Integrated assessment modeling reveals near-channel management as cost-effective to improve water quality in agricultural watersheds

Amy T. Hansen[a,b,1], Todd Campbell[c], Se Jong Cho[a,d], Jonathan A. Czuba[a,e], Brent J. Dalzell[f,g], Christine L. Dolph[h,i], Peter L. Hawthorne[j], Sergey Rabotyagov[k], Zhengxin Lang[k], Karthik Kumarasamy[l], Patrick Belmont[l], Jacques C. Finlay[h], Efi Foufoula-Georgiou[a,m,n], Karen B. Gran[o], Catherine L. Kling[c,i], and Peter Wilcock[l]

[a]St. Anthony Falls Laboratory, University of Minnesota, Minneapolis, MN 55455; [b]Civil, Environmental, and Architectural Engineering Department, University of Kansas, Lawrence, KS 66045; [c]Center for Agricultural and Rural Development, Iowa State University, Ames, IA 50011; [d]Hydrodynamic Branch, US Geological Survey, Reston, VA 20192; [e]Department of Biological Systems Engineering, Virginia Tech, Blacksburg, VA 24061; [f]Department of Soil, Water, and Climate, University of Minnesota, St. Paul, MN 55108; [g]Soil and Water Management Research Unit, US Department of Agriculture Agricultural Research Service, St. Paul, MN 55108; [h]Department of Ecology, Evolution, and Behavior, University of Minnesota, St. Paul, MN 55108; [i]Dyson School of Applied Economics and Management, Cornell University, Ithaca, NY 14853; [j]Institute on the Environment, University of Minnesota, St. Paul, MN 55108; [k]School of Environmental and Forest Sciences, University of Washington, Seattle, WA 98195; [l]Department of Watershed Sciences, Utah State University, Logan, UT 84322; [m]Department of Civil and Environmental Engineering, University of California, Irvine, CA 92697; [n]Department of Earth System Science, University of California, Irvine, CA 92697; and [o]Department of Earth and Environmental Sciences, University of Minnesota, Duluth, MN 55812

Edited by B. L. Turner, Arizona State University, Tempe, AZ, and approved May 28, 2021 (received for review December 3, 2020)

Despite decades of policy that strives to reduce nutrient and sediment export from agricultural fields, surface water quality in intensively managed agricultural landscapes remains highly degraded. Recent analyses show that current conservation efforts are not sufficient to reverse widespread water degradation in Midwestern agricultural systems. Intensifying row crop agriculture and increasing climate pressure require a more integrated approach to water quality management that addresses diverse sources of nutrients and sediment and off-field mitigation actions. We used multiobjective optimization analysis and integrated three biophysical models to evaluate the cost-effectiveness of alternative portfolios of watershed management practices at achieving nitrate and suspended sediment reduction goals in an agricultural basin of the Upper Midwestern United States. Integrating watershed-scale models enabled the inclusion of near-channel management alongside more typical field management and thus directly the comparison of cost-effectiveness across portfolios. The optimization analysis revealed that fluvial wetlands (i.e., wide, slow-flowing, vegetated water bodies within the riverine corridor) are the single-most cost-effective management action to reduce both nitrate and sediment loads and will be essential for meeting moderate to aggressive water quality targets. Although highly cost-effective, wetland construction was costly compared to other practices, and it was not selected in portfolios at low investment levels. Wetland performance was sensitive to placement, emphasizing the importance of watershed scale planning to realize potential benefits of wetland restorations. We conclude that extensive interagency cooperation and coordination at a watershed scale is required to achieve substantial, economically viable improvements in water quality under intensive row crop agricultural production.

water quality | agriculture | wetlands | integrated assessment modeling

Intensive agricultural production, as practiced in the Midwestern United States, is now recognized as the primary cause of impaired surface water quality (1–3). Dominated by corn and soybean row crops, land management in this agricultural system has negative impacts on water quality via both direct losses of nutrients and sediment from fields and indirect effects through modifications of runoff, streamflow, and channel networks (4, 5). Extensive networks of artificial agricultural drainage, such as straightened streams and subsurface tile drainage, have amplified storm runoff intensity, reduced water residence time, and increased sediment erosion from near-channel sources downstream (4, 6, 7). The effects of degraded water quality extend throughout the Mississippi River network and into the northern Gulf of Mexico (5, 8). This degradation of surface water compromises its safety for drinking (9), the suitability of lakes and rivers for recreation (10), and the ability of both inland and coastal waters to support aquatic life (5, 11).

Despite consensus on the overall cause of water quality degradation and financial investment toward more sustainable management of agricultural fields, water quality has not significantly improved in the Midwestern United States (1, 12). Although several assessments show reductions in direct nutrient and sediment losses from agricultural fields and some improvement in river water quality (13, 14), these localized improvements have not translated into meeting water quality targets within the receiving rivers, nor a reduction in the size of the northern Gulf of Mexico hypoxic zone (1, 3, 12). Lack of improvement in river water quality, despite ongoing conservation efforts, is linked to five factors. First,

Significance

Water quality is severely degraded in landscapes cultivated for intensive corn and soybean production. Current water quality policy focuses on reducing nutrient and sediment losses from agricultural fields, yet recent studies have highlighted important roles of near-channel areas as sources of sediment and sinks for nitrogen. We developed an integrated modeling approach to assess water quality cost-effectiveness tradeoffs for watershed management scenarios that include a wide range of both field and near-channel management actions, yielding estimates of reductions in sediment and nitrate loads and associated costs for alternative management actions. Our results indicate that near-channel management, most notably fluvial wetland restoration, was most effective for achieving longstanding policy goals for sediment and nitrate reduction.

Author contributions: A.T.H., T.C., S.J.C., J.A.C., B.J.D., C.L.D., P.L.H., S.R., P.B., J.C.F., E.F.-G., K.B.G., C.L.K., and P.W. designed research; A.T.H., T.C., S.J.C., J.A.C., B.J.D., C.L.D., P.L.H., S.R., Z.L., and K.K. performed research; A.T.H., T.C., S.J.C., J.A.C., B.J.D., C.L.D., P.L.H., S.R., and Z.L. analyzed data; and A.T.H., S.J.C., J.A.C., B.J.D., C.L.D., P.L.H., S.R., Z.L., K.K., P.B., J.C.F., E.F.-G., K.B.G., C.L.K., and P.W. wrote the paper.

The authors declare no competing interest.

This article is a PNAS Direct Submission.

[1]To whom correspondence may be addressed. Email: amy.hansen@ku.edu.

This article contains supporting information online at https://www.pnas.org/lookup/suppl/doi:10.1073/pnas.2024912118/-/DCSupplemental.

