MEASURING THE IMPACT OF NUTRIENT MANAGEMENT PLANNING ON SURFACE WATER QUALITY IN WISCONSIN

BY

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THESIS

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ABSTRACT

In 2009, Wisconsin implemented a new incentive program aimed at encouraging the adoption of nutrient management plans and improving water quality. This program, the Farmland Preservation Program, encourages this adoption via three tiers of tax credit incentives. While prior research has explored this program's impact on statewide water quality, this thesis pioneers the use of farm-level data to scrutinize short-term water quality impacts within a single county. This investigation addresses one key question: does nutrient management plan adoption enhance water quality? I answer this first descriptively by exploring patterns in water quality and nutrient management plan adoption across the state of Wisconsin. Second, I employ a two-stage least squares instrumental variable regression with fixed effects to identify nutrient management plans impact on water quality. The methodology reveals that a 10-percentage point increase in nutrient management plan adoption in a sub-watershed is associated with a 6.4% decrease in total ammonia concentrations. While no impact on total phosphorus levels was found, existing literature suggests that legacy effects would likely inhibit a reduction in total phosphorus levels in the short term.

ACKNOWLEDGMENTS

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CHAPTER 1: INTRODUCTION

Every summer, algal blooms proliferate across the United States, with a notable concentration in the Gulf of Mexico. These blooms inhibit recreational activities such as boating and fishing and can emit toxic fumes that endanger both animals and humans. As summer advances, the blooms decay, consuming all the dissolved oxygen (DO) and engendering a hypoxic zone, commonly referred to as a "dead zone." Dead zones expunge all oxygen from the water, rendering life unsustainable and ravaging aquatic ecosystems. Over the past five years (2017–2022), the Gulf of Mexico's dead zone has averaged the size of Connecticut (Rogener and Scheurer, 2022).

Algal blooms and hypoxia also suffocate economic activities across various industries, including recreation, housing, and fishing. Specifically, economic losses in the Gulf of Mexico alone are estimated to range from $666 million to $2.9 billion (in 2023 dollars) (Anair and Mahmassani, 2013). These blooms are not limited to the Gulf; they can occur wherever water is present, be it fresh or brackish, nationwide. In 2014, half a million residents lost access to clean drinking water due to a blue-green algae bloom in Lake Erie. Similarly, a “super bloom” in Florida’s Indian River Lagoon in 2011 decimated 60% of the seagrass crop, leading to nearly half a billion dollars in losses (NOAA, 2021). The cause of these algal blooms is well known; numerous studies have isolated nitrogen and phosphorus as limiting nutrients to these blooms (Duncan et al., 2019; EPA, 2022; Schindler et al., 2016).

Since 1951, nitrogen and phosphorus fertilizer and manure use have increased sharply. The largest relative increases were found in the "Corn Belt," which already had high fertilizer usage. This increase coincided with an overall decrease in row-cropped acres (Rossi et. al., 2023). The explosion of nutrients on row crops increased the likelihood of agricultural runoff in the Corn Belt, which feeds the largest river system in the Continental United States and drains into the Gulf of Mexico. Specifically, Paudel and Crago (2021) found an increase in fertilizer use coincides with an increase in nitrogen and phosphorus concentrations across watersheds. Additionally, Alexander et al. (2008) identified agriculture as responsible for roughly 70% of the nitrogen and phosphorus pollution in the Gulf. Thus, to reduce nutrient loading, action must be taken in agricultural production.

Nutrient management is key to reducing agricultural runoff. Wisconsin, whose water feeds into two of the nation's three largest river systems, implemented the Farmland Preservation Program (FPP), which encourages nutrient management plan (NMP) adoption. While much is known on the environmental benefits of no-till and cover crops, there is less empirical evidence on the impacts of nutrient management planning.

First, I review the data on nutrient management planning in Wisconsin and how it is correlated with water quality. Then, I use an instrumental variable (IV) approach to estimate the role of the FPP on encouraging NMP adoption. Finally, I estimate how NMP adoption affects local surface water quality within the same sub-watershed (HUC12). I use a unique dataset that captured NMP and FPP participation at the municipal level in Sauk County. The results of the IV two stage least squares (2SLS) regression indicate that for every 10-percentage increase in NMP adoption in a HUC12, total ammonia concentrations decreased by 6.4%, total phosphorus concentrations did not have a statistically significant change.

CHAPTER 2: NUTRIENT BACKGROUND

2.1 NUTRIENT POLLUTION AND WATER QUALITY

Anthropogenic nutrient pollution has become an increasingly important issue in the past few decades. This paper will focus on two main types of nutrient pollution: phosphorus and nitrogen. In their recent report, the Environmental Protection Agency (EPA) estimated that only 35% of rivers and streams have "good" levels of phosphorus. For a river to be classified as good for total phosphorus, it must have less than or equal to 55.9 micrograms per liter (EPA, 2020). In an ecosystem that is at equilibrium, phosphorus will naturally return to the soil or water via animal refuse, carcass decomposition, or erosion. This process tends to replace a similar amount of phosphorus that was extracted from the soil or water, which prevents large algal blooms. In contrast, phosphorus accumulation, or loading, has been occurring over the last several decades due to changes in agricultural practices, e.g., intensified row cropping, nutrient additions, and Concentrated Animal Feeding Operations (CAFOs), (Schindler et al., 2016).

Once excess phosphorus is present in a body of surface water, average algal bloom size increases, which can cause numerous harms. Harms suffered due to algal blooms can be broadly split into two groups: bloom and decomposition. In conditions where surface waters are nutrient-polluted, blooms can become so intense that they block sunlight from penetrating the surface and clog respiration organs and suffocate aquatic life. Certain types of algal blooms (blue-green algae or cyanobacterial blooms) release toxins into the air that can cause extreme symptoms such as vomiting, diarrhea, liver damage, and have been known to kill animals such as otters or dogs (CDC, 2022). Blooms also have severe economic costs, such as declines in property valuation (Wolf and Klaiber, 2017; Wolf et. al., 2022), limiting recreational activities (Anair and Mahmassani, 2013; Brooks et al., 2016; Roelke et al., 2016), and increasing municipal water purification costs (Ribaudo and Mosheim, 2017). Further, Allaire et al. (2019) estimated that areas which have water quality violations that pose an immediate health risk cause affected residents to purchase 14% more bottled water as an averting action, and Liu and Klaiber (2022) estimated that residents of Toledo, Ohio, spent upwards of $800 thousand dollars to avoid harmful algae in the drinking water over a three-day period.

Once an algal bloom has peaked, its decomposition process introduces a new set of ecological harms and economic costs. First, algae, like all organisms, requires oxygen to decompose. As algae dies, it sink to the water floor where it is then decomposed by bacteria; the bacteria consume DO to facilitate decomposition. This process can result in low-oxygen, or hypoxic, conditions where bottom-dwelling organisms, crustaceans, and certain types of fish can no longer survive (Rablais et. al., 2002). Hypoxic conditions cause massive aquatic life kills every year and fish migration. This can devastate coastal fishing communities and aquatic ecosystems in lakes (Anair and Mahmassani, 2013). Further, the "bloom and bust" cycle can permanently change the aquatic ecology. For instance, Rablais et al. (2002) found that hypoxic conditions in the Gulf of Mexico led to a decrease in biodiversity in the region. Additionally, Wang et al. (2007) found that concentrations of phosphorus or nitrogen explain 54% of the variances in fish assemblages in streams across Wisconsin.

Greenhouse gas emission and air pollution are a second harm that is realized as blooms start to decay. Eutrophic lakes emit methane, carbon dioxide, and nitrous oxide as algal blooms decay (Rossi et al., 2023). Beaulieu et al. (2019) estimate that emissions from eutrophic lakes will increase by 30-90% by the end of the century, which is equivalent to 18-33% of annual carbon dioxide emissions from burning fossil fuels.

Nitrogen is another key pollutant whose genesis is in agricultural runoff. The EPA estimates that only 18% of rivers and streams in the Upper Midwest have “good” levels of nitrogen (EPA, 2020). To be classified in good condition in the Upper Midwest, a river or stream must have less than or equal to 583 micrograms of total nitrogen per liter (EPA, 2020). Nitrogen can enter surface waters through direct runoff during significant rain events or through leaching into groundwater aquifers (Water Science School, 2018). In the United States, agriculture is the largest source of nitrogen pollution, accounting for more than 50% (Water Science School, 2018; Ribaudo et al., 2011). The main sources of this pollution are the over-application of nitrogen fertilizers such as urea or manure or via heavy agriculture use such as CAFOs (Meyer & Raff, 2019).

Nitrogen nutrient pollution has many of the same harms as phosphorus loading does. When nitrogen is the limiting factor for algal blooms, its loading can lead to higher intensity blooms that harm or destroy aquatic ecosystems (Water Science School, 2018; Carpenter et al., 1998). Brooks et al. (2016) detail the effects of cyanobacterial blooms, or harmful algal blooms (HABs), that have resulted in toxic contamination of aquatic species and devastating fish kills in freshwater lakes, leaving the lake void of life. While eutrophication is one way that excess nitrogen affects aquatic ecology, nitrogen has also been found to cause skeletal deformities in fish and amphibians and cause other reproductive harms. This has led to a reduction in aquatic species diversity in waters that are nitrogen-loaded (Camargo and Alonso, 2006; Anair and Mahmassani, 2013). Additionally, nitrogen has pernicious effects that are not limited to local ecosystems. During the denitrification and volatilization stages of the nitrogen cycle, nitrous oxide is released. [1] Nitrous oxide is a greenhouse gas that is approximately 300 times stronger

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[1] See appendix A for more information on the nitrogen cycle.

than carbon dioxide and stays in the environment for approximately 116 years. Further, emissions of nitrous oxide have been increasing over the past five decades in the U.S., and agriculture is the main emitter (52%) and the main source of growth in emissions (up to 87%) (Tian et al., 2020).

Another key externality of nitrogen loading happens when it leaches into groundwater. After nitrification occurs, nitrogen is chemically transformed into nitrate. While nitrates are an essential food for plants, excess nitrates leach into groundwater aquifers, where it will either be pulled out of the ground from a well or slowly flow into surface waters (Johnson et al., 2005). Nitrate-loaded water (> 10 mg/L), when ingested, is known to cause humans many ailments (Rossi et al., 2023). For instance, when unfiltered or inadequately filtered water is ingested, nitrate-polluted waters can cause methemoglobinemia, particularly in young children (blue baby syndrome). Nitrate-loaded water is also thought to have a role in causing digestive cancers, birth defects, bladder or ovarian cancers, thyroid hypertrophy, and many other ailments (Camargo and Alonso, 2006; Arbuckle et al., 1988; Majumdar, 2003; Ribaudo and Mosheim, 2017).

Nitrate-polluted waters also increase costs for municipal water providers, as nitrates must be reduced to safe drinking levels. In a technical report on removing nitrates from drinking water, Jensen et al. (2012) found that utilizing Ion Exchange (the most efficient method) to reduce nitrates to safe levels cost $666 per person (2012 dollars). Mosheim and Ribaudo (2017) found that the marginal costs for reducing 1 mg/L of nitrogen in raw water range from $125 to $919, depending on the size of the community water system. In a study on reducing nitrates in private wells, Keeler and Polasky (2014) found that it would cost between $2,600 and $6,710 per well over a 20-year time horizon to reduce nitrate to a safe drinking level.

2.2 NUTRIENT MANAGEMENT

To maximize nutrient uptake while minimizing nutrient losses farmers and conservationists alike start out by assessing the land and considering the source, method, rate, time of nutrient application. These practices combined make up what the USDA defines as SMART nutrient stewardship. First, farmers should understand what nutrients are already available in the soil and what nutrient types are best suited to ensure optimal yield.[2] Second, nutrients should be applied in the right place, using the right method. Some sites may need nutrients incorporated into the soil, so nutrients are injected, while broadcasting can satisfy the nutrient needs of other sites. Third, farmers should assess the site-specific conditions of each plot of land to ensure nutrients are only added when needed. Fourth, nutrients should be applied at the right rate, which may vary depending on the soil characteristics. Farms that practice no-till and cover crops may have reduced nutrient needs. Finally, nutrients should be applied when they are demanded by the crop and when soil and weather conditions permit. Spreading nutrients right before a large precipitation event or on saturated fields can lead to increased runoff potential. Implementing SMART nutrient management practices enables farmers to save on average $30 per acre on input costs, maximize their crop yields and protect surface water by minimizing runoff potential (USDA, 2023).

2.3 NUTRIENT MANAGEMENT PLANS

In Wisconsin, NRCS Standard 590 incorporates SMART nutrient management practices into NMPs. First, farmers must start by conducting one soil test for every five acres at least once

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[2] See appendix E for more information on nutrient source.

every four years. These tests are then submitted to a certified laboratory for analysis of soil acidity, phosphorus, potassium, and organic matter. This assessment ensures farmers can select the right nutrient source and application rate. Second, NMPs must document the yield goal for that year. Nutrient application rates cannot exceed the total nutrient recommendation for the cycle, with some exceptions. Third, manure applications are prohibited from running off the field immediately or during applications, ensuring the correct application method is used. Further, nutrients are strictly prohibited from being applied to surface waters, non-removed crops, fields with a high potential for erosion, or within 50 feet of groundwater conduits, wells, etc. Additionally, timing must be considered to ensure nutrients are not applied to saturated soils, snowmelt areas, or non-harvested vegetative buffers (Wisconsin DATCP, 2015).

