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

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Abstract: Billions of dollars have been dedicated to mitigating nonpoint source pollution from US agriculture. Wetlands provide an ex post natural solution by filtering sediments and excess nutrients from the landscape. 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 fund allocation for nonpoint source pollution abatement. We causally identify whether newly restored wetlands reduce nitrogen and phosphorus loads at the subwatershed level in the Mississippi River Basin. Results show that wetland easements reduce ammonia and total Kjeldahl nitrogen concentrations in the local subwatershed as well as downstream. A short-term increase in local phosphorus immediately following restoration likely reflects soil disturbance. While wetlands have no significant long-term impact on local phosphorus levels, downstream reductions are observed. We estimate that this federal investment in natural infrastructure leads to lower treatment costs for local water systems.

JEL Codes: L95, Q15, Q18, Q50, Q53

Keywords: agricultural pollution, wetland restoration, easement, nonpoint source, water quality, water treatment

NONPOINT SOURCE POLLUTION from agriculture poses challenges to water quality in the United States (Shapiro 2022). Nutrient runoff has been linked to detrimental outcomes such as contaminated drinking water, hypoxia outbreaks, and associated health

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Dataverse data: https://doi.org/10.7910/DVN/PWZKDP

Received April 26, 2024; Accepted September 11, 2025; Published online February 9, 2026.

Journal of the Association of Environmental and Resource Economists, volume 13, number 2, March 2026.
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https://doi.org/10.1086/739287

risks such as developmental and reproductive issues (Del Rossi et al. 2023). Agricultural nutrients are a major cause of the hypoxic zone in the Gulf of Mexico, which is now over 6,000 square miles (Hypoxia Task Force 2013). Excess nutrients in surface waters harm recreation, tourism, and commercial fishing.

Although the negative implications of pollution in our water systems are well documented, estimating the social cost of nutrient pollution has been notoriously difficult (Keiser 2019). There is significant variability in damage estimates from additional nutrients, primarily driven by location-specific differences in exposure, vulnerability, and population preferences (Keeler et al. 2016). In terms of mitigation spending, US investment to improve water quality has reached almost $2 trillion since 1960 (Keiser et al. 2019). Yet, cost-benefit analyses of water improvement policies frequently yield minimal benefits or even negative net benefits (Olmstead 2009; Keiser and Shapiro 2019). These low net benefits may be due to a lack of proper data and measurement error that lead to underestimation of the policy benefits as well as underestimation of the social cost of water pollution (Keiser 2019). Alternatively, policies may have inefficient designs: existing water quality policy has largely used expensive command-and-control regulation (Olmstead 2009). In this study, we look at the cost and benefits of water quality improvements obtained from a market-based instrument: wetland easements.

Quantifying the impact of wetland restoration on water quality in the Mississippi River Basin (MRB) is critical for optimal allocation of funds for nonpoint source pollution abatement.[1] Many reduction strategies aim to prevent runoff before it occurs by optimizing nutrient use (e.g., nutrient management plans) or changing land use (e.g., cover crops or buffer strips). Programs such as the Conservation Reserve (Enhancement) Program and the Environmental Quality Incentives Program largely address future nutrient loss. In the past two decades, US Department of Agriculture (USDA) conservation programs have received $6 billion in federal funding per year; the Agricultural Conservation Easement Program (ACEP), focused on wetland restoration, has consistently received the smallest share of funding ($500 million per year, or approximately 8%) (Wallander 2023).

Thank you to Wes Austin, Eric Edwards, Holly Gibbs, Corbett Grainger, Catherine Kling, Nick Parker, Dan Phaneuf, Andrew Stevens, and Adam Theising for excellent feedback. We thank the attendees of the 2023 AERE annual meeting, the 2023 Agricultural and Applied Economics Association annual meeting, the 2023 Social Cost of Water Pollution Workshop, the University of Wisconsin–Madison Agricultural and Applied Economics seminar, and the University of Illinois Agricultural and Environmental seminar for their helpful comments. Funding was provided by the National Great Rivers Research and Education Center. Any errors or omissions are the sole responsibility of the authors. Edited by Mary Evans.

1. The region we study is referred to as both the Mississippi River Basin (MRB) and the Mississippi Atchafalaya River Basin (MARB) in the existing literature; we choose the convention MRB.

Wetlands are unique among the potential solutions as they provide an ex post natural solution to filter out nutrients already in our soils and water systems (De Steven and Lowrance 2011). Wetlands slow the flow of surface water through a landscape, which allows wetland plants to 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. This ability, one of the many ecological benefits they provide, has earned wetlands the name "nature's kidneys." However, their effect on water quality beyond a micro-watershed level is indeterminate.

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.[2] The WRP, first implemented 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 conditions. The easement program is an example of a market-based tool that leverages incentives and moves beyond the usual command-and-control type policies.

We estimate the effect of newly restored wetlands on nitrogen and phosphorus loads at the subwatershed level in the MRB. We accomplish this by combining over 30 years of water quality data with easement-level data on the WRP. 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 that wetland easements have significantly reduced ammonia and total Kjeldahl nitrogen (TKN) in surface water across the MRB. It takes on average three years after restoration for these water quality improvements to materialize, but our findings suggest that these benefits last for longer than a decade. Using an event study developed by Sun and Abraham (2021) that is robust to heterogeneous treatment effects over time, we find that the first wetland restoration in a HUC12 decreased ammonia concentration by 0.08 milligrams per liter (mg/L), or 62% of the sample mean ammonia concentration.[3] TKN decreased by 0.20 mg/L (37% of the sample mean). Using an ordinary least squares (OLS) model with time and watershed fixed effects, we find that a 100% increase in restored wetland acres in a HUC12 decreased ammonia concentration by 0.004–0.015 mg/L and TKN by 0.010–0.060 mg/L across a range of specifications with valid pretrends. We also find evidence that a 100% increase in upstream wetlands decreases downstream ammonia concentration by 0.006–0.011 mg/L and downstream TKN by 0.009 (not significant)–0.044 mg/L.

2. We refer to these programs in conjunction as WRP for brevity.

3. HUC12 is a 12-digit Hydrologic Unit Code that identifies the smallest watershed unit in the national Watershed Boundary Dataset.

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 at the subwatershed level. Notably, we find a small but statistically significant increase in phosphorus in the initial years after restoration. However, this effect peaks four years after restoration, and we find no evidence that a restored wetland increases net phosphorus over its lifetime.

The overall null result we find for phosphorus is also consistent with the broader literature on solutions for nonpoint source phosphorus pollution (Liu and Wang 2022). Due to its ability to bind with soil, phosphorus remains in the ecosystem for decades as so-called legacy phosphorus. Phosphorus levels in surface water respond slowly due to persistent legacy sources. Additionally, legacy phosphorus is vulnerable to soil disturbance, as occurs during wetland restoration and subsequent reflooding (Aldous et al. 2005). Wetlands appear to play a role in trapping phosphorus within subwatersheds, potentially contributing to downstream water quality improvements.

We value the reduction in ammonia using a cost-benefit analysis by building on the findings of Mosheim and Ribaudo (2017) and data on drinking water treatment costs for public water systems (US EPA 2006a). Restoring 100 additional acres of wetland in a subwatershed has the potential to decrease treatment costs for a small, medium, and large public water system by $1,289, $2,072, and $17,258 per year. The largest absolute benefits occur for water utilities with high operating costs servicing large populations and those sourcing surface water from highly polluted areas. In terms of relative benefits, small systems experience the largest percentage change in operating costs from improved nitrogen levels. Initial wetland investments may pay off via reduced drinking water costs within 20–30 years when targeted effectively, such as to agricultural regions. We find that the initial federal program investment transfers drinking water cost savings to local communities. Based on a snapshot of wetlands restored as of 2018, we estimate that these wetlands generate annual benefits of $200 million at an initial investment of $6 billion.

This study has three main contributions: (1) estimating the ecosystem services of restored wetlands at a broad scale, (2) identifying an effective solution to nonpoint source pollution in surface waters, and (3) providing an estimate of the nonmarket value of wetlands.

We contribute the first study on the water quality impacts of restored wetlands over a broad area and over an extended period of 30 years.[4] Previous studies of the relationship between wetlands and water quality have been site specific (e.g., Jenkins et al. 2010;

4. Restored wetlands are also referred to as manmade, constructed, or artificial wetlands. Notably, nutrient removal is only one of many ecosystem services that wetlands provide, including flood mitigation (Taylor and Druckenmiller 2022), wildlife habitat (Jenkins et al. 2010), improved agricultural productivity (Karwowski 2022, 2023), and removal of other pollutants such as heavy metals (Gill et al. 2017) and pesticides (Tournebize et al. 2013). We focus on nutrient pollution as the external influences and ecological processes differ across the ecosystem services. Also, nutrient removal is relatively understudied at scale.

Hansen et al. 2015) or employ model simulations (e.g., Cheng et al. 2020). These studies have shown that wetlands have oversized impacts on water quality relative to their local area. Other studies focus on the performance of restored wetlands relative to natural wetlands. Al-Isawi et al. (2017) and Zhang et al. (2023) show that restored wetlands can be effective under the right conditions, while Strayer et al. (2003) and Powers et al. (2014) find that restored wetlands have no effect at the watershed level. We build upon this literature by employing a data-driven econometric approach, which demonstrates that wetland restoration can be effective in reducing specific types of nutrient pollutants at scale.

Second, we add to the growing literature on nonpoint source pollution policy. Efforts to address nonpoint source pollution are typically voluntary, and evidence of their success has been mixed, highlighting the challenge of regulating nonpoint source pollution and achieving real reductions in nutrient loss (Ribaudo and Shortle 2019; Del Rossi et al. 2023). Yet the scale of the problem is such that significant efforts are being launched to address nonpoint source pollution, often with high direct or indirect costs. Strategies focus on improving nutrient use (Paudel and Crago 2020; Skidmore et al. 2023), increasing conservation practices (Liu and Wang 2022; Hsieh and Gramig 2023), and land retirement. We find evidence that wetland restoration (and by extension wetland conservation) is a worthwhile incentive-based instrument to decrease nonpoint source pollution.

Finally, we contribute to the extensive literature on the nonmarket valuation of wetlands. The estimated values for wetlands are highly variable and dependent on wetland size, type, conditions, nearby land use, and the beneficiaries of these services.[5] The closest related valuation method to ours is based on replacement costs, or the cost of providing traditional water treatment should the wetland no longer exist or function properly. Kazmierczak (2001) finds savings for municipal wastewater plants ranging from $5 to $9,985/acre/year in Florida and Louisiana. Breaux et al. (1995) estimate cost savings for wastewater treatment of $850–$1,316/acre/year in Louisiana. We estimate that the value of restoration ranges from $13 to $173/acre/year by translating water quality improvements to reduced expenditure on drinking water treatment for public water systems using previous estimates from Mosheim and Ribaudo (2017).

