Nature's Kidneys: the Role of Wetland Reserve Easements in Restoring Water Quality

Nicole Karwowski and Marin Skidmore *

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Abstract

Non-point source pollution, primarily from agriculture, is a threat to water quality in the United States. Many reduction strategies aim to prevent runoff before it occurs by optimizing nutrient use or changing land use. Wetlands provide an ex-post natural solution by filtering sediments and excess nutrients from the landscape. This ability, one of the many ecological benefits they provide, has earned wetlands the name "Earth's kidneys." Quantifying the impact of wetland easements in the Wetland Reserve Program (WRP) and Agricultural Conservation Easement Program (ACEP) on water quality is critical for optimal allocation of funds for non-point source pollution abatement. We causally identify whether newly restored wetlands are effective at reducing nitrogen (ammonia) and phosphorus loads at the subwatershed level in an area spanning 40% of the continental US, the Mississippi/Atchafalaya River Basin. Results suggest that wetland easements reduce ammonia concentrations and that effects are heightened in areas with a higher proportion of vegetation and open water. Effects on phosphorous are complex; wetlands seem to act as both a source and sink for phosphorous nutrients. On average, increasing wetland easement spending by $900,000 in a subwatershed can decrease ammonia levels by 4%.

*Karwowski: University of Wisconsin Madison Department of Agricultural and Applied Economics. Skidmore: University of Illinois at Urbana-Champaign Department of Agricultural and Consumer Economics. Thank you to Eric Edwards, Holly Gibbs, Corbett Grainger, Nick Parker, Dan Phaneuf, and Andrew Stevens for excellent feedback. Thanks to seminar participants at the University of Wisconsin and the University of Illinois for helpful comments. For questions or comments, please reach out via email to nkarwowski@wisc.edu. Any errors or omissions are the sole responsibility of the authors.

1 Introduction

Non-point source pollution, primarily from agriculture, is a threat to water quality in the United States (Shapiro 2022). Among other damages, agricultural nutrients are a major cause of a hypoxic zone in the Gulf of Mexico, which is now over 6,000 square miles (Mississippi River Gulf of Mexico Watershed Nutrient Task Force 2023; River Gulf of Mexico Watershed Nutrient Task Force 2013; Committee on Environment and Natural Resources 2000). This acute problem has garnered significant attention, including the 2008 Gulf Hypoxia Plan and subsequent state-level plans by the twelve states bordering the Mississippi River.[1]

Wetlands provide an ex-post natural solution by filtering sediments and excess nutrients from the landscape (De Steven and Lowrance 2011). Wetlands slow the flow of surface water through a landscape. Wetland plants then take up nutrients for their own growth and facilitate the nitrogen cycle. Nitrogen, and particularly nitrates, that are detrimental in the water system, may be converted to and released as atmospheric nitrogen (N_2). This ability, one of the many ecological benefits they provide, has earned wetlands the name "Nature's kidneys."

Quantifying the impact of wetland easements on water quality in this region is critical for optimal allocation of funds for non-point source pollution abatement. Many reduction strategies aim to prevent runoff before it occurs by optimizing nutrient use or changing land use (Liu and Wang 2022; Paudel and Crago 2020; Skidmore et al. 2022). Topics such as nitrogen use efficiency, cover crops, and buffer strips are a major focus of nutrient loss reduction strategies. Programs to promote them, such as the Conservation Reserve (Enhancement) Program (CRP/CREP) and the Environmental Quality Incentives Program (EQIP) receive $ 6 billion in federal funding per year.[2] These programs, to varying degrees, address future nutrient loss. Wetlands provide an ex-post natural solution to nutrients already in our soils and water systems. Thus, wetlands are an important part of the set of activities to address hypoxia in the United States. However, their effect on water quality beyond a micro-watershed level are unknown.

Here we provide the first causal study of the effectiveness of the Wetland Reserve Program (WRP) and Agricultural Conservation Easement Program (ACEP) in improving water quality at a large-scale. The Wetland Reserve Program, founded in 1990, is a voluntary conservation program in which landowners permanently retire the right to use a field for agricultural production in exchange for a lump-sum payment (∼$3,000/acre). The Natural Resources Conservation Service (NRCS) then restores eligible parcels into their prior wetland condition. We estimate the effect of newly restored wetlands on nitrogen and phosphorous loads at the subwatershed level in the Mississippi/Atchafalaya River Basin. We accomplish this by combining over 30 years of water quality data with easement-level data on the WRP and

[1] https://www2.illinois.gov/sites/agr/Resources/NutrientLoss/Pages/default.aspx
[2] https://www.ers.usda.gov/topics/natural-resources-environment/conservation-programs/

ACEP. We employ both a continuous differences-in-differences strategy and an event study. The timing of this program allows us to distinguish between the endogenous choice to apply to the program, and the ecological treatment effect of a restored wetland.

We find wetland easements have significantly reduced ammonia in surface water across the MARB. We find that a one standard deviation increase in program participation in a HUC12 decreased ammonia concentration by 0.0055mg/L, or 4% of the sample mean. Wetlands were most effective at removing nitrogen when surrounded by higher areas of native vegetation or more surface water, but their effects were weaker in watersheds with high levels of nitrogen from point sources. Our results support the idea that wetlands can become overwhelmed when they receive high levels of nutrients and should be part of a wider set of solutions to water pollution.

In line with the literature, we find that wetlands are an imperfect solution to phosphorus pollution. On average, wetland easements did not significantly reduce phosphorus concentration. However, this masks seasonal heterogeneity in the ecological function of wetlands as a phosphorus sink and source. Wetlands retain phosphorus under warm and dry conditions (i.e., summer); they become a source of phosphorus during high-precipitation events (i.e., fall).

Identifying where wetland restoration can be most effective, as we do here, is an important conservation goal. Hansen et al. (2018) find that wetlands are the single most cost-effective management action to reduce sediment and nutrient loads. They point out the importance of placement for wetland function and motivate the analysis of how wetland function changes based on watershed land use. We build on our difference-in-differences model by adding an interaction term between wetland restoration and land use, weather, and different point source pollutants. This informs us of where wetland easements are effective water quality improvement tools.

We contribute to the literature on the ecosystem services that wetlands provide. Karwowski (2023a) shows that increasing wetland easements at the county-level increases average yields for corn, soybeans and wheat and reduce losses from excess moisture, heat, and disease. Beyond the aggregate level, Karwowski (2023b) uses field-level data to show that wetland easements also reduce flooding in the area near restoration and improve corn yields by 3-4 bushels per acre. Other wetland program analyses estimate their impact mitigating flood damages in resident areas (Taylor and Druckenmiller 2022; Kousky and Walls 2014; Watson et al. 2016).

There are over 3 million acres in the WRP, yet the ecosystem benefits of these conservation efforts in regards to water quality improvements are not well understood. Previous studies have been site-specific (Jenkins et al. 2010; Hansen et al. 2015; Marton et al. 2014) or model simulations (Cheng et al. 2020; Singh et al. 2019). In other cases, WRP is considered as one of a much larger bundle of easement payments such as CRP (Yin et al. 2021; Sprague and Gronberg 2012). These smaller studies have shown that wetlands have oversized impacts on water quality relative to their area. Estimating the impacts of restored wetlands in the Mississippi River Basin is therefore important to appropriately allocate conservation funds, as the role of the Wetland Reserve Program may otherwise be underestimated.

We also contribute to the growing literature on non-point source pollution policy. Efforts to address non-point source pollution are typically voluntary, and there has only recently been evidence as to which have been effective (Kling 2011; Ribaudo and Shortle 2019). Many of these efforts have yielded mixed or null results, highlighting the challenge of regulating non-point source pollution and achieving real reductions in nutrient loss (Paudel and Crago 2020; Skidmore et al. 2022). Yet the scale of the problem is such that significant efforts are being launched to address non-point source pollution, often with high direct or indirect costs.

The paper continues as follows: section 2 provides background, section 3 describes our methods, section 4 presents the results, and section 5 concludes.

2 Background

2.1 Wetland and landuse policy in the USA

The United States has experienced wide-scale wetland loss. Between 1780 and 1980, over 100 million acres of wetlands were drained and filled for urban and agricultural development. Over these two centuries, on average, 60 acres of wetland were destroyed every hour (Dahl 1990). Federal policies including the Swampland Acts of 1849, 1850, and 1860 incentivized landowners to drain wetlands in favor of agricultural expansion. Although the drainage and reclamation were costly, the productive nutrient-dense soils were highly profitable. It is estimated that land drainage increased land value by billions of dollars (Edwards and Thurman 2022). Corresponding, only half of the original wetland cover remains.

Since the 1900s, federal policies have shifted towards wetland ecosystem protection and conservation. Today, approximately 5.5% of the contiguous United States is in wetland (Dahl 201). Information on the ecosystem benefits of wetlands resulted in both policy and funds devoted to reversing losses. Legislation including the Migratory Bird Act (1919), Flood Disaster Protection Act (1973), and Toxic Substances Control Act (1976) led to wildfowl protection, regulated floodplain development, and restricted water pollution. These were some of the first policies that indirectly led to the protection of wetlands.

The Clean Water Act of 1972 explicitly limited development on wetlands. Section 404 of the Clean Water Act required authorization from the Army Corps of Engineers to dredge or fill wetlands for real estate, water resource projects, infrastructure, mining, or industrial activities. For unavoidable impacts, compensatory mitigation was required to replace the loss of the wetland in the watershed (EPA). The following mitigation mechanisms were used: mitigation banks, in-lieu fee mitigation, and permittee-responsible mitigation. The wetland bank program enrolled 700,000 acres of wetlands using cost-share payments and 10-year contracts. Notably, farming and forestry activities were exempt from this regulation.

It was not until the Food Safety Act of 1985 that agricultural producers faced penalties for converting wetlands to croplands. The "Swampbuster" provision prohibited USDA program benefits to producers who converted wetlands (Glaser 1986). Land owners that converted wetlands after the end of 1985 would be ineligible to receive loans, subsidies, crop insurance, and price support. Wetlands that were drained previously, farmable wetlands, and artificial wetlands were exempted from this provision. The USDA additionally defined the term wetland as land that (1) has a predominance of hydric soils; (2) is inundated or saturated by surface or groundwater at a frequency and duration sufficient to support a prevalence of hydrophytic vegetation typically adapted for life in saturated soil conditions; (3) under normal circumstances does support a prevalence of this vegetation (1985 Act, sec 1201(a)).

Besides precluding wetland destruction, the USDA also encouraged additional conservation and active restoration in 1990 through the Wetland Reserve Program (WRP). The WRP continued to receive funding from following Farm Bill allocations and in 2014, was enveloped in the the Agricultural Conservation Easement Program (ACEP).[3] The Natural Resources Conservation Services (NRCS) branch of the USDA provides landowners with financial and technical assistance to promote wetland conservation. Eligible landowners forgo the right to produce and develop on the parcel in exchange for payment. The NRCS then manages and funds the restoration of the wetland. Policy goals include flood mitigation, carbon sequestration, wildlife habitat, and improved soil and water quality.

