NITRATE LEACHING DYNAMICS IN AGROECOSYSTEMS: QUANTIFYING AND INVESTIGATING IMPACTS ON GROUNDWATER QUALITY

by

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A dissertation submitted in partial fulfillment of

the requirements for the degree of

Doctor of Philosophy

(Environment & Resources)

at the

UNIVERSITY OF WISCONSIN-MADISON

2023

Date of final oral examination: 11/8/2023

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Abstract

NITRATE LEACHING DYNAMICS IN AGROECOSYSTEMS: QUANTIFYING AND INVESTIGATING IMPACTS ON GROUNDWATER QUALITY

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Nitrate is an ongoing concern for large sectors of society that rely on municipal and private wells for drinking water. Nitrogen fertilizer and manure additions are standard practices in agroecosystems and are the most significant contributions to Midwestern groundwater nitrate challenges. Nitrate concerns are particularly challenging in the Central Sands region of Wisconsin due to high nutrient requirements of potato/vegetable production and excessively drained soils. To reduce the mounting costs of nitrate remediation, there remains a need to design agricultural systems that lose less nitrate to groundwater while remaining profitable. More extreme weather events and warmer temperatures in Wisconsin have implications for nitrogen management and groundwater quality. As we adapt to our climate reality, communities are increasingly interested in understanding how groundwater quality is changing with respect to nitrate; yet few communities have the necessary systems in place to detect trends at regional scales. This work investigates spatial and temporal nitrate leaching dynamics at both field and regional scales and addresses the following questions: 1) What is the impact of a potato/vegetable rotation on groundwater quality in the irrigated sands of Central Wisconsin? 2) Can interplanting between potato hills lead to reductions in nitrate leaching without negatively impacting yields? 3) Can networks of private wells be used to quantify nitrate trends in a community's groundwater? 4) How can risk communication theory be utilized to better communicate with rural audiences about the importance of testing private wells. In the first experiment, I utilized lysimeters to quantify nitrate leaching over a four-year rotation on a commercial potato/vegetable production farm in Central Wisconsin. Second, I interplanted barley, oats, and millet in the furrows between potato hills to understand whether this practice could reduce nitrate leaching without negatively impacting yield. Third, I worked with two Wisconsin counties to develop spatially extensive networks of private wells for understanding trends. Lastly, I used communication theory to understand how to effectively engage more private well owners in risk reduction behaviors. This work contributes to high-quality long-term data sets that quantify immediate field level impacts and creates a template for communities to track regional well water quality trends.

Acknowledgements

There are many people that I am grateful who have helped me in getting to this point. First, thank you to Dr. Christopher Kucharik for your mentorship and guidance during these past years. You were always generous with your time and may have wondered if I would ever finish, but I will forever be grateful for your encouragement, perspective, and friendship on this journey.

Dr. Francisco Arriaga, thanks for serving as a soil physics role model when I was a MS student and continuing to serve as a mentor during this process. Thank you to Dr. Ken Genskow, who has been a trusted mentor throughout my career; your insight into the extension world and guidance on this project have been invaluable. Thank you Dr. Anita Thompson whose work as a researcher and educator I have always admired and whose advice and perspective on water resources I have always appreciated. And thank you Dr. Yi Wang who has given me a greater appreciation and understanding of the mighty potato and graciously helped with grading of potatoes from my research plots for this project. Each of you has helped me to grow in my learning and become a better scientist. I can't thank you all enough for the time spent mentoring me during this process and I look forward to continued collaborations in the future. Sincere thank you to Jim Miller for all the help and guidance throughout this process, your response time and follow throughout my graduate school experience has always been greatly appreciated.

To Isherwood Family Farms for inviting me onto their farm and into their lives. My time on the farm has been an education of its own. I have a much greater appreciation for what you and your family do on a daily basis. Lynn and Justin, you were always so supportive of any idea I wanted to try. Not only did you allow me to install equipment, but you generously helped with the work. While I will always be grateful for your support of my research and academic pursuits, I appreciate your friendship even more.

To George Kraft who has been a colleague, mentor, and friend for the past 20 years. I would not be where I am today without you taking a chance on me and providing me with the flexibility and encouragement to continue my education. Thanks for all the adventures, lunch conversations, and coffee breaks. Thanks to Paul McGinley, another colleague, friend, and mentor; I am grateful for all the advice and conversations about research and extension work throughout the years. And thanks to the rest of my colleagues who have been patient with me during this process, continually make coming to work enjoyable, and inspire me with their dedication to our natural resources.

To the undergraduate students that have helped with field work along the way: Logan Ebert, Nick Koschak, Liz Belmont, Lily Lefebvre. I could not have succeeded in this project without your time, enthusiasm for learning, and attention to detail that you all brought to these projects. To Dr. Mallika Nocco whose work built an important foundation for the experiments at the Isherwood Farm and Elizabeth McNamee who assisted with various data collection efforts. To my fellow classmates in the Kucharik Lab who have always provided valuable feedback, comradery, and an extra hand for lysimeter installations and potato harvest. I am continually impressed by the creativity and knowledge that you all possess. I look forward to seeing all the great things that you will do in your careers.

Current and past students and staff at the Center for Watershed Science and Education and the Water and Environmental Analysis Laboratory. Your expertise and commitment to high-quality water analysis has been critical to the success of these projects. Despite many obstacles, particularly over these past few years, you have analyzed thousands of samples for these projects. The confidence that I and other researchers have in your analytical work is invaluable, and the water quality data that you provided are the foundation of much of this work.

My sincere gratitude to all the Wisconsin well owners who have submitted samples from their private wells for this work. I have enjoyed many conversations with participants throughout this project and hope that you have found your participation as valuable as I have.

Thanks to my mom and dad Dennis and Mary Masarik, my siblings Brett, Aaron, Jenny, and their families who from my earliest years have continued to provide support and encouragement. To Ryan, thank you for being an all-around great and easy going kid and allowing me to be a small part of your life. Finally, thank you to my partner Shelley. Your patience and encouragement during this process have given me the energy to keep going. I look forward to replacing field work and dissertation writing with vacations, dinners, and home improvement projects.

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Chapter 1

Introduction

1.1 Motivation

Non-point source pollutants from agricultural activities are widespread contributors of nutrients to ground and surface water, affecting the drinking water resources and the biotic integrity of many of the nation's water bodies (EPA, 2009; Dubrovsky, 2010). While nitrogen fertilizers and other nutrient sources (e.g., manure, bio-solids, and legume credits) are valued for their ability to increase crop yields, a portion of nitrogen applied through these methods leaches past the root zone of crops into groundwater as nitrate (Dinnes *et al.*, 2002). Despite the essential role nitrogen plays in agricultural economies, nitrogen losses from agroecosystems represent a contaminant for other sectors of society; a dichotomy that makes nitrogen an ongoing sustainability challenge for many agricultural regions.

Nitrate in groundwater is of particular concern in Wisconsin where nearly 75% of the state's population access groundwater as their primary water source. This includes an estimated 800,000 private wells, where individual homeowners are responsible for the day-to-day decisions regarding testing and safety determination of their drinking water. There is a health-based drinking water standard of 10 mg L^{-1} nitrate-nitrogen. Infants and women who are pregnant or may become pregnant should not consume water above the standard because of the acute health effects such as methemoglobinemia and the potential for neural-tube birth defects. All persons are encouraged to avoid long-term consumption because of research suggesting that nitrate can increase the risk of various cancers (Ward *et al.*, 2005). The most recent statewide survey of agricultural chemicals in Wisconsin groundwater showed that an estimated 8.2% of wells had nitrate concentrations above the drinking water standard of 10 mg/L nitrate-nitrogen (DATCP, 2017b).

Costs to private well owners to reduce exposure to nitrate in their drinking water average $190 per year to buy bottled water, $800 to buy a nitrate removal system plus $100 per year for maintenance, and $7,200 to install a new well (Lewandowski *et al.*, 2008). The Wisconsin Department of Natural Resources’ preferred option is to obtain safe water at the source rather than having to rely on treatment. With an estimated 42,000 exceeding the health standard for nitrate-nitrogen, the cost estimate of abandoning the contaminated well and replacing with a new safe water supply exceeds $440 million. Meanwhile municipalities have incurred more than $40 million in costs to remediate for nitrate contamination (Wisconsin Groundwater Coordinating Council, 2022).

According to the Wisconsin Department of Agriculture, Trade, and Consumer Protection, “Nutrient management planning is one of the best practices farmers can use to reduce excess nutrient applications to their cropland and the water quality problems that result from nutrient runoff to lakes, streams and groundwater” (DATCP, 2013). Wisconsin has a total of 9 million acres of cropland. DATCP estimated that in 2015, 2.9 million acres currently had a nutrient management plan representing an increase of 11% over the previous year. Even so, nutrient management plans are in place on about 31% of cropland acres in Wisconsin (DATCP, 2015).

Nutrient management plans use university nitrogen recommendations based on yield studies to provide nitrogen application rates that maximize yield and profitability (Laboski and Peters, 2012). However, nitrate leaching to groundwater has not been a consideration when setting university nitrogen recommendations or maximum return to nitrogen (MRTN) guidance. Voluntary nutrient management planning among farmers has shown success in getting those that overapply to decrease application rates in accordance with recommendations; but also suggests that similar percentages of farmers that were applying less than recommended rates were encouraged to increase their nitrogen application to meet economic optimal guidelines after participation in the nutrient planning process (Genskow, 2012).

Even optimal nitrogen fertilizer recommendations for row crop agricultural systems have been shown to leach nitrate at rates likely to exceed the drinking water standard, particularly in regions of well-drained soils (Jemison Jr and Fox, 1994; Andraski *et al.*, 2000; Di and Cameron, 2002; Kraft and Stites, 2003; Syswerda *et al.*, 2012; Quemada *et al.*, 2013; Struffert *et al.*, 2016; Wang *et al.*, 2019; Shrestha *et al.*, 2023). Areas of elevated groundwater nitrate in Wisconsin show clear relationships to agricultural land cover, particularly in areas where soil properties and other geologic features exacerbate nitrate leaching losses (CWSE, 2019). As recognition that nitrogen management deserves special considerations in groundwater susceptible areas, the Natural Resource Conservation Service's Conservation Practice Standard 590 has additional restrictions for highly permeable soils or soils with less than 20 inches to bedrock with the specific intent of minimizing entry of nitrogen to groundwater (NRCS, 2015).

Strategies to reduce nitrate leaching in sandy soils include nitrogen fertilizer management practices such as rate, form, timing, nitrification inhibitors, slow-release fertilizers, split-application, and fertigation (Shrestha *et al.*, 2010). Some strategies are aimed at increasing nitrogen use efficiency by increasing yield with the same amount or less of nitrogen fertilizer. Other strategies such as irrigation water management and surfactants are similar in seeking to increase efficiency; however the focus is on managing water to increase yield rather than nitrogen. Efficiency strategies often focus on minimizing the risk for loss and generally have greater impact in wet years and maybe little to no impact in moderate to dry years. Cover crops that scavenge nitrogen from the soil hold potential for reducing leaching if subsequently credited in fertilizer recommendations (Thapa *et al.*, 2018). Perennials or growing crops that require less nitrogen often have greater nitrate reduction potential but require greater disruption to the status quo and can be challenging to implement into potato-vegetable production systems.

The effectiveness of these practices as they relate to water quality needs to be better understood if we are to ensure research and conservation dollars are well spent and provide the best information to 1) farmers and producer-led watershed groups looking to reduce nitrate contamination of groundwater, 2) pollution credit trading programs to ensure practices are worthy of investment, and 3) local, state, and national officials that are making policy decisions related to agricultural practices and water quality.

Significant data exists on baseline nitrate data throughout Wisconsin (CWSE, 2019), however communities are increasingly interested in learning about trends in groundwater quality. Private well owners often have concerns or questions that revolve around whether nitrate concentrations are getting better or worse; meanwhile farmers, conservation professionals, and local leaders are interested in knowing if local changes to agricultural management are making a difference.

Rural private wells provide a window into shallow groundwater aquifers that are most impacted by local land-use decisions. Public water systems are required to submit annual samples for nitrate (WDNR, 2006; WDNR 2014). While these data provide a useful long-term dataset to analyze and investigate trends (CWSE, 2021), public water supply wells are generally not spatially distributed, may have well construction that allows them to access water from deeper aquifers, and lack the ability to learn about groundwater quality in more rural areas where agricultural activity is more prevalent. There has been little to no intentional efforts to collect data in a methodical way that would allow the question of trends to be answered with confidence. Engaging rural residents in these monitoring efforts can help collect valuable data on nitrate trends. Understanding motivations and barriers to participating in long-term monitoring programs can inform strategies for collecting trend data that aids in decision making, increases confidence in the data, and encourages engagement from the broader community.

There is no one-size-fits all solution to improving groundwater quality. Each farm is unique in its management approach and growers would benefit from a menu of whole farm system level strategies to reduce nitrate leaching losses while maintaining profitability. In addition, Wisconsin communities would benefit from systematic approaches to track trends or changes in groundwater nitrate concentrations to inform policy and determine whether groundwater management efforts are working to improve water quality.

1.2 Objectives

The overall objective of my research focuses on the vexing problems associated with nitrogen pollution from agricultural ecosystems with specific emphasis on collecting long-term robust datasets. More specifically I focused on applied research meant to assist farms and communities in studying and addressing nitrate leaching losses to groundwater in Wisconsin. The results of this work will be used to support ongoing water quality modeling, the prioritization of nitrogen reduction strategies, and development of achievable water quality improvement goals.

In Chapter 2, I quantified baseline nitrate leaching for typical cropping systems under commercial management on sandy soils in the Central Sands region of Wisconsin through the collection of year-round water drainage and nitrate leaching data from lysimeters. I expected to see relationships between crop type and nitrate leaching losses. The ability to sample lysimeters year-round helps to understand timing of losses, the impact of weather variability, and aids in the identification of opportunities for minimizing leaching losses. The results were compared to a simple approach for estimating nitrate leaching potential using a nitrogen mass balance.

In Chapter 3, I investigated the potential of inter-planting between potato hills as a potential practice to help minimize water drainage and nitrate leaching while maintaining profitability. Potatoes are grown under irrigation using hill and furrow production systems. In these systems potato plant roots/tubers are concentrated in the hill portion of the field while furrows are essentially void of any significant root mass capable of taking up water and nutrients. As a result, furrows are particularly prone to water drainage and nitrate leaching losses. An experiment was conducted on the potential for interplanting of vegetation (millet, oats, barley) in the furrows between potato hills to reduce nitrate leaching losses to groundwater. I expected that interplanting would minimally compete with potato yet be capable of assimilating significant nitrogen to reduce nitrate leaching losses. Nitrogen uptake of the vegetative biomass between a control and treatment were quantified by measuring potato yields, potato residue, and total above ground biomass of interplanted vegetation in the treatment plot. A detailed sensor array was used to monitor water and solute movement differences between hill/furrow.

In Chapter 4, rural landowners were recruited in a community science project aimed at monitoring trends in groundwater quality. Rural communities who rely primarily on private wells for their drinking water are increasingly interested in knowing whether groundwater quality is getting better or worse. Few communities have appropriate systems to track spatiotemporal changes in groundwater quality. I worked with two Wisconsin counties to develop spatially distributed well water monitoring networks with a goal of testing annually. I expected that data from an extensive network of private wells could be used to understand relationships to land use, well construction, and geologic considerations. And I expected that repeat data from the same wells will allow for determination of trends of individual wells and regional trend assessment that help inform whether well water quality is changing over time.

In Chapter 5, I applied two popular models of risk communication to data from two Wisconsin surveys of attitudes and beliefs surrounding testing of private wells. While public water systems are regularly tested and required to meet drinking water standards, the use of water from a well for drinking purposes is up to the individual well user. Private well owners are essentially their own water utility managers; responsible for decisions about what to test for and whether or not to correct any problems. I applied the extended parallel process model (EPPM) and theory of reasoned action to data from two previous surveys of private well owners to recommend strategies for effective communication with this audience based on risk communication theory.

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Chapter 2

Measurements and modeling of nitrate and chloride leaching from an irrigated potato-vegetable rotation

Abstract

The Wisconsin Central Sands is an important potato and vegetable producing region in the Upper Midwestern US where water quality is significantly degraded because of existing food production agroecosystems. Quantifying both inter- and intra-annual variability of nitrate leaching below the root zone are critical for understanding soil water dynamics and identifying opportunities for improvements to these systems. Equilibrium tension lysimeters (n=4) were installed in a commercial production field and used to quantify drainage and leaching losses of nitrate and chloride below the root zone of an irrigated, moderately-drained loamy sand. Leaching measurements were compared to leaching estimates determined from a nitrogen mass balance model. The study occurred during a 4-year rotation of sweet corn (*Zea mays*), potato (*Solanum tuberosum L.*, var. Russet Burbank), field corn, and peas (*Pisum sativa L.)*. Annual drainage (338-393 mm yr^{-1}) between years was similar for all years of the rotation. The majority of nitrate (60%) and chloride (68%) leaching losses occurred outside of the growing season. Nitrate leaching losses were greatest below sweet corn (150 kg ha^{-1}), followed by potato (133 kg ha^{-1}), field corn (72 kg ha^{-1}), and pea (48 kg ha^{-1}). Chloride losses were greatest under potato (112 kg ha^{-1}), followed by pea (82 kg ha^{-1}), sweet corn (65 kg ha^{-1}), and field corn (28 kg ha^{-1}). The 4-yr flow-weighted mean concentration for the rotation was 28.3 and 20.2 mg L^{-1} for nitrate-N and chloride, respectively. Nitrogen leaching estimates determined using a simple nitrogen budget model compared well with those measured by lysimeters and provide a useful and cost-effective approach for estimating water quality impacts from fields in this type of setting and where monitoring may not be practical.

2.1 Introduction

Non-point source pollutants from agricultural activities are widespread contributors of nutrients to ground and surface water, affecting the drinking water resources and the biotic integrity of many of the nation's water bodies (EPA, 2009; Dubrovsky, 2010). Nitrogen fertilizers and other nutrient sources (e.g., manure, bio-solids, and legume credits) are valued for their ability to increase crop yields, however nitrate ions are highly mobile and susceptible to leaching (Dinnes *et al.*, 2002). Despite the essential role nitrogen plays in agricultural economies, nitrogen losses from agroecosystems represent a widespread contaminant for other sectors of society. This dichotomy makes nitrogen an ongoing sustainability challenge for many agricultural regions.

Nitrate is often elevated above drinking water standards in municipal water supplies and private wells in the Upper Midwestern US (DATCP, 2017a; CWSE, 2019, 2021). The increase in nitrogen fertilizer use over time and the lag time between land-use and groundwater flow to wells that supply drinking water and baseflow to rivers and streams has resulted in increasing concentrations of nitrate in both wells and surface waters (Liu *et al.*, 2021; Basu *et al.*, 2022).

When quantifying the impact of specific agricultural practices on groundwater quality, measuring water drainage and solutes immediately below the root zone provides expedient feedback on temporal leaching losses and patterns of consequential pollutants (Fares *et al.*, 2009). Studies of nitrate leaching are often focused on more standard row crop agricultural systems (Jemison Jr and Fox, 1994; Randall *et al.*, 1997; Masarik *et al.*, 2014; Ochsner *et al.*, 2018). Few studies have quantified water quality impacts over the course of an entire rotation as diverse as those found in vegetable production systems of the Upper Midwest which includes potato, a crop with a high N fertilizer requirement (Laboski and Peters, 2012; Shrestha *et al.*, 2023). Quantifying both inter-annual variability and the intra-annual variability of nitrate leaching over the entire rotation is necessary when evaluating the effectiveness of nitrogen reduction strategies such as nitrogen application rates, slow-release fertilizers, split application techniques, nitrification inhibitors, and cover crops.

Year-round, high-temporal resolution drainage and solute measurements are critical to understanding when leaching occurs and identifying opportunities to reduce leaching losses to groundwater. One category of instrumentation commonly employed to investigate water and solute transport in the vadose zone are lysimeters (Fares *et al.*, 2009). Many forms of lysimeters are used for in-situ measurements, including suction cup (Wang *et al.*, 2012; Jabro, 2016), monolith, passive wick (Gee *et al.*, 2009), zero-tension (Peters and Durner, 2009), and ion-exchange resin (Susfalk and Johnson, 2002). Equilibrium tension lysimeters apply a suction to the lysimeter plate that matches the matric potential of the surrounding soil and sample from a known surface area and are intended to overcome limitations of other lysimeter types (Brye *et al.*, 1999; Masarik *et al.*, 2004). Equilibrium tension lysimeters have been used successfully to investigate water drainage and nitrogen loading below the root zone of highly managed agricultural systems as well as restored or natural ecosystems (Brye *et al.*, 2001; Ochsner *et al.*, 2018; Stenjem *et al.*, 2019). Advantages over suction cup samplers are the ability to obtain direct measurements of drainage while also accounting for preferential flow (Wang *et al.*, 2012). Because they are installed below the frost line, they are also capable of measuring year-round drainage and solute concentrations as opposed to only measuring these during the growing season.

As an alternative to direct measurements of leachate, nitrogen budgets or mass balance models can also be used to define systems of various spatial and temporal scales to understand the magnitude of nitrogen surpluses and may be impactful to guiding crop nutrient management. Furthermore, given the difficulties working with lysimeters to study N losses beneath the root zone, simplified methods that can be applied across larger spatial scales and contribute improved nitrogen use efficiency may be the key to widespread improvements in water quality. Meisinger and Randall (1991) outlined approaches for using simple nitrogen budgets to estimate long-term potential leachable nitrogen in agroecosystems and other studies have used this approach to understand N flows (Stites and Kraft, 2001; Sainju, 2019). Recent work by Byrnes *et al.* (2020) used a nitrogen mass balance approach and a long-term data set of nitrogen to understand county-scale trends in nitrogen use and impacts to water quality over the United States. County-scale N surpluses were determined by taking the difference between N inputs and N outputs. Other researchers have recently been using interactive calculators to determine nitrate leaching losses and assist farmers in Poland with nitrogen management decisions (Dybowski *et al.*, 2020).

The Wisconsin Central Sands (WCS) is an important potato and vegetable producing region in the Upper Midwestern US and is an exemplar of an ongoing conflict between food production and water quality and quantity that currently plagues agriculture. The glacial aquifer system that supplies agricultural irrigation water to the region is also the primary source of water for public and self-supplied domestic use, and industrial applications of the region. The widespread occurrence of nitrate in groundwater, the soil characteristics, and the diversity of cropping systems grown in WCS make it an ideal location for investigating crop rotational effects on nitrate losses to groundwater. Stites and Kraft (2001) previously used the N budget approach of Meisinger and Randall (1991) to quantify typical nitrate leaching losses of 109-203 kg ha^{-1} for sweet corn and potato crops in Central Wisconsin and validated those values using monitoring wells. An inventory of residential wells in this area showed irrigated vegetable production and well drained soils were strong predictors of elevated nitrate-nitrogen levels (Masarik, 2018). Investigations of nitrate-nitrogen concentrations in irrigation wells of the region revealed concentrations from 5-35 mg L^{-1}, with an average around 15 mg L^{-1} (T.A. Campbell, personal communication, December 13, 2022). The recent study by Byrnes *et al.* (2020) identified Central Wisconsin counties as having N surpluses that were largely attributed to fertilizer inputs on cropland, but county level data is too coarse for guiding field scale management in diverse agricultural regions.

This study used equilibrium tension lysimeters to quantify drainage and nitrate and chloride leaching losses below an irrigated potato-vegetable rotation in WCS during a four-year rotation. The objective was to understand the inter and intra-annual variability of nitrate leaching dynamics under highly susceptible soil and variable cropping systems where groundwater quality is degraded. The second goal was to compare nitrate leaching losses derived from ETL measurements with the Meisinger and Randall (1991) N mass balance approach. This allows for an assessment of the modeling approach applied in the WCS. Simplified modeling approaches that can evaluate the impact of nitrogen management strategies with confidence is beneficial for developing decision support tools (DSTs). These types of tools in the form of applications (e.g. computer, tablet, phone), in combination with more effective communication strategies, may work towards improving largescale water quality and nitrogen use efficiency.

2.2 Methods

2.2.1 Site description

An experimental site was established on a commercial farm field near Plover, WI. The field is typical of irrigated potato/vegetable production in the WCS. The field has a four-year rotation that includes potato (*Solanum tuberosum L.*, var. Russet Burbank), sweet corn and field corn (*Zea mays*), and peas (*Pisum sativa L.)* (Table 1). The soil is Friendship loamy sand (moderately well drained, mixed, frigid Typic Udipsamments, USDA Classification) with <3% slopes. The soil profile consists of a loamy sand Ap horizon (0-25 cm), a sand Bw horizon (25-75 cm) underlain by a sand C horizon (>75 cm). Sand with occasional layers of coarse sand and gravel can be found at depths greater than 75 cm to the water table. The water table was encountered at depths of 300-400 cm during the study period.

Fertilizer application rates, planting dates, irrigation events, and other important information were collected through observation or personal communication with the producer (Table 1). Given the high potential for nitrogen leaching losses in sandy soils, fertilizer applications are made through multiple applications. The timing and amounts applied at any given time vary depending on the crop. Starter fertilizer was applied to potato (65 kg ha^{-1}), field corn (34 kg ha^{-1}), and sweet corn (34 kg ha^{-1}) at planting; and in the case of peas, a pre-plant broadcast application (11 kg ha^{-1}). Field corn and sweet corn received additional nitrogen applications (170 and 127 kg ha^{-1}, respectively) as a side-dress application at V4. Potato received a side-dress application of nitrogen (94 kg ha^{-1}) at hilling and fertigation (113 kg ha^{-1}) supplied the remainder of nitrogen to potato. Following pea harvest an additional 6 kg ha^{-1} was applied at planting of a millet cover crop to help with establishment. The nitrogen applications are in accordance with University of Wisconsin recommendations for crops on sandy soils (Laboski and Peters, 2012).

Meanwhile, potash (potassium chloride) was broadcast prior to planting using a spin-spreader in mid to late April; resulting in chloride additions of 158 kg ha^{-1} (potato) and 33 kg ha^{-1} (sweet corn, peas). Potash was not applied to the portion of the field where the lysimeters were located during the field corn portion of the rotation due to an equipment installation that resulted in the contracted spreader avoiding the study area while spreading potash that particular year.

Groundwater is the source of irrigation water for this region; and irrigation wells in the region are often elevated with respect to both nitrate and chloride (Campbell *et al.*, 2023). The amount of nitrate in the irrigation water can be important to consider when attempting to balance crop requirements and groundwater protection (Karim, 1995). The concentration of irrigation water averaged 18 and 22 mg L^{-1} for nitrate and chloride, respectively over the four-year period, and was multiplied by the annual irrigation rate to estimate the mass of nitrate and chloride supplied by the irrigation. Funnels in dryland areas measured weekly precipitation and water was routinely analyzed for nitrate. Precipitation samples generally had less than 0.5 mg L^{-1} nitrate-nitrogen.

A weather station installed 400 m from the experimental field measured precipitation, air temperature, relative humidity, wind speed, and solar radiation. Weather station data contained gaps caused by datalogger issues, and there were additional concerns related to the accuracy of precipitation record during winter months where the technology lacked the ability to accurately quantify snowfall leading to an underestimation of annual precipitation. To account for this, temperature and precipitation data for the study period were downloaded from PRISM (PRISM Climate Group). PRISM is a spatial climate dataset generated from monitoring networks and uses modeling techniques to create a gridded dataset that can be downloaded at a 4 km or 800 m resolution. Data spanning the period of record for lysimeters measurements was downloaded and compared to data collected from the weather station. PRISM data compared well to the weather station data for periods of overlap and provided confidence that modeled data was reliable.

Ultimately, we decided to use the PRISM data set due to the completeness and consistency for the period of interest.

2.2.2 Equilibrium Tension Lysimeters

The field was instrumented in 2016 with four stainless steel equilibrium tension lysimeters (ETLs). The lysimeters were constructed of 1.6-mm-thick stainless steel and were 25.4 cm wide by 76.2 cm long by 15.2 cm tall. A 1-mm-thick, porous stainless steel plate (0.2 μm) was welded to the top, and sidewalls that extend 2.5 cm above the porous plate help to prevent divergence. Each ETL has a collection volume of 29 L or the equivalent of 152 mm of drainage. The ETLs were installed approximately 12 m from the field edge and 1.2 m below the soil surface. Soil structure in potato vegetable production systems is limited given the coarse texture and the repeated tillage and hilling that is performed; as a result, the integrity of the soil at this location did not allow for installation into an undisturbed soil profile. The Ap, Bw, and C horizons were removed, layers were kept separate during installation; following installation the soil horizons were used to cover the lysimeter in the reverse order that they were removed.

The ETLs use a control system to apply suction to a porous stainless-steel plate that is intended to simulate effects of soil matric potential when collecting drainage water to the lysimeter (Brye *et al.*, 2001; Masarik *et al.*, 2004). Ceramic heat dissipation sensors (HDS) which measure soil matric potential were installed adjacent to the upper boundary of the lysimeter. However, we observed that once wetted, the sensors remained static and at values that suggest saturated conditions rather than field capacity (approximately -10 kPa) for a sandy soil substrate. We suspect that water within the ceramic matrix of the HDS was held more tightly by the matrix and the energy gradient between the sandy soil and the ceramic matrix of the HDS was not great enough to equilibrate with the sensor during the periods between wetting events. Rather than apply a suction that was too large, we set a uniform suction of -5 kPa which is half the anticipated equivalent soil suction at field capacity.

Trenches installed below the Ap horizon were used to route the vacuum line and sampling tubes of the ETL to the edge of the field. Special attention was given to installation that provided uninterrupted year-round access to ETLs and ensured the landowner was able to conduct normal field operations (e.g. tillage, planting, fertilization, herbicide applications, harvest, etc.) without having to make accommodations for research equipment.

2.2.3 Water Drainage Sampling and Chemical Analysis

The ETLs were sampled approximately once every two weeks between March and December; and approximately monthly when conditions allowed during frozen soil conditions in winter. Water drainage was removed from the lysimeter using a peristaltic pump. The total volume of leachate was measured and a 125-mL sample of the leachate was brought to the laboratory and stored at 4ºC for water chemical analysis.

Water samples were analyzed for nitrate-nitrogen using a Hanna Instruments® liquid membrane, combination ion selective electrode. Prior to analysis, 1 mL of nitrate ionic strength adjuster (ISA) solution was added to 50 mL. Chloride was measured using a Hanna Instruments® solid state, combination ion selective electrode; 1 mL of halide ionic strength adjuster (ISA) solution was added to 50 mL of sample. Ten percent of measurements were verified at a certified laboratory that employed a Lachat QuickChem 8000 FIA for colorimetric analysis. If ion specific electrode results were more than ten percent different than laboratory analysis, the entire set of samples was rerun. This only happened once during the four-year study period and was attributed to a storage issue with the specific ion electrode which required replacement.

2.2.4 Flow-weighted Mean Leaching Determination

Water volume (V) divided by the collection area of the AETLs allows for determination of drainage per unit area. Concentrations (C) of nitrate and chloride were used to quantify leaching losses as mass per unit area for each individual sampling period as well as the annual cumulative loss for all periods [Eq. 1]. A flow-weighted mean concentration of nitrate and chloride was calculated from the annual cumulative water drainage and nitrate leaching load [Eq. 2]. The multi-year flow-weighted mean concentration provides an important metric for evaluating the overall impact of various cropping systems or climatic variability on the actual groundwater quality below agronomic systems.

[Eq. 1]

$$M_{LEACH} = \sum(V_t \times C_t)$$

[Eq. 2]

$$\text{Annual or Multi-year FWM} = M_{LEACH} / \sum(V_t)$$

Where,

M_{LEACH} = Mass per unit area per leaching year (kg ha^{-1} yr^{-1})

t = Individual Sampling Event

Annual or Multi-year FWM = Flow-weighted mean concentration (mg L^{-1})

Water drainage and solute fluxes (M_{LEACH}) were determined on an annual basis for the period from 1 May to 30 April of the following year. These dates are inclusive of the earliest planting date for the 4-year rotation and roughly coincide with the start of field operations for one crop to the next. We feel that summarizing data based on annual agricultural operations, rather than a calendar year, is a more effective approach for interpreting impacts from one season's management practices and the influence on year-round water drainage and solute fluxes.

While nitrate is the primary reason for this study, chloride serves as a beneficial companion analyte for interpreting water drainage and solute leaching. Potassium chloride was applied prior to planting. Because potassium recommendations for potato are significantly higher than for other crops, chloride leaching helped to validate and provide additional context to nitrate leaching measurements. Chloride like nitrate is highly mobile but not as essential to plant uptake and other transformations. These properties make chloride a useful tracer to track migration of water and solutes through the vadose zone (Stites and Kraft, 2001). Chloride concentrations can be useful for understanding when drainage from one growing season ends and the other begins. Unlike nitrate, there are no health standards associated with chloride. However, in addition to being a conservative tracer, there is increasing interest in the salinization of surface and groundwaters by chloride use and its effects on aquatic life (Dugan *et al.*, 2020).

2.2.5 Potential Leachable Nitrogen

Potential leachable nitrogen (PLN) was estimated using a simple nitrogen budget and compared to actual leaching measurements from ETLs. Estimates of annual PLN were generated for crops grown during the study period and using the approach and tables in Meisinger and

Randall (1991) where PLN (N_{PLN}) is calculated by subtracting nitrogen outputs (N_{OUT}) from nitrogen inputs (N_{IN}) and accounting for any changes in nitrogen storage (N_S).

$$N_{PLN} = N_{IN} - N_{OUT} - \Delta N_S \qquad \text{[Eq. 3]}$$

While this approach is usually more appropriate for estimating long-term leaching potential due to challenges of measuring small changes in nitrogen in the soil profile (Meisinger and Randall, 1991), we were interested in knowing how the approach compares to annual measurements of nitrate leaching. This study location is characterized by low organic matter and excessively drained soil which limit inorganic nitrogen carry over and accumulation of organic matter. These characteristics make quantifying the change in nitrogen storage less critical; therefore, a mass balance with a shorter time step may have more utility in this type of agroecosystems.

The total N inputs were estimated as,

$$N_{IN} = N_{FERT} + N_{DEP} + N_{PPT} + N_{IRR} + N_{SF} + N_{NSF} \qquad \text{[Eq. 4]}$$

Inputs include nitrogen from fertilizer (N_{FERT}), precipitation (N_{PPT}), dry deposition (N_{DEP}), and irrigation (N_{IRR}) water. Nitrogen fertilizer for this area was mostly urea ammonium nitrate (UAN) applied as starter, side-dress, or fertigation. Annual precipitation and irrigation amounts were multiplied by the average nitrate concentration, 0.5 and 18 mg L^{-1} respectively, to determine annual nitrogen inputs from precipitation (N_{PPT}) and irrigation (N_{IRR}). Dry deposition (N_{DEP}) was assumed to be equal to the amount delivered via precipitation. Symbiotic fixation (N_{SF}) was only important during 2019 when peas were grown, here 65% of total N uptake by peas was assumed to have come from fixation. Total uptake was estimated using the assumption that two-thirds of N is removed via the harvested portion of the crop while one-third is returned via residue. Non-symbiotic fixation (N_{NSF}) was estimated to be 3.4 kg ha^{-1} (Meisinger and Randall, 1991).

Total N outputs were estimated as,

$$N_{OUT} = N_{HARV} + N_{AL} + N_{DEN} + N_{SEN} + N_{MISC} \quad \text{[Eq. 5]}$$

N removal via the harvested portion of the crop (N_{HARV}) is the most significant value for determination of nitrogen outputs. Differences in yield can have major implications for the PLN estimate. We initially planned on measuring sweet corn and pea yield, however peas and sweet corn were grown on contract for canning. Timing of harvest is based on optimal conditions for processing and the short notice of harvest did not leave sufficient time to collect samples from the field for years when sweet corn and peas were grown. N removed via sweet corn and pea harvest was estimated using the minimum and maximum annual average for the five-year period from 2015 to 2019 as reported by the National Agricultural Statistics Service for Wisconsin (DATCP, 2016, 2017b, 2018, 2019, 2020). We were able to obtain yield measurements in 2017 (potato) and 2018 (field corn) from the field where we had lysimeters installed. For these years, one standard deviation above and below the mean yield was used to determine the N_{HARV} range for these years. Harvested crop material includes ear with husk (sweet corn), tuber (potato), grain (field corn), and pods (peas). Literature values that report N per harvested unit were multiplied by crop yield to estimate N removal (Meisinger and Randall, 1991; International Plant Nutrition Institute, 2018)

Soil conditions at the field site suggest a low potential for nitrogen loss via denitrification (N_{DEN}). We estimated denitrification to be 3% of the total inorganic nitrogen inputs from fertilizer, precipitation, atmospheric deposition, and irrigation water (Meisinger and Randall, 1991). Ammonia losses from nitrogen fertilizer (N_{AL}) are impacted by application method (incorporated), soil pH (<7), and climate (subhumid, rain within 7 d). Losses of ammonia (N_{AL}) and other miscellaneous losses (N_{MISC}) such as N_2O evolution during nitrification were both estimated to be 1% of the total fertilizer (Meisinger and Randall, 1991). Runoff and erosion which can influence nitrogen budgets at other locations were removed from the calculation of PLN on account of the low slope and high infiltration rate characteristic of the study location.

