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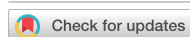
Targeted grassland restoration increases potential for reduced nutrient export in the Upper Mississippi River Basin

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Targeted grassland restoration increases potential for reduced
nutrient export in the Upper Mississippi River BasinKelsie M Ferin^{1,*} , Eric G Booth¹ , Steven P Loheide II², Monica G Turner³ and Christopher J Kucharik¹¹ Department of Plant and Agroecosystem Sciences, University of Wisconsin–Madison, Madison, WI, United States of America² Department of Civil & Environmental Engineering, University of Wisconsin–Madison, Madison, WI, United States of America³ Department of Integrative Biology, University of Wisconsin–Madison, Madison, WI, United States of America

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E-mail: kferin@wisc.edu**Keywords:** agricultural land conversion, nitrogen, phosphorus, agroecosystem, hydrology, modelingSupplementary material for this article is available [online](#)**Abstract**

Agriculture remains a leading cause of poor water quality in the Upper Mississippi River Basin (UMRB), despite ongoing nutrient reduction strategies and incentivized conservation programs. Converting agricultural land to grasslands offers a potential pathway toward achieving nutrient reduction goals for this region. We evaluated whether conversion of agriculture to grassland four decades ago could have lowered contemporary nutrient loads at the basin outlet. Using an integrated modeling framework, we assessed how the extent and spatial configuration (random vs targeted) of land conversion affected nitrogen and phosphorus loading and identified the levels of conversion needed to meet U.S. Environmental Protection Agency goals for nutrient loading. Model results indicate a 45% nitrogen load reduction could have been achieved in 20 years after implementation in 1981 with a targeted 53% conversion of agricultural land, whereas a 45% phosphorus load reduction would require highly extensive conversion of agricultural land and 40+ years, or additional nutrient-reduction practices alongside transformative land-cover change. These simulations demonstrate that earlier and more extensive grassland restoration may substantially reduce nutrient loads, with the targeted approach outperforming random conversion. While a targeted approach offers greater water quality gains and potential for greater return on investment for funded programs, a random approach may be more feasible across large watersheds due to lower social-ecological coordination demands. Nonetheless, achieving UMRB water quality goals by mid-century will require immediate, transformative changes in land use and nutrient management, and support from targeted government policies.

1. Introduction

Although total agricultural area within the Upper Mississippi River Basin (UMRB) has steadily declined in recent decades [1–3], maize production has expanded by 6% while grain production increased by 45% between 2000 and 2024 across UMRB states [4] as driven by global demand, subsidies, and governmental mandates such as the Renewable Fuel Standard. These productivity gains, while supported by advances in technology, breeding, and genetics, have also intensified nitrogen (N) and phosphorus (P) losses while exacerbating nutrient pollution and

degrading water quality throughout the Mississippi River Basin [5–7].

Nutrient management strategies were enacted by states within the UMRB as guided by the US Environmental Protection Agency Hypoxia Task Force to improve water quality by reducing nutrient (N/P) loads by 45% [8] primarily through changes in agricultural management. Recommendations ranged from in-field (e.g. reduced nutrient application, cover crops) and edge-of-field (e.g. saturated-buffers) practices, to conversion of marginal/unproductive land to perennial vegetation [8]. Widespread adoption rates for these practices have decelerated or

become stagnant, despite incentive-based governmental conservation programs (e.g. Conservation Reserve Program (CRP), Environmental Quality Incentives Program (EQUIP)) [9–11]. Additionally, the Conservation Effects Assessment Program estimated these programs have increased soil carbon and reduced soil erosion but have made minimal progress in reducing N and P losses through subsurface flow despite decades of implementation and billions of dollars invested [12, 13]. These losses do not reflect an inherent ineffectiveness of conservation practices, but instead are associated with a combination of factors, including increased infiltration under conservation tillage, continued cropland expansion, and the broad implementation of nutrient reduction practices without targeted placement in vulnerable portions of fields that disproportionately contribute to nutrient losses [12, 13].

Converting agricultural land to grasslands, specifically non-cultivated perennial vegetation (e.g. restoration to native prairies), may be the most efficient method for improving water quality by eliminating pollution at its source. This practice has shown to reduce in-field and downstream losses of N, P, and sediment [14–18]. However, a 20+ year lag in nutrient reductions may occur due to legacy effects [19–21] and heavy precipitation, specific soil and topographic characteristics still pose a risk for increased nutrient loss [15, 22, 23]. Beyond these water quality outcomes, integrating perennial vegetation into the current agricultural landscape can enhance multiple additional ecosystem services [16, 24, 25].

