Environmental Management (2019) 64:313–328
https://doi.org/10.1007/s00267-019-01190-7

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Using Spatially Targeted Conservation to Evaluate Nitrogen Reduction and Economic Opportunities for Best Management Practice Placement in Agricultural Landscapes

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Received: 15 October 2018 / Accepted: 18 July 2019 / Published online: 31 July 2019

Abstract

The US Cornbelt leads North American production of intensively managed, row-crop corn and soybeans. While highly productive, agricultural management in the region is often linked with nonpoint source nutrient pollution that negatively impacts water quality. Presently, conservation programs designed to install best management practices (BMPs) to mitigate agricultural nonpoint source pollution have not been targeted to those areas of the landscape that contribute disproportionately to surface water quality concerns. We used an innovative spatially targeted conservation protocol coupled with a GIS-based landscape planning tool to evaluate the cost and effect on water quality from nitrate-nitrogen loss under alternative landscape scenarios in an Iowa watershed. Outputs indicate large reductions in watershed-level nitrate-nitrogen loss could be achieved through coordinated placement of BMPs on high-contributing parcels with limited reduction of cultivated land, resulting in improved surface water quality at relatively low economic costs. For example, one scenario, which added wetlands, cover crops, and saturated buffers in the watershed, required the removal of <5% of cultivated area to reduce nitrate-nitrogen loss by an estimated 49%, exceeding the Iowa Nutrient Reduction Strategy goal for enhancing water quality. Annualized establishment and management costs of landscape scenarios that met the nonpoint source nitrogen reduction goal varied from \$3.16 to \$3.19 million (2017 US dollars). These results support our hypothesis that water quality can be improved by targeting high-contributing parcels, and highlights the potential to minimize tradeoffs by coupling targeted conservation and planning tools to help stakeholders achieve water quality outcomes within agricultural landscapes.

Keywords Water quality · Best management practices · Spatially targeted conservation · Landscape planning · US Cornbelt

Introduction

Land cover in the United States (US) Cornbelt region is dominated by row crops, primarily corn and soybean (USDA NASS 2017). Collectively in 2017, the market value of corn and soybeans produced in US Cornbelt states was nearly 96 billion US dollars (USDA NASS 2017). Across millions of individual farm fields, the US Cornbelt has been designed, constructed, and managed for the production of these low cost, high-value commodities—yet, obtaining the highest yields at the lowest cost at field scales often compromises landscape-scale ecosystem services, such as maintaining water quality for downstream uses. Extensive annual cropping systems, tillage, and artificial subsurface drainage, along with application of synthetic nitrogen, combine in ways that lead to large nitrate-nitrogen (N) contributions to surface water in the Mississippi River Basin (Schilling and Libra 2000; Petrolia and Gowda 2006; Jones et al. 2018). Excess nitrate-N in surface water is problematic for several reasons. First, nitrate-N in excess of the United States Environmental Protection Agency (USEPA) drinking water standard of 10 mg/L is a human health hazard because it can cause infantile methemoglobinemia (Comly 1945; Johnson, Kross 1990), and has been associated with other human health risks, including cancer (Weyer et al. 2001). Excess nitrate-N in surface water can

Supplementary information The online version of this article (https://doi.org/10.1007/s00267-019-01190-7) contains supplementary material, which is available to authorized users.

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also be ecologically damaging, leading to local eutrophication and regional hypoxia zones (Alexander et al. 2008; Conley et al. 2009). For example, excess nitrate-N from the US Cornbelt Mississippi River Basin has contributed to the annual development of the Gulf of Mexico hypoxic zone (Alexander et al. 2008), the largest of which was recorded during the summer of 2017 (USGS 2017). Globally, over the past 60 years, hypoxic zones in coastal regions have increased exponentially (Diaz and Rosenberg 2008; Rabalais et al. 2010; Conley et al. 2011; Rabotyagov et al. 2014). Similarly to the US Cornbelt, causes of global hypoxic zones are often closely associated with agricultural watersheds that export high amount of nutrients (i.e., nitrogen and phosphorus; Rabalais et al. 2010). Developing policies and tools to aid scientists and ecosystem managers to reduce nutrient export to coastal hypoxic zones provides an opportunity to enhance valuable ecosystem services (e.g., Turner et al. 1999).

In response to local and regional water quality impacts, the Mississippi River Gulf of Mexico Watershed Nutrient Task Force created the Gulf of Mexico Hypoxia Action Plan in 2008 to address the issue (Mississippi River Gulf of Mexico Watershed Nutrient Task Force 2008). The goal of the Gulf of Mexico Hypoxia Action Plan is to advance technology and policy designed to reduce the amount of nitrogen reaching the Gulf of Mexico by 45% with the stated goal of reducing the size of the hypoxic zone to <3219 km^2 (MRGMWNTF 2008). To accomplish this goal, the Action Plan articulated that US Cornbelt states develop state-level nutrient reduction strategies (MRGMWNTF 2008). In accordance with this directive, the Iowa Nutrient Reduction Strategy (Iowa NRS) was released in 2013 with the goal of reducing nutrients in surface water from both point and nonpoint sources in a scientific, reasonable, and cost-effective manner (IDALS et al. 2017a, 2017b). The Iowa NRS calls for a reduction in nonpoint source nitrogen pollution of 41% (IDALS et al. 2017a, 2017b). To meet this reduction goal, the Iowa NRS recommends that combinations of in-field, edge-of-field, and downstream best management practices (BMPs; e.g., nutrient management, cover crops, filter strips, buffers, wetlands, perennials, etc.) be strategically placed using spatially targeted conservation approaches (Iowa Department of Agriculture and Land Stewardship, Iowa Department of Natural Resources, and Iowa State College of Agriculture and Life Sciences 2017).

BMPs, particularly those with diverse, perennial plant communities, provide other environmental benefits in addition to nutrient retention, such as nongame wildlife habitat, pollinator resources, etc. (e.g., Schulte et al. 2017). Spatially targeted conservation is a coordinated approach to implementing select BMPs on specific fields identified as being significant contributors to nutrient loads within a watershed due to a combination of land management practices (e.g., cropping system, tillage, synthetic fertilizer) and biophysical vulnerabilities associated with soil, slope, and proximity to stream (Berry et al. 2005; Secchi et al. 2008). Spatially targeted conservation approaches make use of state-of-the-art geospatial planning tools (e.g., Agricultural Conservation Planning Framework Toolbox—ACPF) and/or hydrologic models, (e.g., Agricultural Policy Environmental eXtender—APEX) to provide a complete picture of nutrient transport on the landscape across large spatial extents. The geospatial findings from these tools can be combined with conservation management frameworks that aid in identifying a suite of opportunities for the application of effective, suitable, and practical BMPs at the lowest cost for landowners (e.g., Tomer et al. 2013; McLellan et al. 2018).

Prior research has demonstrated that states have historically lagged in attaining water quality goals specifically, because BMP application to date has neither been spatially targeted to critical sources and/or pathways of contamination nor applied in accordance with watershed-scale hydrologic considerations (Tomer and Locke 2011). Fortunately, the technical capacity for land management agencies and/or watershed-level entities and allied stakeholders to spatially target BMPs based on high-resolution geospatial analysis is steadily increasing (e.g., Walter et al. 2007; Schilling and Wolter 2009; White et al. 2014; Tomer et al. 2015). Yet comprehensively tracking the cost of BMP application has been a challenge largely because up-to-date data regarding the direct and potential opportunity costs of BMP use is lacking (Tyndall and Roesch 2014). Inadequate cost information, commensurate financial support, and limited decision support have contributed to constraints on landowner adoption of BMPs (Lemke et al. 2010; Tyndall and Roesch 2014; Arbuckle, Roesch-McNally 2015; Zimmerman et al. 2019). The direct and opportunity costs associated with individual BMPs in Iowa alone can be significant, and when applied at watershed scales total costs have been roughly estimated to be in the hundreds of millions of dollars annually (e.g., Rabotyagov et al. 2014). Compared with previous Federal farm bill legislation, the 2014 farm bill significantly reduced total conservation funding and limited the total number of programs that support technical service and conservation planning relevant to BMP application (Claassen 2014). Thus, state-level nutrient reduction strategies will need to be operationalized with cost-effectiveness (e.g., highest environmental gain per dollar spent) as a central component of comparing and selecting implementation strategies (Claassen, Ribaudo 2016). This type of combined hydrologic and cost information is required to more accurately guide understanding of conservation funding needs, and provide policy-oriented technical information required for cost- and outcome-effective implementation of nutrient reduction strategies at regional scales (Duke et al. 2013).