Lastly, the change in nitrogen storage was estimated as:

$$\Delta N_S = \Delta N_{SI} + \Delta N_{SO} \qquad \text{[Eq. 6]}$$

The soils at this location have less than 2% organic matter and a long history of cultivation; conditions that would indicate limited release of nitrogen from soil storage from year to year. We assumed a change in organic nitrogen storage (N_{SO}) of -8.4 kg ha^{-1} (Meisinger and Randall, 1991). A negative value indicates a release of nitrogen from storage while a positive value represents an addition of nitrogen to the soil profile. Given the soil texture and humid climate, we assumed that any inorganic nitrogen remaining in the profile following harvest did not carryover but rather becomes part of the leachable N pool (i.e. $N_{SI} = 0$).

2.2.6 Statistical Analysis

The four individual lysimeters represent repeat measures from our experimental unit and crop type is treated as the independent variable. The cumulative annual drainage and solute fluxes were summarized for each lysimeter. We examined the effect of crop type on measured drainage and solute fluxes using a one-way ANOVA and the Bonferroni procedure post hoc test (α level 0.05). All statistical analysis were performed using R system for statistical computing and the agricolae package (Mendiburu and Yaseen, 2020; R Core Team, 2022).

2.3 Results and Discussion

2.3.1 Precipitation and Drainage

The 30-year average annual precipitation for this location is 847 mm yr^{-1} (PRISM, 2022). Annual precipitation for the 4-year period was 107-144% above average. Annual drainage ranged from 337-393 mm and averaged between 30-38% of annual precipitation (Figure 1). In three of the four years, little to no drainage was measured from December to March when the soil profile was presumed to be frozen. The greatest drainage occurred during the field corn period of the rotation which also saw the greatest annual precipitation.

The 2016 sweet corn year was the only year in which drainage was measured throughout the winter months. The 30-year mean monthly temperatures are 0.8, -6.0, -9.3, -7.4, -1.1, 6.2 °C for the months of Nov, Dec, Jan, Feb, Mar, and Apr respectively (Figure 2). Above average temperatures that occurred in 2016 Nov (5.1 °C), 2017 Jan (-7.5 °C), 2017 Feb (-2.8 °C), and 2017 Apr (8.1 °C) delayed frost development in the soil profile and warmer than normal winter temperatures allowed for significantly more water movement during the winter period in the sweet corn year of the rotation. Fall and winter temperatures during other years in the rotation were closer to long-term 30-year average temperatures and drainage amounts during those years may be more typical of what can be expected during most years.

Annual drainage was not significantly different between years. Drainage during the period between planting and harvest (i.e in-season) determined as a percentage of the total annual drainage was more variable; sweet corn (24%), potato (48%), peas (54%), and field corn (72%). Water holding capacity of the soil at the study location is low. In irrigated potato/vegetable production systems of the region it is standard practice to apply irrigation water throughout the growing season to maintain optimal soil moisture. As a result, drainage was observed anytime rainfall or irrigation exceeds field capacity even during periods when crop is at or near peak evapotranspiration.

2.3.2 Solute Concentrations and Leaching Losses

Nitrate leaching loads for the rotation averaged 101 kg ha^{-1} yr^{-1} and resulted in a flow-weighted mean concentration of 28.3 mg NO_3-N L^{-1}. Sixty percent of the nitrate-nitrogen leaching losses for the 4-yr period occurred during the off-season (i.e. post-harvest to planting) compared to 40% that occurred during the growing season. The greatest nitrate leaching losses occurred under sweet corn, followed by potato, field corn, and pea/millet (Figure 4).

Chloride leaching averaged 72 kg ha^{-1} and a flow-weighted mean concentration of 20.1 mg L^{-1} with the greatest losses occurring under potato, followed by peas, sweet corn, and field corn. Thirty-two percent of chloride leaching losses for the 4-yr period occurred during the growing season while 68% occurred during the off-season. Given similarities in drainage between years, differences in nitrate and chloride concentrations were responsible for inter-annual differences. Nitrogen and chloride inputs, which are applied based on the type of crop grown, would therefore appear to be the primary driver of leaching losses for the agroecosystem studied here.

Nitrate leaching observed under sweet corn was the greatest of all the crops in the rotation. Only 18% of the nitrate-N leaching occurred during the 2016 growing season which was the lowest percentage of all crops. Following sweet corn harvest, nitrate concentrations remained consistent while drainage volumes increased. This resulted in the majority of nitrate leaching below sweet corn occurring during the post-harvest period. Nearly half of the annual nitrate leaching losses occurred during a three-month period from 2017 March 1 to 2017 May 31 following spring thaw conditions. One noteworthy aspect of sweet corn is that it is harvested when the plant is still green, meaning that significant amounts of nitrogen remain in residue that is returned. Combined with warmer than normal conditions in November, February, March, and April, we suspect greater nitrogen mineralization during the spring following sweet corn contributed to the spring nitrate leaching amounts that we observed. Nitrogen mineralization as the driver of spring nitrate leaching is supported by the chloride measurements which continued to decline throughout the spring period while nitrate concentrations in drainage water were simultaneously increasing. If it were residual nitrate in the soil profile, we would have expected concentrations to behave similarly to chloride.

Annual leaching losses under potato were not significantly different than sweet corn, however there were differences in terms of when leaching occurred. During the growing season, the hill/furrow system allows for preferential flow and solute leaching from the furrow while the solutes in the hill may be less susceptible to leaching as a result (Robinson, 1999; Cooley *et al.*, 2007). Forty-four percent of the annual nitrate-N leaching losses during the potato year occurred during the growing season. Following harvest, the field was disked, and rye seed was spin spread onto the field. We suspect the soil profile disturbance resulting from potato harvest and the post-harvest disking results in a redistribution of solutes (nitrate/chloride) both vertically and horizontally within the field. Combined with minimal post-harvest residue and absence of any actively growing vegetation, potato fields post-harvest are particularly vulnerable to nitrate leaching. A few large rainfall events observed post-harvest likely contributed to the large nitrate and chloride leaching losses observed shortly after harvest.

Even though an attempt was made for a rye cover crop following potato harvest, air temperatures at the time rapidly decreased. November to May temperatures following potato were the coldest of all four years, resulting in minimal vegetative growth from the cover crop. If establishment is poor and/or weather does not allow for significant cover crop biomass to accrue, the effectiveness of the cover crop to prevent nitrate leaching will also be limited. That appears to be demonstrated in our measurements which show nitrate leaching losses continued into the spring following potato. In years when spring weather conditions are warm and wet, a rye cover crop may have a greater potential to assimilate nitrogen because of the ability to overwinter and resume growth once the snow melts and soil temperatures begin increasing.

Nitrate leaching during the growing season under field corn was the greatest of any crop in the rotation and represented 86% of the annual flux. Field corn received the second most nitrogen but also experienced the greatest amount of precipitation. Significant drainage resulting from a 60 mm rainfall event on 18 Jun 2018 likely contributed to growing season leaching losses as evidenced by the spike in nitrate-N concentrations from 16.6 to 58.7 mg N L^{-1} following that event. With the exception of July, which was near the long-term average for precipitation, all other months during the growing season saw significantly more precipitation. The highest concentration of nitrate-N observed in any single sample period was 70.4 mg L^{-1} measured on 22 Aug 2018 below field corn. We attribute the greater than average precipitation during periods when nitrogen supply in the soil is near the peak (May-August) as the primary reason for the significant leaching of nitrate from the root zone and elevated nitrate concentrations during the growing season. In an average to dry year where water drainage is minimized, it is possible that we would observe less nitrate leaching from the upper portion of the soil profile during the growing season. However, any nitrate not assimilated by the crop would still be susceptible to leaching post-harvest. Even though potash was not applied to the study area for this particular year, we still measured 28 kg ha^{-1} chloride loss. The annual chloride leaching loss and the late season increases in chloride concentrations would suggest irrigation water as the likely source of chloride. The lack of chloride application for this year combined with measurable chloride leaching highlights the importance of quantifying nitrate and chloride inputs from irrigation water in these types of investigations which can be significant.

The lowest nitrate leaching losses were observed during the peas/pearl millet year of the rotation, leaching during the growing season represented 25% of the annual total loss. Peas can access nitrogen through the relationship with nitrogen fixing bacteria, essentially eliminating the need for nitrogen fertilizers. Without the addition of commercial fertilizers there are limited mobile forms of nitrogen available to leach from soil during the growing season. Following harvest, pea residue is returned to the soil via tillage. Studies have shown greater mineralization rates from pea residue compared to other residues containing higher carbon to nitrogen ratios, however increased mineralization in that study was not observed until after an initial 14 d period when residue was shown to immobilize nitrogen from the soil profile (Jenssen, 1997). Given the early harvest date of peas, we anticipated some leaching post-harvest from pea residue mineralization; however minimal nitrate leaching was measured during the remainder of the period.

A pearl millet (*Pennisetum glaucum (L.) R. Br*) cover crop was planted within two days of pea harvest. Pearl millet is a quick growing warm season grass that can reach heights of 1 m or greater. Leytem (2016) showed that pearl millet reached a peak leaf area index 45 days after planting and assimilated 185 kg N ha^{-1}. Based on the low nitrate concentrations observed in the drainage post-harvest, we suspect that the millet cover crop helped to sequester inorganic nitrogen in the soil profile following pea harvest as well as inorganic nitrogen that mineralized from the pea residue. Pearl millet winterkills at the first hard frost but was not disked into the soil until the following spring. The low nitrate concentrations observed in drainage water for the fall period through the following spring suggest that millet residue did not mineralize until after the measurement period for this study ended.

The majority of annual nitrate leaching loss occurred following harvest of potato and sweet corn, whereas during the field corn and pea years the majority of nitrate losses occurred during the growing season. Similarities of drainage and leaching losses of sweet corn and potato show comparable influence on water quality for the rotation with respect to nitrate. Meanwhile the influence on the flow-weighted mean concentration of chloride was most heavily influenced by the potato year of the rotation. We suspect the relative magnitude of nitrate leaching losses between crop types to hold from year to year, however seasonal patterns are likely to differ depending on weather from year to year. Precipitation during the growing season, particularly intense rainfall events early in the growing season before plant root systems are developed are obvious times for significant leaching. Rescue applications which some growers may employ after significant leaching events add to the overall nitrogen pool and can exacerbate these losses. During years when moderate to dry precipitation are observed during the growing season, such as 2017 and 2018, nitrate appeared to be better conserved in the upper portion of the soil profile only to be lost post-harvest. These observed differences highlight the importance of monitoring leaching losses year-round. Failing to do so could significantly underestimate solute leaching and lead to wrong conclusions about impacts on water quality. Additional years of data under different climatic variability would help to better understand the extent to which precipitation and temperature contribute to drainage and nitrate leaching losses.

2.3.3 Potential Leachable Nitrogen Estimate

Potential leachable nitrogen determined using the nitrogen balance approach suggested average nitrate leaching losses to be greatest under potato, followed by sweet corn, field corn, and peas respectively (Table 3). Leaching measurements quantified using lysimeters were within the range of PLN estimates for three of the four years. The 2016 year, when sweet corn was grown was the lone exception where the actual measured leaching loss was almost 1.5 times greater than the PLN estimate. Some of the differences could be explained by using regional yield estimates as opposed to actual yield measurements, however it is unlikely that yield differences alone explain the discrepancy. As mentioned, previously warm wet conditions may have contributed to greater mineralization. Assumptions regarding nitrogen processes and using literature values for nitrogen content of harvested material could also lead to an over or underestimation of certain components of the nitrogen budget. Additional leaching data from all years, in particular sweet corn, could help to confirm whether 2016 was an anomaly with respect to leaching measurements. Overall, the PLN estimates appear reasonable and show promise for understanding management practices and implications for nitrate leaching in fields where water quality monitoring is not taking place.

The nitrogen fertilizer inputs used in this study are in accordance with University of Wisconsin recommendations for economic optimum rates (Laboski and Peters, 2012). We cannot assume that all growers are following similar application rates; some may be using more and some less. Similarly yield which has a large influence on calculation of PLN is going to vary widely from farm to farm and even within each field. When actual nitrogen fertilizer rates, yield data, and other inputs are known, more accurate assessments of nitrate leaching from individual fields, farms, crops, and year to year variability are possible. The results here are meant to illustrate the role of the simple mass balance approach in providing a reasonable estimate on a study location where actual leaching measurements were collected. Because yield and nitrogen inputs are relatively easy to obtain, this approach can help bracket what might be expected regionally for nitrate leaching from various crops or illustrate scenarios for improved water quality as a results of management alterations.

The leaching measurements help to validate the nitrogen leaching model which can be a useful tool for understanding impacts of management practices on leaching losses in fields where water quality is not actively being monitored. Timing strategies (i.e. split application, fertigation, slow-release fertilizers) that minimize the amount of leachable nitrogen in the active portion of the soil profile during the growing season can be beneficial to avoid rescue applications or reduce the total amount of nitrogen applied; however nitrogen that does not leach during the growing season will be available to leach during the period following harvest through the following spring. When timing strategies are not accompanied by either a reduction in nitrogen applied or increases in yield, the potential for significant reductions in nitrate leaching and improvements to groundwater quality is limited. In addition, nitrogen in irrigation water can be significant (Campbell *et al.*, 2023). Crediting of irrigation water should be considered when nitrate concentrations are sufficiently elevated. Lastly, investigating the addition of crops that require less nitrogen into a rotation, incorporating interplanting and/or post-harvest cover crops that scavenge nitrogen, or incorporating rest periods (i.e. years in which a low-input perennial is grown in place of a crop like field corn) have potential for reducing nitrate leaching losses and improving long term water quality below agricultural ecosystems (Heineman and Kucharik, 2022).

2.4. Conclusion

This study quantifies the impact of crop type on nitrate leaching losses over a four-year rotation. Precipitation influenced the timing of nitrate leaching during individual years, however the main driver of annual nitrate leaching losses was the nitrogen inputs relative to nitrogen removed via harvested crop. These data highlight the difficulty of meeting the nitrate-N drinking water standard in groundwater directly below commercial potato/vegetable production systems of the Upper Midwest. The overall impact of this 4-year rotation would suggest groundwater quality that is nearly three times the nitrate-nitrogen drinking water standard. Particularly when looking at individual crops, meeting the drinking water standard every year in the rotation is not achievable with current management practices.

Potential leachable nitrogen estimates performed using a simple nitrogen budget generally agreed with field measurements of annual nitrate leaching. While additional validation on other farms and cropping systems would be beneficial, simple tools that estimate leachable nitrogen may provide an important metric to assess the impact of various practices without collecting potentially costly and time-consuming water quality data. When looking to minimize nitrate leaching losses in systems as diverse as those found in this region, the entire rotation period should be evaluated for both nitrate optimization and economics rather than focusing solely on the growing season or on one particular crop.

A strategy that warrants further investigation includes the incorporation of rest years where farmers grow vegetation that receives little to no nitrogen fertilizer and is not intended to be harvested, but may have other benefits to future potato crops. Potato have the highest return on investment of any crops in the current rotation, anything that can be done in previous years to improve yield and quality of potato harvests may offset the costs of not growing another crop in the rotation such as field corn. Growing crops like mustard, pearl millet, buckwheat, or oil seed radish in the year proceeding potato that only relies on starter fertilizer and nitrogen supplied from the irrigation water would be expected to leach significantly less nitrogen. Because these crops have possible biofumigant properties, there may be additional savings when it comes to reduced fumigant applications, increased organic matter and water holding capacity, as well as other soil health benefits. Accumulation of biomass also sequesters significant nitrogen which if returned to the soil and credited in the potato year, could also reduce nitrogen fertilizer costs.

Currently, external costs associated with nitrate pollution of groundwater are not incorporated into production costs; meaning that what people pay at the grocery store is therefore artificially low. To incentivize practices that are beneficial to water quality may initially require subsidies to offset the potential reduced income of more diverse cropping systems or incorporation of rest years for water quality benefits. Long-term success, however, requires recognition by consumers and companies that process these crops of the value that certain practices have on the overall health and well being of rural communities and our surface and groundwater resources. Farmers that employ cropping systems that are demonstrably better for water quality, either through actual monitoring or through verified nitrogen budgeting, should be rewarded with higher prices for their efforts.

2.5. Acknowledgements

This work was supported by the Wisconsin Department of Natural Resources and U.S. Environmental Protection Agency through the Wisconsin Groundwater Research Program coordinated by the University of Wisconsin Water Resources Institute (DNR Project #233). We would also like to thank Justin and Lynn Isherwood for their assistance with the project and continued support of scientific research. Field and technical assistance provided by Mallika Nocco, Elizabeth McNamee, Logan Ebert, Nicholas Koschak, Elizabeth Belmont, and Lily Lefebvre were greatly appreciated.

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2.7 Tables

Table 1. Details regarding cropping systems, precipitation, and irrigation for the 4-year study period.

Parameter	2016	2017	2018	2019
Crop	Sweet corn	Potato	Field corn	Peas
Planting date	2 Jun	5 May	15 May	26 May
Harvest date	31 Aug	14 Sep	17 Oct	24 Jul
Annual precipitation (mm)	1075	908	1221	1162
In-season precipitation (mm)†	341	454	684	315
Irrigation (mm)	89	117	103	91
Post-harvest cover	Oats	Rye	Residue	Millet

†In-season precipitation defined as 1 May – Harvest date.

Table 2. Annual drainage, flow-weighted mean (FWM) concentrations, and leaching losses for four equilibrium tension lysimeters.

Growing Season	Crop	Drainage	FWM Nitrate-N	Nitrate-N Leaching	FWM Chloride	Chloride Leaching
		mm	**mg L^{-1}**	**kg ha^{-1}**	**mg L^{-1}**	**kg ha^{-1}**
2016	Sweet corn	338(94)a	44.5(2.7)a	150(43)a	18.8(2.1)b	65(25)ab
2017	Potato	344(69)a	38.9(3.0)a	133(22)ab	32.6(5.3)a	112(26)a
2018	Field corn	393(124)a	17.8(3.0)b	72(29)bc	7.2(1.2)c	28(9)b
2019	Peas/millet	348(60)a	13.9(2.1)b	48(9)c	23.4(5.5)b	82(25)a
	LSD	201	6.1	63	8.9	50
2016-2019	4-yr rotation	1424(292)	28.3(1.3)	403(92)	20.1(2.6)	287(73)

Standard deviation in parentheses
Values within each column followed by the same letter are not significantly different based on Fisher's Least Significant Difference (LSD) test (0.05).

Table 3. Components of Potential Leachable Nitrogen (PLN) budget. All components of nitrogen budget expressed in kg N ha^{-1}.

	2016	2017	2018	2019
Source	**Sweet corn**	**Potato**	**Field corn**	**Pea**
Dry matter yield (Mg ha^{-1})	4.9-5.3†	10.4(±1.7)‡	10.9(±1.8)‡	0.9-1.1†
		N inputs		
N fertilization	160	258	224	17
Dry N deposition	5	5	6	6
Precipitation N	5	5	6	6
Irrigation N	16	21	19	16
Symbiotic N fixation	0	0	0	37
Nonsymbiotic N fixation	3	3	3	3
Total N input	**190**	**291**	**258**	**85**
		N outputs		
Harvest N removal	79-86	138-194	134-186	35-44
Denitrification	5	8	7	1
Ammonia loss	2	3	2	<1
Senescence	4	8	8	2
Miscellaneous gaseous	2	3	3	1
Total N output	**89-92**	**158-217**	**153-207**	**37-46**
		N storage		
Inorganic N	0	0	0	0
Organic N	-8	-8	-8	-8
Change in N storage	**-8**	**-8**	**-8**	**-8**
PLN	**99-107**	**82-141**	**59-113**	**45-54**

†County level yield data used to represent range of anticipated yields for sweet corn and peas. Range represents minimum and maximum average yield for the period between 2015 and 2019.

‡Yield determined from the field measurements for potato and field corn in each respective year. Standard deviation written in parentheses. One standard deviation used to determine lower and upper range for harvest N removal.

2.8 Figures

Figure 1. Cumulative precipitation (solid line) and lysimeter drainage measurements (grey circles) for each cropping season (1 May to 30 April). Error bars represent standard deviation of each sampling period. Vertical lines represent 1 May (solid) and harvest date (dashed) for each respective crop.

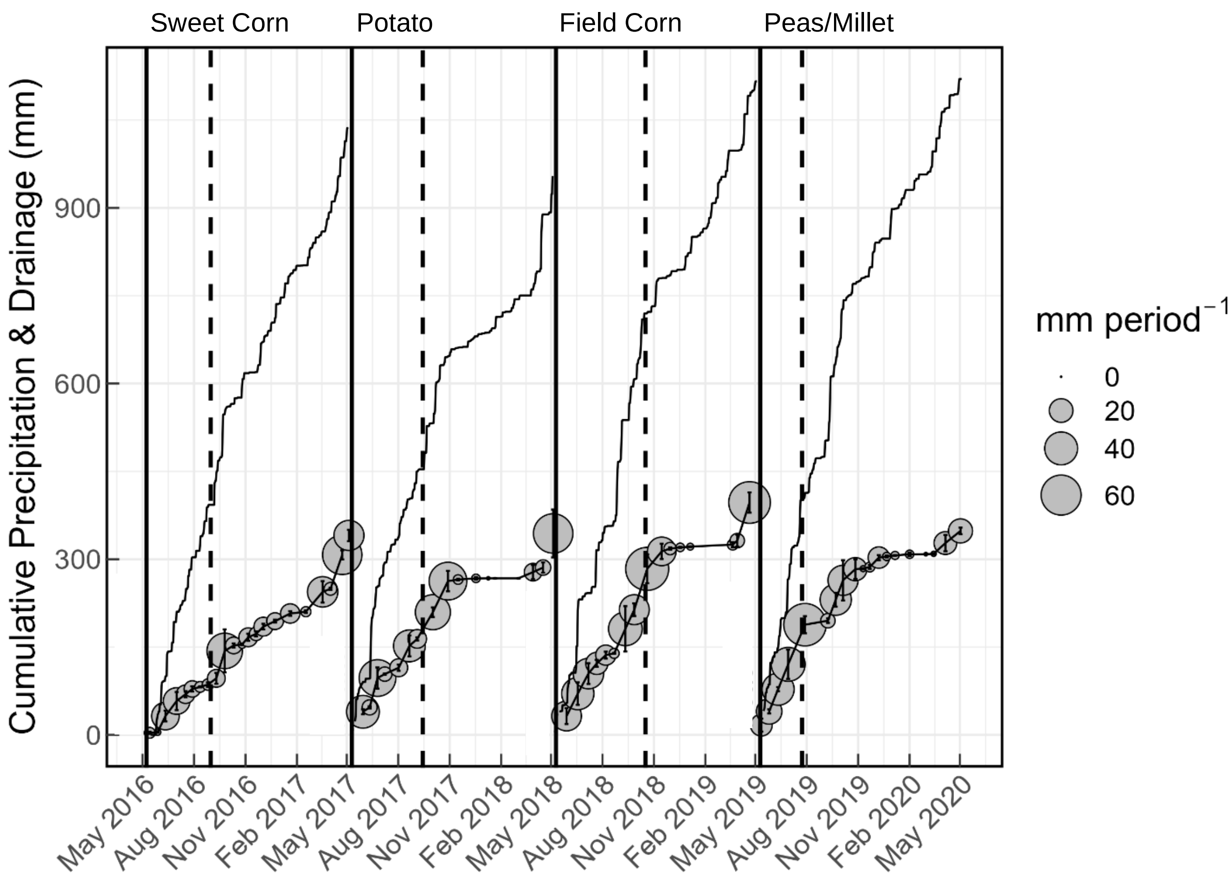


Figure 2. Monthly mean temperature for each cropping year. Dashed line represents 30-year monthly average (PRISM, 2022).

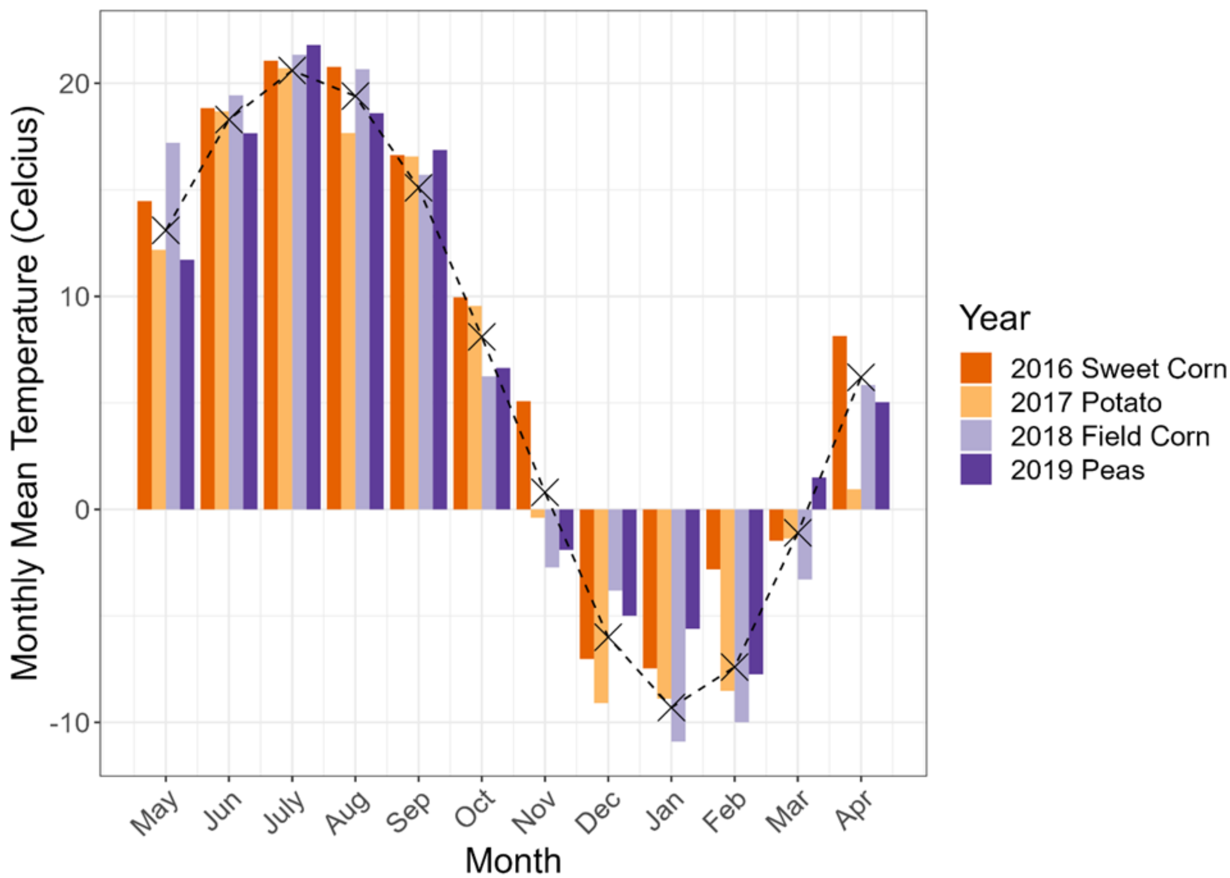


Figure 3. Monthly precipitation measurements for each cropping year. Dashed line represents 30-year monthly average (PRISM, 2022)

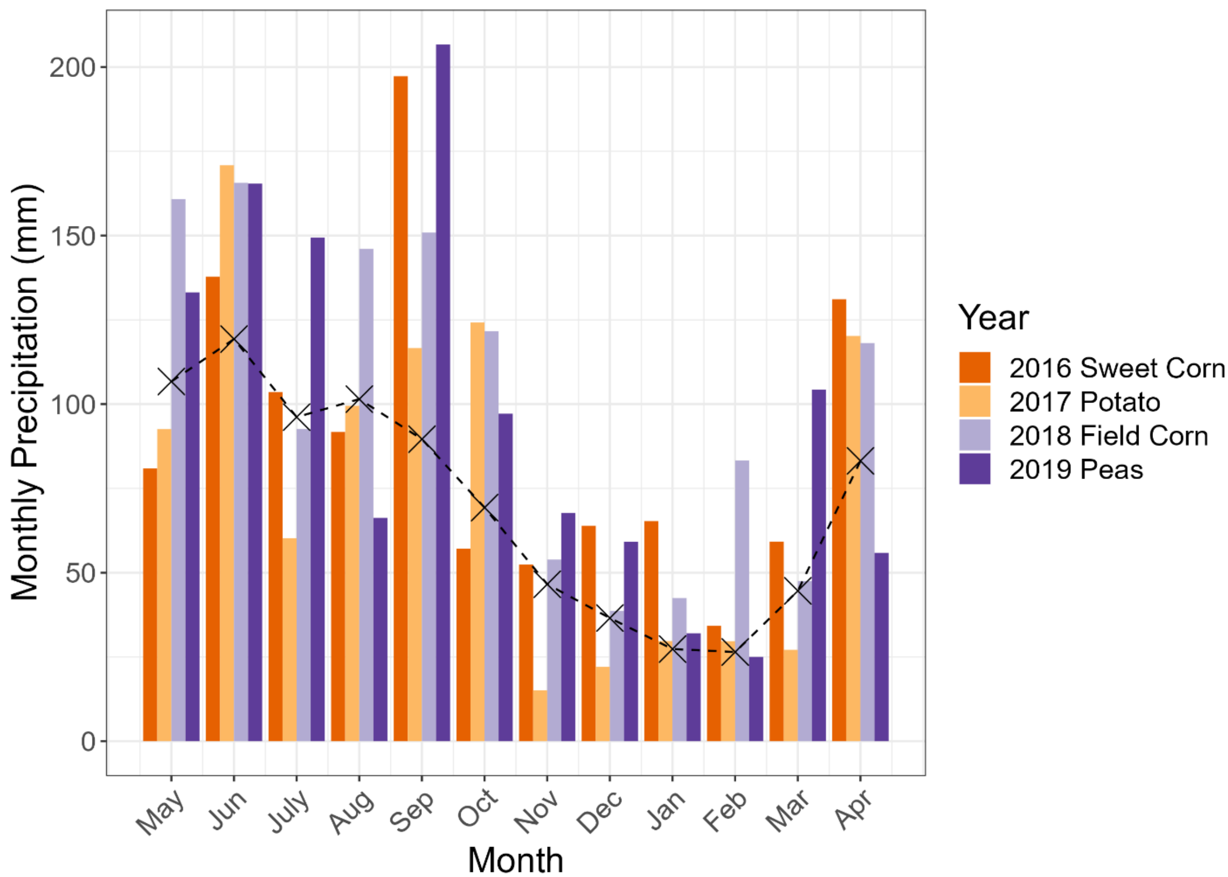
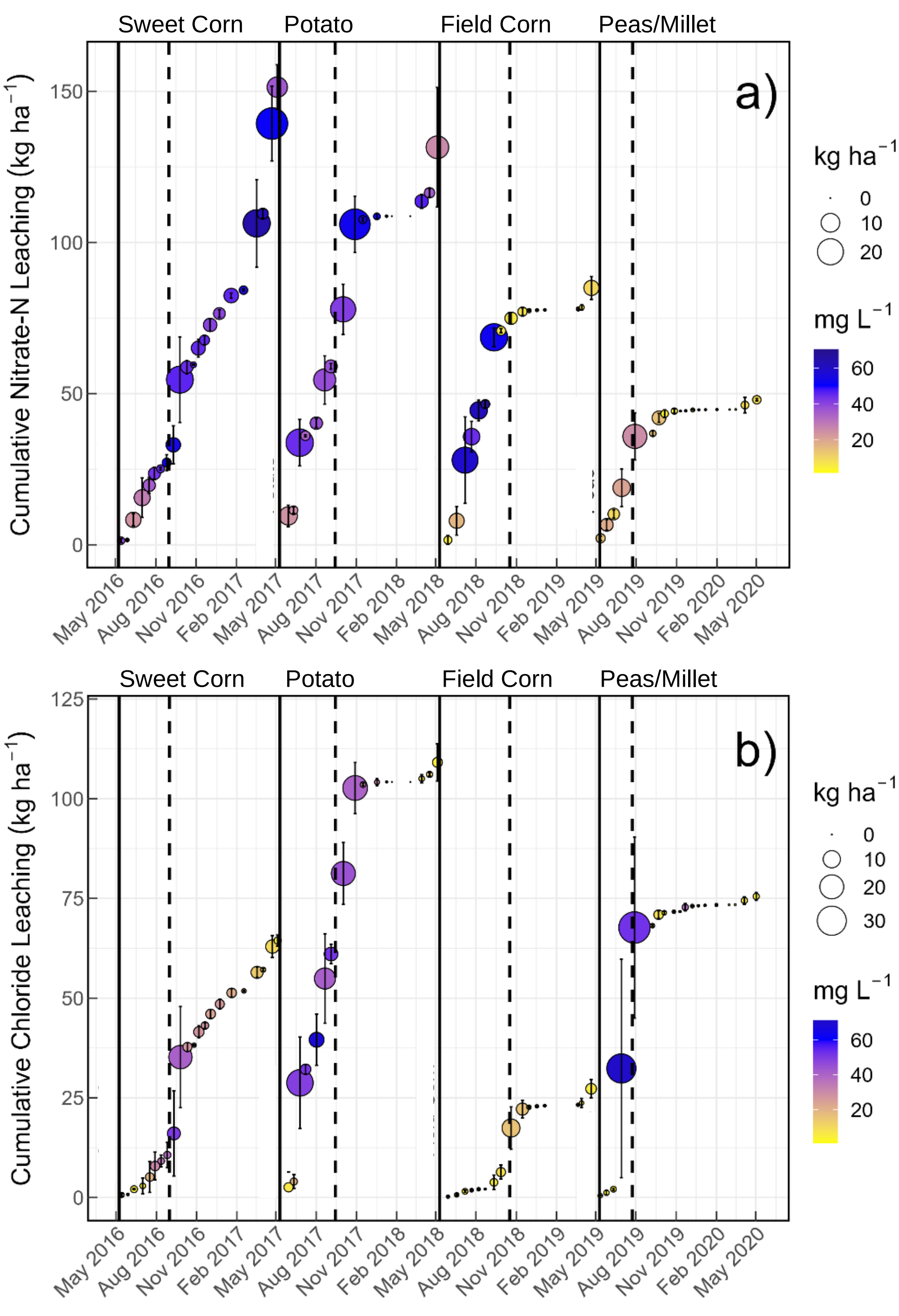


Figure 4. Cumulative nitrate (a) and chloride (b) leaching losses for each cropping season (1 May – 30 Apr). Size of circle corresponds to respective solute leaching loss and error bars denote standard deviation of the four lysimeter measurements for each period. Solid vertical line represents 1 May and dashed line indicates harvest date.



Chapter 3

Investigating the use of interplanting for reducing nitrate loss to groundwater below potatoes

Abstract

Managing nitrate leaching in commercial potato fields is an ongoing challenge because of the coarse textured, well-drained soils and rates of nitrogen fertilizer used in commercial systems. Drainage and solute losses can be exacerbated by the row and furrow configuration of these systems; however, the furrows also represent an unoccupied niche that provide opportunities for interplanting of companion vegetation for potential water quality benefits. Barley (B), oats (O), and millet (M) were interplanted in the furrows between potato rows and measurements collected to determine the impacts on yield, biomass accumulation, and nitrogen uptake. Utilizing a 5x5 Latin-square design used to investigate interplanting treatment effects, no differences in potato yield were observed between the control and interplanting treatments. Barley showed the greatest potential for nitrogen uptake followed by oats. Nitrogen removal represented nearly 10-13% of annual nitrogen fertilizer application. Total nitrogen uptake was greater in all interplanted plots than the control, however, differences were not considered significant ($p = 0.147$). Mean nitrate-nitrogen (46.1 mg L^{-1}) and chloride (35.4 mg L^{-1}) in groundwater following potato were greater than nitrate-nitrogen (17.8 mg L^{-1}) and chloride (16.9 mg L^{-1}) observed below fields when they were planted into peas followed by a millet cover crop. Results were inconclusive that interplanting improves water quality, however we feel improvements to interplant establishment could result in more nitrogen uptake and are worthy of additional experimentation.

3.1 Introduction

Potatoes are a small percentage of cropland acres statewide in Wisconsin; however, they represent 20% of the irrigated landscape of the Central Sands Region. A review of nitrate leaching studies of the upper Midwestern US showed potato to have the highest rates of nitrogen losses among crops investigated (Shrestha *et al.*, 2023). A recent study in Portage County, WI revealed that 24% of wells tested greater than the 10 mg/L EPA drinking water standard; nearly 2.5 times the statewide average. Analysis of the well water data showed that potato/vegetable production acreage and excessively drained soils were the best predictors for elevated nitrate concentrations (Masarik, 2023).