Understanding the value of wetland protection is especially timely in light of the recent *EPA v. Sackett* ruling, which narrowed federal protection on wetlands (Keiser et al. 2022). Wetlands had been protected under the Clean Water Act as Waters of the United States. Wetlands that are not adjacent to navigable waters are no longer protected. Only wetlands that have a continuous surface connection to water features including streams, oceans, rivers, and lakes are protected under the Clean Water Act. This recent ruling lowers the cost of developing on wetland ecosystems and puts them at risk of an

5. See Moeltner et al. (2019) and Turner et al. (2008) for a more comprehensive review.

increased rate of development. Research that clarifies the value of wetlands has the potential to contribute to the ongoing discussion concerning the definition of Waters of the United States (Shapiro 2022).

1. BACKGROUND

1.1. Wetland and Land Use Policy in the United States

The United States has experienced wide-scale wetland loss. Between 1780 and 1980, over 100 million acres of wetlands were drained and filled for development (Dahl 1990). A history of wetland policy can be found in appendix A (apps. A–G are available online).

The Wetland Reserve Program (WRP), created in 1990, incentivizes wetland restoration. Under a wetland easement contract, a landowner agrees to withdraw the eased land from all farming, ranching, and forestry practices while retaining ownership of the land. In exchange, the landowner receives a payment based on appraised value, regional rate cap, or voluntary offer—typically averaging $2,700/acre.[6]

Eligibility depends on land and landowner characteristics. The parcel must be a farmed or converted wetland or a riparian area linking protected wetlands. The landowner must have owned the land for seven years, meet specified income eligibility thresholds, and comply with the highly erodible land and wetland conservation provisions. Program enrollment is limited to no more than 10% of a county's farmable acreage.

The Natural Resources Conservation Services (NRCS) branch of the USDA implements all aspects of the program. Wetland easement applications are evaluated by the state-level NRCS departments.[7] The NRCS assigns applicant parcels a score based on the ecosystem benefits they provide. The most highly ranked applicants are selected for the program. NRCS staff restore the land to its natural wetland condition by planting native species, removing tiling, and building berms.

Wetland restoration costs between $170 and $6,100 per acre, with an average of $650 per acre, based on an analysis of nominal wetland restoration payments completed from 1990 to 2012 (Hansen et al. 2015). Costs fall into two categories: the opportunity cost of production on the land and the expenses from restoring the wetland ecosystem. The largest driver of wetland easement cost is the land's current value, which is based on potential profitability.[8] The engineering cost to restore the wetland varies by the size of the project, flora and fauna reintroduction, and level of construction needed to alter the

6. For easements valued at less than $500,000, the payment is made in a lump sum or no more than 10 annual installments. For easements over $500,000, payments are made in 5 to 10 annual installments.

7. Each state was responsible for its own ranking criteria until 2022, when the federal NRCS implemented the national Conservation Assessment Ranking Tool.

8. This opportunity cost can also include the nuisance cost of altering farm production to maneuver around the wetland (i.e., fuel, time, and fencing).

hydrology (Hansen et al. 2015). Larger wetlands require additional resources to restore. Restoring grass wetlands may only require drill-seeding, whereas forested wetlands require more labor-intensive tree planting. Filling channels, eliminating tile draining, and restoring flow impediments are usually less costly than a full excavation. Reintroducing beavers to a riparian habitat provides a low-tech, and low-cost, strategy to building wetlands, though it removes some control from the restoration staff and landowner (Brazier et al. 2021).

The WRP has received funding from all Farm Bill allocations since 1990. In 2014, it was enveloped in the Agricultural Conservation Easement Program (ACEP).[9] ACEP funding is dependent on Farm Bill budget lines. From 1992 to 2019, the five-year budgets for ACEP were $1.7, $3.3, $4.9, $2.2, and $2.3 billion, respectively (Wallander 2023). In the 2023 Inflation Reduction Act, an additional $19.5 billion dollars were allocated to NRCS climate smart agriculture programs. ACEP received $1.4 billion on top of the 2018 Farm Bill baseline funding. In 2022, there were 3 million acres enrolled in wetland easement programs. Enrollment is strongly correlated with available funding.

Wetlands exist in various forms, and the dominant wetland type varies regionally, each varying in its capacity to filter nutrients. The NRCS practice standards stipulate that the restored wetland be in the same hydrogeomorphic class and same vegetative modifier as the historic wetland (NRCS 2024). Notably, conversations with NRCS scientists highlighted that, in practice, many formerly forested systems are restored to herbaceous vegetation. We do not know the form of each restored wetland in our sample, but a 2019 Fish and Wildlife report to Congress (Lang et al. 2024) breaks down freshwater wetlands in the United States by type: forested (47.5%), emergent (27.2%), shrub (17.3%), ponds (6.2%), and farmed (1.8%). Forested and emergent wetlands are particularly well suited for nutrient filtration because of their dense vegetation, shallower water, and high microbial activity. Based on this, it is likely that NRCS wetlands are being restored to types that are particularly effective nutrient filters.

1.2. The Relationship Between Wetlands and Nutrient Pollution

Wetland easements improve water quality in two stages: first, by retiring land from production and reducing agricultural inputs; second, by removing existing nutrients from the water system. Unlike other strategies, wetlands offer an ex post natural solution by filtering nutrients already in water systems.

Wetlands preserve water quality through both physical and chemical processes (Kostel 2023). These processes differ for nitrogen and phosphorus and also vary by nutrient form (i.e., particulate vs. dissolved, organic vs. inorganic). There is evidence that

9. 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.

wetlands are more effective at reducing nitrogen pollution than phosphorus (Fisher and Acreman 2004). Figure B1 (figs. B1, B2, D1, D2, F1–F6 are available online) shows the cycles that nitrogen and phosphorus follow in a wetland.

Rainfall washes nutrients from upstream lands into surface waters; wetlands then retain and filter nutrients through sediment trapping, soil retention, plant uptake, and microbial processing (Fisher and Acreman 2004). Nitrogen and phosphorus settle on the wetland floor in a process called sedimentation. 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 most 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 depend on season, temperature, precipitation, and water flow.

Empirical estimates of how wetlands filter nutrients largely stem from site-level studies. A study of 203 wetland sites by Land et al. (2016) finds that wetlands remove an average of 93 grams of nitrogen and 1.2 grams of phosphorus per square meter per year. These rates can be higher for artificial wetlands that optimize nutrient removal. The watershed-level estimates are less consistent. Researchers have found a wide range of reductions in nitrogen under the right conditions (Al-Isawi et al. 2017; Zhang et al. 2023) or no significant relationship (Strayer et al. 2003; Powers et al. 2014). Inadequate control for other factors that affect water quality, such as seasonality and surrounding land cover, is a likely source of variation in the estimated effects (Hansen et al. 2018).

We expand upon the existing scientific literature by employing econometric techniques to derive causal estimates over a broader time frame and geographic scale. Analyses of a limited number of sites over shorter periods offer the advantage of precise measurement, monitoring, and direct measurement of other factors influencing water quality. However, they often rely on one-way comparisons across space or across time that may be prone to bias.[10] Additionally, this approach is impractical for large-scale assessments, such as those covering the MRB. We see our findings as a valuable complement to the existing body of

10. For example, if wetlands were restored in an effort to counteract worsening water quality, estimates from a before-and-after study would be biased toward zero. In contrast, a wetland restoration may occur in an area where other conservation practices were simultaneously adopted, leading to overestimation of the wetland's effectiveness. By using a two-way comparison across both space and time, we account for some of these sources of bias.

research. Estimating the impacts of restored wetlands on a broad scale is crucial for the effective allocation of future conservation funds. Without such assessments, the significance of USDA-restored wetland easements may be underestimated.

2. METHODS

2.1. Data

We use data on the locations and contract information of all wetland easements in WRP or ACEP since program inception (1990–2018). Wetland restoration grew steadily over the period (fig. B2). Wetland easements are concentrated in areas with higher nutrient concentrations and abundant agricultural activity (e.g., the Corn Belt). Figure 1 maps the density of wetland easements in 2018 and the average nutrient concentration over the period. At the end of 2018, there were just over 9,000 permanent wetland easements in the MRB, accounting for approximately 1.7 million acres or 0.22% of land in this region.

We pair this with site-monitor water quality data from the Standardized Nitrogen and Phosphorus Dataset (SNAPD). These data provide harmonized measures of nutrient readings at the station level; details on this process can be found in appendix C and in Krasovich et al. (2022). We use surface water readings from five years prior to program inception to the end of the SNAPD sample (1985–2018). The SNAPD water quality data has readings for over 30 nitrogen- and phosphorus-based nutrient compounds. We focus our analysis on three main nutrients: total phosphorus (filtered), ammonia (filtered), and total Kjeldahl nitrogen (filtered), which we refer to as TKN. These three compounds have many observations, which increases the likelihood that the sample is representative of the study region. Compounds with fewer readings may be biased toward areas with frequent sampling due to water quality concerns or may simply be "noisier" as the sampled subwatersheds change from year to year.[11]

Filtered ammonia, TKN, and phosphorus are the most pertinent compounds for an analysis of the economic benefits of wetlands. A filtered reading only includes the nutrient's aqueous form while an unfiltered reading also contains its particulate form. The literature finds some evidence that the aqueous (more soluble) parts of a nutrient are more responsive to wetlands, as these are more readily transformed through chemical and biological processes (Fisher and Acreman 2004). Moreover, ammonia and TKN are measured and managed by water treatment plants because of their important role in environmental quality. Unlike other nitrogen compounds, which cause indirect harm through eutrophication, ammonia (a key compound in TKN measures) has a direct effect at elevated levels. When ammonia is present in surface waters at high

11. We replicate our main results on a set of additional nutrient compounds in app. D. This exercise gives a broader picture of the relationship between wetlands and the various chemical forms of nitrogen and phosphorus. However, we find evidence that these samples are not representative.