In an effort to strengthen the implementation of nutrient reduction strategies, the purpose of this research was to demonstrate a spatially targeted conservation approach in the Upper Big Creek watershed of central Iowa, USA. We use the Agricultural Conservation Planning Framework Toolbox (ACPF), an innovative tool developed by the United States Department of Agriculture (USDA) designed to guide spatially targeted conservation planning, to assess biophysical and economic opportunities for strategic placement of BMPs. We specifically sought to build on the current ACPF by developing new methods to identify areas at high risk for nitrate-N leaching associated with artificial subsurface drainage and for low opportunity costs (i.e., direct costs and forgone income associated with removing land from cultivation to implement BMPs). We applied these new methods to demonstrate the novel use of the ACPF to examine nitrate-N reduction and costs associated with watershed-scale applications of three BMPs: cover crops, saturated buffers, and reconstructed wetlands, which, respectively, address in-field, edge-of-field, and downstream placement of practices within watersheds.

Methods

Study Location

Big Creek watershed is composed of two adjacent HUC-12 watersheds, which extend 20,218 ha across the Des Moines Lobe in central Iowa (Fig. 1a): HUC 071000040801 (hereafter, East Big Creek) and HUC 071000040802 (hereafter, West Big Creek). The topography of the Des Moines Lobe is characterized by low-relief landscapes covered by rich, loamy glacial till (Prior 1991). Historically, the watersheds were composed of highly diverse prairie, savanna, and deciduous forest communities. Today, ~95% of the Upper Big Creek watershed is dedicated to agricultural production. Approximately 96% of agricultural lands are dedicated to crops, and 4% of agricultural land is pasture. Cattle production is prevalent in pastures along stream reaches (Fig. 1b). There are an estimated 530 cattle and 10,862 hogs in confinement in the watershed (Graham 2011). Homesteads and developed areas account for 3.5% of the total landscape in the watershed.

Big Creek watershed is drained by two major streams that discharge into Big Creek Lake. The 306-ha lake is surrounded by a 607-ha State Park and an 830-ha Wildlife Management Area. Located just 32 -km north of the Des Moines metropolitan area, Big Creek State Park receives ~7,20,000 visits per year, which contribute over $40 million to the local economy (Otto et al. 2012). Together, Big Creek State Park and the Wildlife Management Area provide important amenities to their visitors, including a 1.4-ha swimming beach, a marina, and several boat ramps. In addition to its recreational value, water from Big Creek Lake drains into Saylorville Lake, a reservoir located on the Des Moines River, through the Big Creek Spillway. The Des Moines River is one of two primary surface water sources for 5,00,000 water consumers in the Des Moines metropolitan area. The proximity of Big Creek State Park to Des Moines, coupled with its amenities, make the location a valuable recreational resource for swimming, boating, and fishing and an important drinking water resource in central Iowa.

Big Creek Lake was listed on the USEPA 303(d) List of Impaired Waters in 2006 for levels of pathogen indicator bacteria in excess of Iowa's Water Quality Standards (WQS) and in 2016 for levels of cyanobacteria in excess of Iowa's WQS (Iowa Department of Natural Resources 2016). In 2011, a Watershed Quality Improvement Plan (WQIP) and a Total Maximum Daily Load (TMDL) were completed for high concentrations of the pathogen indicator bacteria *Escherichia coli* (*E. coli*) to restore the designated use of the waterbody (Graham 2011). The pathogen indicator bacteria, *E. coli*, can be attributed to wildlife, livestock manure, and poorly functioning septic systems in the watershed (Graham 2011). In addition to cyanobacteria and *E. coli*, the Des Moines Water Works, the public utility charged with providing water to 5,00,000 consumers, has experienced difficulties removing excess nitrate-N pollution from surface water sources. During 2014 and 2015, nitrate-N was observed at levels two to three times the EPA drinking water standard of 10 mg L^{-1} nitrate-N in the Des Moines River watershed and its tributaries (Iowa Water Quality Information System 2019; Iowa Department of Natural Resources 2019). Because of these recreation, aquatic life, and drinking water concerns, the watershed has been the focus of ongoing research related to declining environmental quality and subsequent loss of ecosystem services, primarily of water quality related to drinking water, recreational, and aquatic uses.

Landscape Planning Model: Agricultural Conservation Planning Framework (ACPF)

The Agricultural Conservation Planning Framework (ACPF) is a relatively easy to use GIS-based planning tool designed to provide resource agencies, technical advisors, and farmers with comprehensive information required to target conservation and production opportunities at field and watershed scales (Tomer et al. 2013; Tomer et al. 2015; Tomer et al. 2015a). The planning tool systematically assesses the watershed at the field level, and indicates locations throughout the watershed where specific BMPs would be most appropriate and where opportunities may exist to reduce nutrient and sediment loss (Tomer et al. 2013).