Data summarizing leaching losses over the course of the rotation in a central Wisconsin irrigated commercial potato/vegetable production field revealed that potato generally resulted in greater leaching losses and greater groundwater nitrate concentrations relative to other crops (Masarik and Kucharik, manuscript in preparation). In work by (Nocco *et al.*, 2018), drainage during the growing season was found to be greater below potato than other crops. Commercially grown potatoes rely on hill/furrow production systems and are provided supplemental water via overhead irrigation of groundwater. Because potatoes concentrate most of their root mass and tuber development in the hill portion of the field, furrows generally have little to no active root system to remove water from the soil profile and remain bare for the growing season. Water movement in this system is not uniform and changes during the various growth stages of potato development. Initially stems help route water into the hill toward roots. However, as the vines drop and the canopy begins to open, hydrophobic conditions that limit water infiltration into the hill and promote the shedding of water to furrows have been documented (Saffigna, 1976; Robinson, 1999).

Strategies such as drip irrigation or the use of surfactants have been proposed to increase water use efficiency and yield by limiting the negative impacts from hydrophobic conditions that result later in the growing season (Cooley *et al.*, 2007; Arriaga *et al.*, 2009). The non-uniform water movement combined with a lack of an active root zone in the furrow results in an unoccupied niche that may provide opportunities for reduction in nitrate leaching losses.

Cover crops have been investigated for their potential to scavenge nitrogen and have been shown to reduce drainage nitrate leaching during fall and early spring (Thapa *et al.*, 2018; Meyer *et al.*, 2020). Other research has shown that mixtures of cover crops can be used to balance tradeoffs between N retention and supplementing N to the next crop (Kaye *et al.*, 2019). Less is known, particularly when it comes to potatoes, about the use of interplanting of companion plants during the growing season to aid in water quality improvements. One study that investigated intercropping of cabbage with potato showed that total nitrogen uptake was not significantly different for the treatment (cabbage + potato) versus the control (potato only); here the hypothesis was that cabbage could take advantage of unutilized solar radiation and other resources prior to canopy closure of potato vines (Opoku-Ameyaw and Harris, 2001). Another study looked at intercropping corn with potatoes on small scale systems in Africa and the impacts on productivity and profitability (Kidane *et al.*, 2017). There is a general lack of published studies that investigate the question of plant competition and nitrate reduction potential from vegetation that looks to take advantage of unoccupied niches and resources during the periods after hilling through canopy closure, bulking, and vine kill; particularly when it comes to impacts on water quality.

A cover crop grown in furrows between hills that only minimally competes for soil moisture or sunlight, could potentially assimilate nitrogen that would normally leach past the root zone of the potato crop. This is especially true later in the growing season when above ground biomass begins to transfer energy to tubers during vine senescence and opportunities to utilize solar radiation, moisture, and nutrients from furrows should increase without significant competition to potato crop. If nitrogen in the plant residue could subsequently be credited to the following crop, this could reduce overall nitrogen fertilizer inputs and lead to improved groundwater quality the following year as well.

The primary objectives of this research were to 1) quantify differences in potato yield and nitrogen uptake potential between control and plots where interplanting of companion crops occurred, 2) investigate water movement, temperature, and solute leaching dynamics beneath conventional and interplanting treatment, and 3) determine whether interplanting of companion vegetation can be used to reduce solute leaching (i.e., nitrate-nitrogen/chloride) to groundwater.

3.2 Materials and Methods

3.2.1 Study Site

A commercial farm south of Plover, WI was the site of this experiment. Potato (*Solanum tuberosum L.*) at this farm is grown once every three to four years as part of a rotation that includes sweet corn, field corn, and peas. The region has a humid continental climate with an annual temperature of 7.3°C and 870 mm of precipitation. The fields utilized for this experiment contained Friendship or Richford loamy sand (moderately well drained, mixed, frigid Typic Udipsamments, USDA Classification) soil series with <3% slopes. The soil profiles consist of a loamy sand Ap horizon (0-25 cm), a sand Bw horizon (25-75 cm) underlain by a sand C horizon (>75 cm). Sand with occasional layers of coarse sand and gravel can be found at depths greater than 75 cm to the water table. The water table fluctuates but generally is encountered at depths of

300-400 cm. Temperature and precipitation data for the each of the growing seasons were downloaded from PRISM (PRISM Climate Group).

Potatoes are generally not grown in successive years; as a result different fields were used to establish experimental plots for each year of the study (Figure 1). Field G (2020) and Field H (2021) were preceded by peas and a pearl millet (*Pennisetum glaucum*) cover crop. Pearl millet is used by this particular farmer for its potential as a biofumigant for nematode suppression. Pearl millet terminates at the first hard frost. In Field G the stand was not incorporated into the soil until 20 April 2020, whereas Field H incorporation occurred on 9 October 2020 shortly after frost terminated the stand. Timing of residue incorporation and subsequent mineralization may have implications for drainage and solute leaching.

Seed potatoes were planted in rows spaced 0.9 m apart. Fertilizer application rates, planting dates, irrigation events, and other important information were collected through observation or personal communication with the landowner (Table 1). Nitrogen fertilizer was applied as starter (58 lbs N ac^{-1}), one side-dress application at hilling (84 lbs N ac^{-1}), and two fertigation events supplied the remainder of nitrogen (88 lbs N ac^{-1}). Interplanting was performed following second hilling using a one row hand planter (Earthway 1001-B Precision Garden Seeder). Applications of insecticides, herbicides, and fungicides were typical for commercial potato production of the region. This includes vine killing, in which a desiccant was applied 7-21 days prior to harvest, and is beneficial for harvestability and long-term storage of tubers. Table 1 contains relevant dates and activities pertinent to potato production and the other experimental activities.

3.2.2 Plot establishment

A 2-hectare control plot and a 2-hectare treatment plot were established within a commercial potato field, Field G Control (GC), Field G Treatment (GT) in 2020 and Field H Control (FC) and Field H Treatment (HT) in 2021 with the intent of comparing water quality and estimating leaching losses in conventional practices compared to interseeding. In 2020, the treatment plot consisted of a mix of Japanese millet (*Echinochloas spp.*), rye grass (*Secale cereal*), and oats (*Avena sativa*) interseeded in furrows between the rows of a conventional potato planting. In 2021 rye was replaced with barley (*Hordeum vulgare*) and German millet (*Setaria italica*) was substituted in place of Japanese millet. Interseeding took place following second hilling in both years. The control plot was adjacent and upgradient of the treatment plot.

In 2021 a small section of Field H was used for the establishment of twenty-five small plots (15 m x 15 m). Plots were arranged in a Latin square design (Control (n=5, C1-C5), Millet (n=5, M1-M5), Oats (n=5, O1-O5), Barley (n=5, B1-B5), Millet/Oats/Barley (n=5, MOB1-MOB5)) to investigate the impact of individual cover crop species on potato yield, quality, and harvestability (Figure 1). The Latin square design uses the rows and columns as blocking factors which help control variability that can result from field rows or other soil variability that may impact productivity within the field. An attempt was made to repeat the interplanting experiment again in 2022 on a different field, however the need for an herbicide application following second hilling and miscommunication with the grower resulted in termination of MOB, B, O, and M treatment effects and the experiment was abandoned for the year.

3.2.3 Sensor measurements and drainage estimates

Meter Group Teros 12 sensors and Meter Group ZL6 dataloggers (METER Group, Inc., Pullman, WA, USA) were used to collect measurements of soil water content, soil temperature, and bulk soil electrical conductivity from the control and treatment plots. Each plot had a total of twelve sensors installed at replicated depths below the hill and furrow (Figure 2). The sensor network collected measurements every 10 minutes and allowed for detailed characterization of soil drainage and solute leaching below the hill and furrow. Because these sensors measure soil water content, bulk electrical conductivity (EC), and soil temperature in one sensor, they can convert bulk EC into pore water EC. The solute concentration (pore water EC) is determined using the linear relationship between bulk dielectric permittivity and pore water EC (Hilhorst, 2000; METER Group, Inc., 2018). Pore water EC measurements cannot distinguish nitrate from other solutes in the soil matrix, however comparing even relative measurements from different depths within a soil profile can provide insight into the movement of fertilizer in the soil profile (METER Group, Inc., 2018). Differences between the control plot and treatment may provide insight into potential benefits that interplanting may play in reducing water drainage and solute drainage below the root zone.

Drainage was estimated by measuring changes in soil water content from the -20 and -40 cm soil profile sensors. These depths occur in the layer below the Ap horizon where little to no root growth was observed when removing sensors. It was assumed that water reaching the sensors was beyond the influence of the potato plants and would eventually reach the groundwater. Water content measurements from the -20 cm and -40 cm soil sensors were used to estimate the soil water storage for the -15 to -45 cm depth profile at each measurement interval. The decrease in soil storage following wetting events was assumed to be drainage and these values were summed to determine the cumulative drainage during the growing season.

3.2.4 Measuring biomass and nitrogen removal

Weekly measurements of interplanting vegetation height and percent canopy were collected from each plot to investigate differences in the growth patterns and phenological stages of each type of vegetation. Potato vines were removed from three adjacent potato rows or an area with dimensions of 1.52 m x 1.83 m (5 ft x 6 ft). Removal of vines from a small area of each plot allowed for measurement of interplant vegetation with and without competition from potato.

Percent canopy cover measurements were collected using a Samsung Android phone and Canopeo App. The Canopeo App uses the cell phone camera to take an aerial picture above a portion of the field. The camera was held at a 90-degree angle 1.3 m above the furrow surface. The app analyzes pixels for greenness which is converted into a black (non-green) and white (green material) image that determines canopy cover based on the ratio of black to white (Patrignani and Ochsner, 2015; Wang and Naber, 2018). An Afidus time-elapse camera was also installed in the field to monitor progress of the interplanting treatment throughout the season; the video of the 2021 growing season is available online at: https://youtu.be/X90X35PTdYo.

Following vine kill but prior to commercial potato harvest, above ground biomass of potato residue, interplanting vegetative growth, and tuber yield were measured and collected from 3.05 x 1.83 m sites selected from within control and treatment plots. As the result of poor establishment in 2020, a decision was made to sample for interplant biomass potential; whereas 2021 sample locations were randomly selected. Samples were weighed in the field using a hanging scale and also taken to Hancock Agricultural Research Station for grading. A subsample of plant biomass and potato tuber were collected, dried, and ground for total carbon and total nitrogen analysis. In 2021, the same measurements were made in all C, M, O, B, and MOB plots.

Measurements of pearl millet biomass were collected from HC and HT plots in fall of 2020. Pearl millet is used as a cover crop following peas and we wanted to better understand the nitrogen uptake potential of the millet and how timing may influence the availability of nitrogen to potato the following year. Above ground biomass was removed from a 1 m x 1 m square where it was weighed in the field with a hanging scale and a subsample was collected, dried, and ground for analysis.

All tissue samples were prepped and analyzed for total nitrogen and total carbon using a Perkin Elmer 2400 CHNS/O Series II System. Biomass and N content measurements were used to determine N removal associated with different components of potato and interseeded vegetative types.

3.2.5 Groundwater quality

Previous work in the Central Sands that used 15 x 15 m plots was confounded by high variability of the groundwater quality inherent to the agricultural field and the influence of groundwater flow from outside plot boundaries (Bero *et al.*, 2016). These challenges illustrate the importance of large plots for investigating treatment effects on groundwater quality. The plot size (2 hectares) for GC/GT (2020) and HC/HT (2021) plots was intentionally large in order to allow for sampling capable of accounting for inherent in-field variability and avoid intrusion of groundwater flow from outside the plot boundary.

Samples were collected from each plot prior to field operations for the purpose of establishing the baseline water quality prior to the experiment. Eleven sampling locations within the plots were chosen using a random sample technique. In previous work at this farm, a sample size of 11 was determined to be the minimum necessary to detect a 20% difference in means using a power of 0.80 (probability of correctly finding an effect that is there) and Type I error rate (probability of finding an effect that is not there) of 5% for a 2-sample, 2-sided equality

comparison (Masarik, 2016). This method was specifically developed with the goal of quantifying differences in groundwater quality between a control plot and treatment.
When selecting sampling locations, an area far enough from the treatment border to avoid intrusion of groundwater from outside plot boundaries was divided into 256 grid cells. Eleven numbers between 1 and 256 were chosen at random and the centroid of each grid cell selected was used as a sampling location. The same process was used to select and sample sites the following spring to evaluate the treatment effects of the experiment on groundwater quality. Due to the narrow window for sampling in 2021, we were not able to sample GT before the field was planted.

Water quality sampling was performed using a bucket auger to excavate down to the water table. The water table was encountered at depths ranging from 300-400 cm and underlying sand and gravel materials are excessively well drained. A temporary well consisting of 3.175 cm I.D. polyvinyl chloride (PVC) with a 100 cm screen and plastic point was installed and used to sample water from the top of the aquifer. We estimated that one year of annual recharge (33-41 cm) combined with porosity of 0.395 cm/cm for sand soil (Clapp and Hornberger, 1978) would be expected to occupy the upper 84-104 cm of the aquifer. The well was screened from 0-100 cm below the water table (Figure 3).

A 0.25 in. O.D. and 0.175 in. I.D. polypropylene tubing was inserted near the bottom of the temporary well. A peristaltic pump was used to remove water from the well for sample collection. The well was pumped sufficiently so that suspended sediment noticeably decreased and water temperature stabilized. Samples were field filtered using an inline filter cartridge and glass microfiber filter and 0.45 μm. Samples for cation analysis were collected in a HNO_3 acid preserved 15 mL vial. Samples for nitrate-N and chloride were collected in an unpreserved

HDPE 125-mL bottle. All samples were stored on ice until delivery to the laboratory.

All samples were analyzed at the UW-Stevens Point Water and Environmental Analysis Laboratory. The lab has a formal quality control program in place and holds certification from Wisconsin (DNR State Certification Lab No. 750040280) for a wide-array of elements and matrices. Among the practices that the laboratory employs are periodic analyses of laboratory reagent blanks, fortified blanks, duplicate samples, and calibration solutions as a continuing check on performance. Nitrate-N and chloride were measured colorimetrically by flow injected analysis on a Lachat QuikChem 8000 (Lachat, Milwaukee, WI).

3.2.6 Statistical Analysis

All summary statistics and tests were computed using the R Programming Language (R Core Team, 2022). Differences in yield, above ground biomass, nitrogen uptake, and groundwater quality of control and treatment plots within the same field and year were compared using a 2-sample, 2-sided comparison with unequal variances, a value of $\alpha = 0.05$ was used when determining significance.

Analysis of variance (ANOVA) for a three factor Latin square design was used to understand differences in potato yield and total nitrogen uptake between small plots.

$$X_{ijk} = \mu + T_i + \rho_j + \beta_k + \varepsilon_{ijk} \quad \text{[Eq. 6]}$$

Where, X_{ijk} is the measurement of treatment i, row j, and column k, μ is the mean of all experimental units, T_i is the effect of the treatment (ex. Control, Millet, Oats, Barley, M/O/B), ρ_j is the effect of row j, and β_k is the effect of the column k, and ε_{ijk} are random errors. If ANOVA detected differences between treatments ($p < 0.05$), post hoc Fisher's Least Significant Difference (LSD) test was performed to determine which of the treatment means were different.

Unprotected Fisher's LSD was performed on measurements where ANOVA did not detect differences between treatments.

3.3 Results

3.3.1 Interplanting, crop yield, and nitrogen uptake

Mean tuber yield was 281 and 311 cwt respectively in 2020 and 2021. Significant differences ($p < 0.01$) in tuber yield were observed between control and treatment plots in 2020 (GC/GT) and 2021 (HC/HT) (Table 2). Interplant dry matter was 1,632 lb ac^{-1} in 2020 (GT) and 561 lb ac^{-1} in 2021 (HT); however, sampling locations in 2020 were biased towards areas of successful establishment and are not representative of overall plot average. Observationally, the interplanting was more uniform in 2021 and interplant biomass from this year is a more accurate reflection of performance over the entire plot than what was observed in 2020 (Picture 1). Nitrogen uptake in vines was 22 and 30% of tuber N uptake in 2020 and 2021, respectively. Total nitrogen uptake (tubers + vines + interplant) was not significantly different between large control and treatment plots in either year.

Analysis of small plot tuber data contradicted the yield comparison between HC and HT. Here, no statistical differences in tuber yield was observed between the control (C) and any of the interplant (B, O, M, MOB) treatments (Table 3). Interplant dry matter and nitrogen uptake were greatest in B, followed by MOB, O, M, and C. Both M and C had significantly less interplant biomass and interplant N uptake than the other three plots. Total N uptake was greatest in B plots but was not significantly different from any of the other treatments.

Interplant vegetation height was greatest in O and MOB plots, followed by B, M and C (Figure 4). Height in the no competition (NOCOM) plots taken weekly during the growing season plateaued in early August. Vegetation height in plots where potato canopy was preserved remained at approximately 25 cm for a one-month period from mid-July through mid-August. Small gains in vegetation height of the COM plots were observed until harvest but were approximately half of what was observed in NOCOM plots. Percent canopy cover in NOCOM plots never reached greater than 50%; meanwhile COM plots peaked around 100% in early July and is attributed almost entirely to potato canopy cover (Figure 5).

Measurements of pearl millet residue collected in fall 2020 from HC (n=5) and HT (n=5) to understand potential mineralizable nitrogen from the cover crop were pooled because the two plots up to that point had been farmed as a single unit. Pearl millet dry matter measured 3.0 tons ac^{-1}, had a mean N content of 2.6%, and total N uptake of 153 lb ac^{-1}.

3.3.2 Sensor measurements and drainage estimates

Mean soil temperatures decreased as the sensor depth below the soil surface increased (Tables 4a, 5a). The hill sensor (+10 cm) temperature closely followed diurnal air temperature and solar radiation patterns (Figures 8, 9). Mean and maximum temperatures in the hill were on average 0.2 and 4.4 degrees C greater in 2021 compared to 2020. After vine kill in 2021, soil temperatures in hill showed lower maximum daily temperatures in the HT than in HC (Figure 9). For the period preceding vine kill (12 Aug 2021), soil temperatures in the hill averaged 21.6 and 21.5 °C and after vine kill were 24.6 and 23.9 °C in the HC and HT plots respectively or 0.7 °C cooler in plots with interplanting.

Mean and maximum water contents were greater in sensors beneath the furrow than the equivalent sensor depth below the hill. Mean water contents were greater in the hill (+10 cm sensor) and -10 cm sensor depths than the -20 and -40 cm sensors (Tables 4b, 5b). Noticeable changes in the soil profile occur between the -10 cm and -20 cm sensors where the boundary between the Ap and Bw horizons was located (Figure 10).

When evapotranspiration is negligible and upward movement of water due to matric potential gradients is limited, we assumed that decreases in water storage at the -15 to -45 cm depth were due to drainage (Figure 11). Estimated growing season water drainage was similar between control and treatment plots during both years when sensors were installed. Water drainage in GC and GT was estimated to be 8.8 and 8.2 cm for the period from 28 June 2020 – 30 Aug 2020, while drainage estimates for HC and HT were 14.3 and 12.2 cm for the same time period the following year (28 June 2021 – 30 August 2021).

Summary statistics for the median EC values are provided in Tables 4c and 5c. The EC measurements at some locations occasionally behaved erratically, particularly when soil water content was below 0.10 cm cm^{-1}. The median rather than the mean were summarized because of some extremely high measurements that likely resulted from the erratic measurements and sensor anomalies. Erratic measurements were most noticeable in the hill (+10 cm) and observed more frequently during the 2020 growing season when conditions were dry. Median EC was consistently less below furrows relative to the corresponding sensor depth below hills.

3.3.3 Groundwater quality

Baseline nitrate-nitrogen and chloride concentrations measured prior to planting (Apr 2020 & Apr 2021) were not significantly different between control and treatment (Table 6). Mean nitrate-nitrogen and chloride concentrations both increased following potato (Apr 2021 & May 2022). The resulting nitrate-nitrogen and chloride increases were greater in Field G than were observed the following year in Field H.

3.4 Discussion

3.4.1 Interplanting effects on yield and nitrogen uptake

Barley showed the greatest potential for nitrogen uptake followed by oats. Nitrogen removal represented nearly 10-13% of annual nitrogen fertilizer application and could represent a significant potential reduction of nitrate leaching losses to groundwater. The three species mix (MOB) was dominated by barley and oats; suggesting that the inclusion of oats may have resulted in less biomass production than barley alone. Meanwhile millet was barely evident and had little influence within the MOB plots. With adjustments to establishment and further experimentation we believe it is possible to have greater accumulation of above ground biomass and nitrogen uptake using interplanting with potato.

Plant height and % canopy cover measurements provided additional insight into interplant progression during the growing season. Starting on 14 July (approximately one month after interplant establishment) differences in plant height between no competition (NOCOM) versus plots with potato competition (COM) become noticeable. Divergence corresponds with peak canopy cover in COM plots (Figure 5) and suggests lack of sunlight limited interplant growth during this time. As potato canopy begins to senesce, increases in interplant vegetation height resume.

Browning that occurs after vine kill is reflected in the decrease of percent canopy cover observed on 17 August measurements. The increases in canopy cover observed following the 17 August measurements are the result of regreening of the interplant vegetation and support observations that vine killing did not terminate the interplanted vegetation. Interplant height in NOCOM plots remained steady following 17 August; without competition from potato canopy interplanted vegetation may have reached maximum height prior to vine kill.

Vegetation height in COM plots however continued increasing until harvest. The COM plots at harvest ranged from 50-77% for the different treatments. Because not all of the plants reached physiological maturity prior to harvest, we suspect that if harvest were delayed or the interplanting would have occurred earlier than it did, interplanted vegetation would have continued to grow and add biomass. Experimenting with interplanting on other fields with only one hilling would allow for earlier interplanting establishment and potentially greater benefits if able to produce greater and taller biomass prior to potato canopy closure.

If the interplanted vegetation did compete with potato for water, nutrients, or sunlight, it does not appear that effects were detrimental to potato yields. While no treatment effect was observed, there was a significant row effect that may have resulted from agricultural operations or variability of in-field soil properties. While not considered significantly different, tubers in barley plots resulted in 63 lbs more nitrogen uptake than the control plot. This is the result of tubers in the barley plots measuring 3.0% N compared to 2.0% N in the control plot. The % N differences warrant further investigation as an effect of interplanting on the overall nitrogen content of tubers. Higher protein levels in potatoes are desirable from a food and nutrition standpoint. If the greater total nitrogen corresponds to increased protein nitrogen, then this could be an added benefit to utilizing interplant vegetation such as barley during commercial potato production.

3.4.2 Interplanting species selection

The first year we attempted the experiment, the interplanted vegetation performed poorly. Initial millet germination looked good in 2020; however millet does not tolerate shade (Midwest Cover Crop Council, 2014). Lack of sunlight once potato canopy closure occurred likely resulted in low survival of the millet. Cultivation anomalies which allowed for more sunlight and less canopy competition likely contributed to a few rows of very successful millet establishment. The 5,958 lb ac^{-1} of biomass produced by the millet in those few rows provided optimism that an earlier planting might allow vegetation to grow taller before canopy closure and allow for greater success in subsequent trials. As a result of the potential biomass accumulation that we observed, millet was included in the species selection for 2021.

Emergence of oats was robust and uniform across the plot and resulted in the best overall survival rate of the three species used in the interplanting mixture for 2021. Rapid growth is a trait assigned to oats that likely contributed to success even with competition from the potato canopy (Midwest Cover Crop Council, 2014).

Rye was initially part of the species mix in 2020. Emergence of rye also started off well; however rye is not heat tolerant and in hindsight is likely poorly suited for this particular application (Midwest Cover Crop Council, 2014). With planting occurring in the middle of June just prior to canopy closure, very little of the rye survived beyond vine senescence. Barley was suggested as an alternative to rye and was used in its place during the 2021 season (Ken Schroeder, personal communication). Barley emergence and growth characteristics were similar to oats and appear well suited for this particular application.

3.4.3 Water movement, pore water EC, and soil temperature

The Ap horizon has a higher percentage of silt and organic matter that the underlying subsoil (Nocco *et al.*, 2018), and would result in greater water holding capacity resulting from differences in soil texture. This is supported by the -10 cm sensor measurements that showed consistently higher water contents than sensors at deeper depths in the Bw horizon.

Hill water content measurements generally showed a muted response to wetting events compared to the furrow sensor counterparts. Water ponding in the furrow was often observed and concentrated flow beneath the furrow was confirmed by sensor measurements. This was particularly evident during the 2020 season when hot dry conditions early in the year resulted in what were likely hydrophobic conditions observable in the water content measurements starting around 13 July 2020; wetting events detected at the -10 cm depth below the hill were not observed or were extremely muted in sensor located in the hill. Hydrophobic condition have been observed in other studies that have investigated water movement in potato systems (Cooley *et al.*, 2007). Hydrophobic conditions would be expected to result in less infiltration of water into the hill and route more water into furrows where it will focus infiltration below the furrow.

Starting around vine kill on 12 Aug 2021 there was a two-week period when no wetting events were observed in the soil water content measurements. During this period, the diurnal fluctuations and overall decline in water content is more evident in the hill and furrow measurements at the -10 cm depth of sensors installed below interplanting (HT) than the corresponding sensors in the control plots (HC). We view this as evidence that vegetation in the furrow was actively utilizing water stored in the soil to the -10 cm depth. The lack of a stepwise diurnal pattern of water contents at the -20 and -40 cm sensors suggests that plant roots are not actively interacting with soil moisture at these depths. We assume that water that moves beyond the Ap horizon is likely beyond the reach of plant roots and will ultimately drain to groundwater.

Using changes in water storage measured by sensors in the Bw horizon, we determined average drainage in Field G (2020) was 1.3 mm d^{-1} and in Field H (2021) was 2.1 mm d^{-1} for the respective periods of measurement. Differences in precipitation are at least partially responsible for these differences; 345 mm greater precipitation occurred during the 2021 growing season compared to 2020. Larger drainage events were associated with precipitation driven drainage rather than irrigation. Unsurprisingly precipitation amounts influenced the magnitude of drainage of individual events as well as over the growing season. The 2021 season saw twelve precipitation events greater than 25 mm compared to the six that occurred in 2020 (Supplemental Materials). The largest drainage event measured in either year was 2.5 cm and occurred on 28 August 2021 following an 83 mm precipitation event. Interplanting in 2021 did result in less drainage than the control plot, however there was only one monitoring location per field and future investigation into this question would benefit from replication.

Measurements of EC under extremely dry conditions that occasionally occurred in the hill were not reliable. Growers looking to utilize EC measurements to understand the relative amounts of solutes available to plants in potato hills may find it difficult to interpret real-time measurements and rely on the data to make informed fertilizer management decisions during the season. However, EC measurements at -10, -20, and -40 cm were more stable and may hold potential for understanding the translocation of solutes beneath the hill and furrow throughout the season.

Increases in EC at the -10 cm depth on 1 Aug 2021 observed below the furrow may be the result of fertigation (Supplemental Materials). Fertigation is known to have occurred on 29 July 2021, two days before a wetting event was detected in water content measurements. Increases in water content correlate to the increase in EC measurements at the -10 cm sensor. Another wetting event a couple of days later coincides with an increase in EC measurements at the -20 cm depth.

Starting on 8 Aug 2021, a total of 89 mm of rainfall fell across a two-day period (Supplemental Materials). This event initiated significant water movement through the entire profile and coincided with EC decreases at both the -10 and -20 cm depths and an increase in EC at the -40 cm depth. These data appear to show significant solute leaching events following a gradual movement of solutes down through the profile from a fertigation application and smaller wetting events. While dates of fertigation are not known for 2020, EC measurements starting 17 July 2020 may show a similar more gradual translocation of solutes in the soil profile beneath the furrow.

Greater mean pore water EC measurements in the hill reflect concentration of solutes resulting from early season fertilizer applied as side-dress or incorporating fertilizer into potato hills during the hilling process. The EC measurements beneath the hill portion remained higher throughout the season suggesting that solutes within the hill may not be as prone to leaching as those applied to the furrow and is supported by other studies that have cautioned of significant pools of nitrogen that remain in the potato hills which can be redistributed and leach following harvest (Bohman *et al.*, 2020). This type of information may help growers avoid rescue applications of nitrogen if they can see evidence that solutes remain in hills following rain events.

Meanwhile, pore water EC measurements in the furrow were observed to decrease following the first few drainage events following sensor installation. The data demonstrates concentrated drainage in the furrow flushes any solutes in this part of the profile more readily than what is observed beneath the hills.

Minimum and maximum daily soil temperatures near the surface more closely reflected the diurnal fluctuations. The magnitude of diurnal fluctuations and seasonal temperature patterns decreased with depth but were still evident at -40 cm. One noteworthy difference that may indicate interplant treatment effects was observed in 2021; when the maximum daily soil temperature following vine kill was on average 2.4 °C cooler in HT versus HC. This potentially demonstrates that above ground biomass from interplanting may be providing a soil cooling effect by shading soil that would normally absorb significant solar radiation following vine kill.

3.4.4 Groundwater quality

Baseline nitrate-N and chloride concentrations of groundwater quality comparing the control and treatment plots in 2020 and 2021 showed no significant differences following pea/millet. This provided confidence that groundwater quality beneath the plots was essentially the same prior to starting the interplanting experiment each year. One notable difference between the fields is that nitrate-nitrogen following the pea/millet water year beneath Field H was significantly greater than was observed the previous year on evaluating pea/millet beneath Field G. One potential explanation involves the millet cover crop and timing of mineralization. In Field G where millet was returned to the soil in the spring, there was an extremely small window between residue incorporation and water sampling. Meanwhile in Field H, millet residue was incorporated in the fall and provided more time for mineralization and subsequent leaching prior to water sampling. These results highlight the need for additional research on residue mineralization differences resulting from termination of cover crops and the timing of incorporation.

Nitrate and chloride concentrations measured in the groundwater evaluating the potato water year were significantly greater than measurements evaluating the pea/millet water year. Increased concentrations following potato were expected and support other studies that have shown potato to leach greater amounts of nitrate and chloride relative to other crops (Saffigna and Keeney, 1977; Stites and Kraft, 2001). Potato have higher recommended rates of both nitrogen and potash (Laboski and Peters, 2012). Despite a variety of recommended best management practices for potato, there is no one practice or combination of practices that is capable of eliminating these losses completely (Shrestha *et al.*, 2010).

Interplanting has not been evaluated extensively for its ability to reduce nitrate leaching and our hope was that the large plots would be capable of detecting and demonstrating potential groundwater quality benefits at some scale. By harvest of 2020, it was obvious that the interplanting was not as successful as we may have hoped; and that we would have to improve on interplant establishment and success if water quality benefits are to be achieved. Because significant groundwater quality benefits from the effects of interplanting were not likely, we used this as an opportunity to estimate potential interplant biomass and sampled from locations where interplant vegetation height and uniformity was noticeably better. As a result, interplant biomass measurements from 2020 are biased and not comparable to random samples collected the following year.

Lower potato yields observed in GT compared to the GC plot should not be interpreted as the interplanting negatively impacting yield, although that possibility is also not ruled out. We think it plausible that poor potato plant health which would have resulted in lower yields may also have allowed interplant vegetation to perform better at those locations. Stressed plants may have less extensive canopy cover which allowed interplanting to compete better for sunlight than other parts of the field. Lessons learned in 2020 were applied to the following year.

Interplant biomass was less than what we measured the previous year, but coverage was more uniform in 2021 as evident by the lower standard deviation compared to measurements from 2020 (Table 2). Observationally, the interplanting performed significantly better across the entire 2-hectare plot than the prior year (Picture 1). Yield was 41 cwt less in the treatment plot

(HT) consisting of interplanted barley, oats, millet than the control (HC). These results are contrary to our other results from the Latin square design plots showing no difference in yield between any of the five treatments. Contradictory yield results could partially be attributed to lack of plot replication. The groundwater plots were intentionally large because of the goal of measuring water quality below the plot. However, with only one treatment and one control for comparison, we were not able to control for other factors such as soil variability. Additional plots that allow for randomized block structure would add significant cost to experiments investigating groundwater quality but should be considered to avoid similar concerns with future investigations.

Nitrate-nitrogen and chloride were significantly greater in the Field H treatment plot than the control plot. The nitrogen budget could provide clues as to why; the control plot did have greater tuber nitrogen and greater overall N uptake than was measured in the treatment plot. More N uptake under the same nitrogen inputs would be expected to leach less nitrate and support the groundwater quality results; however, we also had to consider dilution as another possibility.

In potash, chloride is a companion ion to potassium that behaves similarly to nitrate ions. Both ions are negatively charged and have little affinity for soil. Chloride is generally regarded as a conservative and inexpensive tracer because it is not as biogeochemically active as other elements in agricultural systems (White and Broadley, 2001; Svensson *et al.,* 2021). If nitrogen had greater utilization in one plot versus the other, we might expect the ratio of nitrate-nitrogen to chloride to be different between the plots. However, the ratios were essentially the same between plots. This could point to dilution as another potential explanation for lower concentrations observed in the control plot. Greater drainage with similar losses of solute would result in lower concentrations of both ions in the groundwater. Sensor measurements observed more drainage beneath the control plot during the growing season, however it was only one location. And because sensors do not provide accurate water content measurements during frozen conditions, we were not able to quantify the drainage between harvest and spring sampling which is often significant. While greater yields in the control plot remain a plausible explanation for water quality differences, we cannot rule out drainage as the reason for differences in water quality. The lack of a clear or conclusive explanation is another example of the importance of plot replication in the initial study design.

3.5 Conclusions

Managing nitrate leaching in commercial potato fields is challenging because of high nitrogen fertilizer recommendations and coarse textured, well-drained soils. Leaching of solutes is exacerbated by the row and furrow configuration of these systems. An increased frequency of larger and more intense precipitation events predicted for the upper Midwestern US will make these challenges even more important to address moving forward. Investigating unoccupied niches in these systems both during and outside of the growing season may provide opportunities for water quality improvements, however it is important to ensure that they can be economical to perform.

Interplanting of companion crops in the furrows between potato rows did not negatively impact yield. Oat and barley interplanting treatments suggested 24 and 31 lbs ac^{-1} of N uptake as the result of interplanting however the total N uptake was not considered significantly different from conventional methods. Sensor arrays provided important insights into water, temperature, and solute dynamics between rows and furrows, treatments, as well as water leaching dynamics resulting from climatic variability from year to year. While further experimentation is needed to replicate results that demonstrate viability and assess water quality benefits in other years and on other farms, there is optimism that modifications to timing and species selection could significantly improve on the biomass and nitrogen uptake rates measured by interplanting in this study. For instance, additional experimentation with interplanting on fields that utilize one-hilling instead of two would allow for earlier interplant establishment and is anticipated to result in more biomass and nitrogen accumulation.

Achieving actual water quality benefits from interplanting relies on a greater understanding of what happens when the residue is returned to the soil. How long does it take to mineralize and does the nitrogen that is returned to the soil leach prior to the following growing season? If interplanting is viable in commercial potato fields, then there are a series of additional research questions that investigate actual water quality benefits from these practices.

The continued use of split side dress applications, slow-release fertilizers, and multiple fertigation applications that limit how much nitrogen is applied at one time, as well as other strategies that minimize the total amount of nitrogen applied should continue to be used and encouraged. However, the significant challenges of managing nitrogen in potato cropping systems mean that all options must be on the table. Management strategies that investigate unoccupied niches during the growing season, outside of the growing season, and throughout the rotation are critical if we are to make meaningful improvements to groundwater quality beneath potato production systems.

Lastly, quantifying water quality benefits is critical to determining which practices are actually capable of meaningful improvements to water quality. However, water quality monitoring experiments can be time consuming and costly to conduct. To ensure research dollars are spent efficiently, it is critical to first work out the agronomic challenges associated with implementation (i.e. planting date, seeding rates, species selection, etc.) prior to devoting time and resources to measuring water quality where best practices for implementation are still being worked out.

3.6 Acknowledgements

This work was supported by the University of Wisconsin – System (UWS) through the Wisconsin Groundwater Research and Monitoring Program coordinated by the University of Wisconsin Water Resources Institute (University of Wisconsin – System #WR20R001). We would also like to thank Isherwood Family Farm for their assistance with the project and continued support of scientific research. Field and technical assistance provided by Nicholas Koschak, Elizabeth Belmont, and Lily Lefebvre.

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3.8 Tables

Table 1. Timeline of relevant dates and weather summary statistics.

	2020	**2021**
Field	**G**	**H**
Baseline Water Sampling	April 25	April 17
Planting	May 2	April 28
First Hilling	May 21	May 17
Emergence	June 1	May 26
Second Hilling	June 14	June 7
Interplanting	June 17	June 11
Vine Kill	August 20	August 12
Biomass Sampling	September 7	September 1
Harvest	September 11	September 12
Post Water Sampling	April 23, 2021	May 4, 2022
Planting to Harvest (days)	132	138
Precipitation (mm)	443	788
Minimum Temperature (C)	-3.1	-3.4
Mean Minimum Temperature (C)	12.2	11.9
Mean Temperature (C)	18.1	18.0
Mean Maximum Temperature (C)	24.0	24.2
Maximum Temperature (C)	33.0	33.2

Table 2. Summary of large plot data comparing mean biomass measurements and nitrogen uptake between a control plot versus treatment that had mixture of barley, oats, and millet interplanted between potato hills.