Recent studies have considered many methods for targeting grassland placement on agricultural land to reduce nutrient loss, including prioritizing marginal lands, areas with low suitability/performance ratings, low profitability, or areas identified as nutrient-loss hotspots [26–30]. However, these studies focused on forward-looking scenarios and/or small watersheds and have not examined how previous integration of grasslands could have limited the long-term increase in nutrient pollution across the entire UMRB [31, 32]. Moreover, few studies have explored whether unorganized or randomly-sited grassland restoration efforts across large regions would achieve nutrient reduction goals as effectively as targeted or policy-driven placement [19, 33]. While a targeted approach may provide greater return on investment [34], a random approach is likely easier to implement due to less social-ecological coordination but may result in less overall improvement to ecosystem services [35, 36]. Addressing ‘what might have been’ could support future policies that maximize environmental benefits and improve the cost-effectiveness conservation investments.

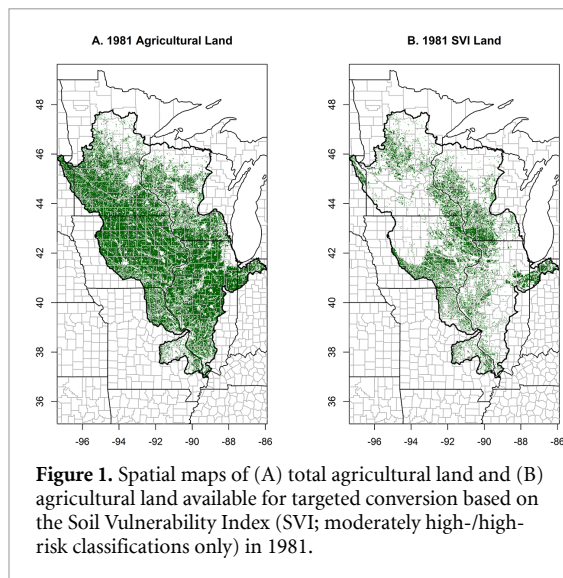
Here, we identified regions within the UMRB that may have high potential for water quality improvement if they were grasslands (i.e. restored,

unmanaged/undisturbed) and used an integrated modeling framework that couples an agroecosystem model (Agro-IBIS) with a hydrologic routing model (THMB). This coupling uniquely links field-scale biogeochemical processes and nutrient dynamics with basin-scale hydrologic and nutrient transportation, allowing us to evaluate differences in water quality with varying amounts and spatial configurations of conversion to grassland from 1981–2021 across the UMRB. This period coincides with the introduction of CRP into the Food Security Act of 1985 and aligns with the availability of transient historical weather data, capturing interannual variability and providing four decades of established grasslands for evaluating long-term water quality impacts. We asked: (1) which UMRB agricultural land and regions are most likely contributing to high nutrient losses given soil and landscape characteristics? (2) How would conversion of agricultural land to grassland approximately four decades ago—including amount and configuration—have affected N and P loads at the outlet of the UMRB compared to historical conditions? (3) What percentage of agricultural land conversion for random vs targeted approaches reduce N and P loading by 45%? Answers to these questions may aid future policies and conservation programs in determining whether agricultural land conversion to grasslands should be prioritized on targeted agricultural areas or implemented more randomly across the landscape.

2. Methods

2.1. Modeling procedure

Simulations were conducted at the 1 km resolution (48 arcsec) across UMRB located in the Midwest U.S. (492 000 km²; figure 1) using the agroecosystem model, Agro-IBIS (Agricultural Version of the Integrated Biosphere Simulator) [37, 38], and a hydrologic routing model, THMB (Terrestrial Hydrology Model with Biogeochemistry) [39, 40]. Agro-IBIS is a physically based, agroecosystem model that simulates leaf level biophysical and biochemical processes. The version of Agro-IBIS used for this study accounts for the exchange of energy, momentum, water, carbon (C), and N for both natural and agricultural land systems, along with coupled interactions of soil erosion and the P cycle on the plant-soil continuum [41]. Hourly calculations involving C and water exchange between plants and the atmosphere are completed at the leaf level and scaled to the field level. The Agro-IBIS model has been previously evaluated and used to simulate a wide range of spatial resolutions, domain extents, and scenarios including changes to land-use/land-cover (LULC), management, and climate [6, 38, 41–46]. For more details on the structure of the original version of Agro-IBIS, refer to Kucharik *et al* [37, 38].



Our Agro-IBIS simulations were driven using historical spatiotemporal weather (gridMET), LULC, area of irrigated agricultural land, and nutrient (N/P fertilizer/manure) inputs along with static gridded soil texture maps derived from the 30 m POLARIS dataset, which is a probabilistic remapping of U.S. Department of Agriculture Soil (USDA) Soil Survey Geographic Database (SSURGO) [3, 6]. Following similar protocol as detailed in Lark *et al* [6] and Motew *et al* [41], the historical spin-up simulation in Agro-IBIS was executed from 1650–1980 to allow for soil C and N pools to reach a quasi-equilibrium to represent contemporary soil biogeochemistry and coupled C-water-energy exchange. Model simulations for the historical baseline and agricultural land conversion scenarios were conducted from 1981–2021.