Published July 6, 2021.

intensively managed agricultural landscapes in the Midwestern United States still receive annual or biannual applications of fertilizer and manure, despite large legacy stores of nutrients (2, 15), leading to increasing nutrient saturation of landscapes. Second, key pollutants (nitrogen, phosphorus, and sediment) have spatially distinct sources from one another and are mobilized by different mechanisms. Conservation strategies that target single contaminants may not be effective for or may even augment delivery of other contaminants (16, 17). Third, the extent of artificial drainage has continued to increase, leading to more rapid movement of water into and through drainage networks, increasing field nutrient losses and riverbank erosion (4, 7, 18). Fourth, climate change has increased rainfall and runoff in the Upper Midwestern United States, increasing nutrient (19, 20) and sediment (18) losses. Finally, agriculture-related water quality programs do not sufficiently account for heterogeneity in water quality benefits nor in costs of the management practices they incentivize, and thus, both the level and allocation may be mismatched with the magnitude of the issue (21–23). In essence, sustainable solutions to management of Midwestern US agricultural watersheds must address the spatial complexity of sediment and nutrient sources in the context of increasing water yield from greater rainfall and rapid drainage.

Recent studies suggest that near-channel processes significantly alter water quality in intensively managed agricultural regions. Wetlands, including fluvial wetlands (e.g., flow-through wetlands, shallow lakes, floodplains, and backwaters) have been shown to reduce river nitrate concentration in intensively managed agricultural watersheds (24–26). Similarly, near-channel sources of sediment (e.g., river bluffs, streambanks, and ravines) often dominate sediment loading (18, 27). However, despite evidence of the potential importance of near-channel processes, current water quality policy heavily promotes field management (e.g., tillage management, precision fertilizer applications) (28, 29). This is partly due to the limited ability of planning tools such as watershed models to predict improvements from near-channel management (30). The lack of a comprehensive, watershed-scale analysis tool that incorporates both near-channel and in-channel processes likely has resulted in misdirected conservation funding to management actions with limited cost-effectiveness for water quality. Given that water quality impairment in intensively managed agricultural watersheds poses a complex and spatially distributed problem, effective management solutions must address near-channel as well as agricultural field contributions to both the problem and the solution.

In this study, we evaluated the capabilities of watershed-scale management plans—consisting of diverse portfolios of field and near-channel management (*SI Appendix*, Table S1)—to cost-effectively restore water quality. In this approach, we approximate trade-offs in cost-effectiveness across multiple monetized and nonmonetized objectives for an intensively managed landscape, similar to refs. 31–38. To date, such analyses have not been possible due to the limited ability of watershed models to capture near-channel processes (30). We overcame this barrier by integrating three biophysical models into one agricultural field-to-river integrated model, hereafter referred to as the AgRiver model. The framework integrates two models of near-channel processes: the Nitrate Network Model (NNM) (39) and the Management Option Simulation Model (MOSM) (40), with the Soil and Water Assessment Tool (SWAT), a watershed model that is widely used to assess field management effectiveness (41). Using the AgRiver model and an evolutionary optimization approach, we compared performance of watershed management portfolios based on their ability to simultaneously reduce nitrate loads (N), sediment loads (S), and cost across a broad range in reduction targets. Field-derived phosphorus (P) was tracked, but P was not an optimization target in this study due to significant gaps in scientific understanding of near-channel P contributions and management action effectiveness for riverine water quality (16, 20). The AgRiver model and an optimization approach allowed us to address the pressing challenge of water quality impairment in intensively managed landscapes through advances in watershed modeling that enabled analyses of the role of near-channel actions and spatial arrangement of management actions on overall conservation cost-effectiveness.

We applied the integrated watershed modeling framework to the Le Sueur River Basin (LSRB), a subwatershed of the Minnesota River basin. At the time of this study, water quality goals for the LSRB were a 45% reduction in total nitrogen and a 65% reduction in total suspended solids over 10 y (42). Average annual spending to improve water quality in the LSRB was \$4.3M USD over 2,900 km^2 or \$14.7 USD/ha/yr during 2004 to 2018 (43). The LSRB contributes S, N, and P to the Minnesota River basin far in excess of its proportion of the drainage area and has been the subject of detailed field studies quantifying the spatially explicit origins of S and N (6, 25, 44). While many watersheds in the Mississippi River Basin share similar water quality problems, the LSRB was chosen due to the availability of extensive observational datasets used to construct, constrain, and calibrate the AgRiver model (45), the degraded quality of water within and exported from the basin (46), and the extent of intensively managed agriculture (47). Near-channel S loading is higher in the LSRB than most in the region due to its historic connectivity to the drainage pathway for glacial Lake Agassiz (6). Land use and nutrient yields from the LSRB are similar to other intensively managed agricultural basins in the region (26, 48), and thus, insight gained from the LSRB can inform management effectiveness throughout the region.

Results and Discussion

Recent analyses show that current conservation efforts are not sufficient to reverse widespread water degradation in Midwestern agricultural systems (48). In contrast, our analyses show that comprehensive water quality improvements for N and S could be achieved at economically viable investment levels, provided that they target the most cost-effective methods for addressing the problem. In particular, we found that the most cost-effective conservation programs must 1) prioritize construction of fluvial wetlands at optimal locations on the river network. Due to wetland construction costs and performance sensitivity to location, this further requires programs to 2) develop one integrated watershed management plan that allows for federal, state, and private entities and 3) pool resources. This broad conclusion is supported by analyses that identified near-channel management, specifically fluvial wetlands, as the most cost-effective watershed management action for all portfolios with budgets large enough to support them (i.e., >\$300K/yr). Fluvial wetland performance was highly dependent on spatial location, however, underscoring the need for coordination across a watershed. While other management actions were also effective for reducing N and S, none were as cost-effective.

Synergies and Trade-offs for Cost-Effective Watershed Management Scenarios. Watershed management portfolios that best met the combined targets for N, S, and cost, (i.e., cost-effective portfolios) were identified by a multiobjective optimization algorithm. The collection of cost-effective management portfolios for all combined targets forms a frontier of optimality in N-S-cost space that consists of the lowest achievable simultaneous N and S reduction and cost targets and provides insight into trade-offs and synergies between targets (49). Cost-effective management portfolios synergistically reduced N and S loads with larger reductions in both N and S loads as spending increased (Fig. 1 and full frontier at *SI Appendix*, Fig. S2). This observation broadly held true regardless of how S versus N were prioritized (S heavily prioritized shown with red outline, and N heavily prioritized shown with black outline; Fig. 1). Scatter between N and S in Fig. 1 is due to trade-offs between the two objectives and increased as cost targets decreased (Fig. 1*B*). The high degree of scatter for low-cost targets indicates that the need to clearly define water quality management objectives is greatest at low investment rates. Although not an optimization target, the reduction in field-derived P load was also synergistic with N and S (*SI Appendix*, Fig. S3). Relatively small increases in investment resulted in large gains in water quality when management actions were optimized. For $2M/yr ($6.90/ha/yr), N loads could be reduced by 32 to 86%, S loads could be reduced by 23 to 50%, and field-derived P could be reduced by 6.5 to 21%, with the range for each depending on prioritization of N versus S targets (Fig. 1 and *SI Appendix*, Fig. S3). At a cost of $12M/yr (~3% of commodity sales in 2017), essentially all achievable reductions in S (77% reduction) and N (~100% reduction) were met, and field-derived P was reduced by 46 to 65% (*SI Appendix*, Fig. S3). Due in part to the inclusion of a broad range in field and near-channel management actions compared to previous models, our analysis predicted sizeable reductions in sediment load for much lower costs than previously reported (40, 50). In comparison to studies investigating nutrient reduction via wetland placement, our results are similar: a 20 to 40% N reduction was achieved via wetland optimization in (51) for $3.30/ha/yr and 25% reduction in N for $4.50/ha/yr (52) compared to ~$2.00/ha/yr in our results.

