. Together with uncertainties in the data, this complexity highlights the need for more detailed and comprehensive monitoring efforts to improve future modeling and better understand nitrate sources in the region.

Using a Groundwater Flow Model to Explore Objective-Based Scenarios in the Pineland Sands Aquifer Region

Author: Leslie Lutke, University of Minnesota

Introduction

The PSA region in north central Minnesota is an important area for agricultural production, fishing and recreation, and is home to many communities, including the Anishinaabeg of the White Earth Nation. As with many regions that rely heavily on groundwater resources, ensuring an adequate amount of clean water for all its uses creates complex management challenges.

The geology of the area is categorized by sandy soils that provide high infiltration capacity for irrigated crops such as potatoes (Stark, Busch, & Deters, 1991) (Stark, Armstrong, & Zwilling , 1994). However, the sandy glacial deposits also increase the potential of contamination of Nitrate-Nitrogen (NO_3-N) in drinking water because of the spaces between sand particles allow water to move more freely throughout the subsurface. A study conducted by the U.S. Geological Survey's (USGS) on public and private drinking water supply aquifers across the between 1988 and 2015 found that groundwater below agricultural land had nitrate concentrations almost three times the national background level of 1 mg L^{-1} nitrate (Ward, et al., 2018). The primary anthropogenic source of nitrogen in intensive agricultural landscapes is nitrogen fertilizer (Ward, et al., 2018).

Elevated levels of nitrate-nitrogen have both human health and ecological impacts. For human health, U.S. EPA sets the MCL for nitrogen as 10 mg L^{-1} under the Safe Drinking Water Act. At concentrations above this level, infants are at risk for developing blue baby syndrome, which reduces the oxygen distributed throughout the body (Ward, et al., 2018). Numerous studies have documented the effects of increase nitrate in drinking water such as congenital birth defects (Holtby, et al., 2014), bladder cancer (Jones, et al., 2016), and pancreatic cancer (Quist, et al., 2018). Alongside human health concerns, there is also risk of contamination affecting aquatic life found in water bodies. The Straight River, one of the major rivers flowing within the study area, is a cold-water trout stream that relies on the upwelling of cold groundwater to provide suitable habitat conditions. McGurk et al. (2006) observed lake trout weight being impacted by chronic exposure to NO_3-N levels above 6.25 mg L^{-1}. In Minnesota, draft ecological chronic exposure standards for salmonid and non-salmonid waters are set at 5 mg NO_3-N L^{-1} and 8 mg NO_3-N L^{-1}, respectively (Monson, 2022).

A modeling framework that integrates groundwater flow and nitrate transport, along with uncertainty analysis and multi-objective optimization, was applied in this study. It was used to evaluate groundwater flow in the Pineland Sands region and to assess the sustainability of agricultural management practices in relation to NO_3-N concentrations in drinking water and in waterbodies used for other purposes. The framework was used to evaluate tradeoffs between farmer social equity, protection of nitrate concentrations in drinking water and trout streams, and agricultural productivity. More detailed information on the framework, methods, results, and other possible uses can be found in the paper by Margarit et al. (2025).

Methods

A groundwater flow model was made using MODFLOW-NWT, a Newton-Raphson formulation for MODFLOW-2005 (Niswonger, Panday, & Ibaraki, 2011). Nitrate-nitrogen transport was simulated using MT3D-USGS (Zheng & Wang, 1999).

The following data were used as inputs into the model:

- The Minnesota Soil Water Balance Model for groundwater recharge inputs using data averaged between 2002 and 2015.
- The 2019 National Land Cover Database for land use information.
- The U.S. Department of Agriculture Soil Survey Geographic Database for soil information.
- Geology and hydrogeology data from the Minnesota County Geologic Atlas for Hubbard, Wadena, and Becker counties.
- Water level observation data for the groundwater heads of 541 observed wells gathered from the MN DNR Cooperative Groundwater Monitoring (CGM) program.
- Daily stream discharge data from the MN DNR Cooperative Stream Gaging Network Database (CSG), collected for two stream gauges along the Straight and Shell Rivers.

The groundwater model realizations were then used to implement the parameter estimation and uncertainty test (PEST) framework. The framework was used to explore how to reduce nitrate pollution in drinking water and rivers while still allowing farmers to grow crops and make a living.

There were three different trade-offs evaluated.

1) economic profit from agricultural practices.
2) nitrate concentrations in streams and drinking water.
3) social equity to ensure that nitrate reductions are shared across more than one farmer and farm.

Two constraints were placed in the framework to ensure that nitrate levels remain below the target. The values 5.4 mg NO_3–N L^{-1} and 8.0 mg NO_3–N L^{-1} are concentrations that represent Mitigations of Level 1 and Level 2, respectively (MN Legislature, 2019).

Instead of finding just one solution, the model tested many possible options and showed the trade-offs among them, forming a Pareto Front. For example, reducing pollution might lower farm profits, or spreading responsibility more evenly might require changes across many farms. The model allowed nitrate levels to vary across different parts of the landscape depending on crop types and management practices. It then compared different scenarios to see which ones best balanced profit, fairness, and water protection. Overall, the goal was to identify realistic strategies that reduce pollution while keeping agriculture productive and fair for farmers.

Results

Curves were created in PEST[7] to show the tradeoffs between different goals under various nitrate limits at the monitoring wells. Each point on the curves (Figure 40) represents one possible solution. The figure shows that these goals conflict with each other. As nitrate leaching increases, the ability to stay safely below nitrate limits decreases. On the other hand, reducing risk (keeping nitrate levels low) creates hardships for farmers because they must cut back on practices that lead to nitrate leaching, which in turn impacts profitability. Figure 40 also shows that risk can never reach a perfect level of 1.0, meaning there is always some chance that nitrate levels will exceed the limits. If nitrate leaching were reduced to zero, the risk could approach 1.0, but this would not be practical. On the right side of the curves, the solutions begin to level off, showing a point of diminishing returns. In this area, even a small reduction in nitrate leaching (about 5 kg N per hectare per year) can greatly reduce farmer hardship while also improving risk. The diamond-shaped points in Figure 40 represent selected solutions that are further described in Figure 41.

[7] PEST is the name of a software package that enables users to conduct model-independent parameter estimation and uncertainty analysis.

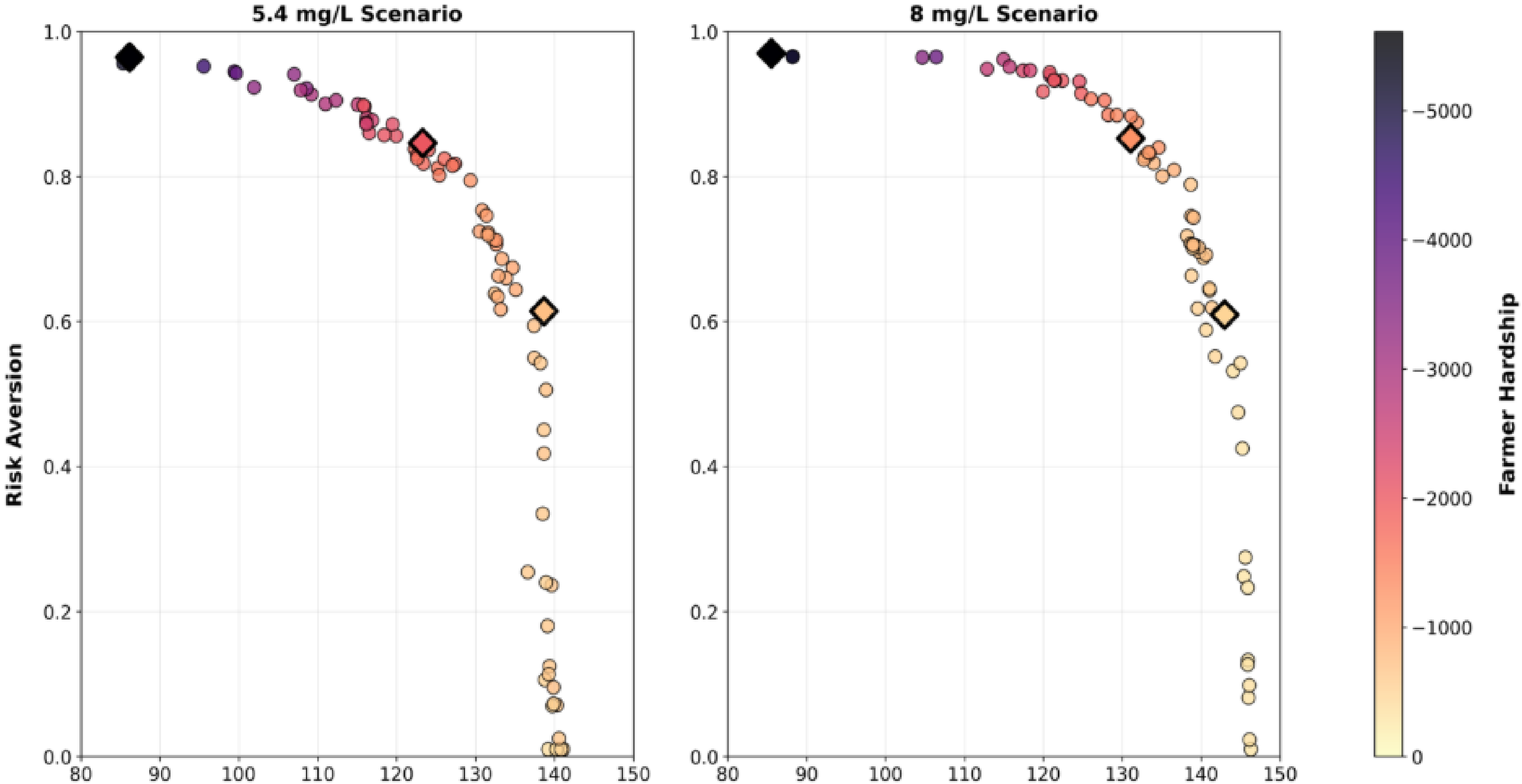

Figure 40: Pareto results representing for two nitrate reduction scenarios, 5.4 and 8 mg L^{-1}