Plot	Tuber	Interplant	Vines	Tuber	Potato Residue	Interplant	Total Uptake
	Yield (cwt)	---Dry matter (lb ac^{-1})----		-------------Nitrogen (lb ac^{-1})---------------			
				2020			
GC	314(35)a	0(0)b	1,146(143)	135(12)	29(12)	0(0)b	164(9)
GT	248(60)b	1,632(920)a	1,141(447)	116(24)	26(12)	49(30)a	192(42)
				2021			
HC	332(18)a	0(0)b	1,695(336)	158(8)a	48(11)	0(0)b	206(16)
HT	291(31)b	561(163)a	1,442(353)	138(15)b	41(16)	16(5)a	196(28)

Standard deviation in parentheses
Values within each column followed by different letters are significantly different (0.10) based on two-sample, two-sided comparison with unequal variance. Comparisons between years not represented.

Table 3. Summary of mean biomass measurements and nitrogen uptake of control (C), interplanted barley (B), oats (O), millet (M), and barley/oat/millet mixture (BOM) plots.

Plot	Tuber	Interplant	Potato Residue	Tuber	Potato Residue	Interplant	Total Uptake
	Yield (cwt)	---Dry matter (lb ac^{-1})---		---------Nitrogen (lb ac^{-1})-----------			
C	313(37)	0(0)b	1,500(37)	115(37)	42(18)	0b	157(36)
B	316(19)	1,089(310)a	1,351(72)	178(29)	36(13)	31(7)a	245(36)
M	306(28)	99(137)b	1,410(249)	153(47)	35(2)	4(6)b	191(49)
O	308(35)	737(264)a	1,199(261)	134(25)	34(16)	18(9)a	186(30)
MOB	298(30)	922(581)a	1,285(132)	152(49)	39(10)	24(20)a	215(65)
ANOVA p-value results							
PLOT	0.818	<0.001*	0.058	0.296	0.808	0.001*	0.147
ROW	0.029	0.337	0.193	0.589	0.378	0.342	0.667
COLUMN	0.888	0.243	0.052	0.990	0.434	0.273	0.920
LSD†	35	411	202	61	17	14	70

Standard deviation in parentheses
*Values within each column followed by a different letter are significantly different based on an observed ANOVA treatment effect (0.05) and post hoc Fisher's Least Significant Difference (LSD) test (0.05)
†Fisher's Least Significant Difference (LSD) were determined for all measurements (0.05)

Table 4. Summary of soil sensor measurements from 2020 in GC and GT plots for soil temperature (a), water content (b), and pore water electrical conductivity (c) at varying depths beneath the hill (H) and furrow (F).

a)

Sensor Depth	Control (GC)						Treatment (GT)					
	Mean		Min		Max		Mean		Min		Max	
	Temperature (degrees C)											
cm	H	F	H	F	H	F	H	F	H	F	H	F
10	21.6	-	13.8	-	28.5	-	21.5	-	13.4	-	29.0	-
-10	20.9	20.7	16.6	15.9	25.4	26.7	21.0	20.9	16.2	15.9	25.7	27.1
-20	20.5	20.5	17.2	17.0	23.8	24.2	20.7	20.5	17.0	16.9	24.3	24.3
-40	20.0	19.8	17.5	17.6	22.5	22.2	20.3	20.1	17.5	17.5	23.0	22.8

b)

Sensor Depth	Control (GC)						Treatment (GT)					
	Mean		Min		Max		Mean		Min		Max	
	cm cm^{-1}											
cm	H	F	H	F	H	F	H	F	H	F	H	F
10	0.10	-	0.07	-	0.26	-	0.11	-	0.07	-	0.33	-
-10	0.15	0.19	0.13	0.13	0.22	0.34	0.16	0.21	0.13	0.14	0.29	0.33
-20	0.13	0.15	0.11	0.12	0.14	0.26	0.13	0.16	0.12	0.13	0.19	0.23
-40	0.11	0.13	0.10	0.11	0.13	0.18	0.10	0.13	0.09	0.11	0.12	0.16

c)

Sensor Depth	Control (GC)						Treatment (GT)					
	Median		Min		Max		Median		Min		Max	
	Pore Water Electrical Conductivity (mS/cm)											
cm	H	F	H	F	H	F	H	F	H	F	H	F
10	3.2	-	0.5	-	1873	-	2.0	-	0.4	-	1259	-
-10	3.4	1.0	1.5	0.4	4.1	13.7	2.6	0.8	1.0	0.4	3.7	9.1
-20	2.2	0.9	1.5	0.4	4.4	5.6	1.7	0.8	0.9	0.4	2.2	6.0
-40	1.4	1.1	0.9	0.7	3.9	1.7	3.8	1.1	1.2	0.8	6.2	6.0

Table 5. Summary of soil sensor measurements from 2021 in HC and HT plots for soil temperature (a), water content (b), and pore water electrical conductivity (c) at varying depths beneath the hill (H) and furrow (F).

a)

Sensor Depth	Control (HC)						Treatment (HT)					
	Mean		Min		Max		Mean		Min		Max	
	Temperature (degrees C)											
cm	H	F	H	F	H	F	H	F	H	F	H	F
10	22.1	-	14.0	-	33.2		22.0	-	13.6	-	33.0	-
-10	21.5	21.5	16.8	15.9	27.8	29.6	21.4	21.3	16.7	16.2	27.0	28.7
-20	21.0	21.0	17.8	17.5	25.7	26.0	20.9	21.0	18.0	17.5	25.0	25.4
-40	20.7	20.7	18.2	18.2	24.3	24.3	20.5	20.4	18.1	18.1	23.9	23.7

b)

Sensor Depth	Control (HC)						Treatment (HT)					
	Mean		Min		Max		Mean		Min		Max	
	cm cm^{-1}											
cm	H	F	H	F	H	F	H	F	H	F	H	F
10	0.17	-	0.09	-	0.31	-	0.14	-	0.09	-	0.29	-
-10	0.21	0.20	0.18	0.17	0.32	0.33	0.16	0.21	0.11	0.12	0.27	0.33
-20	0.14	0.16	0.12	0.13	0.24	0.28	0.12	0.12	0.10	0.10	0.22	0.26
-40	0.18	0.17	0.14	0.14	0.26	0.24	0.17	0.17	0.14	0.13	0.25	0.28

c)

Sensor Depth	Control (HC)						Treatment (HT)					
	Median		Min		Max		Median		Min		Max	
	Pore Water Electrical Conductivity (mS/cm)											
cm	H	F	H	F	H	F	H	F	H	F	H	F
10	0.6	-	0.4	-	16.3	-	0.8	-	0.3	-	526	-
-10	1.1	0.5	0.6	0.3	3.8	2.3	1.1	0.6	0.8	0.4	3.2	9.2
-20	1.1	0.6	0.5	0.4	1.9	1.9	3.7	0.5	0.5	0.3	1130	4.9
-40	0.7	0.5	0.4	0.4	1.8	2.3	1.2	0.6	0.5	0.4	5.4	5.2

Table 6. Mean concentration of nitrate-nitrogen and chloride measured in samples collected from top 100 cm of the water table.

Field	Prior Year Crop	Plot	Sample Date	n	Nitrate-N		Chloride	
					mg L^{-1}	p-val	mg L^{-1}	p-val
G	Peas/Millet	Treatment	25 Apr 2020	11	7.7(2.4)	0.182	6.0(2.4)	0.293
		Control	26 Apr 2020	11	9.8(7.3)		7.3(3.2)	
	Potato	Control	23 Apr 2021	11	56.0(9.5)	-	36.6(15.9)	-
H	Peas/Millet	Treatment	17 Apr 2021	11	28.3(7.4)	0.353	26.2(10.0)	0.967
		Control	18 Apr 2021	11	25.7(4.8)		26.4(8.3)	
	Potato	Treatment	9 May 2022	11	49.0(7.8)	0.029*	50.4(10.7)	0.003*
		Control	4 May 2022	11	36.2(14.7)		34.2(10.9)	

Standard deviation in parentheses
*Values significantly different based on difference in means between treatment and control using Welch’s two-sample t-test (0.05)

3.9 Figures

Figure 1. Aerial view of fields used in 2020 (Field G) and 2021 (Field H) for interplanting experiment. Location of 2 ha plots for control (GC/HC) and treatment (GT/HT) are displayed. In 2021, small plots were located adjacent to HT for studying different interplanting species and impacts on yield and nitrogen uptake.

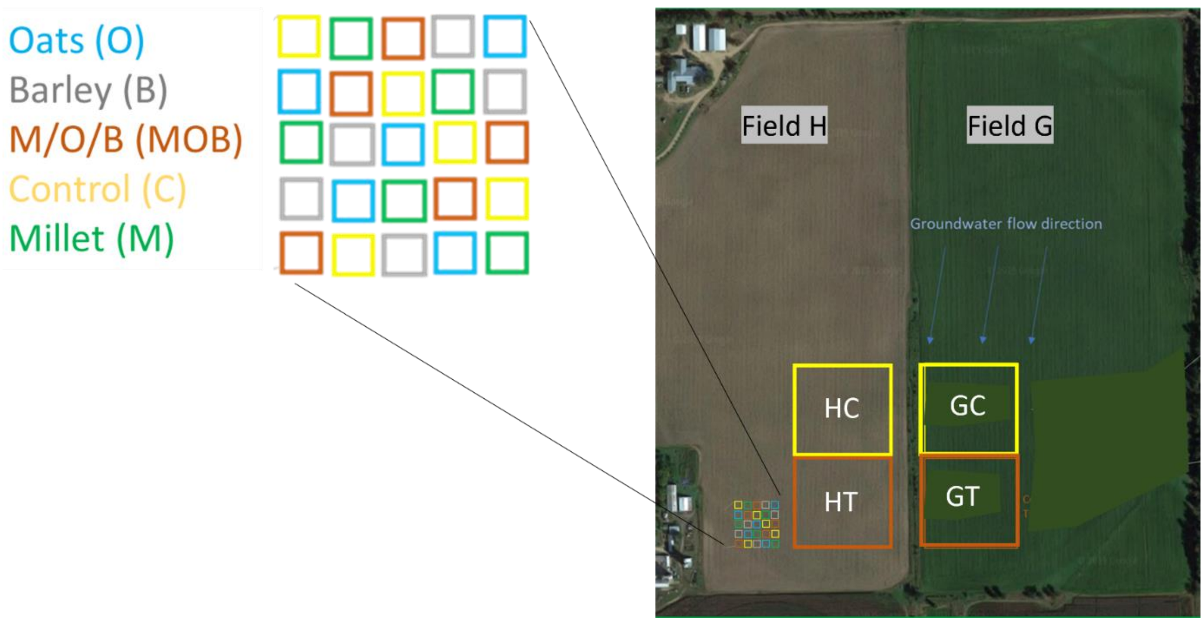


Figure 2. Schematic of sensor installation used to investigate water and solute movement below hill and furrow potato configuration. Sensors were installed at 10, -10, -20, and -40 cm relative to the bottom of the furrow (0 cm).

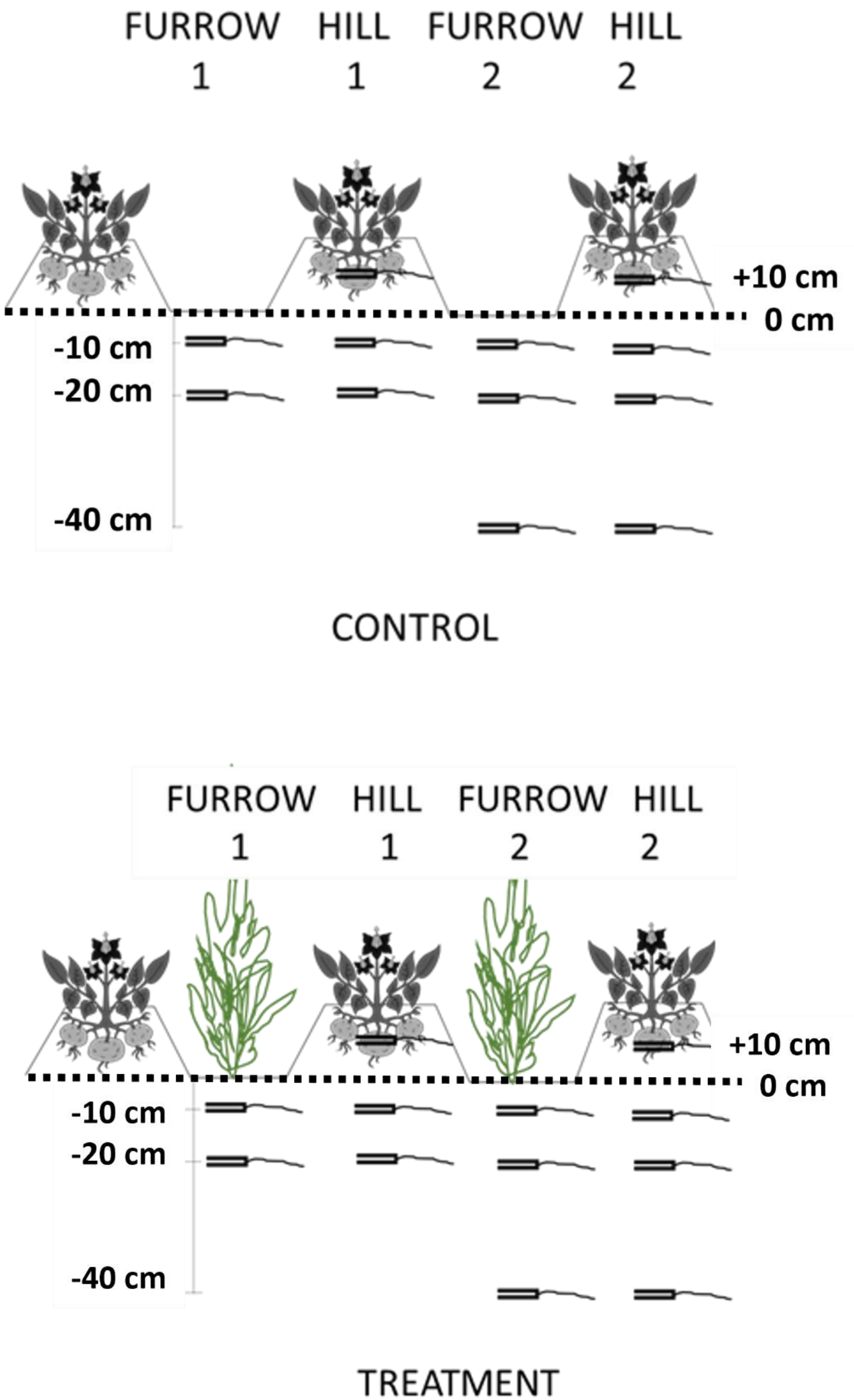


Figure 3. Depiction of sampling design for large plots. Temporary wells were used to sample the upper ~100 cm of the water table. Colors represent differences in concentration for solutes such as nitrate or chloride that might result from different recharge events throughout the year. Sample locations located away from plot edge of primary groundwater flow direction to avoid intrusion from outside of plot boundary.

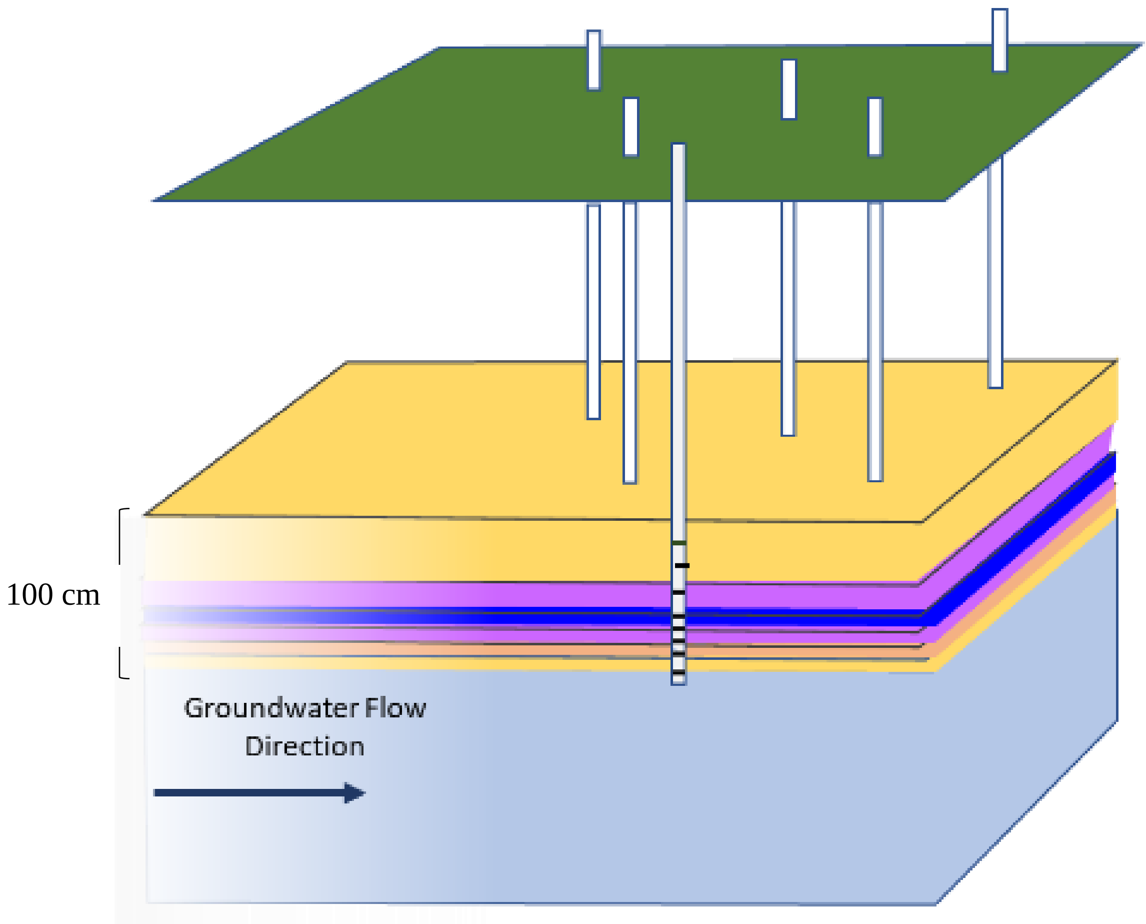


Figure 4. Weekly measurements of vegetation height during the 2021 growing season. Vegetation height from A) no competition (NOCOM) plots where potato vines were removed and B) plots where potato vines remained in competition with interplanting (COM) plots to investigate the effect of canopy cover on interplant growth rates.

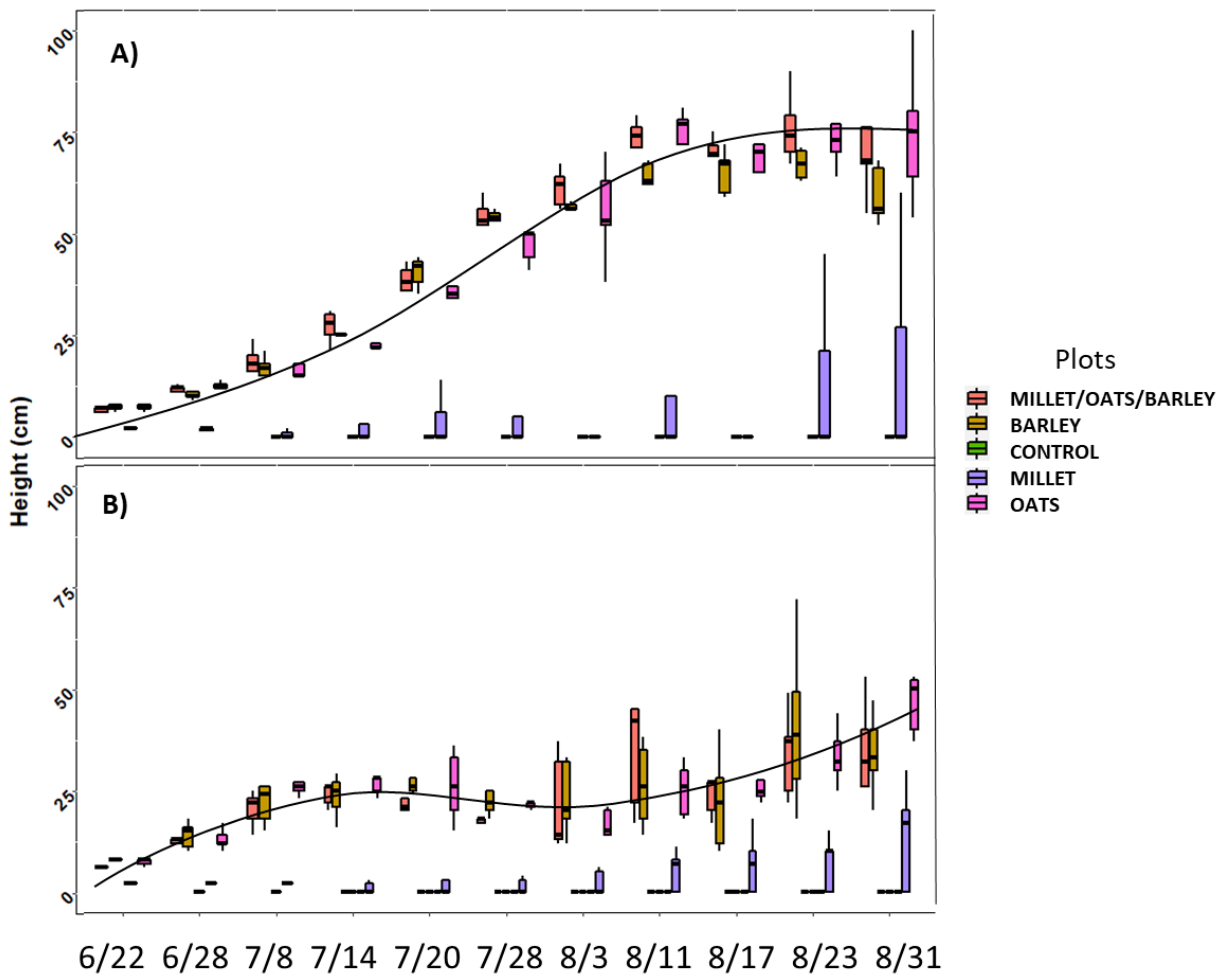


Figure 5. Weekly measurements of canopy cover (%) during the 2021 growing season. Canopy cover from A) no competition (NOCOM) plots where potato vines were removed and B) plots where potato vines remained in competition with interplanting (COM) plots to to investigate the effect of interplanting on canopy cover within furrows.

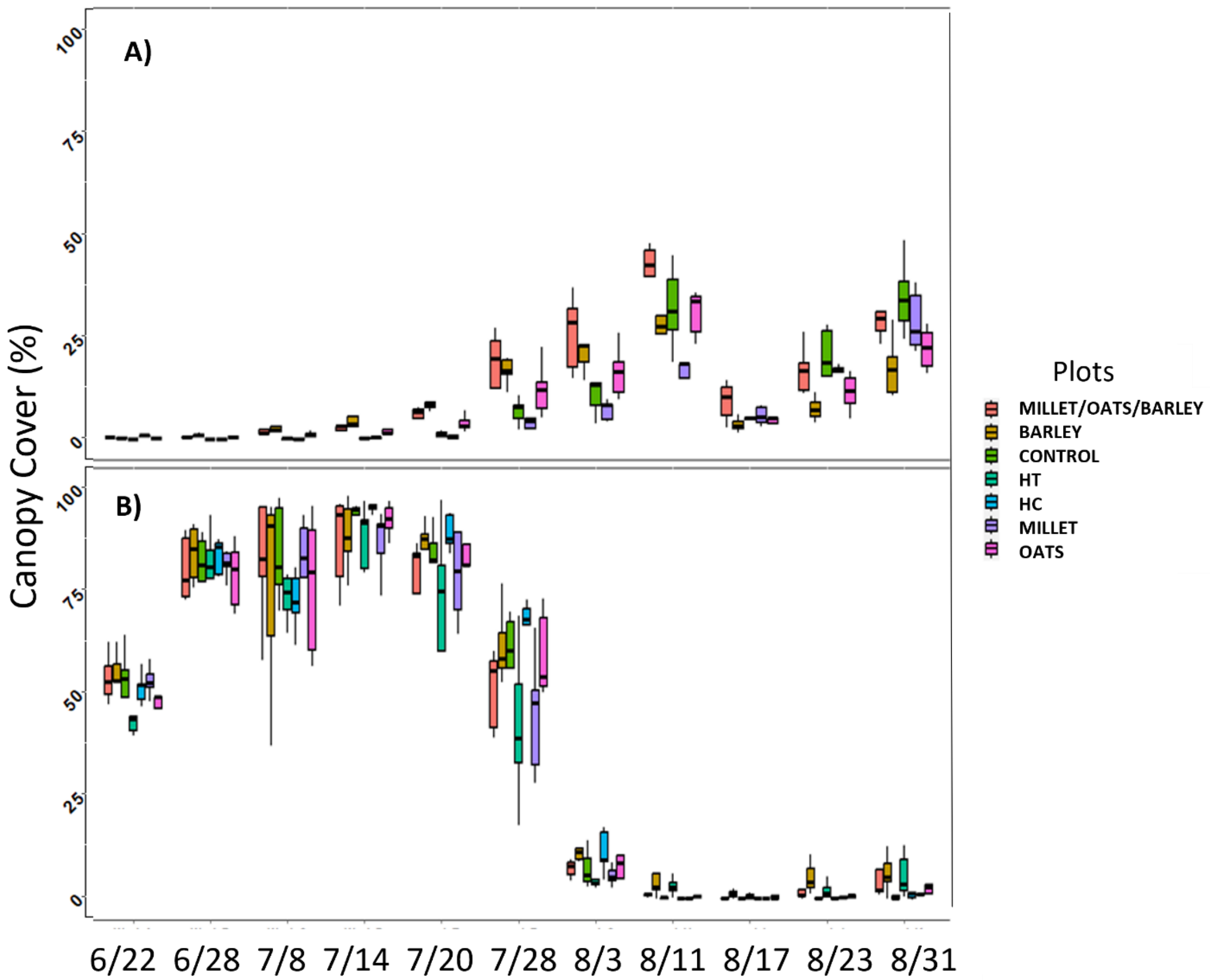


Figure 6. Temperature measurements from 2020 below the hill (+10, -10, -20, -40 cm) and furrow (-10, -20, -40 cm) in control (blue) and interplant treatment (red) plots.

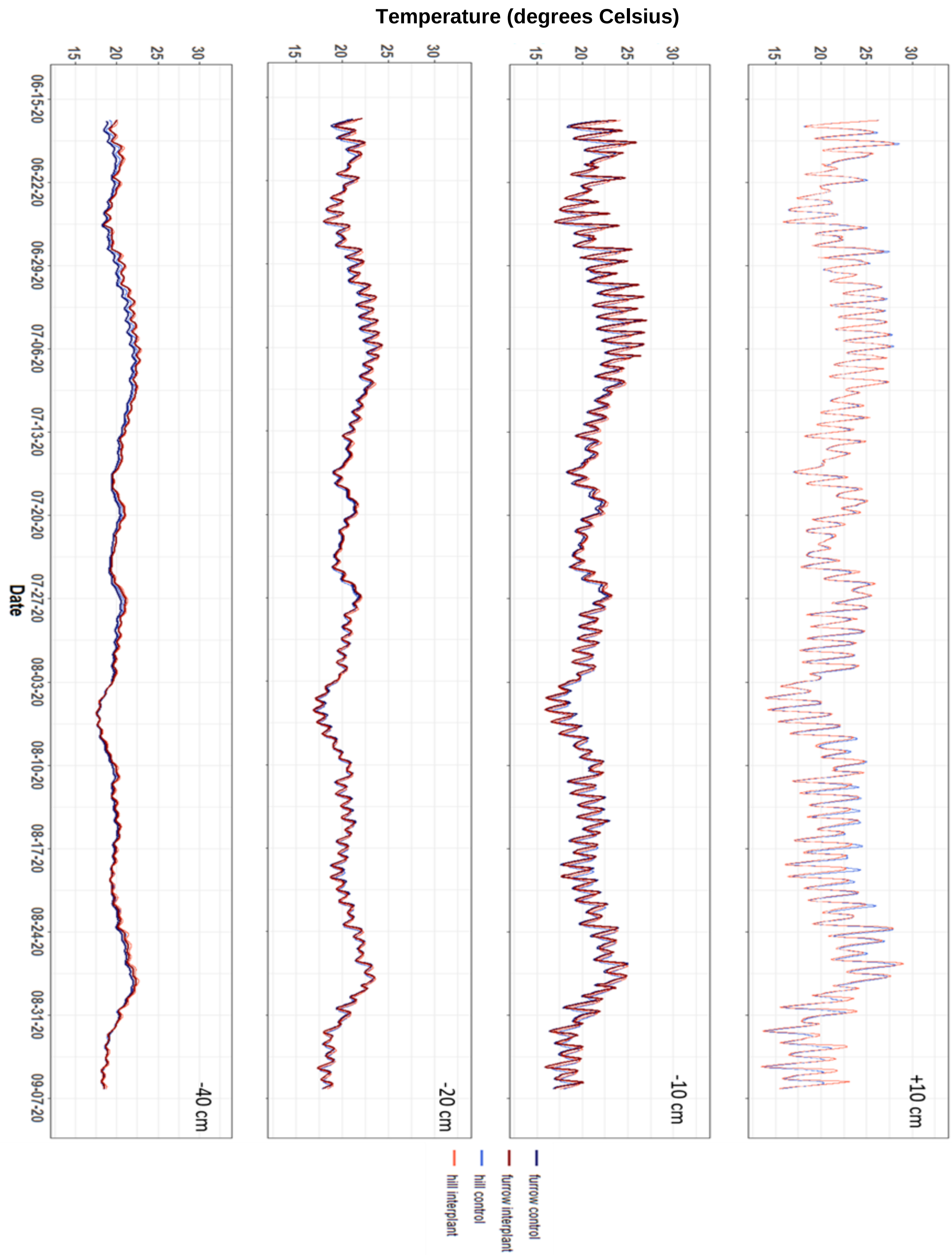


Figure 7. Temperature measurements from 2021 below the hill (+10, -10, -20, -40 cm) and furrow (-10, -20, -40 cm) in control (blue) and interplant treatment (red) plots.

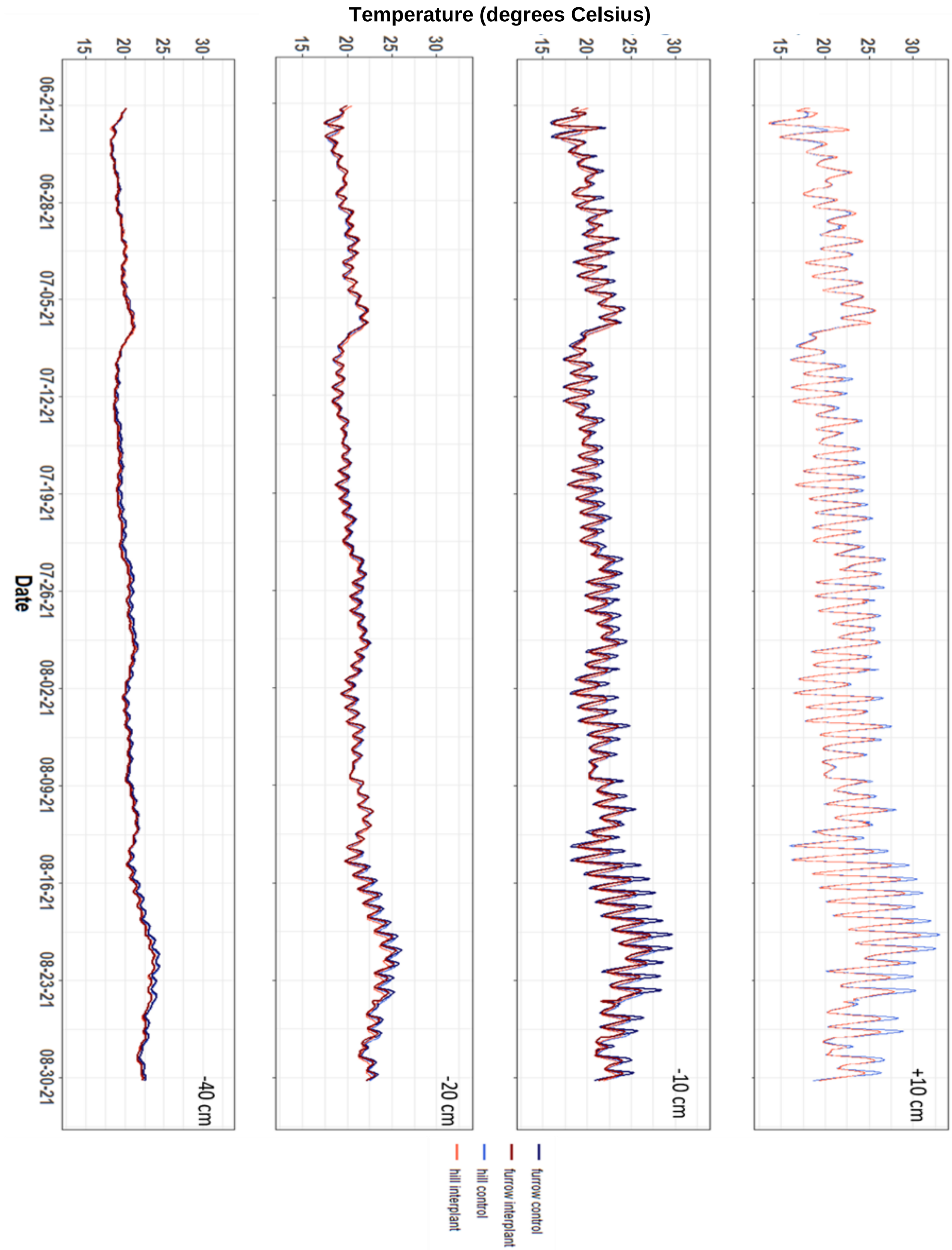


Figure 8. Two-dimensional representation of sensor measurements in HC (control) and HT (treatment) following a precipitation event on 28 July 2021. Rows 6 through 10 represent the 20-40 cm sensor depths.

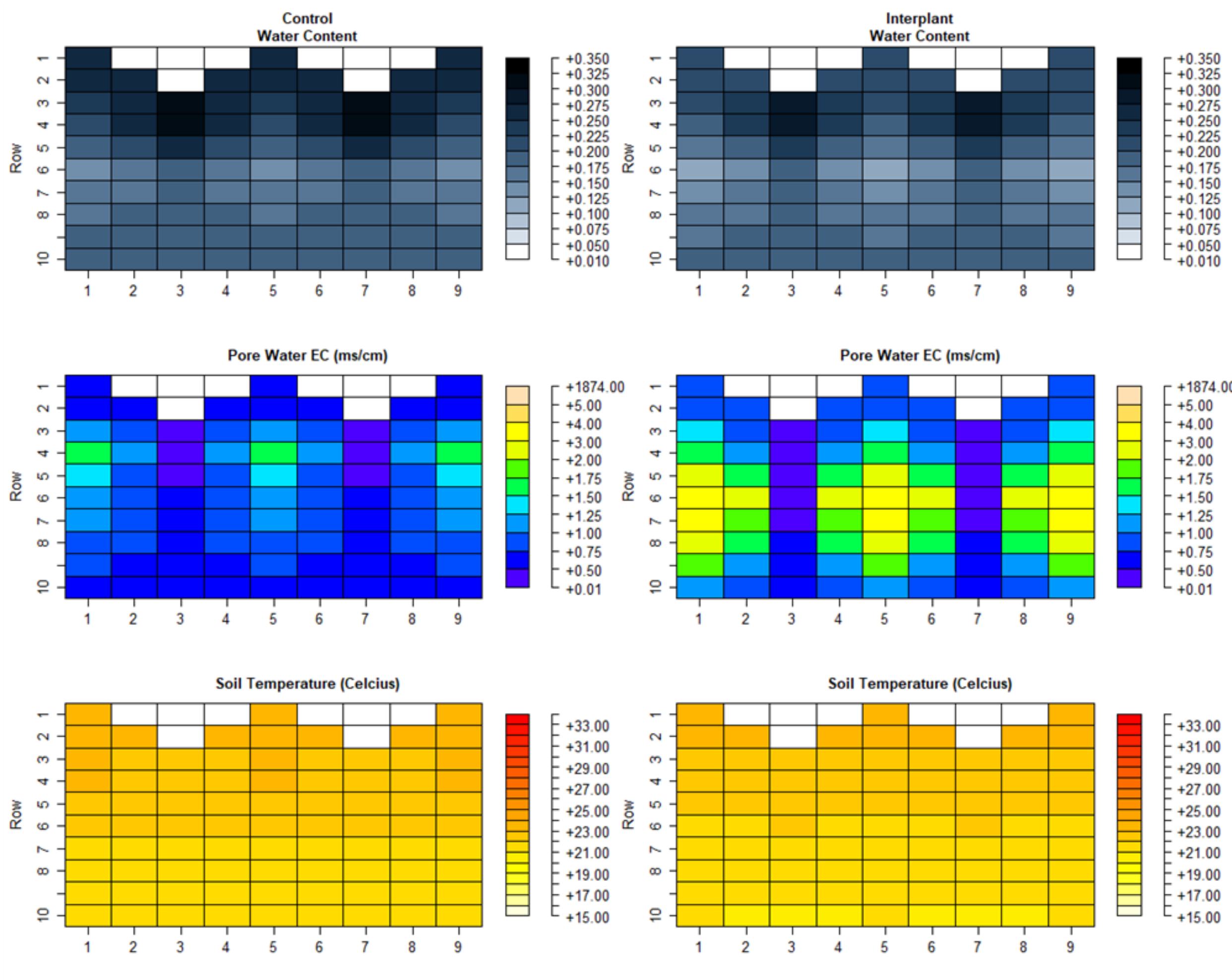
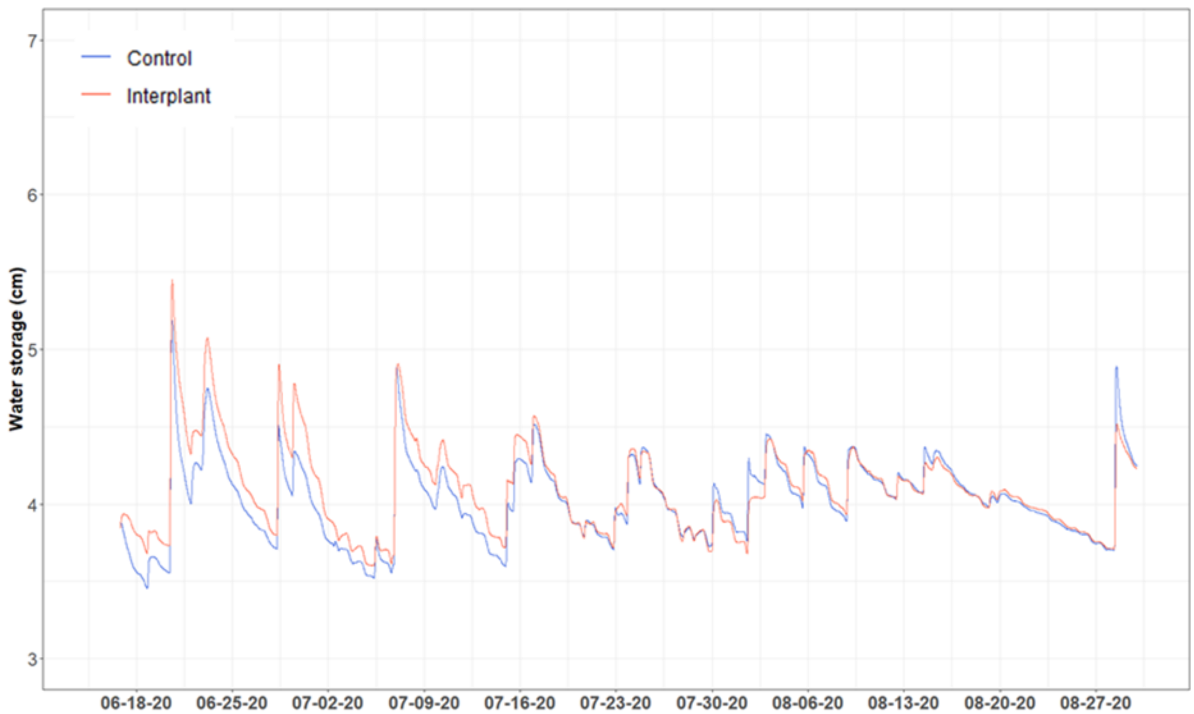


Figure 9. Water storage estimated for a 30-cm depths using soil water content measurements from depths below the root zone. Changes in water storage used to estimate drainage.