Cost-Effective Budget Allocation. Overall, near-channel management emerged as more cost-effective than field management with the exception of scenarios with budgets below $300K/yr, which were incapable of supporting fluvial wetland construction (Fig. 2). For investments at $500K/yr or more above current spending, the budget was primarily allocated for implementing near-channel management actions (Fig. 2 *A* and *B*). For new investments between $300K/yr and $500K/yr, loads were reduced by ~30% (N) and 13 to 17% (S), and spending was more evenly distributed between near-channel and field actions regardless of how N and S were prioritized relative to one another (Fig. 2 *C* and *D*). For new investments less than $300K/yr ($1.03/ha/yr), minimal reductions in N (4%) or S (3%) were achieved, field management was preferentially funded for portfolios prioritizing N reductions, and a balance of field and near-channel management was selected for portfolios prioritizing S reductions (Fig. 2 *C* and *D*).

Individual management actions may reduce both N and S (e.g., cover crops and wetlands) or primarily a single water quality target (e.g., bank stabilization for S or fertilizer management for N). For watershed management portfolios in which N reductions were prioritized, N decreased linearly with the number of wetlands and showed no trend with the extent of field management (Fig. 3*A*). Near complete removal of N was achieved with ~20 fluvial wetlands (out of 103 potential wetland restorations). When S reductions were prioritized, reductions in S were influenced by the extent of fluvial wetlands, ravine stabilization, and field management (Fig. 3*B*). The asymptotic shape of the relationship between S and these three management actions demonstrates a diminishing return on investment in efforts to control S.

For budgets sufficiently large enough to construct at least one wetland, near-channel management in the form of fluvial wetland construction was found to be highly cost effective. This is evident in both the relative allocation of funds to each management action (Fig. 2) and in the extent to which each action was selected relative to its maximum potential extent with increasing costs (Fig. 3*C*). Fluvial wetlands reduce peak streamflow and thus reduce near-channel S loading and at the same time promote internal N removal processes. Interestingly, the number of fluvial wetlands and not the size of selected wetlands increased linearly with increased spending (*SI Appendix*, Fig. S4). Within wetland options, small, shallow fluvial wetlands (individual wetland area = 2.02 ha, average depth <1.1 m) were preferentially selected over larger or deeper wetlands (*SI Appendix*, Fig. S4). The selection preference for numerous small, shallow fluvial wetlands instead of

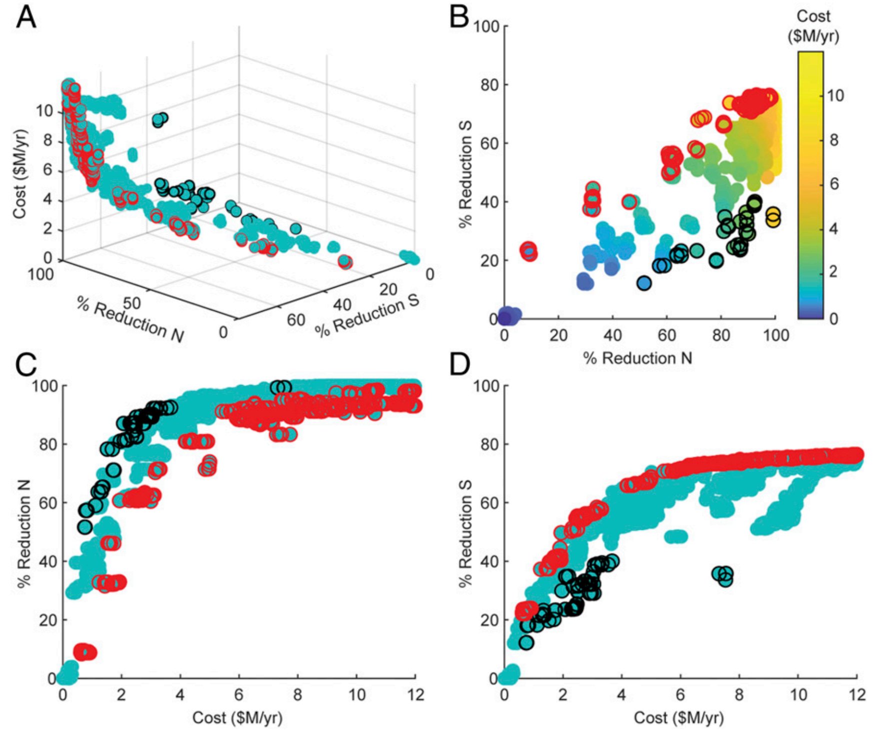

Fig. 1. Cost-effective watershed (Pareto) frontier. (*A*) Three-dimensional frontier of cost-effective watershed management portfolios (individual points) that meet simultaneous targets to reduce S, N, and cost. All cost-effective watersheds meeting cost targets under $12 million/y are shown as solid circles. Outlined circles show water quality target prioritization—watersheds where S was prioritized over N (red outlines) and watersheds where N was prioritized over S (black outlines). (*B*) Two-dimensional plot shows synergy and scatter between N and S objectives with cost shown using color. (*C* and *D*) Two-dimensional plot of N load reductions versus cost and S load reductions versus cost.

