

Abstract

Spatial data on soils, land use, and topography, combined with knowledge of conservation effectiveness, can be used to identify alternatives to reduce nutrient discharge from small (hydrologic unit code [HUC]12) watersheds. Databases comprising soil attributes, agricultural land use, and light detection and ranging–derived elevation models were developed for two glaciated midwestern HUC12 watersheds: Iowa’s Beaver Creek watershed has an older dissected landscape, and Lime Creek in Illinois is young and less dissected. Subsurface drainage is common in both watersheds. We identified locations for conservation practices, including in-field practices (grassed waterways), edge-of-field practices (nutrient-removal wetlands, saturated buffers), and drainage-water management, by applying terrain analyses, geographic criteria, and cross-classifications to field- and watershed-scale geographic data. Cover crops were randomly distributed to fields without geographic prioritization. A set of alternative planning scenarios was developed to represent a variety of extents of implementation among these practices. The scenarios were assessed for nutrient reduction potential using a spreadsheet approach to calculate the average nutrient-removal efficiency required among the practices included in each scenario to achieve a 40% NO3–N reduction. Results were evaluated in the context of the Iowa Nutrient Reduction Strategy, which reviewed nutrient-removal efficiencies of practices and established the 40% NO3–N reduction as Iowa’s target for Gulf of Mexico hypoxia mitigation by agriculture. In both test watersheds, planning scenarios that could potentially achieve the targeted NO3–N reduction but remove <5% of cropland from production were identified. Cover crops and nutrient removal wetlands were common to these scenarios. This approach provides an interim technology to assist local watershed planning and could provide planning scenarios to evaluate using watershed simulation models. A set of ArcGIS tools is being released to enable transfer of this mapping technology.

AGRICULTURAL PRODUCERS in the midwestern United States are being asked to significantly reduce nutrient (N and P) loads to surface waters and thereby mitigate major ecological impacts on aquatic systems in the Gulf of Mexico and the Great Lakes (Michalak et al., 2013; Turner et al., 2012). Although these problems are continental in scope, the challenge in addressing them lies in the management of thousands of small agricultural watersheds and millions of individual farm fields across the Midwest. To be successful, any general strategy must be adaptable to the array of unique combinations of landscape, farm management systems, and the conservation preferences of individuals who own and/or operate farm businesses across this broad region of agricultural production. Although scientific approaches based on watershed modeling and monitoring of conservation effectiveness will be necessary to inform all the management decisions and land use changes that can lead to water quality goals being met and although better modeling approaches to water quality management are being developed (e.g., Arnold et al., 2014; Gebremichael et al., 2013; Kalic et al., 2015), implementing a cost-effective and user-friendly application of these technologies across the breadth of watershed improvement efforts that will be necessary is a daunting task. In the meantime, the need to make measurable progress toward water quality improvement in the near term remains.

Herein, we propose and demonstrate an interim technology that is suited to the hydrologic unit code (HUC)12 watershed scale and that uses detailed data sources that are available across broad areas of the central and northern United States. Based on precision conservation, this approach leverages a recently published framework (Tomer et al., 2013b) that applies consistent criteria to identify appropriate locations for a suite of conservation practices in a given watershed. The application

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Abbreviations: ACPF, Agricultural Conservation Planning Framework; CREP, conservation reserve enhancement program; DEM, digital elevation model; HUC, hydrologic unit code; IRRS, Iowa Nutrient Reduction Strategy; LiDAR, light detection and ranging; RAP, riparian assessment polygon.
of this framework to a watershed provides an inventory of conservation opportunities for water quality improvement that can be considered at the local level. Deliberations at the local level will necessarily take expected costs and producer preferences into account and result in at least one set (scenario) of conservation practices and practice locations being proposed for implementation. The question as to whether each (or any) of the conservation planning scenarios can meet a specified water quality goal is critical to evaluating and comparing proposed planning scenarios.