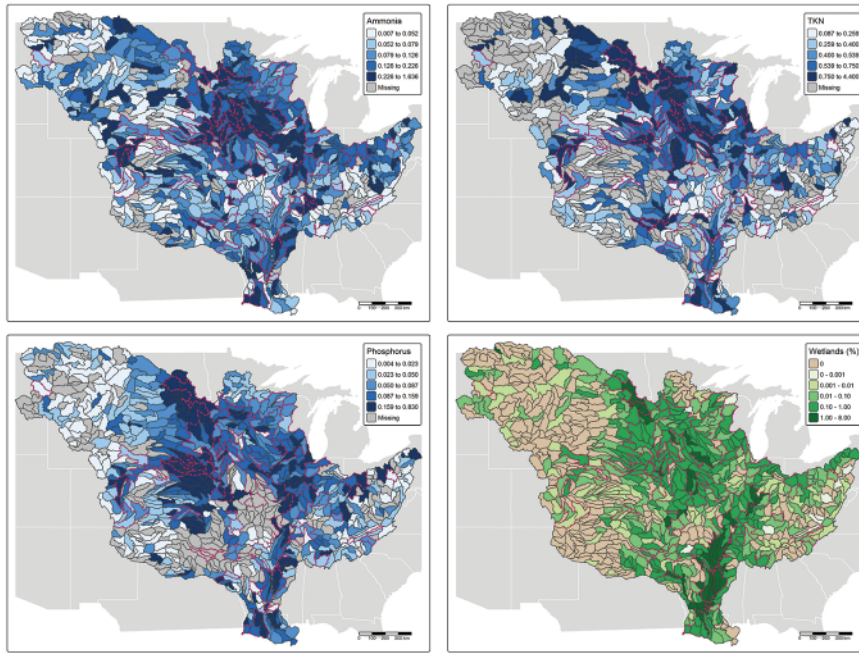


Figure 1. Maps of the HUC8 in the Mississippi River Basin showing average nutrient levels and total restored wetlands. Nutrient concentrations are the mg/L averages of filtered readings of ammonia, total Kjeldahl nitrogen, and phosphorus taken from 1980–2018. The wetland map shows the percentage of the HUC8 with a restored wetland in WRP at the end of 2018. The dashed maroon line outlines the units in our ever-treated sample that have both restored wetlands and water quality readings.

concentrations, it can lead to toxic buildup and death in aquatic organisms (US EPA 2013). Water treatment plants face requirements specific to these compounds and commonly measure ammonia and TKN in their systems (US EPA 2024).

We use data from the US Geological Survey (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 to 2021. We use the categories cropland, developed, grass/shrub, tree cover, and water, which are likely to affect the total nutrient availability and the likelihood of runoff in the surrounding region. We aggregate grass/shrub and tree cover to a native vegetation category.

We also control for other sources of nutrients. The EPA provides information on the release of point source emissions through the Toxic Release Inventory Program since 1987. We control for the pounds of ammonia and phosphorus emitted by permitted facilities in a HUC12-year. While there are measures for total chemical releases (combining

air, water, and land emissions), we find that surface water discharges are most relevant for water quality.

Weather data come from PRISM. We calculate monthly precipitation and mean temperature at the HUC12 level, averaging across pixels. Aggregation reduces error from spatial and interpolation-based correlation (Dell et al. 2014; Hsiang 2016).

The National Hydrography Watershed Boundary Dataset delineates upstream and downstream relationships between HUC12s. When a HUC12 has multiple upstream HUC12s, we calculate an average of upstream water quality and treatment using monthly streamflow as weights (LaFontaine and Blodgett 2022). Further details on data-cleaning procedures are described in appendix C.

2.2. Sample

Our study covers the Mississippi River Basin, which is characterized by high agricultural production, nonpoint source pollution, and harmful algal blooms. This region encompasses 1,245,000 square miles, or 40% of the conterminous United States. Our main unit of analysis is a subwatershed (HUC12) in a distinct month-year. We organize our data in an unbalanced panel. A HUC12 appears in the data only in months in which there was a water quality reading. Not all water quality readings have all nutrient concentrations, so our sample sizes vary by nutrient type.

Summary statistics of the relevant variables are available in table 1. The average nutrient levels in our sample are 0.138 mg/L for ammonia, 0.630 mg/L for TKN, and 0.150 mg/L for phosphorus. Land under wetland restoration is minimal; the average subwatershed only has 27 acres of restored wetland, or 0.10% of land area. Among HUC12s that ever have a wetland restoration, the average is nearly 10 times higher, at 248 acres. The average subwatershed is 26,000 acres and can range from 4,000 to 1 million acres in size. We further explore the differences between the set of HUC12 with a WRP restoration (ever treated) and those without (never treated) in section 3.

2.3. The Impact of Wetland Easements on Subwatershed Water Quality

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

First, we use a continuous difference-in-differences model to investigate the influence of easements on water quality in the surrounding subwatershed. The disparate nature of nonpoint source pollution may lead to impacts at a watershed scale, so it may be appropriate to think of treatment at these aggregate levels (Paudel and Crago 2020; Liu and Wang 2022). This also allows us to study effects of marginal changes in wetland area. We estimate:

$$Y_{imt} = \beta \text{Wetland}_{imt} + \Gamma \mathbf{X}_{imt} + \alpha_w + \delta_m + \lambda_t + \epsilon_{imt}. \quad (1)$$

Our outcome, Y_{imt} is the water quality in HUC12 i in month m and year t. In our primary specification, we use ammonia, TKN, or phosphorus concentrations in mg/L,

Table 1. Summary Statistics and Normalized Differences of HUC12 in the Mississippi River Basin Based on Treatment Status

	All (1)	Ever Treated (2)	Never Treated (3)	N. Difference (4)	*p*-Value (5)
Ammonia	.138	.133	.138	−.010	.104
	(.341)	(.340)	(.341)		
Total Kjeldahl nitrogen	.630	.538	.643	−.120	<.001
	(.687)	(.505)	(.708)		
Phosphorus	.150	.155	.149	.017	.021
	(.258)	(.222)	(.262)		
Restored wetlands (acres)	26.544	247.212	.000	.321	<.001
	(263.433)	(769.285)	(.000)		
Restored wetlands (prop)	.001	.010	.000	.310	<.001
	(.011)	(.031)	(.000)		
Restored wetland (0/1)	.046	.425	.000	.859	<.001
	(.209)	(.494)	(.000)		
Total precipitation (mm)	85.100	87.095	84.860	.026	<.001
	(62.137)	(61.298)	(62.233)		
Average temp (C°)	13.606	13.108	13.666	−.040	<.001
	(9.633)	(9.961)	(9.591)		
HUC12 acres	25,823.514	29,200.166	25,417.347	.159	<.001
	(19,005.896)	(13,732.261)	(19,505.291)		
Percent agland	.406	.589	.384	.574	<.001
	(.294)	(.196)	(.297)		
Percent developed	.100	.073	.103	−.147	<.001
	(.174)	(.101)	(.181)		
Percent grass/shrub	.126	.043	.136	−.349	<.001
	(.234)	(.109)	(.243)		
Percent treecover	.262	.147	.276	−.412	<.001
	(.269)	(.143)	(.277)		
Water	.038	.039	.038	.013	.002
	(.085)	(.053)	(.089)		
TRI ammonia	291.675	319.816	288.290	.002	.641
	(18,232.169)	(6,464.508)	(19,166.909)		
TRI phosphorus	.108	.000	.121	−.008	.003
	(15.033)	(.000)	(15.911)		
Observations	175,624	18,857	156,767		
N HUC12	9,102	348	8754		

Note. This table summarizes key statistics for subwatersheds that are in our full sample, the sample of subwatersheds that ever include a wetland restoration (ever treated), as well as those that never have a wetland restoration (never treated). We find the normalized difference between the ever treated and never treated samples (estimated as the difference in means over the square root of the sum of variances) and report the associated *p*-values.

though we also take an inverse hyperbolic sine (IHS) transformation of concentration for robustness (Bellemare and Wichman 2020). We estimate the model separately for each nutrient compound because we believe there are heterogeneous effects by compound. Feigenberg et al. (2023) show that models including interaction terms that account for treatment-effect heterogeneity with only a subset of variables may introduce omitted variables bias into a model. This is addressed by estimating separate models for each group, as we do here.

We measure treatment in Wetland_{imt}, as the IHS of the total land area in HUC12 i dedicated to wetland easements in month m and year t.[12] The interpretation of the coefficient β is the change in water quality following a one-unit increase in wetland restoration, or a 100% increase in wetland easement acreage. This is a reasonable example in our context, as we observe the program since inception and thus have numerous instances of 100% increases (or more) of restored area in the sample. We prefer an IHS transformation of the treatment variable as we believe that there are nonlinear impacts of wetland area and specifically that the marginal benefits decrease with project size. We consider treatment to be the time after a wetland easement is considered restored by the NRCS, rather than when the wetland easement contract is signed or when the producer stops cropping on the land. There are exogenous factors that influence the process of wetland restoration, as it is a time-intensive process that can take years to achieve and is dependent on specialized contractors to complete the project (Karwowski 2022). Additionally, this allows us to disentangle the initial effect of the land retirement from the ecological function of the wetland.

We control for subwatershed characteristics that affect water quality in $\mathbf{X}_{imt}$. We include land cover (percentage HUC12 area in crop agriculture, developed land, water, and natural vegetation), weather (average temperature and total precipitation), and point source pollution (Toxics Release Inventory [TRI] emissions). In some specifications, we also include the area of restored wetlands upstream of the HUC12 of interest to evaluate downstream water quality impacts. We limit to one subwatershed upstream for empirical simplicity and based on evidence from the scientific literature that wetland benefits attenuate quickly, though exact distance of downstream benefits is heterogeneous (Forgrave et al. 2024).

Weather conditions may alter the effectiveness of a wetland in removing nutrients from a system (Land et al. 2016). To explore this, we modify equation (1) to interact our treatment variable, Wetland_{imt} with the temperature and precipitation in the subwatershed, $\mathbf{X}_{imt}$. We construct bins based on the distribution of both variables, with weather bins defined in appendix F.

Similarly, surrounding land use influences the nutrient exposure that a wetland faces. Following Feigenberg et al. (2023), we split the sample into four groups by agricultural

12. We do not account for whether the wetland is located upstream or downstream of the water quality station within the subwatershed, which may bias our results toward zero.

exposure and estimate the model separately, thereby allowing each subsample specific estimates of all terms, including treatment effects. We use cropped area as a proxy for nutrient exposure.[13] Current nutrient levels are influenced by current wetland area, and less than a third of HUC12s have a nutrient reading from before the start of the program.