The hydrologic routing model, THMB, accounts for the transfer of water and nutrients from Agro-IBIS downstream at the gridded scale of a watershed [39, 40]. Calculations within this model occur at the sub-daily time step and are aggregated up to the monthly scale for analysis. Biogeochemistry is accounted for by simulating the removal of N through denitrification [41]. Recent additions to THMB that simulate the transfer of sediment and P downstream are also included in this study and are described in Motew *et al* [41]. Similar to Agro-IBIS, THMB has previously been assessed across multiple spatial resolutions, watersheds, and land and management scenarios [29, 39–41, 46–50].

Output from Agro-IBIS is used to drive THMB and includes precipitation, surface runoff, subsurface drainage, dissolved N in subsurface drainage, dissolved and sediment P in surface runoff, sediment loads in surface runoff, and soil surface/water temperature. Unlike Agro-IBIS, the model spin-up period to reach equilibrium is much shorter for THMB and

simulations were conducted over 1960–2021. Three decades (1992–2001, 2002–2011, 2012–2021) were used to assess past land-cover/management and associated nutrient losses across the UMRB as a historical baseline simulation relative to the agricultural land conversion to grassland scenarios. The baseline simulation was evaluated against monthly average streamflow and N and P loads from the U.S. Geological Survey observations for 2000–2021 at two locations along the Mississippi River [51, 52]. See SI for details on THMB inputs, calibration, and evaluation.

2.2. Scenario development

We defined agricultural land by the land class types that currently have fertilizer or manure applied (i.e. maize, soybean, wheat, alfalfa, pasture, hay). It should be noted that Agro-IBIS does not have specific plant functional types for alfalfa, pasture, and hay and these are assumed to function as a C3 cool grass (alfalfa) and mix of C3 cool season and C4 warm season grasses (pasture/hay). A map of agricultural land was generated using 1981 LULC [3] and used to determine areas of potential conversion to grasslands (274 347 km²). Two methods were used to determine where and how much agricultural land should be converted to grasslands: targeted and random selection.

We generated maps of the USDA Natural Resource Conservation Services (NRCS) Soil Vulnerability Index (SVI) to classify agricultural land for runoff and leaching potential to determine which regions should be targeted for conversion to grasslands [23, 53]. To generate these indices, slope data were obtained from EDNA [54] and soil textural data were obtained from the SSURGO derived POLARIS dataset. Soil order, rock fragment, and *k*-factor maps at 800 m resolution were obtained from the Soil Properties dataset created by Walkinshaw *et al* [55]. All maps were regridded to the 1 km resolution and clipped to the existing UMRB domain for this study. Hydrologic soil groups were determined by the soil textural map and soils were considered organic when classified as a histosol in the soil order map [23]. Our study followed the criteria presented in Thompson *et al* [23] for both runoff and leaching vulnerability as it was developed and evaluated across the UMRB (table S1). Due to limitations in data availability of artificial tile drainage locations at the 1 km resolution across the UMRB, we did not include this adjustment to the leaching vulnerability rating following protocol from Audia *et al* [53].

Once maps were generated for runoff and leaching potential, we selected the agricultural grid cells that fell into the moderately high-/high-risk categories for either runoff or leaching. This method selected 35% of the agricultural land area in the UMRB. From there, we used the *sample* package in R [56] to randomly select 25, 50, and 75% of these grid

cells to be replaced with grassland [56]. This resulted in 8.7, 17.5, and 26.2% of agricultural grid cells to be replaced for each targeted scenario, respectively. Three random conversion scenarios were then generated using the same agricultural land conversion rates as determined in the targeted scenarios (8.7, 17.5, and 26.2%).

The randomized selection process for both approaches were repeated three times for all percentages to account for variability associated with the random selection technique, resulting in a total of 18 scenarios. All land-cover change scenarios begin with grassland establishment in the year 1981 and maintained through 2021, allowing comparison between these conversion scenarios and the historical baseline scenario. These scenario simulations consist of the same Agro-IBIS and THMB model procedure as the historical baseline and analysis of nutrient loads for the outlet of the UMRB was conducted over the same time periods.

Additional scenarios were created for assessing the percentage of agricultural land conversion that would be required to reach a 45% reduction in nutrient loads at the outlet of the UMRB. Since the peer-reviewed literature has demonstrated that there is a strong linear relationship between agricultural land area and its associated nutrient loss [19, 28, 57, 58], we used linear extrapolation via a linear model in R [56] to determine the relationship between the average nutrient load reductions relative to the percent of agricultural land change determined from the three conversion rates previously discussed for the 2012–2021 period. From there, linear extrapolation was used to obtain the percent of agricultural land conversion to grasslands required to reach a 45% reduction in both N and P loads for both approaches (table S2), resulting in 10 additional simulations (3 targeted N, 3 random N, 3 targeted P, 1 random P). For the targeted scenarios, we assumed that all SVI targeted land was first converted and then randomly selected from the remaining agricultural land available to reach the total amount of agricultural land conversion determined by the linear equation for each scenario. This resulted in 100% of SVI land plus an additional 24.6% of the remaining agricultural land for targeted N load reduction and 100% of SVI land plus an additional 67.6% of remaining agricultural land for P load reduction (figures S1 and S2).