Although watershed models have been used to evaluate conservation planning scenarios in terms of anticipated nutrient load reductions (Ghebremichael et al., 2013; Kalcec et al., 2015), the expertise required to apply watershed models to agricultural conservation planning activities in thousands of small Midwestern watersheds is not widely available to local conservation planners. Our interim approach, demonstrated here, could assist local agricultural watershed improvement efforts in many watersheds in a relatively short time period and at little cost while requiring modest technical expertise. At the same time, information that could augment modeling efforts to support the management of agricultural watersheds would be developed. The approach, denoted the Agricultural Conservation Planning Framework (ACPF), generates results that comprise a suite of possibilities for placement of conservation practices from which planning scenarios can be developed and compared for their potential to meet water quality goals. The comparison of planning scenarios is based solely on available literature on conservation practice effectiveness and a spreadsheet approach. The approach requires no assumptions about the relative nutrient reduction effectiveness of individual practices included in the scenarios but rather calculates an average effectiveness required among the practices to meet a specified water quality goal while considering how the practices included in the proposed scenario are distributed across the tested watershed. A comparison of this average effectiveness with published research results on practice effectiveness can be used to identify which scenarios can meet (or substantially progress toward) the nutrient reduction goal. This likelihood can be ranked against the amount of cropland removed from production to install conservation treatments or other cost information. The objective of this study was to develop and demonstrate a precision conservation–based approach to identify watershed-scale conservation planning scenarios and compare their relative capacities to cost-effectively meet nutrient reduction goals. For this demonstration, we evaluate scenarios for potential N reduction in two HUC12 watersheds in the upper Mississippi River basin. Practices installed in fields and below field edges are included in the scenarios we tested.

Materials and Methods

Example Watersheds and Input Data

Two HUC12 watersheds were selected to demonstrate our planning approach; one is located in Iowa and one in Illinois, two states identified to be responsible for disproportionate fractions of the N loads contributing to Gulf of Mexico hypoxia (Robertson et al., 2014). Beaver Creek is a tributary to the Upper Cedar River located in north-central Iowa in a landform region known as the Iowan Surface (or Iowa Erosional Surface [Prior, 1991]). Dominated by Illinoian-age (~500,000 yr) glacial till that is typically loess mantled, this area was in a periglacial environment during the most recent glacial advances of Wisconsinan age (~14,000 yr) and comprises rolling terrain that is well dissected by streams. Beaver Creek has a drainage area of 4480 ha and exhibits a naturally formed drainage network with 44.7 km of streams, which gives a drainage density of 1.00 km⁻¹. As is often found in headwater catchments in the Midwest, several of the Beaver Creek’s tributary streams have been straightened and extended by the installation of ditches. Beaver Creek is identified as watershed 3 in our companion paper (Tomer et al., 2015).

Lime Creek is located in north-central Illinois within an area of recent Wisconsinan-age glaciation that was covered by a westward glacial advance from present-day Lake Michigan. This watershed, described by Tomer et al. (2013a), is dominantly tile drained but includes sloping lands near the northern boundary of the watershed where a terminal moraine forms the watershed divide. The moraine grades to a glacio-fluvial plain that dominates the southern half of the watershed. The watershed is 6960 ha in area and is drained via 51.3 km of streams and drainage ditches; drainage ditches dominate the drainage network of Lime Creek watershed, which has a drainage density of 0.74 km⁻¹. Lime Creek is denoted as watershed 6 in our companion paper (Tomer et al., 2015). These two watersheds represent headwater HUC12 agricultural watersheds from two different but common glacial landform regions found in the Midwest (Fig. 1).

Geographic analyses were conducted to identify potential locations for a variety of conservation practices in the two test watersheds. Input data were comprised of soil survey information, land use, and high-resolution digital elevation models (DEMs) derived from LiDAR (light detection and ranging) survey data (Fig. 2). Soil characterization data were extracted from the NRCS Web Soil Survey (Soil Survey Staff, 2013). The dominant soil types (Table 1) are Mollisols in these two watersheds, but Aquic moisture regimes are more common among soils in the Lime Creek than in the Beaver Creek watershed, as are hydric soils (Table 1).