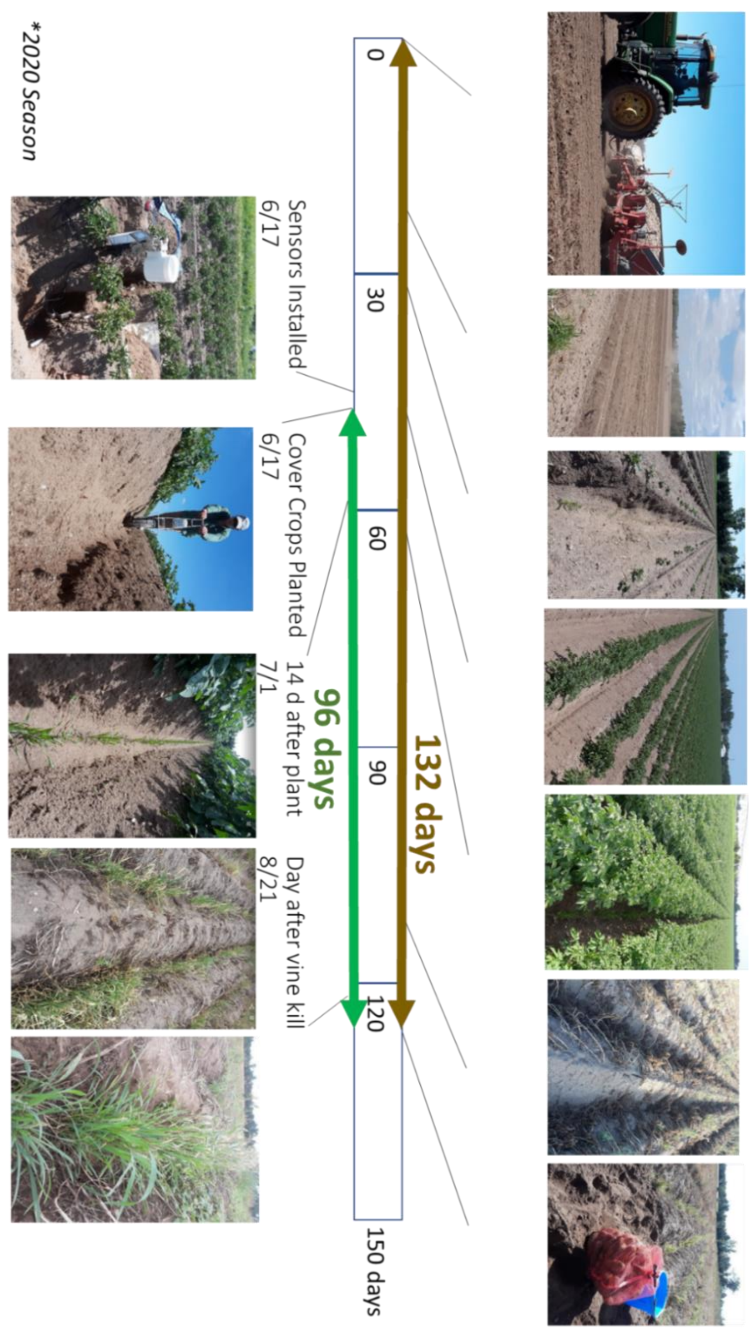


3.10 Pictures

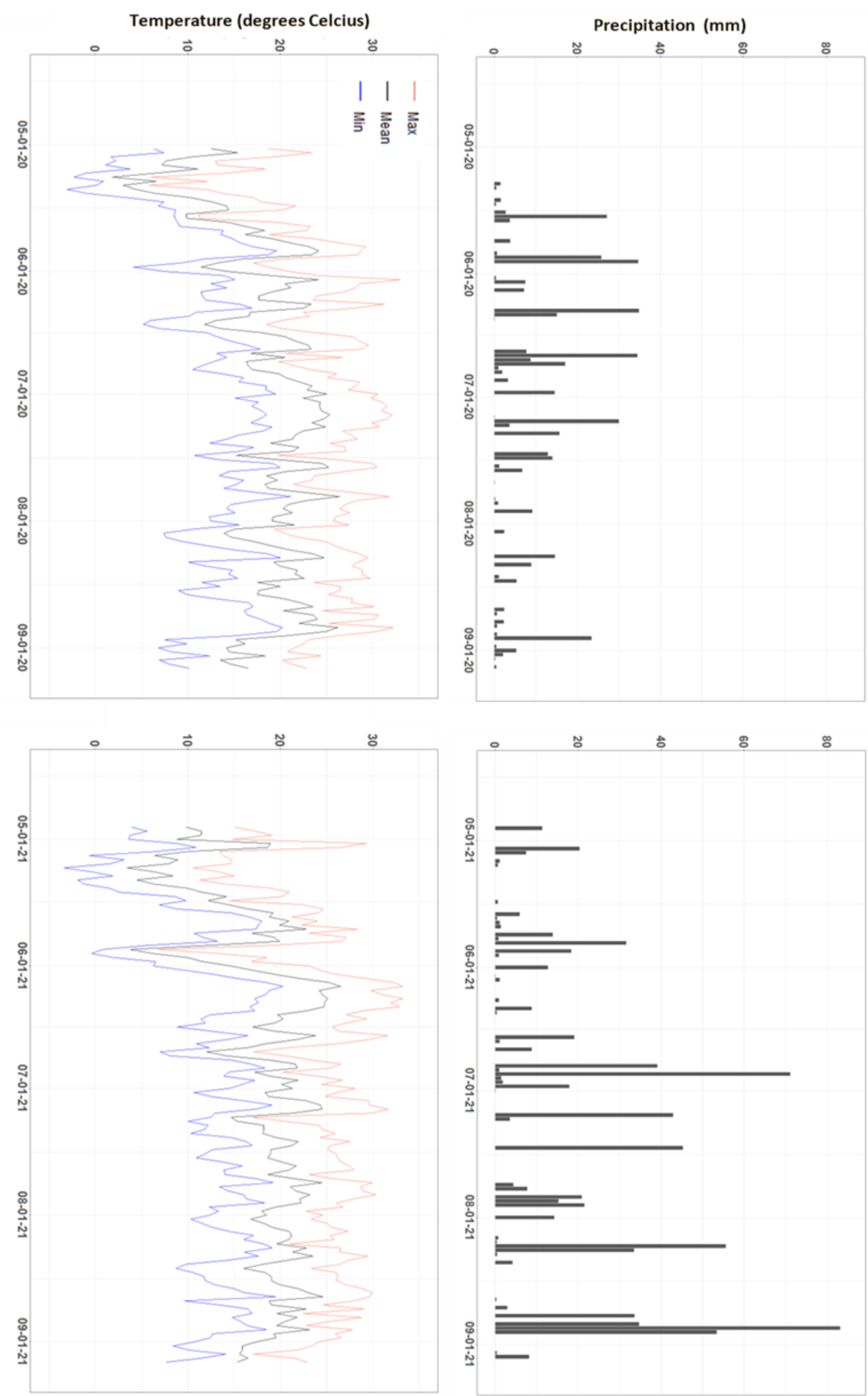
Picture 1. Interplanting plots just prior to harvest in 2020 (top) and 2021 (bottom) comparing coverage between the two years.

3.11 Supplemental Materials

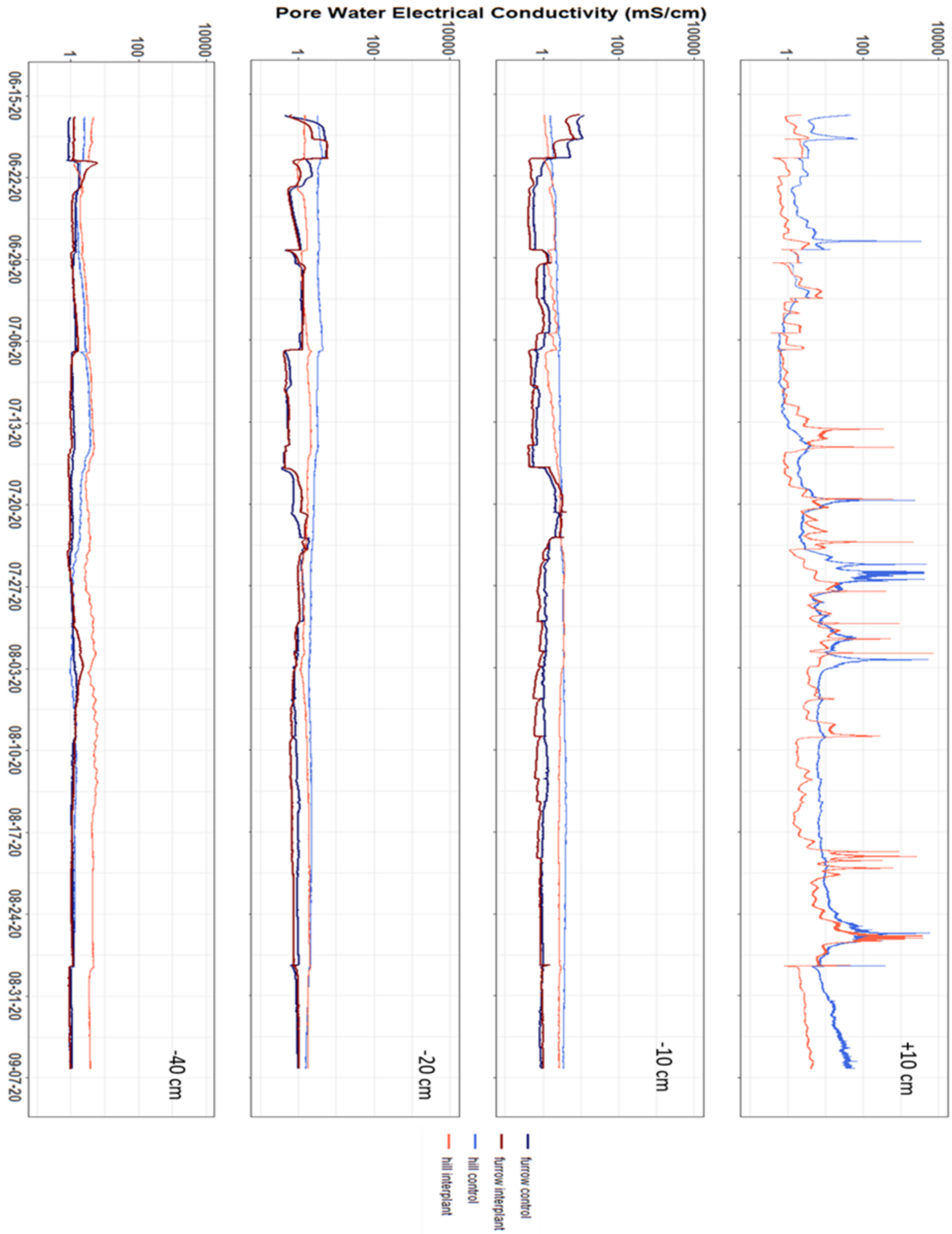
Supplemental Figure 1. Photo timeline of field operations during the 2020 growing season. Normal field operational activities are shown on top culminating with harvest 132 days after planting. Activities having to do with interplanting are shown on the bottom.



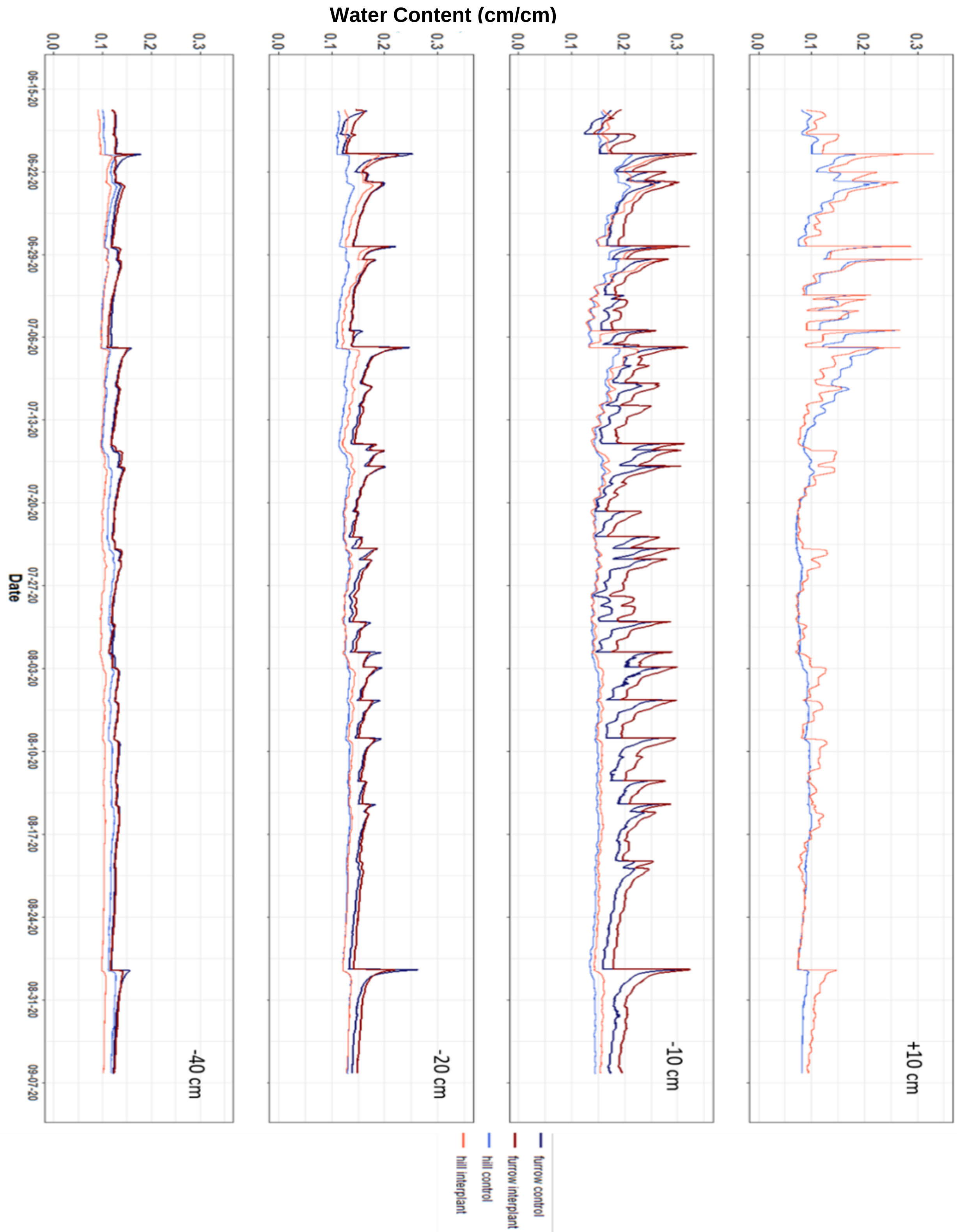
Supplemental Figure 2. Daily precipitation (top) and daily maximum, mean, minimum air temperatures during the period between planting and harvest for 2020 and 2021 for the study site.



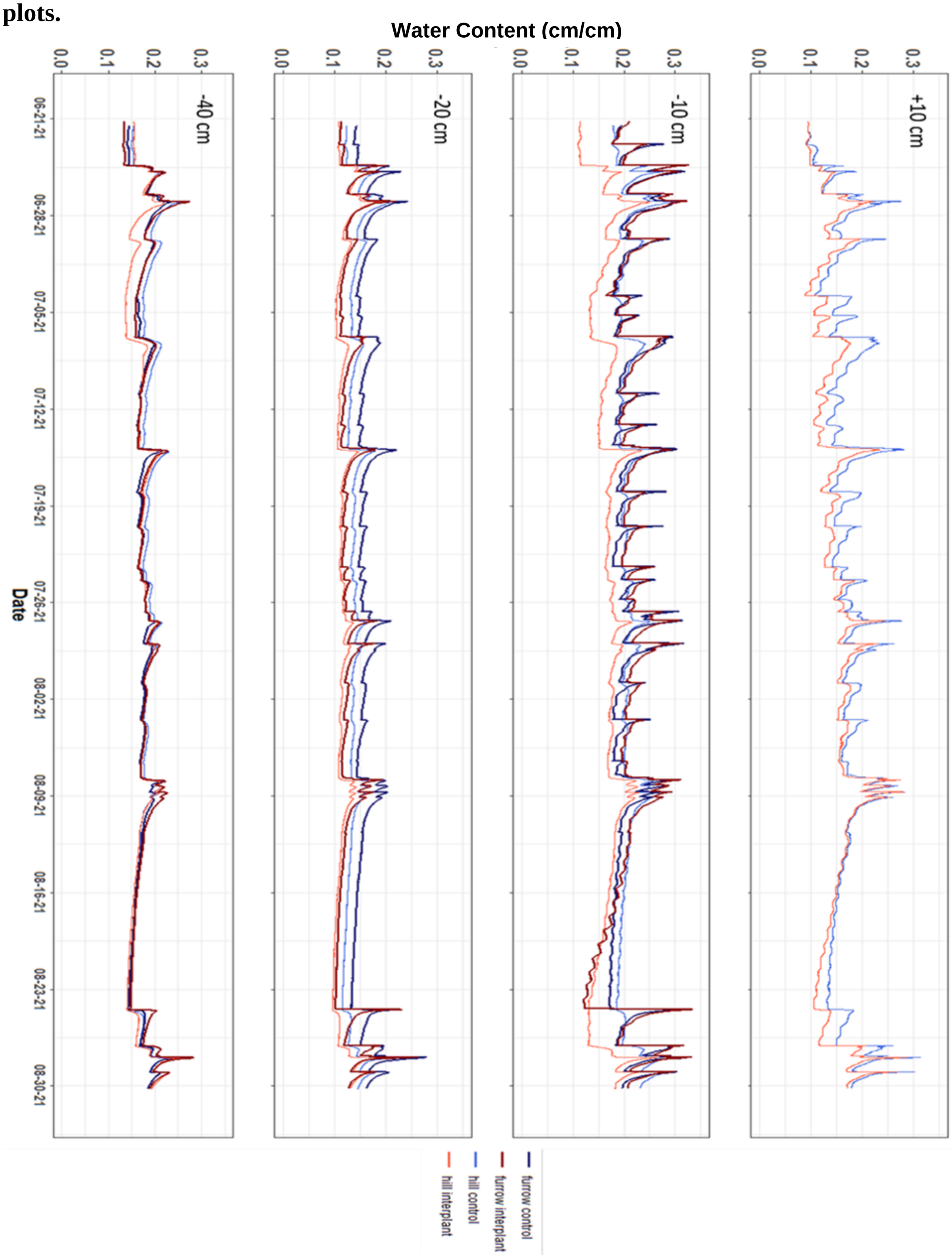
Supplemental Figure 3. Pore water electrical conductivity measurements from 2020 below the hill (+10, -10, -20, -40 cm) and furrow (-10, -20, -40 cm) in control (blue) and interplant treatment (red) plots.



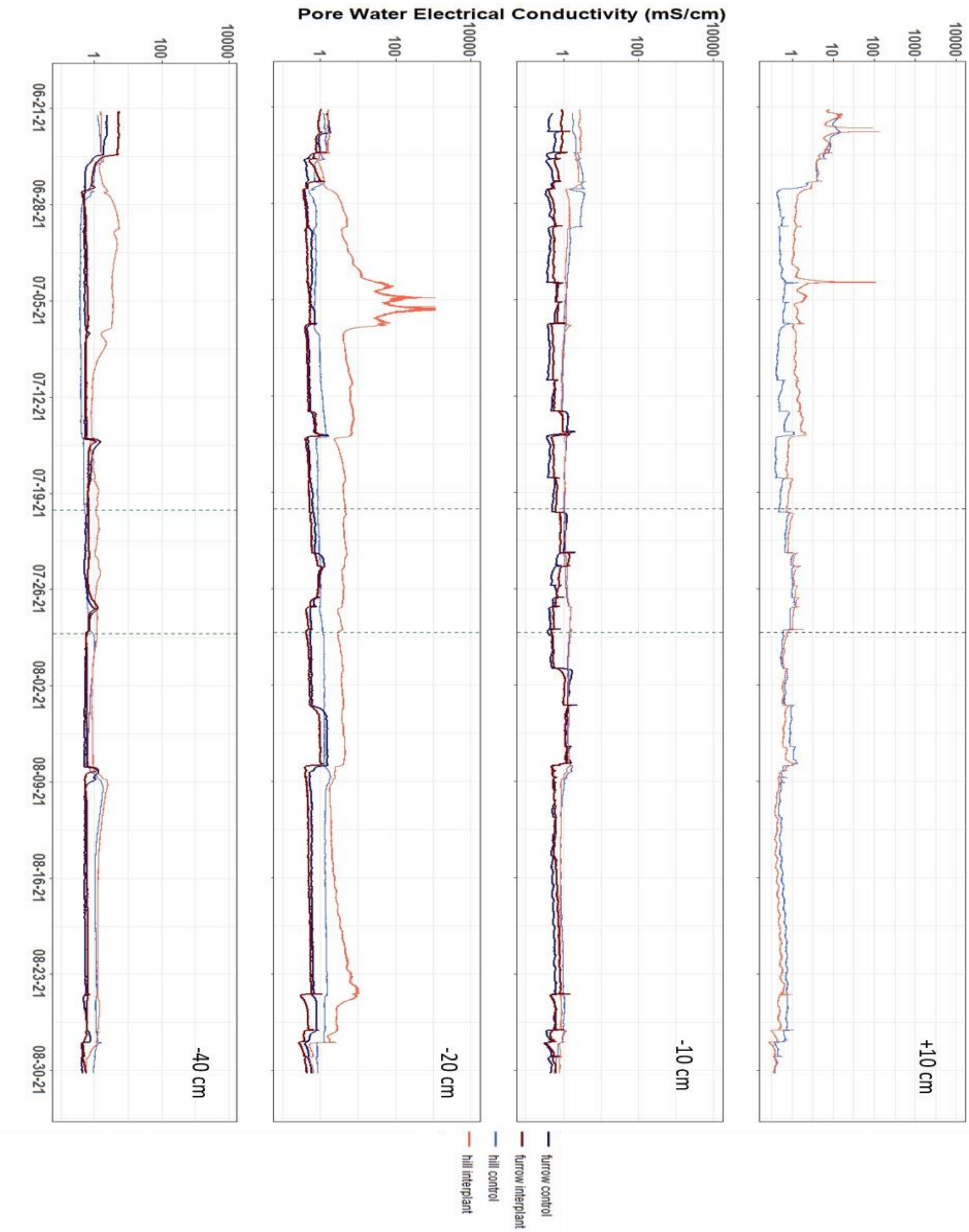
Supplemental Figure 4. Water content measurements from 2020 below the hill (+10, -10, -20, -40 cm) and furrow (-10, -20, -40 cm) in control (blue) and interplant treatment (red) plots.



Supplemental Figure 5. Water content measurements from 2021 below the hill (+10, -10, -20, -40 cm) and furrow (-10, -20, -40 cm) in control (blue) and interplant treatment (red) plots.



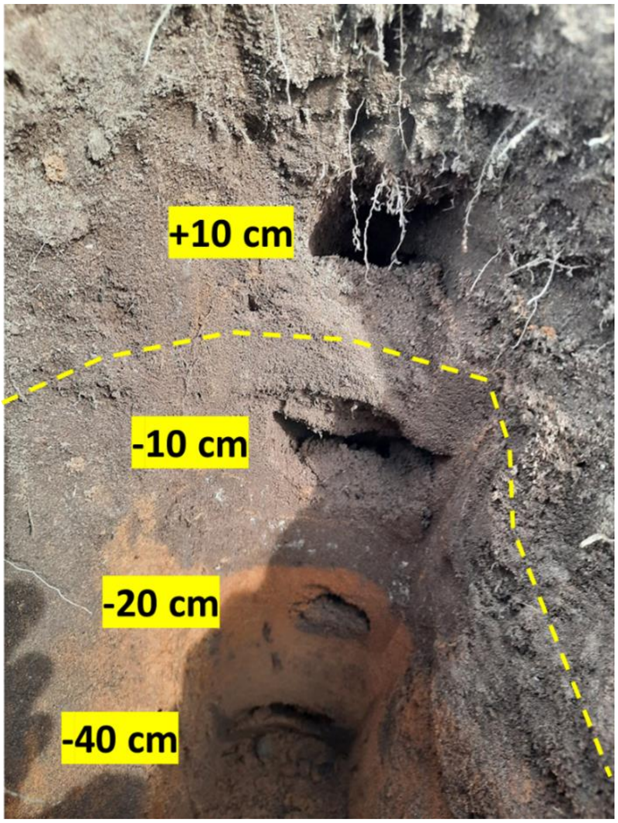
Supplemental Figure 6. Pore water electrical conductivity measurements from 2021 below the hill (+10, -10, -20, -40 cm) and furrow (-10, -20, -40 cm) in control (blue) and interplant treatment (red) plots.



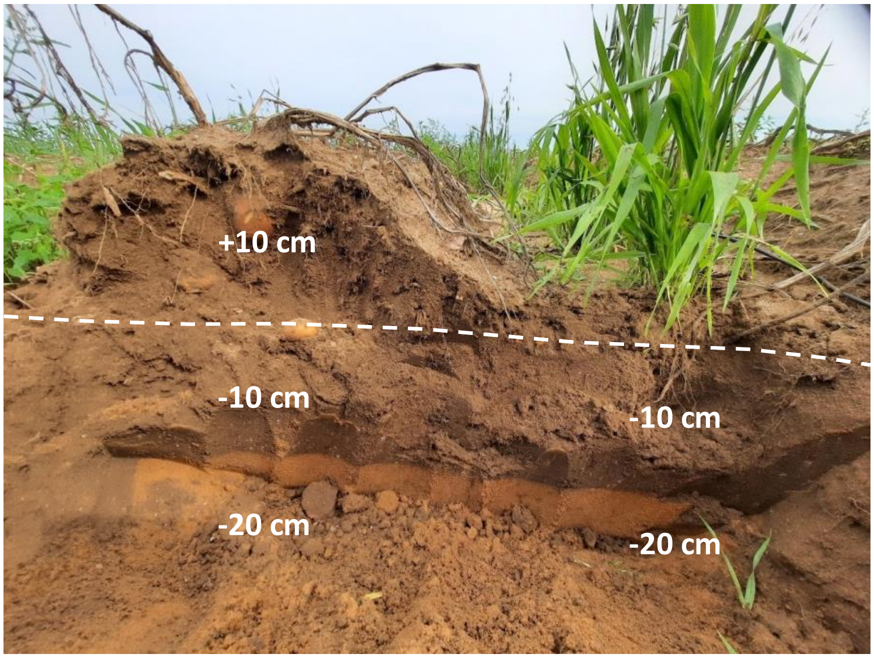
Supplemental Picture 1. View of Teros 12 sensor installation. Visible are the hill (+10 cm), -10 cm, and -20 cm sensors beneath the hill.

Supplemental Picture 2. View of the Meter ZL6 datalogger which allowed for viewing and accessing of data remotely. Afidus time-elapse camera is mounted to the top.

Supplemental Picture 3. Soil profile following removal of sensors. Dashed line represents the location of the furrow bottom.



Supplemental Picture 4. View of hill and furrow soil sensor measurement locations following removal. Dashed line represents bottom of furrow.



Supplemental Picture 5. Photos of interplanting progression during the 2021 season. Furrow just after placing of seed for plot establishment (left), emergence of seed with evidence of erosion after heavy rain event (center), inteplanting two weeks after planting (right).

Supplemental Picture 6. Close up view of a furrow in HC control plot (left) and HT treatment plot (right) showing the interplanting.

Supplemental Picture 7. View of control plot (left) and interplanting treatment plot (right) just prior to harvest in 2021.

Chapter 4

Development of a community well monitoring network to identify factors influencing nitrate-nitrogen concentrations and trends

Abstract

State and local officials, resource professionals, and rural residents are increasingly interested in knowing whether groundwater quality is getting better or worse. Few if any communities in Wisconsin have appropriate systems to track spatiotemporal groundwater quality. A multi-year citizen-based well monitoring network was initiated to provide an annual assessment of well water quality. Testing of the same wells each year provided the ability to assess nitrate trends at an individual well or regionally. Groundwater nitrate-nitrogen concentrations were shown to have strong relationships to areas of agricultural activity. A multiple linear regression model highlighted areas where nitrate is likely to be low (<3 mg L^{-1}), moderate (3-6 mg L^{-1}) or high (> 6.5 mg L^{-1}). Trends in nitrate-nitrogen concentrations were not detected in 86% of wells tested over a 4-year period. Fourteen percent of wells indicated significant trends with nearly equal numbers showing increasing/decreasing nitrate-nitrogen concentrations. Regional trend analysis suggested that nitrate-nitrogen concentrations were increasing in Green County and decreasing in Sauk County.

4.1 Introduction

In Wisconsin, nearly one-third of residents rely on one of the estimated 800,000 rural residential wells (i.e. private wells) as their primary source of water. While public water systems are regulated and required to meet drinking water standards, deciding what to test for, how often to test, and what to do if problems are detected are largely the responsibility of the individual well owner. Testing is the only way to determine if a private well exceeds safe drinking water standards for one or more parameters; including nitrate, which is the most widespread groundwater contaminant in Wisconsin.

A recent statewide survey of agricultural chemicals in Wisconsin groundwater showed that an estimated 8.2% of wells had nitrate concentrations above the drinking water standard of 10 mg/L nitrate-nitrogen (DATCP, 2017b). Sampling of groundwater-dominated rivers and streams in Wisconsin revealed significant evidence of nitrogen saturation; agriculturally rich areas commonly contained nitrate-nitrogen concentrations with implications for stream biota and negative consequences for coastal eutrophication (Stanley and Maxted, 2008).

State and local officials, resource professionals, and rural residents are increasingly interested in knowing whether groundwater quality is getting better or worse, particularly with respect to proposed or recently enacted land-use changes as they relate to nitrate-nitrogen. Well monitoring networks have been used to monitor groundwater levels and quality at national and regional scales (Roy *et al.*, 2007; Stuart *et al.*, 2007; Hodgkins, 2017); however information at these scales is generally not adequate to inform questions specific to watersheds or local units of government.

Most communities do not have appropriate systems to track spatiotemporal changes in groundwater quality necessary to understand localized or regional trends in groundwater quality. In Wisconsin, public water supplies (i.e. municipal wells, bars, restaurants, churches, schools, mobile home parks, etc.) are required to test for nitrate annually and submit results to the Wisconsin Department of Natural Resources. The long testing history of many public supply wells have provided an opportunity to investigate nitrate trends (CWSE, 2021); however, concerns exist about relying solely on public water supplies to understand regional trends in groundwater quality because of potential biases. Objections include: (1) construction of these wells may allow for accessing of groundwater that is deeper and less representative of shallow groundwater accessed by typical rural landowners; (2) public water supply wells are located in more densely developed areas not representative of rural areas; and (3) public water system wells are often replaced as nitrate concentrations approach the drinking water standard limiting the ability track concentrations of wells that are high in nitrate.

Installation of new wells that are extensive enough to capture variability at local scales would have a significant cost associated with drilling and maintenance; it is therefore logical to utilize the expansive network of private wells where residential owners have a vested interest in their water quality. Beyond the obvious monetary savings of using existing private wells as an alternative to drilling new wells, involving well owners in the collection of data has other potential benefits.

Private wells are encouraged to be tested routinely for common parameters; however, a recent survey suggests only half of well owners tested their well in the last 10 years (Malecki *et al.*, 2017). The top three reasons given for not testing included lack of perceived problems, not knowing how to test or what to test for, or not seeing a need to test because they were actively treating their water. Community programs that ensure well water sampling is convenient were mentioned as one of the top reasons for having water tested.

Incorporating private well owners into the data collection efforts not only assists private well owners in the safety evaluation of their own well water, but also provides an important public service when data are aggregated and used to identify local/regional concerns, target outreach efforts, and assist with groundwater management decisions. Volunteer monitoring or citizen-science has been shown to be a valuable method of building scientific knowledge

(Brossard *et al.*, 2005; Bonney, 2009). Some studies also suggest that inclusion of the general public in science can be beneficial for informing policy and encouraging public action (McKinley *et al.*, 2017).

Land-use, soil properties, and other characteristics of the well (i.e. depth, primary aquifer, etc.) have been used to model various groundwater quality parameters, including nitrate (Johnson and Belitz, 2009; Tesoriero *et al.*, 2017; Jurgens *et al.*, 2020). The Mann-Kendall test, which uses the nonparametric correlation coefficient Kendall's tau, can be used to test for water quality trends of individual wells (Mann, 1945; Helsel and Frans, 2006; Jurgens *et al.*, 2020). Helsel and Frans (2006) proposed a Regional Kendall Test that can be used to determine whether there is a consistent regional trend among environmental variables. Jurgens et al. (2020) used spatial weighting to determine areal proportion of trends in a study area or hydrologic province.

For this research, we set out to develop a community well water monitoring network capable of quantifying drivers of spatiotemporal variability. The use of rural residential private wells takes advantage of existing wells to monitor the groundwater quality typically accessed by rural residents who have a vested interest in the information and maintaining the quality of their local groundwater resource. By making it convenient and easy for participants to test, we hope to overcome one of the most common barriers of testing, which is not knowing how to test or what to test for. Testing the same wells each year provides an opportunity to investigate trends of individual wells and regional trends based on shared characteristics The objectives of this research were to determine: 1) what factors are most critical for predicting water quality, 2) which wells contain evidence of trends, and 3) where and why well water quality is changing (i.e. region, aquifer, land-use, soils, geology, and/or well construction).

4.2 Materials and Methods

4.2.1 Study Site

Sauk County (849 mi^2) and Green County (584 mi^2) are located in south-central Wisconsin. The population of Sauk County is 65,763 people (77.5 persons per mi^2), while Green County has a population of 37,093 (63.5 persons per mi^2). It is estimated that 43% and 60% of residents rely on private wells as their primary water supply in Sauk County and Green County, respectively. Both counties have similar climates, but different land-use, geology, and soils (Figures 1-4). We set out to test 250-300 wells in each of the two counties for five years. Wells were selected to be representative of the regional land cover, soils, and geology.