3. Results

3.1. Agricultural land and regions likely to contribute high nutrient losses

Agricultural land classified as moderately high/high leaching and runoff potential through the SVI was concentrated along the Driftless Region of southwest Wisconsin and northeast Iowa, south of the Des Moines Lobe, sandy regions of northern Minnesota

and central Wisconsin, and south of Lake Michigan in Illinois and Indiana (figures 1(b) and S2). Random conversion scenarios consistently selected a higher percentage of maize and soybean land relative to the targeted scenarios (table 1). In contrast, targeted conversion scenarios consistently selected a greater percentage of wheat, alfalfa, hay, and pasture than the random scenarios. These relationships were observed for both 1981 and 2021 land-cover (tables 1 and S3). Additionally, the percentage of total agriculture land replaced in the random approach was proportional to all agricultural land types while the targeted approach was proportional to agricultural land grown preferentially on the more vulnerable landscape positions.

Under the targeted approach, the SVI classification for runoff potential led to a selection of 23.8% of total agricultural land while the leaching potential classification led to a selection of 10.9%, with only 0.2% of this land overlapping one another (table 2). Between 45.8 and 54.7% of alfalfa, hay, and pasture was selected from the total agricultural land under the SVI runoff potential targeting scheme. The SVI leaching potential targeting scheme was more evenly distributed across all agricultural land types, with the lowest selection percentage for soybean (7.3%) and the highest for hay (18.0%).

3.2. Effects of conversion to grassland on N and P loading

From the gridded historical weather data used to drive these simulations, average annual precipitation varied spatially and temporally across the basin over the past four decades (figure 2(a)). Additionally, the last decade during this period (2012–2021) was the most variable and had the highest precipitation rates compared to the two preceding decades (figure 2(b)).

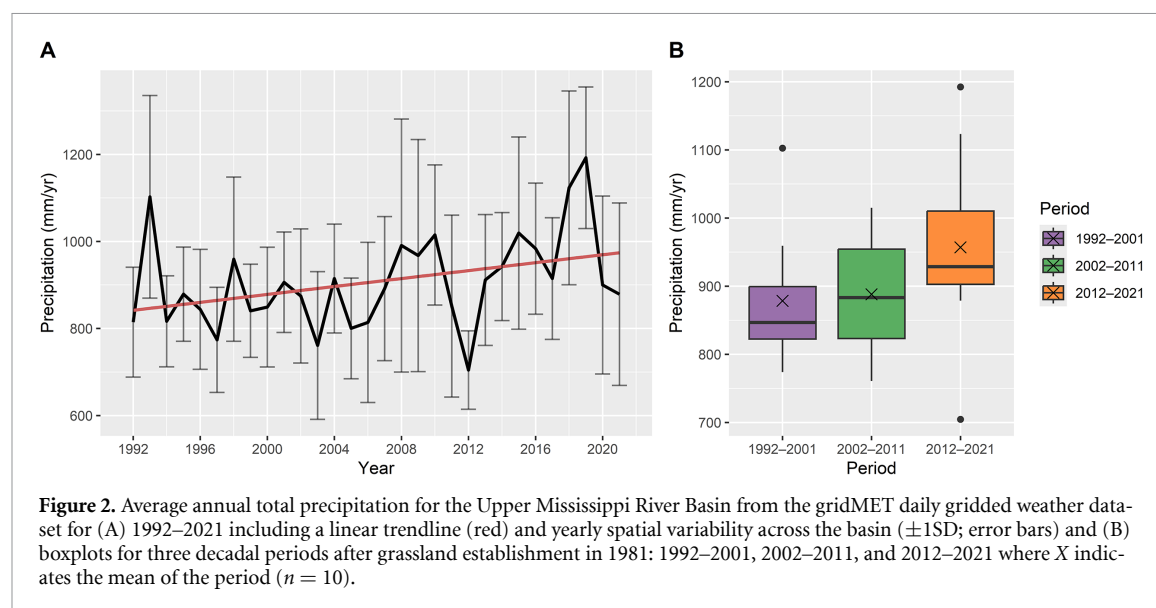
Both N and P loads were reduced relative to the historical baseline with increasing agricultural land replaced with grasslands (figures 3 and S4). The percent reduction in N loads was more variable among replicates compared to P loads under both approaches. There was a statistical difference between P loads at the initial rates of agricultural land conversion ($n = 3$, $P < 0.05$). While there was no statistical difference between the N load reductions from random and targeted approaches when 8.7% and 17.5% of agricultural land was replaced, the N load reduction under the targeted approach was significantly greater than the random approach for 26.2% conversion ($n = 3$, $P < 0.05$). Compared to random scenarios, reductions in N and P loads were greater in the targeted approach relative to the baseline simulation for the same amount of agricultural land replaced. The average percent N load reduction (relative to baseline) was approximately equal to the percent agricultural land conversion (1:1), but P load reductions were less responsive ($<1:1$), especially for the random approach (figures 3 and S4).