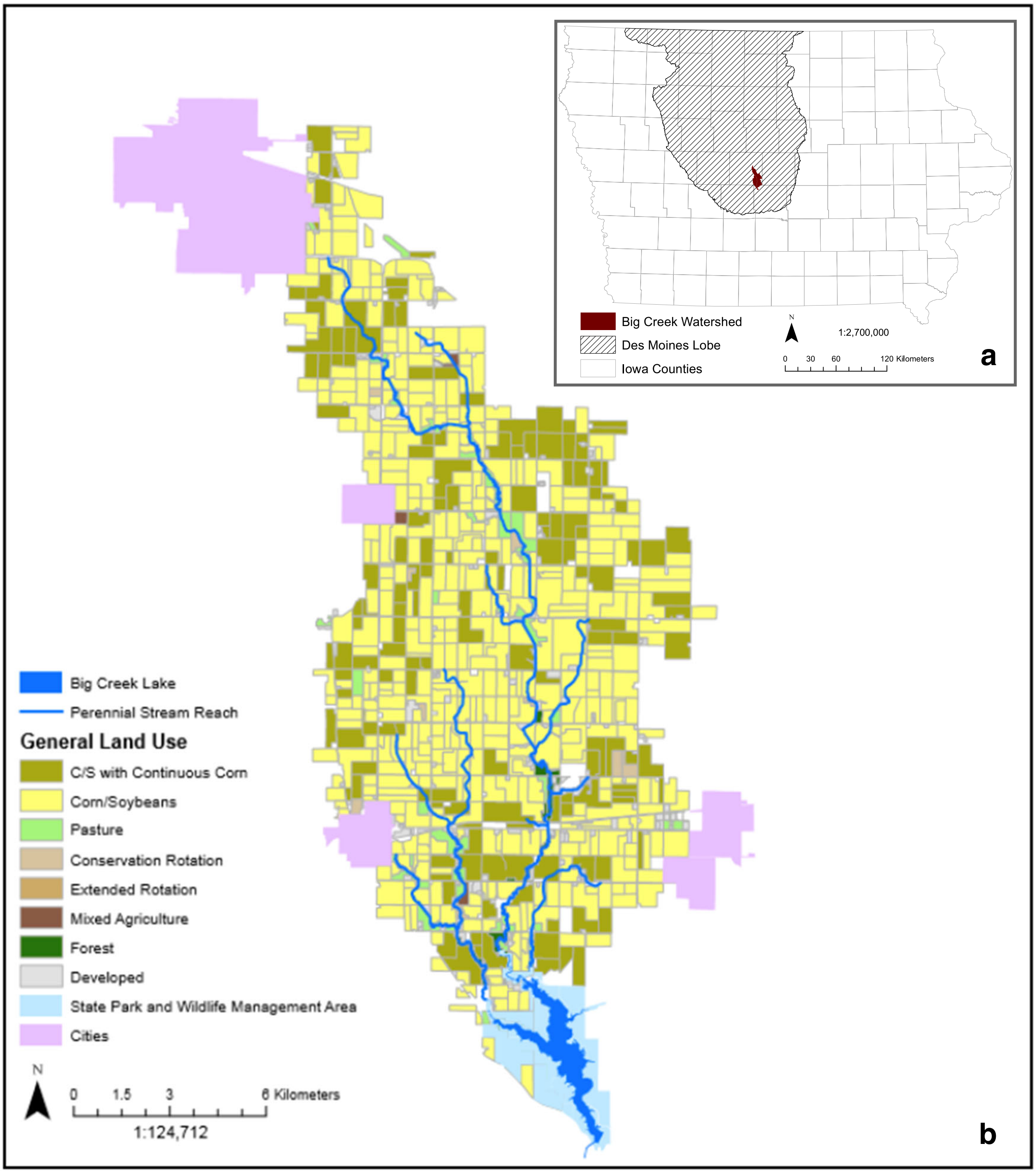

Fig. 1 **a** Des Moines Lobe geological formation (striped), Big Creek watershed (brown) within Iowa, USA. **b** By-parcel land use in Big Creek watershed. Parcel data originated from the publically available Common Land Unit data from 2006. The data layer was updated in 2013 by the USDA ARS to reflect changes in parcels. Land-use data from 2014 were extracted from USDA NASS Cropland Data Layers and aggregated by USDA ARS. State Park and Wildlife Management Areas are from the Conservation and Recreation Lands in Iowa data set published by the Iowa Department of Natural Resources in 2012

BMPs included in the toolbox version used in this study are drainage management, grassed waterways, contour buffer strips, water and sediment control basins, nutrient removal wetlands, and riparian management. While the ACPF does not include all potential BMPs (e.g., two-stage drainage ditches, living mulches, etc.), it is designed specifically for small, HUC-12 watersheds, has been applied in the US Cornbelt (e.g., Tomer et al. 2015), and is structurally and quantitatively suitable for the unique challenges (e.g., sub-surface drainage, intensive management, etc.) of an agricultural landscape specific to the US Cornbelt (Tomer et al. 2013; Tomer et al. 2015). ACPF uses a three-step process: (1) stream network development, (2) field characterization, and (3) precision BMP siting (Fig. 2; Tomer et al. 2013). We used ACPF V1_beta, released in summer 2015 for this research.

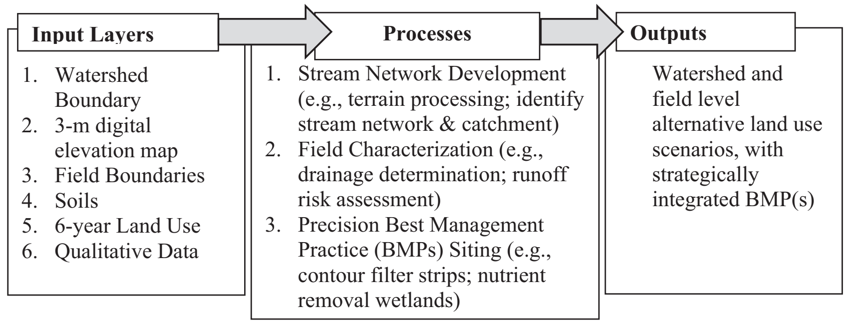

Fig. 2 Simplified schematic of Agricultural Conservation Planning Framework (ACPF; Tomer et al. 2013)

Geospatial Data

We used geospatial data to identify biophysical and economic opportunities for strategic placement of BMPs. Base data layers were generated and aggregated by the USDA Agricultural Research Service (ARS; Porter et al. 2015; Tomer et al. 2017) and downloaded for Big Creek watershed. Primary input data layers used were: (1) watershed boundary layer; (2) field boundary layer; (3) soil data layer and tables; (4) land cover data layers and tables; and (5) digital elevation model (DEM) layer. The watershed boundary layer is a polygon data layer derived from the United States Geological Survey (USGS) National Hydrography Dataset (NHD). The field boundaries layer is a polygon data layer of agricultural field boundaries that were manually updated from publicly available 2005 USDA Farm Service Agency (FSA) data. The soils data layer is a raster data layer derived from the USDA Natural Resource Conservation Service (NRCS) gridded gSSURGO database. Customized tables containing data on surface horizon, surface texture, and soil profile data are provided alongside the soils data layer. The land cover data layers are raster USDA National Agricultural Statistics Service (NASS) Cropland Data Layers for the six most recent years (2009–2014). The resolution of the Cropland Data Layers are 56 m × 56 m. Customized tables containing information on the majority crop and crop history over the 6 years are provided alongside the land cover data layers (Tomer et al. 2017). An unfilled, LiDAR-derived digital elevation map (DEM) with 3-m horizontal resolution was also provided by USDA ARS. Available base data layers can be downloaded from https://www.nrrig.mwa.ars.usda.gov/st40_huc/dwnldACPF.html. Base data layers were used as inputs for the ACPF, and for developing a spatial targeting protocol for nitrate-N and opportunity cost.

Spatial Targeting for Nitrate-N and Opportunity Cost

We identified areas of the agricultural landscape at high risk for nitrate-N leaching associated with artificial subsurface drainage, and for low opportunity costs. Opportunity costs account both for the direct costs associated with BMPs (e.g., planting and terminating cover crops, wetland construction) as well as forgone income associated with removing land from cultivation to implement BMPs (e.g., wetlands, buffers). We developed methodology for prioritizing areas of the agricultural landscape at high risk for nitrate-N leaching and for relatively low land-use costs (i.e., low opportunity costs) for BMP implementation of practices that require removing land from cultivation. To first identify areas of the landscape at high risk for nitrate-N leaching, we used the soils data associated with the gSSURGO data provided by the USDA ARS to quantify the proportion of each field classified as dual drainage (classification B/D). A dual drainage classification refers to fields that have moderate infiltration when artificial subsurface tile drainage is installed. We used soils data to infer artificial subsurface drainage, because tile drainage maps are not available for this watershed. For nitrate-N loss, proportion of dual drainage values were classified into a high, medium, or low classification consistent with the ACPF structure for evaluating runoff risk (Tomer et al. 2013). In this case study, fields were categorized into three nitrate-N leaching risk bins using a 40:40:20 split—high, medium, and low—based on their individual risk values. We used a binning process to identify areas of priority in the landscape, meaning fields where conservation opportunities are greatest for reducing the largest amount of nitrate-N loss. Consequently, changing the binning classification proportions will influence the proportion of fields that are given high, medium, and low priority. This method provides flexibility in the spatial targeting protocol, and should be adjusted based on watershed characteristics and user/stakeholder interests. In this case study, fewer fields were classified in the low category because the Des Moines Lobe has limited topographic relief and fertile, yet poorly drained soils, which makes this region particularly susceptible to nitrate-N loss via artificial subsurface drainage. In addition, this case study was focused on identifying fields in a HUC-12 watershed because watershed improvement efforts are frequently targeted at the HUC-12 scale (e.g., National Water Quality Initiative), and conservation has to be operationalized at the field level (i.e., BMPs are installed in specific fields).