PRISM is a spatial climate dataset generated from monitoring networks and uses statistical modeling techniques to create a gridded dataset at 4-km or 800-m resolutions (PRISM Climate Group). Annual precipitation data for the period 2019-2022 were downloaded for each well location and these data were used to calculate a mean annual precipitation for each county.

4.2.2 Recruitment

During the initial well selection phase, we only considered wells drilled after 1988. Significant revisions to Wisconsin Legislative Code NR 812 in 1988 updated the regulations concerning the construction of wells and private well water systems. Selecting wells drilled post-1988 was important for two reasons, 1) we could dispel any doubts that wells were not constructed up to "modern standards", and 2) well records began to be stored in a digital database and were assigned a Wisconsin Unique Well Numbers (WUWN). The WUWN ensures access to information on the well geology, static water level, casing/well depth, etc. and also makes it easier for water quality results from future testing efforts to be reconciled with previous test results. We used well construction report (WCR) records to match candidate wells with specific property owners by utilizing the statewide parcel data layer (Wisconsin Land Information Program (WLIP), 2018). Additional information on well depth, static water level, casing depth, uppermost bedrock unit, and aquifer unit the well finished in were obtained for the majority of wells.

Recruitment of participating well owners involved sending a letter explaining the project along with a pre-paid postcard that allowed landowners to opt in/out. We planned for an opt-in rate of 35% but ended up with 50.6% and 45.2% in Sauk County and Green County respectively. Attrition of participants was expected. To avoid identifying and recruiting replacement wells each year when people drop out of the program, we opted to sample everyone that responded positively in the first year of the project. We lost an average of 18 participants per year in Green County, while Sauk County lost an average of 26 participants per year. Attrition was often the result of participants selling the property. Of the initial set of wells tested in Year 1, 84.5% of participants in Green County and 80.5% of participants in Sauk County participated in all four years of the well water monitoring program.

4.2.3 Well Water Sampling

Participants were mailed a 125-mL sample bottle and a pre-paid mailer. Participants were asked to select an untreated faucet and run the water for 5 minutes prior to collecting a sample. If the participant was unsure or indecisive of which faucet to use, we suggested the cold-water kitchen faucet as the default sampling location as most cold-water kitchen faucets bypass the water softening process. Following sample collection, participants were instructed to take the pre-paid mailer containing the sample to a postal service counter where it could be weighed to ensure proper postage and sender could declare that there were no hazardous substances contained within the package. A deadline of approximately 14-21 days was given for collecting and returning samples. Following the deadline, any samples that had yet to be returned were sent a postcard indicating that we would still accept samples for an additional period of time. Samples were generally received at the laboratory within 1-3 days of when they were collected and mailed. Upon arrival at the laboratory, samples were stored at 4 degrees Celsius until the time of analysis.

All samples were analyzed at the UW-Stevens Point Water and Environmental Analysis Laboratory, which is state-certified to perform the tests of interest. Samples were tested for nitrate-nitrite-nitrogen, chloride, alkalinity, total hardness, pH, and conductivity. Nitrate and chloride are the primary anions of interest for trend analysis and understanding the influence of land-use with respect to groundwater. Companion parameters (alkalinity, hardness, pH, and conductivity) provide some diagnostic ability to verify if samples were treated and add insight into geologic influences on water quality. Any samples with water quality profiles indicating treatment capable of artificially reducing nitrate-nitrogen (i.e. reverse osmosis) were flagged as treated samples and removed from any subsequent data analysis.

4.2.4 Well characterization

Land cover and soil drainage classification within a 500 m buffer around the well along with other characteristics (i.e. static water level, well depth, casing depth below the water table, primary aquifer, etc.) were determined for each well. We utilized spatial datasets that are publicly available at statewide scales in hopes that modeling results can be applied to other areas of the state. Previous research investigating the relationship of land cover and buffer size to well water quality concluded that 500 m was adequate for assigning land cover to a well (Burow *et al.*, 1998; Johnson and Belitz, 2009). Given that many private wells are accessing relatively shallow groundwater, we felt that 500 m was an appropriate size for purposes of our investigation. The R Programming Language (version 4.3.0) in combination with the rgdal, raster, sp, rgeos, and terra packages were used to obtain percentage of land cover from the Wiscland datalayer (Wisconsin Department of Natural Resources, 2016). The Level 3 land cover data allows for greater categorization of agricultural land cover into: continuous corn, cash grain, dairy rotation, and potato/vegetable production.

Land cover categories were sometimes binned when summarizing data or performing certain statistical analysis. This included the percentage of agricultural land cover within a 500 m buffer (sum of cash grain, continuous corn, dairy rotation, and potato/vegetable) broken out into intervals of 25%. Nitrate associated with developed areas (i.e. septic systems and lawn fertilizers) represent a potential source of nitrate to groundwater (Wilcox *et al.*, 2010; Rayne *et al.*, 2019). Urban land cover from the Wiscland layer is an easily accessed layer that was used to investigate these effects. Most private wells are expected to have a low percentage of urban land cover and is highlighted by the significant left skewed data for percent urban land cover (Figure 5). Greater and less than 10% were categories of urban land cover used for summarizing and investigating effects of this variable on nitrate-nitrogen concentrations.

The rate at which water infiltrates into and drains through soils can be an important factor for determining the amount of groundwater recharge and the ease with which contaminants such as nitrate leach into groundwater. Drainage class is a layer within the SSURGO database that classifies soils into seven categories depending on the rate that water moves through the soil (Soil Survey Staff, 2020). Drainage class areas within a 500 m buffer of each well were determined and used to calculate a weighted average soil drainage rank (1, Very Poorly Drained – 7, Excessively Drained).

In Wisconsin, wells installed into unconsolidated aquifers are required to have a minimum depth of 25 ft or screen installed 10 feet below the static water level (SWL) when the SWL is greater than 15 feet below the surface. Meanwhile wells drilled into bedrock are required to have a minimum of 30 ft of casing when drilled into sandstone aquifers; and 40 ft of casing when top of dolomite is greater than 20 ft below the ground surface or 60 ft of casing when top of dolomite is less than 20 ft below the ground surface (Wisconsin Administrative Code, NR 812.13 and NR 812.14). As a result, it is possible that well casing does not extend into the water table. Thirty-nine percent of wells (n=259) had casing that terminated prior to reaching the water table and were categorized as "ABOVE" or assigned a casing depth below the water table (DBWT) value of 0. The next three DBWT categories consisted of 25 ft increments; the remainder of wells were grouped together as having a DBWT of >75 ft.

4.2.4 Statistical Analysis

Summary statistics (mean, median, standard deviation, minimum, maximum) were generated for nitrate-nitrogen concentrations aggregated by potential explanatory variables including county, land cover, weighted soil drainage rank, uppermost bedrock unit, and casing depth below the water table.

The Kruskal-Wallis test or one-way analysis of variance on ranks was used to test for differences in mean nitrate concentrations between different explanatory variables. The Kruskal-Wallis test is a non-parametric method that is useful when there are unequal sample sizes and/or non-normality. When the Kruskal-Wallis test indicated significant differences, the Wilcoxon rank sum test for pairwise comparison was used post-hoc to determine which categories differed. For non-parametric analysis of means, a value of $\alpha = 0.05$ was used when determining significance.

Multiple linear regression (MLR) was used to investigate the relationship between nitrate-nitrogen concentrations and continuous explanatory variables identified by the Kruskal-Wallis test as having different means. It is appropriate to use MLR when scientific knowledge on causal relationships are understood or widely known; and can be useful for understanding the degree to which explanatory variables influence the dependent variable (Helsel *et al.*, 2020):

$$y = \beta_0 + \beta_1 x_1 + \beta_2 x_2 + \cdots + \beta_k x_k + \varepsilon \qquad \text{[Eq. 1]}$$

where y is the response variable, β_0 is the intercept, x_i is the i^{th} explanatory variable, β_1 is the coefficient for the first explanatory variable, β_2 is the coefficient for the second explanatory variable, β_k is the coefficient for the k^{th} explanatory variable, and ε is the error term.

Because the nitrate-nitrogen concentrations were not normally distributed, a square root transformation was applied to the dataset and the lm function was used to analyze data with multiple linear regression. Backward elimination was used to evaluate various models for the ability to explain variability in the dependent variable. A model with all explanatory variables initiates the elimination process; variables with a p-value greater than 0.05 are removed and the model is analyzed again. The process is repeated until only explanatory variables with significant p-values remain (Faraway, 2002; Haque *et al.*, 2018).

The centroid of every land parcel in Green and Sauk Counties was used to obtain relevant data on explanatory variables using a 500 m buffer. These data allowed the best fitting MLR model to be applied countywide and to predict nitrate-nitrogen concentrations for each parcel in the two counties. Nitrate-nitrogen categories of low (< 3 mg L^{-1}), moderate (3 – 6.5 mg L^{-1}), or high (> 6.5 mg L^{-1}) were used to represent data on maps.

Nitrate-nitrogen concentration trends within individual wells were analyzed using the Mann-Kendall rank correlation (Mann, 1945; Helsel and Frans, 2006; Jurgens *et al.*, 2020). Tests were computed using the R Programming Language (R Core Team, 2022) for private wells that had nitrate-nitrogen data spanning four continuous years. Seasonality was not considered because samples kits were sent to participants and returned around the same time of year. The Mann-Kendall test, which uses the nonparametric correlation coefficient Kendall's tau, was computed using the mk.trend function in the trend package (Non-Parametric Trend Tests and Change-Point Detection) to test for nitrate-nitrogen concentration trends of individual wells. First, the nitrate concentrations of individual wells for the four-year period were arranged in the order they were collected. Increases or decreases were determined by subtracting the time sequential pairs of ordered data. Kendall's S, is determined by the following equation:

$$S = P - M \qquad \text{[Eq. 2]}$$

where P = number of pluses (increases) and M = number of minuses (decreases).

When there are no increases/decreases or the number of increases is equal to the number of times that the concentration decreases, S will be zero or close to zero. An S that is significantly different from zero would indicate a positive or negative trend depending on the sign of S. Tau (τ) is the nonparametric correlation coefficient that accounts for the sample size and is determined by:

$$\tau = \frac{S}{n(n-1)/2} \qquad \text{[Eq. 3]}$$

where S = Kendall's S and n = number of data points. If the null hypothesis were true, S would equal zero, and therefore τ would equal or be close to zero as well. The p-value is determined by comparing τ to the Z statistic which is generated by a normal distribution of the data when S equals zero.

When using the non-parametric Mann-Kendall rank correlation and Sen's slope estimator to assess trends in water-quality data, trends in individual wells were accepted as statistically significant when MK rank correlation p-values were below a significance level (α) of 0.1. With only 4 samples, the lowest p-value possible was 0.08.

The Sen's slope is a measure of the linear rate of change and was computed with the sens.slope function (trend package). Positive Sen's slopes indicate increasing concentrations while negative slopes indicate decreasing concentrations. The Sen's slope provides the ability to compare rates of nitrate-nitrogen concentration increases or decreases between wells.

A repurposing of the Seasonal Kendall test has been used in other climate and water work to investigate trends within regions or categories. In this approach Mann Kendall is performed on individual wells and the results combined for the region or category of interest (Helsel and Frans, 2006; Renard *et al.*, 2008; Skinner, 2022). Here a Regional Kendall test was performed to understand trends at both the county, municipal, and other categorical levels.

$$S_R = \sum_{i=1}^{m} S_i \qquad \text{[Eq 4]}$$

Where *R* = region or category of interest, *m* = number of wells within *R, and i* = individual well.

Briefly, values at or near zero for the summed S_i values of the region (S_R) would suggest the number of wells increasing and decreasing are similar. In the regional Mann-Kendall approach, S_R becomes the Z test statistic and is compared to a normal approximation for when Z is zero. If the test statistic determines that S_R is significantly different than zero, the region is determined to have a significant trend and the slope used to indicate whether the trend is positive or negative. Other researchers provide more detailed descriptions of the regional and seasonal

Mann-Kendall trend analysis (Hirsch *et al.*, 1982; Morton and Henderson, 2008; Helsel *et al.*, 2020).

4.3 Results

4.3.1 Spatial variability and other controls on nitrate-nitrogen concentrations

Concentrations of nitrate-nitrogen were significantly higher in Green County than were measured in Sauk County (Table 1). Data shows increasing mean/median concentrations by percent agricultural category (i.e. sum of continuous corn, cash grain, dairy, and potato/vegetable); this was observed regardless of county (Figure 6). The Kruskal-Wallis test indicated significant differences between all categories with the exception of the 50-75% and >75% categories. No difference in nitrate-nitrogen concentration was observed between areas with <10% and >10% urban land cover (Table 1).

Increasing concentrations of nitrate-nitrogen were observed as the weighted soil drainage rank increases; however, the Kruskal-Wallis test did not indicate that differences were significant at the $\alpha = 0.05$ level. The weighted drainage ranks were clustered within the soil drainage category from 3-4 (n=548) and 4-5 (n=126); these categories also had similar mean and median nitrate concentrations.

The uppermost bedrock units encountered by wells in Green and Sauk County included dolomite (n = 273) and sandstone (n = 285). An additional 71 wells were in areas where private wells accessed water from unconsolidated materials (i.e., sand/gravel) rather than underlying bedrock aquifers. Sandstone resulted in the lowest mean and median nitrate concentration; however, the Kruskal-Wallis test did not indicate differences to be significant (Table 1). Analysis of aquifer influences on nitrate-nitrogen concentrations and percent agricultural activity differs when separated by individual county (Figure 7 and Figure 8). Analysis of casing DBWT was limited to the 88% of wells with detailed well construction information. The >75 ft category had the lowest mean and median nitrate-nitrogen concentrations and was the only DBWT category that was significantly different (Table 1).

4.3.2 Multiple linear regression analysis

Following the backward elimination procedure Model 3 was selected as the best fitting model (Table 2). Approximately 17.6% of the variability in nitrate-nitrogen concentration can be explained by Model 3. All six explanatory variables in Model 3 were significant ($\alpha < 0.01$) and the intercept was not significantly different from zero.

Model 3 was used to develop a parcel level nitrate-nitrogen concentration map for both counties (Figure 9 and 10). Measured nitrate-nitrogen concentrations generally agreed well with predicated values, particularly for low and high categories. It was more common for wells to be miscategorized by one category (ex. predicted moderate when actual value was low or high) then for wells to be off by two categories (ex. predicted high when actual value was low).

4.3.3 Temporal variability of nitrate-nitrogen concentrations

No significant differences were detected in the mean annual nitrate-nitrogen concentrations between any of the four years using the Kruskal-Wallis non-parametric test (Table 3). Mann-Kendall trend analysis was performed on each well with four years of annual testing data (Table 4). The combined dataset for Green and Sauk County suggests no trend in 85.8% of wells, 5.8% of wells increased, while 7.6% decreased. The average nitrate-nitrogen increase was 0.58 mg L^{-1} yr^{-1} and the mean decrease was 0.70 mg L^{-1} yr^{-1}. Green County had approximately 5% more wells showing increasing trends than decreases (Figure 11), while Sauk County had 5% more wells decreasing than increasing (Figure 12). Most wells detecting trends had mean concentrations greater than 1 mg L^{-1} of nitrate nitrogen, which is generally considered to be background or natural levels.

We were able to assess whether trends in individual wells indicated regional trends by applying the Regional Kendall test to our dataset. Trends were investigated first by county and then by individual municipalities (i.e. Town). Overall, the analysis of Green County suggested an increasing trend while Sauk County suggested a decreasing trend ($\alpha = 0.10$ level). Because the slope for the Regional Kendall test utilizes the median slope of all individuals wells to determine the regional slope of the overall dataset, slopes of zero may result from the analysis even when trends are determined to be significant. In these situations, the sign of tau (τ) provides insight into whether the trend is increasing (+ τ) or decreasing (- τ). Because the τ value for Green County was positive, we can determine the trend is increasing, although likely at a low rate. Adams and Mount Pleasant were two towns in Green County identified as having significant increasing trends over the 4-yr study period (Table 5). Seven towns were determined to have decreasing trends, while the Town of Sumpter was the only town in Sauk County determined to have an increasing trend (Figure 11).

The Regional Kendall test was used to investigate categorical variables for potential trends. When analyzing the relationship to agricultural land cover, the combined dataset suggested significant increasing trends in the 33-67% category and decreasing trends in the <33% category (Table 6). When breaking out the data by individual counties, only the 33-67% category had a significant increasing trend in Green County and only the <33% category had a significant decreasing trend in Sauk County.

The same process was applied to categories of weighted soil drainage rank. The combined data set showed an increasing nitrate trend in the >4.840 drainage rank category, but no trend in the <4.840 category (Table 7). When investigating counties separately, Sauk County showed decreasing nitrate trends in both categories, while Green County detected an increasing trend in the <4.840 category only.

Cumulative annual precipitation during the 2019-2022 study period ranged from 676-1258 mm in Green County and 739-1346 mm in Sauk County. Spatial variability of precipitation in Sauk County was greatest in 2019 while in Green County was 2022 (Figure 13 and 14). Annual precipitation values were highly correlated between the two counties ($r = 0.937$). Significant differences were detected between annual precipitation for each year of the study, but differences between counties were not significant (Table 3). Thirteen percent of wells showed a significant correlation ($p < 0.05$) between annual precipitation and nitrate-nitrogen concentrations; however, wells detecting a negative or positive trend and also detecting a correlation between nitrate-nitrogen concentrations and annual precipitation represented <1% of wells tested.

4.4 Discussion

4.4.1 Spatial variability and controls on nitrate concentrations

Well water nitrate-nitrogen concentrations increased as the percentage of agricultural land cover near the well increased. Agricultural activity is a major source of nitrate to water resources (Dinnes *et al.*, 2002; Shrestha *et al.*, 2023) and results presented in this study confirm that private well water quality is influenced by this relationship. The MLR analysis quantified the influence of individual agricultural land cover types on nitrate-nitrogen concentrations.

Potato/vegetable had more than twice the influence on well water nitrate than the next most influential explanatory variable. Dairy rotation was second followed by hay, continuous corn, and finally cash grain. These findings are similar to other research highlighting the role that cropping systems have on nitrate leaching losses (Randall *et al.*, 1997; Heineman and Kucharik, 2022; Shrestha *et al.*, 2023).

In addition to crop type, soil properties influence water drainage and conveyance of contaminants like nitrate to groundwater. Better drained soils are more prone to nitrate losses than areas of finer textured materials and higher organic matter content (Meisinger and Randall, 1991; Tesoriero *et al.*, 2017; Shrestha *et al.*, 2023). Comparison of mean nitrate-nitrogen concentrations by soil drainage categories was inconclusive; however, weighted drainage rank was considered a significant explanatory variable in the MLR analysis. The positive coefficient of weighted drainage rank observed in the best fitting MLR model suggests increasing nitrate-nitrogen concentrations as the soil drainage category around a well becomes more well drained. We suspect the uniformity of soil drainage classification in Green and Sauk Counties may have contribute to finding of no significance in the Kruskal-Wallis analysis of means, and that soil drainage is important to consider in regions with greater soil drainage variability.

While Model 3 is considered highly significant, it is only able to explain 17.6% of nitrate-nitrogen variability. There are a few potential explanations for why the model cannot explain more variability. First, the 500 m buffer used to characterize land cover for analysis is not a true representation of the capture zone. Groundwater flow to private wells is generally not towards the well from all directions but more likely comes from one direction based on changes in the potentiometric surface; therefore a true capture zone would be better thought of as a wedge extending upslope from the well rather than a circle (Johnson and Belitz, 2009).

Without detailed knowledge and datasets on groundwater elevation and aquifer properties, actually defining the groundwater flow direction is challenging. Private wells pump relatively small quantities of water, as a result the captures zone is likely to be quite discrete (Wilcox *et al.*, 2010). Given uncertainties in identifying the true capture zone, there is a risk of assigning too much specificity which could result in a worse fitting model or the introduction of additional error. While a buffer may seem overly simplistic, it appears to be suitable for simple characterization of the dominant land covers responsible for influencing shallow groundwater quality accessed by private wells.

Use of statewide land cover datasets for understanding sources of nitrate is another potential explanation for low MLR model performance. In addition to crop type, actual nitrate losses below agricultural fields are impacted by nitrogen application amounts, yield, tillage practices, cover crops, irrigation, rainfall, use of drain tile, etc. (Dinnes *et al.*, 2002; Quemada *et al.*, 2013; Thapa *et al.*, 2018). The Wiscland layer used here treats land cover as a homogenous unit regardless of farm or individual field management practices. Additional information on these factors would likely improve overall model performance but currently there is no consistent source or easy to way to access that type of information.

The aquifer properties can be important for understanding the conveyance of water and contaminants to the groundwater. Particularly in areas where the soil layer is thin or absent, the physical properties of the uppermost bedrock unit can contribute to groundwater contamination susceptibility. Comparison of mean nitrate-nitrogen concentrations by uppermost bedrock type did not result in significant differences. Further investigation into nitrate separated by county and geologic units showed conflicting results (Figure 7). For example, nitrate-nitrogen concentrations in Sauk County were greatest in the wells installed into unconsolidated materials, while the lowest concentrations were observed in those wells encountering sandstone. The greatest concentrations in Green County were observed in the sandstone and lowest in unconsolidated materials. It is likely that at a county level scale, agricultural suitability is influenced by geology. Boxplots for nitrate-nitrogen concentrations by bedrock type (Figure 7) and percent agricultural activity by bedrock type (Figure 8) show similarities between nitrate and agricultural activity. As a result, we suspect that it is the land cover (mainly agricultural land cover) rather than the uppermost bedrock unit that is reflected in the nitrate/aquifer comparisons (Figure 8).

Casing depth below the water table was another variable that previously has been identified as being an important explanatory variable for nitrate occurrence (Tesoriero *et al.*, 2017). The main function of casing is to prevent unconsolidated materials from collapsing in on the open borehole, but also determines where within the aquifer a well receives its water from. Wells cased deeper below the water table are generally considered to access water that may be older and potentially less likely to contain nitrate (Tesoriero *et al.*, 2017). Casing DBWT was not significant in any of the MLR models. Comparison of DBWT categories detected differences but only in the >75 ft category, suggesting that casing depth below the water table may not significantly lower the risk of elevated nitrate until casing depths extend a minimum of 75 ft into the water table. These results are specific to Sauk and Green counties and findings should not be considered a blanket recommendation since casing depth influences are likely to vary depending on geology, landscape position, and land use. Rather it supports other research indicating casing DBWT is a consideration that can influence nitrate-nitrogen concentrations of wells (Tesoriero *et al.*, 2017).

Applying the best fitting MLR model to predict nitrate-nitrogen concentrations of individual parcels suggests more widespread nitrate impacts in Green County compared to Sauk

County. Greater mean nitrate-nitrogen concentrations observed in Green County and predicted by the MLR model correlate well with areas of agricultural land cover with minor influences attributable to soil drainage.

4.4.2 Individual well and regional nitrate trends

We expected that trends or differences in water quality between years would not likely be detectable with simple random sampling and countywide mean comparison tests on such a large and variable private well data set. The differences in annual mean nitrate-nitrogen concentrations is likely too small while variability was too large for significant differences to be detected. Because groundwater accessed by private wells is likely to be influenced by what happens locally (i.e. near the well) rather than at a countywide scale (Wilcox *et al.*, 2010), we correctly suspected that trends would be more evident within the data of individual wells rather than comparing a random sample from the region.

Trends were detected in 13.4% of private wells; regional analysis provided evidence of increasing and decreasing trends at both county and town scale. The dominant nitrate trend in Sauk County was decreasing while Green County suggested an increasing trend. Three towns detected increasing trends, while seven towns (all in Sauk County) detected decreasing trends. In the case of Town of Sumpter, the trend determination may be the result of the small sample size (n = 3) rather than indicative of an actual trend.

Differences in precipitation could be a potential explanation for dominant trend direction differences between counties. More groundwater recharge would be expected to dilute nitrate-nitrogen concentrations. However, precipitation differences between the two counties were not significant and mean annual precipitation between counties were highly correlated. In addition, the lack of correlation between nitrate-nitrogen concentrations and annual precipitation amounts of individual wells suggests that precipitation is not a contributing factor to trends. Given delays in impacts from land use activities on groundwater quality, 4-years of data presented here potentially limits our ability to correlate climate variability with groundwater quality results. Phase lags have been observed by other studies investigating trends in nitrate concentration (Roy *et al.*, 2007), and may be a possibility here. Additional years of data collection could allow for more detailed investigation of climatic variability on the interpretation of trends which would be particularly helpful in analysis of phase lags.

Categorical analysis of trends by agricultural category suggested that wells with a greater percentage of agricultural land cover were less likely to contain trends. Meanwhile investigating trends using categories of soil drainage classification were inconclusive. Analysis of trends using categorical variables provided limited insight into understanding why certain wells are indicating trends. Machine learning methods such as random forest is an avenue for future investigation that may provide insight into factors or combinations of factors that influence or explain trends in nitrate concentrations (Tesoriero *et al.*, 2017). Regardless of methods used, additional years of data on well water quality would be beneficial in reaffirming trends in wells or identifying trends in additional wells where trends were not observable with only four data points.

4.5 Conclusions

Private well owners are willing and capable partners for characterization and monitoring of regional well water quality. The work reinforces other studies showing that groundwater is influenced by what happens locally and that groundwater nitrate-nitrogen concentrations are heavily influenced by agricultural activity. The majority of wells did not show significant trends; however, regional trend analysis suggested Sauk County nitrate-nitrogen concentrations to be decreasing while Green County concentrations were increasing over the four year period . Trends could not be explained with generalized data sets. Additional years of data will help confirm whether these changes in water quality are short-term or long-term trends. Meanwhile detailed data (i.e., nitrogen fertilizer rates, yield measurements, manure applications, etc.) could be beneficial in interpreting trends, but obtaining this type of detailed data is not readily available, particularly at regional scales.

Well owners have a responsibility when it comes to maintaining the integrity of their well water system, this includes routine testing to determine its safety. It is not feasible nor necessary that every single private well be tested annually, however valuable information can be learned about well water quality regionally if a subset of wells is monitored and the data is collected in a systematic way. Local governments or state agencies do not need to be responsible for testing every single private well but should be more intentional about collecting data that can be used for understanding temporal changes in well water quality over time. Encouraging well owners to participate in community monitoring programs can be done with something as simple as a free and convenient well water test.

This approach provides a template for other communities to collect useful information regarding changes in well water quality that can be useful for addressing land management decisions and future well water outreach activities. The longer the period of record, the more useful this type of data becomes. However, sustaining these networks can be challenging because of participant engagement and lack of designated funding sources for this type of monitoring. Communities looking to replicate this type of work should consider a minimum sample size to ensure that trend analysis is representative of the region of interest. Lastly, standard approaches should be developed to address attrition of well testing participants and well replacement procedures that maintain the long-term statistical integrity of the trend monitoring network.

4.6 Acknowledgements

This work would not have been possible without the willingness of well owners to submit samples from their wells every year, or the County Boards of Green and Sauk Counties who voted for the water testing projects. Funding for the Green County well testing was provided by Green County, while in Sauk County the well testing was funded by Sauk County with additional funding support from the Ho-Chunk Nation for the initial year of the project. Victoria Solomon and Jennifer Erickson were instrumental in getting these projects started in their respective counties. The following entities assisted with the project in Green County: Extension Green County, Green County Health Department, Green County Land & Water Conservation Department, Green County Land Information Office, and Green County Land Use & Zoning Department. County. The following entities assisted with the project in Sauk County: Sauk County Extension, Sauk County Health Department, Sauk County Land Resources and Environment. The staff and students at the Water and Environmental Analysis Laboratory were instrumental in processing the thousands of samples and managing data.

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4.8 Tables

Table 1. Nitrate-N concentration summary statistic by factor for variables of interest. Kruskal-Wallis and Wilcoxon pairwise comparison rank sum test results reported for each variable.

			Nitrate-N (mg L^{-1})			Kruskal-Wallis		Wilcox
Variable	**Factor**	**n**	**Mean**	**Median**	**SD**	**Chi-square**	**p-val**	**Pairwise Comp.**
County	Green	348	5.4	5.0	4.4	25.4	<0.001	a
	Sauk	406	4.1	2.5	4.9			b
Bedrock	Dolomite	273	4.9	4.0	4.2	4.94	0.085	-
	Sandstone	285	4.3	2.9	4.1			-
	Unconsolidated	71	5.6	4.0	6.7			-
Percent Ag	<25%	317	3.2	2.1	3.6	87.5	<0.001	d
	25-50%	219	4.7	3.7	4.8			c
	50-75%	143	6.3	6.1	4.5			a
	>75%	75	7.8	7.4	6.2			ab
Weighted Drainage Rank	<3	57	3.6	2.2	3.7	10.3	0.067	-
	3-4	548	4.7	3.5	4.3			-
	4-5	126	4.8	3.1	6.0			-
	5-6	19	7.4	4.4	8.7			-
Urban	<10%	642	4.6	3.5	4.6	0.002	0.964	-
	>10%	112	5.0	3.5	5.6			-
Casing	Above	259	4.1	3.4	3.8	29.9	<0.001	a
	0-25 ft	118	5.6	3.5	6.0			a
	25-50 ft	111	5.5	4.6	4.9			a
	50-75 ft	87	5.4	4.4	4.8			a
	> 75 ft	91	3.2	1.7	4.0			b

Values within each column followed by the same letter are not significantly different based on post-hoc pairwise comparison using Wilcoxon rank sum test

(-) Kruskal Wallis not significant, post-hoc pairwise comparison not performed

Table 2. Multiple linear regression analysis applied to modeling of square root transformed nitrate-nitrate concentrations and explanatory variables related to land cover, soils, and well construction.

	Model 1	**Model 2**	**Model 3**	**Model 4**
Variable	Coefficients			
Cash Grain	1.695***	1.387***	1.393***	1.241***
Continuous Corn	1.636***	1.561***	1.573***	1.345**
Dairy Rotation	1.730***	1.812***	1.816***	1.796***
Potato/Vegetable	4.290**	4.357**	4.363**	4.971***
Hay	1.604***	1.694***	1.675***	1.574***
Pasture	-0.180	-0.051	-	-
Drainage Rank	0.261***	0.214**	0.216**	-
Depth Below WT	-0.001	-	-	-
Intercept	-0.096	0.141	0.126	1.193***
Degrees of freedom	662	746	747	748
R-squared	0.189	0.176	0.176	0.164
p-value	<2.2e-16	<2.2e-16	<2.2e-16	<2.2e-16

***Significant at the p < 0.001 level
** Significant at the p < 0.01 level
* Significant at the p < 0.1 level
(-) Explanatory variable not included in the model

Table 3. Mean annual precipitation and nitrate-nitrogen concentrations for wells tested in all four years of the study. Standard deviation in parentheses. Chi-square and p-values reported for Kruskal-Wallis test investigating differences within county between study years.

			Precipitation			Nitrate-N		
County	**Year**	**n**	**Mean**	**Chi-square**	**p-val**	**Mean**	**Chi-Square**	**p-val**
			mm			**mg L-1**		
Green	2019	294	1258(46)a	910.1	<0.001	5.4(4.4)†	0.811	0.847
	2020	294	1010(31)c			5.8(4.8)		
	2021	294	676(22)d			5.7(4.7)		
	2022	294	1026(50)b			5.5(4.4)		
Sauk	2019	327	1346(94)a	1089	<0.001	4.2(5.1)†	0.531	0.912
	2020	327	1023(57)b			4.4(5.5)		
	2021	327	739(29)d			4.1(4.8)		
	2022	327	913(35)c			4.1(4.8)		

Values within each column followed by the same letter are not significantly different based on post-hoc pairwise comparison using Wilcoxon rank sum test
†Kruskal Wallis not significant, post-hoc pairwise comparison not performed

Table 4. Summary of nitrate-nitrogen concentration and Sen slope estimates by county and trend category identified using Mann-Kendall trend analysis.

		Wells		Nitrate-Nitrogen (mg L^{-1})			Slope (mg L^{-1} yr^{-1})			
County	**Trend**	**N**	**%**	**Mean**	**Median**	**StDev**	**Mean**	**Median**	**Min**	**Max**
Green	Decrease	16	5	5.9	5.0	4.0	-0.67	-0.69	-1.25	-0.14
	No trend	249	85	5.4	4.8	4.5	0.0	0.0	0.0	0.0
	Increase	29	10	6.3	5.9	3.8	0.52	0.30	0.14	1.75
Sauk	Decrease	31	9	3.8	2.7	2.7	-0.72	-0.42	-6.52	-0.10
	No trend	284	87	3.9	2.1	4.9	0.0	0.0	0.0	0.0
	Increase	12	4	7.5	6.7	4.3	0.74	0.73	0.20	1.13

Table 5. Regional Mann Kendall summary by municipality and county. Bold p-values highlight municipalities with significant trends (α = 0.10).

County	Municipality	N	p-value	Z	τ	Slope	Intercept
Green	Adams	12	**0.030**	2.174	0.333	0.200	-733.2
	Albany	17	0.183	1.332	0.167	0.117	-229.4
	Brooklyn	18	0.612	0.508	0.061	0.000	-61.9
	Cadiz	15	0.848	-0.192	-0.012	0.000	-95.2
	Clarno	18	0.732	-0.343	-0.062	0.000	17.3
	Decatur	22	0.523	-0.638	-0.068	0.000	0.1
	Exeter	20	0.722	0.356	0.053	0.000	-181.3
	Jefferson	12	0.434	-0.783	-0.111	0.000	0.1
	Jordan	16	0.217	1.236	0.143	0.000	-16.1
	Monroe	21	0.299	1.038	0.104	0.000	0.1
	Mount Pleasant	20	**0.020**	2.331	0.248	0.100	-161.3
	New Glarus	27	0.412	-0.820	-0.084	0.000	0.1
	Spring Grove	15	1.000	0.000	-0.017	0.000	96.3
	Sylvester	23	0.879	0.152	0.022	0.000	36.4
	Washington	17	0.113	1.587	0.196	0.117	-28.9
	York	21	0.670	0.427	0.052	0.000	0.1
		293	**0.064**	1.849	0.050	0.000	0.1
Sauk	Baraboo	18	0.303	-1.030	-0.120	0.000	21.3
	Bear Creek	22	**0.008**	-2.641	-0.238	0.000	35.5
	Dellona	18	**0.082**	-1.742	-0.204	-0.075	19.3
	Delton	19	**0.055**	-1.917	-0.193	0.000	35.9
	Excelsior	16	0.930	-0.088	-0.021	0.000	38.3
	Fairfield	17	0.847	0.193	0.035	0.000	0.1
	Franklin	19	0.930	0.087	0.018	0.000	0.1
	Freedom	7	0.882	-0.149	-0.048	0.000	1.4
	Greenfield	11	0.333	0.968	0.136	0.000	1.5
	Honey Creek	18	**0.081**	-1.743	-0.194	-0.013	60.4
	Ironton	11	**0.022**	-2.282	-0.333	-0.100	205.3
	La Valle	20	0.795	-0.260	-0.051	0.000	34.9
	Merrimac	9	0.229	-1.203	-0.185	0.000	0.1
	Prairie du Sac	10	0.909	-0.115	-0.033	0.000	74.6
	Reedsburg	16	0.857	-0.181	-0.031	0.000	0.2
	Spring Green	26	0.533	-0.623	-0.064	0.000	34.9
	Sumpter	3	**0.046**	2.000	0.611	0.175	-332.1
	Troy	13	0.835	-0.208	-0.038	0.000	0.1
	Washington	8	0.467	0.727	0.125	0.000	0.1
	Westfield	14	0.255	-1.139	-0.133	0.000	2.8
	Winfield	15	**0.002**	-3.082	-0.391	-0.100	274.8
	Woodland	13	0.740	0.331	0.051	0.000	7.4
		326	**<0.001**	2.661	-0.104	0.000	6.1

Table 6. Mann Kendall trend summary by agricultural land cover. Bold p-values highlight categories with significant trends (α = 0.10).

County	Agricultural Land Cover	N	p-value	Z	τ	Slope	Intercept
Green	<33%	84	0.651	0.453	0.025	0.000	-15.7
	33-67%	101	**0.001**	3.450	0.167	0.100	-129.4
	>67%	65	0.285	-1.070	-0.067	-0.067	96.3
Sauk	<33%	175	**<0.001**	-5.487	-0.195	-0.05	135.3
	33-67%	74	0.773	-0.288	-0.018	0.000	6.7
	>67%	18	0.806	0.245	0.037	0.000	109.2
Combined	<33%	260	**<0.001**	-4.183	-0.123	-0.033	85.0
	33-67%	175	**0.015**	2.432	0.089	0.033	-65.3
	>67%	83	0.426	-0.796	-0.044	-0.017	96.3

Table 7. Mann Kendall trend summary by drainage classification. Bold p-values highlight categories with significant trends (α = 0.10).