We estimate the model with either HUC8 or HUC12 fixed effects. HUC8 fixed effects control for characteristics of the broader watershed that are consistent across time. This includes hydrologic, geologic, or climate characteristics that remain consistent during the time period. It may be sufficient to control for the broader watershed (i.e., HUC8) rather than the subwatershed (i.e., HUC12) for two reasons. First, HUC8 cover a single watershed, or "the geographic boundary of a drainage divide." Thus, by construction, many of the time-invariant characteristics that affect water quality are shared within a HUC8. Second, including HUC8 fixed effects effectively allows for comparison within a HUC8 rather than only within a HUC12.

The MRB has 35,105 HUC12 units, though only 8,969 HUC12 (895 HUC8) have a relevant water quality reading. Of these, 344 HUC12 (170 HUC8) also have a wetland easement. By controlling for time-invariant watershed factors, we can leverage cross-sectional variation to identify treatment effects. However, HUC12 fixed effects precisely control for all unobserved factors at our unit of observation. The inclusion of these fixed effects controls for 40%–50% of variation in water quality but may be important to address sources of endogeneity.

Similarly, we define time fixed effects λ_t two ways. In two specifications, we include year fixed effects, which account for sample-wide phenomena that occur in a given year. This could include large-scale weather trends (e.g., a year with wide-scale flooding) or changes in national policy. As an alternative, we also estimate a model with HUC4-by-year fixed effects. This allows us to control for time trends specific to a smaller region.

We estimate the model with HUC8 and year, HUC8 and HUC4-by-year, and HUC12 and year fixed effects. We face a trade-off between controlling for more unobserved factors that may introduce endogeneity while also leaving sufficient variation in the outcome variable unexplained. The model with HUC8 and year fixed effects controls for the fewest unobservable factors, while the other two models account for more unobservable unit or time factors.

We estimate standard errors that are robust to clustering at the HUC8-year level. Given the nature of watersheds, we expect that water quality is related within a HUC8, including through downstream treatment effects, so we employ standard errors that reflect the idea that treatment assignment is correlated at the level of the HUC8 (Abadie et al. 2023).

13. We define bins as 0%–25% of land area in cropped agriculture (low), 25%–50% (medium), 51%–75% (medium-high), and 75%–100% (very high).

In order for β to be an unbiased measure of the true treatment effects, treated and control HUC12 must have been on parallel trends in absence of the restored wetland. In our sample of ever-treated HUC12, this assumes that the timing of restoration was unrelated to changes in water quality. If, instead, easement contracts were more likely to be approved and subsequent restoration occurred after a change (either for better or for worse) in water quality, this would bias the estimator. We empirically test for evidence of pretrends using an event-study framework.

Our estimator could also be biased if wetland restoration occurred concurrently with other activities that affect future water quality. For example, this would be the case if wetland restoration occurred simultaneously with widespread adoption of agricultural best management practices by farmers in the watershed. This assumption cannot be empirically tested, as we do not observe counterfactual water quality and cannot gather data to rule out all possible activities that may have occurred simultaneously. However, we believe that this is not a major threat for several reasons. First, wetland easement contracts are a two-sided decision by both landowners and USDA officials. Second, restoration is carried out by NRCS agents, and the amount of time to complete a restoration varies greatly across projects. We believe that this introduces variation such that, even if a landowner decided to apply for WRP as one of a set of actions that improve water quality, we would still be identifying off of quasi-random timing in the restoration of the wetland.

2.4. Parallel Trends and Timing of Filtration

We use an event study to empirically test whether HUC12 were on parallel trends prior to the restoration of a new wetland. Additionally, the event-study framework allows us to investigate how long after restoration a wetland has a significant impact on water quality. We estimate:

$$Y_{imt} = \Sigma_{k=-5,k\neq-1}^{10}(\beta_k \text{Wetland}_{imt} \quad \mathbf{1}\{K_{imt} = k\}) + \Gamma \mathbf{X}_{imt} + \alpha_w + \delta_m + \lambda_t + \epsilon_{imt}. \tag{2}$$

Here, Wetland_{imt} is a binary indicator for the presence of a restored wetland easement in subwatershed i in a given month, m, and year, t. We denote the years since (or until) restoration using k, and a HUC12 has a value of 1 for K_{imt} when it is k years from restoration. Thus, the coefficients β_k represent annual event-time estimates relative to the restoration year, with the year immediately prior to restoration ($k = -1$) omitted as the reference category. We estimate (2) using the full sample (i.e., all HUC12 in the MRB) 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 the main text, we report the event studies using our least restrictive set of fixed effects (i.e., HUC8 for α_w and year for λ_t) as this model controls for the fewest potential sources of endogeneity, though we also estimate the model with HUC12 fixed effects for robustness.

We estimate the equation with and without the vector of control variables, X_{imt}. Though equation (1) includes controls and, therefore, requires the relatively weaker assumption of conditional parallel trends, we use the unconditional models to confirm that a stricter assumption (i.e., unconditional parallel trends) is valid in our setting. We again cluster standard errors at the HUC8-year level.

A challenge in our setting is that a HUC12 may have multiple wetlands restored in the period.[14] We estimate the event-study model with only the first restored wetland in a HUC12. We also pool the indicators for more than five years before the first treatment or more than 10 years after treatment to avoid cases with multiple restored wetlands separated by a large gap. Thus, our event study effectively tests whether HUC12s 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. This also lends to a different interpretation of the results of equation (2) compared to equation (1). The former only estimates the marginal effect of the first wetland restoration, while the latter estimates the total effect of all restored wetland area in a given time period.

This event study is vulnerable to bias due to heterogeneous treatment effects over time (Sun and Abraham 2021), particularly in the full MRB sample with never-treated units. For this reason, we estimate equation (2) using ordinary least squares and the method of Sun and Abraham (2021). We use the cohort of subwatersheds that are treated in the final year, 2018, as the reference group.[15]

3. RESULTS

3.1. Placement of Wetland Easements Is Endogenous to Land Use and Water Quality Trends

First, we report the summary statistics of HUC12s that ever receive a wetland easement (ever treated) and those that never receive one (never treated) in table 1. We find large differences between subwatersheds with and without wetlands, and we report normalized differences in column 4. The ever-treated sample has significantly more agricultural activity; 59% of total land area in the average ever-treated HUC12 is in agriculture, compared to 38% in the never-treated sample. Ever-treated HUC12s also have significantly less tree cover (15% vs. 28%) and are slightly less developed (7% vs. 10%).

14. Of the 344 treated HUC12, there are 81 HUC12 (24%) with more than one wetland restoration project. In places with multiple wetland easements, the median time difference between projects is 1 year with a mean difference of 2.2 years. In these cases with two projects restored in the same year, we consider the two projects to be one effective "restoration event" and still interpret the model as measuring the effect following the first restoration. This could potentially bias our posttreatment event-study coefficients upward.

15. We also remove the 2018 year data from our estimation data in this specification.

These results demonstrate that the placement of a wetland easement is not random, as easements are targeted to land that has been in agricultural use. We focus on the ever-treated sample for our main analyses, as this provides a set of HUC12s that are comparable in terms of many of the activities that are linked to water quality outcomes. Limiting our analysis to the ever-treated sample tests the response of water quality to wetland easements using variation in the timing of the establishment of wetland easements, rather than whether the easement was ever established. Thus, subwatersheds serve as "control" units in the periods before their first easement is established during the time they are "not yet treated."

We use an event study to explore the nutrient trends before and after a subwatershed's first wetland restoration project (fig. 2). This tests whether our sample design uses suitable counterfactual units for treated subwatersheds at their time of treatment. Using the Sun and Abraham model with controls in our ever-treated sample, we find no significant differences in the pre-period for ammonia, TKN, or phosphorus.[16] We also report the full set of coefficients from the Sun and Abraham model and from a standard fixed-effect model (fig. E1), the same models without controls (fig. E2), and a standard fixed-effect model with an IHS-transformed outcome variable (fig. E3).

These results are largely consistent when we include HUC12 fixed effects with some meaningful exceptions (fig. E4).[17] TKN was slightly lower three years before restoration in the model with controls (significant at the 10% level) and significantly lower in both the three and four years before restoration in the model without controls. However, these estimates do not exhibit a clear pattern (e.g., there is no consistent downward trend leading up to the restoration date).

When we include all the subwatersheds, we see that ammonia, TKN, and phosphorus levels differed between treatment and control units in the years before the first restoration. Ammonia levels were lower in treated units in the earliest window (i.e., five or more years pre-restoration) as well as two years before restoration. TKN levels were lower in treated units in all periods before treatment. Phosphorus levels were higher in treated units in the four and five or more years before the first restoration. These results confirm that a sample of HUC12s that ever receive treatment provides a better counterfactual. The lack of pretrends suggests that the effects on ammonia, TKN, and phosphorus in the ever-treated sample may be interpreted causally.

16. We replicate fig. 2 for alternative compounds in fig. D1 and fig. D2.

17. We find that ammonia concentrations were significantly higher five or more years before the first restoration, though this may be largely attributed to the earlier window. We find no significant differences from four years prior to treatment until three years after the restoration has been completed. We also find weak evidence (significant at the 10% level) that phosphorus dropped two years before the restoration; this may be due to the transition out of crop production after contract closing.