To identify and prioritize areas of the landscape with low opportunity costs for potential BMP placement, we calculated area-weighted opportunity costs based on Corn Suitability Rating (CSR) by soil units in the gSSURGO soil database. The CSR is a point-based, indexing approach unique to the state of Iowa used to measure potential soil productivity relative to expected corn yields based on soil profile, slope characteristics, and weather conditions; CSR is also used to set land rental rates in Iowa. In Iowa, CSR varies from 5 to 100 points, with 100 points indicating a soil most ideal for producing corn. In each field, for each soil unit, we multiplied the CSR of each soil unit by the area occupied by that soil unit. We summed those values and divided the summation by the area to quantify the area-weighted CSR for each field. We used 2009–2014 cash rental rates, adjusted for inflation to 2017 dollars, for Boone County (Iowa State University Extension and Outreach 2014) to calculate rental rate per CSR by dividing the average per hectare cash rental rate for Boone County ($617.27) by the average CSR for Boone County (83). The rental rate per CSR point ($7.43) was multiplied by the area-weighted CSR for each field to quantify by-field opportunity costs. We chose to use data on cash rental rates from 2009 to 2014 to match the years used to develop the general land cover data layer in the model. Field-based opportunity costs were classified into a high, medium, or low classification using a 40:40:20 split. Thus, fields were categorized into three opportunity cost bins—high, medium, and low—based on their individual cost values similarly to nitrate-N risk. Similar to nitrate-N, the binning process for opportunity costs was also skewed high to reflect the highly productive soils associated with central Iowa. In regions with greater soil variability, and potentially greater variability in production, the binning could be altered to classify fewer fields with high opportunity costs (i.e., fewer fields with relatively high forgone income associated with removing land from cultivation to implement BMPs). In addition, while land rental rates in Iowa have been somewhat volatile and have been trending downward, we expect relative classifications/rankings to remain the same across time because they are based on soil capability for crop production. To integrate nitrate-N leaching risk and opportunity costs for potential BMP placement, we created a simple matrix designed to evaluate these risks to prioritize areas of the agricultural landscape where BMPs will provide the largest biophysical difference (Fig. 3).

Opportunity Costs \ Nitrate-N Leaching Risk	High	Medium	Low
High	3	4	5
Medium	2	2	4
Low	1	2	3

Fig. 3 Nitrate-N leaching risk (columns) combined with opportunity costs to prioritize nitrate-N leaching and opportunity costs, allowing for by-field prioritization based on biophysical vulnerability and cost. In the matrix, numbers correspond with by-field prioritization order, where a number one indicates low costs and high vulnerability for nitrate-N leaching and a number five indicates high costs and low biophysical vulnerability. In a prioritization protocol, fields classified with a number one would be highest priority, while fields classified with a number five would be lowest priority

Development of Spatially Targeted BMPs

Specifically, we used ACPF to examine BMPs designed to address nitrate-N loss and associated opportunity costs at in-field, edge-of-field, and downstream positions; cover crops, saturated buffers, and nutrient removal wetlands. These practices were chosen for this study because they reduce losses of nitrate-N, phosphorus, and sediment to surface water, are representative of in-field, edge-of-field, and downstream BMPs, and have been strongly promoted as part of the Iowa NRS (Iowa Department of Agriculture and Land Stewardship, Iowa Department of Natural Resources, and Iowa State College of Agriculture and Life Sciences 2017a). Saturated buffers have been shown to remove nitrate-N carried in artificial subsurface drainage (Jaynes and Isenhart 2014). Saturated buffers divert tile-drained, nitrate-N laden water into the soil profile beneath riparian buffers. This raises the water table, providing an anaerobic, organic matter-rich medium for denitrification to occur. Nutrient removal wetlands have been shown to reduce nitrate-N (Helmers et al. 2008) by creating residence times and anaerobic conditions favorable to denitrification. These BMPs have also been shown to reduce phosphorus loss and *E. coli* to surface water by stabilizing the soil surface to prevent wind and water erosion detachment and transport of sediment and sediment-bound nutrients and contaminants and by trapping and retaining transported sediment and sediment-bound nutrients and contaminants (Lee et al. 2000; Dinnes 2004; Knox et al. 2008). Cover crops utilize water-soluble nitrate-N for growth during fall, winter, and spring when crops are not present, and have the potential to decrease nitrate-N export in agricultural landscapes (Kaspar et al. 2007, 2012).

We used ACPF to site saturated buffers and nutrient removal wetlands within East and West Big Creek watersheds. Similar to Tomer et al. (2015), we developed a method to site saturated buffers in locations adjacent to a perennial stream reach receiving a low amount of runoff, with a high water table, and where at least 50% of the soil had an organic matter content >4%. The amount of runoff and the water table level classifications are calculated and assigned categorical values using the ACPF riparian function assessment (RAP) tool (Porter et al. 2015). Using the ACPF riparian buffer tool, saturated buffers were placed in

250-m long, RAPs along perennial stream reaches, which met the above criteria. The area of each of each of the saturated buffers was calculated by multiplying the length of the saturated buffer adjacent to the stream by 30 m to represent the width of the associated grassed riparian buffer. Belt et al. (2014) demonstrated that buffers with widths >30 m had diminishing effects on nitrate-N treatment; nonetheless, we recognize that this assumption is non-dynamic, and in practice variable-width buffers would need to be based on site-specific flow patterns and stream morphology (Dosskey et al. 2005). The present version of the ACPF saturated buffer tool (version 3) is much more sophisticated, and accounts for many of these site-specific components that this early exercise does not. Grassed riparian buffers and saturated buffers were assumed to be constructed simultaneously (i.e., no grassed riparian buffer was assumed to be present prior to the saturated buffer; the land was assumed to be cropped). Key parameters for sited wetlands were: (1) depressional areas with a drainage range of >60 ha—maximum watershed drainage where a wetland could be sited; (2) an impoundment height of 0.9 m measured from the top of the bank; (3) a buffer height of 1.5 m measured from the top of the wetland pool; (4) a pooled area/drainage area ratio of 0.5–2; and (5) a buffer area/pooled area ratio <4.0. The ACPF searched suitable locations and returned possible wetland locations for consideration. To site wetlands, the ACPF generates potential impoundment locations along flow pathways (Porter et al. 2015). Potential locations are sorted by contributing area, and at each location an impoundment simulated in the DEM, creating a wetland and wetland buffer above the impoundment (Porter et al. 2015). The key parameters listed above are evaluated, and if the wetland meets those suitability criteria, it is added to the feature class output for selection in the scenario (Porter et al. 2015). The location of wetlands and saturated buffers was drainage-ownership independent. Cover crops were considered suitable on any hectares that had a general land cover of corn followed by corn and corn/soybean rotation. Based on spatial targeting and prioritization of fields and areas identified using the ACPF and biophysical data, alternative landscape scenarios were created using the three BMPs described above.

Evaluating Nitrate-N Reduction and Costs

Nitrate-N reduction from installing BMPs in alternative landscape scenarios were evaluated using the spreadsheet method described by Tomer et al. (2015). This approach assumes that stacked BMPs will have a multiplicative effect on nitrate-N reduction (Lazarus et al. 2014). Each field was represented in the spreadsheet as a row. Columns were used to represent the proportion, or relative size, of each field in the watershed; the impact of crop rotation on nutrient losses; and each BMP included in each scenario. Consistent with Tomer et al. (2015), crop rotation values were varied as a proportion between 0.9 and 1.10 based on hypothesized nutrient application and leakiness of each cropping system. Cropping systems incorporating more crop rotation were given lower values while cropping systems with a greater frequency of corn followed by corn were given higher values, reflecting higher nutrient applications (Helmers et al. 2012). Cells in BMP columns were populated with (1-E), where E represented the average nitrate-N removal efficiency of each BMP. Average nitrate-N removal efficiencies for nutrient removal wetlands, cover crops, and saturated buffers were 50%, 30%, and 90%, respectively (Iowa Department of Agriculture and Land Stewardship, Iowa Department of Natural Resources, and Iowa State College of Agriculture and Life Sciences 2017). Absence of a BMP in or below a field was represented with a value of 1. Using the spreadsheet method to calculate the hypothesized nitrate-N reduction for each of the alternative landscape scenarios at the watershed level, as follows:

1. For each field (row), relative size of field was multiplied by the crop rotation value. These products were summed to represent the current nitrate-N loss at the watershed level.
2. For each field (row), the product of relative size of field and crop rotation value was multiplied by the BMP column, where cells reflect the absence of a BMP (1) or presence of a BMP in or below the field (1-E) in the alternative landscape scenario. These products were summed to represent nitrate-N loss at the watershed level under an alternative landscape scenario, with the given BMP(s).
3. Nitrate-N losses under the alternative landscape scenario was divided by the sum of current nitrate-N loss at the watershed level to represent the fraction of nitrate-N load reduction under the alternative landscape scenario.