Table 1. Percentage of total agricultural land and crop type classification selected for conversion to grassland in each scenario in 1981. See SI for 1981 land-use/land-cover map (figure S3).

Conversion to grassland scenario	Total percentage of agricultural land replaced (%)	Maize (%)	Soybean (%)	Wheat (%)	Alfalfa (%)	Hay (%)	Pasture (%)
Random 8% of agricultural land	8.7	8.5	8.4	8.5	8.2	8.6	8.6
25% of targeted agricultural land	8.7	6.4	4.6	11.7	16.8	16.1	16.9
Random 17% of agricultural land	17.5	17.4	17.5	17.7	17.9	17.1	17.5
50% of targeted agricultural land	17.5	12.7	9.3	23.5	33.7	32.6	33.9
Random 26% of agricultural land	26.2	26.2	26.3	26.6	25.9	26.4	26.3
75% of targeted agricultural land	26.2	19.2	14.0	35.0	50.7	49.1	51.0

Table 2. Percentage of total agricultural land and crop type classification in 1981 selected in each Soil Vulnerability Index (SVI) classification for conversion to grassland. See SI for 1981 land-use/land-cover map (figure S3).

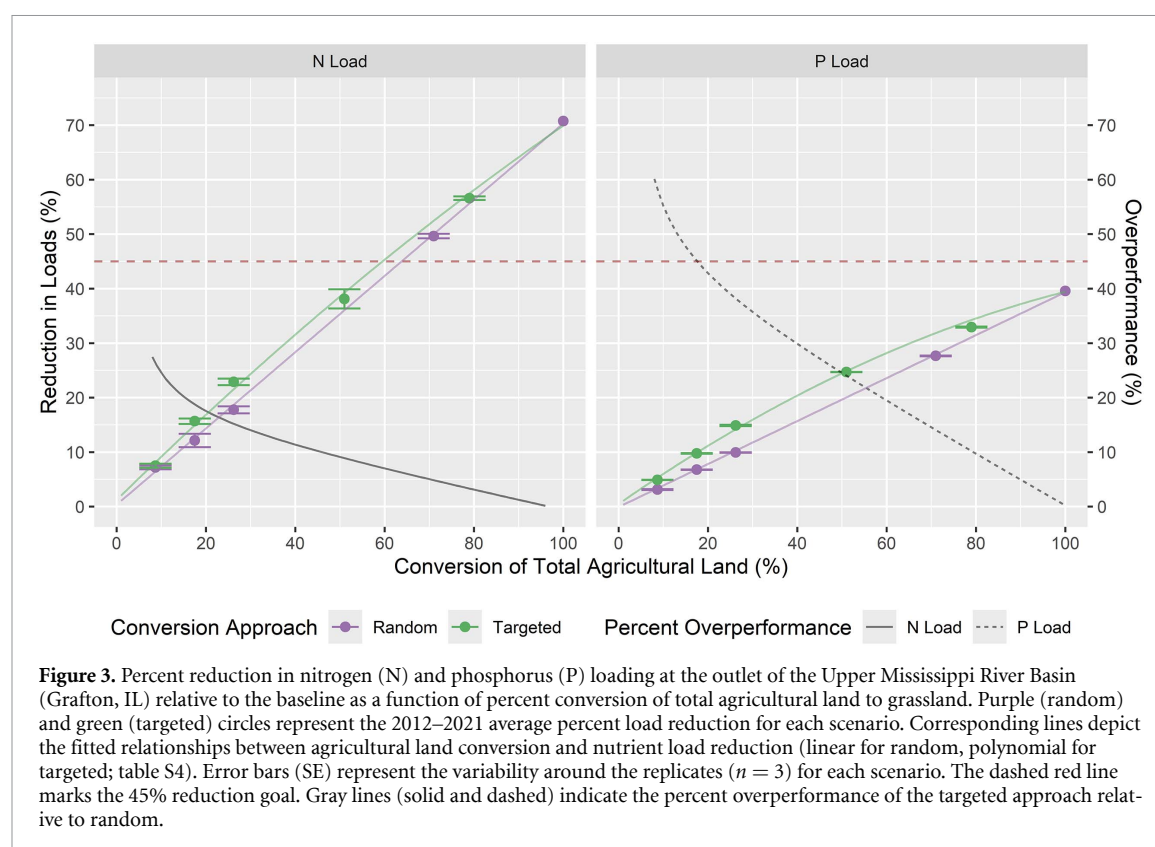
SVI classification	Total percentage of agricultural land replaced (%)	Maize (%)	Soybean (%)	Wheat (%)	Alfalfa (%)	Hay (%)	Pasture (%)
Runoff potential	23.8	15.3	11.2	30.2	52	45.8	54.7
Leaching potential	10.9	10.3	7.3	16.6	15.2	18	12.9
Overlap	0.2	0.1	0.02	0.3	0.6	0.7	0.5