Standard 590 also provides specific guidance for minimizing the potential of nutrient leaching by restricting high-permeable soils or soils with high bedrock. It recommends practices such as split application or timing nitrogen application with crop growth needs. Additionally, it suggests sources such as slow-release fertilizers or nitrification inhibitors, using cover crops, and prescribes specific temperature guidelines for nutrient applications. Furthermore, it requires vegetative cover in all areas of concentrated flow. Finally, Standard 590 highlights specific practices to protect air quality, such as installing edge-of-field windbreaks or using conservation tillage practices (Wisconsin DATCP, 2015).

2.4 WHEN NMPS ARE REQUIRED IN WISCONSIN

Chapter NR 151 requires NMP implementation for several reasons.[3] First, NMPs are required if a farm is participating in the FPP (Wisconsin DATCP, 2023). Second, if a Wisconsin county offers cost-sharing to develop a plan a farm must either accept and implement an NMP or decline and implement an NMP. Third, a farm may be governed under a local ordinance that requires them to implement an NMP. The second and third reasons are closely linked together as counties and municipalities tend to pass a city ordinance then offer cost-share to farms in violation. Further, cost-share is primarily used as a tool to target the highest risk areas of nutrient runoff e.g., manure storage facilities. Finally, the farm may be required to implement a NMP due to federal regulations or if they are participating in a federal program. For instance, under federal and state regulations, all CAFOs require NMP implementation, which is governed under the Clean Water Act (CWA) and Chapter NR 243. Specifically, NR 243 requires that all CAFOs have a Wisconsin Pollution Discharge Elimination System (WPDES) permit, which in turn requires an NMP.

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[3] Chapter NR 151 governs runoff management and mandates certain practices to be implemented. First, all land that grows crops must meet tolerable soil loss conditions. These conditions vary based on the type of land, such as pasture and row crop. Second, tillage is prohibited within 5 feet of surface water. Next, it limits significant discharge of wastewater into surface waters. Fourth, livestock access to open bodies of water is limited to preserve vegetative cover. Fifth, all newly constructed or reconstructed manure storage facilities must be built to code. Further, NR 151 requires all farms that apply nutrients or soil amendments to implement an NMP that adheres to Natural Resources Conservation Service (NRCS) 590 standards (Wisconsin State Legislature, 2023). However, NRCS 590 only applies under certain circumstances such as if cost-sharing is offered or participation in the FPP. Therefore, most farmland is currently exempt from implementing an NMP. (DATCP, 2023). Finally, special restrictions are in place in Eastern Wisconsin, distinguished by its Silurian bedrock and rapid groundwater recharging. The presence of this bedrock has caused many "brown water" events (Skidmore et al., 2023). Therefore, there are a series of strict restrictions on nutrient spreading depending on the depth of the bedrock, with spreading prohibited on bedrock depths less than or equal to two feet (Wisconsin State Legislature, 2023).

CHAPTER 3: FARMLAND PRESERVATION PROGRAM

3.1 BACKGROUND

In 2009, Wisconsin implemented the FPP with the intent to preserve farmland and encourage soil and water conservation standards. To preserve farmland, Wisconsin created two new land development categories: Farmland Preservation (FP) zoning and Agriculture Enterprise Agreements (AEAs). FP zoning is a special zoning category that is designated at the municipal level. AEAs are a tool in which private farmers can band together and apply to Wisconsin's Department of Agriculture, Trade and Consumer Protection (DATCP) for at least a 15-year agreement to limit land use to agriculture. To encourage soil and water conservation standards, they nested incentives in the land development categories to encourage NMP adoption.

The FPPs incentive mechanisms are designed to entice farmers to implement soil and water conservation standards. There are three levels of tax credits: $5, $7.50, and $10 per acre. To claim these tax credits, a farmer must meet two conditions. First, the farmer must be in either a FP Zone, AEA, or both. If a farm is in an AEA, they qualify for a $5 per acre credit. If a farm is in a FP Zone, they qualify for a $7.5 per acre credit. If a farm is in both an FP Zone and an AEA, they qualify for a $10 per acre credit. Second, in all cases, to claim a tax credit, they must implement an NMP. (Wisconsin DATCP, 2023).

3.2 FARMLAND PRESERVATION LAND DEVELOPMENT CATEGORIES

There are two land development categories for the FPP, they are FP zoning districts and AEA. The first step to defining these categories is for a county to submit their FP plan to DATCP. Broadly speaking, an FP plan is non-binding and designates land that is agricultural in nature and is not anticipated to be developed for non-agricultural uses over the next 15 years.[4] Once a county has an approved FP plan, counties, towns, cities or villages, are eligible create FP zoning districts or farmers may band together to form an AEA (Wisconsin State Legislature, 2023).

For a municipality to create a FP zoning district, they must first implement general zoning within the municipality. [5] Once the municipality has general zoning, they may further restrict land use and development by creating a FP zoning district.[6] FP zoning districts create a legal restriction on land use whereas FP plans do not. Farms that are located within a FP zoning district are eligible to claim a $7.50 per acre tax credit. Unlike FP zoning districts, AEAs are established by a group of farmers coming together to voluntarily restrict their land's use. To create an AEA there must be at least five separate farms that meet minimum size and income requirements. They must then apply to DATCP and once approved, they enter into an agreement to restrict their land to agricultural use for at least 15 years[7]. Farms which are located within an

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[4] For farms to be eligible for FP planning, the land needs to include a key agricultural resource or be critical for agricultural infrastructure. It also must be in an area that the county hopes to maintain or develop the land for agricultural use. If all the requirements are met, an FP plan may be submitted to DATCP for approval. FP plans may be certified for up to ten years at a time without requiring recertification (Wisconsin State Legislature, 2023). Additionally, in Wisconsin, FP plans are broadly determined at the county level and are non-binding. Nearly all land that meets the requirements is planned for FP.

[5] From this point on I while use the term municipality to refer to a broader set of authorities. Under DATCPs guidance a county, town, city, or village may be the zoning authority (Wisconsin DATCP, 2023).

[6] For land to be either meet or be more restrictive than provisions listed in Chapter 91 (Wisconsin State Legislature, 2023). Further, when zoning is implemented, state law requires that districts are at least 80% compliant with the FP plan. This means that large blocks of land are restricted to agricultural development, which is intended to prevent land use conflicts (Wisconsin DATCP, 2023). The zoning body must also define the jurisdictional, organizational, and enforcement provisions for proper administration. Once the FP zoning application is complete, it must be opened for public comment for a set period. After comments have been made, the application is submitted to DATCP for certification (Wisconsin State Legislature, 2023).

[7] To apply for an AEA, there must be at least five separate farm owners with contiguous land of at least 1,000 acres applying. Each farmer must also meet the minimum income threshold of $6,000 per year in gross farm revenues. Next, the farmers must fill out an extensive application with the rationale for the petition spelled out clearly. Once the application is completed, it is submitted to the DATCP for approval or denial. DATCP may approve up to two million acres of agricultural land in total for AEAs. If DATCP approves the area as an AEA, the parties enter into an agreement for at least 15 years. Like FP zoning, AEAs are enforceable by law. However, landowners may terminate an AEA by submitting a termination request in writing, gaining unanimous consent of all the farmers, and paying a conversion fee equal to three times the per-acre value of the land (Wisconsin State Legislature, 2023)..

AEA are eligible to claim a $5.00 per acre tax credit. Farms located within both AEA, and a FP zoning district are eligible for a $10.00 per acre tax credit.

CHAPTER 4: WISCONSIN DISCRIPTIVE ANALYSIS

4.1 DATA

To conduct my analysis, I use both publicly available data as well as private data. The public data I use includes water quality data from the EPAs water quality portal, Parameter-elevation Regressions on Independent Slopes Model (PRISM) data, United States Department of Agriculture's (USDAs) Cropland Data Layer (CDL), HUC12 and municipal boundary data from the Wisconsin Department of Natural Resources (DNR), I also use private data on FP planning, FP zoned districts, and AEA shapefiles from the Wisconsin DATCP. Lastly, I requested private data which covers size and location of CAFOs in Wisconsin as well as cross sectional geospatial data containing 2022 NMP and FPP participation for Sauk County.

4.1.1 Water Quality Data

The water quality data I use contains water quality observations in Wisconsin spanning the years 2010 – August 2023. There are 161,820 observations for total phosphorus and 45,533 observations for total ammonia concentrations. The data contains coordinates which identify where observations were taken. It also labels which type of samples were taken as well as the location type e.g., stream, lake, or groundwater. The observations list the total concentration of the measured water quality indicator in milligrams per liter and it has observations for nitrogen, phosphorus, and ammonia. Finally, the data contains the date and time it was sampled and what the detection limit is. Observations which were below the detection limit were given the detection limit as their value (EPA, 2023).

4.1.2 PRISM Data

PRISM data was pulled from Oregon State University's PRISM Climate Data portal. To pull the data I had to provide coordinates for specific requested locations. The locations which I requested were based on the centroids of each HUC12 in Sauk County. I then pulled annual data from 2021 – August of 2023 at the daily level. This data includes daily temperatures, precipitation, and elevation for the requested period (PRISM Climate Group, 2023).

4.1.3 USDA CDL

I pulled the USDA's CDL for the years 2010 – 2022 using the USDA's CropScape tool. This data is a geo-referenced raster which has crop-specific land cover data, where each pixel represents a 30x30 meter square. To classify the data, researchers physically sample each field and match the land use category to a pixel. I use this data to identify the percentage of a county or HUC12 that was cropped in each year. I am also able to calculate the total acreage in each land use category, such as corn or soy crops (USDA, 2023).

4.1.4 Wisconsin Data

Using the Wisconsin DNR data portal, I pulled publicly available data on municipal boundaries and HUC12s. The municipal boundary data is a shapefile which delineates boundaries for all counties in Wisconsin. Watershed data contains upstream downstream sub-watershed data as well as the sub-watershed code and name (Wisconsin DNR, 2023). I also requested and received Wisconsin DNR data on CAFO location. Then, I requested data on FP zoned districts, FP planning, AEAs, annual reports, and NMP annual survey from the Wisconsin DATCP. The first three items were shapefiles which included geospatial boundaries on all three areas, date of implementation and expiration. The annual report data was composed of DATCPs annual reports from 1995 – 2022, which detailed the progression of nutrient management planning in Wisconsin. The reports highlighted areas of success and areas which need improvement in nutrient planning. They also highlighted select counties NMP implementation. The NMP annual survey listed the reasons and acreage for NMP adoption by county and individual NMP. This survey contained data from 2019 – 2022.

Finally, I requested and obtained parcel level NMP and FPP participation data from Sauk County. The parcel level NMP shapefile contained geospatial data for all NMPs in Sauk County for the year 2022. The FPP participation shapefile contained geospatial data for all NMPs attributable to the program for 2022. Both shapefiles contained information on the size and date of implementation of the NMP.

4.2 DESCRIPTIVE METHODS

To effectively analyze the NMPs effect on water quality, I first investigate the state of agriculture and water quality in Wisconsin. To do this, I use the USDA’s CDL to conduct a land use analysis in Wisconsin from 2010 – 2022. Using the CDL and ArcGIS Pro 3.1.2, I was able to distinguish annual land uses across all of Wisconsin. Specifically, I calculated two metrics, first, I was able to count how many acres in Wisconsin were cropped each year. Second, I used the CDL to calculate the percent of land that was cropped in each year by each county and by each HUC12 in Sauk County. I used the second metric as a control variable in later regressions.

At the beginning of this period, the FPP introduced stronger incentives for farmers to adopt NMPs. Using DATCPs annual reports on nutrient management, I chart out NMP adoption since 2010. Additionally, the annual reports only list out aggregate NMP acres, which are not broken down by county. However, since 2019, DATCP has been collecting a NMP acreage survey which disaggregates NMP data by county and reason implemented. This survey disaggregates each individual NMP, how many acres are in that NMP and what reason is given for implementing the NMP. For the years 2019 and 2020, farmers could only choose one reason. From 2021 onwards, farmers can list multiple reasons for their NMP.

To gain a better understanding of NMP by county I leveraged the latest NMP acreage survey, to compare 2021 NMP acres to crop acreage in counties with over 10% cropped land. I limited the comparison to counties with more than 10% of their land cropped, to ensure I am comparing similar counties. Then I calculate each county's NMP adoption rate by taking their ratio of NMP acres to crop acres. This rate helps me understand what counties are most motivated to adopt NMPs and can highlight what areas may have higher levels of pollution.

Next, I leverage the NMP surveys to reveal the reasons which farmers are implementing NMPs. I start out by calculating the percent of NMP acres for each reason by each year of the survey. Since 2021, the NMP survey has allowed farmers to select multiple reasons for implementing an NMP. Therefore, I use the 2021 NMP survey to run a correlation matrix and analyze which motivations are correlated.

4.3 DESCRIPTIVE EMPIRCAL STRATEGY

Since the introduction of the FPP in 2010, NMP adoption has increased substantially. Given that NMP adoption has increased during this period, concentrations in phosphorus and ammonia in the surface water may be trending downward in absolute terms. Therefore, I use equation 1 to reveal any county level trends.

(1) $$WaterQuality_{c,y} = \alpha + \beta_1\eta_c * YearTrend + \delta X_{c,y} + \eta_c + \theta_q + \varepsilon_{c,q}$$

I run this simple regression with minimal controls since the FPPs inception, 2010 – 2022. The outcome of interest is $WaterQuality_{c,q}$ which is the concentration of total phosphorus or ammonia in milligrams per liter in surface water in county (*c)* in year *(y)*. The variable $\beta_1\eta_c * YearTrend$ is the county linear trend; $\delta X_{c,y}$ are controls for percentage of a county (*c)* in year *(y)* that is cropped and whether the water quality station was in a lake or a river; η_c is county fixed effects; and θ_q is quarter fixed effects.