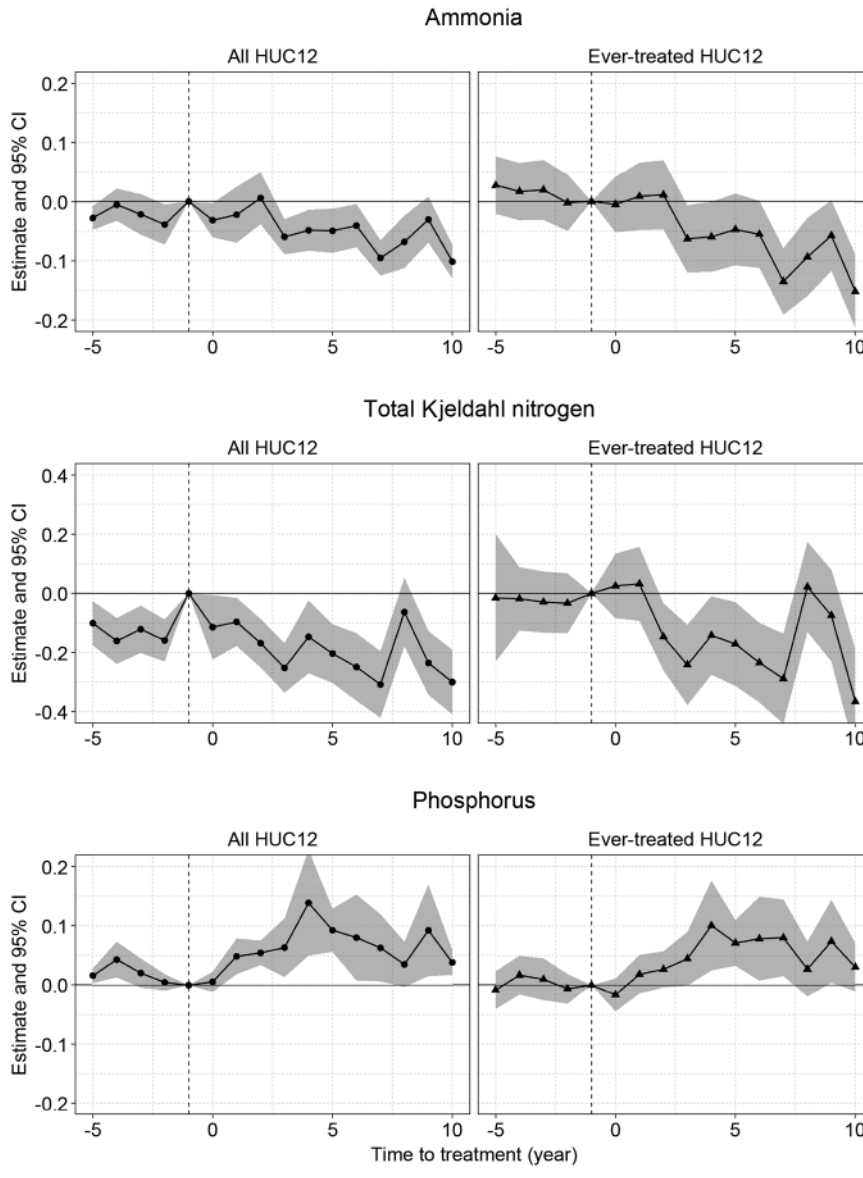


Figure 2. Effect of first wetland easement restoration on HUC12 nutrient concentrations using a Sun and Abraham (2021) event-study model with controls for the full sample and ever-treated sample. Observations are at the HUC12-month-year level. Models include HUC8, year, and month fixed effects and controls for percentage of HUC12 area in cropland, developed land, water, and natural vegetation, average temperature, total precipitation, and TRI emissions. Confidence intervals shown are at the 95% level.

We find a lag between the time a wetland is considered fully restored and when the effects are fully realized. This lagged response of water quality improvement projects is common for conservation practices, habitat restoration, and nutrient management (Meals et al. 2010; Tomer and Locke 2011). Wetlands are no exception and can take years to mature and become fully functional (Gutrich and Hitzhusen 2004; Al-Isawi et al. 2017). On average, it takes two to three years after restoration before we observe decreases in ammonia and TKN. Effects increase over the period, showing that water quality improvements from wetland restorations are long-lasting. Ammonia falls further over time, from a 0.06 mg/L reduction in year 3 to 0.15 mg/L in year 10 and beyond. Similarly, TKN decreases by 0.22 mg/L three years after restoration and 0.36 in year 10 and beyond.

We explore this further by defining the treatment timing using contract closure rather than wetland restoration (fig. E5). In the early years after the contract is closed, this tests whether our results are driven by the direct effect of removing land from production, since a producer can no longer crop after contract closing. We find no significant reduction in ammonia until the seventh year, compared to year 4 in the event study considering wetland restoration completion (fig. 2). The significant effects we detect in year 7 onward are after most restorations are completed (average time from closing to restoration is two to three years). We find no significant impact on TKN in any year. For phosphorus, levels are unchanged until 10 or more years after contract closure, when they significantly decrease.

This suggests that our results are largely driven by the ecological function of the wetland rather than the immediate impact from land retirement. Notably, we estimate the impact of restoration compared to a counterfactual where the land remained in agricultural production. Alternatively, the land could have been abandoned or converted to another land use. In our study region, most abandoned cropland becomes grassland or pasture (Xie et al. 2024). A benefit comparison to these alternatives is beyond the scope of this study, but we note that forested land has also been shown to improve downstream water quality (Caldwell et al. 2023).

We estimate the average post-restoration treatment on the treated (ATT) from the event study with controls (table 2). As expected based on the single-year coefficients, we find a negative and significant effect of restoration on ammonia and TKN. In our preferred model (col. 1), ammonia concentration falls by 0.079 mg/L (compared to a sample mean of 0.13) and TKN falls by 0.20 mg/L (compared to a sample mean of 0.53) on average in the years after the first wetland restoration in the HUC12.

Notably, we find a modest increase (0.039 mg/L average or 23% of the mean value) in phosphorus following the first wetland restoration in a HUC12 (table 2). This effect peaks in the fourth year after restoration, and there are no significant effects 10 or more years after restoration (fig. 2). Though demonstrating the exact mechanism is beyond the scope of this study, we believe it may be due to soil disturbance during restoration and reflooding, which could release legacy phosphorus into the water system (Aldous

Table 2. Average Effect (ATT) of First WRP Restoration from Sun and Abraham Event Study With and Without Controls

	Ammonia		Total Kjeldahl Nitrogen		Phosphorus	
	(1)	(2)	(3)	(4)	(5)	(6)
ATT	−.079***	−.078***	−.206***	−.194***	.039**	.024*
	(.025)	(.025)	(.063)	(.065)	(.016)	(.012)
No. HUC12	299	299	178	178	240	240
Dependent variable mean	.13459	.13459	.54119	.54119	.15552	.15552
Observations	13,756	13,756	2,953	2,953	8,579	8,579
Adjusted R^2	.37473	.36960	.48264	.48042	.40709	.37710
Controls	✓		✓		✓	

Note. Observations are at the HUC12-month-year level. Only HUC12 that are ever treated are in this sample. Models include HUC8, year, and month fixed effects and controls for percentage of HUC12 area in cropland, developed land, water, and natural vegetation, average temperature, total precipitation, and TRI emissions. The Sun and Abraham (2021) methodology is used to account for staggered timing in treatment. The not-yet-treated cohort of 2018 is used as the control group. Standard errors are clustered at the HUC8-year. WRP = Wetland Reserve Program.

* $p < .10$.

** $p < .05$.

*** $p < .01$.

et al. 2005). While this estimate is informative, particularly given the evidence that this precise specification delivers causal estimates, it does not capture the net effect of all restored wetlands. We rely on the pooled effects for such an answer.

3.2. Wetlands Filter Nutrients from Broader Surface Water Systems

We find that wetland easements reduce nitrogen-compound concentrations in the surrounding subwatersheds after restoration. We report the range of coefficients from models with three sets of fixed effects: HUC8 and year, HUC8 and HUC4-by-year, and HUC12 and year. We find that a 100% increase in wetland restoration land area decreases ammonia concentrations by 0.004–0.015 mg/L relative to a sample mean of 0.13 mg/L of ammonia (table 3, cols. 1–3). For TKN, we find a 0.03–0.06 mg/L decrease relative to a sample mean of 0.54 mg/L when we include HUC8 fixed effects. When we include HUC12 fixed effects, we find a 0.010 mg/L increase in TKN, though we interpret this result with caution due to the significant coefficients in the pretreatment period for the event study of TKN with HUC12 fixed effects (fig. E4).

Next, we include upstream restored wetlands to test whether there is a meaningful impact on downstream water quality (table 3, cols. 4–6). For ammonia, the estimates of the impact of wetlands within their own watershed are largely unchanged, with decreases in ammonia between 0.004 and 0.014 mg/L in response to a 100% increase in wetland easement area. We find evidence that upstream wetlands reduce ammonia

Table 3. Effect of Current WRP area (IHS acres) on HUC12 Water Quality (mg/L) in HUC12 That Ever Had a WRP Restoration

	(1)	(2)	(3)	(4)	(5)	(6)
	A. Ammonia					
IHS restored wetland (acres)	−.010***	−.015***	−.004*	−.009***	−.014***	−.004**
	(.002)	(.003)	(.002)	(.002)	(.003)	(.002)
IHS upstream restored wetland (acres)				−.011***	−.007***	−.006**
				(.003)	(.002)	(.003)
No. HUC12	302	302	302	255	255	255
Dependent variable mean	.13346	.13346	.13346	.13320	.13320	.13320
Observations	14,257	14,257	14,257	12,466	12,466	12,466
Adjusted R^2	.25191	.41669	.40463	.27828	.43757	.43156
	B. Total Kjeldahl Nitrogen					
IHS restored wetland (acres)	−.025***	−.060***	.010***	−.025***	−.055***	.009**
	(.009)	(.015)	(.003)	(.009)	(.015)	(.004)
IHS upstream restored wetland (acres)				−.009	−.044***	.004
				(.006)	(.016)	(.005)
No. HUC12	178	178	178	161	161	161
Dependent variable mean	.53813	.53813	.53813	.53445	.53445	.53445
Observations	3,058	3,058	3,058	2,786	2,786	2,786
Adjusted R^2	.35684	.43130	.57472	.33154	.41721	.55772
	C. Phosphorus					
IHS restored wetland (acres)	.0006	−.003	.001	.002	−.002	.002
	(.002)	(.003)	(.001)	(.002)	(.003)	(.001)
IHS upstream restored wetland (acres)				−.003	−.007**	−.004**
				(.002)	(.003)	(.002)
No. HUC12	240	240	240	212	212	212
Dependent variable mean	.15511	.15511	.15511	.15608	.15608	.15608
Observations	8,708	8,708	8,708	8,138	8,138	8,138
Adjusted R^2	.33252	.33908	.47555	.35369	.37245	.47944
HUC8 fixed effects	✓	✓		✓	✓	
HUC12 fixed effects			✓			✓
Year fixed effects	✓		✓	✓		✓
HUC4 × year fixed effects		✓			✓	

Note. Observations are at the HUC12-month-year level. Only HUC12s that are ever treated are in this sample. Models include month fixed effects and controls for percentage of HUC12 area in cropland, developed land, water, natural vegetation, and average temperature, total precipitation, and TRI emissions. Standard errors are clustered at the HUC8-year. IHS = inverse hyperbolic sine; WRP = Wetland Reserve Program.

* $p < .10$.

** $p < .05$.

*** $p < .01$.

levels downstream by a similar magnitude to those in the HUC12 with the restoration (0.006–0.011 mg/L).

For TKN, we again find that increasing restored wetlands decreases TKN in the same watershed (0.025–0.055 mg/L) when we control for HUC8 fixed effects. In these models, we also find a 0.009 (not significant)–0.044 mg/L decrease in downstream TKN. When we include HUC12 fixed effects, we again find a small increase (0.009 mg/L) in TKN in HUC12 with restored wetlands with no significant effect on downstream TKN.