Because decisions regarding implementation of conservation practices concern the use and management of individual farm fields, land use coverages were developed to represent agricultural fields and the types and rotations of agricultural crops and other land cover types in both watersheds. Land use boundaries were produced by editing a publicly available USDA field boundaries dataset (pre-2008), with all ownership and county-level attributes removed. To ensure these field polygons were consistent with recent land use, the 2009 Cropland Data Layer (USDA–NASS, 2013) was examined for all fields larger than 16 ha. For those fields with multiple cover types, 2009 National Agricultural Imagery Program aerial photography was used as a basis to manually edit field boundaries. A field was considered to have multiple cover types and was edited if the dominant cover occupied <75% of the field, as indicated by the 2009 Cropland Data Layer. Updated field boundaries were then overlaid with data from USDA–NASS (2013) Cropland Data Layer for 2007 through 2012, and each field was classified to represent crop rotations and land cover using the 6-yr (2007–2012) sequence of land cover. Six-year land-cover strings (e.g., corn–corn–soybean–corn–soybean–corn) generated for each
Topographic data were obtained from LiDAR survey data in both watersheds, and 3-m grid DEMs were developed. The LiDAR data for Lime Creek were obtained as described in Tomer et al. (2013a). The 3-m grid DEM developed for Beaver Creek used LiDAR data collected through a statewide Iowa survey (University of Northern Iowa, 2013). Both DEMs were processed to enforce the correct routing of hydrologic flows toward and along streams to the watershed outlet. This process involves removal of artificial impoundments that occur where channels and pathways of concentrated flow are obscured by infrastructure (bridges and roads) and occasionally by riparian vegetation. The DEM for Lime Creek was processed for enforcement of overland flows as described by Tomer et al. (2013a). The DEM for Beaver Creek was similarly processed but was conducted under software control (B. Gelder, written communication, April 2014) as part of an effort to bulk-process the hydrologic enforcement of Iowa’s LiDAR data. The resulting DEM coverages were manually checked and corrected using aerial photographs and shaded-relief imagery to confirm locations of stream channels and pathways of concentrated flow. Once this hydrologic processing was completed, topographic slopes and up-gradient runoff contributing areas were mapped using tools included in ArcGIS v. 10.2 and TauDEM v. 5.1.2 (ESRI, 2014; Tarboton, 2014). Finally, a stream network coverage was obtained for both watersheds. A stream threshold initiation algorithm (Peucker and Douglass, 1975) provided an estimated distribution of stream channels, which was edited using aerial photography to represent the actual location of streams in the watersheds as accurately as possible. This facilitates subsequent analyses to classify and map riparian settings in the watershed (Tomer et al., 2015).

A D8 flow routing algorithm was applied to each hydrologically corrected DEM to generate a flow-direction coverage for each watershed. This process directs overland flow from each grid cell to a single neighboring cell, which is determined by the steepest downward slope gradient. Flow direction rasters were then used to generate flow-accumulation rasters for each watershed, whereby the value of each grid cell is equal to the count of upstream cells flowing into that cell. Upstream cell counts are converted to area measurements to determine the upstream drainage to each point in the DEM. A slope coverage in percent rise (rise/run) was also generated for each watershed using the slope tool available in ArcGIS v. 10.2 (ESRI 2014). Conceptually, the slope tool fits a plane to the z-values of a 3 × 3 cell neighborhood around the processing or center cell.

**Conservation Practice Selection and Siting Criteria**

Based on the above information, there were four input data sources (Fig. 2) that were developed for each watershed, including (i) soil survey information; (ii) agricultural field boundaries with land use and crop rotations; (iii) a processed 3-m grid DEM with derived data including local slope, up-gradient areas of runoff-contribution; and (iv) a manually edited stream network (vector coverage). These data were used to identify locations suited for
Fig. 2. Mapped representation of input data for the Beaver Creek and Lime Creek watersheds. CB, corn/soybean.

Table 1. Dominant soil series in the two study watersheds, and extent of hydric soils.