An easement contract allows the landowner to retain ownership while selling the right to produce on that field. They retain the right to recreate, sell, and lease the land. [4] With a program emphasis on long-term conservation, most wetland easements are permanent or 30-year contracts. The landowner agrees to withdraw the eased land from all farming, ranching, and foresting practices. In exchange, the landowner receives a payment that is equal to the fair market value of the land (appraised value), the geographical area rate cap, or a voluntary landowner offer. Most often, payment is based on the geographical area and averages \$2,700/acre. For easements valued at less than \$500K, the payment will be lump sum or no more than 10 annual installments. For easements more than \$500K, payments are made in 5-10 annual installments. The NRCS restores the land to its natural wetland condition by planting native species, removing tiling, and building berms. Restoration costs on average \$650/acre.

[3]The Wetland Reserve Program, Grassland Reserve Program, and Ranch Lands Protection Program were bundled under one easement program. The ACEP wetland easement program largely parallels the former WRP.

[4]There are some possibilities of allowing grazing on the land or timber harvest if the practices are consistent with long-term wetland enhancement.

Eligibility for the wetland easement programs is dependent on the characteristics of the land as well as the landowner. The intended parcel needs to be one of the following: (1) farmed wetland, (2) converted wetland, (3) land that was used for agricultural production prior to natural flooding, or (4) riparian area that links protected wetlands. The landowner must have owned the land for seven years, be below an certain income threshold, and be compliant with the highly erodible land and wetland conservation provisions. Finally, only 10% of a county's farmable acreage can be enrolled.

Wetland easement applications are evaluated by the state-level NRCS departments. The NRCS assigns applicant parcels a score based on the ecosystem benefits it provides. The most highly ranked applicants are selected to enroll in the program. Each state was responsible for their own ranking criteria until 2022, when the federal NRCS implemented the national Conservation Assessment Ranking Tool (CART). Guiding ranking principles include performance of the proposed conservation practices, resource concerns, national priority concerns, environmental benefits, and compliance with other regulation. [5] The NRCS ranks sites based on the current land uses; previous wetland conditions; expected soil, water, air, plant and animal resource outcomes; planned restoration activities; vulnerability; resource and program priorities for wildlife species; maintenance requirements; proximity to other protected areas; and cost efficiency.

In 2022, there were 3 million acres enrolled in wetland easement programs. Enrollment is strongly correlated with available funding. ACEP funding is dependent on Farm Bill budget lines. From 1992 to 2019, the five-year budgets for ACEP have been 1.7, 3.3, 4.9, 2.2, and 2.3 billion dollars respectively.[6] In the recent Inflation Reduction Act (H.R. 5376), an additional $19.5 billion dollars was allocated to NRCS climate smart agriculture programs. Specifically, ACEP received $1.4 billion on top of the 2018 Farm Bill baseline funding. Conservation is becoming increasingly important as a tool for climate change adaptation, meriting further research into its economic, societal, and ecosystem impacts.

2.2 Background on water pollution

Water pollution is a critical challenge in the United States and worldwide. As such, the effects of policy's that govern pollution sources is a central question for environmental economists.

The Clean Water Act was passed in 1972 and was a historic pivot in the nation's efforts to restore and maintain the chemical, physical and biological integrity of the Nation's waters (33 U.S.C. 1251). The Environmental Protection Agency (EPA) required industrial, municipal, and other point source facilities to obtain permits for pollution discharges that directly entered surface waters. These permits are tracked in the National Pollutant Discharge Elimination System (NPDES). The Clean Water Act has significantly altered land use decisions, this has led to substantial improvements in water quality. There is

[5] https://www.nrcs.usda.gov/sites/default/files/2023-04/IRA-ACEP-Ranking-Criteria.pdf
[6] https://www.ers.usda.gov/topics/natural-resources-environment/conservation-programs/

a body of evidence showing the Clean Water Act has been responsible for large declines in water pollution (Shapiro 2022). These efforts are observable using measures of the overall health of a surface water body, including total dissolved oxygen and whether the body is fishable (Keiser and Shapiro 2019). The overall cost-efficiency of the Clean Water Act is difficult to estimate; while program costs were high relative to typical measures of environmental benefits, like housing value, clean water yields myriad societal benefits, including public health (Keiser and Shapiro 2019; Flynn and Marcus 2021).

Despite these gains, non-point source pollution remains a pervasive threat to water quality in the United States. Surface-water levels of nitrogen and phosphorus, which largely originate from agricultural fertilizers, have remained flat or are increasing (Paudel and Crago 2020; Keiser and Shapiro 2019).[7] This is often credited to the distinct policy approach to non-point source pollution, which is not subject to enforcement under the Clean Water Act (Kling 2011; Ribaudo and Shortle 2019).[8] Instead, federal policy governing agricultural land use and the use of agricultural nutrients is largely voluntary (Liu and Wang 2022). States or localities may implement more stringent policies, and these policies form a patchwork across the United States (Skidmore et al. 2022).

Efforts to reduce non-point source pollution have largely focused on three strategies: increased nutrient efficiency, in-field and edge-of-field practices, and land retirement. Cao et al. (2018) show that nitrogen fertilizer increased dramatically over the second half of the 20th century, from an average of 0.22 $gm^{-2}year{-}1$ in 1940 to 9.04 $gm^{-2}year{-}1$ seventy-five years later in 2015. Farmers continue to apply over the optimal rates in terms of economic efficiency (Sellars et al. 2020), despite the myriad private and societal benefits for increase nitrogen use efficiency (Langholtz et al. 2021; Lamkowsky 2021). As such, a primary focus of practitioners is to increase use of the "4-Rs," or applying the right source of fertilizer at the right rate in the right place and the right time (Banger et al. 2020; Beegle et al. 2000; King et al. 2018). Skidmore et al. (2022) find promising evidence that local policies to support these practices via nutrient management plans yields short-term reductions in nitrogen and phosphorus runoff.

In-field and edge-of-field practices affect how much of the nutrients applied to a soil's surface ultimately reach surface water. To this end, the USDA developed the Environmental Quality Incentives Program (EQIP) to provide cost-sharing for farmers that adopt a number of conservation practices, many of which reduce nutrient runoff. Among others, farmers may participate in EQIP for adopting cover crops, buffer strips, conservation tillage, wetland construction, and waste or fertilizer management. The USDA spent over \$15 billion through EQIP cost-sharing contracts in the ten years from 2009–2019 (Liu and Wang

[7] Agricultural fertilizers take the form of both chemical fertilizers and livestock manure that provide nitrogen, phosphorus, and potassium to crops.

[8] Funds for non-point source programs are available under section 319 of the Clean Water Act, but non-point activities, with the exception of large-scale animal agriculture, are not subject to monitoring, permitting, or enforcement in the same way that point-source activity is.

2022). Liu and Wang (2022) find that increased EQIP participation and payment led to reduced nitrogen levels but increased phosphorus levels. Their results highlight the pernicious problem of phosphorus; legacy nutrients challenge practitioners ability to decrease phosphorus through current practices as well as researcher's ability to identify reductions (Skidmore et al. 2022).

Finally, Ribaudo and Shortle (2019) theorize that land retirement is a potential, albeit expensive, pollution control tool. Retirement programs like CRP and ACEP effectively incentivize this strategy. CRP was not developed as a water quality policy, and its benefits for water quality have not yet been studied at a large scale. Similarly, the other two components of ACEP, the Grasslands Reserve Program and the Farm and Ranch Lands Protection Program, theoretically benefit water quality by reducing production on those acres, although their effects have not yet been quantified.

WRP, which we study here, yields two-fold benefit for water quality. First, restored wetlands require that land be retired from production, potentially providing a small reduction in water pollution. The majority of the water quality benefits of WRP, however, follows from the ecological processes of wetlands, which removes existing nutrients from the water system. In this way, wetlands diverge from the rest of the practice and literature on water pollution. Rather than reducing the additional nutrients we add to the system, wetlands provide an ex-post natural solution by capturing and removing nutrients, as we describe below.

2.3 The relationship between wetlands and nutrient pollution

Wetlands preserve water quality through a number of different physical and chemical processes (Kostel 2023). These processes differ for nitrogen and phosphorous and also vary by nutrient form (i.e., particulate vs. dissolved, organic vs. inorganic). There is evidence that wetlands are more effective at reducing nitrogen pollution than phosphorous. Nitrogen nutrients are directly removed from the waters through microbial activity (Fisher and Acreman 2004). Nitrogen nutrients are directly removed from the waters through sedimentation, plant uptake, and notably, microbial activity (Fisher and Acreman 2004). Phosphorous is removed from surface waters by settling on the wetland floor and through plant uptake of phosphates. Nitrogen nutrients are more often in a soluble form, so are more easily filtered from the surface waters.

First, wetlands remove nitrogen and phosphorous from agricultural runoff in surface water Miller (Miller). An international meta-analysis of 57 studies finds that natural wetlands reduce nutrient loading: the influx of nutrients into an specified body of water during a given time period (Fisher and Acreman 2004) The authors also point out that in some cases, wetlands may increase nutrient loading, particularly during autumn and winter when wetlands flush plant litter, water flows are higher, and erosion is more likely. A more recent study by Land et al. (2016) focused on 203 wetland sites confirms that on average, wetlands reduce the transport of total nitrogen and total phosphorous. The median removal rates for wetlands are 93 and 1.2 grams/meter^2 per year. These rates can be higher for created wetlands that optimize nutrient removal. In the eastern half of the United States where urban and agricultural runoff is high, wetland-nutrient models suggest that removal rates are 10-40 grams/meter^2 per year for nitrogen and 0.5-5 grams/meter^2 per year for phosphorous. However, wetlands in former farmlands are significantly less efficient at nutrient retention and removal.

Wetlands also retain and filter nutrients through sediment trapping, soil retention, plant uptake, and microbial processing (Johnston 1991). Figure 1 breaks down the different cycles that nitrogen and phosphorous nutrients experience in a wetland. Wetlands cause nitrogen and phosphorous to settle on the wetland floor in a process called sedimentation. Rainfall washes nutrients from upstream lands into surface waters; wetland vegetation traps flowing nutrients into the wetland. These nutrients can bind to sediment materials through a process called sorption. Wetland plants, algae, and bacteria can then use these nutrients as fuel. Wetland plants uptake nitrates, ammonia, and phosphates to grow during the spring and summer months. These nutrients are stored until the plants decompose; some of these nutrients remain in the plant material, but a majority of these are released back into the water during the fall and winter. The microbes in the wetland soil are responsible for the actual removal of nitrogen nutrients from the water through ammonification, nitrification, and denitrification. These chemical processes break down organic nitrogen, ammonia, and nitrates into less harmful substances which leads to cleaner water. Denitrification in particular plays a large role in converting nitrates into gaseous forms and returning nitrogen to the atmosphere (Martínez-Espinosa et al. 2021). These processes are highly dependent on season, temperature, precipitation, and the flow of water and nutrients.

Our understanding of how wetlands filter nutrients stems from site-level studies; the effects of wetlands at the watershed-level are less consistent. This could be due in part to a heterogeneous wetland effects, insufficient wetland coverage, or the complex interactions between different land uses masking the true wetland effect (Hansen et al. 2018). Strayer et al. (2003) find an insignificant correlation between wetland cover and nitrate levels using watershed-wide land use models. In a catchment-level study in Wisconsin, Powers et al. (2014) find no significant effect of wetlands on total nitrogen removal. Rather, they observe that lakes and reservoirs are responsible for the majority of nitrogen removal at the watershed scale.