Next, I conduct a straightforward regression using water quality data (expressed in milligrams per liter), isolated to the summer months—June, July, and August—of 2021. The exclusive focus on 2021 is due to its status as the most recent year with reliable NMP acreage data for each county. Then I run a second regression analyzing the relationship between the NMP adoption rate and water quality for this same sample.

4.4 DESCRIPTIVE RESULTS

The results of my CDL analysis are depicted in Figure 1. It shows that since the inception of the FPP there has been roughly a 13% increase in cropped acres, from 8.3 million to 9.4 million.

An increase in cropped acres without a similar increase in nutrient managed acres could lead to worse water quality overall. However, after aggregating and depicting NMP adoption in Figure 2, I find that NMP adoption has increased during this period too. Specifically, figure 2 shows that since 2010 there was a 226% increase in NMP adoption, from 1.5 million acres in 2010 to 3.4 million acres in 2017. Since 2017, NMP adoption has been relatively flat.

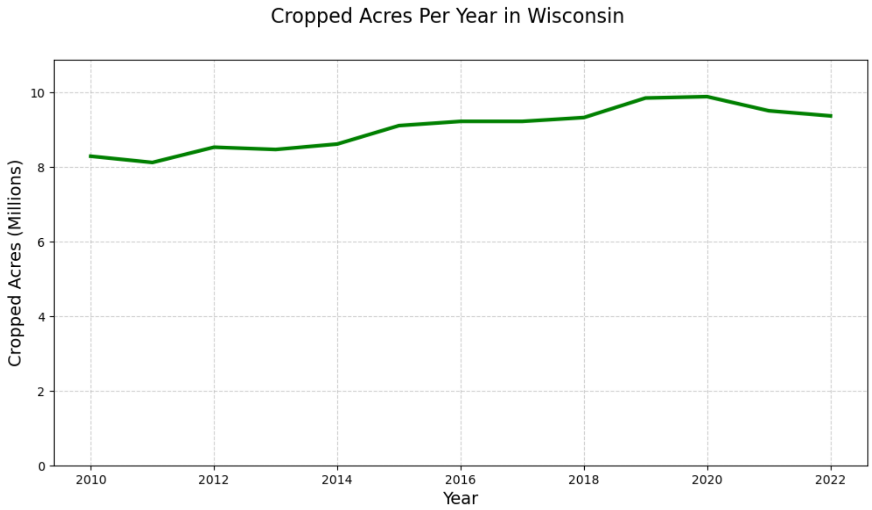

Figure 1 – Depicts millions of cropped acres by year.

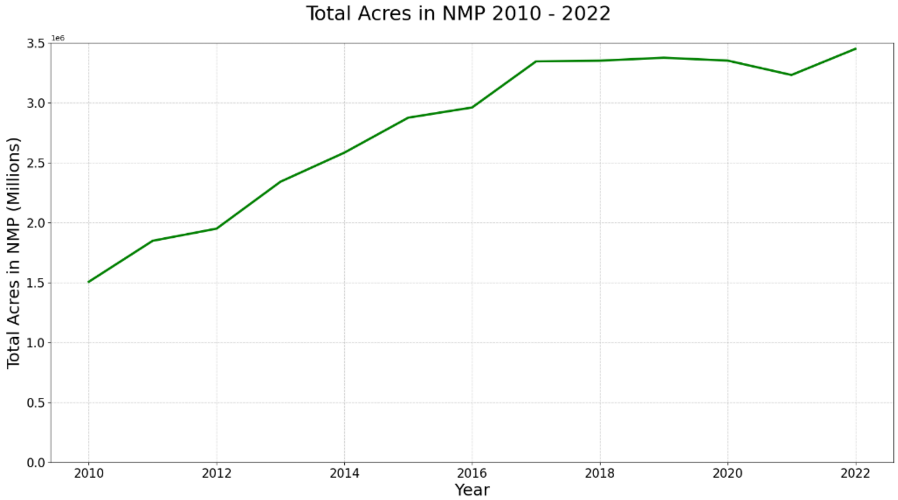

Figure 2- Depicts millions of NMP acres by year.

Pairing these two results together, I can reveal the NMP adoption rate by county. Figure 3 shows that eastern Wisconsin has the highest rates of adoption, which coincides with where Wisconsin's Silurian bedrock is.[8] Counties adoption rate for NMP span anywhere from 0% to 87% in 2021. Understanding where NMP adoption is the highest, presents a new question; why are farmers implementing NMP?

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[8] See appendix B, footnote 3 and NR 151.

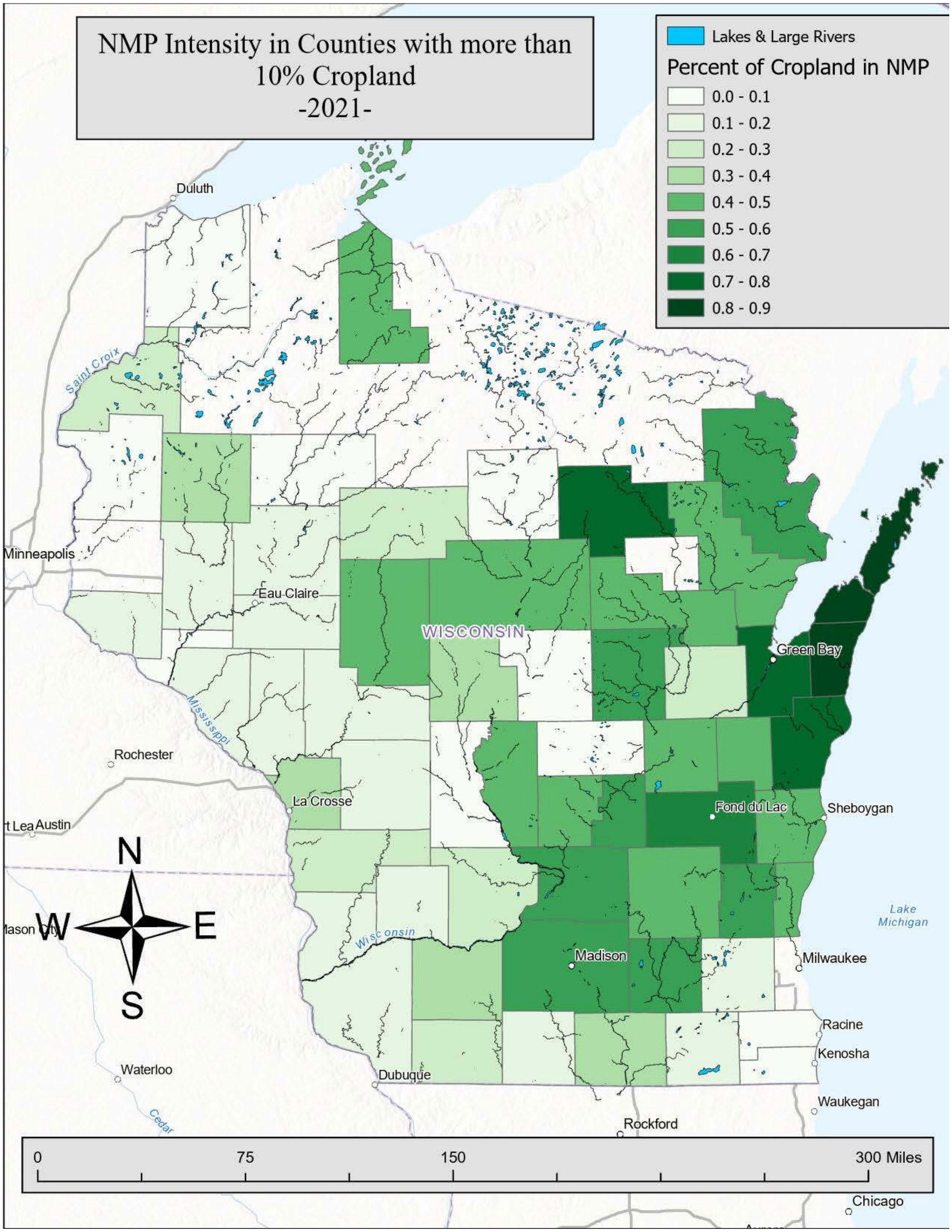

Figure 3 – NMP Intensity in Counties with more than 10% Cropland in 2021.

I found that the primary reasons farmers are adopting NMPs are FPP, NR 243 and County ordinances (Figure 4). First, it shows that an increasing number of farmers are participating in the FPP with the most recent year accounting for almost 60% of NMPs. The second most prevalent reason given is NR253, which addresses CAFO compliance pursuant to the CWA. The third major reason farms adopt NMP is due to a county ordinance which requires the implementation of abatement practices, such as manure storage ordinances; if the county offers cost-sharing to implement the required practices, the farm must comply and maintain compliance indefinitely. Additionally, farms may have multiple reasons for adopting an NMP.

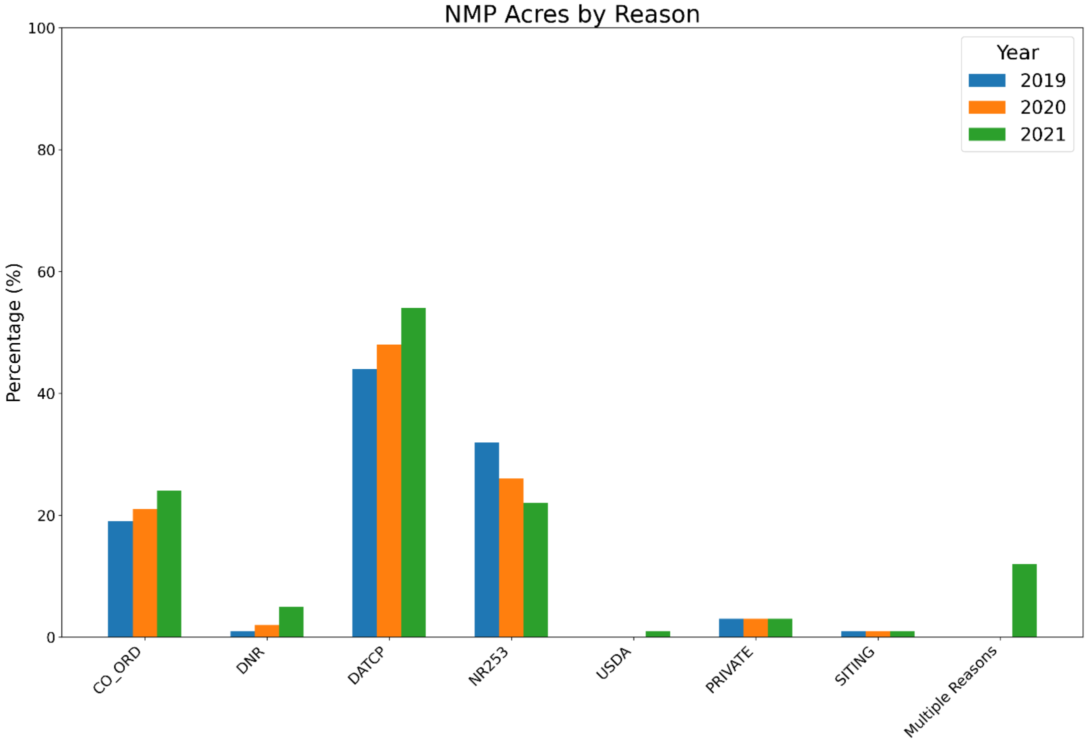

Figure 4 – Farms may be required to adopt NMP due to a county ordinance requiring the implementation of abatement practices, such as manure storage ordinances; if the county offers cost-sharing to implement the required practices, the farm must comply and maintain compliance indefinitely. Siting is also a form of county ordinance dealing with livestock. Second, the Wisconsin Department of Natural Resources (DNR) manages the state's Wisconsin Pollutant Discharge Elimination System (WPDES) program, which may necessitate NMPs if users discharge wastewater into Wisconsin's waters. Third, NMPs may be implemented due to the FPP, which DATCP oversees. Fourth, NR253 addresses CAFO compliance pursuant to the CWA. Fifth, participation in various USDA payments for ecosystem services programs, such as the Environmental Quality Incentives Program (EQIP), may necessitate NMP. Sixth, individual farms may voluntarily implement an NMP. SITING is a form of county ordinance which counties may implement to mandate NMPs for farms with livestock under the animal unit threshold in NR 253.

The three major reasons for implementing an NMP are all positively correlated with each other (Figure 5). Second, it highlights that farms subject to NR243 are most positively correlated with farms subject to county ordinances. Third, farms participating in the FPP are also most positively correlated with participation in a USDA program such as Environmental Quality Incentives Program (EQIP). Finally, outside of the major reasons for implementing an NMP, there are no significant correlations depicted in figure 5.