Costs of installing and managing the BMPs were calculated using the framework presented in the Iowa Nutrient Reduction Strategy Decision Support Tool (Tyndall and Bowman 2016) and from Roley et al. (2016), and were updated to 2017 US dollars. For cover crops, direct cost was estimated at \$158 ha^{-1} for cereal rye winter cover crop establishment and termination. Cover crops were assumed to have no effect on subsequent crop yields (e.g., Marcillo and Miguez 2017). Annualized costs for saturated buffers and wetlands were calculated using a 4% real discount rate and a 20-year management horizon. Forgone income associated with removing land from cultivation to implement saturated buffers and wetlands was calculated based on area-weighted opportunity costs using CSR by soil units in the gSSURGO soil database. Similarly to opportunity costs calculated for field prioritization, for each saturated buffer and for each wetland, we multiplied the CSR of each soil unit by the area occupied by that soil unit. We summed those values and divided the summation by the area to quantify the area-weighted CSR for each saturated buffer and for each wetland. We multiplied the rental rate per CSR point ($7.43) calculated for Boone County by the area-weighted CSR for each saturated buffer and each wetland to quantify by-individual saturated buffer and by-individual wetland opportunity costs. Rental rates in cropland converted to saturated buffers varied from $232 ha^{-1} to $624 ha^{-1}. The average rental rate for cropland converted to saturated buffers was $550 ha^{-1}. Rental rates in cropland converted to wetlands varied from $475 ha^{-1} to $628 ha^{-1}. The average rental rate for cropland converted to wetlands was $600 ha^{-1}. For saturated buffers, annualized establishment and management costs for the grassed riparian area (excluding per ha rental rates) were estimated at $21 ha^{-1}, and saturated buffer control structures and infrastructure costs were estimated at $74 ha^{-1} (Tyndall and Bowman 2016). For nutrient removal wetlands, annualized construction and management costs (excluding per ha rental rate) were estimated at $368 ha^{-1}. To calculate the total annualized costs for saturated buffers and wetlands, we summed by-individual saturated buffer and by-individual wetland, area-weighted rental rates, and annualized establishment and management costs.

Results

The two HUC-12 watersheds comprising the Big Creek watershed are dominated by row crops and pastures. Together these agricultural land use comprise 95% of the Upper Big Creek watershed area, covering 19,256 ha. The remaining 5% of land use is predominately composed of residential areas and built infrastructure with some forest, particularly around waterbodies. In the east watershed, agricultural land use accounts for 13,141 ha of 13,734 ha (96%), and in the west watershed, agricultural lands use accounts for 6115 ha of 6483 ha (94%).

Spatial Targeting for Nitrate-N and Opportunity Cost

Using the dual drainage values in the gSSURGO soils attribute table, we classified each field into one of three nitrate-N leaching risk categories. In Upper Big Creek overall, we identified 297 fields (40%) as high risk, 298 fields (40%) as medium risk, and 150 fields (20%) as low risk for nitrate-N loss. In the east watershed, we identified 480 fields as agricultural, and evaluated them for nitrate-N leaching risk. Of those fields, 191 (40%) fields were identified as high risk, 192 (40%) fields were identified as medium risk, and 96 (20%) fields were identified as having low risk for nitrate-N leaching. In the west watershed, 266 fields were identified as agricultural, and evaluated for nitrate-N leaching risk: 106 (40%) were at high risk, 106 (40%) were at medium risk, and 54 (20%) were at low risk for nitrate-N leaching. Areas of nitrate-N leaching risk were diffuse and were often not co-located with perennial stream reaches (Fig. 4a).

By-field opportunity costs for all agricultural fields in the watersheds were classified into a high, medium, or low classification using a 40:40:20 split (Fig. 4b). All opportunity costs are expressed in 2017 US dollars. In the Upper Big Creek watershed overall, including crops and pasture, opportunity costs ranged from $190 to $681 ha^{-1}, with a mean of $590 ha^{-1}. The mean opportunity cost for cropped hectares was $595 ha^{-1}, and the mean opportunity costs for pasture hectares was $501 ha^{-1}. Overall, 297 fields (40%) had high opportunity costs, 298 (40%) fields had medium opportunity costs, and 150 (20%) fields had low opportunity costs. The mean of opportunity costs classified as high, medium, and low was $632 ha^{-1}, $593 ha^{-1}, and $502 ha^{-1}, respectively. In the east watershed, of the 480 fields identified as agricultural, opportunity costs ranged from $190 to $671 ha^{-1}, with a mean of $581 ha^{-1}. In the east watershed, 191 fields (40%) had high, 192 fields (40%) had medium opportunity costs, and 96 fields (20%) had low opportunity costs. In the west watershed, of the 266 fields were identified as agricultural, opportunity costs ranged from $341 to $681 ha^{-1}, with a mean of $602 ha^{-1}. Of these, 106 fields (40%) had high opportunity costs, 106 fields (40%) had medium opportunity costs, and 54 fields (20%) had low opportunity costs.

By integrating nitrate-N leaching risk and opportunity cost into a combined matrix (Fig. 3), we identified 30 fields (4%) as critical priority, denoted with 1 in Fig. 3 (high nitrate-N leaching risk, low opportunity costs); 164 fields (22%) as high priority, denoted with 2 in Fig. 3; 223 fields (30%) as moderate priority, denoted with 3 in Fig. 3; 303 fields (41%) as low priority, denoted with 4 in Fig. 3; and 26 fields (3%) as marginal priority, denoted with 5 in Fig. 3 (low nitrate-N leaching risk, high opportunity costs, low biophysical risk) within the Upper Big Creek watershed (Fig. 4c). In the east watershed, 16 fields (3%) were identified as critical priority; 104 fields (22%) as high priority; 151 fields (31%) as moderate priority; 193 fields (40%) as low priority; and 16 fields (3%) as marginal priority. In the west watershed, 14 fields (5%) were identified as critical priority; 60 fields (23%) as high priority; 72 fields (27%) as moderate priority; 110 fields (41%) as low priority; and 10 fields (4%) were marginal priority.