We do not find a significant effect of wetlands on average HUC12 phosphorus concentration (table 3, cols. 1–3). We find evidence that upstream wetlands decrease downstream phosphorus concentrations by 0.003–0.007 mg/L (cols. 4–6), though this effect is not significant in the model with HUC8 and year fixed effects. When controlling for upstream wetland restorations, we again find no significant effect of phosphorus in the same HUC12. The largely insignificant change in phosphorus that we highlight here also mirrors challenges previously highlighted by Liu and Wang (2022) in their analysis of the EQIP program and Skidmore et al. (2023) in their analysis of county ordinances governing nutrient management. While nitrogen (including in its ammonia form) is water soluble and travels easily through the watershed, phosphorus particles attach to rocks and sediments, making phosphorus slow moving. "Legacy phosphorus," or previously applied phosphorus that is bound to soils, may affect water quality for decades and challenges efforts to estimate short-term impacts on phosphorus. Moreover, our results may reflect the different processes through which wetlands store and release phosphorus compared to nitrogen.

Next, we explore heterogeneous impacts. We estimate the model separately based on cropped area in the HUC12 in figure 3 and table E1 (tables D1, E1, F1–F6 are available online). Notably, not all coefficients are significant across all specifications; we discuss only significant coefficients. Our coefficients increase in magnitude as the percentage of the HUC12 in cropped agriculture increases. In a HUC12 with low crop agriculture, a 100% increase in restored wetland area reduces ammonia by 0.003–0.004 mg/L in ammonia. This increases to a 0.004–0.007 mg/L reduction for medium agriculture HUC12s, 0.011–0.013 for medium high, and 0.020 for very high HUC12. Average ammonia concentration increases with cropped agriculture; the relative effect sizes are relatively stable or modestly increasing. The coefficients range from 5% to 6% for low, 4% to 7% for medium, 9% to 11% for medium high, and 8% for HUC12s with very high agricultural activity.

We again find a modest effect of wetland restoration on phosphorus levels. We find no significant coefficient in HUC12s with low, medium, or very high levels of crop agriculture. We do find negative and significant reductions (0.004–0.012 mg/L) in phosphorus in HUC12s with medium-high agriculture. These equate to 3%–8% reductions in phosphorus.

We report the coefficients for the models of TKN, though we interpret these results cautiously due to relatively few observations and limited numbers of HUC12 per

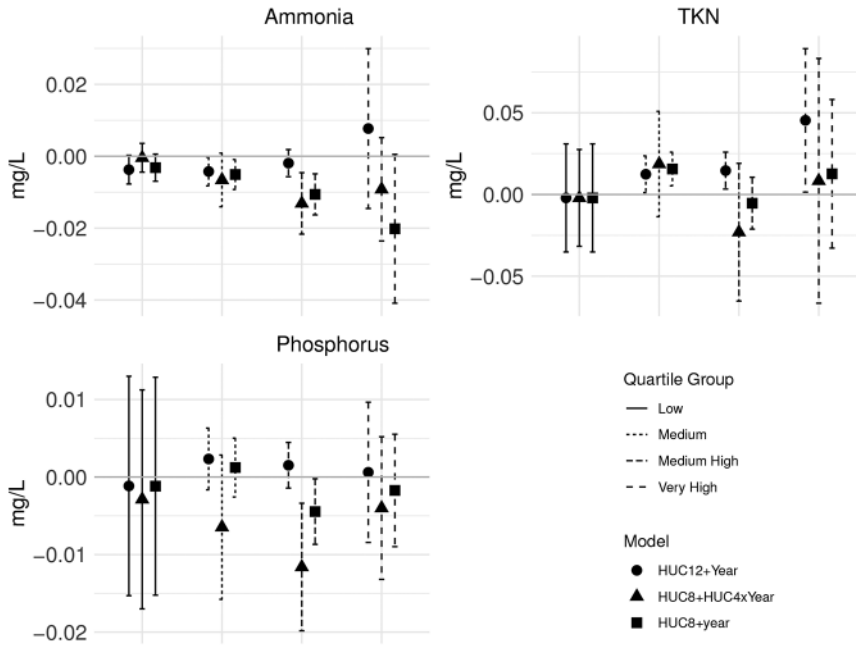


Figure 3. Impact of inverse hyperbolic sine (IHS) wetland area on nutrient concentration by percentage of area in the HUC12 dedicated to cropland. Quartiles are defined as low (0%–25% of area), medium (25%–50%), medium high (50%–75%), and very high (>75%). Only HUC12s that are ever treated are in this sample. Models include HUC8, year, and month fixed effects, and controls for percentage of HUC12 area in cropland, developed land, water, and natural vegetation, average temperature, total precipitation, and TRI emissions. Confidence intervals are shown at the 95% level.

category. We find positive and significant effects in medium and very high HUC12s, though these samples have 1,114 observations across 43 HUC12s and 522 observations across 70 HUC12s, respectively.

We do not attribute causality to the difference in coefficient size across agriculture categories, as agricultural activity itself is endogenous to water quality. We note, however, that the idea that wetlands remove more nitrogen in areas with greater agricultural activity is in line with the scientific literature. Hansen et al. (2018) find that wetlands remove more nitrogen when they receive more nitrogen, while Ross and McKenna (2023) find that the filtration rates of wetlands increase with cropped area. Our findings, taken with the existing evidence, suggest that situating wetlands within agricultural hubs may be an efficient approach for addressing nitrogen concerns.

We estimate a series of robustness tests to address the complexity of the relationship between wetland restoration and water quality in appendix F. First, we include a direct

measure of upstream water quality on a limited sample (table F1). Additionally, we include results from the sample of all HUC12s in the MRB and test alternative fixed effects (table F2). We explore the role of weather in wetland function (table F3). Table F4 uses alternative control groups with the Sun and Abraham estimator. Finally, we test alternative functional forms of the treatment and outcome variable (table F5, table F6). Our results are largely robust to these alternative specifications. We find consistent evidence that nitrogen (measured as ammonia or TKN) falls with wetland restoration and little evidence of an average effect on phosphorus.

3.3. Cost-Benefit Analysis

Nonmarket valuation of improvements in surface water quality is vital to better understand the costs and benefits of programs like the WRP, yet a disconnect remains between many analyses of policy impact and the current valuation literature. Most policy evaluations estimate impact in terms of scientific indicators, such as nutrient concentration in surface water, but nutrient concentration is rarely the focus in valuation studies (Del Rossi et al. 2023). Instead, many studies consider biophysical measures of water quality such as Secchi depth (Ji et al. 2020), the biological condition gradient (Vossler et al. 2023), or harmful algal bloom events (Wolf et al. 2022). Linking the impact of nutrient reductions on these valued outcome measures is challenging, as the relationships are nonlinear and rely on many other complex factors influencing the water body. One notable exception is the literature that links source water quality to avoided drinking water treatment expenditure (Mosheim and Ribaudo 2017; Price and Heberling 2018, 2020; Cullmann et al. 2024). This approach offers a clearer pathway for translating biophysical outcomes into monetary terms in our setting, helping to quantify tradeoffs in policy-relevant terms.

Managing high levels of nitrogen is costly for water treatment facilities.[18] Nitrate standard violations have been a persistent issue in community water systems, with approximately 4,000 violations recorded between 1982 and 2015 (Pennino et al. 2017). The Safe Drinking Water Act sets a maximum contaminant level for nitrogen levels in drinking water to protect against human health effects (e.g., blue baby syndrome) (Knobeloch et al. 2000). Nitrate removal treatments are only implemented when concentrations persist above regulatory limits. Treatment requires expensive technologies such as ion exchange, reverse osmosis, electrodialysis, and centralized biological

18. There is no enforceable regulatory limit for phosphorus in drinking water, as there is no evidence of a direct link from phosphorus levels to human health. Phosphorus is not monitored at all treatment facilities, and the financial impact of phosphorus is comparatively lower (US EPA 2024). A meta-analysis by Price and Heberling (2018) finds that a 1% increase in nitrogen load correlates with a 0.06% rise in drinking water treatment costs, whereas a similar increase in phosphorus load results in a 0.02% cost increase. This discrepancy underscores differing regulatory and financial priorities between nitrogen and phosphorus management.

treatment (Jones et al. 2007; Rauh and Hughes 2024). These high costs may force treatment plants to invest in new technologies or suffer financial losses from decreased water sales (Dunlap et al. 2015). As a result, nitrate levels have remained one of the top three violations of safe drinking water standards for over two decades (Pennino et al. 2017).

An analysis by Mosheim and Ribaudo (2017) considers the impact of nitrogen concentration in source water on water treatment costs. They estimate how variable treatment costs respond to nitrogen levels, with elasticities varying by plant size and ambient nitrogen.[19] These elasticities range from 0.0007 in a region with low nitrogen (and thus low risk of surpassing regulatory standards) to 0.05 for small utilities or those in regions with high nitrogen.[20] Notably, the elasticities are the same for increases or decreases in the amount of nitrogen in the source water; we make the same assumption in our cost-benefit analysis. A recent paper by Cullmann et al. (2024) emphasizes the impact of source water contamination on total cost in addition to variable cost, as utilities may incur fixed capital costs. As we only consider the effect on variable cost, we likely underestimate the total cost faced by water treatment utilities.

We pair these elasticities with the most recently reported average variable expenditure by public water system size (US EPA 2006a, 2006b) to calculate the marginal effect of adding a wetland restoration on a system's variable treatment costs. We calculate payback periods and break-even cost thresholds for wetland investments. We then use average estimates of the cost per customer and the cost per million gallons of water to provide rough estimates of the break-even customer base and water-processing level for each plant and nutrient-level category.

We estimate the break-even levels based on an 8% reduction in ammonia per 100% increase in restored wetland area (table 3, col. 1). We focus on the results of the model with HUC8, year, and month fixed effects. We believe this specification controls for potential sources of endogeneity (as evidenced by the event study) while retaining sufficient variation for identification. We assume that a 100% increase is on average equivalent to 100 acres of restoration as this roughly approximates the restoration of a second large wetland. The sample median project was 80 acres, and the sample average total restored area was 248 acres.

19. Mosheim and Ribaudo (2017) categorize utilities by size according to the population they serve: fewer than 10,000 people (small), between 10,000 and 100,000 (medium), and more than 100,000 (large). Low, medium, and high nitrogen refers to the amount of nitrogen in the source water. Within these groups, an average of 0.04 mg/L, 0.21 mg/L, and 2.73 mg/L must be removed, respectively.

20. Cullmann et al. (2024) find nationwide elasticities of groundwater nitrate on drinking water variable cost in Germany that are at the upper end of the elasticities in Mosheim and Ribaudo (2017) (0.048–0.052).