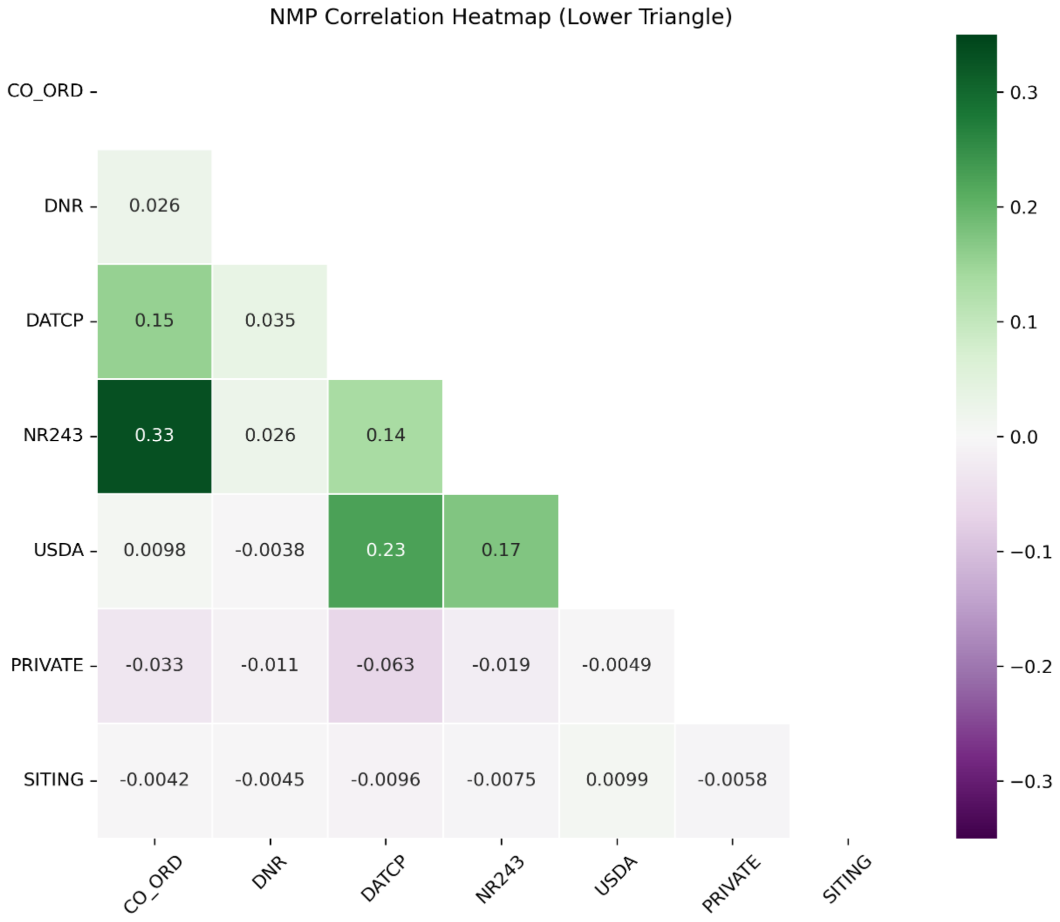

Figure 5 – depicts a correlation matrix for NMP acres for 2021.

4.4.1 Water Quality Since FPP

Results from regression analysis indicate heterogeneity across counties in Wisconsin during the time for total phosphorus. Figure 6 illustrates the total change in total phosphorus readings by county since the inception of the FPP. Some counties experienced an increase in total phosphorus readings up to 61% during this period, while other counties reduced their total phosphorus readings by as much as 21% percent. Figure 7 illustrates the overall shift in ammonia since the FPP. Changes in total ammonia were also heterogeneous during this period, with some counties experiencing an increase in total phosphorus readings of 105% while other counties experienced decreases of as much as 108% during the period.

Through this simple analysis on water quality since the start of the FPP, I was able to learn that measuring macro trends in water quality due to NMPs will be more complex due to the confounding relationship between NMPs and cropped acres. Additionally, changes in water quality in percentage terms may be more difficult to interpret, since ideal concentrations tend to be small and subject to external and seasonal shocks. Given these results, I run a narrower analysis while comparing counties NMP adoption rate to highlight county level relationships between water quality and NMP.

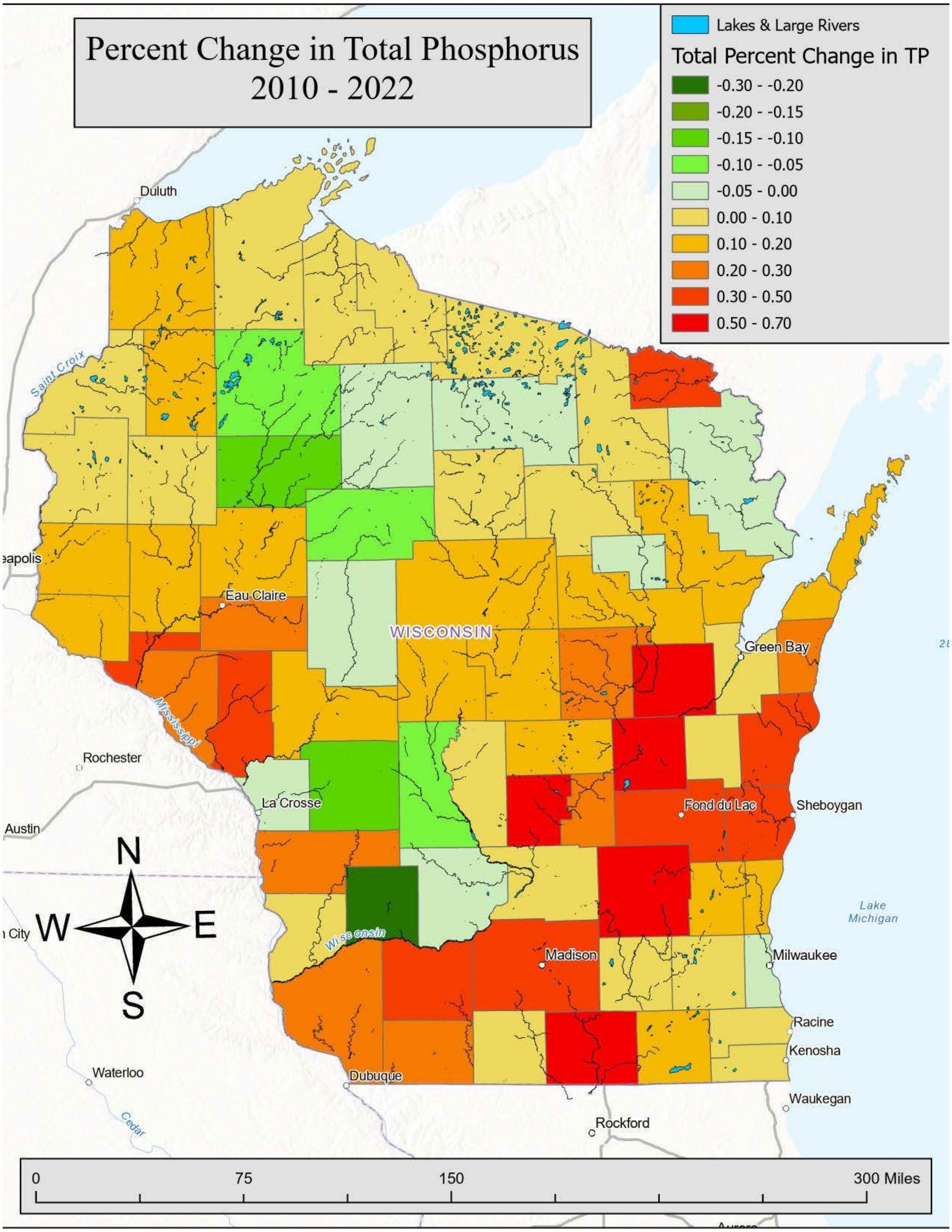

Figure 6 – Depicts total phosphorus change from 2010 – 2022 based on the β_1 variable in equation (1). Results are given in percentage changed during the entire time, for example, Sauk County reduced total phosphorus between 0 to 5% during this period.

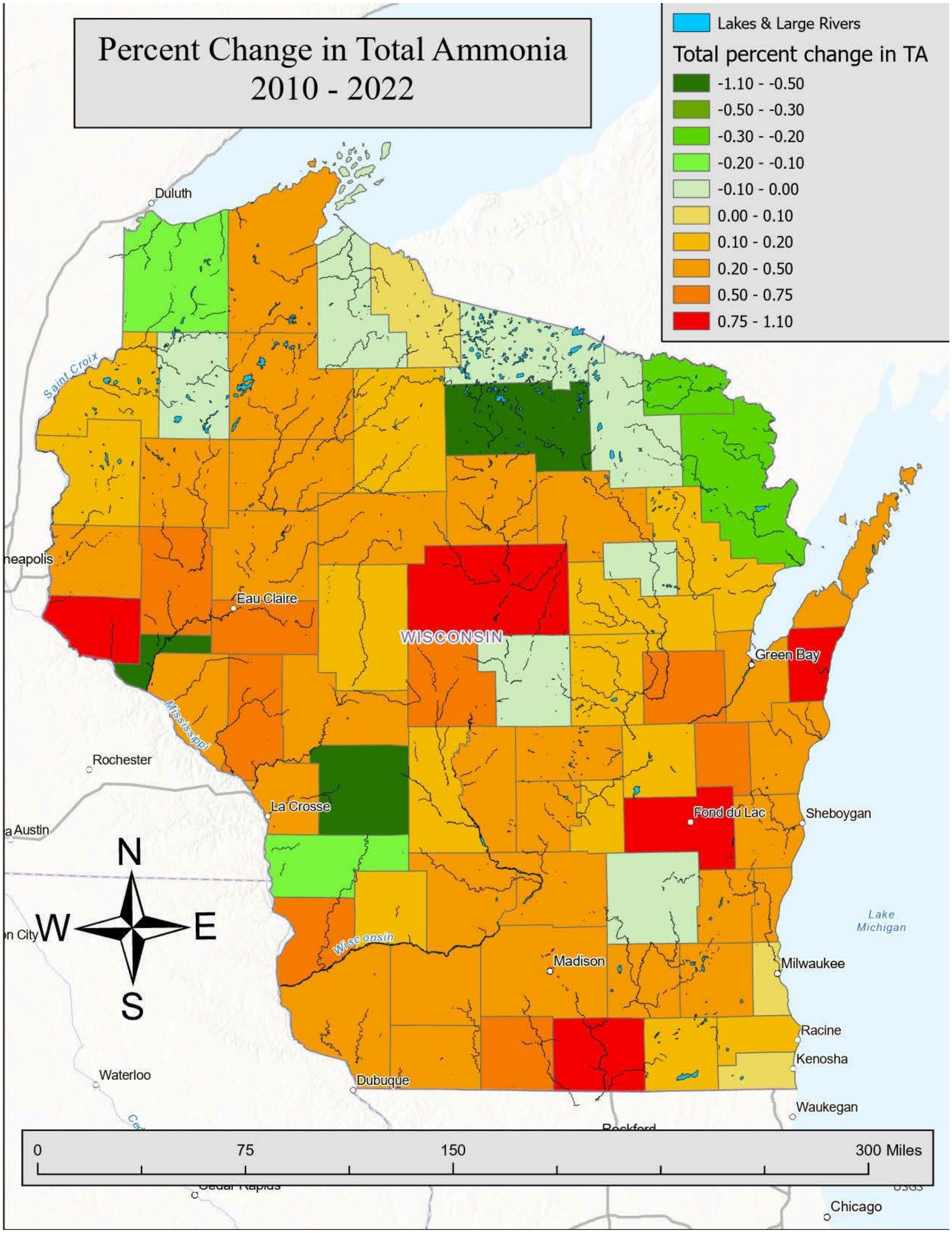

Figure 7 - Depicts total ammonia change from 2010 – 2022 based on the β_1 variable in equation (1). Results are given in percentage changed during the entire time.

4.4.2 Comparing Water Quality and NMP Adoption

Figure 8 visualizes the comparison between absolute water quality levels and NMP adoption rate. Notably, the highest adoption rates are concentrated in the eastern region of Wisconsin, aligning with the Silurian bedrock area. Moreover, roughly half of the counties in Wisconsin appear to meet at least one of the thresholds for total phosphorus in surface water. Certain counties, particularly those near Green Bay, exhibit exacerbated total phosphorus readings and are downstream from a major phosphorus source. Known for its algal blooms, Green Bay's poor water quality and elevated phosphorus levels may permeate through adjacent counties via the Fox River. Lastly, poorer readings (TP > 0.1 mg/L) are observable in the state's primary agricultural areas, with most of the counties manifesting yellow readings or worse.

A comparison of total ammonia levels reveals that most counties were beneath the ammonia limits. Figure 9 illustrates that most counties with observations remained under 0.06 mg/L of total ammonia in surface water. Only a handful of counties, such as Dane, exhibit higher total ammonia levels. However, the number of observations for total ammonia is substantially lower than those for total phosphorus, 881 versus 5,054, attributable to Wisconsin implementing a phosphorus standard but, has yet to establish a nitrogen standard (Wisconsin State Legislature, 2022).