<table>
<thead>
<tr>
<th>Soil information</th>
<th>Beaver Creek</th>
<th>Lime Creek</th>
</tr>
</thead>
<tbody>
<tr>
<td>Three most common soil types</td>
<td>Typic Hapludoll</td>
<td>Mollic Hapludalfs</td>
</tr>
<tr>
<td></td>
<td>Cumulic Hapludolls</td>
<td>Cumulic Endoaquolls</td>
</tr>
<tr>
<td></td>
<td>Typic Argiudolls</td>
<td>Typic Endoaquolls</td>
</tr>
<tr>
<td>Hydric soil map units percentage of watershed (area)</td>
<td>9.7% (436 ha)</td>
<td>19.6% (1362 ha)</td>
</tr>
</tbody>
</table>

Table 2. Classes of crop rotations identified from a sequence of 6 yr (2007–2012) of majority crop cover. All agricultural fields in the two study watersheds that produced annual crops according to crop data layer information (USDA–NASS, 2013) fell into one of these classes.

<table>
<thead>
<tr>
<th>Crop rotation</th>
<th>Description</th>
<th>Rotation weight†</th>
</tr>
</thead>
<tbody>
<tr>
<td>Corn/soybean</td>
<td>The dominant cover during the 6-yr rotation alternated between corn and soybean; consecutive years of corn not observed.</td>
<td>1.00</td>
</tr>
<tr>
<td>Continuous corn</td>
<td>The dominant cover during the 6-yr rotation showed corn was planted all 6 yr; no other majority crop in any of the 6 yr.</td>
<td>1.10</td>
</tr>
<tr>
<td>Corn/soybean with continuous corn</td>
<td>The dominant cover during the 6-yr rotation included only corn and soybean, but corn was the dominant cover in at least two consecutive years. The number of years of consecutive corn could be up to five.</td>
<td>1.05</td>
</tr>
<tr>
<td>Conservation rotation</td>
<td>The dominant cover during the 6-yr rotation was corn, soybeans, and perennial cover classes. The intent is to identify fields where a forage crop was included in the rotation.</td>
<td>0.90</td>
</tr>
<tr>
<td>Extended rotation</td>
<td>The dominant cover during the 6-yr rotation included corn, soybean, and other annual crops. The intent is to identify fields that include annual crops other than corn and soybeans in the 6-yr rotation.</td>
<td>0.90</td>
</tr>
<tr>
<td>Mixed agriculture</td>
<td>The dominant cover during the 6-yr rotation showed a mix including corn, soybeans, other annual crops, and/or forages. Other crop rotations not described above, or fields without any dominant crop in at least some years (e.g., contour strips).</td>
<td>1.00</td>
</tr>
</tbody>
</table>

† The rotation weight provides for relative differences in nutrient losses among rotations to be considered, and changes in crop rotation to be included, in nutrient reduction scenarios. Assigned values are estimates and for demonstration only.
a variety of conservation practices using a planning framework described by Tomer et al. (2013a). The processes (or steps) within the framework comprise automated routines developed to identify appropriate locations for conservation practices placed in fields, below fields, and in riparian zones. However, before conducting these analyses, the planning framework begins with an emphasis on practices that promote healthy functioning of soils to minimize soil erosion, enhance infiltration and water retention, and minimize loss of plant nutrients (N and P). These practices, such as zero or zonal tillage, cover crops, and nutrient management, carry the potential benefit of increased farm profitability and/or soil productivity, which is likely to be of interest to all farm producers and are therefore emphasized in the planning framework without geographic prioritization.

To represent the importance of these agronomic conservation practices, in both watersheds the planning scenarios evaluated herein included winter cover crops implemented at four extents: on all, two thirds, one third, and none of the fields in the watershed, with the two levels of partial implementation (one third and two thirds) assigned to fields selected at random. This practice was selected for soil improvement because it is known to decrease NO$_3$–N loss to tile drainage water (Malone et al., 2014). Studies reviewed under the Iowa Nutrient Reduction Strategy (INRS) found an average NO$_3$–N reduction of 31% when winter rye (Secale cereal L.) was used as a cover crop (Nitrogen Science Team, 2013).