We also assume a one-to-one relationship between nitrogen and ammonia and a 10-year lifetime per wetland.[21] We consider these to be conservative estimates in each case. Notably, we find that wetlands have the largest impact 8–10 years after restoration in the dynamic model, suggesting that a wetland's productive lifespan exceeds 10 years. We report variable cost, marginal variable cost reduction, and break-even variable costs for small, medium, and large facilities and medium-sized facilities in a HUC12 with low, medium, and high nitrogen levels in table 4.

Nitrogen reductions yield the highest relative declines in variable costs for small water treatment plants or plants whose source water has high levels of nitrogen. In either case, a 1% reduction in nitrogen yields a 0.05% reduction in variable costs (Mosheim and Ribaudo 2017). For any small utility, a 100-acre wetland restoration will decrease utility variable cost expenditure by 0.004% (or $1,289). For a medium-sized utility in a HUC12 with high nitrogen levels, a wetland easement project reduces treatment expenditure by 0.004% or approximately $17,265.

For small treatment plants or plants drawing from sources with high nitrogen, the cost of a 100-acre wetland restoration is offset by a reduction in variable cost once annual variable costs exceed $8.13 million per year. Mosheim and Ribaudo (2017) report a cost per million gallons of water around 1995$1,800 across firms (2024$3,642), suggesting that firms will break even at 900–1,000 million gallons of water treatment per year. This is far below the mean treatment level in the Mosheim sample (10,678 million gallons per year) and the more recent mean treatment level for surface water sources of 2.17 million gallons per year (US EPA 2006b, table 10).

Absolute benefits increase with utility size. Even though the elasticity between nitrogen levels and variable costs is low (0.0002%) for large utilities (not in high-nitrogen HUC12), the high processing volume and total variable costs ($70 million annually) increase the total value of water quality improvements. As expected, however, the break-even annual variable cost is significantly higher for large utilities ($135 million) or those with low nitrogen levels in their source water ($580 million), as both of these experience far lower cost reductions per unit reduction in nitrogen. However, our findings of wetland impact on surface water may not extrapolate to highly populated areas or areas with low levels of pollution.

We graph the number of years it would take to pay off the wetland restoration through drinking water treatment savings for each public water system type (small, medium, and large) as well as for a medium system in a high, medium, and low nitrogen setting (fig. 4). The shortest payback period of 20 years exists for large public water systems and medium-sized systems sourcing water from surface waters with a high level of nitrogen. The other payback periods are greater than 100 years, though even relatively

21. In models using HUC8 fixed effects, we estimate a similar reduction relative to the mean for ammonia and TKN (8%–11% vs. 5%–11%). We rely on the ammonia estimates, as we have the largest and most representative sample for these readings.

Table 4. Impact of Wetland Easements on Water Treatment Utility Variable Costs (VC) Based on Marginal Impacts from Mosheim and Ribaudo (2017) Under Different Utility Scenarios

	Marginal VC Reduction				Break-Even Values			
Setting	Variable Cost	Elasticity	WRP VC (%)	WRP VC ($)	Payback Period	Annual VC	M Gallons per Year	Population Serviced
Small PWS	312,943	.05	.0041	1,289	260	8,131,364	914	13,906
Medium PWS	4,190,771	.006	.0005	2,072	162	67,761,364	9,464	144,047
Large PWS	69,816,914	.003	.0002	17,258	19	135,522,727	21,649	329,513
High nitrogen	4,190,771	.05	.0041	17,265	19	8,131,364	1,136	17,286
Medium nitrogen	4,190,771	.004	.0003	1,381	243	101,642,045	14,196	216,070
Low nitrogen	4,190,771	.0007	.0001	242	1386	580,811,688	81,119	1,234,687

Note. All estimates assume restoration costs based on a 100-acre wetland, net contract and restoration costs of $3,350 per acre, wetland lifetime of 10 years, average cost per million gallons of $1,800, and average water consumption of 180 gallons per person per day. Variable costs are based on public water system (PWS) size, except in the comparison of nitrogen levels which uses the average medium utility costs. WRP = Wetland Reserve Program.

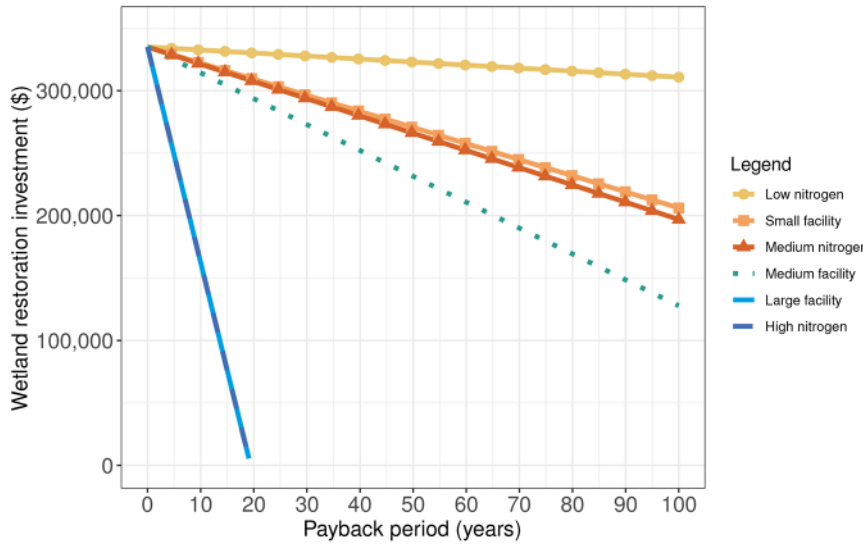


Figure 4. Number of years to pay off the HUC12 wetland restoration investment through reduced water treatment expenditure by public water system (PWS) size and nitrogen concentration. The marginal change in variable costs for water treatment utilities is based on elasticities from Mosheim and Ribaudo (2017), variable cost estimates from US EPA (2006b), and the estimated wetland effect on ammonia (table 3, col. 1).

small cost savings can be meaningful for small local governments. We also estimate only the impact on variable cost, which ignores the potentially high capital cost of adopting new treatment technology.[22]

We map the potential and current value of wetland restoration on water quality in figures 5 and 6.[23] First, we estimate the local value of restored wetlands based on HUC8-level nutrient levels and the nutrient-defined elasticities from Mosheim and Ribaudo (2017). We assume that the effects we find at the HUC12 level are the same for a HUC8. In figure 5*A*, we estimate the potential effect of an additional 100-acre easement assuming the HUC8 has one medium-sized public water system and varying ammonia levels. A 100-acre increase in wetland easements corresponds to an annual

22. An average restoration for small- or medium-sized utilities sourcing from low- or medium-nitrogen surface water would break even in 20 years with avoided capital costs of $293,563–$330,165. This is the low end of the capital expenses that small- and medium-sized utilities report paying for improved treatment technology (US EPA 2006b).

23. The lifetime value of the program is beyond the scope of this study. It would require a panel dataset on public water system locations, which are not readily available.

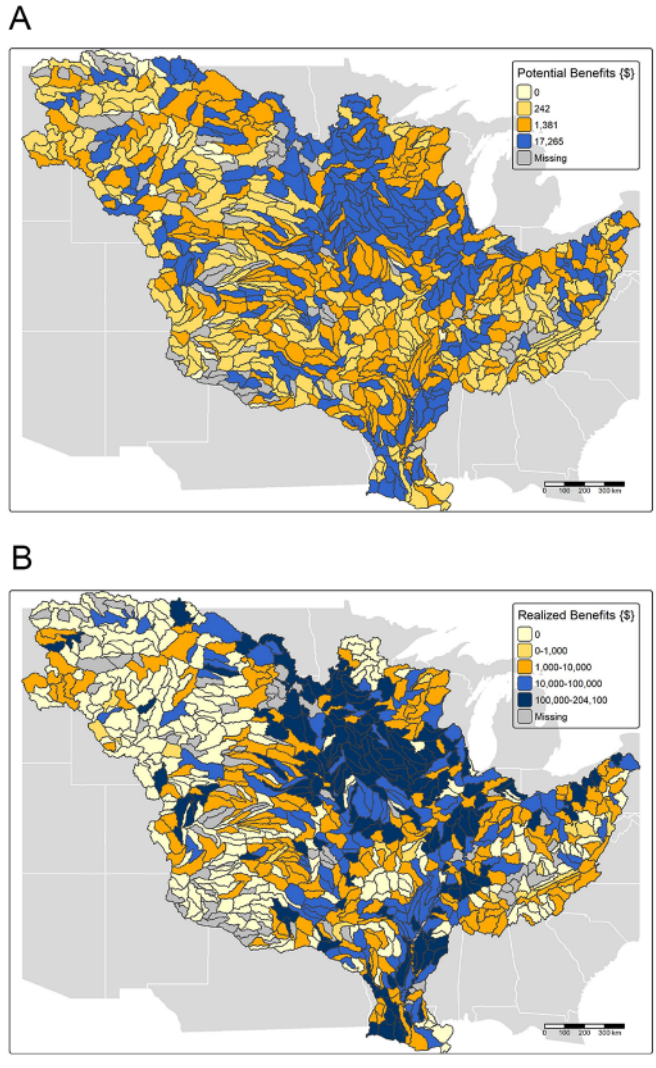


Figure 5. Estimated annual treatment cost savings from wetland restorations by watershed nutrient level: comparison of benefits from a hypothetical 100-acre restoration and actual NRCS restorations since 2018. *A*, Potential benefits from a 100-acre wetland restoration by nutrient level. *B*, Realized benefits from total wetland restorations since 2018 by nutrient level. Benefits are estimated as cost savings = wetland area × nitrogen reduction × treatment cost elasticity. We use elasticities by nitrogen levels from Mosheim and Ribaudo (2017): 0.0007 for low nitrogen, 0.004 for medium nitrogen, and 0.05 for high nitrogen. We assume that HUC12-level wetland effect (0.08) generalizes to HUC8-level benefits and that the HUC8 has one medium-sized public water system. In panel *A*, we estimate the benefit from a 100-acre change in wetland area. In panel *B*, we estimate the benefit based on the total area of restored wetlands in 2018. Missing values correspond to HUC8 with no ammonia readings, and 0 values correspond to HUC8 without a public water system.