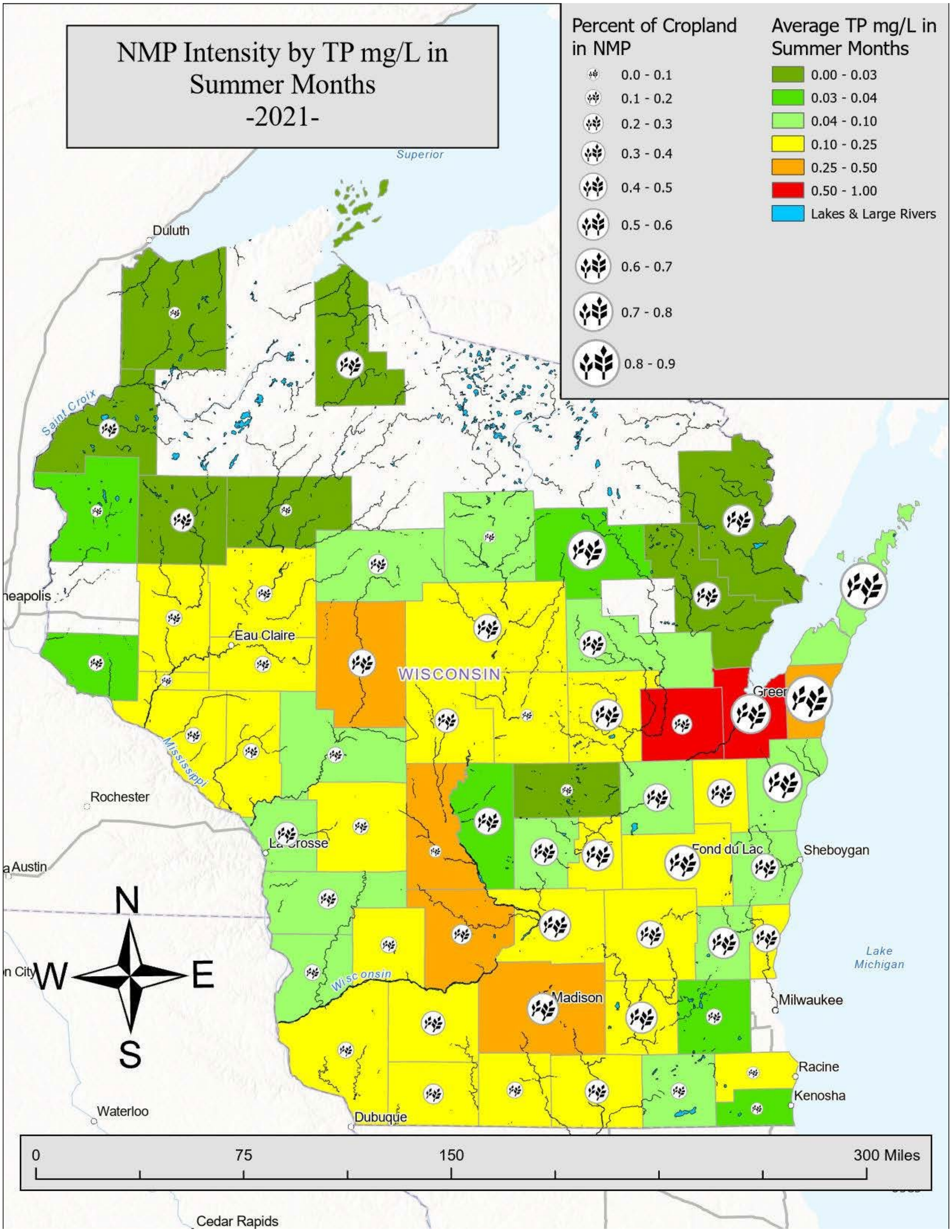

Figure 8 – Depicts average total phosphorus mg/L for June, July, and August in 2021. The ranges for total phosphorus are pulled from Wisconsin law Chapter NR 102. Stratified Lakes and reservoirs have a total phosphorus limit of 0.03 mg/L; non-stratified lakes and reservoirs have a limit of 0.04 mg/L; rivers and streams have a limit of 0.1 mg/L. The other limits are arbitrarily set. Of note, two-story fishery lakes have a limit of 0.015 mg/L, none of the counties met that average (Wisconsin State Legislature, 2022).

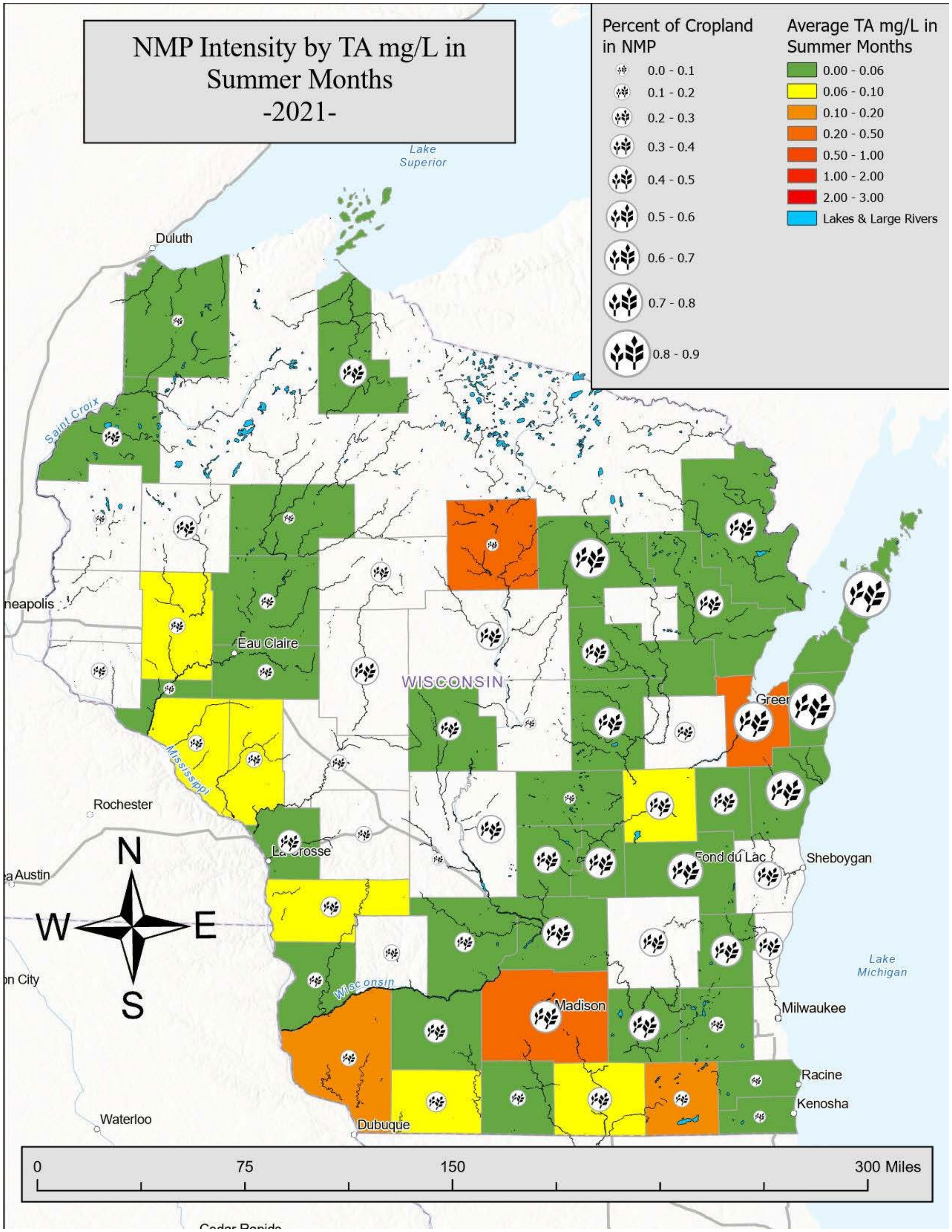

Figure 9 – Depicts average total ammonia mg/L for June, July, and August in 2021. The ranges for safe levels of ammonia in the surface water are pulled from the literature. Total ammonia greater than 0.06 mg/L can lead to gill damage in fish; levels greater than 0.1 mg/L indicate polluted waters; at levels greater than 0.2 mg/L sensitive fish species such as trout and salmon begin to die; The National Academy of Science recommends a drinking water standard of no more than 0.5 mg/L; at levels of total ammonia greater than 2.0 mg/L, ammonia-tolerant fish like carp begin to die (Anderson et. al., 2002; (Minnesota Department of Health, 2018).

Lastly, Table 1 illustrates that counties which adopt NMP at a higher rate correlate with higher concentrations of ammonia and phosphorus in the surface water. Specifically, a 10% increase in the NMP crop ratio is associated with a 0.1 mg/L increase in phosphorus concentrations or a 0.0387 mg/L increase in ammonia concentrations. This result likely captures the confounding relationship between NMP adoption water quality. That is, counties which have high adoption rates, also tend to have confounding factors which impair surface water quality such as Silurian bedrock or higher rates of cropped acres. Therefore, a deeper analysis on NMPs impact on water quality is warranted.

Table 1 – NMP Crop Ratio to Water Quality in Summer Months 2021

	Phosphorus	Ammonia
NMP Crop Ratio	1.000***	0.387***
	(0.0386)	(0.0807)
Sample Size	3951	682

Note - Standard errors in parentheses, * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

CHAPTER 5: CASE STUDY – SAUK COUNTY

5.1 DESCRIPTIVE ANALYSIS OF SAUK COUNTY

Given that a statewide database of NMP and FPP adoption does not exist, I use Sauk County as a case study to measure NMPs impact on water quality. Sauk County is an ideal candidate to analyze because it has a complete accounting of NMP use, both for FPP or otherwise (see figure 11, map 4). Sauk County also has a significant agriculture sector with an ideal mix of land use categories with roughly half of its land in FP zoning or AEAs. Table 3 provides the summary statistics for this sample and figure 10 depicts the water quality station locations. All HUC12s have varying levels of NMP, with the average HUC12 having 20% of its acres under NMP (table 1). However, the HUC12s range from having .6% to 58% of their acres in NMP. FP zoning in Sauk's HUC12s ranges more widely with as little as 0% in zoning to as much as 96% in zoning (see figure 11, map 2). The percentage of acres within an AEA ranges from 0 to 17% in each HUC12 (see figure 11, map 3). In terms of cropped acres, Sauks HUC12s ranged from 0% cropped to 44% of its area being cropped; these estimates changed year to year. Finally, the data also includes information on where water quality measurements were taken, either a lake or a stream/river. Out of the 581 water quality measurements, 441 were taken from a stream, and 140 were taken from a lake.

Table 2 - Summary Statistics

Variable	N	Mean	Std. Dev.	Min	Max
Phosphorus Sample	466	.2010489	.3201347	0	2.64
Ammonia Sample	114	.05344	.0704692	0	.58
Percent in NMP	581	.1720757	.1640185	.0066802	.5821059
Percent FP Zoned	581	.1932297	.3531183	0	.9569068
Percent in AEA	581	.02193297	.04815696	0	.1724051
Percent of HUC12 Cropped	581	.2201173	.1369381	0	.4441842
Precipitation in Inches	581	.2538898	.5427691	0	3.02
Average Temperature	581	57.05766	16.01567	3.2	81.1
CAFO Density	581	.1135972	.3282729	0	2
Sampled at Lake	140				
Sampled at River or Stream	441				

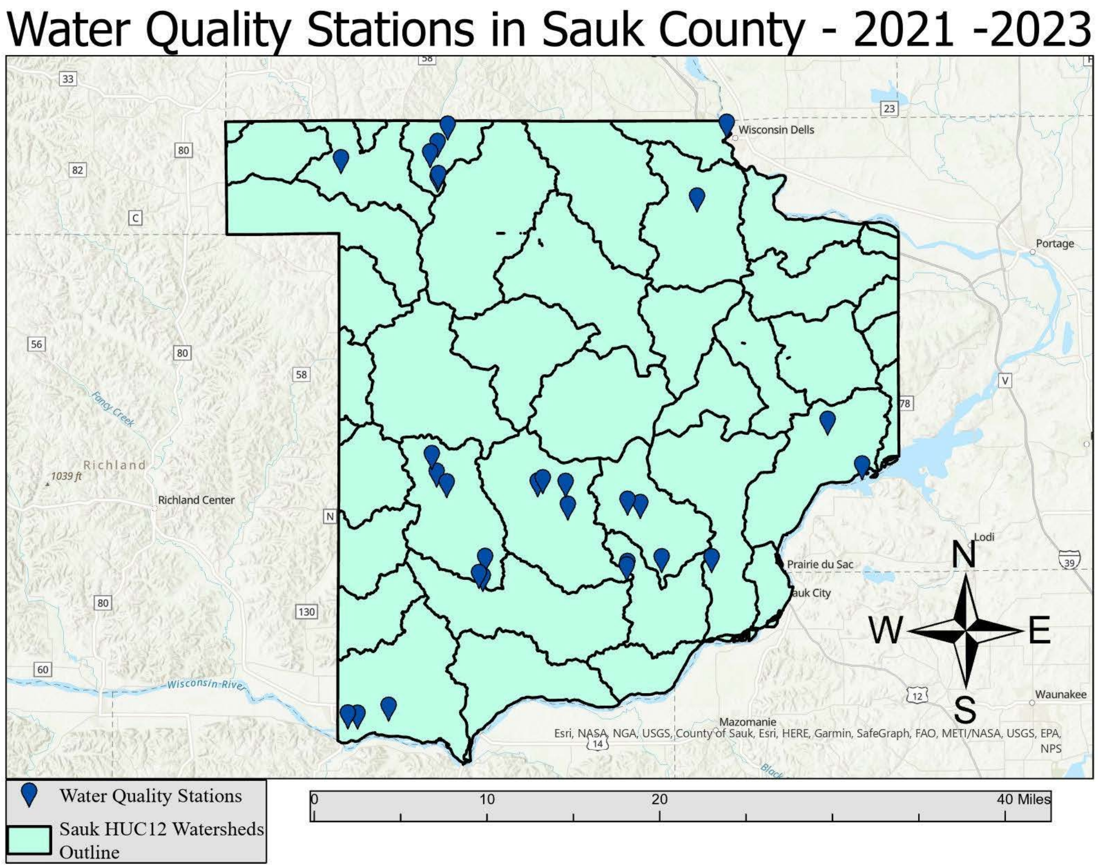

Figure 10 - depicts water quality stations in Sauk County and in which HUC12 they reside in. There are 581 observations across the 29 water quality stations depicted.