Each scenario A through F shows a different pattern of nitrate leaching across the landscape. These patterns highlight which areas need the largest reductions to meet MDA set drinking water well water quality limits, thereby reducing the need for mitigation. Cooler or darker colors indicate lower allowable nitrate leaching, which can translate to less farm profitability. Across all scenarios, two main areas are consistently affected by lower allowed nitrate leaching: the region between the lower Shell River and Fish Hook River in the east-central part of the model, and the area southwest of Park Rapids. Figure 41 also displays the different combinations of agricultural land use that can shift to achieve nitrate leaching reduction goals. In other words, to reduce nitrate leaching, crops that require less nitrogen would need to be planted instead of ones with a higher nitrogen demand.

Scenario D could be viewed as the most ideal option because it has a risk aversion score of 0.61 and a farmer hardship of -488, which is minimized. On the other hand, the constraint of 8.0mg L^{-1} nitrate-nitrogen may not be sufficient to reduce nitrate in drinking water or to reduce nitrate concentrations that affect stream inhabitants, such as trout. Scenarios B, C, and F could be considered a bigger win for drinking water and the environment, but they come with significant hardship for farmers. The results in Figure 41 show clear trade-offs between economic opportunities and environmental protection.

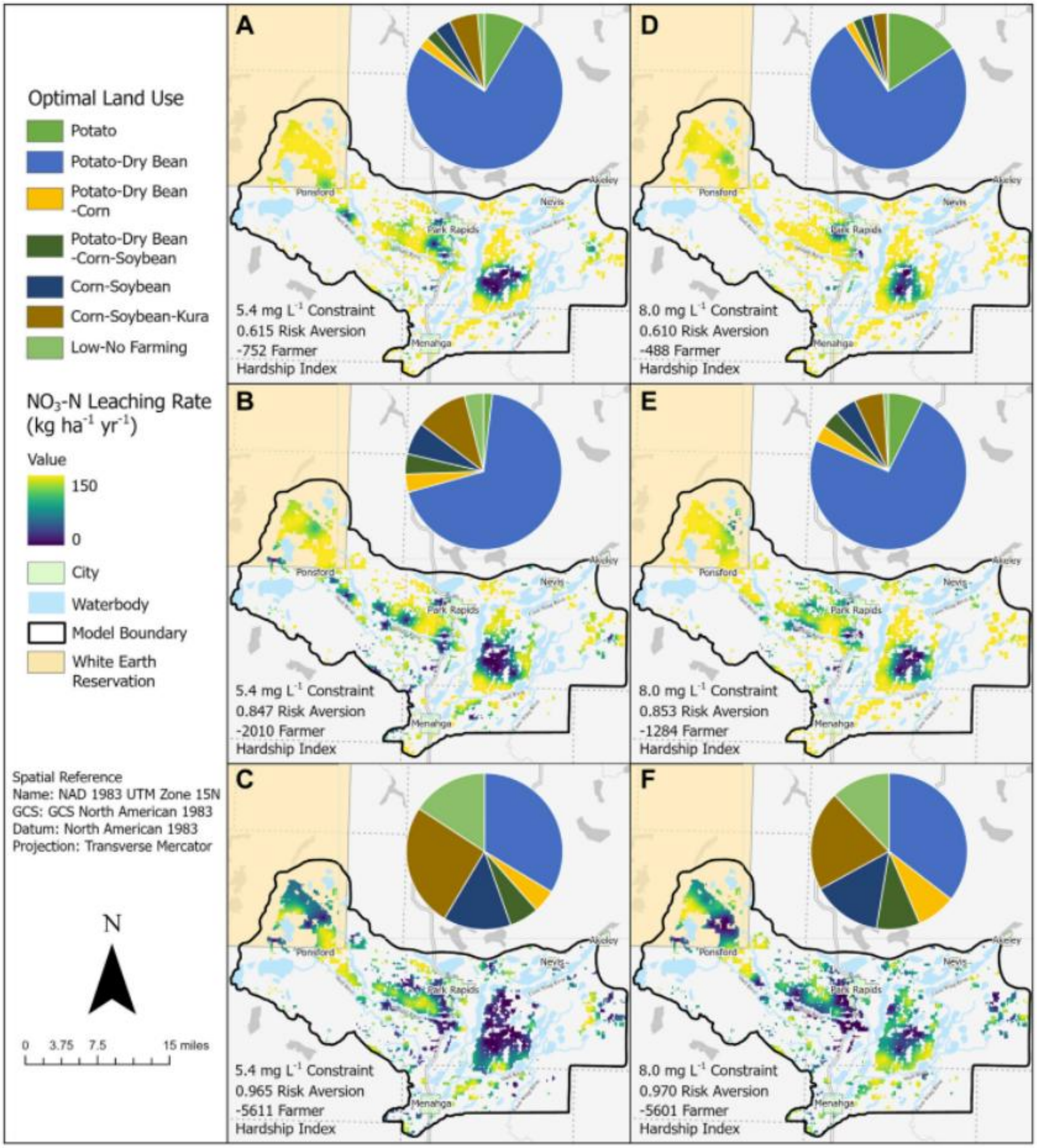

Figure 41: Optimization results of nitrate-nitrogen leaching rates

Conclusion

The optimization results presented a platform for discussing solutions and trade-offs to address nitrate leaching among stakeholders in the Pineland Sands area. The Pareto Front trade-offs presented solutions but do not incorporate stakeholder preferences, compare scenarios, or help determine which solutions constitute the best compromise among stakeholders. Margarit et al. (2025) discussed ways to take the Pareto tradeoffs a step further. The technique for order preference by similarity to ideal solution (TOPSIS) converts trade-offs into ranked alternatives, assigning weights or preferences to each stakeholder group. Using TOPSIS yields a clearer understanding of the ideal best or ideal worst solution for each stakeholder group. The Borda count is a voting method that can then be used to aggregate rankings from each stakeholder group to find a compromise that holds the most consensus. Both TOPSIS and Borda are useful approaches to support more equitable decision-making across multiple perspectives within the watershed.

9 Conclusions and Recommendations

The scientific findings from this study show that the Pineland Sands region is undergoing major environmental changes, driven largely by land-use changes and increased groundwater use over time. The shift from forests and natural landscapes to irrigated agriculture has had a clear impact on both water and ecosystems. Water movement through the region's sandy soils has caused applications of fertilizers and pesticides to reach groundwater and nearby surface waters. Elevated nitrate levels and widespread pesticide detections are now common in groundwater, with some levels exceeding health or environmental guidelines. Groundwater pumping for irrigation has grown rapidly. Because groundwater and surface water are closely connected in this region, these withdrawals can also affect rivers, lakes, and wetlands that depend on steady groundwater flow. While the area still supports diverse and valuable plant and animal communities, there are signs of decline and stress. Some rivers do not meet water quality standards due to low oxygen levels and degraded habitat. Sensitive species—such as mussels and fish—are showing reduced reproduction or are under pressure from changing conditions.

The modeling results reinforce these observations and show how much conditions have changed over time. Compared with pre-settlement conditions, modern land use has greatly increased the amount of water flowing through the system and the amount of nitrogen entering groundwater and surface waters. These changes are not small, they represent several-fold increases and confirm that the impacts are widespread and cumulative. Looking ahead, climate change is expected to add further pressure. Warmer temperatures, longer growing seasons, and more intense rainfall events are likely to increase both water demand and pollutant transport across the landscape.

Sulfide also plays an important role in the health of lakes and their ability to produce wild rice beds. All but one lake with wild rice had pore water sulfide levels below 120 ug L^{-1}; two of five where wild rice was absent had levels above 120 ug L-1; and one had 112 ug L^{-1}pore water sulfide. Most of the lakes sampled are vulnerable to further additions of sulfate from multiple sources, which would be expected to further harm wild rice populations. Care should be taken to monitor lakes with high sulfate sensitivity, especially those with healthy wild rice populations.

Without careful management, these pressures are likely to intensify over time. There are no simple solutions; addressing these challenges will require strong partnerships, coordinated efforts, and honest conversations about tradeoffs. All actions should also account for both current and future climate impacts on water and land use. Even so, it is possible to move forward by developing targeted strategies that balance agricultural productivity with long-term water sustainability, ecological protection, and cultural or subsistence resource uses.

On agricultural landscapes:

- Increase living cover and cover crops to retain nutrients in the landscape year-round.
- Optimize fertilizer and pesticide usage so overapplied amounts do not migrate to the groundwater or runoff in surface waters at levels dangerous to human and ecological health.
- Rotate crops that require less nitrate to grow.
- Evaluate efficient water use systems to reduce water usage during the growing season.
- Expand the use of conservation practices like buffer strips, wetlands, and cover crops that help filter pollutants before they enter waterways.