Conversely, Hansen et al. (2018) show that wetlands reduce nitrates at the watershed level using spatially extensive water sampling data in the Minnesota River Basin. Wetlands decrease nitrate levels exponentially in the spring season during moderate and high streamflows. Identifying the effect of wetlands conditional on current corn and soybean crop cover, they find this effect is magnified in watersheds with a higher proportion of cropland. In areas with crop range percentages of of 50-55, 55-60, 60-65, 65-70, 70-75, 80-85, 85-90, wetlands decrease nitrate levels by 0.56, 0.43, 1.03, 1.01, 1.12, 1.55, and 2.35 mg/L respectively. On average, 73% of the land use in their study area was in row crop agriculture while 5% was wetland. In one of their sample subwatersheds, the Le Sueur River Basin had an average nitrate level 14.1 mg/L in June. Adding 1 percentage point of wetland cover to this subwatershed would result in a -1.12 mg/L (-8%) reduction in nitrates.

3 Methods

3.1 Data

We use three decades of data (1990-2020) on the locations of all wetland easements since program inception from the USDA NRCS. These data provide spatial boundaries and contract information on every wetland easement that has participated in the WRP. The locations of wetland easements in the MARB are shown in Figure 3. For each easement, we have the application date, the closing date of the contract, and the date on which the wetland was fully restored. In total, there are 9,000 easements in the MARB accounting for approximately 905,000 acres or 0.17% of land in this region. Wetland restoration grew steadily over the period (Figure 5).

We pair this with site-monitor water quality data from SNAPD. These data provide harmonized measures of nutrient readings at the station-level; details on this process can be found in Krasovich et al. (2022). We use surface water readings from the same period as our easement and land use data (1985 - 2018). We focus on ammonia and phosphorous readings in our analysis. Maps showcasing average nutrient levels at the watershed level in Figure 3 show the spatial relationship between wetland easements and water quality. Wetland easements are concentrated in areas with have higher nutrient concentrations and abundant agricultural activity (i.e., the corn belt). Variation in average nutrient concentration levels in the MARB is displayed in a timeline in 4.

We use data from the USGS Land Change, Monitoring, Assessment, and Projection (LCMAP) (version 13) database to measure non-wetland land cover. LCMAP performs analysis on Landsat images to produce annual information on land cover from 1985 - 2021. In particular, we use the categories barren, cropland, developed, grass shrub, ice/snow, tree cover, and water, as each of these are likely to affect the total nutrient availability and the likelihood of runoff in the surrounding region. More information on how these categories are defined and the data are produced is available from United States Geological Survey (2022)

We use data from the USDA Agricultural Census (1990 - 2020) to measure the livestock activity in a region. We aggregate the number of cattle, chickens, and hogs, all of which were consistently measured and reported across the period. Further division of types of animals within these categories limits data availability. From these categories, we estimate the total animal units by assigning weights of 1.0 to cattle, 0.4 to hogs, and 0.01 to chickens. These weights are based on a rough average animal units within each category (Conservation 2023). We impute units in non-census years by assuming linear trends between census years.

The EPA provides information on the locations of point-source polluting firms that are registered with a National Pollutant Discharge Elimination System Permit from 2008-present. NPDES permits are required for all firms emitting point-source pollution under the Clean Water Act (Keiser and Shapiro 2019). Farms are largely excluded, except for livestock operations with over 1,000 animal units (Raff and Meyer 2019).

Weather data comes from the PRISM Climate Group. We aggregate readings to the average mean monthly precipitation and temperature across all pixels in the HUC12. Aggregation of weather data to larger spatial units is the best practice due to spatial correlation of weather and further correlation in cases where data are interpolated between weather stations (Dell et al. 2014; Hsiang 2016)

The National Hydrology Dataset delineates upstream and downstream relationships between HUC12s and the distance to nearest stream or river. In cases where a HUC12 has multiple upstream HUC12, we assign weights of equal value to each upstream HUC12.

3.2 Sample

Our study focuses on the Mississippi/Atchafalaya River Basin, which is characterized by high agricultural production, heavy non-point source pollution, and harmful algal blooms (Ritter and Rao Chitikela 2020). The boundaries of our study are the Missouri, Upper Mississippi, Ohio, Arkansas-White-Red, Lower Mississippi, and Tennessee HUC4. This region encompasses 1,245,000 square miles, or 40% of the conterminous United States.

Summary statistics on the relevant variables are available in Table 1 and 2. Note that not all observations have both ammonia and phosphorous readings, so our sample sizes vary by nutrient type. The average nutrients levels in our sample are 0.138 mg/L for ammonia and 0.150 mg/L for phosphorous. Land under wetland restoration is minimal; the average subwatershed only has 27 acres of restored wetland, or 0.01% of land area. Meanwhile, the average subwatershed is 26,000 acres and can range from 4,000 to 1 million acres in size.

There are notable differences between the ever-treated HUC and the HUC that never enroll any wetland easements. Comparing the means between the treated and untreated HUC, we note that HUC with wetland easement tend to experience higher rainfall, lower temperatures, have more cropland, and are nearer riverbeds.

3.3 Identification strategy/empirical model

We employ a number of empirical strategies to estimate the impact of wetland easements on water quality.

First, we estimate the impact of current easement area on water quality using a continuous differences-in-differences model:

$$Y_{imt} = \beta Wetland_{imt} + \Gamma \mathbf{X}_{imt} + \alpha_w + \delta_m + \lambda_t + \epsilon_{imt}. \tag{1}$$

Our outcome, Y_{imt} is the water quality in HUC12 i in month m and year t. In our primary specification, we use ammonia or phosphorus concentrations provided by (Krasovich et al. 2022). Nutrient concentration is the standard outcome measure in analyses of the impact of non-point source policy (Paudel and Crago 2020; Raff and Meyer 2019; Skidmore et al. 2022).

We measure treatment in $Wetland_{imt}$, the presence of wetland easements in HUC12 i in month m and year t. Our preferred measure is the restored wetland area as a percent of total area in the HUC12. We control for watershed characteristics that affect water quality in $\mathbf{X}_{imt}$. These include land cover, livestock agriculture, point-source activity, and weather.

In some specifications, we also include the upstream nutrient concentrations of the relevant nutrient to control for the quality of the water that is flowing into the HUC12. This controls for differences in water quality in i that are affected by changes in upstream activity, and in particular accounts for potential spillover effects of upstream wetlands. In order for an observation to be included in this specification, there must have been a water quality reading in both i and its upstream HUC12 in the same month. This decreases our sample size.

In our preferred specification, we control for HUC8 fixed effects, α_w, which control for characteristics of the broader watershed that are consistent across time. This includes hydrologic, geologic or climate characteristics that change slowly. We control for the broader watershed (i.e., HUC8) rather than the HUC12 for two reasons. First, many of the time-invariant characteristics that affect water quality are likely shared within a HUC8. Second, including HUC8 fixed effects effectively allows for comparison within HUC8 rather than only within a HUC12; this broader comparison region allows for more variation from which we identify treatment effects.

We also include month fixed effects, δ_m and year fixed effects, λ_t. Month fixed effects are particularly important to control for seasonal variation in water quality, while year fixed effects account for region-wide phenomena that occur in a given year. This could include large-scale weather trends (e.g., a particularly wet year) or changes in national policy. As a robustness check, we also estimate a model with HUC4-by-month and HUC4-by-year fixed effects. This allows us to control for time trends specific to a smaller region.

There are a number of watershed characteristics that we expect will alter the effectiveness of a wetland in removing nutrients from a system. To explore this, we modify equation 1 to interact our treatment variable, $Wetland_{imt}$ with the watershed activities, $\mathbf{X}_{imt}$. This allows us to shed light on which subwatersheds should be targeted to optimize water quality improvements of wetland restoration.

3.4 Parallel trends

The validity of our estimated treatment effects rests on the assumption that treated and control HUC12 would have been on parallel trends in absence of the restored wetland. We use an event study to empirically test whether HUC12 were on parallel trends prior to the restoration of a new wetland.

$$Y_{imt} = \varSigma_{k=-5,k\neq-1}^{1}0\Big(\beta_k Wetland_{imt} * \mathbb{1}\{K_{imt} = k\}\Big) + \varGamma\mathbf{X}_{imt} + \alpha_w + \delta_m + \lambda_t + \epsilon_{imt}. \quad (2)$$

We denote the years since (or until) restoration using k, and a HUC12 has a value of 1 for K_{imt} when it is exactly k periods from restoration. We test this for the entire sample (i.e., all HUC12 in the MARB) and for the sample that was ever treated. The coefficients in the periods before treatment serve as a falsification test of whether the program had a significant effect on water quality prior to restoration.

In our setting, there may be a a small effect of the removal of the land from production in the periods between the contract signing and wetland restoration. This is similar in nature to the anticipatory effect discussed in Sun and Abraham (2021). However, we expect that this effect will be small, as the wetlands themselves are a very small proportion of area. Moreover, this effect should be controlled for in the control for total cropland area.

Our study, as with all cases where treatment was implemented occurred over time, is vulnerable to biased estimators due to heterogeneous cohort effects and negative weights (Sun and Abraham 2021; Callaway and Sant'Anna 2020; de Chaisemartin and D'Haultfœuille 2020). This is particularly true for the full MARB sample, as it includes never-treated units. As a robustness test, we test equation 2 using the method of Sun and Abraham (2021) to assign positive weights to each cohort.

Another challenge is that HUC12 may have multiple wetland restored in the period. We estimate a model with only the first restored wetland in a HUC12. This tests whether HUC12 were on parallel trends prior to the first restored wetland, although we cannot rule out that HUC12 with multiple restored wetlands were already improving in water quality by the time the subsequent wetlands were restored.

4 Results

4.1 Wetlands filter nitrogen from surface water systems

We find that wetland easements reduce ammonia concentration after restoration. We estimate effects among HUC12 that have ever received treatment in Table 4. In our preferred model (column 2), which uses variation within a HUC8, we find that a 1 percentage point increase in wetland restoration land area decreases ammonia concentrations by 0.0058 mg/L. In our sample, one standard deviation of wetland as a percent of HUC12 area is 1 percentage point; our presentation of results can therefore be interpreted as the effect of a one standard deviation increase in wetlands as a percent of HUC12 area. This negative and significant effect is consistent when we include HUC4-by-year and HUC4-by-month fixed effects, with coefficients between -0.56 and -0.83. We find one exception: including a HUC12 fixed effect (column 3) results in a positive and significant coefficient of 0.21.

We also examine how restored wetlands impact ammonia concentrations using a sample of all of the HUC12 units, even those that are never treated. We find similar evidence that restored wetlands have a negative, significant impact on ammonia concentrations. In Table 5, we find that increasing the proportion of restored wetlands significantly decreases ammonia levels. In our preferred model (column 2), a 1 percentage point increase in land area under wetland restoration decreases ammonia concentration by 0.0055 mg/L on average. Even when controlling for a number of different unit fixed effects, the effect remains significant besides when using a HUC12 fixed effect, which accounts for 40% of the variation in the outcome. When we include HUC4-by-month and HUC4-by-year fixed effects, ammonia concentrations are -0.0038 to -0.0049 mg/L lower after a 1 percentage point increase in restored wetlands.