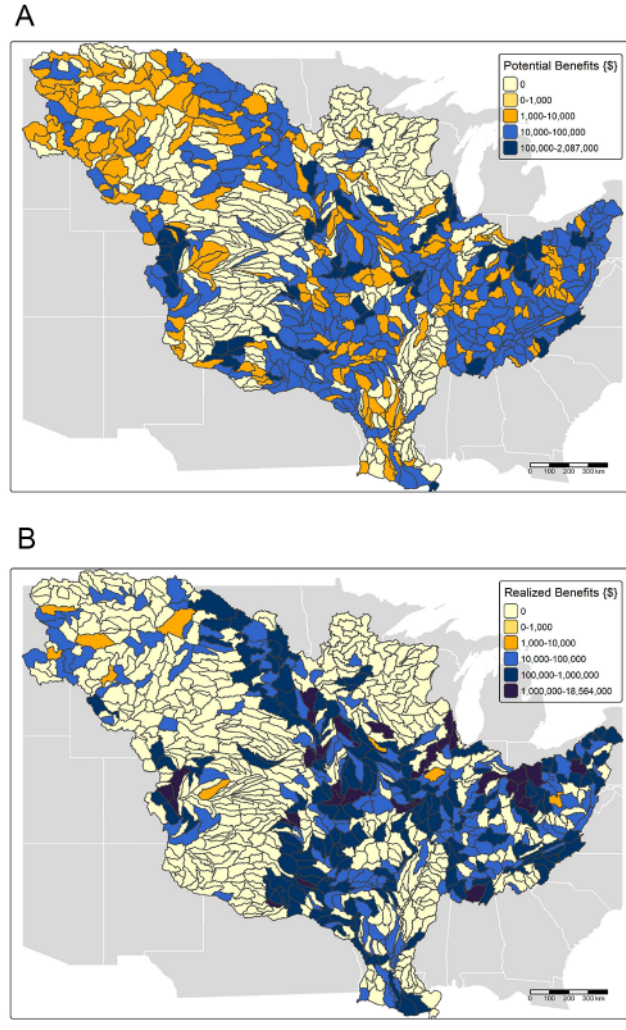


Figure 6. Estimated annual treatment cost savings from wetland restorations by public water system size: comparison of benefits from a from a hypothetical 100-acre restoration and actual NRCS restorations since 2018. *A*, Potential benefits from a 100-acre wetland restoration by public water system size. *B*, Realized benefits from total wetland restorations since 2018 by public water system size. Benefits are estimated as cost savings = wetland area × nitrogen reduction × treatment cost elasticity. We use elasticities by public water system size from Mosheim and Ribaudo (2017): 0.003 for large systems, 0.006 for medium systems, and 0.05 for small systems. We count the number of active small, medium, and large systems within each HUC8 that rely on surface water. We assume the HUC12-level wetland effect (0.08) generalizes to HUC8-level benefits for medium and large systems with low or medium nitrogen levels. In panel *A*, we estimate the benefit from a 100-acre change in wetland area. In panel *B*, we estimate the benefit based on the total area of restored wetlands in 2018. Missing values correspond to HUC8 with no ammonia readings, and 0 values correspond to HUC8 without a public water system.

reduction of $242, $1,381, and $17,265 in system costs in HUC8 with low, medium, and high ammonia levels, respectively. In figure 5*B*, we estimate the current annual value of program benefits by multiplying these avoided costs by the total wetland easements in the HUC8 in 2018. Regions with heavy agricultural activity as well as high levels of restoration (e.g., the Cornbelt and along the lower Mississippi River) see the largest water treatment savings ($100,000–$204,000 per HUC8 per year).

Additionally, we estimate the value based on the size and number of public water systems within a HUC8 and the elasticities based on system size (fig. 6).[24] This assumes that system size determines elasticity (i.e., nitrogen levels are not high). Again, we estimate the potential benefit of an additional 100-acre wetland restoration (fig. 6*A*) and the current value based on the total easements restored as of 2018 (fig. 6*B*).

Here we find that regions with higher populations will accrue more benefits, as large systems accrue the largest absolute benefits. We estimate high potential benefits of restoration in areas surrounding metropolitan areas (e.g., Milwaukee, Denver) and denser nonmetropolitan regions (e.g., the Ohio and Tennessee HUC2s vs. the Plains).[25] Future wetland targeting may also consider the size and number of public water systems in the watershed.

The total benefits from the restored wetlands in 2018 amount to $194 million per year, based on an aggregation of the watershed benefits presented in figure 5*B*. The total cost of restoring 1.7 million acres under the program was $5.9 billion, assuming a per-acre cost of $3,350. With a conservative 10-year lifespan, the annual cost is $590 million, resulting in a benefit-to-cost ratio of 0.33—indicating that wetland restoration offsets about one-third of its costs through water treatment savings. However, if we assume a 30-year lifespan, the annual cost drops to $196 million, making the benefit-to-cost ratio 0.99—suggesting that over a longer horizon, the wetland investment would nearly break even.

Our results suggest that the WRP is an appropriate solution for the water quality issues driven by agriculture. Local communities in agricultural regions face a high cost of water treatment because they are faced with nitrogen levels that may exceed those permitted under the Safe Drinking Water Act and do not benefit from economies of scale for water treatment. The benefits that wetlands provide therefore yield the highest relative cost savings in exactly the places where WRP easements are commonly located.

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24. We use EPA data on the locations of public water systems in 2022. We count the number of active small, medium, and large facilities within each HUC8 that rely on surface water based on the definitions in Mosheim and Ribaudo (2017) with the exception that small systems support fewer than 1,000 people.

25. As urban areas were not included in our sample, our results may not be externally valid to urban HUC8s.

Through this lens, WRP is a form of benefit transfer from the federal to local governments. Per our calculations, many WRP easements may have been net neutral for public expenditure or have in fact yielded a net reduction in public expenditure. The positive expenditure by the federal government was offset by a significant reduction in the burden borne by local utilities and raised by local taxes. Considering that nutrient management is an increasing issue, wetland easements offer an economically feasible solution to these smaller systems with less financial capacity for costly infrastructure investments and upgrades (Rauh and Hughes 2024).

Our cost-saving estimates are conservative in comparison to the valuation estimates of Breaux et al. (1995) and Kazmierczak (2001). Our highest savings to a single public water system is $17,260 per wetland per year, or roughly a $173 reduction in treatment costs per acre per year for a public water system. This is within the range of savings to wastewater treatment facilities estimated by Kazmierczak (2001) ($5–$10,000) and lower than Breaux et al.'s (1995) range of $850–$1,316. They both focus on a smaller subset of states (Louisiana and Florida) where water pollution costs are high and wetlands are abundant. Our sample in the MRB represents a larger area over a longer time span; our cost-saving estimates may represent average expected savings for a larger number of drinking systems in the United States. Moreover, we also highlight benefit heterogeneity by system size and surrounding water pollution levels.

Notably, the benefit calculation we estimate here captures only one narrow benefit provided by improvements in water quality and wetlands. The benefits of improved water quality are myriad, including improved recreation, home value, and human health (Del Rossi et al. 2023). Communities also incur mitigation costs beyond water treatment, including aeration, dredging, and chemical treatment directly to the water body (US EPA 2015). Left unaddressed, nutrient loading will lead to further eutrophication on inland surface water, which will in turn increase methane emissions (Beaulieu et al. 2019).

Wetlands also yield numerous benefits beyond the water quality improvements we study. Wetlands support wildlife (Jenkins et al. 2010), improve agricultural productivity (Karwowski 2022, 2023), filter out nonnutrient chemical pollutants (Tournebize et al. 2013; Gill et al. 2017), and reduce flooding (Taylor and Druckenmiller 2022). The value of these benefits may be higher than the treatment costs we calculate here. Thus, the analysis we perform here should be considered a lower bound on the overall benefits of the WRP. We compare the effectiveness of WRP at reducing nutrient pollution with other policy studies in the literature appendix G.

4. CONCLUSION

Water pollution, primarily from nonpoint agricultural sources, continues to be a challenge in the Mississippi River Basin. We show that wetland restoration by the USDA is one potential abatement strategy to lower nitrogen concentration. We leverage the quasi-random timing of restoration completion to estimate how the stock of land in wetland easement impacts ammonia, TKN, and phosphorus levels.

Increasing the acres of wetland easement by 100% reduces ammonia and TKN concentrations in the subwatershed by 0.004–0.015 mg/L (3%–11%) and 0.025–0.060 mg/L (5%–11%) across specifications with valid pretrends. We find that surrounding land use and weather conditions meaningfully impact the rate at which wetlands filter nitrogen from the system. Wetlands in areas with higher levels of nutrient loading, defined by the baseline area in crop agriculture, experience the largest ammonia reductions.

On average, we did not find a significant impact of wetlands on phosphorus among the sample of ever-treated HUC12s. There is suggestive evidence that trapped phosphorus within subwatersheds may be reducing phosphorus levels in downstream water bodies. We hypothesize that the initial increase in phosphorus is due to the release of legacy phosphorus as soils are disturbed during restoration. Aldous et al. (2005) provide best practices for minimizing release of legacy phosphorus during and following restoration.

We also highlight which areas and under which conditions wetlands yield the highest savings for local water treatment utilities. Wetlands yield high reductions in nitrogen concentration as well as the highest water treatment savings in areas with more agricultural activity. A back-of-the-envelope calculation suggests that WRP easements are a cost-effective nutrient reduction strategy for agricultural regions. The cost to the federal government of acquiring and restoring the land is offset by water treatment savings enjoyed by local communities.

Easements to date are densest in these agricultural regions, demonstrating that program targeting has been relatively efficient in terms of water treatment savings; though this is only one of many benefits that wetlands provide. We also show that the size of the nearby population served by public water systems may be a secondary targeting dimension. These results can aid policymakers as they target future wetland restorations in a cost-effective manner and lead to improved water quality.

Since we find no net change in phosphorus over the lifetime of the wetland and because phosphorus is not a major source of water treatment costs, we provide no estimate of the impact of wetlands on the social cost of phosphorus. While we do not believe that wetlands are a long-term source of increased phosphorus concentrations, it bears mentioning that even temporary excess phosphorus causes damage through eutrophication, algal blooms, and recreation losses, among others.

We shed light on how manmade restored wetlands in the USDA easement programs impacted water quality at a large scale over a span of 30 years. Our benefit estimation builds on previous willingness-to-pay studies assessing the value of wetlands (Moeltner et al. 2019). Future research could more precisely estimate the benefits of wetlands to downstream urban watersheds with high water treatment costs. Such an analysis is important for better understanding the distribution of benefits by income and other demographics. Additionally, it is important to understand heterogeneity in the water quality benefits provided by wetlands (e.g., by connected vs. isolated wetlands), particularly in light of changes to the definition of Waters of the United States.

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