In developed areas:

- Maintain septic systems to prevent the leaking of pollutants into nearby water sources.
- Increase vegetation cover around lakeshores in developed areas.

Continue to manage and protect groundwater through the Straight River GWMA and other efforts:

- Improve tracking and management of groundwater withdrawals, especially in areas near sensitive streams and lakes and during drought periods.
- Continue to expand water quality and water level monitoring to track changes over time.
- Focus monitoring efforts on areas showing the greatest ecological stress or highest contamination.
- Consider limits or adjustments in high-use areas to prevent further declines in groundwater levels.
- Make findings accessible to communities, landowners, and decision-makers for informed action.

Protect and restore natural landscapes:

- Preserve remaining forests, wetlands, and natural areas that help absorb water and filter pollutants.
- Restore degraded habitats, especially wetlands and stream corridors, which play a key

role in maintaining water quality and ecosystem health.

- Remove infrastructure (dams, weirs, etc.) to improve the movement and connectivity for the aquatic ecosystem.

Create policies, laws, and ordinances that support environmental protection:

- Consider zoning and ordinance changes such as setbacks from lakeshore development or programs that take agricultural land out of production or establish conservation easements.
- Ensure that scientific understandings of hydrology are incorporated into laws and policies that recognize the connectivity of surface and ground waters as well as the cumulative impacts of users on a single aquifer.
- Incorporate adequate protections and monitoring programs for culturally important species guaranteed by treaty for continual hunting, fishing, and gathering rights by the people of White Earth Nation and other tribal nations.
- Develop policies and incentives to increase the efficiency of water use across agricultural, business, and residential water uses (e.g., rotating irrigation schedules and rebates for smart water monitors).

10 References

Advisory Council on Historic Preservation. (n.d.). *Integrating NEPA and Section 106.* Retrieved from https://www.achp.gov/integrating_nepa_106

Ali, A., Jabeen, N., Farruhbek, R., Chachar, Z., Laghari, A., Chachar, S., . . . Yang, Z. (2025). Enhancing nitrogen use efficiency in agriculture by integrating agronomic practices and genetic advances. *Frontiers in Plant Science, 16*. Retrieved from https://doi.org/10.3389/fpls.2025.1543714

Allan, J. (2004). Landscapes and riverscapes: The influence of land use on stream ecosystems. *Annual Review of Ecology, Evolution, and Systematics, 35*, 257–284. Retrieved from https://doi.org/10.1146/annurev.ecolsys.35.120202.110122

Alva, A., Paramasivan, S., Fares, A., Sajwan, K., & Delgado, J. (2006). Nitrogen and irrigation management practices to improve nitrogen uptake efficiency and minimize leaching losses. *Journal of Crop Improvement*, 171-204.

Arnold, J., Moriasi, D., Gassman, P., Abbaspour, K., White, M., Srinivasan, R., . . . Arnold, J.-F. (2012). *SWAT: Model Use, Calibration, and Validation.* Retrieved from ASABE: Engineering a Sustainable Future: https://doi.org/10.13031/2013.42256

Bauer, E. J., Chandler, V., Jirsa, M. A., Marshall, K. J., Gowan, A. S., Hamilton, J. D., & Radakovich, A. L. (2016). *C-42, Geologic Atlas of Becker County, Minnesota.* Retrieved from University of Minnesota Digital Conservancy: , https://hdl.handle.net/11299/184650

Benton-Banai, E. (2010). *The Mishomis Book: The Voice of the Ojibway.* University of Minnesota Press.

Berman, J., & Duffy, A. (2025, July). *University of Minnesota School of Public Health*. Retrieved from Talking wildlife smoke and health with UMN: https://www.sph.umn.edu/news/talking-wildfire-smoke-and-health-with-university-of-minnesota/

Bouwman, L., Goldewijk, K., van der Hoek, K., Beusen, A., van Vuuren, D., Willems, J., . . . Stehfest, E. (2013). Exploring global changes in nitrogen and phosphorus cycles in agriculture induced by livestock production over the 1900-2050 period. *Proceedings of the National Academy of Sciences of the United States of America, 110*(52), 20882–20887. Retrieved from https://doi.org/10.1073/pnas.1012878108

Brooks, K., Ffolliott, P., & Magner, J. (2013). Hydrology and the Management of Watersheds. Wiley-Blackwell Publication.

Camargo, J., Alonso, A., & Salamanca, A. (2005). Nitrate toxicity to aquatic animals: a review with new data for freshwater invertebrates. *Chemosphere*, 1255-1267.

Canby, W. C. (2015). *American Indian Law in a Nutshell (6th Edition).* West Academic Publishing.

Cardenas, M. (2015). Hyporheic zone hydrologic science: A historical account of its emergence and a prospectus. *Water Resources Research*, 3601-3616.

Carlisle Indian School Digital Resource Center. (1892). *Kill the Indian in him, and save the man: R. H. Pratt on the Education of Native Americans*. Retrieved from Carlisle Indian School Digital Resource Center: https://carlisleindian.dickinson.edu/teach/kill-indian-him-and-save-man-r-h-pratt-education-native-americans

City of Park Rapids. (2018, January). *Wellhead Protection Plan.* Retrieved from https://webgen1files1.revize.com/parkrapidsmn/Documents/Departments/Water%20and%20Sewer/Park%20Rapids%20WHP%20Plan%20Final.pdf?t=202503130813550&t=202503130813550

Coffman, D., Black, K., Boyd, K., Clark, S., Greene, B., Mosel, J., . . . Weske, C. (2024). *Climate Change in North Central Minnesota.* Retrieved from University of Minnesota Climate Adaptation Partnership: www.climate.umn.edu/regional-climate-summaries

Dewitz, J. (2021, July 12). *National Land Cover Database (NLCD) 2019 Products (ver. 3.0, February 2024).* Retrieved from USGS: https://www.usgs.gov/data/national-land-cover-database-nlcd-2019-products-ver-30-february-2024

Di, H., & Cameron, K. (2002). Nitrate leaching in temperate agroecosystems: sources, factors and mitigating strategies. *Nutrient Cycling in Agroecosystems, 46*.

Diaz, R., & Rosenberg, R. (2008). Spreading dead zones and consequences for marine ecosystems. *Science, 321*(5891), 926–929. Retrieved from https://doi.org/10.1126/science.1156401

EPA. (1995, April). *Metolachlor.* Retrieved from EPA Prevention, Pesticides and Toxic Substances: https://www3.epa.gov/pesticides/chem_search/reg_actions/reregistration/fs_PC-108801_1-Apr-95.pdf

Esworthy, R., & Yen, J. (2012, November). *Pesticide Law: A Summary of the Statutes*. Retrieved from U.S. Library of Congress: https://www.congress.gov/crs-product/RL31921

Foley, J., DeFries, R., Asner, G., Barford, C., Bonan, G., Carpenter, S., . . . Snyder, P. (2005). *Global Consequences of Land Use.* Retrieved from https://doi.org/10.1126/science.1111772

Getches, D. H., Wilskinson, C. F., Williams, R. A., & Fletcher, M. L. (2011). *Cases and Materials on Federal Indian Law.* West Publishing Co.

Glennon, R. (2002). *Water Follies: Groundwater Pumping and the Fate of America's Fresh Waters.* Island Press. Retrieved from Island Press.

González Perea, R., Daccache, A., Rodríguez Díaz, J., Camacho Poyato, E., & Knox, J. (2018). Modelling impacts of precision irrigation on crop yield and in-field water management. *Precision Agriculture, 19*(3), 497–512. Retrieved from https://doi.org/10.1007/s11119-017-9535-4

Goodrum, J. (2019). Taking on Water: Winters, Necessity, and the Riparian East. *William and Mary Environmental Law and Policy Review Volume 807*.

Greer, B. W. (n.d.). *The American Indian Movement*. Retrieved from American Archive of Public Broadcasting - a collaboration between the Library of Congress and GBH: https://americanarchive.org/primary_source_sets/american-indian-movement

Hansel, C., Lentini, C., Tang, Y., Johnson, D., Wankel, S., & Jardine, P. (2015). Dominance of sulfur-fueled iron oxide reduction in low sulfate freshwater sediments. *ISME Journal*, 2400-2412.

Heglesen, J. (1977). *Groundwater Appraisal of the Pineland Sands Area, Central Minnesota*.