We use an event study to explore the ammonia trends before and after a subwatershed's first wetland restoration project. This confirms whether our sample design uses suitable counterfactuals units for treated subwatersheds at their time of treatment. When using a subsample of HUC12 that ever receive treatment, we find no significant differences in the pre-period or post-period (Figure 6) When we include all the subwatersheds, we see that 2-3 years before treatment, ammonia levels are not significantly different between treated and control units. However, ammonia levels seem lower for HUC12 that enroll in the wetland easement program 4 to 5 years before the first treatment. We also find that the year of treatment, and 1-2 years after, ammonia levels fall by 0.04-0.06 mg/L each year. We run the event study using the Sun and Abraham specifications, and make the same conclusions (Table 7)

These results confirm that a sample of HUC12 that ever receive treatment provide better counterfactuals. The lack of pre-trends suggest that the results in this sample may be interepreted causally. While we do not find a significant decrease in post-treatment among the ever-treated sample, these coefficients only capture the single-year effects after the first wetland restoration. We rely on the pooled effects in Table 4 to estimate the total effect of wetland restoration on watershed ammonia concentrations.

Next, we include upstream restored wetlands to identify whether there is a meaningful impact on downstream water quality (Table 8). Again, we find that wetlands significantly reduce ammonia concentrations within their own watershed. We do not find evidence that upstream wetlands reduce ammonia levels downstream. This suggests that wetland effects on ammonia are more localized and limited to the subwatershed level. As expected, we find that upstream water quality has a meaningful impact on downstream ammonia levels. Increasing upstream ammonia by 1 mg/L will increase downstream subwatershed ammonia level by approximately 0.2 mg/L.

We explore whether restored wetlands perform differently based on surrounding land use in Table 9. Across specifications, we consistently find that wetlands are more effective at removing ammonia when there is more natural vegetation and there is more surface water in the surrounding region. In contrast, wetlands are less effective when surrounded by more barren land or more point-source pollution sources. Notably, we no longer estimate a significant main effect of wetlands. Our preferred specification uses the full time period and ever-treated HUC12. In the HUC12 with the mean natural vegetation and mean restored wetland proportion, an additional one percentage point of area in wetlands reduces ammonia by an additional 0.049 mg/L. At the mean area of surface water in the HUC12, wetlands reduce ammonia by an additional 0.0041 mg/L. A HUC12 with the mean barren area (0.002 percent of area) experiences 0.0055 mg/L less mitigation from an additional percentage point of wetland area.

These results are in line with the scientific literature. Surrounding vegetation buffers intercept sediments and keep the wetland from becoming saturated (Skagen et al. 2008). Furthermore, wetlands that are more connected to the surface water (as opposed to isolated wetlands) play a larger role in the water treatment processes (Craft and Casey 2000; Racchetti et al. 2011; Marton et al. 2014). The denitrification rates in well-connected wetlands are 1-2 orders of magnitude higher than the rates measured in isolated wetlands (Racchetti et al. 2011). Wetlands were most effective at removing nitrogen when surrounded by higher areas of native vegetation or more surface water.

The category of barren lands captures a number of land-use scenarios that have high erosion rates, such as sandy soils with minimal vegetation and transitional states. Transitional areas can include, for instance, land conversion from forest to agricultural, wetland to developed, or from grassland to mining activity (Anderson et al. 1976). The conversion of lands can increase nutrient loading rates as well as sedimentation (Berhane et al. 2020). These high-erosion settings may overwhelm the capacity of wetlands to filter nitrogen from the surface waters. Similarly, wetlands may not be effective means of reducing point source pollution for the watershed given the highly-concentrated nature of point-source pollution, which may overwhelm a wetland's capacity to trap and filter nitrogen.

We find that weather also has a perceptible impact on the relationship between wetland restoration and nutrient quantities. Higher temperatures have the expected effect on ammonia concentrations: a one unit increase in temperature leads to a 0.01 mg/L reduction in ammonia. Warmer environments increase microbial activity and significantly decrease nitrate concentrations (Veraart et al. 2011). We find that wetlands slow down this process: at the mean value of temperature, wetlands, an additional percentage point of wetland area dampens this effect by by 0.00275 mg/L.

Wetlands also significantly alter the effect of rainfall on nutrient concentrations. Precipitation events flush nutrients out of the watershed and into surface waters downstream (Skidmore, Foltz, and Andarage 2023). Wetlands mitigate these run-off events by slowing the rate of floodwaters and trapping sediments (Acreman and Holden 2013; Bullock and Acreman 2003). In our model, an additional percentage point of wetland increases ammonia levels by 0.0009 mg/L in subwatersheds with the average wetland and precipitation levels, precluding it from flowing downstream.

4.2 Wetlands act as a phosphorus sink and a source over the course of their lifetime

While the effect of wetlands on ammonia levels seem quite clear, the relationship between wetlands and phosphorous is more complex. We find evidence that wetlands can act as both phosphorous sinks (i.e., decrease phosphorus levels) as well as sources (i.e., increase phosphorous levels). While nitrogen (and ammonia) is water soluble and travels easily through the watershed, phosphorous particles travel attached to rocks and sediments, making phosphorous slow-moving.

Among the sample of HUC12 that are ever treated, we do not find significant average effect of wetlands on phosphorus concentration (Table 10). The only exception is a model including HUC4, year, and month fixed effects; we interpret this result with caution, as this model does not control for variations between HUC8 within a HUC4. When we include never-treated units, we find that a 1 percentage point increase in wetland area increases phosphorous levels by around 0.004 mg/L, on average (Table 11).

Again we evaluate the trends in phosphorus concentration before and after the first wetland restoration in a subwatershed. We find that phosphorous levels are on average higher in the 2-5 years in areas before restoration occurs in a standard event study (Figure 7 and Table 12). Pre-treatment differences are marginally smaller within the sample of ever-treated HUC12, although these coefficients still fail a parallel trends test. These results raise concern about analysis of phosphorus using the full sample and suggest that wetland placement and timing may violate the parallel trend assumption even among the ever-treated sample. Indeed, the worsening phosphorus trend prior to treatment is likely the source of the positive and significant results in the full sample. This analysis may be a good candidate for further methods such as matched difference-in-differences.

We control for upstream water quality and the proportion of wetlands in the upstream subwatersheds in Table 14. In our sample of ever-treated units, we find that restored wetlands do not have a significant impact on the phosphorous levels within the subwatershed, and we find weak evidence that restored wetlands increase phosphorus downstream (significant at the 10% level). In the full sample, we find weak evidence that restored wetlands decrease phosphorous levels in their own HUC12 (significant at the 10% level) and no downstream effect. We again find that an increase in upstream nutrient levels raises downstream nutrient concentrations; 1 mg/L upstream increases downstream phosphorus by 0.3-0.4 mg/L.

The complexity of our results mirror those of Liu and Wang (2022) in their analysis of the EQIP program. They find that the conservation practices associated with the EQIP program reduce nitrogen levels but also lead to an increase in erosion, which increases phosphorous levels.

Although we find no average effect of wetlands on phosphorus, we expect that this may mask the fact that wetlands act as sources or sinks of phosphorous nutrients under different conditions. We explore these mechanisms in Table 15 and confirm that weather is a main determinant of whether wetlands are a source or a sink of phosphorus (Land et al. 2016).

We find that wetlands are more effective at removing phosphorus from the system when temperatures are high, while they are less effective when precipitation is high. At the mean value of temperature, wetlands, an additional percentage point of wetland area reduces phosphorus by 0.00327 mg/L, while at the mean level of precipitation, wetlands increase phosphorus by 0.00261.

These findings reflect the expected relationships based on the agronomic literature (Ercoli et al. 1996; Mackay and Barber 1984). Soils will more readily adsorb phosphates in higher temperatures, allowing plants to increase their phosphorous uptake. In our model, we find that a one unit increase in temperature, decreases phosphorous levels in the subwatershed by 0.007 mg/L. In an area with more wetland plants, this effect is magnified, leading to larger reductions in phosphorous concentrations in the subwatershed.

During times of low precipitation, wetlands retain phosphorus. Higher levels of precipitation may disrupt the sedimentation of phosphorous on the wetland bottom. Heavy rainfall events can flush these sediments and the accumulated phosphorous downstream, resulting in lower subwatershed concentrations at the cost of higher phosphorous downstream. As described earlier, wetlands trap the sediments during run-off events, and lead to higher phosphorous nutrient levels in the watershed.

There is also a significant interaction effect between wetland areas and and animal units in the later period in the all HUC sample (column 3), suggesting that wetlands decrease phosphorous levels in these contexts. In a subwatershed with mean wetland area and mean animal units (2,286), an additional percentage point of wetland reduces subwatershed phosphorous levels by 0.001 mg/L. Cow manure is a large source of nutrient loading and phosphorous pollution in surface waters (Raff and Meyer 2022; Waller et al. 2021). Wetland plants may uptake more of these nutrients in areas with significant animal production, resulting in better water quality. Once again, our results point to the fact that wetlands may be effective tools at improving water pollution in areas with poor water quality from agricultural, non-point sources.

4.3 Cost benefit analysis and results comparison

We carry out a back-of-the-envelope calculation to determine the cost-effectiveness of wetland restorations in regards to improving ammonia water quality levels in the MARB. We estimate that a 1 percentage point increase in wetland restoration leads to a 0.0058 mg/L reduction in ammonia levels. The average HUC12 is 25,819 acres, so increasing the wetland area by 1 percentage point would increase wetland restoration by 258 acres (approximately 3 wetland easement contracts). With an average purchasing cost of $2,700/acre and restoration cost of $650/acre, this 1 percentage point increase would cost $864,000. Thus, spending $900,000 in a subwatershed on wetland restoration leads to a decrease in ammonia levels by 4% each year.

How does this cost to benefit ratio compare to other conservation programs and natural wetlands? Liu and Wang (2022) find that a 1 standard deviation increase in EQIP spending per HUC10 ($62,000/HUC10/year) reduces nitrogen 0.041mg/L, or 2.89% of the sample mean. In their paper on fertilizer use, Paudel and Crago (2020) estimate that a 10% increase in fertilizer per HUC8 leads to a 1.52% increase in nitrogen and a 1.37% increase in phosphorous. Finally, in their paper on natural wetlands, Hansen et al. (2018) finds that adding 1 percentage point of wetland cover to a subwatershed in the Minnesota River Basin results in a -0.0112 mg/L (-8%) reduction in nitrates. We find that WRP yields comparable benefits to other water quality programs.

5 Conclusion

Water pollution, primarily from non-point agricultural sources, continues to be a problem in the Mississippi River Atchafalaya River Basin. We show in this paper that restoring wetlands through the USDA easements program is one potential abatement strategy that leads to lower nutrient levels, especially for ammonia. Leveraging the quasi-random timing of restoration completion, we use a continuous difference-in-differences strategy to estimate how the stock of land in wetland easement impacts ammonia and phosphorous levels. We add to the literature by showing that these wetland retention and filtration effects impact water quality at the subwatershed (HUC-12) level.

Increasing the proportion of land in wetland easement by one standard deviation reduces ammonia concentrations in the subwatershed by 0.0055 mg/L, or 4%, under a variety of different fixed-effects specifications. We find that surrouding land use and weather conditions meaningfully impact the rate at which wetlands filter ammonia from the system. Wetlands in areas with higher levels of vegetation and surface water filter more ammonia, while places with more barren land and heavy in point-source pollution see reduced filtration effects. Wetlands are less effective at filtering ammonia during times of high temperature and high rainfall; this could be because of their local cooling abilities and tendency to act as a sink, trapping sediments within the watershed.