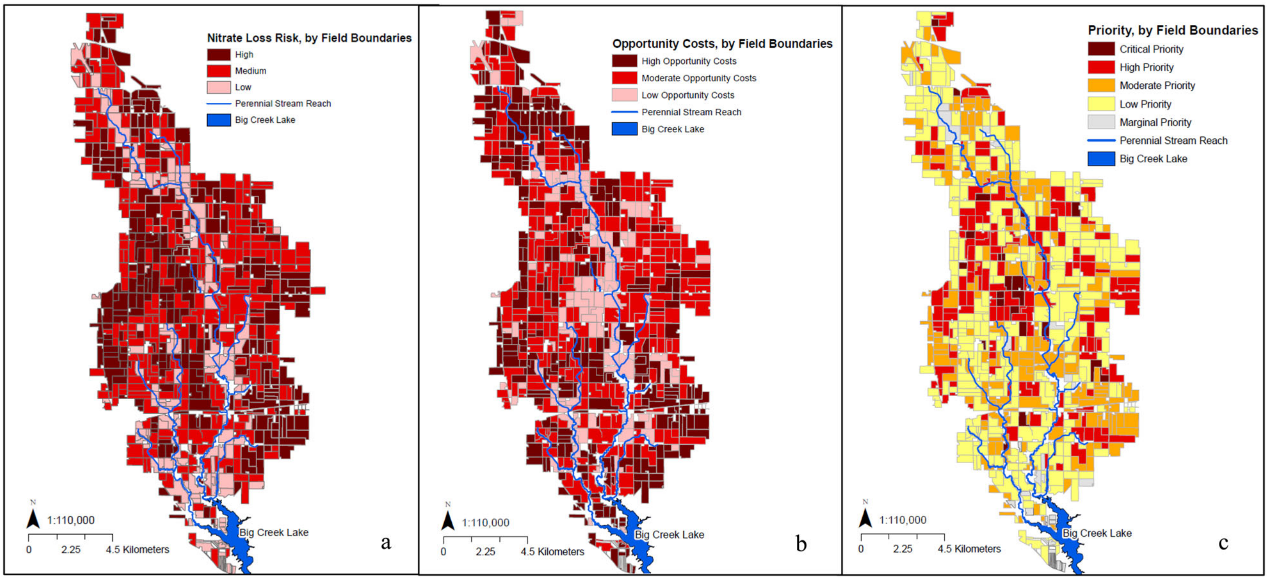

Fig. 4 a In-field nitrate-N leaching risk for Upper Big Creek watersheds, Iowa, USA. **b** Opportunity costs for Upper Big Creek watersheds, Iowa, USA. **c** Fields prioritized for the application of nutrient reduction practices, based on nitrate-N leaching risk and opportunity costs, within Upper Big Creek watersheds, Iowa. Prioritization and colors correspond to the matrix shown in Fig. 3

Evaluating Nitrate-N Reduction and Costs through Landscape Scenarios

We developed nine alternative landscape scenarios using cover crops, nutrient removal wetlands, and saturated buffers BMPs to understand options for reducing nitrate-N leaching. In this paper, we present the two scenarios that met the Iowa NRS nitrate-N reduction goal of 41% (Table 1). The remaining seven scenarios are provided in the Table A1 in the Supplementary Appendix. Alternative landscape scenarios were visualized using ACPF outputs (Fig. 5a–d). Spatial prioritization based on nitrate-N and opportunity costs aided in the selection of the best-performing wetland scenario, wherein nutrient removal wetlands were placed in headwater locations where nitrate-N leaching was expected to be high, and opportunity costs were relatively low, and wetland drainage areas corresponded heavily to critical and high priority fields (Figs. 4c, 6). Cover crops and saturated buffers were located based on the targeted detailed in the Methods section. The two scenarios that met the Iowa NRS goal for nitrate-N reduction combined cover crops, nutrient removal wetlands, and saturated buffers.

Expected nitrate-N reduction differed between the two scenarios in each of the four HUC-12 watersheds from 41 to 49% nitrate-N reduction and the extent of BMP coverage for both scenarios included BMPs on 685 (91.8% of agricultural fields in the watershed) and 690 fields (92.4% of agricultural fields in the watershed), respectively (Table 2). The amount of cultivated land removed varied between the two scenarios by <1%, from 347 ha (2% of total watershed area) to 406 ha (2% of total watershed area), respectively (Table 1). The annualized establishment and management costs of the two landscape scenarios that met the nonpoint source nitrogen reduction goal in Upper Big Creek watershed varied marginally in 2017 dollars, from $3.16 million to $3.19 million (Table 3).

Discussion

Spatially targeted conservation approaches that integrate biophysical vulnerabilities and costs offer an informed and efficient means for adapting the current row-crop agricultural landscape in the US Cornbelt to meet nutrient reduction goals. Our research in the Upper Big Creek watershed of Iowa, which is dominated by corn and soybean production, demonstrates the utility and efficiency of the USDA ACPF Toolbox, a spatially targeted conservation approach, to meet the Iowa NRS reduction goal for nonpoint source nitrate-N pollution. Using soils data from gSSURGO and opportunity costs, we developed an innovative spatial targeting protocol that prioritizes fields based on nitrate leaching potential and opportunity costs. This represents a new application of the ACPF. We used the spatial targeting protocol to guide placement of wetlands and saturated buffers the Upper Big Creek watershed, and developed nine alternative landscape scenarios that integrated cover crops, wetlands, and saturated buffers (Table 1; Table A1, Supplementary Appendix). Two of the nine scenarios exceeded the Iowa NRS goal for nonpoint source nitrate reduction of 41% and removed little land from cultivation (Table 1).

Table 1 Two alternative land-use scenarios created for the two HUC-12 watersheds in Upper Big Creek watersheds that met the Iowa NRS nitrate-N reduction goal of 41%

Watershed	Area in cover crops (ha)	Area in wetland (ha)	Area in saturated buffers (ha)
Scenario 1: Cover crops on all corn and soybean acres and headwater wetlands			
East	12, 616 ha	158 ha	0 ha
West	5,698 ha	189 ha	0 ha
Whole watershed	18,314 ha	347 ha	0 ha
Scenario 2: Cover crops on all corn and soybean areas, headwater wetlands, and saturated buffers			
East	12, 616 ha	158 ha	31 ha
West	5,698 ha	189 ha	28 ha
Whole watershed	18,314 ha	347 ha	59 ha

Note that the only difference between Scenario 1 and Scenario 2 is the addition of saturated buffers in Scenario 2

Similar to our research findings, other studies indicate the importance of using a targeted conservation protocol to efficiently place BMPs to maximize nonpoint source nutrient reduction (e.g., Tomer and Locke 2011; White et al. 2014). Combining the free-to-use ACPF toolbox combined with recently developed BMP cost data provides a comprehensive way to guide the strategic placement of BMPs to mitigate nitrate-N leaching to surface water while minimizing the costs associated with water quality protection. The methodology used in this research (1) incorporates publicly available data and tools to create spatially targeted conservation scenarios, (2) evaluates the environmental and economic outcomes from those scenarios, and (3) communicates those scenarios using visual outputs. While our evaluation was confined to the Upper Big Creek watershed, the approach can be geographically expanded and has the potential to guide watershed-level conservation planning, reduction estimation, and cost evaluation throughout the whole US Cornbelt Mississippi River Basin and other watersheds in the US (e.g., Chesapeake River watershed). Resource managers need tools and frameworks, such as the ACPF, that integrate biophysical data (e.g., elevation, soil, land cover) and socio-economic data (e.g., opportunity costs) that may be applied in other agricultural watersheds that export high levels of nutrients to coastal hypoxic areas (e.g., Baltic Sea, Black Sea, Yangtze and Pearl River Estuaries, etc.). In agricultural landscapes that do not presently have artificial subsurface drainage (e.g., Brazil), this type of spatial targeting protocol could be amended to identify and target areas where soils are well-drained, infiltration is rapid, and residence time of water, and water-soluble nitrate, is low. Moreover, our approach, of using publicly available data and tools, is well-suited for implementation of BMPs in agricultural landscapes because of their availability and adaptability for a variety of users across varying spatial extents and resolutions. Our approach could be adapted to identify appropriate locations for other BMPs that may be more suitable in other locations, with differing biophysical characteristics and socio-economic opportunities (e.g., application of living mulches, continuous living cover) that would produce varying levels of water quality outcomes.

Water quality benefits produced on-farm (e.g., installation of BMPs to reduce nitrate-N leaching to surface water) are largely experienced off-farm (e.g., higher water quality for aquatic life, recreation, and drinking). Consequently, there are many and varied stakeholders and decision-makers in the US Cornbelt region that must cooperate, over potentially long-term time scales, to reach mutually acceptable agricultural and environmental outcomes, including on-farm producers of environmental benefits and off-farm consumers of environmental benefits. This may be particularly difficult when water quality improvements from BMPs may take years to decades to realize (e.g., Meals et al. 2010; Van Meter et al. 2016).