3.3. Strategies that would reduce N and P loading by 45%

Linear extrapolation of N load reductions with the extent of agricultural land conversion estimates that conversion of 71% and 51% of agricultural land area placed at random or targeted, respectively, may have been needed to reduce N loading by 45% within the 41 year timeframe (table S2). Conversion of 79% of targeted agricultural land may have been required

to reduce P loading by 45%. When capping conversion at 100% of agricultural land, linear extrapolation suggests P loads could have been reduced by 39% by 2012–2021. Results from simulating these additional scenarios through the modeling framework showed that a 45% reduction in N loads at the outlet could have been reached by 2012–2021 if 64% of agricultural land was randomly converted to grasslands or 60% of agricultural land was converted through a



targeted approach (figure 3). These percentages differ from the initial estimates provided by the linear regression analysis mentioned above (71% and 51%, respectively; figure S2). However, a 45% reduction in P loads was never achieved in the 41 year simulations (figure 3).

Percentage of overperformance, defined as the percent change between the targeted approach relative to the random approach, indicated that the targeted approach consistently outperformed the random approach (figure 3). The targeted approach was substantially more effective at lower levels of land conversion. This relationship eventually plateaued, more so for N load reductions, resulting in minor differences between approaches. Moreover, the magnitude of overperformance for the targeted approach was considerably greater for P load reductions than for N load reductions.

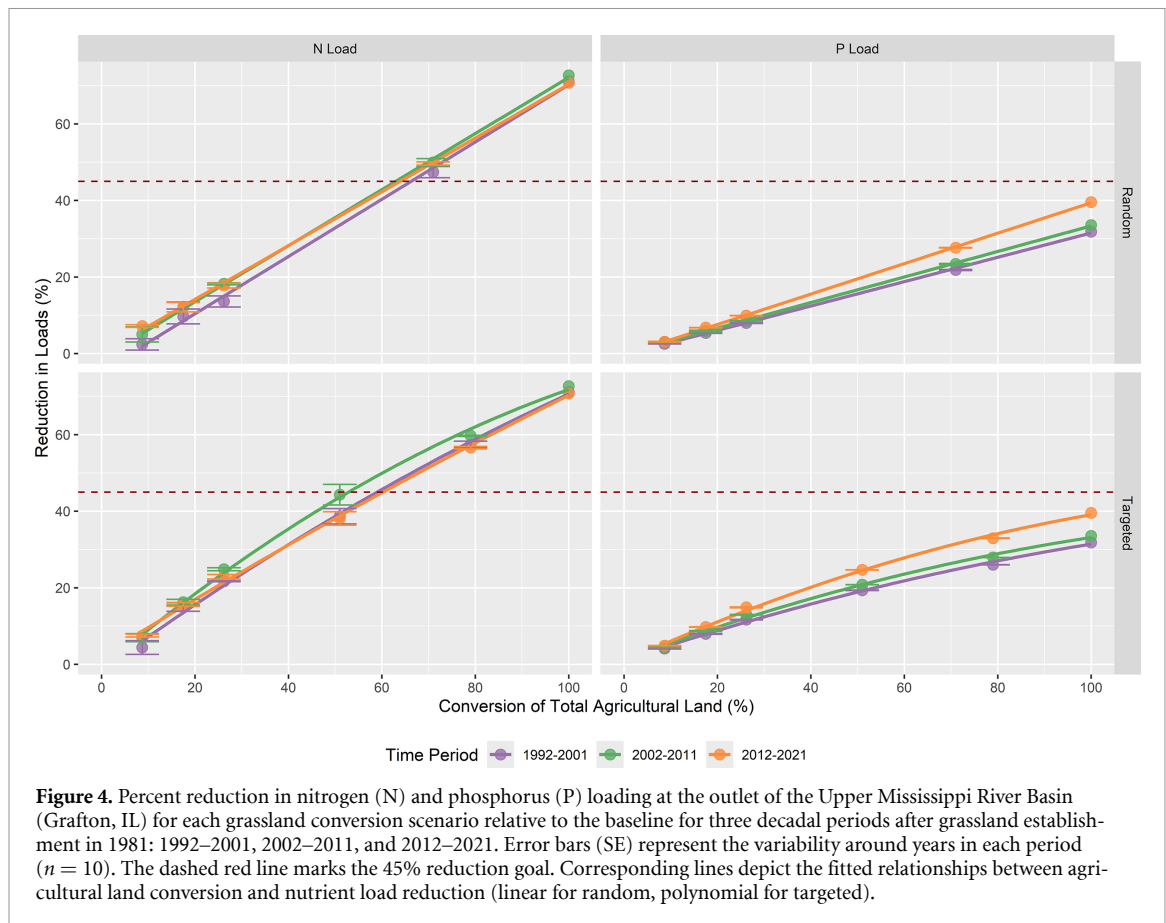
P load relative to the baseline was reduced as time progressed for both random and targeted scenarios (figures 4 and S5). N load reductions under the random grassland placement scenario plateaued by the 2002–2011 period and were maintained through 2012–2021 on average (figures 4 and S5). However, the targeted N load reductions during the 2002–2011 period achieved the greatest reduction relative to the respective baseline period and were less effective by the 2012–2021 period (figures 4 and S5). This observation was more prominent for the targeted 26.5, 51, and 79% of total agricultural land conversion rates.