Hobbs, H., & Goebel, J. (1982). *S-01 Geologic Map of Minnesota, Quarternary Geology.* Retrieved from University Digital Conservancy: https://conservancy.umn.edu/items/3ea19cd3-7737-4375-832e-2659c57d6587

Holtby, C. E., Guernsey, J. R., Allen, A. C., VanLeeuwen, J. A., Allen, V. M., & Gordon, R. J. (2014). A population-based case-control study of drinking-water nitrate and congenital anomalies using geographic information systems (GIS) to develop individual-level exposure estimates. *International Journal of Environmental Research and Public Health, 11*(2), 1803-1823. Retrieved from https://doi.org/10.3390/ijerph110201803

Horn, J. E., & Harter, T. (2009). Domestic Well Capture Zone and Influence of the Gravel Pack Length. *Ground Water, 47*, 277–286. doi:10.1111/j.1745-6584.2008.00521.x

Hoss, A. (2025). *Federal Statutory and Regulatory Definitions of "Indian".* Retrieved from Iowa Law Review: https://ilr.law.uiowa.edu/online/volume-110-essays/federal-statutory-and-regulatory-definitions-indian

Jasechko, S., Perrone, D., Befus, K., Bayani Cardenas, M., Ferguson, G., Gleeson, T., . . . Kirchner, J. (2017). Global aquifers dominated by fossil groundwaters but wells vulnerable to modern contamination. *Nature Geoscience, 10*(6), 425–429. Retrieved from https://doi.org/10.1038/ngeo2943

Jenks, A. (1900). *The wild rice gatherers of the upper lakes: a study in American primitive economics.* Retrieved from Wisconsin Historical Society - Turning Points in Wisconsin History: https://legacy.wisconsinhistory.org/turningpoints/search.asp?id=1065

Johnson, C. (2023, May). *Crow Wing River Watershed Stressor Identification Update 2023* . Retrieved from Crow Wing River Watershed: https://www.pca.state.mn.us/sites/default/files/wq-ws5-07010106b.pdf

Jones, R. R., Weyer, P. J., DellaValle, C. T., Inoue-Choi, M., Anderson, K. E., Cantor, K. P., . . . Silverman, D. T. (2016). Nitrate from drinking water and diet and bladder cancer among postmenopausal women in Iowa. *Environmental Health Perspectives, 124*(11), 1751-1758. Retrieved from https://doi.org/10.1289/EHP191

Kappler, C. (1838, June 15). *Treaty with the Chippewa, 1837*. Retrieved from Oklahoma State Tribal Treaty Database: https://treaties.okstate.edu/treaties/treaty-with-the-chippewa-1837-0491

Klein, C. A. (1995). The Constitutional Mythology of Western Water Law. *Environmental Law Journal*.

Klein, C. A. (2022). Groundwater Exceptionalism: The Disconnect Between Law and Science. *Emory Law Journal*.

Klein, C. A., Cheever, F., & Birdsong, B. C. (2013). *Natural Resources Law: A Place-Based Book of Problems and Cases*. Wolters Kluer, 3rd Edition.

Lacher, I., Ahmadisharaf, E., Fergus, C., Akre, T., Mcshea, W., Benham, B., & Kline, K. (2019). Scale-dependent impacts of urban and agricultural land use on nutrients, sediment, and runoff. *Science of the Total Environment, 652*, 611–622. Retrieved from https://doi.org/10.1016/j.scitotenv.2018.09.370

LaFond-Hudson, S., Johnson, N., Pastor, J., & Dewey, B. (2020). Interactions between sulfide and reproductive phenology of an annual aquatic plant, wild rice (Zizania palustris). *Aquatic Botany*, 130230.

Lamers, L., Grovers, L., Janssen, I., Geurts, J., van der Welle, M., van Katwijk, M., & Smolders, A. (2013). Sulfide as a soil phytotoxin. *Frontiers in Plant Science*, 1-14.

Lewis, K., Tzilivakis, J., Warner, D., & Green, A. (2016). An international database for pesticide risk assessments and management. *Human and Ecological Risk Assessment, 22*(4), 1050-1064. doi:10.1080/10807039.2015.1133242

Li, H., Si, B., Zhang, Z., & Miao, C. (2022). Deep soil water storage and drainage following conversion of deep rooted to shallow rooted vegetation. *Agricultural Water Management, 261*. Retrieved from https://doi.org/10.1016/j.agwat.2021.107359

Lindgren, R. (1995). *Availability and Quality of Water from Drift Aquifers in Marshall, Pennington, Polk, and Red Lake Counties, Northwestern Minnesota.* Retrieved from USGS: https://pubs.usgs.gov/wri/1995/4201/report.pdf

Linnakoski, R., Kasanen, R., Dounavi, A., & Forbes, K. M. (2019). Editorial: Forest health under climate change: Effects on tree resilience, and pest and pathogen dynamics. *Frontiers in Plant Science, 10*, 1157. doi:https://doi.org/10.3389/fpls.2019.01157

Lusardi, B. (2016). *C-40, Geologic Atlas of Wadena County, Minnesota.* Retrieved from University of Minnesota Digital Conservancy: https://hdl.handle.net/11299/183206

Lusardi, B. (2018). *C-41, Geologic Atlas of Hubbard County, Minnesota.* Retrieved from University of Minnesota Digital Conservancy: https://hdl.handle.net/11299/198898

Magdoff, F., & van Es, H. (2021). *Chapter 3: Amount of Organic Matter in Soils.* Retrieved from SARE: https://www.sare.org/publications/building-soils-for-better-crops/amount-of-organic-matter-in-soils/

Margarit, P., Dohery, J., Ludtke, L., Nieber, J., Wilson, B., & Magner, J. (2025). Investigating tradeoffs between competing agricultural objectives using multi-objective optimization under uncertainty. *Frontiers Environmental Science, 13*. doi:10.3389/fenvs.2025.1634272

Mayer, T. V., & Jennings, C. E. (2022). *Groundwater Governance, Well Cobbled?* Retrieved from Freshwater Society: https://freshwater.org/reports-archive/white-papers-groundwater-governance/

McClurken, J., Cleland, C., Lund, T., Nichols, J., Tanner, H., & White, B. (2000). *Fish in the lakes, wild rice, and game in abundance : testimony on behalf of Mille Lacs Ojibwe hunting and fishing right.* East Lansing, Mich.: Michigan State University Press.

McGlothlin, R. M., & Slater, S. (2013). No Fictions Required: Assessing the Public Trust Doctrine in Pursuit of Balanced Water Management. *University of Denver Water Review, Volume 53*.

McGurk, M. D., Landry, F., Tang, A., & Hanks, C. C. (2006). Acute and chronic toxicity of nitrate to early life stages of lake trout (Salvelinus namaycush) and lake whitefish (Coregonus clupeaformis). *Environmental Toxicology and Chemistry, 25*(8), 2187-2196. Retrieved from https://doi.org/10.1897/05-270r.1

MDA. (2015, March). *Minnesota Nitrogen Fertilizer Management Plan (NFMP).* Retrieved from MN Department of Agriculture: https://www.mda.state.mn.us/pesticide-fertilizer/minnesota-nitrogen-fertilizer-management-plan

MDA. (2018). *Final Township Testing Nitrate Report: Hubbard County 2016-2017.* Retrieved from MDA Pesticide and Fertilizer Management Division: https://wrl.mnpals.net/islandora/object/WRLrepository%3A3813

MDA. (2025). *2024 water quality monitoring report: January through December 2024.* Retrieved from https://www.mda.state.mn.us/pesticide-fertilizer/water-monitoring-reports-and-resources

MDA. (2026). *Metribuzin Herbicide*. Retrieved from MDA Licensing and Inspections: https://www.mda.state.mn.us/metribuzin-herbicide#:~:text=Metribuzin%20is%20a%20selective%20and,of%2Daction%205%20photosynthesis%20inhibitor.

MDH. (2013, September). *Toxicological Summary for Metribuzin.* Retrieved from Environmental Risk: https://www.health.state.mn.us/communities/environment/risk/docs/guidance/gw/metribuzin.pdf

MDH. (2016a, September). *Alachlor and Drinking Water.* Retrieved from Environmental Risk: https://www.health.state.mn.us/communities/environment/risk/docs/guidance/gw/alachlorinfo.pdf

MDH. (2016b, May). *Clothianidin and Drinking Water.* Retrieved from Environmental Risk: https://www.health.state.mn.us/communities/environment/risk/docs/guidance/gw/clothinfo.pdf

MDH. (2016c, October). *Thiamethoxam and Drinking Water.* Retrieved from Environmental Risk: https://www.health.state.mn.us/communities/environment/risk/docs/guidance/gw/thiamethinfo.pdf

MDH. (2018, April). *Metolachlor and Drinking Water.* Retrieved from Environmental Risk: https://www.health.state.mn.us/communities/environment/risk/docs/guidance/dwec/metolinfo.pdf

MDH. (2020, September). *Imidacloprid and Groundwater* . Retrieved from Environmental Risk: https://www.health.state.mn.us/communities/environment/risk/docs/guidance/gw/imidainfo.pdf

MDH. (2025). *Nitrate Report for 2023-2024.* Retrieved from Drinking Water Protection Program: https://www.health.state.mn.us/communities/environment/water/docs/contaminants/nitrpt20232024.pdf

MDH. (2026, February). *Updated mercury analyses for fish consumption guidelines*. Retrieved from https://www.health.state.mn.us/communities/environment/fish/guidance/hganalyses.html

MDH. (n.d.). *Risk Assessment Advice for Groundwater: Metribuzine Degradates DA, DK, and DADK*. Retrieved from https://www.health.state.mn.us/communities/environment/risk/docs/guidance/gw/metribuzindeg.pdf

Meyer, G. (1986). *Subsurface Till Stratigraphy of the Todd County Area, Central Minnesota.* Retrieved from University of Minnesota Conservancy: https://conservancy.umn.edu/server/api/core/bitstreams/e7eacf5d-07d6-4333-bcc2-67f549028ef7/content

Meyer, M. L. (1994). *The White Earth Tragedy: Ethnicity and Disposession at a Minnesota Anishinaabe Reservation, 1889-1920.* University of Nebraska Press.