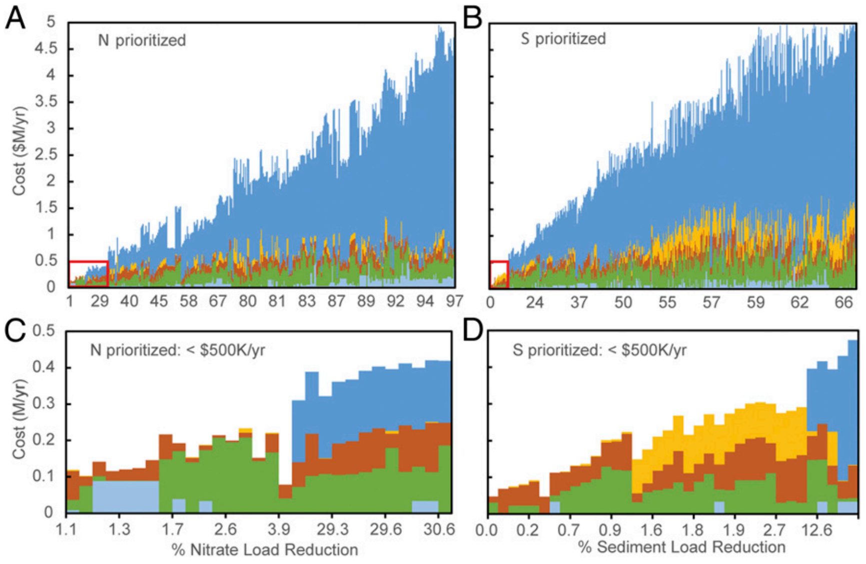

Fig. 2. Cost allocation by management action. Stacked bar charts showing how spending is allocated across management actions within cost-effective watershed portfolios that prioritize N reduction (*A* and *C*) or S reduction (*B* and *D*) in which the bottom panels are enlargements of red boxes region in the upper panels. Candidate management actions include the following: isolated wetlands (light blue), all field management actions (green), bank/bluff stabilization (red), ravine stabilization (yellow), and fluvial wetlands (blue). On average, fluvial wetlands account for a linearly increasing proportion of spending for cost targets above $500,000/y for both N and S prioritization (*A* and *B*). In contrast, at cost targets <$500,000/y, field management and other near-channel management actions are selected (*C* and *D*). Note that *x*-axes are categorical, thus not linear, and sorted by increasing load reduction.

fewer, larger, or deeper wetlands may be due to their preferable ecological function and lower construction costs. Rates of denitrification, a primary N removal mechanism, depend on both N and organic carbon supply so small, shallow wetlands with high rates of internal dissolved organic carbon production from emergent vegetation likely have higher N removal rates compared to deeper or larger wetlands. Economics likely also plays a role in the selection preference for small, shallow fluvial wetlands since dredging is one of the highest costs of wetland construction (53).

When S reduction was prioritized, near-channel management in the form of ravine stabilization was preferentially selected (Figs. 2 *B* and *D* and 3*B*). Ravines form through focused erosion in ephemeral channels linking uplands with deeply incised mainstem channels and are found throughout the lower watershed. Ravine erosion leads to high sediment concentrations, but they are limited in area. Although highly cost-effective, the potential of ravine stabilization to improve water quality was restricted by the limited contribution of ravines to total S loading in the LSRB (14% of total S loading under baseline conditions).

Field management was found to be a persistent component of cost-effective watershed management portfolios regardless of the rate of new investment or N versus S prioritization at cost targets under approximately $20M/yr. Because there were only minor increases in investment in field management, as total investment increased (Fig. 2 *A* and *B*), the relative allocation of funds to field management decreased. Although relatively less cost-effective than near-channel management, there are reasons to continue to promote field management that are not addressed within this study. First, field management is a preventative solution and may be more effective at reducing byproducts of excess fertilizer application that were not included in this study including greenhouse gas emissions (nitrous oxide) and contributions to legacy stores of N

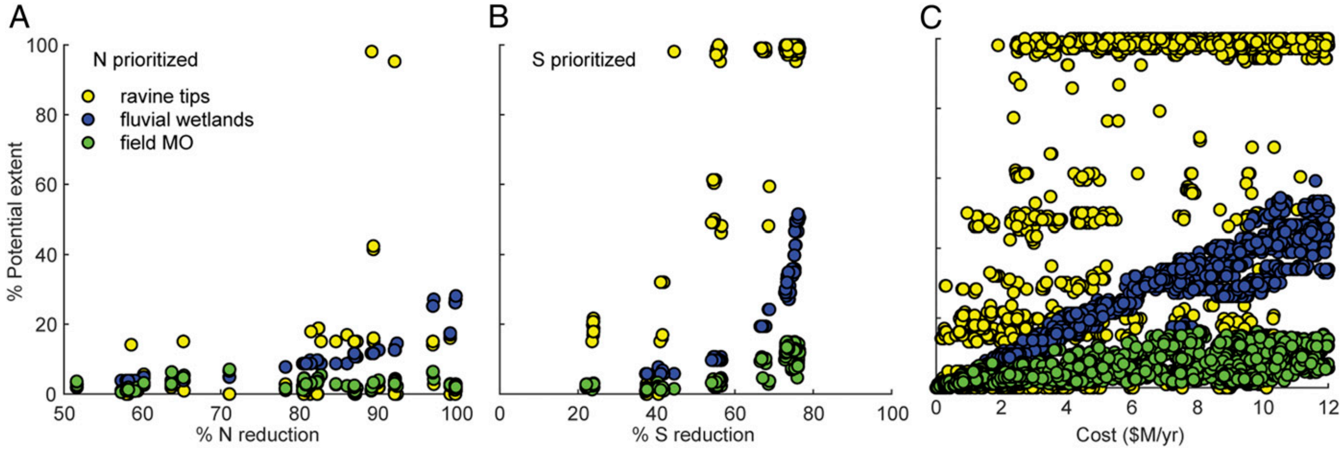

Fig. 3. Utilization of potential management action extent. Percent of potential locations for each management action that were selected for N reduction when N was prioritized (*A*), S reduction when S was prioritized (*B*), and extent of each management action versus cost for all cost-effective solutions (*C*). A 100% potential extent means all possible locations for a management action within the watershed were selected.

and P. Second, off-site management actions are removed from the input location and "out of sight" of upstream farmers, which may reduce the sense of personal responsibility of landowners which could, in turn, lead them to disregard the externalized, downstream costs of excess fertilizer and increase application rates (54).

Dependency on Location. To understand the flexibility with which managers can accommodate political or private preferences yet still achieve their water quality goals, we evaluated the extent of preferential spatial placement of individual management actions within clusters of cost-effective watershed management portfolios (Fig. 4). The first cluster consisted of 30 cost-effective watershed management portfolios that met the current policy target to reduce N by 45% at minimal cost (42). The second cluster consisted of 31 cost-effective watershed management portfolios which met the current policy targets for both N and S (reduce S by 65%) at minimal cost (42). Within these two clusters, near-channel management actions were consistently positioned in the same locations in the watershed (Fig. 4 *B* and *D*). In cost-effective watershed management portfolios that met the N target only, wetlands were positioned near the outlet (Fig. 4*B*). Portfolios that satisfied both N and S targets contained more wetlands, and these were positioned further upstream, typically along the three major tributaries (Fig. 4*D*). The preferential placement of wetlands along major tributaries may be due to a trade-off between high N interception rates and sufficient wetland volume to reduce peak streamflows and thus downstream near-channel S generation. In contrast to wetland placement, no strong location preference was observed for field management actions (Fig. 4 *A* and *C*).