4. Discussion

4.1. Regions in the UMRB that have the greatest potential for nutrient loss reduction

Our study used a modified version of the USDA NRCS SVI [23] for identifying land with moderately high/high runoff and leaching potential across the UMRB (table S1). This methodology identified a higher percentage of land located in relatively steep regions of the basin (figure 1(b)), which was also noted in previous literature across multiple spatial scales [53, 59]. Additional areas were selected throughout the basin and classified under the leaching potential of SVI. While we did not evaluate the accuracy of SVI selection to in-field observations due to the large spatial resolution and domain extent [23], this area compared well to Agro-IBIS leaching and slope thresholds (section S2 of SI; figure S6).

The SVI methodology resulted in a classification preference weighted towards locations with high runoff potential (table 2). The methodology does not capture the influence of artificial tile drainage on increasing nutrient transport which could have increased the area of high leaching potential [53, 60]. Furthermore, performing the classification at a coarser resolution than the initial SSURGO 30 m could also exclude localized regions of high N leaching potential across the basin in this study. However, the likelihood that land managers and farmers would convert prime cropland with high crop yield potential to grassland after installing tile drainage is lower



due to past investment [61]. These factors may result in a lower total area of SVI land and reduce the nutrient reduction potential across the UMRB under our conversion scenarios.

4.2. Increased grassland area could have reduced historical nutrient loss and transport

The conversion of agricultural land to grasslands has shown great potential for significantly reducing nutrient loads [16, 17, 28, 30, 32]. Our model results indicated that both random and targeted approaches could have reduced N and P loads at the UMRB outlet with increasing agricultural land conversion, had they been established and maintained over past decades. However, a substantially greater reduction in nutrient loading occurred when implementing a targeted spatial approach. The increased effectiveness of the targeted approach is likely attributed to the SVI classification preferentially selecting land that is more vulnerable to nutrient loss, including regions with greater leaching risks, steeper slopes, and higher particulate P runoff due to erosion [53, 59]. Targeting steeper agricultural regions for grassland restoration has been shown as one conservation approach to reduce water and sediment loss [62, 63]. More than 75% of P loss from cultivated landscapes is in the form of particulate P bound to soil and lost through erosion [64], which suggests this targeted approach may aid in

reducing P runoff across the UMRB. Additionally, replacing areas of N intensive agriculture with grasslands on high leaching potential areas removes additional N inputs to the landscape and provides greater nutrient recovery of N from the soil, aiding in reductions to N leaching and loading downstream [15, 18, 65].

Unlike the P load responses, the percent reduction in N loading at the outlet of the UMRB was more variable across the replicates in both the random and targeted approaches (figure 3). This may be attributed to the complexity and bi-directional feedbacks between the N and carbon cycling within AgroIBIS. Furthermore, the spatial and temporal variability of weather may have large implications on water and N loss across a watershed scale by altering the hydrologic pathways and biogeochemical processes which control both movement and transformation of N [66–69].

4.3. Targeted grassland restoration is more likely to achieve water quality reduction goals with less land conversion

The assumption of a linear relationship between the percentage of agricultural land conversion and nutrient load reduction has been previously supported throughout the literature [19, 30, 57, 58, 70]. By linearly extrapolating the initial model results,

we estimated that the targeted approach is likely to reduce N and P loads by 45% with less agricultural land conversion to grasslands than the random approach (table S2). This finding coincides with previous studies which have also shown that targeted land conversion approaches have typically been more effective than random approaches at reducing nutrient and sediment loads across multiple spatial scales [19, 33].

The results of these model simulations showed that the targeted approach follows a curvilinear relationship between the percentage of agricultural land replaced with grassland and percent reduction in N and P loads, while the random approach follows more of a linear pattern (figure 3 and table S4). Additionally, the targeted approach consistently outperformed the random approach for all levels of agricultural land conversion (figure 3). We found this advantage diminished once the 35% of agricultural land identified by the SVI classification was targeted, and additional agricultural land was randomly selected to meet the total conversion rates as specified by the regression analysis. A similar pattern was observed by Tuppad *et al* [33], where initial targeted BMP implementation in a small watershed was more effective at reducing nutrient loss than random placement.