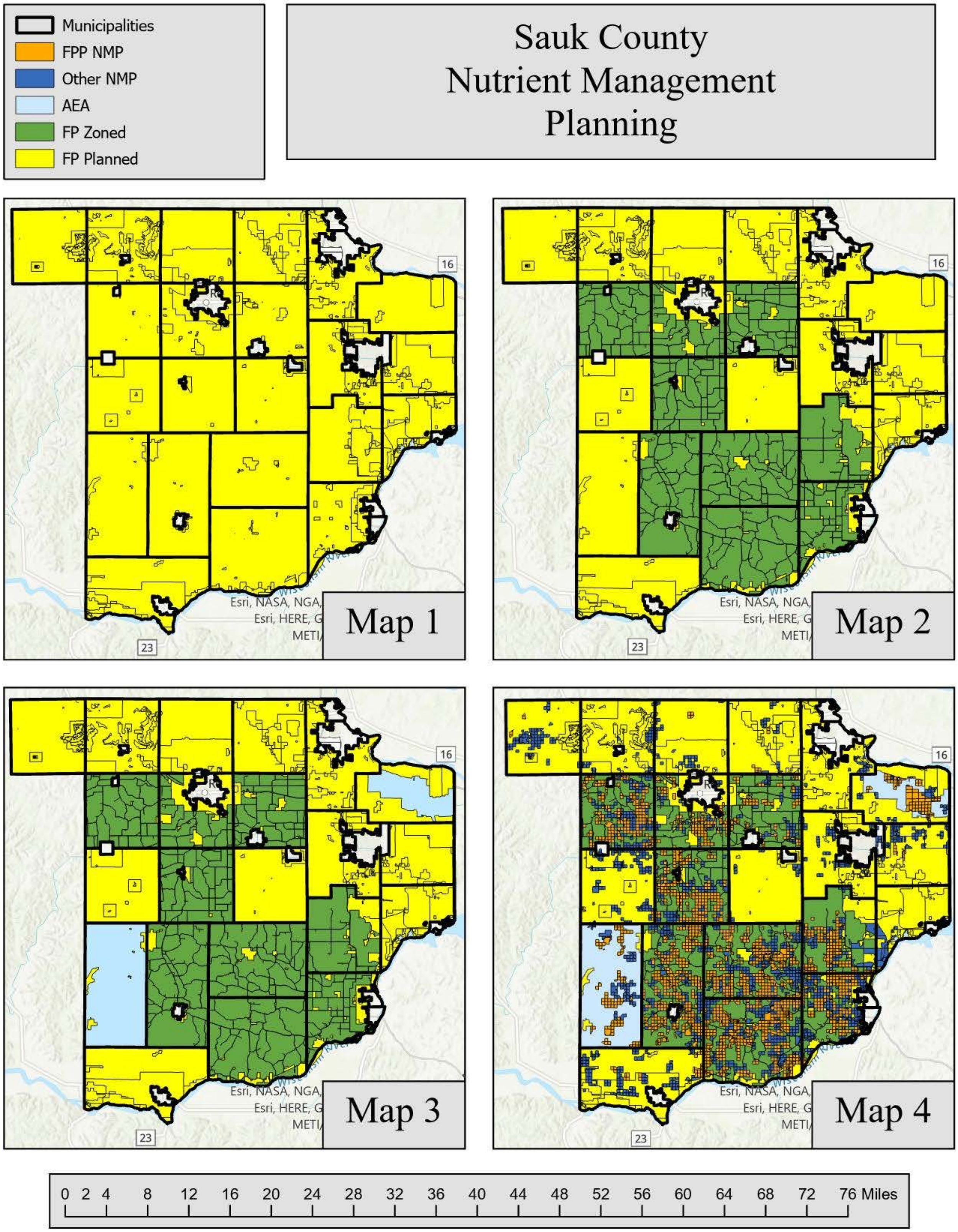

Figure 11 - Map 1 – 4 depicts the FP land use process. The FP plan is the first step, then FP Zoning and AEA designation may happen concurrently. Once FP zoning or an AEA is designated FPP NMPs can be established. Map 4 depicts Sauk Counties NMPs for 2022.

Table 2 also shows significant variance in precipitation throughout the year, which I anticipate a strong connection between it and nutrient concentrations. For instance, when a significant rainfall event occurs, it washes sediment bound nutrients such as phosphorus off the land and into surface water (Duncan, et al., 2019). Thus, intuitively, I would expect to see precipitation spikes correlate with spikes in phosphorus observations. Additionally, these rainfall events also increase the prospect of nitrates leaching into the groundwater. Therefore, rainfall may not correlate immediately to ammonia spikes. Rainfall events may also dilute the concentration of nutrients in the surface water; therefore, results may be heterogeneous. Figure 12 illustrates several events where phosphorus spikes with rainfall and seems to show a lagged reaction in ammonia concentrations.

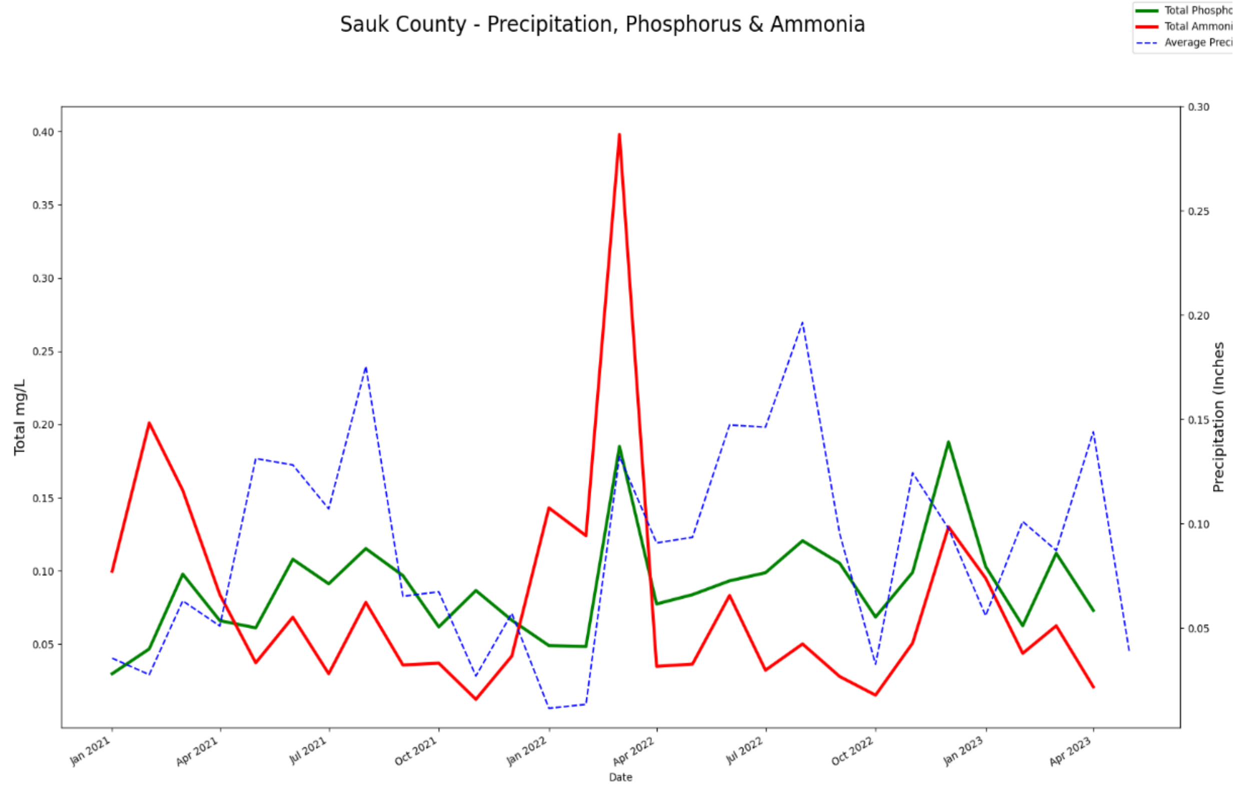

Figure 12- The left y-axis depicts Total mg/L of either Ammonia of Phosphorus. These are average monthly readings for Total Phosphorus and Total Ammonia. The right y-axis depicts average precipitation in inches by month. The x-axis ranges from January 2021 to April 2023.

Finally, it's important to understand how Sauk County's agricultural growth compares to other analogous counties. To make this comparison, I executed a comparative CDL analysis across counties, excluding those with under 10% cropped land. Figure 13 demonstrates that Sauk County growth is 98% correlated with comparable counties. From 2010 – 2022, Sauk County had roughly 12 thousand more cropped acres than comparable counties in 2010, most of the growth happening between 2014 and 2019. Since, 2019 the ratio of Sauk's cropped acres to other comparable counties has remained constant.

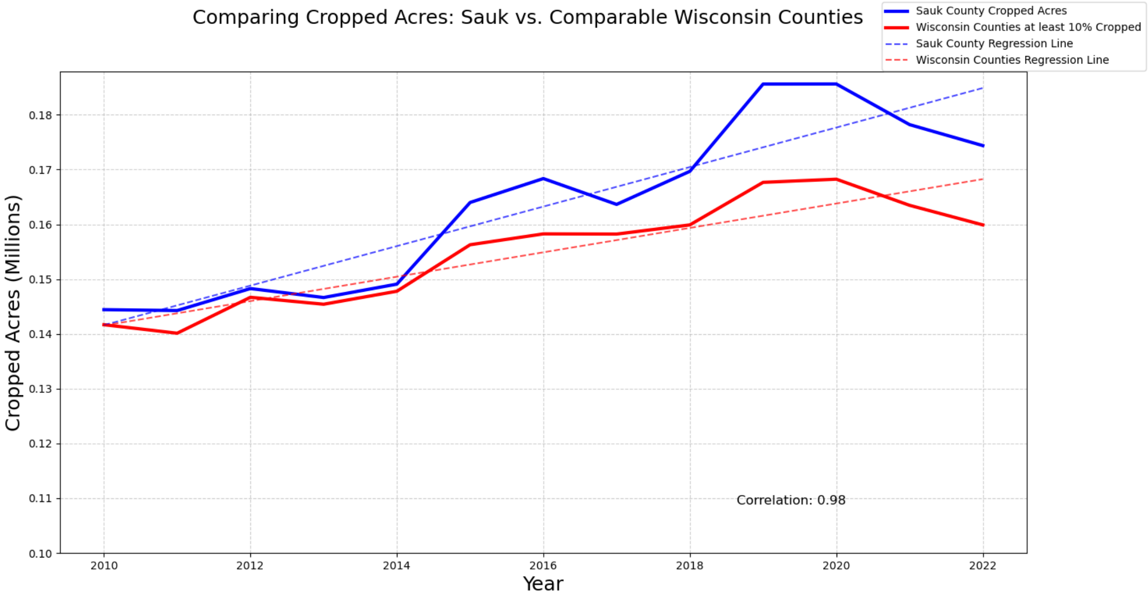

Figure 13 – Comparable counties are counties other than Sauk County with at least 10% of their county cropped.

5.2 IV METHODS

To analyze the impact NMP adoption has on total phosphorus or ammonia concentration, I employ a 2SLS strategy using FP zoning and AEA acres as IV. The first stage is depicted in equation 2.

$$NMP_i = \alpha + \delta_1 FPZone_i + \delta_2 AEA_i + \lambda_2 X_{i,t} + \theta_q + \varepsilon_{i,q} \qquad (2)$$

Here, NMP_i is given by the percent of NMPs in each watershed i; $FPZone_i$ denotes the percentage of land in watershed i that is located in a FP zoning district; AEA_i denotes the percentage of land in watershed i that is located in an AEA; $X_{i,t}$ is a vector of controls that includes precipitation, temperature, CAFO presence, percent of HUC12 that is cropped in a given year and water quality station type (stream or lake); θ_q is time fixed effects given by quarter; $\varepsilon_{i,q}$ is the error term.[9] Quarter fixed effects control for all factors that vary seasonally in all HUC12s, including rainfall and temperature. I define quarters as three-month periods e.g., January to March is quarter 1. I bootstrap the standard errors to account for low total sample size and potential HUC12 clusters (Cameron & Miller, 2015).

The 2SLS estimation equation is given by equation 2.

$$WaterQuality_{i,t} = \alpha + \lambda_1 \widehat{NMP}_i + \lambda_2 X_{i,t} + \theta_q + \varepsilon_{i,q} \tag{3}$$

Here, $WaterQuality_{i,q}$ is the total concentration of ammonia or phosphorus in milligrams per liter, which is transformed using the inverse hyperbolic sine transformation, of surface water in HUC12 watershed i in quarter q; NMP_i is the percent of area that is nutrient managed in HUC12 watershed i and is instrumented in equation 2; $X_{i,t}$ is the vector of controls; θ_q is quarter fixed effects; and $\varepsilon_{i,q}$ is the error term, which is bootstrapped.

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[9] The water quality data listed several location options which I categorized as either a lake or river. Location types which were defined as Lake, Reservoir or Impoundment were recoded as Lake. All other observations were coded as otherwise e.g., interpreted as river.

5.3 THREATS TO IDENTIFICATION

For an IV strategy to be causally identified, there are two main assumptions that must be satisfied: the Relevance Assumption and Exclusion Restriction. The relevance assumption states that the IV is highly correlated with the endogenous variable, NMP adoption. The exclusion restriction states that the IV only affects the outcome of interest through the endogenous variable. In our context, FP zoning district must only affect water quality through its effect on NMP adoption (Cunningham, 2021).