Tile Drainage Extent and Identifying Drainage
Management Opportunities

A variety of conservation practices can provide water quality benefits if they are installed in places where they can intercept and treat water that moves along specific flow pathways. In Midwest watersheds, a dominant hydrologic pathway that carries substantial NO$_3$–N loads is subsurface (tile) drainage (Royer et al., 2006), and there are particular conservation practices that can mitigate NO$_3$–N loads from tile drainage. Therefore, those fields likely to be tile drained are identified as an early step in the ACPF framework (Tomer et al., 2013b). The actual extent of tile drainage is usually not publicly available and needs to be estimated in most Midwest watersheds. In the Beaver and Lime Creek watersheds, we designated a field as tile drained if it was in row-crop production, had <5% slope on >90% of the field area, and had either at least 10% of the field’s soils mapped as hydric (formed under saturated conditions; Vepraskas et al. [2002]) or at least 40% of the field’s soils mapped in a dual hydrologic group (typically B/D). The dual hydrologic group designation indicates that improved drainage is necessary for production of row crops (Natural Resources Conservation Service, 2007).

We next identified individual fields in the two watersheds where opportunities to manage artificial drainage systems were apparent. In flat, tile-drained fields where slopes are <1%, drainage water management (a.k.a. controlled drainage) systems can be installed. This practice comprises the installation of gate structures that are used to raise the water table during part of the year. Volumes of tile drainage and associated NO$_3$–N loads can be substantially reduced without affecting crop yield (Fang et al., 2012; Helmers et al., 2012a). The gates are lowered to provide unrestricted drainage for planting, harvest, and other field operations and are used to raise the water table to a depth of 0.6 to 0.8 m during the growing season. During winter, the water table can be raised to within 0.3 m of the soil surface, which minimizes winter drainage volumes. Controlled drainage is most readily implemented in flat fields, which are common in glacial-fluvial plains in the Midwest and which are most extensive in Ohio, Indiana, and Illinois. Fields with >1% slopes will usually need multiple gate structures and custom-engineered patterns of tile drains to install drainage control systems. A single control gate will typically be effective in controlling the water table within the root zone of a drained area with at most a 0.5-m variation in ground surface elevation. In the Beaver Creek and Lime Creek watersheds, we identified fields where controlled drainage was most feasible by determining the largest part of each tile-drained field that was within any 1-m contour interval. If the largest area within a 1-m contour in a field comprised >50% of the field (meaning two gates could control the water table depth in at least half the field), then that field was designated as a potential location to install controlled drainage. Studies on controlled drainage that were reviewed under the INRS (Nitrogen Science Team, 2013) had an average nitrate load reduction of 33%, which was achieved by a decrease in drainage volume.

Assessing Runoff Risk at Field Scale

A number of conservation practices can be placed within individual fields where steeper slopes occur and where there is a need to mitigate risks of runoff and erosion. Grassed waterways, contour filter strips, and terraces are examples of these practices. The ACPF includes a utility to rank individual farm fields in a watershed according to proximity to streams and steepness of slopes to identify fields where runoff is most prone to directly enter streams. We anticipate that installing runoff control practices in the prioritized fields should be most effective for water quality improvement from a watershed management perspective. Fields are first ranked according to steepness, based on the 75th percentile slope value to represent the steepest quarter of each field; that is, 25% of the field is comprised of slopes greater than this value. The concept to rank fields based on the steepest part of each field is justified because erosion losses do not occur evenly from all parts of a field but come from limiting areas “of significant extent,” meaning “at least 20% of the field” (Lewandowski et al., 2006). Accordingly, we chose to use the 75th percentile slope values, which were classified to place each field into one of three categories: ≥5% slope (high), 2.5 to 5% slope (medium), or ≤2.5% (low). These demarcations are arbitrary, but they provided a reasonable basis to compare slope distributions of the fields in these two watersheds that are dominantly tile drained.