Need for Collaboration. Our results provide much needed guidance toward cost-effectively achieving water quality goals. Nonetheless, the costs are substantial in comparison to a single agency's annual budget, and the performance for the most effective management actions (i.e., fluvial wetlands) is spatially dependent, suggesting that strong coordination across agencies in spending and planning is needed. During the study period, an average of $4.3 M/yr was spent on water quality measures in the LSRB by federal, state, and local agencies with an average budget per agency of $610K/yr (43). However, budgets must be above $500K/yr for fluvial wetlands, the most cost-effective management action, to be feasible. For example, based on the results in Fig. 2, four agencies working independently with annual budgets of $250K/yr would reduce S and N by ∼10% of current loads. However, if they were to coordinate their spending the $1M/yr total investment would collectively achieve a 30% reduction in S and ∼50% reduction in N. By collaboratively developing a whole-watershed plan as well as combining financial resources, S load reductions would be three times greater, and N load reductions would be five times greater than if agencies worked separately, due to their ability to pool resources and thus construct more wetlands as well as choose more optimal locations for wetland construction. An ongoing policy challenge is creation of a system of incentives to implement an (approximately) cost-effective allocation in the context of system-wide interdependencies of the effectiveness of management actions and informational asymmetries with respect to costs of private management efforts (55). While theoretical and empirically grounded advances have been made (56, 57), practical implementation will require substantive agency and stakeholder collaboration. Furthermore, although it is likely that our results are fairly robust to the estimates of the water quality effectiveness and cost (*SI Appendix*), investments in long-lived wetlands would need to be evaluated for their performance in light of ongoing climate change and under deep uncertainty (58).

Future Directions. This analysis concludes that near-channel management, primarily in the form of fluvial wetlands, was most cost effective toward reducing both N and S loads. Because of this, we expect the results of this study to be transferable, in concept, to agricultural basins throughout the Midwestern United States where near channel sources are known to be an important yet poorly constrained source of S (59). Additional research is to constrain the proportion of sediment derived from near-channel sources in other watersheds and better represent near-channel sediment sources in watershed models. Similarly, field and modeling studies that constrain near-channel P sources and transport are needed in order to include P as an optimization target and better align model output with the full suite of typical goals for water quality programs. Finally, management action effectiveness was modeled as a static function, but many actions have a limited life-expectancy that should be considered for full cost-benefit analysis.

Our focus with this research was to consider an expanded suite of management actions that included near-channel in order to identify more cost-effective watershed management portfolios for improved water quality. To facilitate this, we use a simplified economic component in the form of estimates of exogenously determined annualized costs of management actions. Future research could expand on these results by 1) considering the structure of economic incentives for cost-effective outcomes under the challenges presented by nonpoint source pollution problems (55, 56, 60) in the context of integrated assessment models (61, 62), 2) incorporating the broad set of factors known to influence private conservation and program participation (63), and 3) considering collaborative management of complex systems under changing external regimes and uncertainty (58).

Conclusion

Our analyses show that achieving cost-effective management of riverine water quality in intensively managed agricultural systems requires a watershed perspective and collaborative cross-agency decision making. Near-channel management actions, specifically small, shallow fluvial wetlands and ravine stabilization, were clearly more cost effective than field management. However, wetland performance was highly dependent on optimal positioning, and wetlands can be prohibitively expensive for individual farms or agencies. Thus, a comprehensive watershed planning strategy that considers the watershed as a system, combines fiscal resources, and carefully selects fluvial wetland location will yield the most efficient reductions in N and S loads. Our results are supported by decades of scientific investment in understanding watershed scale processes in an intensively managed watershed and will enable better use of limited conservation investments to achieve water quality goals.

Materials and Methods

Biophysical Modeling Framework. Three biophysical models of the LSRB were linked to fully capture both terrestrial and near-channel processes (*SI Appendix*, Fig. S5). Terrestrial inputs, transformations, and transport of water, N, S, and P were modeled using the SWAT (41). The base unit in SWAT is the hydrologic response unit (HRU) in which each HRU represents a distinct combination of land use, land cover, soil type, and slope within a subbasin. The computational unit in SWAT is the subbasin, which accounts for the spatial distribution of basin characteristics and land management. The LSRB SWAT model consisted of 103 subbasins (average area 15 km^2) and 934 HRUs. Output from SWAT was routed to two river network models to model near-channel processes. Near-channel N removal was modeled using the NNM (39). Upland S delivery and near-channel S loading were modeled with the MOSM (40). NNM, MOSM, and SWAT are all publicly available (39–41). Weather was modeled at a daily time step and as spatially uniform in order to separate the effect of watershed spatial context from localized variability in weather patterns. Persistent or future spatial patterns in weather may also contribute to decisions about appropriate conservation actions and location and are the subject of future study. Further model details, including model calibration and validation, are provided in *SI Appendix*.

Management Actions. This analysis considered a broad suite of candidate management actions that have previously been shown to reduce N or S loads (*SI Appendix*, Table S1). Management actions were classified as either field management (i.e., actions on current agricultural land) or as near-channel management (i.e., actions within the riverine network). Field management actions