MHS. (n.d.). *Minnesota Treaty Interactive*. Retrieved from The US Dakota War of 1862: https://www.mnhs.org/usdakotawar/history/treaties/minnesota-treaty-interactive?__hstc=187326327.dc56a7cea8a1cacb2dc016ca401ea2c7.1758848493811.1759106424913.1759108919259.7&__hssc=187326327.42.1759108919259&__hsfp=1732968120&_gl=1*ao8dt4*_gcl_au*MTkyNjY1O

Minnesota Legislature. (2009). *Minnesota Administrative Rules*. Retrieved from 8420.0100 PURPOSE.: https://www.revisor.mn.gov/rules/8420.0100/

Minnesota Legislature. (2025a). *10.65 Government-to-Government Relationship with Tribal Governments*. Retrieved from 2025 Minnesota Statutes: https://www.revisor.mn.gov/statutes/cite/10.65

Minnesota Legislature. (2025b). *Chapter 103G. Waters of the State*. Retrieved from 2025 Minnesota Statutes: https://www.revisor.mn.gov/statutes/cite/103G

Minnesota Legislature. (2025c). *103G.271 Appropriation and Use of Waters*. Retrieved from 2025 Minnesota Statutes: https://www.revisor.mn.gov/statutes/cite/103G.271

Minnesota Legislature. (2025d). *103G.287 Groundwater Appropriations*. Retrieved from 2025 Minnesota Statutes: https://www.revisor.mn.gov/statutes/cite/103G.287

Minnesota Legislature. (2025e). *103G.291 Public Water Supply Plans; Appropriation During Deficiency*. Retrieved from 2025 Minnesota Statutes: https://www.revisor.mn.gov/statutes/cite/103G.291

Minnesota Legislature. (2025f). *Chapter 103H. Groundwater Protection*. Retrieved from 2025 Minnesota Statutes: https://www.revisor.mn.gov/statutes/cite/103H

Minnesota Legislature. (2025g). *Chapter 103I. Wells, Borings, and Underground Uses*. Retrieved from 2025 Minnesota Statutes: https://www.revisor.mn.gov/statutes/cite/103I

Minnesota Legislature. (2025h). *Chapter 115.03. Powers and Duties*. Retrieved from 2025 Minnesota Statutes: https://www.revisor.mn.gov/statutes/cite/115.03

Minnesota Legislature. (2025i). *Chapter 116D. Environmental Policy*. Retrieved from 2025 Minnesota Statutes: https://www.revisor.mn.gov/statutes/cite/116D

Minnesota Legislature. (2025j). *Chapter 144. Department of Health*. Retrieved from 2025 Minnesota Statutes: https://www.revisor.mn.gov/statutes/cite/144

MN DNR. (2009). *Minnesota Statewide Drought Plan*. St. Paul, MN. Retrieved from https://files.dnr.state.mn.us/natural_resources/climate/drought/drought_plan_matrix.pdf

MN DNR. (2015). *System-Wide Low-Flow Management Plan: Mississippi River above St. Paul, Minnesota*. St. Paul, MN. Retrieved from https://files.dnr.state.mn.us/natural_resources/climate/drought/Mississippi_River_Low_Flow_Management_Plan.pdf

MN DNR. (2017a). *Straight River Groundwater Management Area Plan*. Retrieved from https://files.dnr.state.mn.us/waters/gwmp/area-sr/sr_gwma_plan.pdf

MN DNR. (2017b). *Watershed Context Report: Crow Wing River*. Retrieved from MN Department of Natural Resources: https://files.dnr.state.mn.us/natural_resources/water/watersheds/tool/watersheds/context_report_major_12.pdf

MN DNR. (2022, March 15). *Natural Resouces Atlas: Vegetation - Presettlement*. Retrieved from University of Minnesota Duluth - Natural Resources Research Institute: https://mnatlas.org/resources/vegetation-presettlement/

MN DNR. (2024a). *Natural Resource Atlas: Wild Rice Water Bodies*. Retrieved from University of Duluth - Natural Resources Research Insitute: https://mnatlas.org/gis-tool/?id=k_0015

MN DNR. (2024b). *Groundwater Atlas of Wadena County, Minnesota*. Retrieved from Groundwater Hydrogeology: https://files.dnr.state.mn.us/waters/groundwater_section/mapping/cga/c40_wadena/wade-report.pdf

MN DNR. (2024c). *Minnesota Water Use Data: Permit Index Report (1988-2024)*. Retrieved from https://files.dnr.state.mn.us/waters/watermgmt_section/appropriations/mpars_index_permits_installations_uses.xlsx

MN DNR. (2024d, April). *Straight River Groundwater Management Area Monitoring and Analysis Report.* Retrieved from GWMA: https://files.dnr.state.mn.us/waters/gwmp/area-sr/april2024-straight-river-monitoring-report.pdf

MN DNR. (2025a). *DNR Hydrography Dataset*. Retrieved from https://gisdata.mn.gov/dataset/water-dnr-hydrography

MN DNR. (2025b, December 16). *Infested Waters List*. Retrieved from https://www.dnr.state.mn.us/invasives/ais/infested.html

MN DNR. (n.d.-a). *Jack pine (Pinus banksiana)*. Retrieved from Trees: https://www.dnr.state.mn.us/trees/jack-pine.html

MN DNR. (n.d.-b). *Straight River Groundwater Management Area*. Retrieved from Groundwater Management: https://www.dnr.state.mn.us/gwmp/area-sr.html

MN DNR. (n.d.-c). *Groundwater*. Retrieved from https://www.dnr.state.mn.us/waters/groundwater_section/index.html

MN DNR. (n.d.-d). *Dinner Creek SNA*. Retrieved from Scientific and natural areas: https://www.dnr.state.mn.us/snas/detail.html?id=sna02049

MN DNR. (n.d.-e). *Badoura Jack Pine Woodland SNA*. Retrieved from Scientific and natural areas: https://www.dnr.state.mn.us/snas/detail.html?id=sna02066

MN DNR. (n.d.-f). *MBS Site Biodiversity Significance Ranks*. Retrieved from https://www.dnr.state.mn.us/eco/mbs/biodiversity_guidelines.html

MN DNR. (n.d.-g). *WHAF Watershed Health Scores*. Retrieved from https://www.dnr.state.mn.us/whaf/about/scores/index.html

MN DNR. (n.d.-h). *Invasive terrestrial plants*. Retrieved from https://www.dnr.state.mn.us/invasives/terrestrialplants/index.html

MN Legislature. (2019, July 11). *1573.0040 DRINKING WATER SUPPLY MANAGEMENT AREA; MITIGATION LEVEL DESIGNATION.* Retrieved from Office of the Revisor of Statues, Minnesota Administrative Rules: https://www.revisor.mn.gov/rules/1573.0040/

MN Tribal Wild Rice Task Force. (2018). *1854 Treaty Authority.* Retrieved from https://www.1854treatyauthority.org/images/NEW_TWRTFFINALREPORT2018.pdf

Monson, P. (2022, October). *Aquatic Life Water Quality Standards Draft Technical Support Document for Nitrate.* Retrieved from MPCA: https://www.pca.state.mn.us/sites/default/files/wq-s6-13.pdf

Moyle, J. (1945). Some chemical factors influencing the distribution of aquatic plants in Minnesota. *The American Midland Naturalist*, 402-420.

Moyle, J. (1956). Relationships between the chemistry of Minnesota surface waters and wildlife management. *Journal of Wildlife*, 303-320.

MPCA. (2014a). *Assessment Manual Guidance for Assessing the Quality of Minnesota Surface Waters for Determination of Impairment: 305(b) Report & 303(d) Impaired Waters List*. Retrieved from https://www.pca.state.mn.us/sites/default/files/wq-iw1-04m.pdf

MPCA. (2014b). *Crow Wing River Watershed Monitoring and Assessment Report.* Retrieved from https://www.pca.state.mn.us/sites/default/files/wq-ws3-07010106c.pdf

MPCA. (2017). *Final Technical Support Document: Refinements to Minnesota's Sulfate Water Quality Standard to Protect Wild Rice.* Retrieved from MN Environmental Quality Board: https://www.eqb.state.mn.us/sites/eqb/files/documents/MPCA%202017%20Final%20Technical%20Support%20Document.%20Refinements%20to%20MN%20sulfate%20WQ%20standard%20to%20protect%20wild%20rice.%20wq-rule4-15n.pdf

MPCA. (2023, June). *Crow Wing River Upper Mississippi River Basin.* Retrieved from Watershed assessment and trends update: https://www.pca.state.mn.us/sites/default/files/wq-ws3-07010106e.pdf

MPCA. (2024). *MPCA Impaired Waters Viewer (GIS data extraction)*. Retrieved from https://gisdata.mn.gov/dataset/impaired-waters-viewer

Mugani, R., Messana, G., & Di Lorenzo, T. (2015). The hyporheic zone and its functions: Revision and research status in Neotropical regions. *Brazilian Journal of Biology*, 524-534.

Mullen, M. (2016, February). *House Research: Short Subjects - The Tribal Law and Order Act in Minnesota* . Retrieved from Minnesota House of Representatives: https://www.house.mn.gov/hrd/pubs/ss/sstola.pdf

Myrbo, A., Swain, E., Engstrom, D., Coleman Wasik, J., Brenner, J., Dykhuizen Shore, M., . . . Blaha, G. (2017). Sulfide Generated by Sulfate Reduction is a Primary Controller of the Occurrence of Wild Rice (Zizania palustris) in Shallow Aquatic Ecosystems. *Journal of Geophysical Research: Biogeosciences*, 2736-2753.