On-farm, agricultural land management and conservation planning on private lands integrates multiple sources of information for decision-making, including guidance from traditional government services (e.g., USDA NRCS), private industries (e.g., commodity groups), nonprofit organizations (e.g., The Nature Conservancy), and others (Prokopy et al. 2014; Reimer et al. 2017). Given the extensive coverage of privately held agricultural land in Iowa and the costs associated with the scenarios that met nutrient reduction goals, land management policies, such as spatially targeted conservation, will require wide landowner and stakeholder buy-in and extensive coordination between landowners, institutions (e.g., government agencies, private agribusiness firms, nonprofit organizations, etc.), and taxpaying citizens to meet nutrient reduction goals. Publicly available data and tools (i.e., ACPF), which can be individualized for specific land owners and/or watersheds, can serve as a cross-actor tool for consistent communication and planning when various and diverse actors are involved in on-farm decision-making, which often has off-farm consequences. In the US Cornbelt, farmland owners and farmers are generally amenable to a spatially targeted conservation approach (Arbuckle 2013; Kalcic et al. 2014, 2015; Zimmerman et al. 2019), and are particularly interested in innovative incentives (e.g., alternative markets) and institutional arrangements to achieve on-farm and off-farm environmental goals (Zimmerman et al. 2019). Citizens are also interested in a spatially targeted approach to producing environmental benefits from agricultural landscapes. In a survey conducted in 2011 and 2012, 64% of survey respondents indicated that they would support such

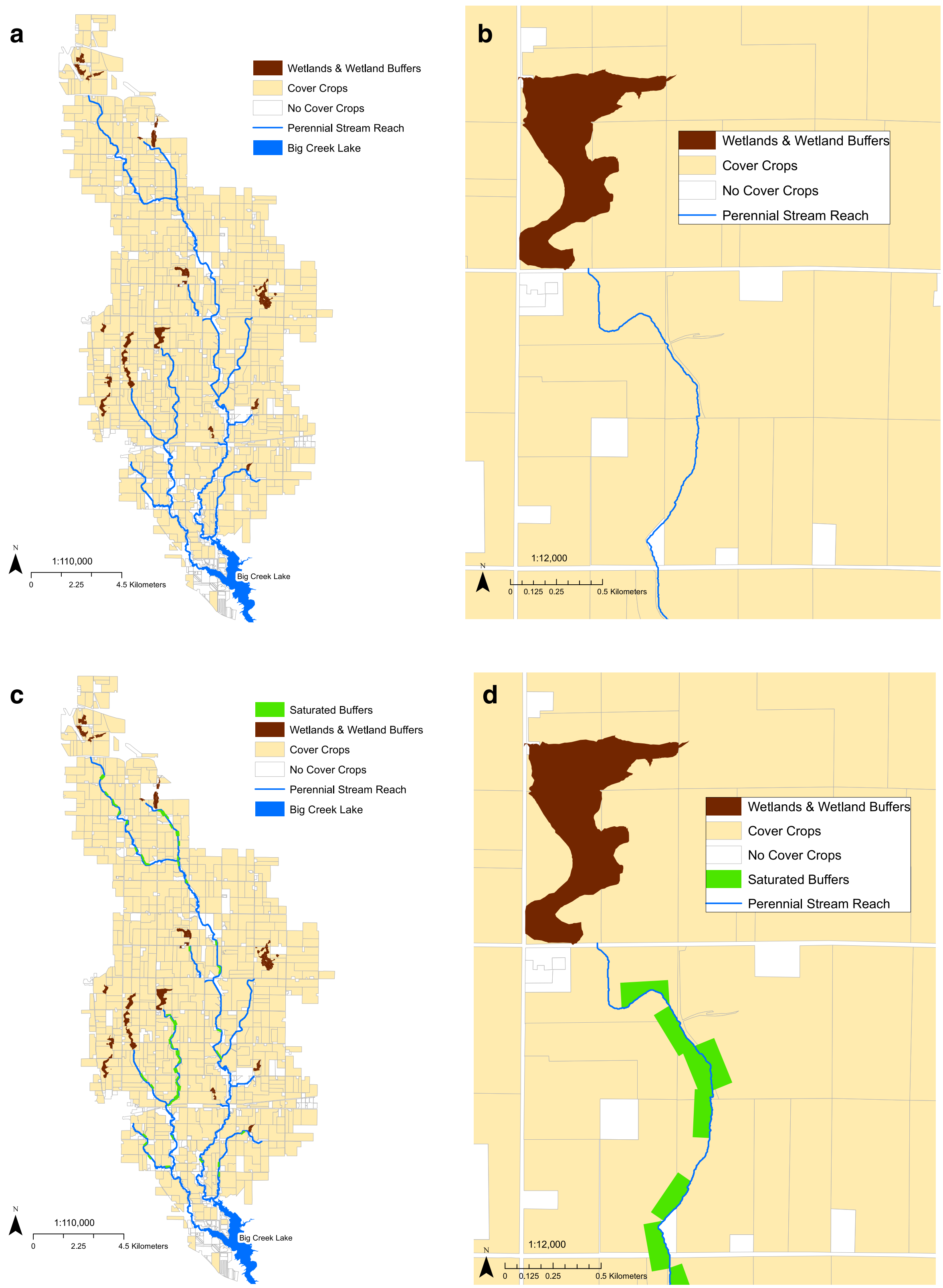

Fig. 5 **a** Alternative land-use Scenario 1 from the Agricultural Conservation Planning Framework (ACPF; Tomer et al. 2013) for Upper Big Creek watersheds, Iowa that includes wetlands and cover crops. **b** Detailed output from the same alternative land-use scenario. **c** Alternative land-use Scenario 2 from the ACPF for Upper Big Creek watersheds, Iowa that includes saturated buffers, wetlands, and cover crops. **d** Detailed output from the same alternative land-use scenario