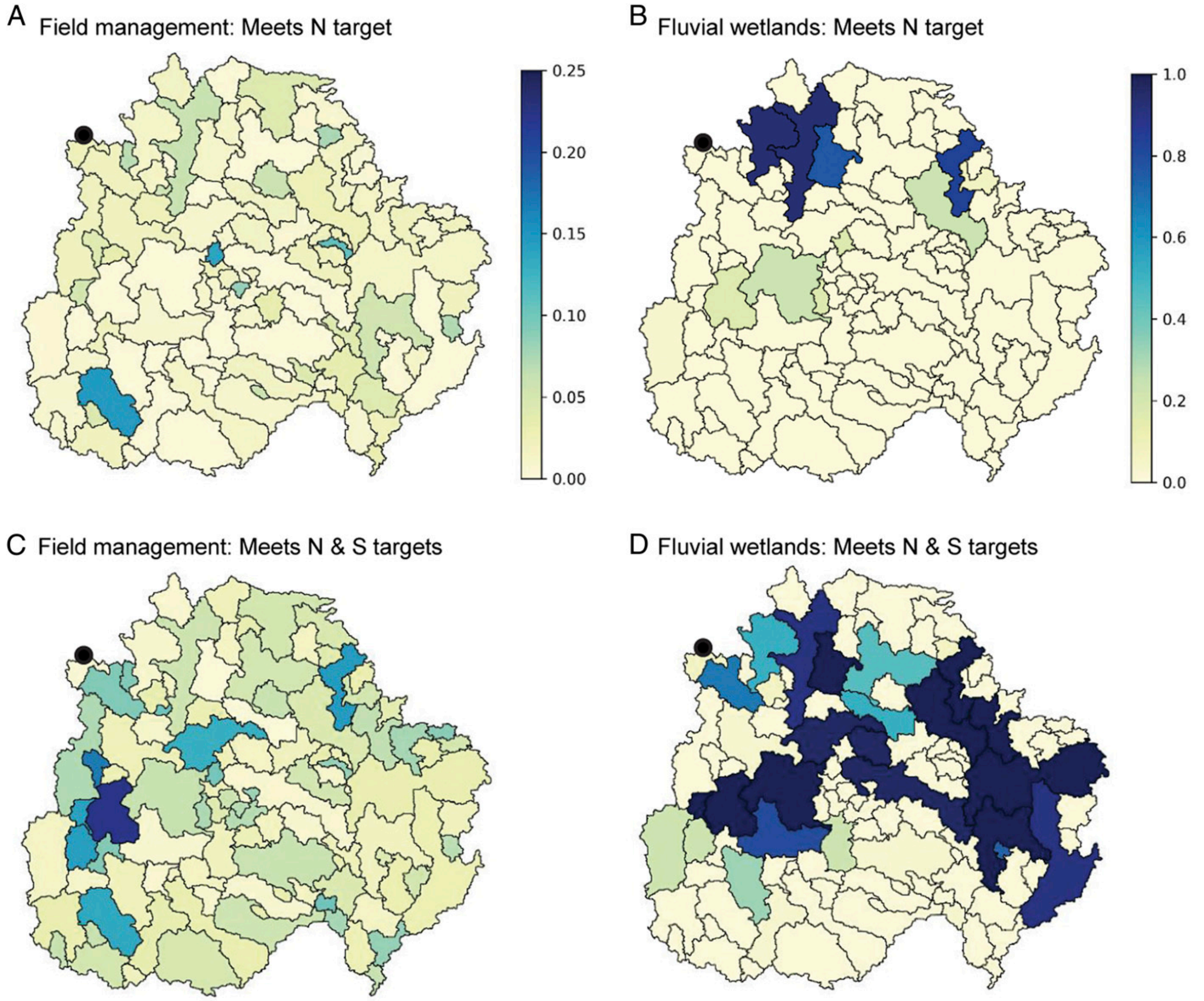

Fig. 4. Spatial dependency of management actions. Watershed subbasins colored by the fraction of watershed portfolios in which they were selected for either field management or wetland placement within clusters of portfolios meeting the N target reduction (*A* and *B*; 30 portfolios) or both N and S policy target (*C* and *D*; 31 watersheds). Note that the color bar scales are different for field management (*A* and *C*) and fluvial wetlands (*B* and *D*). The subbasins selected for wetland remediation (*B* and *D*) are more spatially persistent than those selected for field management actions (*A* and *C*). The watershed outlet is shown with a solid black circle located in the top left corner.

included cover crops, grassed waterways, isolated wetlands or ponds, which all retain N and S on agricultural fields, and fertilizer management to reduce the quantity of N applied. SWAT architecture restricts isolated wetlands to one per subbasin; in the model, isolated wetlands were sized to reflect aggregated relative area for individual wetlands (1, 3, or 5% of the subbasin area) and shape (i.e., shallow marsh versus deep pond). Near-channel management actions included fluvial wetlands, ravine stabilization, and toe protection for banks and bluffs. Ravine stabilization and toe protection for banks or bluffs both reduce the magnitude of near-channel contributions to S (40). Previous research has identified 106 ravines and 480 mapped bluff or exposed banks within the LSRB (64). Fluvial wetlands reduce N by increasing removal rates and reduce S by reducing the magnitude of peak streamflow. Similar to isolated wetlands, the number of modeled fluvial wetlands was constrained by SWAT to one per subbasin. Fluvial wetlands were further specified by aggregated size (70, 450, or 1,700 ha and shape [marsh versus pond]). Spatially explicit costs were assigned to each management action within the candidate watershed. These costs included land opportunity costs modeled using a real options analysis (finding a critical payment sufficient for private landowners to devote their land to wetlands), construction, engineering and maintenance costs, and losses due to yield reduction. Further details describing the representation and costs of management actions are provided in *SI Appendix*.

Optimization Framework. A multiobjective evolutionary optimization algorithm (MOEA) was used to identify watershed portfolios that most cost-effectively satisfied simultaneous targets for cost, N load reduction, and S load reduction. We used an elitist modification of a strength Pareto evolutionary algorithm 2 (SPEA2) algorithm, in which nondominated solutions are maintained in the archive (65–67), to solve the multiobjective optimization problem. We followed the recent work of Lang et al. to overcome the "curse of dimensionality" in large-scale MOEAs (68). Evolutionary algorithm iterations were stopped upon reaching the consolidation ratio of 0.9 (68). P load was not included as an optimization target due to insufficient understanding of near-channel P dynamics and legacy storage (16, 20, 69). Optimizing field-derived P only without an adequate representation of near-channel storage and generation processes would not reflect true P load reductions.

Data Availability. Optimization genome and output file data have been deposited in Open Science Framework (DOI 10.17605/OSF.IO/JEMKN) (67).

ACKNOWLEDGMENTS. This research was funded by the NSF under the Water Sustainability and Climate Program (WSC) through an Observatory grant (EAR-1209402): REACH (Resilience under Accelerated Change), a WSC Science, Engineering and Education for Sustainability Fellows grant (EAR-1415206) to A.T.H., US Department of Agriculture–Agriculture and Food Research Initiative (USDA AFRI) National Institute of Food and Agriculture grant to S.R., and a USDA National Resource Conservation Service grant to P.B. (CPT0011193). Special thanks to Iowa State's Center for Agriculture and Rural Development for model integration and executing the optimization algorithm.