Myrbo, A., Swain, E., Johnson, N., Pastor, J., Dewey, B., Engstrom, D., & Peters, E. (2017b). Increase in nutrients, mercury, and methylmercury as a consequence of elevated sulfate reduction in experimental wetland mesocosms. *Journal of Geophysical Research: Biogeosciences*.

NALC. (n.d.). *Water Law: An Overview*. Retrieved from https://nationalaglawcenter.org/overview/water-law/.

Namugize, J., Jewitt, G., & Graham, M. (2018). Effects of land use and land cover changes on water quality in the uMngeni river catchment, South Africa. *Physics and Chemistry of the Earth, 105*, 247–264. Retrieved from https://doi.org/10.1016/j.pce.2018.03.013

National Archives. (n.d.). *Executive Orders*. Retrieved from Federal Register: https://www.federalregister.gov/presidential-documents/executive-orders

National Integrated Drought Information System. (n.d.). *Conditions for Hubbard County, Minnesota*. Retrieved from Drought.gov: https://www.drought.gov/location/56470

National Water Quality Monitoring Council. (n.d.). *Water Quality Portal.* Retrieved March 1, 2026, from https://www.waterqualitydata.us/

NAU ITEP. (n.d.). *Other Tribal Partnership Groups*. Retrieved from Tribal Waste and Response Program: https://itep.nau.edu/twarp/twarsc/other-tpgs/

Niswonger, R. G., Panday, S., & Ibaraki, M. (2011). *MODFLOW-NWT, a Newton formulation for MODFLOW-2005.* U.S. Geological Survey. Retrieved from https://doi.org/10.3133/tm6A37

Nolan, B., Ruddy, B., Hitt, K., & Helsel, D. (1997). *Risk of Nitrate in Groundwaters of the United States - A National Perspective.*

NTAA & MCAF. (2021, April). Retrieved from Indigenous People and Air Pollution in the U.S.: https://www.ntaatribalair.org/wp-content/uploads/2021/04/indigenousairpollution_041421.pdf

Onstad, C., & Jamieson, D. (1970). Modeling the Effect of Land Use Modifications on Runoff. *Water Resources Research, 6*(5), 1287–1295. Retrieved from https://doi.org/10.1029/WR006i005p01287

Owen, D. (2021). Law, Land Use and Groundwater Recharge. *Stanford Law Review, Volume 73*.

Pastor, J., Dewey, B., Johnson, N., Swain, E., Monson, P., Peters, E., & Myrbo, A. (2017). Effects of sulfate and sulfide on the life cycle of Zizania palustris in hydroponic and mesocosm experiments. *Ecological Applications*, 321-336.

Patrick, W., & DeLaune, R. (1972). Characterization of the oxidized and reduced zones in flooded soil. *Soil Science Society of America*, 573-576.

Perez-Lucas, G., Vela, N., El Aatik, A., & Navarro, S. (2019). Environmental risk of groundwater pollution by pesticide leaching through the soil profile. In *Pesticides: Use and misuse and their impact in the environment.* IntechOpen. Retrieved from https://www.intechopen.com/chapters/64602

Pollman, C., Swain, E., Bael, D., Myrbo, A., Monson, P., & Shore, M. (2017). The evolution of sulfide in shallow aquatic ecosystem sediments: An analysis of the roles of sulfate, organic carbon, and iron and feedback constraints using structural equation modeling. *Journal of Geophysical Research: Biogeosciences*, 2719-2735.

Puckett, L., Tesoriero, A., & Dubrovsky, N. (2011). Nitrogen Contamination of Surficial Aquifers—A Growing Legacy. *Environmental Science and Technology*, 839–844.

Quist, A. J., Inoue-Choi, M., Weyer, P. J., Anderson, K. E., Cantor, K. P., Krasner, S., . . . Jones, R. R. (2018). Ingested nitrate and nitrite, disinfection by-products, and pancreatic cancer risk in postmenopausal women. *International Journal of Cancer, 142*(2), 251-261. Retrieved from https://doi.org/10.1002/ijc.31055

Ramankutty, N., Mehrabi, Z., Waha, K., Jarvis, L., Kremen, C., Herrero, M., & Rieseberg, L. (2018). *Annual Review of Plant Biology Trends in Global Agricultural Land Use: Implications for*

Environmental Health and Food Security. Retrieved from https://doi.org/https://doi.org/10.1146/annurev-arplant-042817-040256

Randall, G., & Mulla, D. (2001). Nitrate Nitrogen in Surface Waters as Influenced by Climatic Conditions and Agricultural Practices. *Journal of Environmental Quality*, 337-344.

Ransom, K. M., Grote, M. N., Deinhart, A., Eppich, G., Kendall, C., Sanborn, M. E., . . . Harter, T. (2016). Bayesian nitrate source apportionment to individual groundwater wells in the Central Valley by use of elemental and isotopic tracers. *Water Resources Research, 52*, 5577-5597. doi:https://doi.org/10.1002/2015WR018523

Ransom, K., Bell, A., Barber, Q., Kourakos, G., & Harter, T. (2018). A Bayesian approach to infer nitrogen loading rates from crop and land-use types surrounding private wells in the Central Valley, California. *Hydrologic Earth Systems Science*, 2379-2758.

Ransom, K., Grote, M., Deinhart, A., Eppich, G., Kendall, C., Sanborn, M., . . . Harter, T. (2016). Bayesian nitrate source apportionment to individual groundwater wells in the Central Valley by use of elemental and isotopic tracers. *AGU Publications: Water Resources Research*, 5577-5597.

Reagan, R. (1983). *American Indian Policy.* Retrieved from https://www.epa.gov/sites/default/files/2013-08/documents/president-reagan83.pdf

Reid, L., Kplinger, M., Miller, J., Eldstein, B., & Oiumet, S. (2021). High-Capacity Wells: A Survey of Groundwater Withdrawal Rights and Regulations. *NACL*.

Scanlon, B., Jolly, I., Sophocleous, M., & Zhang, L. (2007). Global impacts of conversions from natural to agricultural ecosystems on water resources: Quantity versus quality. *Water Resources Research, 43*(3). Retrieved from https://doi.org/10.1029/2006WR005486

Schaefer, B., Busch, J., Johnson, H., Meyer, R., & VanRyswyk, B. (2022). *Private Well Pesticide Sampling Project.* Retrieved from MN Department of Agriculture: https://www.mda.state.mn.us/pesticide-fertilizer/private-well-pesticide-sampling-project

Schilling, K., Jha, M., Zhang, Y., Gassman, P., & Wolter, C. (2008). Impact of land use and land cover change on the water balance of a large agricultural watershed: Historical effects and future directions. *Water Resources Research, 45*(7). Retrieved from https://doi.org/10.1029/2007WR006644

Sietman, B. (2026). Research Scientist, Malacologist, MN DNR. (B. DaZiel, Interviewer)

Simkin, S., Bedford, B., & Weathers, K. (2013). Phytotoxic sulfide more important than nutrients for plants within a groundwater-fed wetland. *Ecosystems*, 1118-1129.

Smolders, A., Lucassen, E., Bobbink, E., Roelofs, J., & Lamers, L. (2010). How nitrate leaching from agricultural lands provokes phosphate eutrophication in groundwater fed wetlands: The sulphur bridge. *Biogeochemistry*, 1-7.

Society, S. C. (n.d.). *The Cherokee Nation Cases (1831-1832)*. Retrieved from Supreme Court Historical Society Civics: https://civics.supremecourthistory.org/article/the-cherokee-nation-cases/

Spalding, R., & Exner, M. (1993). *Occurrence of Nitrate in Groundwater-A Review*.

Stark, J. R., Armstrong, D. S., & Zwilling , D. R. (1994). *Stream-Aquifer Interactions in the Straight River Area, Becker and Hubbard Counties, Minnesota*. U.S. Geologic Survey. doi:10.3133/wri944009

Stark, J., Busch, J., & Deters, M. (1991). *Hydrogeology and Water Quality of Glacial-Drift Aquifers in the Bemidji-Bagley Area, Beltrami, Clearwater, Cass, and Hubbard Counties, Minnesota*. U.S. Geologic Survey.

Stark, J., Busch, J., & Deters, M. (1994). *Stream-Aquifer Interactions in the Straight River Area, Becker and Hubbard Counties, Minnesota*. U.S. Geologic Survey.

Strommer, G. D., & Osborne, S. D. (2015). The History, Status, and Futures of Tribal Self-Governance under the Indian Self-Determination and Education Assistance Act. *American Indian Law Review Volume 39, No. 1*.

Stroom, K. (2014). *Crow Wing River Watershed Stressor Identification Report*. Retrieved from https://www.pca.state.mn.us/sites/default/files/wq-ws5-07010106.pdf

Stroom, K. (2024, July). *Straight River (Hubbard County, Minnesota) Nutrient Study*. Retrieved from MPCA: https://www.pca.state.mn.us/sites/default/files/wq-ws5-07010106c.pdf

Tribe, M. C. (1989). *White Earth: A History*. Minnesota Chippewa Tribe.