County	Drainage Classification	N	p-value	Z	τ	Slope	Intercept
Green	<4.840	165	**0.001**	3.234	0.122	0.067	-65.3
	>4.840	86	0.231	-1.199	-0.064	-0.033	1.1
Sauk	<4.840	94	**0.003**	-2.946	-0.144	-0.033	97.1
	>4.840	173	**<0.001**	-3.413	-0.123	-0.050	69.3
Combined	<4.840	259	0.396	0.841	0.026	0.000	6.25
	>4.840	259	**<0.001**	-3.494	-0.104	-0.033	37.1

4.9 Figures

Figure 1. Map of land cover and percentages of each category for Green County.

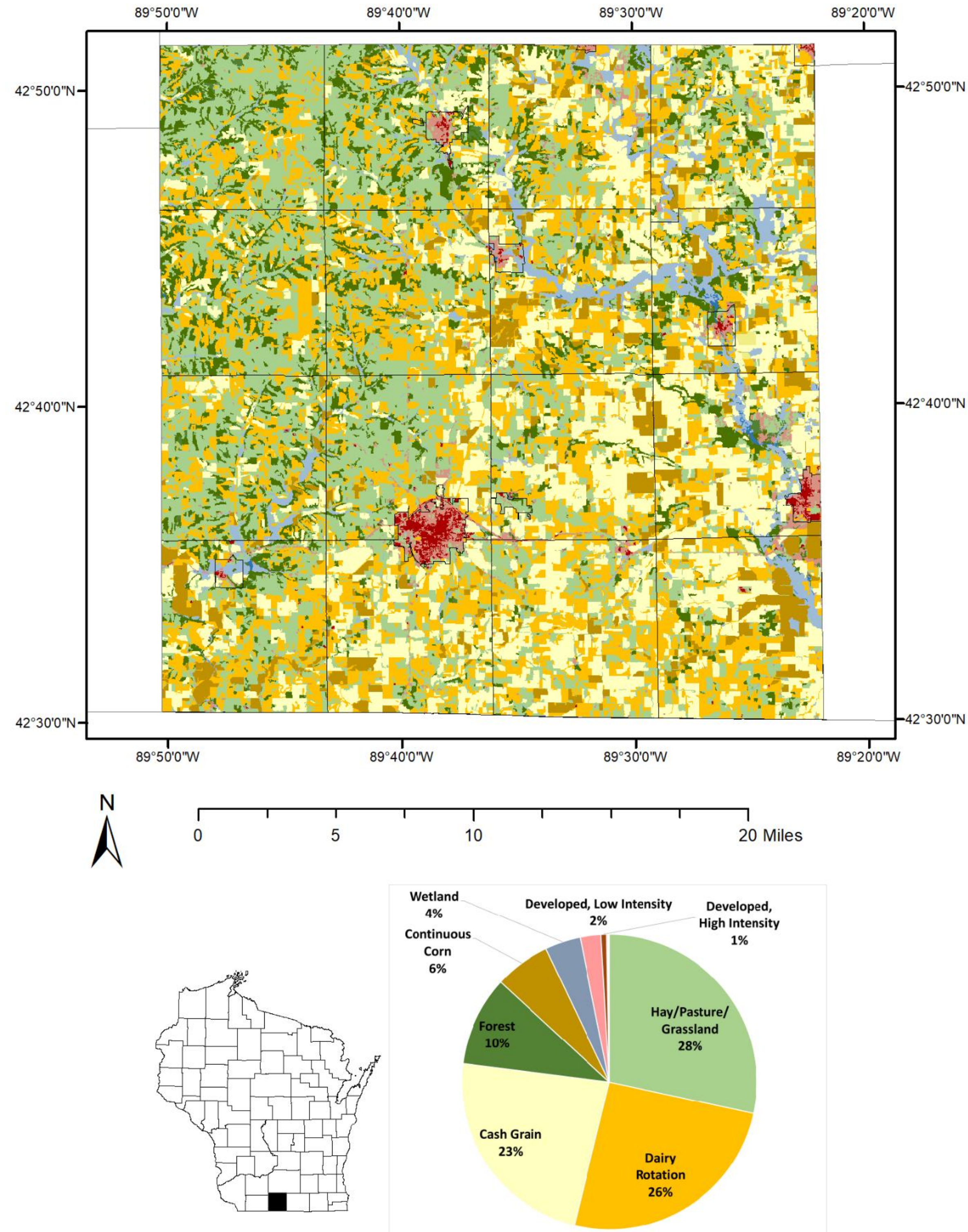


Figure 2. Map of land cover and percentages of each category for Sauk County.

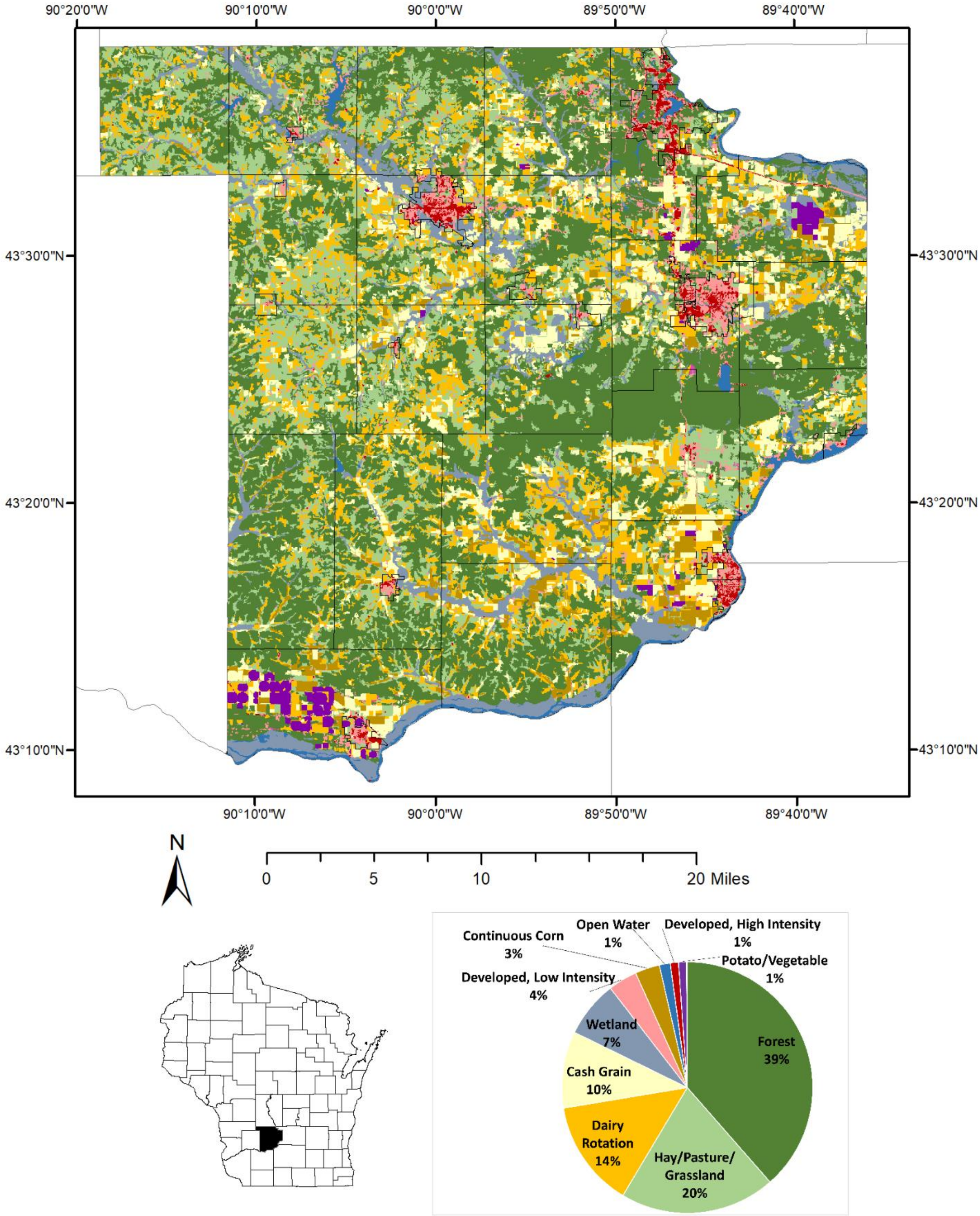
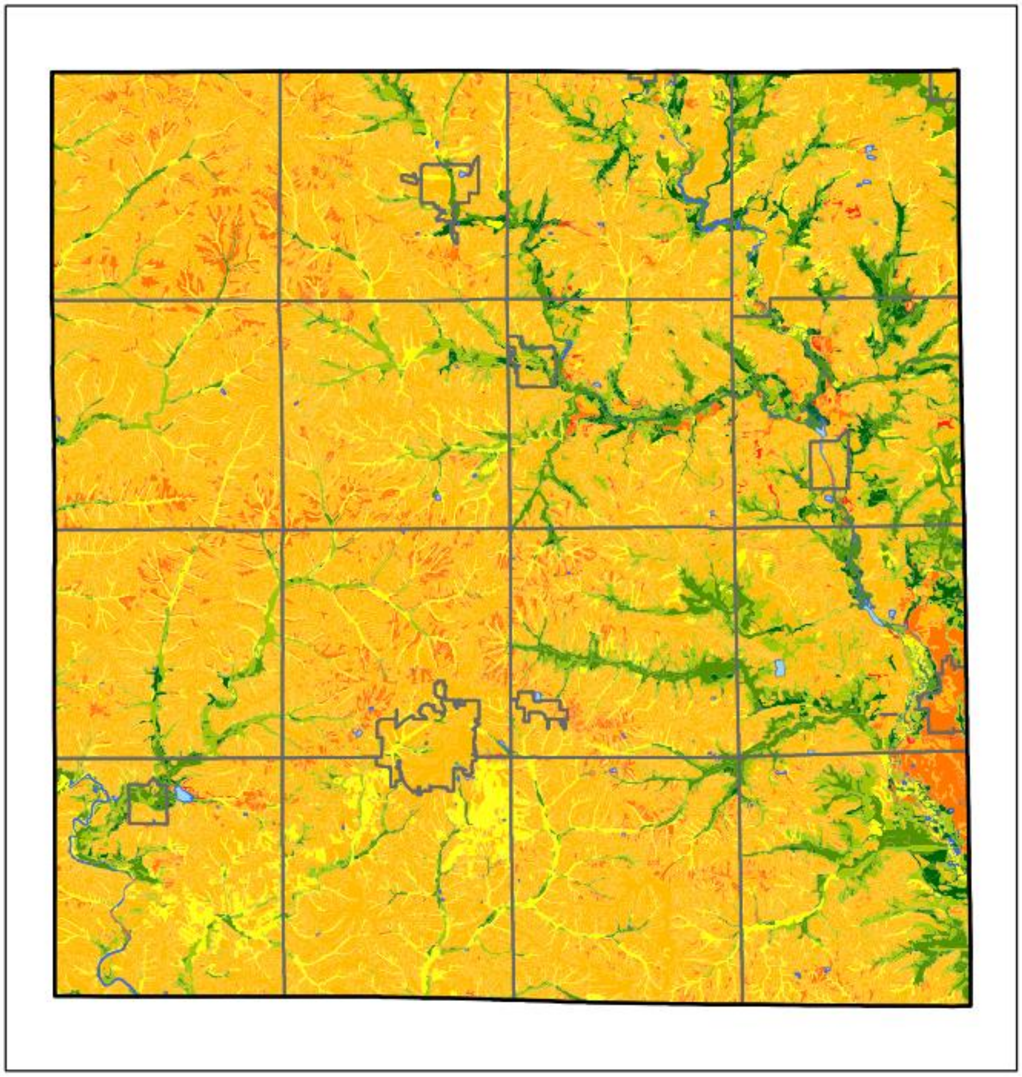


Figure 3. Soil drainage classification map for Green County



Source: Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture. Soil Survey Geographic (SSURGO) Database

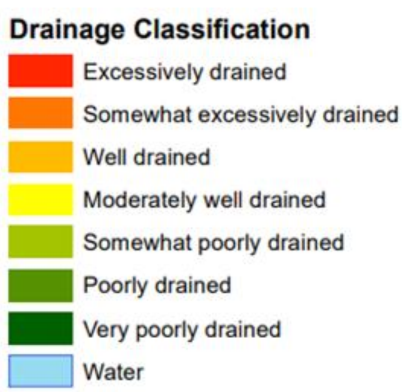
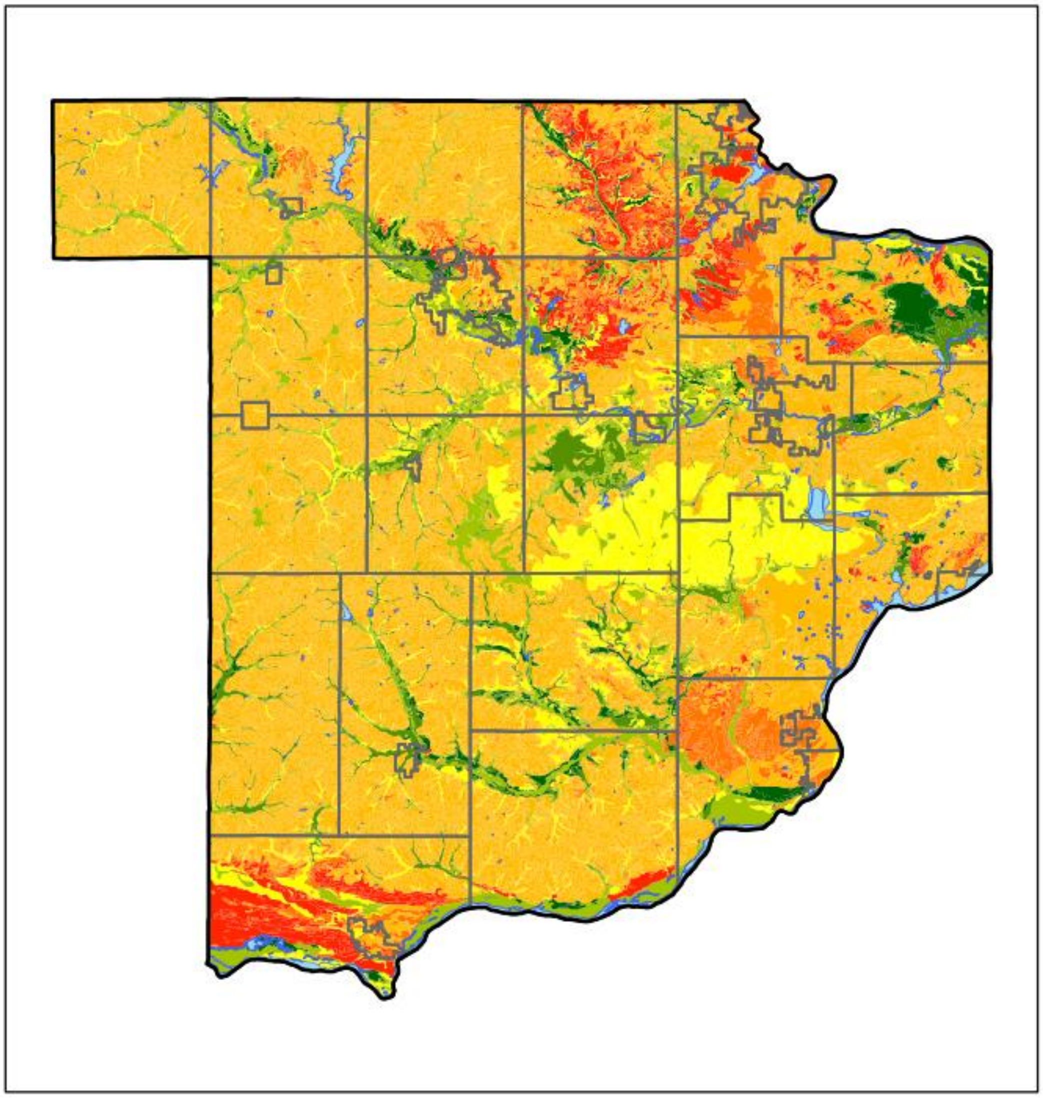


Figure 4. Soil drainage classification map for Sauk County.



Source: Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture.
Soil Survey Geographic (SSURGO) Database

Drainage Classification

- Excessively drained
- Somewhat excessively drained
- Well drained
- Moderately well drained
- Somewhat poorly drained
- Poorly drained
- Very poorly drained
- Water

Figure 5. Density of land cover and drainage characteristics within 500 m buffer of wells sampled in Green County (green) and Sauk County (yellow).

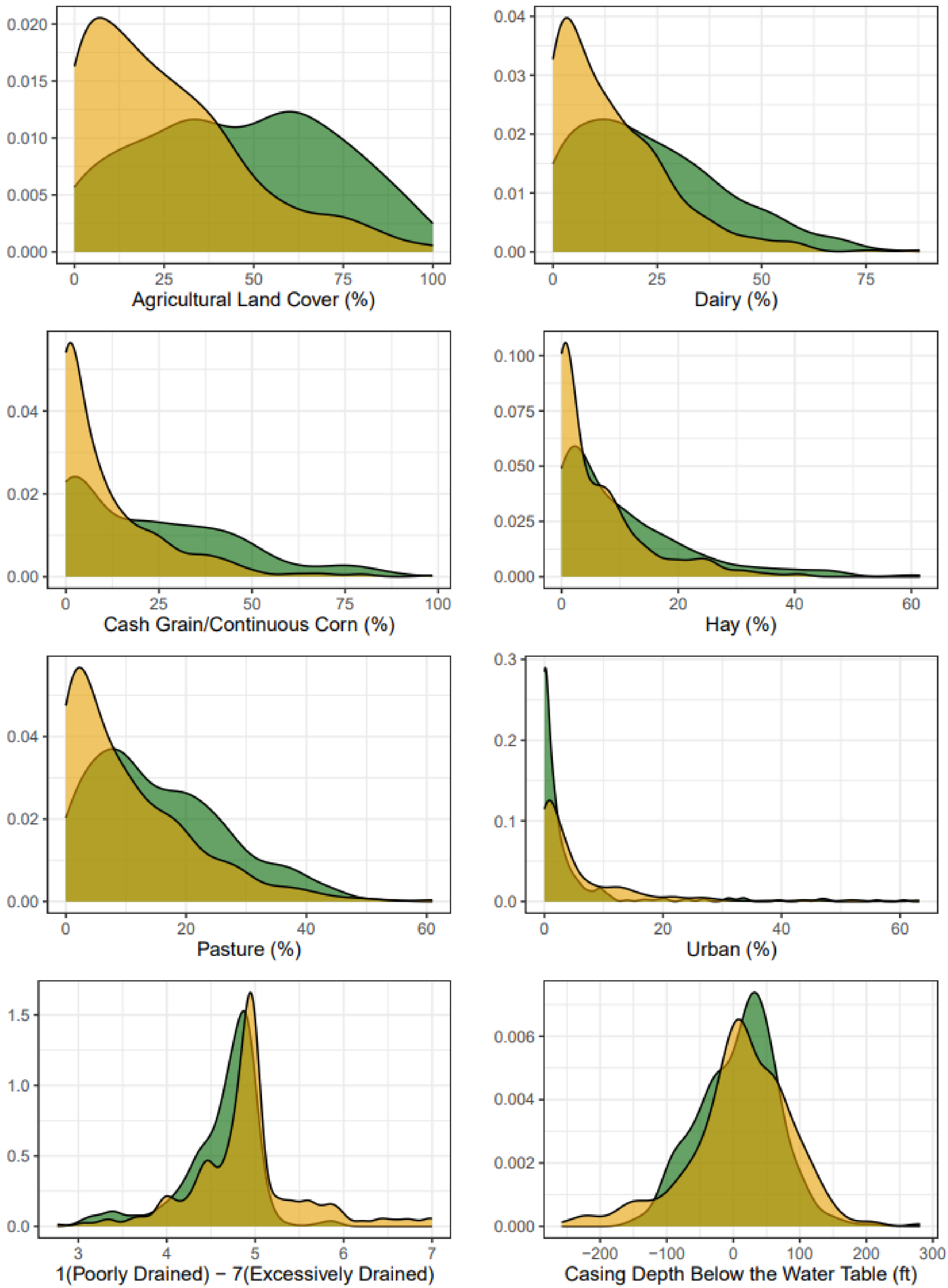


Figure 6. Box plots of nitrate-nitrogen concentrations by percent agricultural land cover (sum of continuous corn, cash grain, dairy, and potato/vegetable).

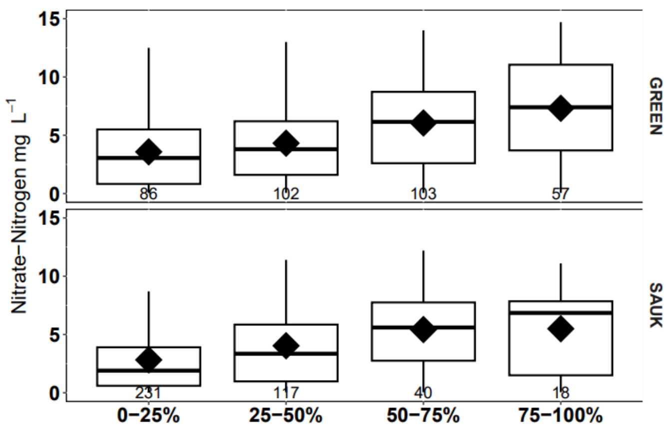


Figure 7. Box plots of nitrate-nitrogen concentrations by aquifer classification.

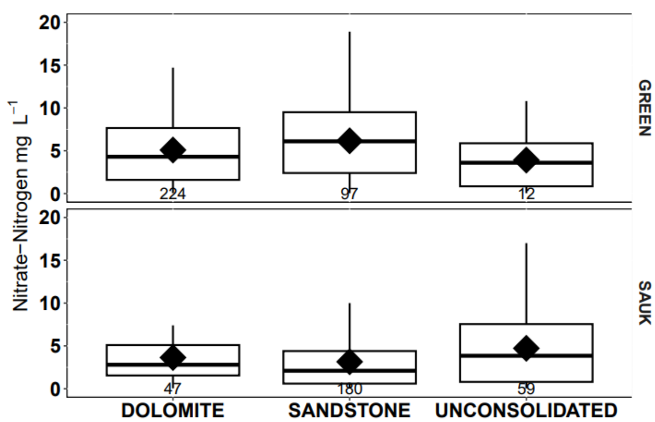


Figure 8. Box plots of percent agricultural activity within 500 m buffer of well by aquifer classification.

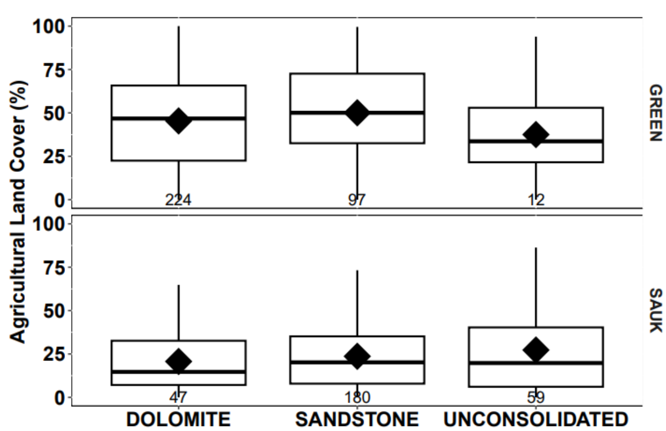


Figure 9. Nitrate-nitrogen concentrations by individual wells sampled in 2019 with output of predicted nitrate-nitrogen concentrations in Green County.

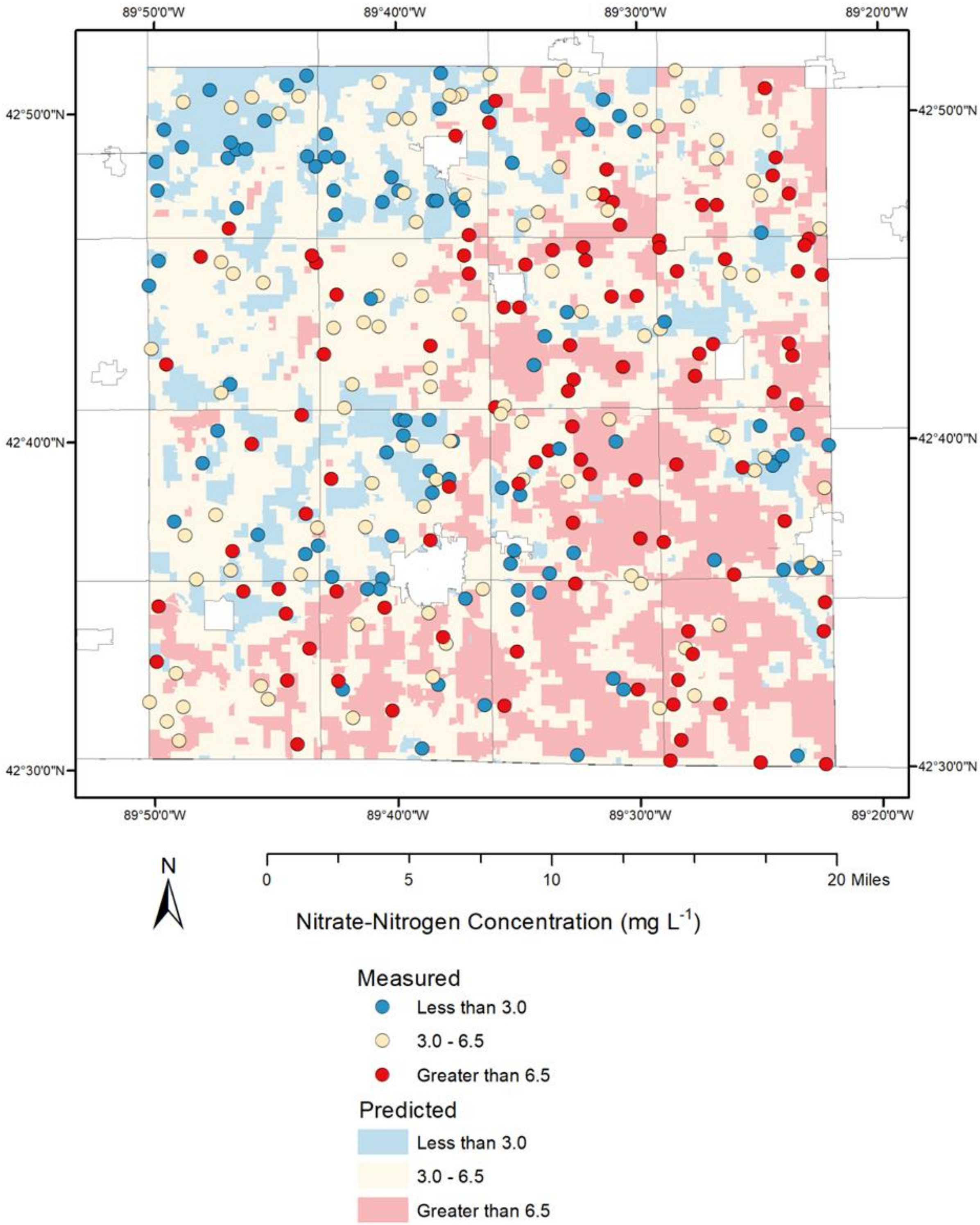


Figure 10. Nitrate-nitrogen concentrations by individual wells sampled in 2019 with output of predicted nitrate-nitrogen concentrations in Sauk County.

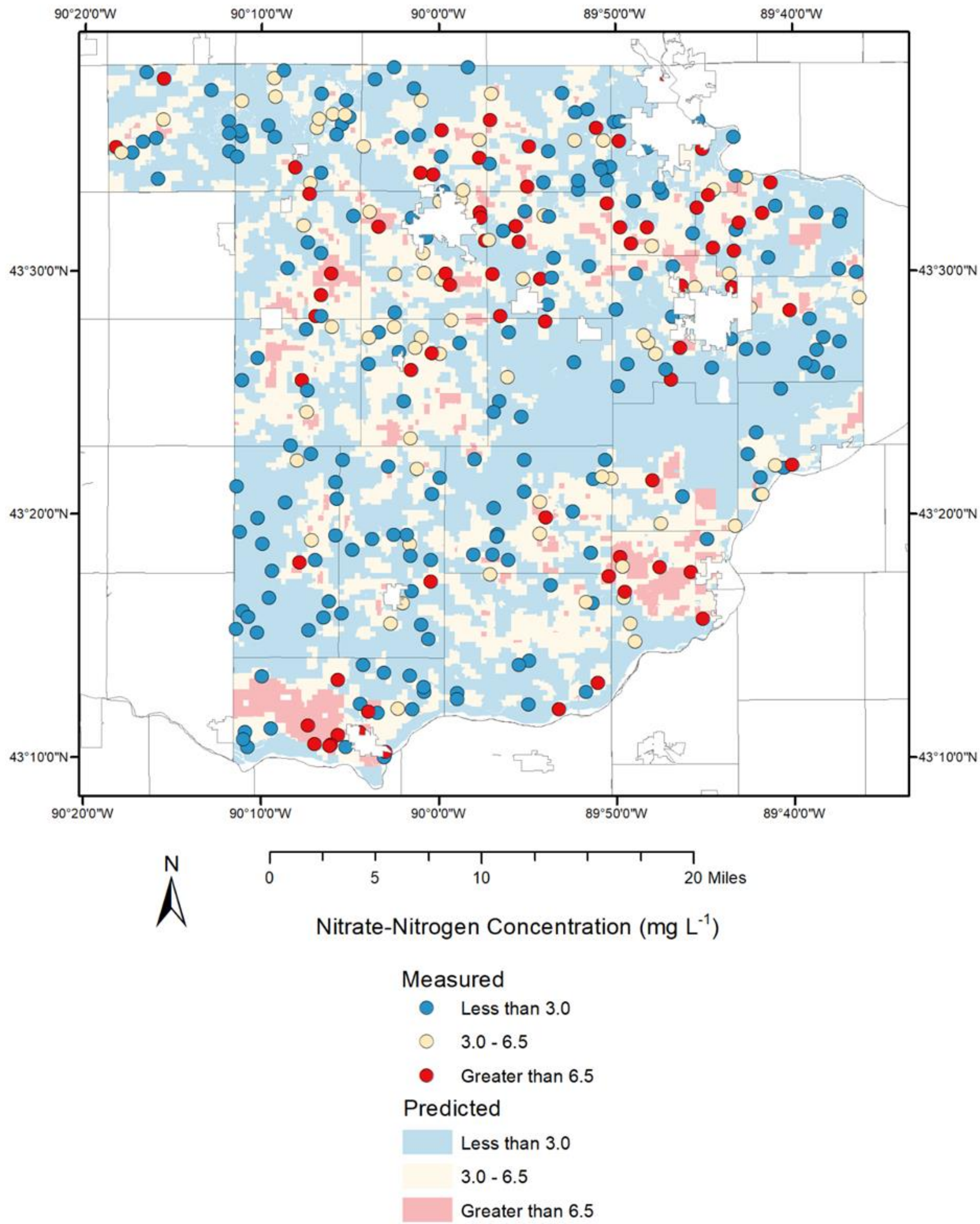


Figure 11. Mann Kendall trend analysis for Sauk County. Increasing (red) or decreasing (blue) symbols represent individual wells with a significant trend, the symbol size represents the Sen's slope estimate or rate of change (mg L^{-1} yr^{-1}). Municipalities with significant increasing (pink), decreasing (light blue), or no trend detected (grey) as determined by the Regional Kendall test.

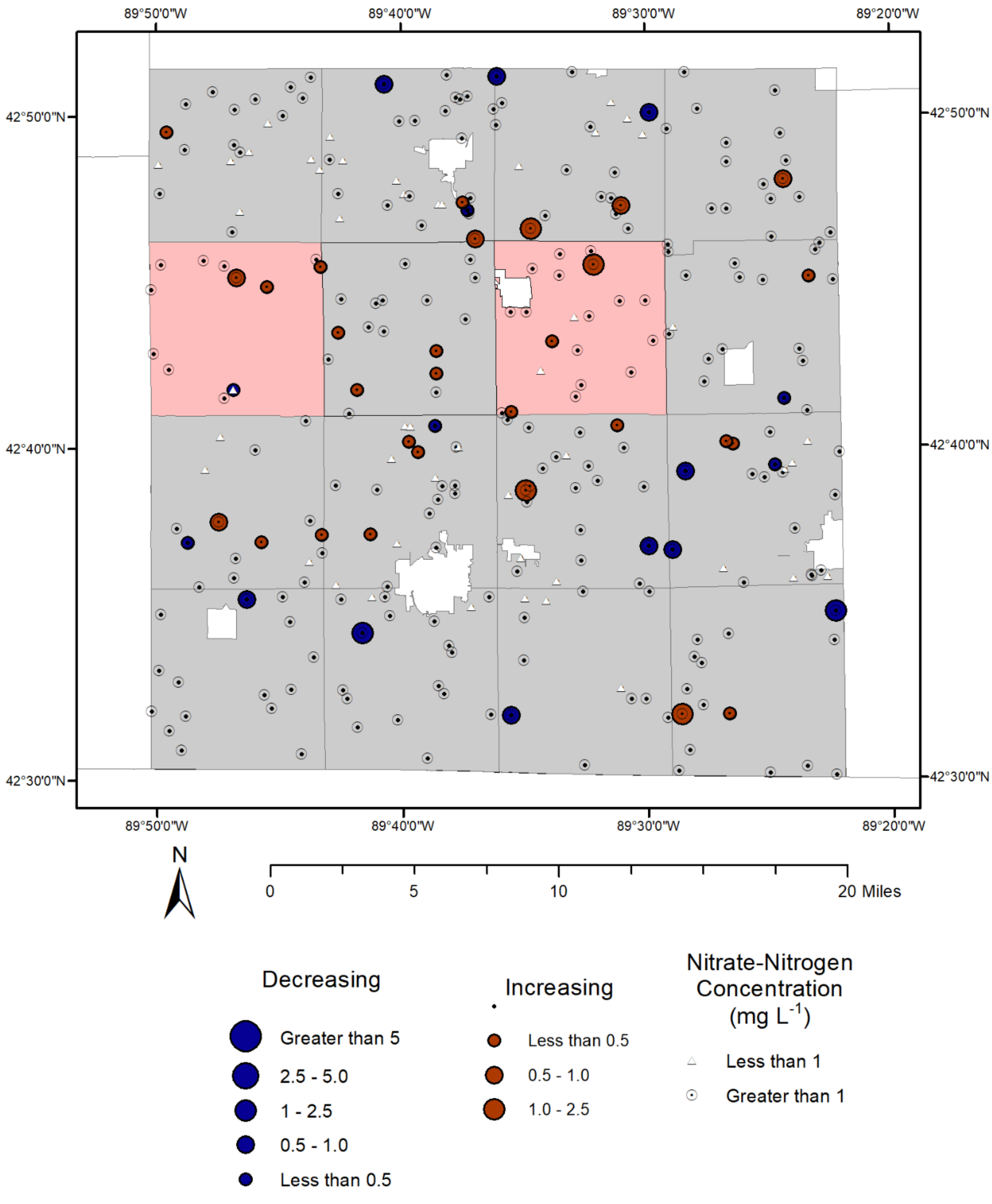


Figure 12. Mann Kendall trend analysis for Sauk County. Increasing (red) or decreasing (blue) symbols represent individual wells with a significant trend, the symbol size represents the Sen's slope estimate or rate of change. Municipalities with significant increasing (pink), decreasing (light blue), or no trend detected (grey) as determined by the Regional Kendall test.