Wetlands did not have a unidirectional impact on phosphorous levels. On average, we did not find a significant impact of wetlands on phosphorous among the sample of ever-treated HUC12. However, there are conditions under which wetlands decrease or increase levels of phosphorous in a subwatershed. During times of high precipitation, the sedimentation on the wetland floor is disrupted, resulting in lower levels of phosphorous within the subwatershed, but potentially higher concentrations downstream. When temperatures are higher, wetlands are more likley to decrease subbain phosphorous levels, perhaps because wetland plants will uptake more phosphate nutreints to grow during the summer months. Wetlands also seem to filter higher levels of phosphorous in areas with heavy agricultural production, particularly those with more animal units.

Our findings are in line with the literature of how wetlands impact nutrients broadly Hansen et al. (2018), at the site-level (Marton et al. 2014) as well as studies on how conservation activity impacts nutrient pollution Liu and Wang (2022). We shed light on how man-made restored wetlands in the USDA easement programs impact water quality at a larger scale over a span of thirty years. We also highlight which areas and under which conditions wetlands are more efficacious water treatment solutions. These results can aid policy-makers target future wetland restorations in a cost-effective matter, and lead to improved water quality.

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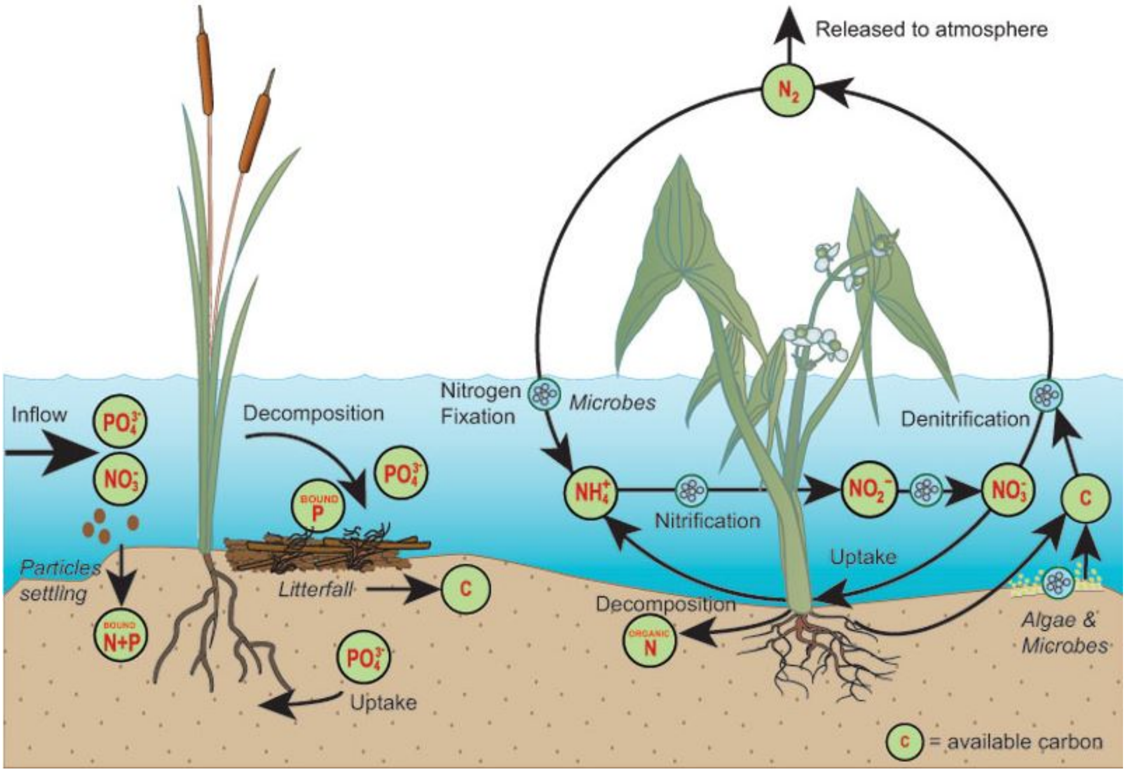

Figure 1: A graphic illustrating how nitrogen and phosphorous cycles in a wetland ecosystem from The Wetlands Initiative.

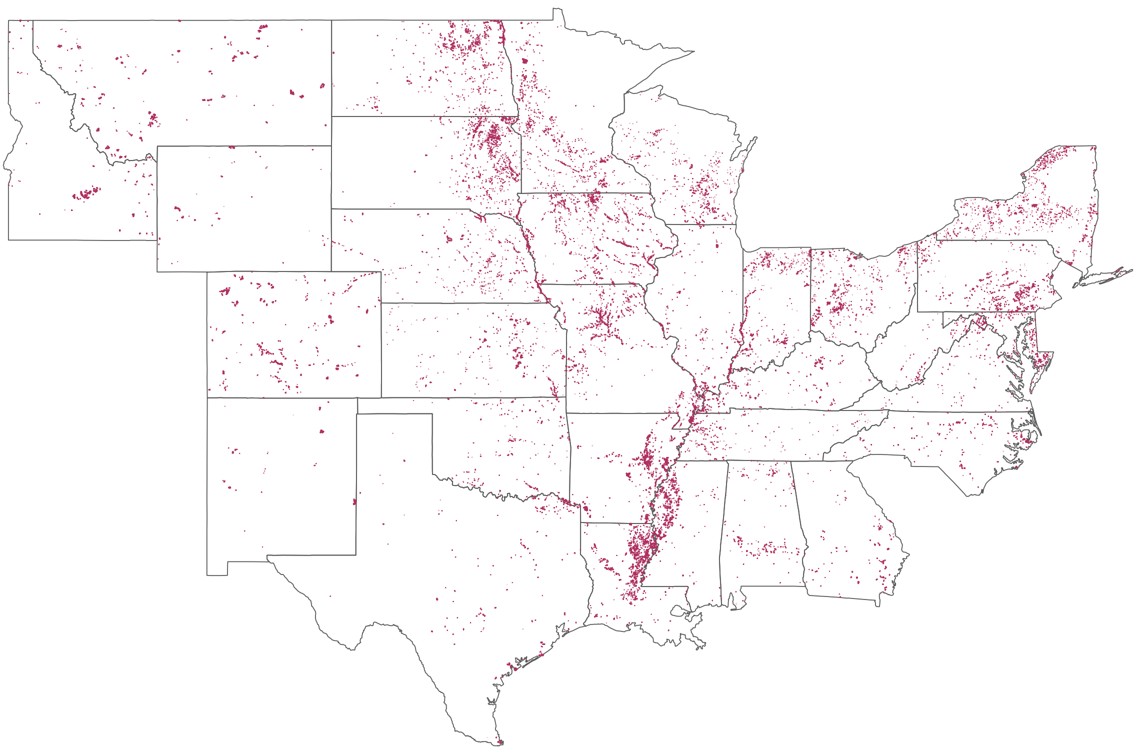

Figure 2: A map of restored wetlands in the WRP and ACEP at the end of 2018.

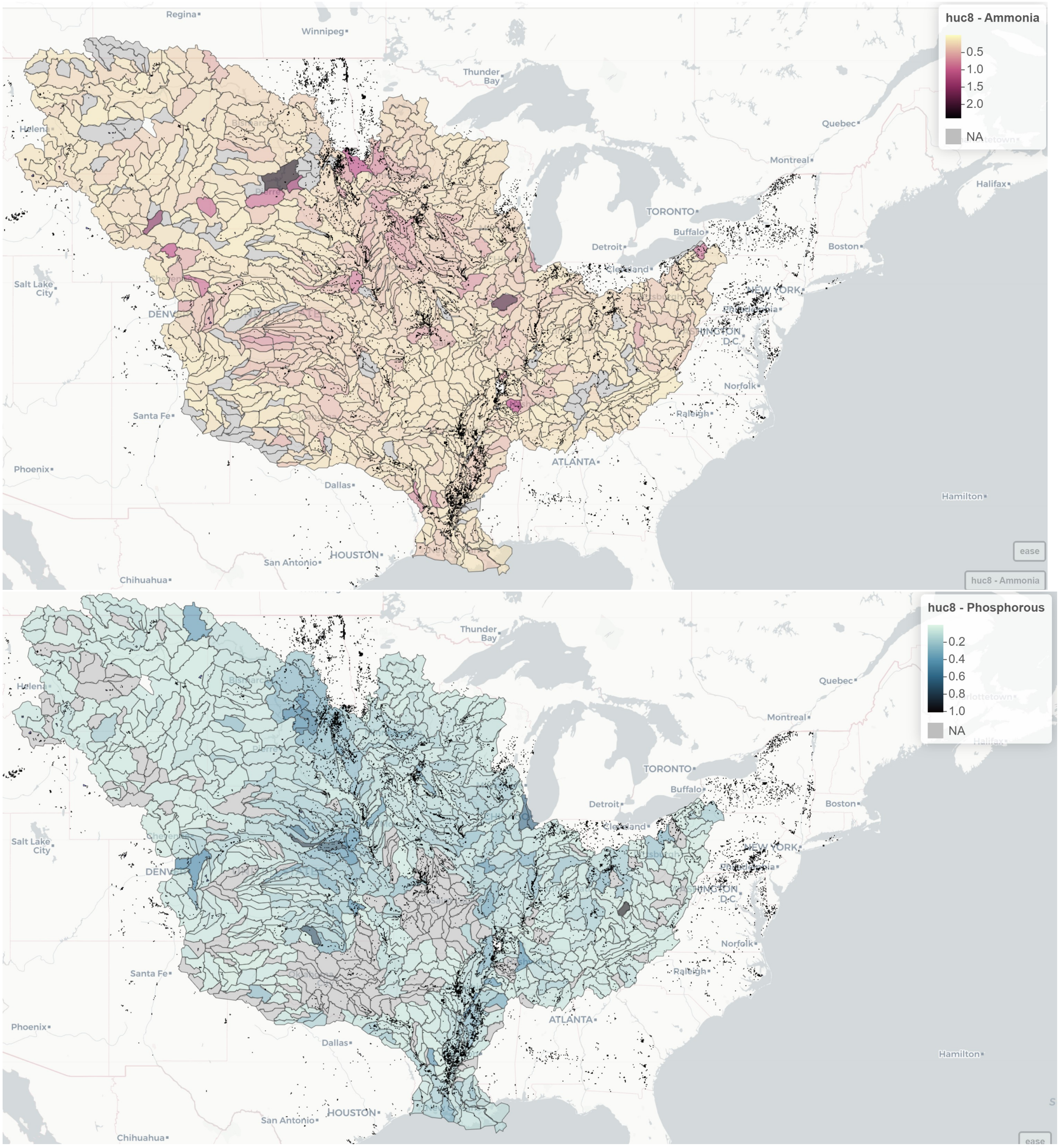

Figure 3: Maps of restored wetlands overlaid with average HUC12 ammonia and phosphorous concentrations in the Mississippi Atchafalaya River Basin.