Proximity to the stream channel was next ranked using an equation to estimate sediment delivery ratio (SDR) from the edge of the field to the nearest channel. We selected the equation used in Minnesota’s Phosphorus Index [SDR = (X/3.28) $^{0.207}$] in which $X$ is the distance (m) from the field edge to the stream channel (Lewandowski et al., 2006). To calculate the SDR for all fields in Beaver and Lime Creek watersheds, distance to stream rasters were generated to provide horizontal distances to the stream channel for each 3-m grid cell in the DEMs. The distance assigned to each field was the topographically lowest grid cell along the field boundary and provided the flow-path length to the channel (rather than the shortest horizontal distance). The
SDR decreases significantly in a short distance; therefore, the 3-m resolution of the DEM had to be considered. Sediment delivery ratio values decline from a value of 1.00 for a field that borders a stream to a value of 0.62 for a stream just one grid cell (3 m) away. To account for this effect of DEM resolution, all fields within one grid cell of a stream channel were given a SDR value of 1.00. In practice, 3 m was subtracted from the minimum distance-to-stream value for every field; a SDR of 1.00 was assigned where this subtraction gave a negative result, and the above SDR equation was applied where the subtraction gave a positive result. This shift provided a consistent result for fields close to the stream where small errors in digitizing a field edge could otherwise affect the SDR ranking. For Lime Creek and Beaver Creek, the SDR values calculated for each field were classified into three categories as follows: >0.4 (high), 0.2 to 0.4 (medium), and <0.2 (low). The SDR calculation decreases rapidly within the first few meters, and therefore an SDR of >0.4 was considered high.

Cross-classifying fields according to the above SDR and slope steepness groupings provides a “runoff-risk assessment” (3 x 3) matrix for the agricultural fields in both watersheds; the risk assessment classifies fields as (i) “critical” fields, where steeper slopes occur near the stream; (ii) “very-high” fields, where steep slopes are relatively near the stream or where moderately steep fields are near the stream; or (iii) “high” fields that are steep or near the stream and other “smaller-risk” fields that are neither steep nor near the stream relative to other fields in the watershed. A “smaller-risk” classification does not identify fields where runoff control practices are not needed; it identifies fields with the least risk of direct runoff delivery to the stream.

Practice Selection to Address Runoff Risk

Our planning scenarios in Lime Creek and Beaver Creek suggested locations for grassed waterways in those fields identified by the runoff risk prioritization. Several approaches have been suggested to determine where grassed waterways can be placed within fields to best mitigate the possibility of gully formation (e.g., Pike et al., 2009). Essentially, all these approaches identify pathways of concentrated (or collective) flow. Terrain analysis of high-resolution DEMs allows flow pathways to be identified by a simple threshold approach, which is applied to a flow accumulation raster. For both Lime Creek and Beaver Creek, grid cells with a contributing area >1 ha were identified as suitable for grassed waterway installation. Waterways were assigned a constant width of 10 m to calculate the total land area taken out of production under three scenarios using results from the runoff risk assessment.

Grassed waterways are known and usually installed to reduce runoff and sediment transport to field edges (Fiener and Auerwald, 2003); however, trapping and treatment of N in grassed waterways has received little attention. Clearly, particulate N in runoff can be trapped in grassed waterways. In addition, Schilling et al. (2007) found evidence of subsurface NO$_3$–N loss via denitrification along an ephemeral waterway in Iowa, and narrow zones of perennial grass cover have been associated with decreased transport of total N in surface runoff and/or subsurface nitrate in several studies (Udawatta et al., 2002; Zhou et al., 2010; Balestrini et al., 2011; Zhou et al., 2014). If combined with installation of shallow drainage tiles (Luo et al., 2010) along ephemeral waterways, as suggested by Schilling et al. (2007), grassed waterways could be designed to provide adequate surface drainage while providing nutrient sinks for runoff and shallow subsurface flows (see Schilling et al. [2013] for further discussion). We suggest that, with careful design, grassed waterways could reduce some fraction of total N and nitrate loads from fields, and we include it in our scenarios as a N-reducing practice. Shallow drainage tiles, installed at <0.8-m depth, are considered a N-reducing practice in the INRS (Nitrogen Science Team, 2013) when compared to deeper drainage (tiles installed at >1.0 m depth). Although grassed waterways are not listed as a N-reducing practice in the INRS, grass filters can reduce total N concentrations in surface runoff by trapping particulate N (Zhou et al., 2014). Herein, we consider shallow-drained grassed waterways an experimental practice for N reduction. Their inclusion in planning scenarios also helps address contaminants other than NO$_3$–N, such as sediment and P; the benefits of including multiple nutrients in water quality planning efforts have been discussed elsewhere (Rabotyagov et al., 2010; Tomer and Locke, 2011).