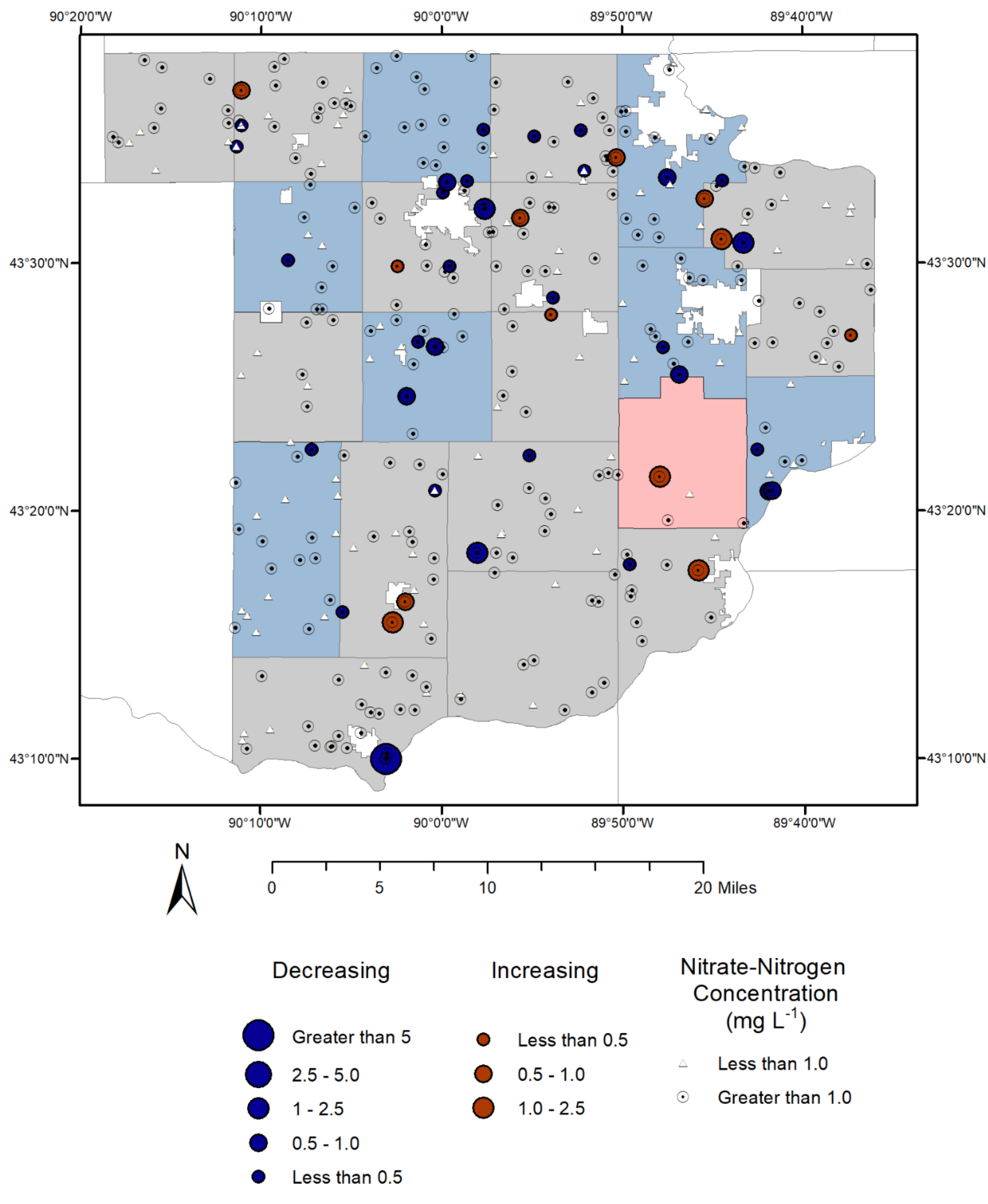


Figure 13. Annual precipitation for well locations in Green County for the 4-year study period.

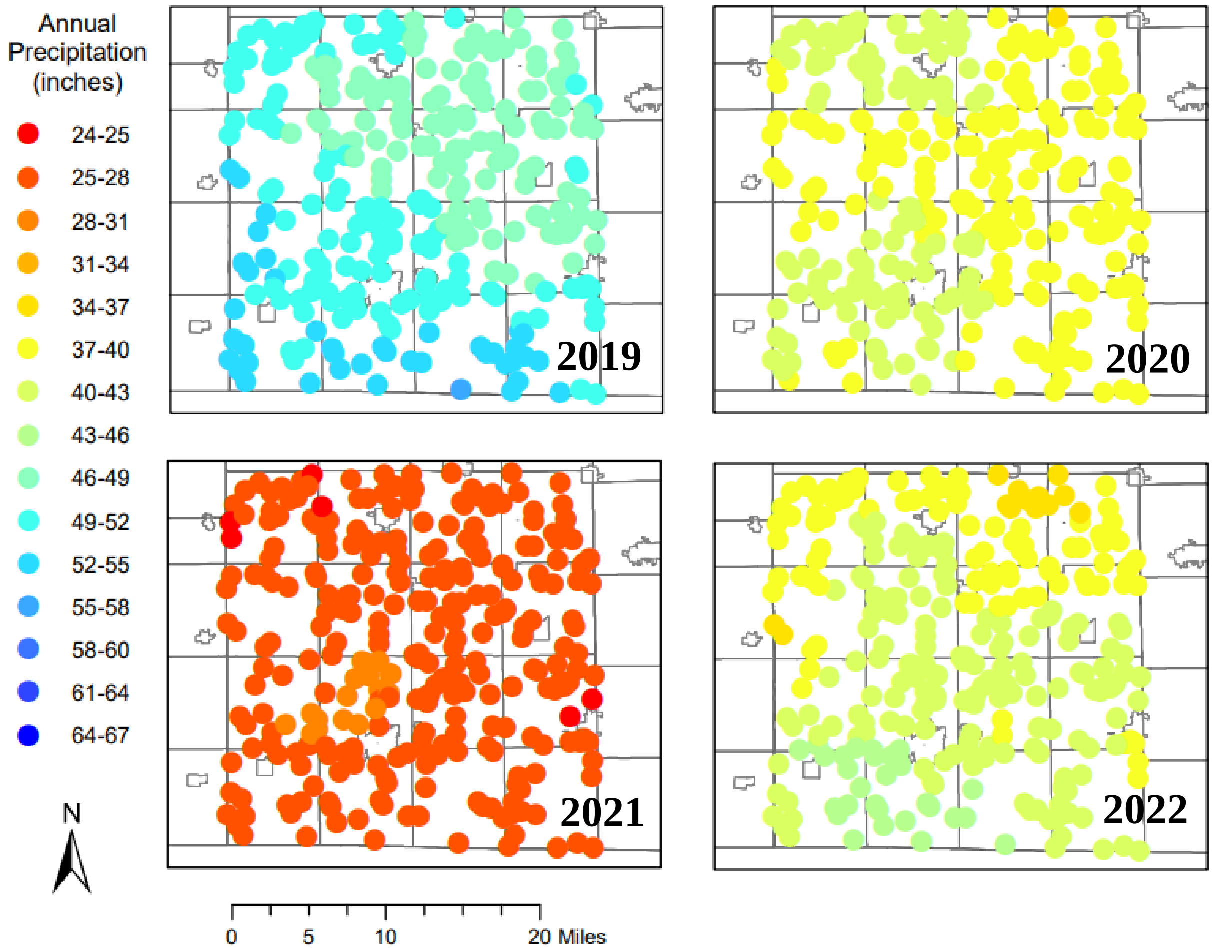
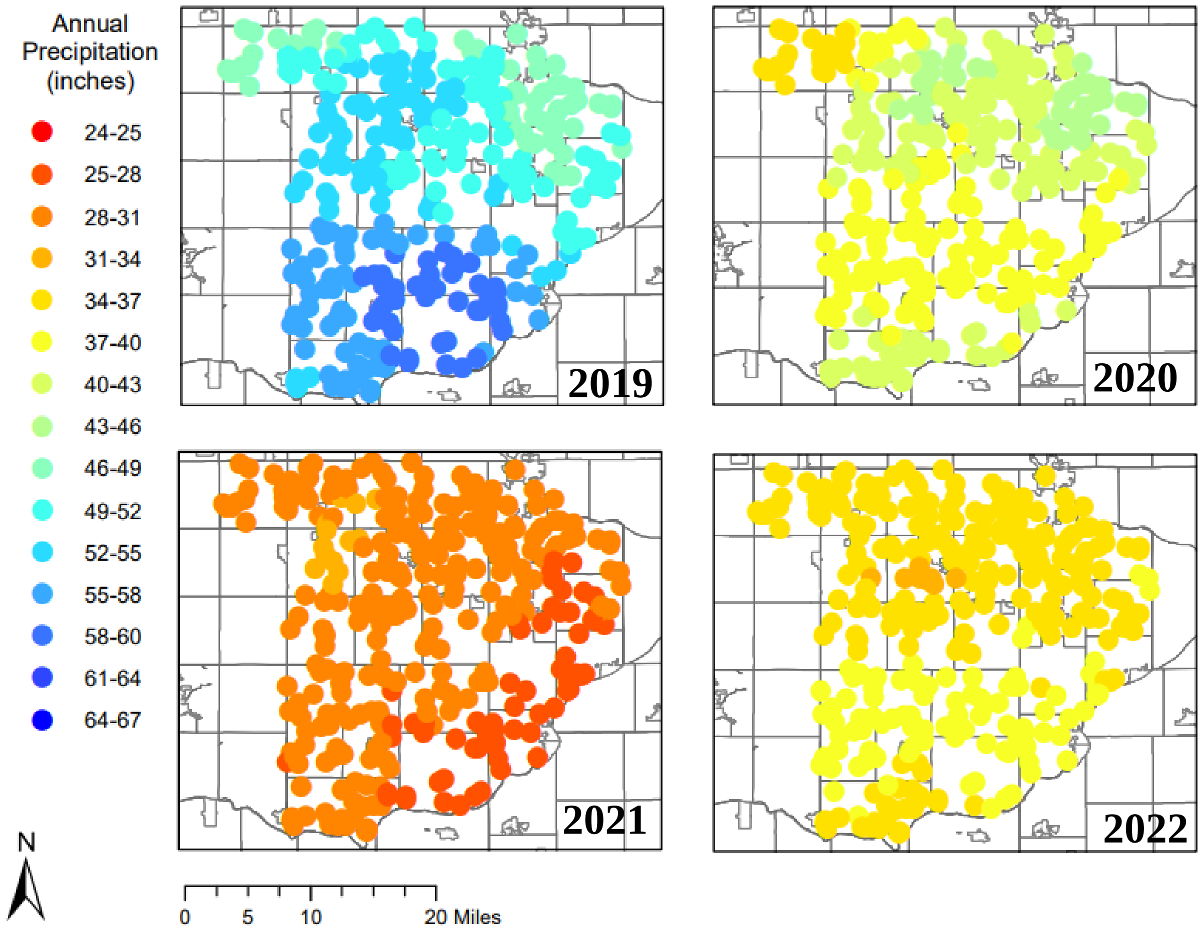


Figure 14. Annual precipitation for well locations in Sauk County for the 4-year study period.



Chapter 5

Applying risk communication theoretical frameworks to understand behavior of private well water users

Abstract

Drinking water is an ongoing risk communication challenge, particularly when messaging to private well owners. Private well owners consist largely of rural residents who, besides lacking government safeguards from exposure to drinking water contaminants, have fewer mechanisms to secure safe water and may have less access to diagnosis and health care when health problems arise. Private well owners are expected to be aware of the risks, do their own water testing, decide on appropriate courses of action, and implement solutions to prevent or reduce health risks when appropriate. I applied the extended parallel process model (EPPM) and theory of reasoned action to data from two previous surveys of private well owners to 1) better understand the attitudes and beliefs of private well owners as they may relate to motivation or intention and 2) identify potential barriers to performing certain risk-reducing behaviors such as well water testing. The results are utilized here to identify potential strategies for effective communication with this audience.

5.1 Introduction

5.1.1 Overview of Well Water Users and Concerns

Improvements in sanitation and water quality are arguably the United States' greatest public health achievements. But an estimated two and a quarter million people, or 42% of Wisconsinites get their water from more than 900,000 private wells that have none of the systematic scrutiny that ensures the safety of those serviced by municipal water systems (Hutson, 2004). Statewide, 10% of wells tested exceed the state and federal drinking water standards for nitrate, the most common health-related contaminant; with this increasing to more than 20% in agricultural areas, and more than 50% in some towns (DATCP, 2017a; CWSE, 2019).

Waterborne pathogens (bacteria, viruses, protozoa, etc.) cause serious gastrointestinal illnesses. Statewide, an estimated 18% of private wells contain evidence of coliform bacteria, indicating a pathway for waterborne pathogens to enter the water supply (Knobeloch *et al.*, 2013).

Nitrate causes “blue baby” syndrome or methemoglobinemia and is linked to reproductive disorders including increased risk of miscarriage. When nitrate converts to nitrite in the body it can then convert into the carcinogen N-nitroso compounds (NOC’s). NOC’s are some of the strongest known carcinogens and have been found to cause cancer in several organs. Increased cancer risks linked to nitrate contaminated drinking water include: non-Hodgkin’s lymphoma; gastric cancer; and bladder and ovarian cancer in older women (Ward *et al.*, 2010).

Naturally occurring arsenic is found in wells statewide but is of particular concern in Outagamie and Winnebago counties (Luczaj and Masarik, 2015). Arsenic exposure is associated with various diseases including cancer, cardiovascular disease, and diabetes (National Research Council, 2001). Portions of deep sandstone aquifers and crystalline bedrock aquifers in Wisconsin naturally exceed standards for radium in well water (Luczaj and Masarik, 2015). Radium exposure is associated with increased cancer risk (WHO, 2011).

Fluoride is important to development of teeth that are resistant to dental caries (CDC, 2001). While most municipal water contains appropriate fluoride levels via supplementation, Wisconsin well water varies in its native fluoride content. Although supplementation is usually required, too much fluoride is toxic. Without a centralized information source for local well water fluoride concentrations, dentists and physicians are often uncertain whether, and how much, supplement is required, leading to lost opportunities to prevent poor oral health and lifelong consequences.

These numbers add up to a large number of Wisconsin residents at risk for a variety of acute and chronic health problems as a result of the water that they drink every day. While there often is not the detailed medical history data to tie individual health problems to contaminated well water, the number of potential exposures and the variety and severity of health effects is compelling.

5.1.2 Defining Risk in the Context of Drinking Water

Risk is often defined in different ways, but one proposed definition describes risk as a situation where something of human value is at stake and the outcome is uncertain (Aven and Renn, 2009). In the case of drinking water, health of oneself or ones family is the value at stake. Uncertainty in this situation is whether health will be negatively affected by drinking their water.

Risk can be defined in both subjective and objective terms – containing both fact and value laden judgements (Hansson, 2010). The risks of negative health effects from contaminants are often expressed as probabilities, which may appear like a strictly objective process. However, drinking water standards are in fact both objective and subjective.

Toxicology studies utilized when setting drinking water standards are an objective process that determines likelihoods of suffering a health outcome from some environmental exposure. For chronic related health outcomes a typical standard may be conveyed as a probability, for example a 2-3 per 10,000 mortality risk estimate associated with a lifetime exposure to 10 $\mu g\ L^{-1}$ of arsenic in drinking water (National Research Council, 2001). However, most drinking water standards weigh the scientific risk estimate or probability of illness (fact laden) against what is economically feasible given current technologies (value laden). For municipal water systems, this type of standard approach to providing water that meets socially defined levels of safe water takes out much of the complexity associated with individual risk perception and variability within the population (Humphreys, 2022). It is an effective strategy for water utilities because of the systems that are in place and the culture of those relying on municipal water.

Cultural theory proposes that societies are arranged into groups varying from low order to highly structured entities (Rayner, 1992). The way in which societies or particular groups of individuals interact and are arranged will affect perceptions of risk. There are significant differences between characteristics of populations served by municipal water versus those that rely on private wells. According to cultural theory, those populations served by municipal water would be defined as a high-group, high-grid structure (Rayner, 1992). Characteristics of this group include interconnected networks, frequent interactions (e.g., access to public utilities and other government services), closed boundaries (homes serviced by utility are connected in discrete area), many shared activities, vertical accountability, great specialization (i.e., designated water staff) and hierarchical resource allocation.

Drinking water standards are a social construct (regulatory framework) that is likely to be effective because of the social organization within the high-grid, high group structure. Standards have become not only a regulatory framework, but also a heuristic tool that is utilized to communicate a reasonable assurance of safety to consumers. The majority of city water utilities have proven track records of providing good, safe water. Research has shown that trust in the institution or messenger is often more important than the message itself. As a result, trust in water utilities tends to be quite high in most cities served by municipal water.

The opposite can be true in situations where a water utility has failed to provide safe water such as Walkerton, Canada, where trust in the water utility was easily lost and even decades later has not fully been restored after an E.coli contamination event (Driedger *et al.*, 2014). A more recent example of this is the lead crisis in Flint, MI where officials ignored strong guidance and warnings of water utility staff of the negative consequences that would result from switching to a more corrosive water source (Masten *et al.*, 2016). A quick search will highlight countless news articles mentioning the lack of trust and even criminal charges that have resulted from the handling of the crisis. Once the problem was finally acknowledge there was swift response at both the state and federal level, and post hoc analysis of health implications and what went wrong continues and likely will for some time (Flint Water Advisory Task Force, 2016; Masten *et al.*, 2016; Ezell *et al.*, 2023). Even when there is a lack of trust, there is a structure in place to respond and be held accountable when situations do arise in municipal water systems. A uniform structure with the ability to respond to well water quality concerns is not as easily applied to private well users even when those concerns may be local.

There are cultural differences between municipal water users and private well water users that make it important to have different strategies for communicating risks associated with drinking water safety to rural private well owners. As opposed to municipal water users, private well users are low-grid, low group. Characteristics of this group include less formalized networks, fewer interactions among the population, open boundaries, fewer shared activities, horizontal accountability, less specialization and more egalitarian allocation of resources (Rayner, 1992). Within rural landowners there is little in the way of social stratification and group control over individual behavior with respect to well water, therefore regulations are likely to be less effective.

Private well owners are in a sense their own water utility manager. While there are regulations that apply to the construction of a well, those regulations do not address testing behavior or corrective measures if water is contaminated. Drinking water standards are not actually enforceable for this population of interest but are often used to provide guidance to those that have questions about the safety of their water supply. Research shows that the average private well owners ability to interpret whether test results are better or worse than drinking water standards is actually quite limited (Johnson, 2008).

This means that private well owners have less heuristic tools available when making determinations about the safety of their water supply. There are additional measures or steps in the risk evaluation process for private well owners that those on city water do not have to perform. Private well owners are responsible for their own fact-finding; and must also sort through information and attempt to place risk into context. To ensure risks are communicated effectively to make informed decisions it is important to understand how messaging to this audience can be improved.

Individuals perceive risk differently based on preexisting values and beliefs (Slovic, 1992). Uncertainty of a particular risk and the amount of dread associated with risk will also play an important role in placing risk into context. Greater uncertainty and higher associations of dread have been shown to increase individual perception of risk (Slovic, 1992).

Even though a particular hazard may not be particularly dreadful, the public may attribute a higher degree of dread because effects are often delayed and not immediately observable. This is likely to be the case with drinking water quality health outcomes where standards are set based on long-term exposure to a particular contaminant in drinking water. Using cancer as an illustration, the cancer would develop as a result of years or decades to exposure, the risk increases depending on the length of exposure, amount of contaminant, the genetics of an individual, even the amount of water that an individual consumes (National Research Council, 2001). All these factors together result in a high degree of uncertainty that can make the interpretation of risk by well owners frustrating and the communication of risk by public health professionals challenging.

Risks related to drinking water have varying degrees of dread associated with them. A salient illustration is provided by a study investigated risk ratings of different contaminants (Johnson, 2008). The data showed that at identical standards and contamination levels, contaminants that historically have received attention as causing harm such as mercury, arsenic and lead were perceived as having greater risk than less familiar contaminants.

In the absence of additional information, people will often make their decisions based on existing attitudes and beliefs using heuristics. If risk is perceived to be sufficiently great, it is up to the individual homeowner to decide what additional measures or self-efficacy strategies they would like to attempt. Whether or not an individual chooses to adjust their behavior to reduce risk will depend on 1) whether the value they have assigned to the benefits outweighs the risk, and 2) whether there are effective and achievable strategies available to them.

5.1.3 Risk Communication Theories and Frameworks

The goal of health professionals that work with private well owners is to encourage those with contaminated water to undertake appropriate risk reducing behavior. To facilitate that behavior however, it is necessary to try and understand the various attitudes and beliefs that affect the decision-making process. Two existing frameworks that have utility for interpreting current attitudes and beliefs are the extended parallel processing model (EPPM) and the theory of reasoned action.

5.1.4 Extended parallel processing model

Fear is an inevitable component of messaging regarding environmental health risks. The EPPM is particularly effective for understanding how people respond to fear messaging. When intentionally used as a risk communication technique, the EPPM is useful for understanding when fear appeals are likely to be effective; or alternatively when fear appeals can be potentially damaging to the intended goal or objective (Witte, 1995).

The EPPM suggests that when an individual is confronted with risk, they will be motivated to either control the danger or control their fear. The components of messages or information processing include external stimuli of self-efficacy, response efficacy, susceptibility and severity. This model evaluates two appraisals made by an individual during the processing of risk information. The first is the appraisal of risk (is this something that I should be concerned about) and the second is the appraisal of self-efficacy (is it in my capacity to protect myself from this risk).

If an individual perceives no threat or feels the threat is sufficiently low; there is not necessarily a reason for that individual to respond with additional actions or a different behavior. Alternatively, when an individual perceives a threat, understands appropriate actions, and has the capacity to perform risk reducing behavior – the person is likely to accept messages and engage in risk reducing behavior. However, if an individual perceives a threat and either the fear of the message is too great, or the person does not understand which actions are appropriate, or is not capable of carrying out appropriate recommendations – they are likely to rely on defense mechanisms to control that fear (Witte, 1995). Defense mechanisms in such situations include rejecting the message outright or potentially becoming an active opponent to desired behavior.

5.1.5 Theory of Reasoned Action

Another framework useful for understanding motivations of individuals is the theory of reasoned action. This theory posits that a person will undertake voluntary action as a result of existing attitudes and subjective norms (Ajzen and Fishbein, 1980). Attitudes as they relate to risk would be formulated from an individual's perception of perceived consequences/risks and the evaluation of outcomes that could arise under different scenarios. The second factor, subjective norms, relate to an individual's motivation to comply based on social pressures from others in their social network or sphere of influence. If information lines up with current attitudes and/or subjective norms; it is more likely that an individual will engage in certain behavior based on these two factors.

5.1.6 Improving Risk Communication to Private Well Audiences

A goal of risk communication is to encourage some sort of preventative action or steps that will lead to reduced risk. In the case of private well water this can involve a series of steps that are often quite complicated and are not navigated using heuristics. It is has been shown to be difficult for private well owners to find credible information on well water testing and interpretive information (Malecki *et al.*, 2017; Malecki *et al.*, 2022). This can lead to confusion, the accessing and spreading of misinformation and overall low feelings of self-efficacy when it comes to making decisions regarding one's own drinking water. Given the current trends in state agency programming, expanded government services may be unlikely and may not even be effective if implemented without consideration of the population being served. It is important for health and other agencies serving this audience to evaluate existing methods and information delivery systems with a goal of improving adoption rates of best management practices.

Strategies to improve communication efforts on this topic would therefore benefit from applying risk communication theories and frameworks to understand how current information could be more effectively targeted to have a greater influence on attitudes and behaviors towards private well water. Here I provide an overview of two independently conducted studies which sought to 1) better understand the attitudes and beliefs of homeowners regarding their well water quality and 2) identify potential barriers to performing certain risk-reducing behaviors such as well water testing.

5.2 Methods

In investigating this topic, I utilized results from two fairly extensive surveys of private well water users in Wisconsin. These include a 2015 statewide survey conducted by Survey of the Health of Wisconsin (SHOW) and a 2015 survey of Extension well water programming efforts in Fond du Lac County, WI conducted by the University of Wisconsin – River Falls. Survey results were obtained from publicly available reports summarizing the data.

5.2.1 Survey of the Health of Wisconsin

The Survey of the Health of Wisconsin (SHOW) is a population health monitoring program that gathers information about the health of the state’s residents. Started in 2008 by the University of Wisconsin School of Medicine, the survey has over 4,500 participants. The main goal of the survey is to conduct annual health surveys of Wisconsin residents that allows for longitudinal follow-up surveys to be performed (Malecki *et al.*, 2022).

A survey of SHOW participants investigated private well water testing behaviors and barriers (Schultz, 2015; Malecki *et al.*, 2017). The mail-based survey followed up with 719 SHOW participants, or approximately one-third of housesholds participating in the program. One-third is also the estimate of Wisconsin residents that rely on rural well water as their primary water supply.

5.2.2 Fond du Lac Survey of Private Well Owners

This survey compared populations of individuals that previously sampled well water through organized testing events conducted by UW-Extension and Fond du Lac County (Well-Testers) to a random sample of individuals within the county (Trechter, 2015). The intent was to understand differences in behavior between the two groups in an effort to evaluate the effectiveness of county efforts to promote well water testing and education.

5.3 Results

The SHOW survey results indicated that 75% of respondents always used their well water for drinking. Fifty-one percent of households submitted a water sample in the past ten years with just under 60% having tested in the last 5 years. The most common reason for having water tested was simply wanting to know if the well water is safe to drink (34%) followed by real estate transaction (19%). A majority (63%) of respondents who had water tested indicated no problems were detected through testing.

Of those responding to the survey that did not have water tested in the last ten years, the most common reason given as to why was that they “Have been drinking the well water for years without problems” (66%), followed by “Our water is probably fine” (46%). Knowledge of “Not knowing how to have water tested” and “Not knowing what to test for” were the fourth (34%) and fifth (28%) most common reasons given, respectively.

As indicated by a previous question, knowledge may also be a potential barrier as to why an individual may not have water tested. An almost equal percentage of respondents felt they had adequate information to make decisions about testing versus those who did not. Not surprisingly, those who indicated they had adequate information were more than twice as likely to have had water tested in the past 10 years (73%) than those that did not have the information (33%).

When asked what would prompt future testing, “Change in taste, smell or appearance” resulted in the most common response (49%) followed by “Learning a neighbors well is contaminated” (14%). On the issue of convenience, preferred methods for encouraging future testing were to “Pick up a test kit at a local location and return the sample to a local location a few days later” (45%) and “Order a test kit on a website and return the sample by mail” (28%).

The survey also investigated population characteristics for understanding testing behavior and treatment preferences of water. Females were 1.3 times more likely to test than males. Younger individuals (21-40) and older (>60) were 2.1 and 1.5 times more likely to have water tested than those age 41-60. Those with incomes above $25,000 were 1.4 times more likely to have water tested than the alternative. Those with a high school education or less and Bachelor’s degree or higher were both 1.5 times more likely to have water tested than someone with only some college. Former smokers or those that have never smoked were 1.3 and 1.5 times more likely to have had water tested. Having children in the home made it slightly more likely (1.2) that water would be tested.

Knowledge, attitudes, and beliefs were also evaluated as predictors of testing behavior. Those who felt that they had the information to make decisions about testing well water were 5.8 times more likely to have had water tested. Similarly, those who felt that they had the information to manage the safety and quality of well water were 5.0 times more likely to test water. The next strongest predictor was "Knowing someone else that had water tested" which made it 4 times more likely an individual would have their own water tested. This was followed by "I feel better knowing what is in my water" (3.1) and "Homeowners are responsible for having their well water tested" (1.9).

People that held certain beliefs and attitudes were less likely to have had water tested. Believing that "Adverse health effects from drinking water tend to be overstated" showed an adjusted odds-ratio of 0.59. Those who believed they were "Happy with the appearance of their water" (0.91) were also less likely to have had water tested.

Knowing patterns of water testing is important because it is the first step in evaluating the safety of a private well. The Fond du Lac survey results indicate only slight differences when it came to opinions of water safety; 9% of well-testers felt that their well water was unsafe compared to 6% of those in the random sample population. Expectedly, well-testers were more than twice as likely to have tested within the past 5 years (67%) than those in the random sample (36%). When asked on thoughts related to future testing 35% of those in the random sample indicated that they had <u>not</u> thought about it or <u>don't</u> plan to test compared to only 10% who indicated those options in the well-tester group.

Another barrier that can prevent people from testing water is simply not knowing what to test for. Those in the well-tester group were more likely to agree on what the most important tests to consider are than those in the random sample. When asked to recall what was last tested for, well-testers were much less likely (28%) to indicate that they didn't know what was tested for than those that tested within the random sample (51%).

Motivations for testing differed extensively between the two sample populations. Well-testers were more likely than random sample participants to indicate that their motivation for sampling was because of a well testing program in the area (78% compared to 21%) and to verify that water is safe (49% compared to 29%). While "Verify water is safe" was the highest motivating factor for random sample participants, the second highest listed was "Property transfer" (25% compared to only 4% of well-testers).

When it comes to perception about water quality, the researchers observed virtually no difference between groups for those that participated in organized testing programs to those in the random sample that had water tested. Approximately 88% (random sample) versus 91% (well-testers) indicated that their water was fine. While most water tests will not indicate problems, these results are slightly higher than what might be expected considering that 15-25% of wells contain coliform bacteria, 9% contain nitrate-nitrogen greater than the drinking water standard, and smaller percentages of various other compounds such as arsenic, manganese, and lead are often identified through testing.

When asked why people had not had water tested the most common reason given was "they had been drinking the water without any problems" (41%). Second on the list was "Don't know how to test" (17%); this question obviously did not apply to well-testers. Both well-testers and the random sample were asked about motivation or factors that may prompt a water test.

While the most common reason was “Change in Taste, Smell or Appearance” for both the well-testers (64%) and random sample group (71%), there was significant differences in secondary choices. Second highest among the well-tester group was a “Well test program in the area” (55% compared to 24% among random sample group), while “Know other wells in town contaminated” was the second highest choice for the random sample group (48% compared to 29% for the well-tester group).

5.4 Discussion

The results from two independent surveys yielded similar findings and provide confidence regarding the results related to questions on the attitudes, behaviors, and beliefs of private well water users. Both surveys include important information related to risk communication; the application of two prominent theoretical frameworks related to cultural theory and risk communication may lead to improved strategies for reaching this audience.

Based on information in these studies it was determined that the majority of private well owners rely on their well as their primary drinking water supply. Just over half of respondents in each random sample of populations indicated testing their water in the past ten years. However, public health experts recommend annual testing for certain constituents; this suggests a deficiency between what is recommended with actual testing frequency. Finding ways to encourage the other half to test their water is an obvious first goal; secondly, there is a need to ensure those that have tested in the past ten years are continuing to test on more routine basis. Because the only way to know for sure whether a well contains health-related contaminants is to have the water tested, this is a logical first step in the decision-making process for assessing risk from drinking water.

Results from both studies indicate a low-perceived threat from individual well water. For those that tested it may be influenced by test results, for those that have not tested the concern is that the low perceived risk may be partially based on having used the water for years without having any issues. Despite low-perceived actual threat, there is evidence within the two surveys to suggest people are generally aware that there are risks associated with water as indicated by the number of people selecting the reason for testing to be "verify that water is safe". The non-testing group also indicated they would be motivated to test if they noticed a change in appearance, smell, or taste of water, or knew of other contaminated wells in the area. This indicates that among non-testers they understand that there may be risks, but their perception of that risk has not warranted additional actions at the current time.

Increasing the perceived threat may be one way to increase the number of people that test, however care must be taken to ensure that effects are not overstated, or it may have the opposite effect. The EPPM provides evidence that fear appeals if they are too strong may lead to defensive behavior to control the fear rather than risk-reducing behavior. This is particularly true when there is a lack of perceived self-efficacy measures to help people deal with the risks. Results from the SHOW survey may provide evidence that information on drinking water safety have resulted in defensive mechanisms and rejecting the message; those that believed "Adverse health effects from drinking water tend to be overstated" were 40% less likely to have had water tested. Ensuring that fear appeals are accompanied by self-efficacy steps (i.e., here is what to do next or this is how you can address the problem) has the potential to increase likelihood that messages will be accepted and acted upon.

There is strong evidence in both surveys that feelings of self-efficacy lead to increased testing rates. In the SHOW survey, those that felt they had the necessary information to manage their private well water system were more than 5 times as likely to have had water tested. In the Fond du Lac survey, only 10% of those who had tested previously through an organized testing program did not plan to test again compared to 35% in a random sampling. Both surveys clearly show that empowering people with information or processes to get water tested not only make it more likely that they will have their water tested, but will continue that behavior into the future.

While the survey shows areas where efforts have had success in encouraging testing of private wells, a key question that remains is how to increase testing among non-testers. Both surveys provided evidence that locally arranged testing opportunities or mailing samples to households may be strategies to consider for reaching this group. These two options address general self-efficacy and ultimately overcome some of the commonly identified barriers (i.e. “Not knowing how to test” or “What to test for”). With minimal support by local agencies or organizations, these barriers are easily addressed and should continue to be supported or expanded in areas where testing options may be limited.

Messaging should be investigated for its potential to increase testing participation; specifically, the use of gain frames or fear appeals when communicating about best behaviors for private wells. The percentage of people that do not plan on testing who indicated that water quality concerns have been overstated suggests a rejection of potential risks and may have other unintended consequences. As a result, gain frames may help reduce the number of people that choose to manage fear rather than danger. People who agreed with the statement that “I feel better knowing what is in my water” were three times more likely to have water tested; this provides evidence that this type of gain frame may be effective for increasing rates of testing.

The theory of reasoned action suggests that subjective norms or social networks can influence behavior. One of the more enlightening statistics from the SHOW survey was that

"Knowing someone else had water tested" made them 4 times more likely to have had their own water tested. This supports the idea that normative beliefs – what others close to you think and do – influence ones' own individual behavioral intent and actions regarding well water testing. Utilizing peer networks or encouraging testers to share information with other family or friends is worth exploring as a strategy for increasing adoption of well testing behavior.

5.5 Conclusion

A large percentage of Wisconsin's population relies on private wells as their primary water supply. It is important that households routinely test their water to evaluate its safety. While a significant percentage of private well owners participate in some of the appropriate risk reducing behaviors such as testing, it is important to understand why some people choose not to undertake measures to ensure the water they use for drinking is safe.

We applied the EPPM and Theory of Reasoned Action to two Wisconsin specific studies on private well owners to understand behaviors of this audience. While current efforts such as locally arranged well water testing events and outreach activities show demonstrated success, additional gaps or areas of improvement have also been identified. Investigating the use of gain frames rather than fear appeals; and increasing feelings of self-efficacy have the potential to expand participation rates. In addition, finding ways to utilize those that have tested their well water to communicate with friends and family about how and why to test water could also help improve efforts as well. Lessons learned from these surveys will be useful for developing campaigns and outreach materials that meet the objective of increased testing rates of private well water users.

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Chapter 6

Conclusions

6.1 Synthesis

This research adds to the understanding of spatial and temporal nitrate leaching losses at both field and regional scales. In Chapter 2, I quantified nitrate loading from each crop during a typical rotation for the region, as well as the four-year flow-weighted meant nitrate-nitrogen concentration for the rotation. The results highlight the challenges of meeting drinking water standards in nitrogen intensive cropping systems with irrigation and coarse textured soils. Opportunities for water quality improvements in these systems should not only utilize best practices for in-field growing season management, but also investigate methods of managing the entire rotation for water quality. In Chapter 3, I explored the use of interplanting as an in-season management strategy in potato productions systems for increasing nitrogen uptake and minimizing leaching losses during the growing season. The results suggest optimism that interplanting can be performed without negatively impacting tuber yield, however, impacts on water quality were inconclusive. Additional experimentation is needed to better understand 1) whether nitrogen uptake by interplanted residue can be sequestered long enough to be credited to the following years crop, and 2) investigate ways to increase initial establishment of interplanted residue to increase nitrogen uptake. Chapter 4 utilized a network of private well owners to 1) develop models for predicting nitrate-nitrogen concentrations, and 2) detect trends in individual private wells as well as regionally. Findings revealed various agricultural landcover to be the most important predictor of elevated well water nitrate concentrations. While significant trends were detected at local and regional scales, the publicly available datasets used to investigate cause and effect for trends were inconclusive. This project highlights the value in having systems in place to track changes in well water quality through space and time. However, well water quality being a local resource, more detailed and accessible data on in-field management is needed to be able to answer the questions of what factors are contributing to changes in well water quality. Chapter 5 utilized two communication theories and the results of two independent surveys of private well owners to suggest improvements for communicating and interacting with rural landowners who rely on private wells for drinking water. Application of these theories suggest that using fear-based messages as a motivator may have the opposite impact of what is desired. Inclusion of self-efficacy strategies (i.e. what are the next steps) with initial testing messages and recruitment materials are thought to make people more likely to participate in desired risk reducing behavior. Lastly, the utilization of peer-to-peer communication strategies that take advantage of existing trust between individuals should also be used to improve outcomes.

6.2 Future Work

The work presented here seeks to provide a robust dataset of nitrate leaching in potato/vegetable production systems for which data is currently lacking. I hope to continue building on this long-term dataset through additional years of data collection that can be used to calibrate and validate models capable of simulating the diverse crops and changing climate of the region. In addition, I would like to continue exploring the use of interplanting as it relates to nitrate and water dynamics in potato systems. There are a variety of lessons that were learned during the experiment that were not able to be implemented. Partnering with other farms that only perform one hilling may allow interplanting to occur sooner. Earlier interplanting would hopefully result in improved establishment and greater nitrogen uptake. In addition, investigating changes to the rotation and rates of mineralization within these systems is critical to quantifying effectiveness of things like cover crops for improving water quality.

Most communities lack appropriate systems for tracking changes in well water and groundwater quality over time. The existing networks in Green and Sauk Counties provide a template for other communities. For this research topic, I hope to explore funding sources that would allow for the expansion of these networks to other counties or even statewide. Given attrition among participants during the five-year period, developing appropriate methods for replacing wells that maintain the statistical integrity of the network is also something that will be pursued.

Wisconsin has a robust network of water resource professionals and innovative farmers that make the state well suited to addressing the current and future groundwater nitrate challenges. Whether it is the amount of fertilizer that a farmer chooses to apply, the crops that someone chooses to plant, the field that is converted to managed grazing, the price that a consumer is willing (or able) to pay for their food, or the policies that local, state, and federal government officials choose to enact; each of these activities can have implications for the quality of our current and future drinking water quality. Because we are all in this together, it is important to maintain open lines of communication and good working relationships between stakeholders. If we are interested in moving the quality of water resources forward, then it is critical that we understand each person's respective point of view and utilize the best available science to make management and policy decisions. I plan to continue facilitating conversations and collaboration between groundwater stakeholders in hopes of identifying and implementing those strategies that maintain or enhance water quality while allowing farmers to have a high quality of life for themselves and their families.