Turner, R., Rabalais, N., & Justic, D. (2006). Predicting summer hypoxia in the northern Gulf of Mexico: Riverine N, P, and Si loading. *Marine Pollution Bulletin, 52*(2), 139–148. Retrieved from https://doi.org/10.1016/j.marpolbul.2005.08.012

U.S. BIA. (n.d.). *NEPA Compliance on American Indian and Alaska Native Trust or Restricted Lands*. Retrieved from NEPA Compliance : https://www.bia.gov/service/nepa-compliance

U.S. Census Bureau. (n.d.-a). *Resident Population in Becker County, Minnesota*. Retrieved from https://fred.stlouisfed.org/data/MNBECK5POP

U.S. Census Bureau. (n.d.-b). *Resident Population in Hubbard County, Minnesota*. Retrieved from https://fred.stlouisfed.org/data/MNHUBB7POP

U.S. Census Bureau. (n.d.-c). *Resident Population in Wadena County, Minnesota*. Retrieved from https://fred.stlouisfed.org/series/MNWADE9POP

U.S. DOI. (2012). *Indian Affairs NEPA Guidebook*. Retrieved from https://www.bia.gov/sites/default/files/dup/assets/public/raca/handbook/pdf/59_IAM_3-H_v1.1_508_OIMT.pdf

U.S. EPA. (1998, February). *Federal Register Volume 63, No. 29*. Retrieved from U.S. Government Information: https://www.govinfo.gov/content/pkg/FR-1998-02-12/pdf/98-3451.pdf

U.S. EPA. (2002). *Onsite Wastewater Treatment Systems Manual*. Retrieved from National Service Center for Envrionmental Publications (NSCEP): https://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=30004GXI.TXT

U.S. EPA. (2004, June). *Understanding the Safe Drinking Water Act*. Retrieved from Overview of the Safe Drinking Water Act: https://www.epa.gov/sites/default/files/2015-04/documents/epa816f04030.pdf

U.S. EPA. (2016, May). *Revised Interpretation of the CWA Tribal Provision*. Retrieved from Federal Register: https://www.federalregister.gov/documents/2016/05/16/2016-11511/revised-interpretation-of-clean-water-act-tribal-provision

U.S. EPA. (2023, December 7). *EPA Policy on Consultation with Indian Tribes*. Retrieved from United States Environmental Protection Agency: https://www.epa.gov/system/files/documents/2025-04/epa-policy-on-consultation-with-indian-tribes.pdf

U.S. EPA. (2025a). *Origins of EPA*. Retrieved from https://www.epa.gov/history/origins-epa

U.S. EPA. (2025b). *319 Grant Program for States and Territories*. Retrieved from Nonpoint Source Pollution: https://www.epa.gov/nps/319-grant-pr

U.S. EPA. (2025c). *Tribal Nonpoint Source Program*. Retrieved from Nonpoint Source Pollution: https://www.epa.gov/nps/tribal-nonpoint-source-program

U.S. EPA. (2025d). *National Primary Drinking Water Regulations*. Retrieved from Groundwater and Drinking Water: https://www.epa.gov/ground-water-and-drinking-water/national-primary-drinking-water-regulations

U.S. EPA. (2025e). *SDWA Enforcement*. Retrieved from Enforcement: https://www.epa.gov/enforcement/safe-drinking-water-act-and-federal-facilities#state-enforcement

U.S. EPA. (2025f, September 4). *EPA Updates Aquatic Life Benchmarks for Registered Conventional and Antimicrobial Pesticides*. Retrieved from Pesticides: https://www.epa.gov/pesticides/epa-updates-aquatic-life-benchmarks-registered-conventional-and-antimicrobial-0

U.S. EPA. (2025g). *Atrazine*. Retrieved from Ingredients in Pesticide Products: https://www.epa.gov/ingredients-used-pesticide-products/atrazine

U.S. EPA. (2025h). *National Primary Drinking Water Regulations*. Retrieved from Groundwater and Drinking Water: https://www.epa.gov/ground-water-and-drinking-water/national-primary-drinking-water-regulations

U.S. EPA. (2026a). *Clean Air Act Text*. Retrieved from Clean Air Act Overview: https://www.epa.gov/clean-air-act-overview/clean-air-act-text

U.S. EPA. (2026b). *Summary of the Clean Water Act*. Retrieved from EPA Laws and Regulations: https://www.epa.gov/laws-regulations/summary-clean-water-act

U.S. EPA. (2026c). *Tribal Governments Role in SDWA on Tribal Lands*. Retrieved from Tribal Drinking Water: https://www.epa.gov/tribaldrinkingwater/tribal-governments-role-safe-drinking-water-tribal-lands-3#What%20are%20the%20advantages%20and%20responsibilities%20of%20primacy

U.S. EPA. (2026d). *Summary of the National Environmental Policy Act*. Retrieved from https://www.epa.gov/laws-regulations/summary-national-environmental-policy-act

U.S. EPA. (2026e). *Federal Insecticide, Fungicide, and Rodenticide Act and Federal Facilities*. Retrieved from Enforcement: https://www.epa.gov/enforcement/federal-insecticide-fungicide-and-rodenticide-act-fifra-and-federal-facilities

U.S. EPA. (2026f). *Tribal Pesticide Program*. Retrieved from https://www.epa.gov/pesticide-advisory-committees-and-regulatory-partners/tribal-pesticide-programs

U.S. EPA. (2026g). *Clean Water in Indian Country*. Retrieved from https://www.epa.gov/tribal/clean-water-indian-country

U.S. EPA. (2026h). *Aquatic life benchmarks and ecological risk assessments for registered pesticides*. Retrieved May 28, 2026, from https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/aquatic-life-benchmarks-and-ecological-risk

U.S. EPA. (n.d.). *Tools for Automated Data Analysis (TADA)*. Retrieved March 1, 2026, from Water Data and Tools: https://www.epa.gov/waterdata/tada

UMN. (2020). *County Well Index*. Retrieved from Minnesota Geological Survey: https://cse.umn.edu/mgs/cwi

USDA. (2022). *Cropland Data Layer*. Retrieved from National Agricultural Statistics Service : https://gisdata.mn.gov/dataset/agri-cropland-data-layer-2022

USGS. (2024). Zebra and Quagga mussels in the United States-Dreissenid mussel research by the U.S. Geological Survey. USGS. Retrieved from https://pubs.usgs.gov/fs/2024/3009/fs20243009.pdf

Van Meter, K., Basu, N., Veenstra, J., & Burras, C. (2016). The nitrogen legacy: Emerging evidence of nitrogen accumulation in anthropogenic landscapes. *Environmental Research Letters, 11*(3). Retrieved from https://doi.org/10.1088/1748-9326/11/3/035014

Walker, R., & Pastor, J. (2006). Delays in nutrient cycling and plant population oscillations. *Oikos*, 698-705.

Walker, R., Pastor, J., & Dewey, B. (2010). Litter quantity and nitrogen immobilization cause oscillations in productivity of wild rice (Zizania palustris L.) in northern Minnesota. *Ecosystems*, 485-498.

Wan, R., Cai, S., Li, H., Yang, G., Li, Z., & Nie, X. (2014). Inferring land use and land cover impact on stream water quality using a Bayesian hierarchical modeling approach in the Xitiaoxi River Watershed, China. *Journal of Environmental Management, 133*, 1–11. Retrieved from https://doi.org/10.1016/j.jenvman.2013.11.035

Ward, M. H., Jones, R. R., Brender, J. D., de Kok, T. M., Weyer, P. J., Nolan, B. T., . . . van Breda, S. G. (2018). Drinking water nitrate and human health: An updated review. *International Journal of Environmental Research and Public Health, 15*(7), 1557. Retrieved from https://doi.org/10.3390/ijerph15071557

Warren, W. W. (1885). *History of the Ojibway Nation.* St Paul, Minn.: Pioneer Press Co.

Waters, T. (1977). *The streams and rivers of Minnesota.* Minneapolis: University of Minnesota Press.

Weil, R. H. (1989). Destroying a Homeland: White Earth, Minnesota. *American Indian Culture and Research Journal, Volume 13 Issue 2*, 69-95.

Wilkinson, C. F. (1985). Western Water Law in Transition. *University of Colorado Law Review*.

Winchell, N., & Upham, W. (1888). *The Geology of Minnesota, Volume II of the Final Report.* Retrieved from University of Minnesota Conservancy: https://conservancy.umn.edu/handle/11299/56281

Winter, T., Harvey, J., Franke, O., & Alley, W. (1998). *Groundwater and Surface Water: A Single Resources.* Retrieved from U.S. Geologic Survey: https://pubs.usgs.gov/circ/circ1139/

Yohannes, A., Cotter, M., Kelboro, G., & Dessalegn, W. (2018). Land use and land cover changes and their effects on the landscape of Abaya-Chamo basin, Southern Ethiopia. *Land, 7*(1). Retrieved from https://doi.org/10.3390/land7010002

Yost, J., & Hartemink, A. (2019). Soil Organic Carbon in Sandy Soils: A Review. *Advances in Agronomy, Volume 158*, 217-230.