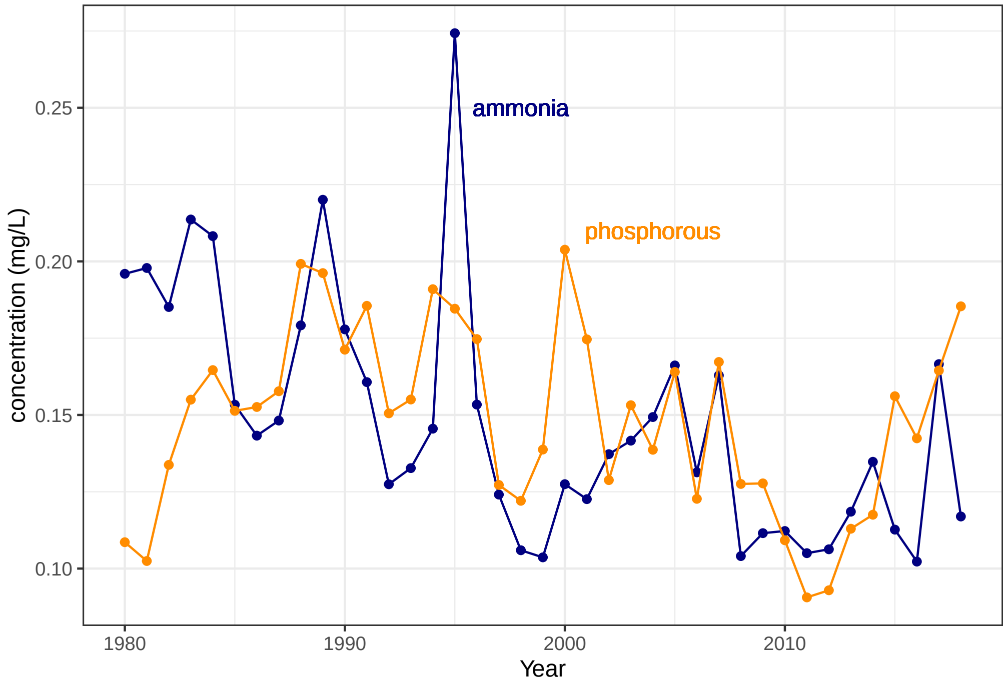

Figure 4: Average ammonia and phosphorous levels over time.

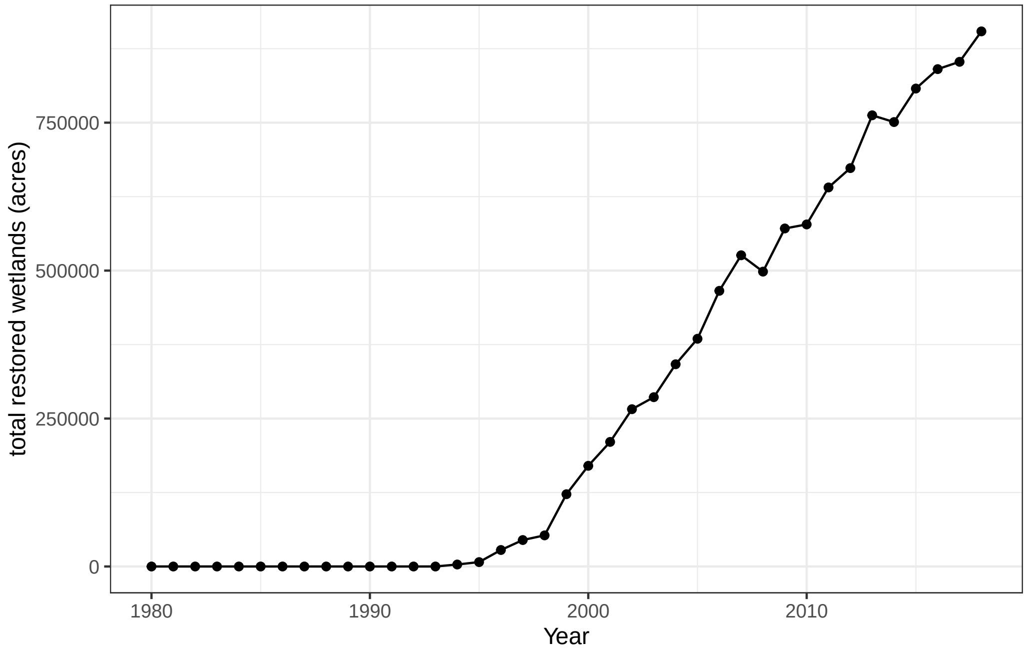

Figure 5: Total restored wetland acreage in the MARB over time.

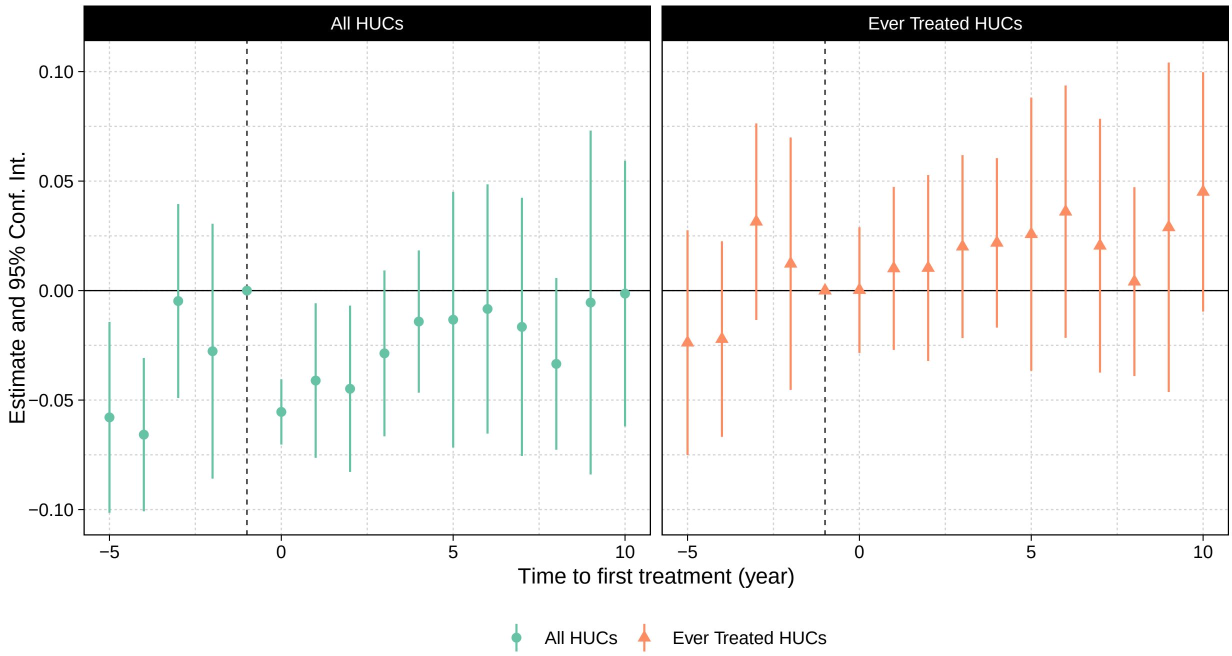

Figure 6: Effect of first WRP restoration on ammonia concentrations using a standard event study model. Models include HUC8, year, and month fixed effects and controls.

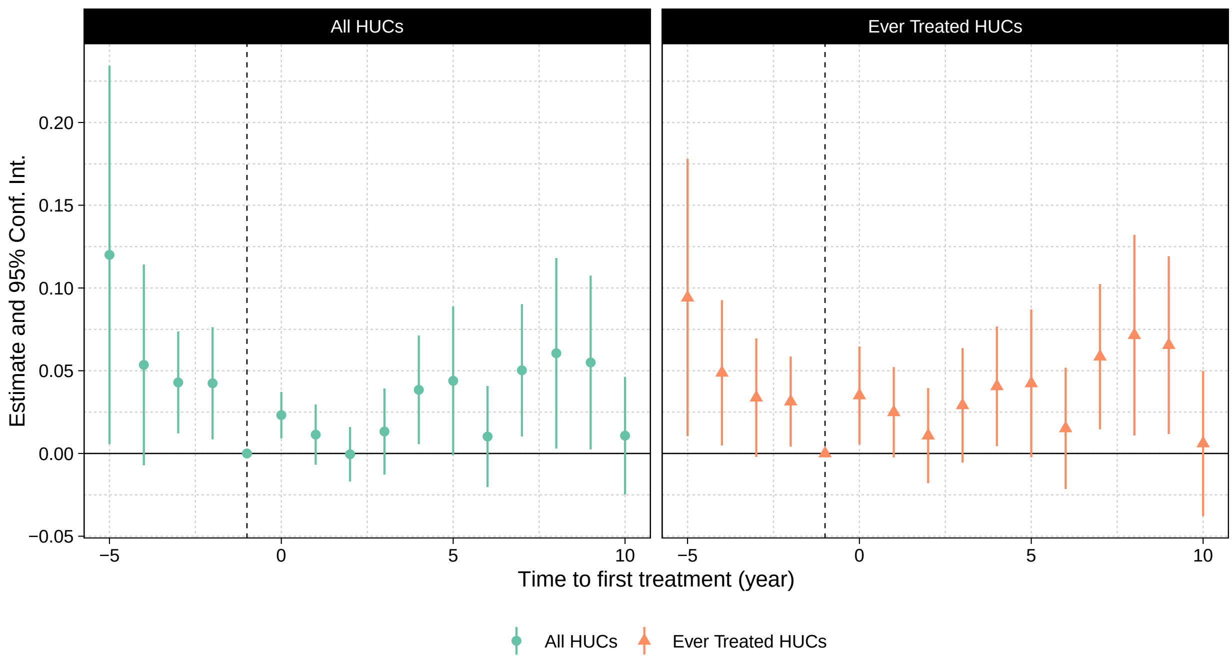

Figure 7: Effect of first WRP restoration on phosphorus concentrations using a standard event study model. Models include HUC8, year, and month fixed effects and controls.

Table 1: Summary statistics for all HUC

Statistic	N	Mean	St. Dev.	Min	Max
ammonia (mg/L)	121,933	0.138	0.341	0.004	4.940
phosphorous (mg/L)	81,587	0.150	0.258	0.003	2.490
restored wetland (acres)	173,370	26.850	265.067	0.000	6,600.260
restored wetland (prop)	173,370	0.001	0.011	0.000	0.288
ppt (monthly avg mm rainfall)	173,370	84.919	61.987	0.000	632.815
temp (monthly avg C)	173,370	13.547	9.643	−20.765	34.447
HUC 12 acres	173,370	25,819.430	19,104.160	4,410.880	1,052,467.000
distance to nearest river (m)	173,370	31.876	383.605	0.000	22,298.100
nitrogen facilities	55,047	1.173	2.799	0	45
phosphorous facilities	55,047	0.323	0.872	0	18
animal units	117,100	2,285.701	2,543.742	0.000	44,386.030
barren	173,370	0.004	0.021	0.000	0.620
cropland	173,370	0.406	0.294	0.000	0.994
developed	173,370	0.100	0.175	0.000	0.998
vegetation	173,370	0.389	0.299	0.000	1.000
ice snow	173,370	0.001	0.007	0.000	0.204
water	173,370	0.037	0.081	0.000	1.000
wetland	173,370	0.062	0.105	0.000	0.963
upstream restored wetland (acres)	39,812	7.067	68.435	0.000	3,700.490
upstream restored wetland (prop)	39,812	0.0003	0.003	0.000	0.096
upstream ammonia	18,677	0.141	0.327	0.004	4.900
upstream phosphorous	14,602	0.179	0.290	0.003	2.470