1. E. G. Stets *et al*., Landscape drivers of dynamic change in water quality of U.S. rivers. *Environ. Sci. Technol.* **54**, 4336–4343 (2020).
2. K. J. Van Meter, P. Van Cappellen, N. B. Basu, Legacy nitrogen may prevent achievement of water quality goals in the Gulf of Mexico. *Science* **360**, 427–430 (2018).
3. G. P. Oelsner, E. G. Stets, Recent trends in nutrient and sediment loading to coastal areas of the conterminous U.S.: Insights and global context. *Sci. Total Environ.* **654**, 1225–1240 (2019).
4. K. L. Blann, J. L. Anderson, G. R. Sands, B. Vondracek, Effects of agricultural drainage on aquatic ecosystems: A review. *Crit. Rev. Environ. Sci. Technol.* **39**, 909–1001 (2009).
5. P. M. Vitousek *et al*., Human alteration of the global nitrogen cycle: Sources and consequences. *Ecol. Appl.* **7**, 737–750 (1997).
6. P. Belmont *et al*., Large shift in source of fine sediment in the upper Mississippi river. *Environ. Sci. Technol.* **45**, 8804–8810 (2011).
7. E. Foufoula-Georgiou, Z. Takbiri, J. A. Czuba, J. Schwenk, The change of nature and the nature of change in agricultural landscapes: Hydrologic regime shifts modulate ecological transitions. *Water Resour. Res.* **51**, 6649–6671 (2015).
8. N. N. Rabalais, R. E. Turner, W. J. Wiseman, Gulf of Mexico hypoxia, a.k.a. "The Dead Zone". *Annu. Rev. Ecol. Syst.* **33**, 235–263 (2002).
9. M. H. Ward *et al*., Drinking water nitrate and human health: An updated review. *Int. J. Environ. Res. Public Health* **15**, 1–31 (2018).
10. B. L. Keeler *et al*., Linking water quality and well-being for improved assessment and valuation of ecosystem services. *Proc. Natl. Acad. Sci. U.S.A.* **109**, 18619–18624 (2012).
11. A. Wagenhoff, J. E. Clapcott, K. E. M. Lau, G. D. Lewis, R. G. Young, Identifying congruence in stream assemblage thresholds in response to nutrient and sediment gradients for limit setting. *Ecol. Appl.* **27**, 469–484 (2017).
12. L. A. Sprague, R. M. Hirsch, B. T. Aulenbach, Nitrate in the Mississippi river and its tributaries, 1980 to 2008: Are we making progress? *Environ. Sci. Technol.* **45**, 7209–7216 (2011).
13. U.S. Department of Agriculture, Natural Resources Conservation Service, Effects of conservation practices on nitrogen loss from farm fields: A national assessment based on the 2003-06 CEAP survey and APEX modeling databases. (2017). https://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcseprd1365657.pdf. Accessed 13 June 2019.
14. U.S. Department of Agriculture, Natural Resources Conservation Service, Effects of conservation practices on water erosion and loss of sediment at the edge of the field: A national assessment based on the 2003-06 CEAP survey and APEX modeling databases. (2017). https://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcseprd1365654.pdf. Accessed 13 June 2019.
15. J. O. Goyette, E. M. Bennett, R. Maranger, Low buffering capacity and slow recovery of anthropogenic phosphorus pollution in watersheds. *Nat. Geosci.* **11**, 921–925 (2018).
16. R. J. Dodd, A. N. Sharpley, Conservation practice effectiveness and adoption: Unintended consequences and implications for sustainable phosphorus management. *Nutr. Cycl. Agroecosyst.* **104**, 373–392 (2016).
17. A. Lintern, L. McPhillips, B. Winfrey, J. Duncan, C. Grady, Best management practices for diffuse nutrient pollution: Wicked problems across urban and agricultural watersheds. *Environ. Sci. Technol.* **54**, 9159–9174 (2020).
18. S. P. Schottler *et al*., Twentieth century agricultural drainage creates more erosive rivers. *Hydrol. Processes* **28**, 1951–1961 (2014).
19. E. Sinha, A. M. Michalak, and V. Balaji, Eutrophication will increase during the 21st century as a result of precipitation changes. *Science* **357**, 405–408 (2017).
20. C. L. Dolph *et al*., Phosphorus transport in intensively managed watersheds. *Water Resour. Res.* **55**, 9148–9172 (2019).
21. D. Osmond *et al*., Improving conservation practices programming to protect water quality in agricultural watersheds: Lessons learned from the National Institute of Food and Agriculture-Conservation Effects Assessment Project. *J. Soil Water Conserv.* **67**, 122A–127A (2012).
22. L. A. Kurkalova, Cost-effective placement of best management practices in a watershed: Lessons learned from conservation effects assessment project. *J. Am. Water Resour. Assoc.* **51**, 359–372 (2015).
23. USGAO, "Agricultural conservation: USDA's environmental quality incentives program could be improved to optimize benefits, GAO-17-225" (2017). https://www.gao.gov/assets/gao-17-225.pdf. Accessed 13 June 2019.
24. H. E. Golden *et al*., Non-floodplain wetlands affect watershed nutrient dynamics. *Environ. Sci. Technol.* **53**, 7203–7214 (2019).
25. A. T. Hansen, C. L. Dolph, E. Foufoula-Georgiou, J. C. Finlay, Contribution of wetlands to nitrate removal at the watershed scale. *Nat. Geosci.* **11**, 127–132 (2018).
26. F. Y. Cheng, K. J. Van Meter, D. K. Byrnes, N. B. Basu, Maximizing US nitrate removal through wetland protection and restoration. *Nature* **588**, 625–630 (2020).
27. A. A. Vaughan, P. Belmont, C. P. Hawkins, P. Wilcock, Near-channel versus watershed controls on sediment rating curves. *J. Geophys. Res. Earth Surf.* **122**, 1901–1923 (2017).
28. USDA-NRCS, Minnesota Payments Schedules. https://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/programs/financial/?cid=nrcseprd1328245. Accessed 5 December 2020.
29. C. Lenhart *et al*., Agricultural BMP Handbook for Minnesota 2017 (2017). https://wrl.mnpals.net/islandora/object/WRLrepository:2955. Accessed 4 September 2020.
30. C. Wellen, A.-R. R. Kamran-Disfani, G. B. Arhonditsis, Evaluation of the current state of distributed watershed nutrient water quality modeling. *Environ. Sci. Technol.* **49**, 3278–3290 (2015).
31. T. L. Veith, M. L. Wolfe, C. D. Heatwole, Optimization procedure for cost effective BMP placement at a watershed scale. *J. Am. Water Resour. Assoc.* **39**, 1331–1343 (2003).
32. M. Arabi, R. S. Govindaraju, M. M. Hantush, Cost-effective allocation of watershed management practices using a genetic algorithm. *Water Resour. Res.* **42**, 1–14 (2006).
33. C. Maringanti, I. Chaubey, M. Arabi, B. Engel, Application of a multi-objective optimization method to provide least cost alternatives for NPS pollution control. *Environ. Manage.* **48**, 448–461 (2011).
34. B. M. Gramig, C. J. Reeling, R. Cibin, I. Chaubey, Environmental and economic trade-offs in a watershed when using corn stover for bioenergy. *Environ. Sci. Technol.* **47**, 1784–1791 (2013).
35. N. K. Singh *et al*., Optimizing wetland restoration to improve water quality at a regional scale. *Environ. Res. Lett.* **14**, https://doi.org/10.1088/1748-9326/ab1827. (2019).
36. M. Bostian, G. Whittaker, B. Barnhart, R. Färe, S. Grosskopf, Valuing water quality tradeoffs at different spatial scales: An integrated approach using bilevel optimization. *Water Resour. Econ.* **11**, 1–12 (2015).
37. H. Wu, A.-X. Zhu, J. Liu, Y. Liu, J. Jiang, Best management practices optimization at watershed scale: Incorporating spatial topology among fields. *Water Resour. Manage.* **32**, 155–177 (2018).
38. A. Piemonti, M. Babbar-Sebens, S. Mukhopadhyay, A. Kleinberg, Interactive genetic algorithm for user-centered design of distributed conservation practices in a watershed: An examination of user preferences in objective space and user behavior. *Water Resour. Res.* **53**, 4303–4326 (2017).
39. J. A. Czuba, A. T. Hansen, E. Foufoula-Georgiou, J. C. Finlay, Contextualizing wetlands within a river network to assess nitrate removal and inform watershed management. *Water Resour. Res.* **54**, 1312–1337 (2018).
40. S. J. Cho, P. R. Wilcock, P. Belmont, K. B. Gran, B. F. Hobbs, Simulation model for collaborative decision making on sediment source reduction in an intensively managed watershed. *Water Resour. Res.* **55**, 1544–1564 (2019).
41. J. G. Arnold, N. Fohrer, SWAT2000: Current capabilities and research opportunities in applied watershed modelling. *Hydrol. Processes* **19**, 563–572 (2005).
42. J. Boettcher, P. Davis, Le Sueur River WRAPS report, Minnesota Pollution Control Agency. (2015). https://www.pca.state.mn.us/sites/default/files/wq-ws4-10a.pdf. Accessed 27 June 2019.
43. Minnesota Pollution Control Agency, Spending for watershed implementation projects (2019). https://www.pca.state.mn.us/water/spending-watershed-implementation-projects. Accessed 12 February 2021.
44. C. F. Lenhart, K. N. Brooks, D. Heneley, J. A. Magner, Spatial and temporal variation in suspended sediment, organic matter, and turbidity in a Minnesota prairie river: Implications for TMDLs. *Environ. Monit. Assess.* **165**, 435–447 (2010).
45. K. B. Gran *et al*., The power of environmental observatories for advancing multidisciplinary research, outreach, and decision support: The case of the Minnesota River Basin. *Water Resour. Res.* **55**, 3576–3592 (2019).
46. W. P. Anderson *et al*., The Minnesota Nutrient Reduction Strategy (2013). https://www.pca.state.mn.us/water/minnesota-nutrient-reduction-strategy. Deposited 11 March 2016.
47. C. G. Homer *et al*., Completion of the 2011 National Land Cover Database for the conterminous United States - Representing a decade of land cover change information. *Photogramm. Eng. Remote Sensing* **81**, 345–354 (2015).
48. E. McLellan *et al*., Reducing nitrogen export from the Corn Belt to the Gulf of Mexico: Agricultural strategies for remediating hypoxia. *J. Am. Water Resour. Assoc.* **51**, 263–289 (2015).
49. K. Deb, Nonlinear goal programming using multi-objective genetic algorithms. *J. Oper. Res. Soc.* **52**, 291–302 (2001).
50. D. N. Pennington *et al*., Cost-effective land use planning: Optimizing land use and land management patterns to maximize social benefits. *Ecol. Econ.* **139**, 75–90 (2017).
51. S. C. Newbold, A combined hydrologic simulation and landscape design model to prioritize sites for wetlands restoration. *Environ. Model. Assess.* **10**, 251–263 (2005).
52. Z. Qi, G. Kang, X. Wu, Y. Sun, Y. Want, Multi-objective optimization for selecting and siting the cost-effective BMPs by coupling revised GWLF model and NSGAII algorithm. *Water* **12**, 1–12 (2020).
53. Louise Berger *et al*. Inc., Costs for wetland creation and restoration projects in the glaciated Northeast for the U.S. Environmental Protection Agency, Region I. EPA Contract No. 68-05-0171. (US EPA, 1997).
54. J. McGuire, L. W. Morton, A. D. Cast, Reconstructing the good farmer identity: Shifts in farmer identities and farm management practices to improve water quality. *Agric. Human Values* **30**, 57–69 (2013).
55. J. Shortle, R. Horan, Nutrient pollution: A wicked challenge for economic instruments. *Water Econ Policy* **3**, 1650033 (2017).
56. D. J. Lewis, S. Polasky, An auction mechanism for the optimal provision of ecosystem services under climate change. *J. Environ. Econ. Manage.* **92**, 20–34 (2018).
57. Y. Kuwayama, N. Brozović, The regulation of a spatially heterogeneous externality: Tradable groundwater permits to protect streams. *J. Environ. Econ. Manage.* **66**, 364–382 (2013).