To test the relevance assumption, I use a series of F-statistics tests. The first is the Kleibergen-Paap rk LM statistic, which tests the null hypothesis that the equation is under identified. Thus, a sufficiently large F statistic indicates that the null hypothesis is rejected, and the excluded instruments are correlated with the endogenous regressor (Kleibergen & Paap, 2006). The second test I run is the Cragg-Donald Wald F statistic, which is a test for weak identification. This statistic tests the null hypothesis that the instruments are weak; thus, a sufficiently large F statistic confirms that the instruments are strong (Cragg & Donald, 1993). The final test is the over-identification test using the Sargan statistic. In this test, the null hypothesis states that the instruments are valid, and the excluded instruments were correctly selected. Thus, a high F statistic indicates invalid instruments (Hayashi, 2000).   Table 3 reports the results for the three statistics. The Kleibergen-Paap rk LM statistic is sufficiently high in both IVs to reject the null that the equation is under identified. Similarly, the Cragg-Donald Wald F statistic indicates that the instruments are strong. Finally, the Sargan statistic is sufficiently small to not reject the null hypothesis, indicating that the instruments are valid.

Table 3 – IV Tests

IV Test: FP Zoned Acres - Phosphorus	
Under identification test (Kleibergen-Paap rk LM statistic):	427.243
Chi-sq (1) P-value =	0.0000
Weak identification test (Cragg-Donald Wald F statistic):	1668.233
Over identification test (Sargan statistic):	1.933
Chi-sq (1) P-value =	0.3804
Instrumented:	Nutrient Managed Acres
Excluded Instruments:	FP Zoned Acres, AEA Acres & Interaction
IV Test: FP Zoned Acres - Ammonia	
Under identification test (Kleibergen-Paap rk LM statistic):	99.741
Chi-sq (1) P-value =	0.0000
Weak identification test (Cragg-Donald Wald F statistic):	237.826
Over identification test (Sargan statistic):	1.449
Chi-sq (1) P-value =	0.4847
Instrumented:	Nutrient Managed Acres
Excluded Instruments:	FP Zoned Acres, AEA Acres & Interaction

The exclusion restriction would be violated if FP zoning affected water quality in another way other than NMP adoption. One concern is if farmer could influence FP zoning decisions, specifically if they are already implementing SMART nutrient management. If this were the case, then the estimates of the impact of NMPs on water quality could be biased. However, this case is unlikely for a few reasons. Counties or municipalities make the decision to to adopt general zoning and then FP zoning districts, rather than individual farmers. Additionally, FP zoning is a tool used to restrict the development of large tracts of agricultural land. Thus, while there is a public comment period to the FP zoning process, individual farmers are unlikely to weild much influence. Therefore, zoning decisions are made on the objective criteria spelled out in Chapter 91, versus catering to individual farmers. As such, it is unlikely that there are unobservable differences between farmers whose land is in a FP zoning district versus those who were not.

5.4 IV RESULTS

The relationship between FP zoning districts and NMP adoption is critical to the identification strategy of this paper. Therefore, it is imperative to start by identifying whether FP zoned districts influence NMP adoption. Table 4 highlights the first-stage results which shows that 100% coverage in each HUC12 is associated with 49.7% of that HUC12 being under an NMP. Likewise, 100% AEA coverage of a HUC12 is associated with 33.1% coverage in NMP. These results are both significant at the 99% level and show that FP zoning districts and AEAs are effective at encouraging NMP adoption.

Table 4 – First-stage Results

Percent of HUC12 in NMP	
Percent of HUC12 in FP Zoning	0.497***
	(0.0400)
Percent of HUC12 in an AEA	0.331***
	(0.287)
Sample Size	11

Note - Standard errors in parentheses, * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

Next, table 5 displays the results of both a naïve regression and the 2SLS IV regression. The results indicate that when control variables and fixed effects are not included, NMP has is correlated with an increase in total phosphorus concentration. Once controls and fixed effects are included in the naïve regression, the results are still correlated with an increase in phosphorus concentrations albeit not a statistically significant one.

However, when examining the results for the IV 2SLS regression, the sign switches in all three equations and the percent of a HUC12 in NMP is correlated with a decrease in total phosphorus concentrations. Specifically, when the IV regression is run with minimum controls and no fixed effects, total phosphorus concentrations see a significant decrease of 2.6% for every

10-percentage point increase in NMP area in a HUC12. However, including controls in the second IV regression without fixed effects seems to explain much of the impact being captured in the first regression. In the preferred regression, I see a negative point estimate that is not statistically significant from zero.

The results in table 6 highlight the relationship between the percentage of a HUC12 in a NMP and total ammonia concentrations. The naïve regression results show a similar trend to table 5, where NMP adoption in a HUC12 is correlated with higher concentrations of ammonia. However, when I run the IV regressions the coefficients are all negative. In the first IV regression with minimum controls, I see a negative and insignificant coefficient. However, when I include controls in the IV regression, I see that a 10-percentage point increase in NMP area in a HUC12 is associated with a 6.98% decrease in total ammonia. In the third IV regression, I include fixed effects which account for some of the impact captured in the second regression, but the coefficient remains statistically significant at the 95% level. Specifically, I show in my preferred IV regression that a HUC12 with a 10-percentage increase in NMP coverage would have a 6.4% decrease in total ammonia concentrations.

Table 5 – Change in Total Phosphorus Concentrations

	Naïve OLS			IV 2SLS		
	Minimum Controls	Maximum Controls	Maximum Controls & FE	Minimum Controls	Maximum Controls	Maximum Controls & FE
Percent of HUC12 in NMP	1.499***	0.190	0.103	-0.260***	-0.0329	-0.0193
	(0.204)	(0.145)	(0.156)	(0.0404)	(0.0525)	(0.0446)
Percent of HUC12 in FP Zoning	-0.778***	-0.0908	-0.0301			
	(0.105)	(0.0787)	(0.0792)			
Percent of HUC12 in an AEA	-2.499***	-0.267	-0.123			
	(0.236)	(0.653)	(0.554)			
AEA FP Zoning Interaction	2.877***	-0.138	-0.547			
	(0.411)	(1.045)	(0.913)			
Precipitation Inches		0.114***	0.119***		0.111***	0.115***
		(0.0333)	(0.0310)		(0.0267)	(0.0307)
Mean Temp in Fahrenheit		0.00208*	0.00369**		0.00212**	0.00372**
		(0.000814)	(0.00135)		(0.000691)	(0.00124)
Sampled at Lake (Stream if otherwise)		0.0922***	0.0935***		0.0996***	0.101***
		(0.0140)	(0.0149)		(0.0152)	(0.0145)
CAFO Density		-0.0484	-0.0312		-0.0526**	-0.0282
		(0.0487)	(0.0378)		(0.0201)	(0.0186)
Percent of HUC12 Cropped		-0.0554	-0.0282		-0.0599	-0.00387
		(0.286)	(0.244)		(0.0642)	(0.0586)
Quarter Fixed Effects	No	No	Yes	No	No	Yes
Sample Size	466	466	466	466	466	466

Note - Standard errors in parentheses, * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

Table 6 – Change in Total Ammonia Concentrations

	Naïve OLS			IV 2SLS		
	Minimum Controls	Maximum Controls	Maximum Controls & FE	Minimum Controls	Maximum Controls	Maximum Controls & FE
Percent of HUC12 in NMP	0.250	0.409	0.349*	-0.0543	-0.698**	-0.640**
	(0.207)	(0.295)	(0.178)	(0.0354)	(0.228)	(0.233)
Percent of HUC12 in FP Zoning	-0.168	-0.171**	-0.144*			
	(0.115)	(0.0642)	(0.0645)			
Percent of HUC12 in an AEA	-0.629*	3.740	3.546*			
	(0.308)	(2.146)	(1.444)			
AEA FP Zoning Interaction	0.956**	-5.631	-5.421*			
	(0.303)	(3.226)	(2.238)			
Precipitation Inches		0.0256**	0.0332**		0.0405*	0.0447**
		(0.00869)	(0.0110)		(0.0198)	(0.0168)
Mean Temp in Fahrenheit		-0.000541**	-0.0000347		-0.000489*	0.000304
		(0.000210)	(0.000444)		(0.000242)	(0.000408)
Sampled at Lake (Stream if otherwise)		0.404**	0.384**		0.264**	0.246**
		(0.156)	(0.141)		(0.0833)	(0.0876)
CAFO Density		-0.00847	-0.00696		-0.0458	-0.0387
		(0.0586)	(0.0186)		(0.0312)	(0.0254)
Percent of HUC12 Cropped		-0.0968	-0.0729		0.665**	0.633*
		(0.307)	(0.118)		(0.240)	(0.249)
Quarter Fixed Effects	No	No	Yes	No	No	Yes
Sample Size	114	114	114	114	114	114

Note - Standard errors in parentheses, * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

CHAPTER 6: DISCUSSION

I investigated the relationship between nutrient management plans, a key practice in many nutrient reduction strategies. Despite its widespread popularity and importance, there is little causal evidence as to how much NMPs improve local water quality.

I address this using a two-pronged approach. First, I describe state-wide trends in water quality and nutrient management planning. I use correlative methods to describe the direction of their relationship, finding higher ratios of NMP to Cropped acres correlates with higher concentrations of phosphorus and ammonia. This positive association is likely due to the confounding relationship between higher use of planning in areas with more crop agriculture in the current moment as well as historically. Indeed, these confounding factors have challenged causal estimates of the impact of NMP on water quality.

Second, I use a 2SLS IV regression to account for these factors and discern the causal effect. I do so using Sauk County as a case study, employing a detailed dataset that identified all NMPs in the county, both from FPP and other programs. I determined that a 10-percentage increase in NMP coverage in a HUC12 would reduce ammonia by 6.4%. While there was no statistically significant impact on total phosphorus in the IV 2SLS, a negative point estimate was observed. The results were anticipated for several reasons. First, they are consistent with the existing literature on NMP in Wisconsin (Skidmore et. al., 2023). Second, phosphorus, a sediment-bound nutrient, is recognized to have potent legacy effects spanning decades (Phillips & Lindsey, 2003). Therefore, a two-and-a-half-year analysis period is unlikely to reveal any improvement in total phosphorus, whereas dissolved nutrients, like ammonia, can improve more rapidly. Third, the improvement in ammonia might be attributed to the composition of Sauk's crop rotation. Specifically, corn and soy make up Sauk’s predominant crops, accounted for approximately 51.4% of the cropped acres in 2021. These crops require substantial levels of nitrogen fertilizer throughout their cycle, and any level of nitrogen deficiency will adversely impact yields (Motasim et. al., 2022). Consequently, farmers have a potent incentive to engage in risk-avoidance behaviors by over-fertilizing these crops, which amplifies the risk for runoff events.

NMPs encourage practices that mitigate this over-application of nutrients. Specifically, farmers must conduct one soil test per five acres, which informs them of the requisite fertilization for optimal yields. It also promotes the use of slow-release fertilizers and split applications, both of which curtail the risk of nutrient runoff (Wisconsin DATCP, 2015). Moreover, cover crops act as nitrogen scavengers, absorbing excess nutrients and fixing them in the ground, while conservation tillage enhances water filtration/absorption in the soil, deterring nitrogen runoff (Blanco-Canqui, 2018; Liu et. al., 2014).

6.1 NMP ADOPTION IN WISCONSIN

Upon the initiation of the FPP, NMP adoption witnessed a substantial increase, enveloping approximately one-third of all agricultural lands. Nevertheless, since 2017, this figure has plateaued, mirroring the stability in cropped acres. Several factors may explain this stagnation. Since the inception of the FPP, farmers have been able to claim credits of $5, $7.50, or $10 per-acre. However, since then, they have remained nominally static and have seen a real term decrease due to inflation. For these credits to have equivalent incentive power to 2009 dollars, they would have needed to be increased to $7.30, $11.00, and $14.50 respectively, by August 2023 (BLS, 2023). Therefore, one of the simplest explanations to NMP adoption stagnation is that the incentive is no longer strong enough to induce individuals to implement NMP via the FPP.

It may also be the case that farms with lower barriers to implementing an NMP may have signed up initially, leaving behind those with higher implementation costs. NMP implementation tends to be more cost-effective when farms don't have livestock or surface water conduits. Thus, NMP adoption might have been primarily taken up by farms where implementation was more affordable. In contrast, farms with livestock or streams/surface water conduits may not participate in the program due to elevated upfront costs, such as upgrading their manure storage facilities or relocating these facilities if they are near streams. Farms housing livestock must also account for all nutrient sources during the planning process. Further, if these farms have challenging topography they may have to implement costly infrastructure upgrades such as diversions. Moreover, they might fear heightened scrutiny, owing to an annual certificate of compliance, particularly if they anticipate a high likelihood of NR 151 violations. Simply put, for these farms, the costs may outweigh the benefits.

A third potential explanation I posit is that achievable demand for NMP adoption still exists, provided tax credits are available, but zoning decisions impede further uptake. While a farm might be inclined to implement an NMP, to qualify for a tax credit, they must be situated either in an FP zone or an AEA. Figure 11 illustrates that only about half of Sauk County’s land is within one of these zones. Sauk assigns zoning decisions to individual municipalities and townships, while the DATCP must confer AEA designation following a comprehensive process. Furthermore, municipalities without general zoning ordinances are unlikely to adopt them to implement FP zoning districts. Thus, the need to first implement general zoning presents a significant barrier to adoption. Thus, it's likely that additional demand could be tapped in unzoned counties were they to be FP zoned. However, neither the incentives nor the demand for the FPP tax credit are robust enough to coax farms into pursuing an AEA or municipalities into adopting zoning.