Our results show the lag time of reducing legacy N is much shorter than legacy P across this basin (figures 4 and S5). The decadal analysis we conducted indicated that a 45% reduction in N loads may have been achieved within 20–30 years after implementation in 1981 if a targeted 53% or a random 63% of agricultural land was converted to grasslands. This is a smaller percentage of agricultural land conversion and shorter timeframe than the initial 30–40 years presented in the *results* section. The lag time of reducing legacy N across the basin (figures 4 and S5) coincides with previous findings from Van Meter *et al* [71] and Van Meter and Basu [19, 20]; however, our model may underpredict the legacy N response due to lack of a groundwater reservoir representation for N. Our analysis also indicated that greater N loss may have occurred after targeted grassland conversion was implemented during 2012–2021 relative to 2002–2011 (figures 4 and S5). This final decade corresponds with greater precipitation rates across the basin (figure 2(b)), which is likely driving the increase in N loss on land under high runoff and leaching potential used in the targeted approach.

On the contrary, our results indicate that achieving P reduction goals in the basin by converting agricultural land to grasslands may take longer than the 41 years simulated in this study. The general consensus in lag of P reductions has been approximately 30–40 years for different regions across the world [56] but could greatly depend on factors including climate and basin drainage area [72, 73]. Additional P inputs also originate from non-agricultural sources (e.g.

wastewater treatments plants, urban) [74]; however, this study was not designed to quantify these contributions. In our simulations, the increasing trend towards heavier precipitation during the last 15 years may be inhibiting greater reductions in P loads [41]. This lag is also a consequence of the strong legacy effect of soil P storage in agricultural fields, with evidence suggesting that many decades of zero nutrient applications and land-cover changes may be required to improve water quality [21, 28, 41].

5. Limitations

Uncertainty in model results may arise from the simplification of parameterization within THMB (discussed in SI section S4.3). In addition, uncertainties in Agro-IBIS input data and within both models, particularly when applied over large spatial domains at fine resolutions, may propagate and contribute to cumulative error across the broader modeling framework. To account for this uncertainty and variability, our study included three random repetitions for each model scenario within our analysis.

Our methodology identified areas suitable for land-cover conversion and shows promise for large-scale water quality improvement. However, replacing highly productive cropland without accounting for the economic value of grassland biomass and ecosystem services may reduce profitability for land managers [27, 75]. Prior research suggests targeting unprofitable land for bioenergy perennial grass integration has potential to improve water quality [29] and support positive economic outcomes [27, 76]. Similarly, converting cropland to a managed grazing system may support ecosystem services while increasing profitability [77, 78]. While these strategies offer multiple benefits [79–81], careful consideration is needed to balance environmental gains with economic outcomes for landowners. For example, NRCS programs like EQUIP and CRP incentivize land management practices that enhance ecosystem services, including clean water. Although this study did not include an economic analysis, future research for this region should evaluate the potential tradeoffs of achieving nutrient reductions on producer economic returns to better inform program design, land management decisions, and policy changes [27, 75].

These model results may be helpful in guiding future land management and policy decision-making in the context of a changing climate, which may pose increasing challenges to improving water quality [82, 83]. Climate change is likely to increase N losses on agricultural land due to reductions in crop productivity and corresponding fertilizer N uptake, increases in mineralization and soil N availability, and increased likelihood for runoff and nitrate leaching due to an increasing frequency and magnitude of extreme rainfall events [65, 84–87]. Perennial grasses hold the potential for reducing large scale nutrient losses due

to their extended growth periods, greater rooting density, and ability to translocate more nutrients to the root system than annual row crops [18, 24, 88, 89]. Previous studies that examined smaller watersheds have demonstrated similar benefits of grassland conversion in the context of future conditions [28, 29, 58, 90]; however, our study extends these insights to a regional scale, demonstrating that large-scale implementation enables the targeting of specific watersheds regions. This distinction is important because not all areas are equally effective in reducing nutrient losses.

6. Conclusions

This study provides the first basin-wide assessment of how varied placement and extent of agricultural land conversion to grassland could have reduced nutrient loading in the UMRB over the past four decades. Our simulations suggest that earlier and more extensive grassland restoration would likely have resulted in substantial reductions in N and P loads, with the targeted approach outperforming random conversion. For example, model results indicate that a 45% N load reduction may have been achievable within two decades following implementation in 1981 through a targeted 53% conversion of agricultural land. In contrast, comparable P reductions appear to be less attainable through land conversion alone and likely require highly extensive conversion in combination with additional nutrient management practices.

While these results demonstrate the effectiveness of targeted strategies, they reflect idealized scenarios and do not fully capture the socio-economic and logistical constraints of implementing targeted change across the UMRB. Achieving long-term water quality goals by mid-century will likely require immediate, transformative change across the basin and major shifts in nutrient management practices. These actions should be guided by targeted government policies to prioritize the areas where these transformations will yield the fastest and most impactful change while offering the greatest return on investment for these funded programs.

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Data availability statement

All data that support the findings of this study are included within the article (and any supplementary files).

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