SUSTAINABILITY SCIENCE

58. E. A. Moallemi, J. Kwakkel, F. J. de Haan, B. A. Bryan, Exploratory modeling for analyzing coupled human-natural systems under uncertainty. *Glob. Environ. Change* **65**, 102186 (2020).
59. M. T. Streeter, K. E. Schilling, C. L. Burras, C. F. Wolter, Erosion and sediment delivery in southern Iowa watersheds: Implications for conservation planning. *J. Soil Water Conserv.* **76**, 103–115 (2021).
60. C. L. Kling, Economic incentives to improve water quality in agricultural landscapes: Some new variations on old ideas. *Am. J. Agric. Econ.* **93**, 297–309 (2011).
61. S. H. Hamilton, S. ElSawah, J. H. A. Guillaume, A. J. Jakeman, S. A. Pierce, Integrated assessment and modelling: Overview and synthesis of salient dimensions. *Environ. Model. Softw.* **64**, 215–229 (2015).
62. F. Lupi *et al*., Linking agricultural nutrient pollution to the value of freshwater ecosystem services. *Land Econ.* **96**, 493–509 (2020).
63. L. S. Prokopy *et al*., Adoption of agricultural conservation practices in the United States: Evidence from 35 years of quantitative literature. *J. Soil Water Conserv.* **74**, 520–534 (2019).
64. K. B. Gran, *et al*., “An integrated sediment budget for the Le Sueur River Basin” (MPCA Report wq-iw-29o., 2011).
65. S. Rabotyagov *et al*., Least-cost control of agricultural nutrient contributions to the Gulf of Mexico hypoxic zone. *Ecol. Appl.* **20**, 1542–1555 (2010).
66. M. Laumanns, E. Zitzler, L. Thiele, “On the dffects of archiving, elitism, and density based selection in evolutionary multi-objective optimization” in *Evolutionary Multi-Criterion Optimization Proceedings*, E. Zitzler et al., Eds. (Springer, Berlin, 2001), pp. 181–196.
67. P. Hawthorne, S. Rabotyagov, A. Hansen, *AgRiver model optimization – Le Sueur River Basin* 2021. Open Science Framework. http://dx.doi.org/10.17605/OSF.IO/JEMKN. Deposited 31 March 2021.
68. Z. Lang, S. S. Rabotyagov, S. J. Cho, T. D. Campbell, C. L. Kling, Good seeds bear good fruit: Using benefit-to-cost rations in multi-objective spatial optimization under epistasis. *Land Econ.* **96**, 531–551 (2020).
69. R. M. Records, E. Wohl, M. Arabi, Phosphorus in the river corridor. *Earth Sci. Rev.* **158**, 65–88 (2016).