6.2 POTENTIAL THREATS

The question remains: is Sauk County a general model for other counties? The answer is it depends. Sauk County has a significant agriculture industry with cropped acres ranging from 26.5% - 34.1% spanning the years 2010 – 2022. In 2021 28.2% of their cropped acres were in an NMP. Additionally, the Wisconsin River and Lake Wisconsin border a substantial portion of the county. Considering these factors, the findings are most likely generalizable to counties with analogous agricultural profiles, NMP adoption, and geographical features. Variables that may impact its external validity include urbanization, protected forests, and elevation profiles. Further, an increase in overall crop coverage may lead to deteriorating water quality readings, even if the adoption rate of NMP increases at the same or a faster pace. Therefore, if Sauk County is experiencing increased crop production compared to other agricultural counties in Wisconsin, this could bias overall water quality results.

A second threat to my analysis is upstream water quality. Upstream water quality is known to effect downstream water quality (Skidmore et. al, 2023). Since I studied a relatively brief timeframe and controlled for quarter fixed effects, I make the assumuption that there are no water quality shocks due to land use changes. Water quality changes due to weather events should be captured in via precipitation and temperture data and seasonal variations in my quarter fixed effects. However, land use changes such and agriculture intensification, increase in CAFO density or other point source emitters upstream, are not accounted for in my analysis. My future research will control for upstream water quality either controling for the water quality at the station immeadiately upstream of my observation or the mean water quality of the upstream HUC12.

6.3 POLICY IMPLICATIONS

Policy implications regarding water quality hinge on whether the incentive structure is drawing new participants to implement NMP or merely reallocating funds to farmers who would have adopted the practices regardless. A certain cohort of farmers would have deployed conservation practices even without the state's incentives, driven either by a strong environmental ethic, perceived cost savings in implementing such practices, or an auxiliary profit motivation stemming from marketing their products as eco-friendly. The question is: is this cohort large enough to bias FPPs impact on NMP adoption? In seeking answers to this question, I reviewed the annual nutrient management briefings from 2001 onward. According to these reports, 31% of NMPs were voluntary in 2003. Of note, in 2003 acres in NMP was less than one-fifth of 2021 acres and NMPs do not equate to acres e.g., just because 31% of plans were voluntary doesn't mean 31% of acres were. Post the FPP's inception, voluntary plans sharply decreased and have stabilized at 2% since 2016. From 2019 on, DATCP worked to accurately document NMP acres by reason, with their reports indicating that voluntary NMP acres comprised 5% in 2019, 4.7% in 2020, and 3.2% in 2021. Bearing these reports in mind, the FPP has markedly amplified NMP adoption across the state. Consequently, the surge in NMP adoption has catalyzed advancements in environmental practices during the same period, practices that would not have been adopted in the FPP's absence. Therefore, if policy makers wish to improve water quality further across Wisconsin, they could broaden accessibility to the program and adjust tax credit rates.

Accessibility to the FPP could be increased in two ways. First, policymakers could aim to harness latent demand by reconfiguring how FPP tax credits are claimed. Currently, about half of Sauk County's land is ineligible to participate in the FPP. By altering the incentive structure to accommodate farms in FP planned areas (see figure 11, map 1), more latent demand could be captured. For instance, they might offer $3 per acre if a farm is in FP planning, $5 if it's in an AEA, $7.50 if it's in FP zoning, and $10 if it's in both. An alternative strategy would centralize decision-making authority at the county level, mandating a certain percentage of all FP planned land be FP zoned. This would broaden access by enabling more farms to be located in FP zoned districts. Either of these options would likely enhance NMP adoption and could mitigate administrative burdens on municipalities.

Additionally, focusing on marketing the benefits of NMP may also drive further adoption. Periodically, DATCP's annual reports highlight cost savings as a benefit to NMP through reduced fertilizer costs or topsoil savings. However, finding ways to communicate the benefits quickly and effectively may improve FPP participation. For instance, the USDA markets SMART nutrient management saves, on average, $30 per acre. A similar tactic in Wisconsin may encourage further adoption.

Finally, future studies into the effect of NMP adoption at a small scale are needed to identify NMPs impact on total phosphorus and ammonia over the medium term. To accomplish this, additional data is needed, specifically, consecutive years of NMP shapefiles for a cluster of counties. These NMP shapefiles should include all instances of NMP adoption in each county and identify which NMPs fall under the FPP. Further, a cluster of agricultural counties would enable future research to capture upstream and downstream effects of NMP adoption. These counties would be useful insofar as they either share HUC12s or have adjacent HUC12s to other counties in the dataset. I believe having a five-year panel dataset that includes these attributes would enable future research to casually identify the impacts of NMP.

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APPENDIX A: NITROGEN CYCLE

Nitrogen is transformed into different forms through the nitrogen cycle and depending on the environmental conditions; these forms have different environmental impacts. Under normal or natural conditions, plants such as legumes will undergo the process of fixation, which pulls nitrogen gas out of the atmosphere and fixes it to the soil, making it available for future use. When manure is applied and optimal conditions are present, mineralization occurs, which is when microbes break down organic nitrogen (manure) and transform it into ammonium. Once in this stage, nitrification can occur, which changes ammonium into nitrate. Nitrate is negatively charged, just like the soil. As a result, nitrate easily moves with water and leaches into groundwater aquifers. Once in groundwater aquifers, it is slowly released into surface water. After nitrification, denitrification can occur when the soil is saturated and bacteria use nitrates as an oxygen source. This causes nitrogen gas and nitrous oxide to be released into the atmosphere. Finally, when urea organic nitrogen is applied, volatilization may occur. Volatilization occurs when nutrients are applied in hot and windy conditions and are not incorporated into the soil quickly (Johnson et al., 2005). This process releases ammonia gas, which can harm the local ecology and is a powerful greenhouse gas (RAND Corporation, n.d.).

APPENDIX B: SOIL TYPE AND TESTING

Sandy soil has low water retention and therefore, low water availability for the plant, whereas a fine and dense soil, such as clay, has higher water retention but prevents root elongation and therefore can limit water and nutrient availability for plants (Passioura, 1991). Loam is a soil that is in between the two; it has good water retention yet ample macropores, which allow for easy root growth; therefore, it is ideal for many types of plants (Ball, n.d.).

Understanding the nutrient and soil profile of a farm is the first step in successful nutrient management. Soil type dictates how quickly water infiltrates through the soil and if water and nutrients are readily available. Bedrock depth is a critical factor as it can affect how quickly ground water aquifers recharge, influencing the chances for runoff or "brown water" events (Muldoon, 2016). Understanding soil properties can inform farms of their optimal nutrient application practices, if needed (e.g., applying multiple small nutrient applications or using slow-release nutrients on sandy soil). However, soil testing informs farmers whether nutrients need to be added, where they need to be applied, and how much needs to be applied. Micha et al. (2023) found that soil-testing led to a 6.4% reduction in the use of phosphorus fertilizer on farms in Ireland. A further study found that increasing adoption of soil testing would be a catalyst in transforming Ireland's agriculture practices from unsustainable to sustainable (Macintosh et al., 2019).

APPENDIX C: SOIL CONSERVATION PRACTICES

Soil conservation or soil trapping techniques are the most effective techniques at controlling phosphorus. These techniques aim to reduce or eliminate soil erosion; they include a bevy of practices such as cover crops, conservation tillage, buffer strips, and others. Cover crops are an exceptional abatement tool in that they can reduce soil erodibility anywhere from 22-100%. Additionally, they reduce nitrogen leaching by an average of 53% and can outcompete weeds, reducing the need for pesticides. However, cover crops can take more than three years to fully establish and have limited effectiveness at reducing dissolved nutrients (Blanco-Canqui, 2018; Liu et. al., 2014). Conservation tillage refers to a variety of tillage practices that seek to minimize soil disturbances. These practices include no-till, strip till, ridge till, and mulch till systems. No-till has been exceptionally effective at sharply reducing topsoil erosion on U.S. farms (1.9 billion tones to 3.1 billion tones between 1982-1997). Specifically, no-till farms have 45-55% lower phosphorus loss than conventionally tilled farms (Daryanto et al., 2017). Other forms of conservation tillage yield these benefits to a lesser degree. For example, mulch tillage reduces soil losses anywhere from 5-40% (Busari et al., 2015). However, tillage practices decisions should be made with soil types in mind, as soils with high clay content will have high rates of runoff with no-till. Further, no-till farms have much more mixed results with reducing dissolved nutrient runoff and should be paired with other abatement practices such as cover crops (Daryanto et al., 2017).

Most waters enter receiving water bodies via first or second order headwater streams; thus, implementing buffers around these streams can be effective at reducing nutrient loading (Correll, 2005). Riparian buffers are perennial vegetated barriers between farms and waterways with the purpose of reducing runoff. They can either be grassed, wooded, or a mix of the two, with each type having its own benefits and drawbacks. Perennially grassed buffers excel at flood control, increasing hydraulic resistance, dispersing concentrated water flows, absorbing nitrates and phosphorus, and preventing erosion (Dosskey et al., 2010). However, grassed buffer strips' effectiveness can be nullified and even become a source of pollution if saturated with nutrients. To combat this, grass should be annually mowed and removed, and nutrient inputs controlled (Cole et al., 2020). Wooded buffer strips are best at stabilizing soil and can excel at reducing nitrates in shallow water, yet they are less effective at reducing phosphates (Rood et al., 2015). Therefore, trees should not be densely planted and grassed buffers should be incorporated into the abatement plan (Dosskey et al., 2010).

Correll (2005) describes the most effective vegetative buffer consisting of three zones. The first zone is immediately adjacent to the river or stream and consists of native trees that stabilize the riverbanks and water temperatures. The second zone closest to the stream consists of a wider zone of native trees, which reduce nitrates and the acidity of water. The third zone, farthest from the river, is a narrow and dense grass buffer, which traps sediment, sediment-bound pollutants, and dissolves pesticides (Correll, 2005). The ideal width of a buffer strip is 7.5 meters for reducing sediment-bound pollutants and 15 meters for removing dissolved nutrients, e.g., dissolved phosphorus and nitrates (Dosskey et al., 2008). Lastly, soil types should be factored into buffer strip design as soil can be a major factor in the effectiveness of the barrier.

APPENDIX D: LEGACY EFFECTS

Legacy effects are effects from historical actions, much like stock externalities. In terms of nutrient loading, phosphorus, and nitrogen both have long term lags that can prevent water quality measurements from showing improvement immediately. However, there is significant heterogeneity, depending on several variables. Phosphorus legacy effects are caused by sedimentary release. This is the process in which phosphorus, over time, is deposited into the sediment of a body of surface water. During periods of high external loading of phosphorus, sedimentation can provide a source of phosphorus that keeps total phosphorus levels high, even when external loading is normal. Several factors affect the rate of sedimentary release, including warmer water temperatures, anoxic conditions, and extreme pH levels (Hoverson, 2008). In one study, Hoverson found that sedimentary release accounted for 71% of Lake Shawano's annual phosphorus budget (Hoverson, 2008). Further, various studies have analyzed this process and found that lag times for total phosphorus in surface water may not be present for decades (Hamilton, 2012; Meals et. al., 2010; Rippey B. , Rippey et. al., 2021), In one scenario, the Baltic Sea, water quality improvements are not expected for at least 70 years (Gren I.-M. , 2009). However, the rate of surface water improvement depends on the hydrology and type of nutrient being measured. Typically, streams and rivers have a shorter time to recovery when compared to lakes or bodies of water with less flow (Hamilton, 2012).

Unlike phosphorus, nitrogen nutrient loading is heavily dependent on groundwater flows. While phosphorus is typically transported via sediment loss, nitrogen, depending on its form, can leach into groundwater flows or be transported via runoff. For instance, nitrates readily leech into groundwater and do not bind to soils while ammonia can be dissolved into a water column or associate with sediment (EPA, 2023). This leads to substantial heterogeneity in lag times for nitrogen. Meals et. al. (2010) reviewed the literature on lag times and found wide variances in responses, depending on the type of hydrology and the treatments being enacted. They found that smaller watershed or bodies of water generally responded more quickly to treatments than larger ones. However, nitrogen nutrient loading could have legacy effects measured in decades depending on the relevant variables (Meals et. al., 2010). Phillips and Lindsey (2003) were able to devise general guidelines based these variables. They found that abatement of point source pollutants delivered the most rapid results. Second, dissolved nutrients, if associated with the soil, would show rapid improvements as well. However, if dissolved nutrients are associated with groundwater, the median time to recovery would be a decade. Finally, sediment associated nutrients have the longest lag time measured in decades or longer can also display significant time lags depending on the local hydrology (Phillips & Lindsey, 2003).

APPENDIX E: NUTRIENT APPLICATION PRACTICES

Application practices of nitrogen and phosphorus can limit or exacerbate nutrient loading in nearby surface waters. One of the main obstacles to overcome in nutrient application is nitrogen leaching, as approximately 19% of nitrogen applied to the soil is lost due to leaching (Puga et al., 2020). Liquid urea, when compared to granular urea, significantly reduces nitrogen losses from ammonia volatilization and nitrous oxide emission, and enhances nitrogen use efficiency (Wang et al., 2020). Using liquid urea while splitting nutrient applications can further reduce nitrogen losses. Specifically, aligning applications when crops are demanding those nutrients can greatly enhance nutrient uptake efficiency. Wang et al. (2016) found that by applying liquid urea to corn during three application periods, which were lined up with their growth cycle, reduced nitrogen losses to 12.7% lost versus 27.9% loss from one application (Wang et al., 2016). Further, ensuring nutrient availability throughout the growing cycle has been shown to enhance crop growth (Olaiya et al., 2020) and induce higher yields (Motasim et al., 2022). Ultimately, using smaller applications of nitrogen ensures the targeted crop has enough capacity to absorb and utilize the nutrients being applied, which reduces nitrogen leaching (Motasim et al., 2022).