Yousaf, A., Khalid, N., Aqeel, M., Noman, A., Naeem, N., Sarfraz, W., . . . Khalid, A. (2021). Nitrogen Dynamics in Wetland Systems and Its Impact on Biodiversity. *Nitrogen (Switzerland), 2*(2), 196–217. Retrieved from https://doi.org/10.3390/nitrogen2020013

Zhang, P., Cai, Y., He, Y., Xie, Y., Zhang, X., & Li, Z. (2022, November 15). *Changes of vegetational cover and the induced impacts on hydrological processes under climate change for a high-diversity watershed of south China.* Retrieved from Journal of Environmental Management: https://doi.org/10.1016/j.jenvman.2022.115963

Zheng, C., & Wang, P. P. (1999). *MT3DMS: A modular three-dimensional multispecies transport model.* U.S. Army Engineer Research and Development Center. Retrieved from https://apps.dtic.mil/sti/tr/pdf/ADA373474.pdf

11 Appendix

Appendix 1: AI policy for Pineland Sands project

This policy is intended for students, faculty, and consultants providing written information, data, or other services for this report and overall project. While the use of artificial intelligence (AI) is not strictly prohibited on this project, this policy provides guidelines for using AI in a manner that maintains written originality and data reliability.

For UMN students and faculty, please also refer and adhere to UMN's guidance entitled "Appropriate Use of Generative AI Tools" for UMN-specific AI policies. This document is intended for concurrent use with all UMN-specific AI policies.
If you have questions on any of the guidelines below, please contact Dr. Joe Magner (jmagner@umn.edu) for additional information.

1. **If Generative AI is used, an AI statement or acknowledgment must be created.**
    a. Examples:
        i. "I acknowledge the use of <insert AI platform(s)> to generate key words to enter into a database for my background research"
        ii. "I acknowledge the use of <insert AI platform(s)> to create an outline structure for this paper"
2. **AI may not be used to originate language or ideas.**
    a. Users can employ AI to brainstorm or ask questions which lead to independent thought. However, users cannot explicitly use or claim ownership of answers provided by AI. Users are intended to employ AI as a means of discussion, not as a means of gathering answers to questions.
3. **AI can be used to rephrase or reword previously drafted original content.**
    a. Though using AI to draft content is prohibited, users may input content into AI which they have independently drafted for rephrasing or rewording. It is still expected that users review reworded content to ensure that the meaning of their work or any references therein have not been altered in an untruthful manner by AI. It is also expected users acknowledge their use of Generative AI as a statement in their work product.
4. **AI platforms must be licensed and privacy settings are adequate.**
    a. Licensed platforms include Google Gemini and Microsoft Copilot. Privacy settings must be set to not allow the platform to share inputted data to other AI platforms or the internet broadly or otherwise set privacy settings to best maintain security. The use of unlicensed AI platforms is prohibited.
5. **AI can be used to collect sources for research, but all sources must be validated.**

    a. To ensure data quality, sources generated during an AI search must be validated before use. If the credibility of a data source is in question, please contact your supervisor or the Young Environmental staff member listed above for assistance.
    b. Citations may not be generated by AI.

6. **Articles, books, or other copyrighted information may not be uploaded into AI**
7. **Furthermore, confidential information must not be entered into AI under any circumstances.**
    a. Before inputting information into AI, determine the security level of that information.
    b. Inputting confidential information into AI has the possibility of compromising sensitive information. If you are unsure whether the information you wish to input into AI is deemed confidential, please contact your supervisor or the Young Environmental staff member listed above.
    c. As a best practice, make note that all information entered into AI may end up as public information. Therefore, use AI tools cautiously.

Below are individual author declarations about the use of AI.

Name	Description of use
Harprabhjot Kaur Dahliwal	AI was used for proofreading after the original report language was drafted.
Bella Radler	AI was used to troubleshoot R code errors after the original code was developed.
Leslie Ludtke	AI was used to trouble shoot code errors after the original code was developed.
Brady Hartmann	AI was used for the following: • Explaining and troubleshooting ArcGIS Pro workflows, map layout issues, and chart formatting. • Revising wording in manuscript sections that were initially drafted by the author. • Organizing benchmark information and tables for manual review. • Assisting with reference organization and citation-management checks. • Identifying possible source-search directions for author review.

Name	Description of use
Xiang Li, Ph.D.	AI was used for proofreading after the original report language was drafted.
Kerry Holmberg	AI was used for the following: • to improve grammar, readability, and sentence structure • To format citations

Appendix 2: EPA aquatic life benchmark values

Pesticide	Freshwater Vertebrates (μg L^{-1})		Freshwater Invertebrates (μg L^{-1})		Nonvascular Plants (μg L^{-1})		Vascular Plants (μg L^{-1})	
	AcuteA	Chronic B	AcuteC	ChronicD	IC50E	NOAEC	IC50F	NOAEC
Alachlor	900	187	1250	110	1.64	0.035	2.3	0.339
Atrazine	2650	5	360	60	< 1	N/A	4.6	N/A
Metolachlor	1600	30	11750	3200	8	1.5	14	4.4
Metribuzin	21000	< 3000	2100	1290	8.1	2.3	130	18
Clothianidin	> 50750	9700	11	0.05	64000	6350	> 280000	520
Imidacloprid	114500	9000	0.385	0.01	N/A	N/A	N/A	N/A
Thiamethoxam	> 57000	20000	17.5	0.74	> 99000	12000	> 90200	22000

Acute: A short-term exposure benchmark. Acute values are commonly used to evaluate whether a concentration may be high enough to cause harmful effects after a brief exposure period.

Chronic: A longer-term exposure benchmark. Chronic values are used to evaluate whether a concentration may be high enough to affect growth, survival, reproduction, or similar endpoints over a longer exposure period.

AcuteA: EPA acute benchmark for freshwater vertebrates, such as fish and other aquatic animals with backbones.

ChronicB: EPA chronic benchmark for freshwater vertebrates.

AcuteC: EPA acute benchmark for freshwater invertebrates, such as aquatic insects, crustaceans, and similar organisms without backbones.

ChronicD: EPA chronic benchmark for freshwater invertebrates. This endpoint is especially important for several neonicotinoid insecticides because aquatic invertebrates can be sensitive to that pesticide class.

EC50: The effect concentration where 50 percent of the tested organisms show a measured effect, such as reduced growth, movement, or reproduction. It does not always mean death.

LC50: The lethal concentration where 50 percent of the tested organisms die during the toxicity test.

IC50: The inhibition concentration where a measured function, such as algal or plant growth, is reduced by 50 percent.

NOAEC: No observed adverse effects concentration. This is the highest tested concentration where harmful effects were not observed in the study used to support the benchmark.

Freshwater vertebrates: Fish and other aquatic organisms with backbones.

Freshwater invertebrates: Aquatic organisms without backbones, including aquatic insects, crustaceans, and similar organisms.

Nonvascular plants: Algae and similar aquatic plants that do not have vascular tissue.

Vascular plants: Larger aquatic plants, such as duckweed, that have vascular tissue.

μg L^{-1}: Micrograms per liter. For water, 1 μg L^{-1} is approximately equal to 1 part per billion (ppb).

Ng L^{-1}: Nanograms per liter. For water, 1 ng L^{-1} is approximately equal to 1 part per trillion (ppt).

mg L^{-1}: Milligrams per liter. For water, 1 mg L^{-1} is approximately equal to 1 part per million (ppm), or 1,000 μg L^{-1}

MRL: Method reporting limit. This is the lowest concentration that the laboratory can reliably quantify and report for an analyte.

ND: Not detected or not derived, depending on the table context. In result tables it generally means the analyte was not detected; in benchmark tables it may mean a value was not derived.

N/A: Not applicable or not available for the benchmark category shown.

Appendix 3: Minnesota Department of Health, human health based guidance for pesticides

Pesticide Name	Value Type	Exposure Duration	Value ($\mu g\ L^{-1}$)	Health Endpoint(s)
Alachlor	HRL18	Acute	ND	
		Short-term	100	Developmental; Kidney system
		Subchronic	60	Blood system; Liver system; Kidney system
		Chronic	9	
		Cancer	NA	
Metolachlor	HRL23	Acute	ND	
		Short-term	300	Developmental
		Subchronic	300	
		Chronic	300	
		Cancer	NA	
Metribuzin	HRL13	Acute	30	Developmental; Nervous system
		Short-term	10	Thyroid (E)
		Subchronic	10	
		Chronic	10	
		Cancer	NA	
Clothianidin	HRL18	Acute	ND	
		Short-term	200	Developmental
		Subchronic	200	
		Chronic	200	
		Cancer	NA	
Imidacloprid	HRL23	Acute	100	Nervous system
		Short-term	2	Immune system
		Subchronic	2	

Pesticide Name	Value Type	Exposure Duration	Value ($\mu g\ L^{-1}$)	Health Endpoint(s)
		Chronic	2	
		Cancer	NA	
Thiamethoxam	HRL18	Acute	ND	
		Short-term	400	Developmental; Female reproductive system; Liver system
		Subchronic	200	Male reproductive system
		Chronic	200	
		Cancer	NA	

HRL: Health Risk Limit. HRLs are promulgated in rule through the Minnesota rulemaking process and are developed to protect human health for a specified exposure duration.
HRL18: A Health Risk Limit derived by MDH through the 2018 HRL rulemaking/update process.
HRL23: A Health Risk Limit derived by MDH through the 2023 HRL rulemaking/update process.
HRL13: A Health Risk Limit derived by MDH through the 2013 HRL rulemaking/update process.
HRLMCL: An MCL-based Health Risk Limit. MDH uses this code when the HRL is based on a U.S. EPA MCL because the MCL was lower than the HRL value in effect at that time.