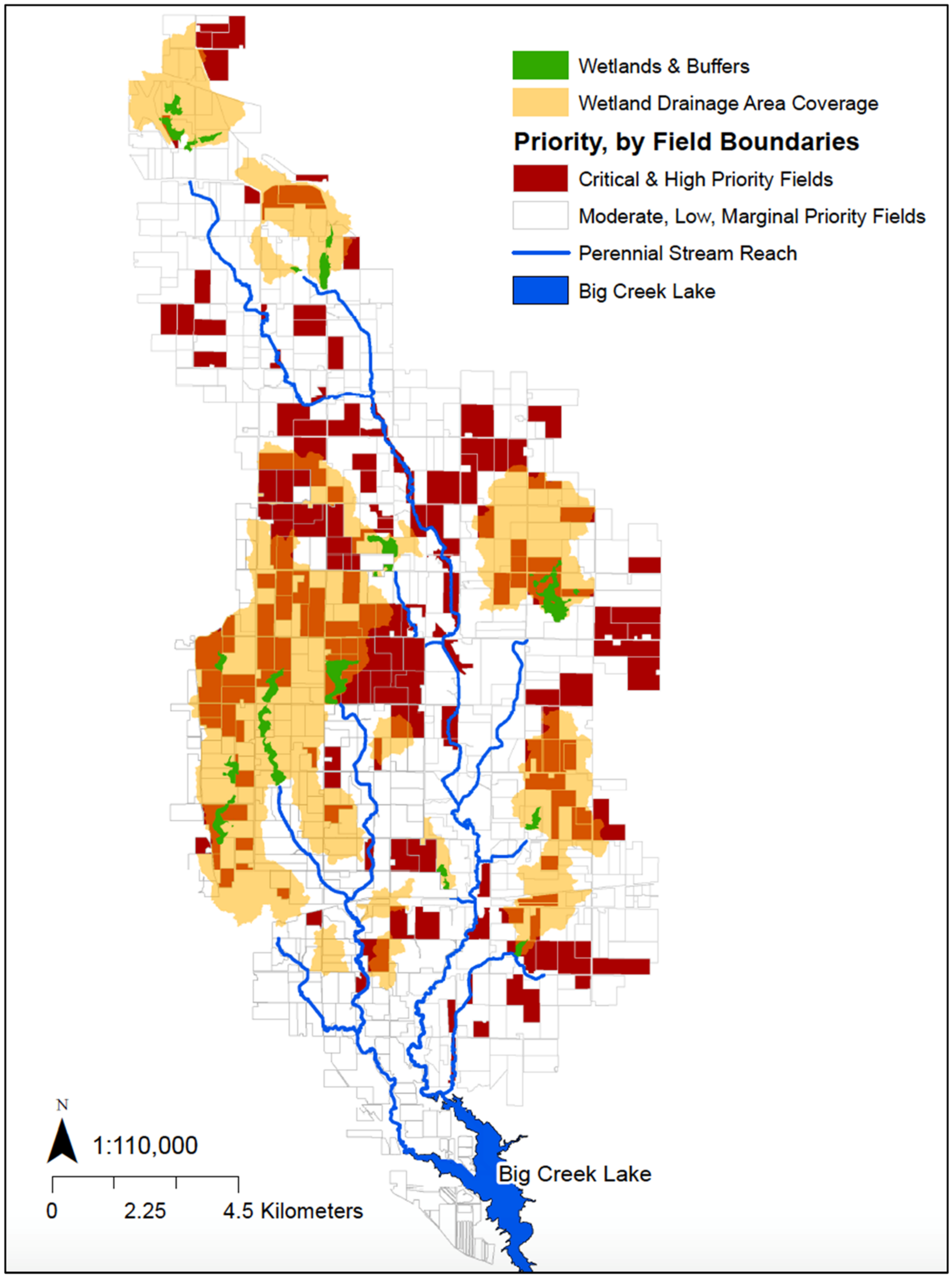

Fig. 6 The wetland output from Agricultural Conservation Planning Framework showing wetland drainage area and wetlands/buffers for the Upper Big Creek watershed, Iowa. Scenario was developed using spatial prioritization of fields identified as critical and high priority (combined together, denoted with deep red color) based on nitrate-N leaching risk and opportunity costs. Selected wetlands were chosen for drainage coverage of critical and high priority areas (denoted with the yellow color, which when placed over fields, creates an orange color in the figure). These wetlands appear in both of the scenarios that meet the Iowa NRS nitrate-N reduction goal of 41%

an approach, and would be willing to pay an average of $33 per year to achieve enhanced environmental benefits (Arbuckle et al. 2015). Developing targeted conservation scenarios, such as those presented here, that spatially evaluate opportunities for, biophysical outcomes of, and the economic costs of change may be an important step forward in initiating conservation strategies to meet Iowa NRS goals and to realizing broader environmental benefits at local and regional scales.

The cost of achieving the Iowa NRS goal for nitrate-N reduction is not going to be inexpensive, and funds to support it are limited, underscoring the need for strategic, effective approaches. Kling (2013) and Hayes et al. (2016) estimate annual costs in Iowa for BMPs to range between $77 million and $1.4 billion. Over the past 20 years, Federal conservation funding from the Conservation Reserve Program (CRP), Conservation Stewardship Program (CSP), Environmental Quality Incentives Program (EQIP), and Wildlife Habitat Incentives Program (WHIP) has spent $4.36 billion in the state of Iowa (EWG 2018). In 2014, Federal conservation funding from those same four Federal programs spent $285 million on water quality and biodiversity programming on agricultural lands in the state of Iowa (Environmental Working Group 2018). At the state

Table 2 Metrics associated with two alternative land-use scenarios created for the two HUC-12 watersheds in Upper Big Creek watershed, Iowa, USA that met the Iowa NRS nitrate-N reduction goal of 41%

Watershed	Nitrate-N reduction (%)	Cost ($)	Cost-effectiveness	Number of fields with BMPs	Area removed from cultivation (ha)
Scenario 1: Cover crops on all corn and soybean acres and headwater wetlands					
East	41%	$2,142,181	$52,248	447 fields	158 ha
West	41%	$1,014,283	$24,739	238 fields	189 ha
Whole watershed	NA	$3,156,464	NA	685 fields	347 ha
Scenario 2: Cover crops on all corn and soybean areas, headwater wetlands, and saturated buffers					
East	47%	$2,161,603	$45,992	448 fields	189 ha
West	49%	$1,033,323	$21,088	242 fields	217 ha
Whole watershed	NA	$3,194,926	NA	690 fields	406 ha

Metrics included in this table are estimated nitrate-N reduction, cost, cost-effectiveness, the number of fields with best management practices (BMPs), and the area removed from cultivation. Cost-effectiveness can be interpreted as cost per 1% of nitrate-N reduction, and was calculated by dividing the cost by nitrate-N reduction percent. All costs are reported in 2017 US dollars

Table 3 Costs associated with each of the best management practices (BMPs) associated with two alternative land-use scenarios created for the two HUC-12 watersheds in Upper Big Creek watershed, Iowa, USA that met the Iowa NRS nitrate-N reduction goal of 41%

Watershed	Cost cover crops	Cost wetlands	Cost saturated buffers
Scenario 1: Cover crops on all corn and soybean acres and headwater wetlands			
East	$1,993,328	$148,853	$0
West	$900,284	$113,999	$0
Whole watershed	$2,893,612	$262,852	$0
Total cost	$3,156,464 per year		
Scenario 2: Cover crops on all corn and soybean areas, headwater wetlands, and saturated buffers			
East	$1,993,328	$148,853	$19,422
West	$900,284	$113,999	$19,040
Whole watershed	$2,893,612	$262,852	$38,462
Total cost	$3,194,926 per year		

All costs presented are annual costs in 2017 US dollars

level, Senate File 512 was signed in 2018 by Governor Kim Reynolds, which commits more than $280 million—~$23.3 million per year—to water quality initiatives over the next 12 years (Des 2018). This money will be invested in conservation infrastructure and programming on agricultural lands—similar to Federal programming initiatives. Based on historical Federal funding and new State funding, future annual conservation funding for Iowa might expected to be ~$300–$325 million—this falls on the low end of the estimated annual investment required for Iowa to meet its nitrate-N reduction goal—despite expressed interest from key stakeholders.

One key challenge associated with improving environmental benefits, such as water quality, is that these benefits are often nonmarket in nature and that those stakeholders producing the benefit are not those experiencing the benefit. Making progress to meet the Iowa NRS nitrate-N reduction or other water quality goals will likely require new approaches to signal that the production of non-market environmental benefits, such as enhanced water quality, are as important as commodity production and that nonmarket benefits are capable of providing immediate and comparable economic return. While current government programming does provide incentives, there are few market-based opportunities. Innovative incentive approaches that provide market-based approaches (e.g., water quality trading—Selman et al. 2009, payment for ecosystem services approaches—Wunder et al. 2008, banking programs—Robertson 2006, etc.) have the potential to incentivize the production of these environmental benefits in the United States and globally (e.g., Greenhalgh, Selman 2012; Grima et al. 2016). To efficiently identify areas of the landscape most appropriate for these efforts—that is areas with the greatest potential to reduce nutrient and sediment loss at the lowest opportunity costs—we posit that harnessing geospatial technology and economic tools like the one demonstrated here present a new way to foster engagement and participation from farmers, farmland owners, and citizens.

Conclusion

Nutrient reduction goals in agricultural landscapes can be met using a spatially coordinated conservation approach that accounts for ecological and economic outcomes, while involving the social and cultural needs of stakeholders. One such spatially coordinated conservation scenario removed less than 5% of cultivated area and reduced nitrate loss by an estimated 49%, exceeding the Iowa NRS goal for enhancing water quality. This framework is particularly well-suited for engaging, collaborating, and communicating with diverse stakeholders across varying spatial extents and resolutions, and is a timely tool for meeting the increasing agricultural and environmental demands placed on the US Cornbelt.

Acknowledgements This research was funded by the Leopold Center for Sustainable Agriculture (E2013-08). The Agricultural Conservation Planning Framework (ACPF) was developed by the United States Department of Agriculture Agricultural Research Service (USDA ARS). The authors thank USDA ARS employees Mark Tomer, David James, and Sarah Porter for their development, assistance, and guidance with the ACPF and its associated data.

Compliance with Ethical Standards

Conflict of Interest The authors declare that they have no conflict of interest.

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