Table 2: Summary statistics for ever-treated HUC

Statistic	N	Mean	St. Dev.	Min	Max
ammonia (mg/L)	14,257	0.133	0.340	0.004	4.850
phosphorous	8,708	0.155	0.222	0.003	2.490
restored wetland (acres)	18,760	248.134	770.994	0.000	6,600.260
restored wetland (prop)	18,760	0.010	0.032	0.000	0.288
ever treated	18,760	1.000	0.000	1	1
max restored wetland (prop)	18,760	0.023	0.045	0.0001	0.288
ppt (monthly avg mm rainfall)	18,760	87.152	61.382	0.000	483.176
temp (monthly avg C)	18,760	13.091	9.967	−16.334	31.109
HUC 12 acres	18,760	29,198.790	13,745.220	8,503.260	152,174.000
distance to nearest river (m)	18,760	5.405	76.422	0.000	1,666.772
nitrogen facilities	6,841	1.278	2.624	0	25
phosphorous facilities	6,841	0.331	0.942	0	11
animal units	13,993	3,114.457	3,486.504	0.000	36,711.540
barren	18,760	0.002	0.005	0.000	0.068
cropland	18,760	0.588	0.196	0.014	0.936
developed	18,760	0.073	0.101	0.003	0.554
vegetation	18,760	0.189	0.160	0.000	0.845
ice snow	18,760	0.000	0.000	0	0
water	18,760	0.039	0.053	0.000	0.335
wetland	18,760	0.109	0.113	0.000	0.609
upstream restored wetland (acres)	4,597	37.583	148.024	0.000	3,700.490
upstream restored wetland (prop)	4,597	0.002	0.006	0.000	0.096
upstream ammonia	1,972	0.177	0.368	0.004	4.900
upstream phosphorous	1,614	0.194	0.274	0.004	2.400

Table 3: Difference in means between ever-treated and never-treated HUC

	Ever-treated		Never-treated		
	Mean	N	Mean	N	Difference
ammonia	0.13	13,280	0.14	108,653	-0.007*
phosphorous	0.15	7,226	0.15	74,361	0.003
restored wetlands (acres)	276.90	16,811	0.00	156,559	276.9***
restored wetland (prop)	0.01	16,811	0.00	156,559	0.012***
max restored wetland (prop)	0.02	16,811	0.00	156,559	0.023***
ppt	88.49	16,811	84.54	156,559	3.956***
temp	13.13	16,811	13.59	156,559	-0.458***
HUC 12 acres	29,505	16,811	25,424	156,559	4081.7***
distance to river (m)	4.51	16,811	34.81	156,559	-30.30***
nitrogen facilities	1.30	6,675	1.16	48,372	0.149***
phosphorous facilities	0.34	6,675	0.32	48,372	0.0170
animal units	3,056	13,321	2,187	103,779	868.8***
barren	0.00	16,811	0.00	156,559	-0.003***
cropland	0.59	16,811	0.39	156,559	0.200***
developed	0.07	16,811	0.10	156,559	-0.030***
vegetation	0.18	16,811	0.41	156,559	-0.228***
ice snow	0.00	16,811	0.00	156,559	-0.0006***
water	0.04	16,811	0.04	156,559	0.004***
wetland	0.11	16,811	0.06	156,559	0.057***
upstream restored wetland (acres)	39.64	4,239	3.19	35,573	36.45***
upstream restored wetland (prop)	0.00	4,239	0.00	35,573	0.002***
upstream ammonia	0.17	1,846	0.14	16,831	0.030***
upstream phosphorous	0.20	1,420	0.18	13,182	0.018*

Signif. Codes: ***: 0.001, **: 0.01, *: 0.05

Table 4: The effect of restored wetlands on ammonia concentrations: Ever-treated HUC

	ammonia					
	(1)	(2)	(3)	(4)	(5)	(6)
restored wetland (prop)	0.0002	-0.579***	0.210**	-0.825***	-0.564***	-0.817***
	(0.057)	(0.146)	(0.107)	(0.157)	(0.142)	(0.152)
Dependent variable mean	0.13118	0.13118	0.13118	0.13118	0.13118	0.13118
Observations	13,280	13,280	13,280	13,280	13,280	13,280
Adjusted R^2	0.20872	0.26406	0.40397	0.43882	0.29570	0.47598
year fixed effects	✓	✓	✓		✓	
month fixed effects	✓	✓	✓	✓		
HUC4 fixed effects	✓					
HUC8 fixed effects		✓		✓	✓	✓
HUC12 fixed effects			✓			
HUC4 x year fixed effects				✓		✓
HUC4 x month fixed effects					✓	✓

Clustered (HUC8-year) standard-errors in parentheses. Signif. Codes: ***: 0.01, **: 0.05, *: 0.1

Table 5: The effect of restored wetlands on ammonia concentrations: All HUC

	ammonia					
	(1)	(2)	(3)	(4)	(5)	(6)
restored wetland (prop)	-0.369***	-0.549***	0.013	-0.444***	-0.491***	-0.386***
	(0.072)	(0.126)	(0.080)	(0.116)	(0.112)	(0.104)
Dependent variable mean	0.13781	0.13781	0.13781	0.13781	0.13781	0.13781
Observations	121,933	121,933	121,933	121,933	121,933	121,933
Adjusted R^2	0.07133	0.13941	0.43992	0.19605	0.16205	0.21617
year fixed effects	✓	✓	✓		✓	
month fixed effects	✓	✓	✓	✓		
HUC4 fixed effects	✓					
HUC8 fixed effects		✓		✓	✓	✓
HUC12 fixed effects			✓			
HUC4 x year fixed effects				✓		✓
HUC4 x month fixed effects					✓	✓

Clustered (HUC8-year) standard-errors in parentheses. Signif. Codes: ***: 0.01, **: 0.05, *: 0.1

Table 6: Standard event study: the effect of first wetland restoration on ammonia concentrations

	ammonia	
	(1)	(2)
period = -5	-0.058***	-0.024
	(0.022)	(0.026)
period = -4	-0.066***	-0.022
	(0.018)	(0.023)
period = -3	-0.005	0.032
	(0.023)	(0.023)
period = -2	-0.028	0.012
	(0.030)	(0.029)
period = 0	-0.055***	0.0002
	(0.008)	(0.015)
period = 1	-0.041**	0.010
	(0.018)	(0.019)
period = 2	-0.045**	0.010
	(0.019)	(0.022)
period = 3	-0.029	0.020
	(0.019)	(0.021)
period = 4	-0.014	0.022
	(0.017)	(0.020)
period = 5	-0.013	0.026
	(0.030)	(0.032)
period = 6	-0.008	0.036
	(0.029)	(0.029)
period = 7	-0.017	0.020
	(0.030)	(0.030)
period = 8	-0.034*	0.004
	(0.020)	(0.022)
period = 9	-0.005	0.029
	(0.040)	(0.038)
period = 10	-0.001	0.045
	(0.031)	(0.028)
Dependent variable mean	0.13781	0.13346
Observations	121,933	14,257
Adjusted R^2	0.14013	0.25208
Ever treated		✓
HUC8 fixed effects	✓	✓
month fixed effects	✓	✓
year fixed effects	✓	✓

Clustered (HUC8-year) standard-errors in parentheses.
Signif. Codes: ***: 0.01, **: 0.05, *: 0.1

Table 7: Sun and Abraham event study: the effect of first wetland restoration on ammonia concentrations

	ammonia	
	(1)	(2)
period = -5	-0.046***	-0.056*
	(0.018)	(0.030)
period = -4	-0.059***	-0.061**
	(0.015)	(0.028)
period = -3	0.002	0.005
	(0.020)	(0.028)
period = -2	-0.021	0.008
	(0.026)	(0.029)
period = 0	-0.061***	-0.045**
	(0.009)	(0.023)
period = 1	-0.035**	0.002
	(0.017)	(0.023)
period = 2	-0.038**	0.0009
	(0.016)	(0.024)
period = 3	-0.021	0.019
	(0.016)	(0.024)
period = 4	-0.007	0.011
	(0.013)	(0.023)
period = 5	-0.014	0.003
	(0.027)	(0.033)
period = 6	-0.002	0.032
	(0.025)	(0.031)
period = 7	-0.014	0.003
	(0.020)	(0.027)
period = 8	-0.029**	-0.004
	(0.014)	(0.024)
period = 9	-0.004	-0.002
	(0.017)	(0.026)
period = 10	0.004	0.032
	(0.014)	(0.024)
Dependent variable mean	0.13781	0.13346
Observations	121,933	14,257
Adjusted R^2	0.15259	0.36798
Ever treated		✓
HUC8 fixed effects	✓	✓
month fixed effects	✓	✓
year fixed effects	✓	✓

Clustered (HUC8-year) standard-errors in parentheses.
Signif. Codes: ***: 0.01, **: 0.05, *: 0.1

Table 8: The effect of upstream restored wetlands and water quality on downstream ammonia concentrations

	ammonia	
	(1)	(2)
restored wetland (prop)	-0.983***	-2.05***
	(0.298)	(0.649)
upstream restored wetland (prop)	3.73	-0.191
	(2.64)	(0.985)
upstream ammonia	0.219***	0.163***
	(0.023)	(0.044)
Ever treated		✓
Dependent variable mean	0.13413	0.13907
Observations	17,638	1,722
Adjusted R^2	0.28622	0.39354
HUC8 fixed effects	✓	✓
year fixed effects	✓	✓
month fixed effects	✓	✓

Clustered (HUC8-year) standard-errors in parentheses.
Signif. Codes: ***: 0.01, **: 0.05, *: 0.1

Table 9: The effect of restored wetlands and other land use on ammonia concentrations

	ammonia			
	(1)	(2)	(3)	(4)
restored wetland	0.742	1.46	3.34**	2.72
	(2.06)	(1.80)	(1.40)	(2.75)
ppt	-7.08×10^{-5}***	-4.97×10^{-5}	-5.32×10^{-5}***	-5.9×10^{-5}*
	(1.63×10^{-5})	(4.66×10^{-5})	(1.77×10^{-5})	(3.05×10^{-5})
temp	-0.005***	-0.010***	-0.006***	-0.009***
	(0.0006)	(0.001)	(0.0007)	(0.001)
barren	-0.150*	-13.2***	-0.303*	-8.05
	(0.086)	(4.33)	(0.174)	(10.4)
cropland	0.266***	0.540***	0.237***	0.396***
	(0.022)	(0.082)	(0.028)	(0.102)
developed	0.271***	1.03***	0.291***	0.424**
	(0.026)	(0.192)	(0.032)	(0.180)
vegetation	0.113***	-0.149	0.148***	-0.149
	(0.020)	(0.123)	(0.025)	(0.179)
ice snow	0.161		0.809**	
	(0.243)		(0.340)	
water	0.124***	-2.02***	0.209***	-0.487
	(0.033)	(0.551)	(0.046)	(0.339)
distance to river (m)	1.06×10^{-5}*	-0.0004	-3.11×10^{-5}***	0.0002
	(6.44×10^{-6})	(0.0006)	(8.98×10^{-6})	(0.0002)
restored wetland × ppt	0.0010*	0.001**	0.0002	-5.56×10^{-5}
	(0.0005)	(0.0006)	(0.0004)	(0.0004)
restored wetland × temp	0.009*	0.021***	-0.004	0.003
	(0.005)	(0.005)	(0.006)	(0.006)
restored wetland × barren	154.3***	425.7***	133.0**	321.2**
	(53.1)	(84.8)	(64.5)	(139.2)
restored wetland × cropland	-0.149	-1.18	-4.23**	-3.94
	(2.64)	(2.34)	(1.88)	(3.38)
restored wetland × developed	4.50	5.27*	-1.78	3.41
	(2.88)	(3.19)	(2.53)	(3.86)
restored wetland × vegetation	-18.6***	-24.7***	-7.77*	-11.4
	(4.79)	(6.38)	(3.98)	(9.67)
restored wetland × water	-13.0**	-16.2***	-11.8**	-14.8
	(5.33)	(5.66)	(5.54)	(10.6)
restored wetland × distance to river (m)	-0.039	0.198*	0.016	0.108
	(0.034)	(0.113)	(0.024)	(0.082)
nitrogen facilities			0.002*	-0.009***
			(0.001)	(0.002)
animal units			7.48×10^{-7}	7.83×10^{-6}**
			(1.03×10^{-6})	(3.51×10^{-6})
restored wetland × nitrogen facilities			0.050	0.160***
			(0.034)	(0.053)
restored wetland × animal units			-6.28×10^{-6}	-5.56×10^{-5}**
			(3.07×10^{-5})	(2.3×10^{-5})
Ever treated		✓		✓
Years	1985-2018	1985-2018	2008-2018	2008-2018
Dependent variable mean	0.13781	0.13118	0.11784	0.10122
Observations	121,933	13,280	43,898	5,890
Adjusted R^2	0.14042	0.26939	0.18857	0.21825
HUC8 fixed effects	✓	✓	✓	✓
year fixed effects	✓	✓	✓	✓
month fixed effects	✓	✓	✓	✓

Clustered (HUC8-year) standard-errors in parentheses. Signif. Codes: ***: 0.01, **: 0.05, *: 0.1

Table 10: The effect of restored wetlands on phosphorus concentrations in a HUC12: Ever-treated HUC

	phosphorous					
	(1)	(2)	(3)	(4)	(5)	(6)
restored wetland (prop)	0.945***	-0.147	-0.041	-0.194	-0.231	-0.257
	(0.169)	(0.156)	(0.162)	(0.244)	(0.153)	(0.246)
Dependent variable mean	0.15232	0.15232	0.15232	0.15232	0.15232	0.15232
Observations	7,226	7,226	7,226	7,226	7,226	7,226
Adjusted R^2	0.27011	0.44446	0.49193	0.46965	0.48079	0.50916
year fixed effects	✓	✓	✓		✓	
month fixed effects	✓	✓	✓	✓		
HUC4 fixed effects	✓					
HUC8 fixed effects		✓		✓	✓	✓
HUC12 fixed effects			✓			
HUC4 x year fixed effects				✓		✓
HUC4 x month fixed effects					✓	✓

Clustered (HUC8-year) standard-errors in parentheses. Signif. Codes: ***: 0.01, **: 0.05, *: 0.1

Table 11: The effect of restored wetlands on phosphorus concentrations: All HUC

	phosphorous					
	(1)	(2)	(3)	(4)	(5)	(6)
restored wetland (prop)	0.186	0.487***	0.117	0.491***	0.387**	0.389**
	(0.125)	(0.158)	(0.162)	(0.157)	(0.163)	(0.162)
Dependent variable mean	0.14980	0.14980	0.14980	0.14980	0.14980	0.14980
Observations	81,587	81,587	81,587	81,587	81,587	81,587
Adjusted R^2	0.15216	0.28341	0.51682	0.30732	0.29906	0.32119
year fixed effects	✓	✓	✓		✓	
month fixed effects	✓	✓	✓	✓		
HUC4 fixed effects	✓					
HUC8 fixed effects		✓		✓	✓	✓
HUC12 fixed effects			✓			
HUC4 x year fixed effects				✓		✓
HUC4 x month fixed effects					✓	✓

Clustered (HUC8-year) standard-errors in parentheses. Signif. Codes: ***: 0.01, **: 0.05, *: 0.1

Table 12: Standard event study: The effect of first wetland restoration on phosphorous concentrations

	phosphorous	
	(1)	(2)
period = -5	0.120**	0.094**
	(0.058)	(0.043)
period = -4	0.053*	0.049**
	(0.031)	(0.022)
period = -3	0.043***	0.034*
	(0.016)	(0.018)
period = -2	0.042**	0.031**
	(0.017)	(0.014)
period = 0	0.023***	0.035**
	(0.007)	(0.015)
period = 1	0.011	0.025*
	(0.009)	(0.014)
period = 2	-0.0005	0.011
	(0.008)	(0.015)
period = 3	0.013	0.029*
	(0.013)	(0.018)
period = 4	0.038**	0.041**
	(0.017)	(0.018)
period = 5	0.044*	0.042*
	(0.023)	(0.023)
period = 6	0.010	0.015
	(0.016)	(0.019)
period = 7	0.050**	0.058***
	(0.020)	(0.022)
period = 8	0.061**	0.071**
	(0.029)	(0.031)
period = 9	0.055**	0.066**
	(0.027)	(0.027)
period = 10	0.011	0.006
	(0.018)	(0.022)
Dependent variable mean	0.14980	0.15511
Observations	81,587	8,708
Adjusted R^2	0.28448	0.35974
Ever treated		✓
HUC8 fixed effects	✓	✓
month fixed effects	✓	✓
year fixed effects	✓	✓

Clustered (HUC8-year) standard-errors in parentheses.
Signif. Codes: ***: 0.01, **: 0.05, *: 0.1

Table 13: Sun and Abraham event study: The effect of first wetland restoration on phosphorous concentrations

	phosphorous	
	(1)	(2)
period = -5	0.140***	0.111***
	(0.046)	(0.039)
period = -4	0.070**	0.061***
	(0.028)	(0.023)
period = -3	0.061***	0.047***
	(0.011)	(0.017)
period = -2	0.053***	0.037**
	(0.016)	(0.015)
period = 0	0.026***	0.032**
	(0.006)	(0.014)
period = 1	0.019**	0.016
	(0.008)	(0.012)
period = 2	0.009	0.016
	(0.007)	(0.013)
period = 3	0.023*	0.030*
	(0.012)	(0.018)
period = 4	0.045***	0.038**
	(0.014)	(0.016)
period = 5	0.044***	0.036**
	(0.017)	(0.016)
period = 6	0.011	0.007
	(0.012)	(0.014)
period = 7	0.055***	0.036**
	(0.015)	(0.017)
period = 8	0.064***	0.045*
	(0.021)	(0.023)
period = 9	0.058***	0.067***
	(0.021)	(0.023)
period = 10	0.010	0.033
	(0.014)	(102.4)
Dependent variable mean	0.14980	0.15511
Observations	81,587	8,708
Adjusted R^2	0.28954	0.42314
Ever treated		✓
HUC8 fixed effects	✓	✓
month fixed effects	✓	✓
year fixed effects	✓	✓

Clustered (HUC8-year) standard-errors in parentheses.
Signif. Codes: ***: 0.01, **: 0.05, *: 0.1

Table 14: The effect of upstream restored wetlands and water quality on downstream phosphorous concentrations

	phosphorous (1)	phosphorous (2)
restored wetland (prop)	-1.02*	0.456
	(0.563)	(0.635)
upstream restored wetland (prop)	-1.77	4.12*
	(1.13)	(2.12)
upstream phosphorous	0.361***	0.306***
	(0.032)	(0.062)
Ever treated		✓
Dependent variable mean	0.19324	0.15557
Observations	13,115	1,330
Adjusted R^2	0.49432	0.62451
HUC8 fixed effects	✓	✓
year fixed effects	✓	✓
month fixed effects	✓	✓

Clustered (HUC8-year) standard-errors in parentheses.
Signif. Codes: ***: 0.01, **: 0.05, *: 0.1

Table 15: The effect of restored wetlands and other land use on phosphorus concentrations

	phosphorous			
	(1)	(2)	(3)	(4)
restored wetland	-0.961	1.88	1.44	3.00
	(2.99)	(3.60)	(9.65)	(7.70)
ppt	1.52×10^{-5}	3.56×10^{-5}	1.87×10^{-5}	1.58×10^{-5}
	(1.86×10^{-5})	(3.5×10^{-5})	(2.08×10^{-5})	(3.56×10^{-5})
temp	-0.0009	-0.007***	-0.0008	-0.007***
	(0.0006)	(0.002)	(0.001)	(0.002)
barren	0.157	-0.552	-0.315	-3.53
	(0.106)	(0.916)	(0.221)	(2.27)
cropland	0.135***	1.07***	-0.075	1.08***
	(0.043)	(0.243)	(0.108)	(0.323)
developed	0.121***	2.51***	-0.180*	2.00***
	(0.042)	(0.502)	(0.097)	(0.653)
vegetation	-0.082*	1.54***	-0.382***	1.04**
	(0.042)	(0.302)	(0.112)	(0.419)
ice snow	-0.941***		-0.784***	
	(0.168)		(0.270)	
water	-0.122**	-0.711*	-0.238*	-1.42**
	(0.050)	(0.370)	(0.128)	(0.557)
distance to river (m)	$-7.93 \times 10^{-6***}$	0.001***	$-1.1 \times 10^{-5***}$	0.001***
	(2.62×10^{-6})	(0.0002)	(2.78×10^{-6})	(0.0003)
restored wetland × ppt	0.002**	0.003**	0.002**	0.003***
	(0.0010)	(0.001)	(0.001)	(0.001)
restored wetland × temp	-0.052***	-0.025**	-0.026**	-0.015
	(0.011)	(0.012)	(0.013)	(0.015)
restored wetland × barren	-294.6***	25.8	-339.7***	57.2
	(93.5)	(123.6)	(129.5)	(121.7)
restored wetland × cropland	3.60	-2.72	-2.47	-6.99
	(3.94)	(4.55)	(9.33)	(9.06)
restored wetland × developed	14.2*	1.68	24.0	17.6
	(8.24)	(5.44)	(21.2)	(14.5)
restored wetland × vegetation	-8.51**	-1.90	-1.31	4.61
	(3.87)	(4.31)	(9.81)	(7.80)
restored wetland × water	10.5	-3.77	2.32	-8.95
	(9.86)	(12.1)	(27.9)	(22.5)
restored wetland × distance to river (m)	-0.004	-0.008	-0.004	-0.012
	(0.005)	(0.008)	(0.007)	(0.012)
phosphorous facilities			-1.44×10^{-5}	-0.018*
			(0.005)	(0.009)
animal units			2.52×10^{-6}	3.25×10^{-6}
			(1.7×10^{-6})	(3.54×10^{-6})
restored wetland × phosphorous facilities			0.100	0.390**
			(0.129)	(0.185)
restored wetland × animal units			$-6.37 \times 10^{-5***}$	-4.05×10^{-5}
			(2.14×10^{-5})	(3.14×10^{-5})
Ever treated		✓		✓
Years	1985-2018	1985-2018	2008-2018	2008-2018
Dependent variable mean	0.14980	0.15232	0.12531	0.12601
Observations	81,587	7,226	20,092	2,422
Adjusted R^2	0.28377	0.44458	0.33341	0.53411
HUC8 fixed effects	✓	✓	✓	✓
year fixed effects	✓	✓	✓	✓
month fixed effects	✓	✓	✓	✓

Clustered (HUC8-year) standard-errors in parentheses. Signif. Codes: ***: 0.01, **: 0.05, *: 0.1