Edge-of-Field Practice: Nutrient Removal Wetlands

Below field edges, it is possible to map locations that are suitable for different types of practices that intercept and treat tile discharge and/or runoff. A number of these practices comprise impoundments that slow and detain water flows to provide a variety of water quality and ecosystem benefits. Nutrient removal wetlands, sediment control basins, and farm ponds are all practices that use impoundments. We identified locations that could be suited for nutrient removal wetlands in these watersheds because of the wide extent of tile drainage in Lime Creek and Beaver Creek and the well-recognized effectiveness of wetlands for reducing NO$_3$–N loads from tile drainage (Kovacic et al., 2000; Tomer et al., 2013b). Nutrient removal wetlands have the potential to remove >50% of incoming NO$_3$–N loads (O’Geen et al., 2010) if the area of the wetland is sufficient to provide an adequate hydraulic residence time (about 1 d).

The 3-m DEMs were used to determine where nutrient-removal wetlands could be placed in the Lime Creek and Beaver Creek watersheds without impeding drainage from significant areas of cropland. Points were generated at 100-m intervals along streams and flow pathways where upslope contributing areas exceeded 20 ha. Along each stream reach/flow pathway, these points were tested for wetland suitability beginning with the lowest (most downstream) location and then proceeding upstream. At each location, an impoundment was simulated to mimic the installation of a constructed wetland at the site. Criteria used to site wetlands under Iowa’s Conservation Reserve Enhancement Program (CREP) were adapted (Tomer et al., 2013b) to test for wetland suitability of each site; the most significant modification was to reduce the minimum contributing area from 300 to 20 ha to provide the greatest possible number of wetland sites. This is consistent with the goal of applying this framework, which is not to recommend where practices should be placed, but to provide an inventory of potential sites for a full range of practices that landowners may be willing to consider for improving water quality.

The procedure followed to test candidate sites was much as described by Tomer et al. (2013b), but the capability to test...
the sites under software control allowed a greater number of sites to be tested. At each test point, the (focal) minimum and range in elevation within a 20-m radius were determined. The focal minimum provided the estimated channel elevation, and the bank height was estimated by the focal range in elevation within the 20-m buffer. Sites with bank heights >4 m were omitted from further consideration to avoid locations with incised channels. The resolution of the LiDAR-driven DEMs allowed a simulated impoundment depth to be specified, chosen here as the height of the bank plus a constant of 0.9 m, which, when added to the channel elevation, provided the wetland pool elevation. The 0.9-m pool depth would facilitate establishment of emergent wetland vegetation, providing the C source needed for denitrification to occur. The area adjoining and up-gradient of the test point with a surface elevation less than the pool elevation provided an estimated wetland area for that site. A buffer around the wetland, where the surface elevation is within 1.5 m of the wetland pool elevation, was determined to identify where the wetland could impede soil drainage in the upslope area. The areas of wetlands, buffers, and contributing areas were then tabulated for each point and used to test for wetland suitability. To pass this test, a wetland pool area had to be between 0.5 and 2.0% of the contributing area to provide an adequate residence time, and the wetland buffer area had to be less than four times the size of the wetland pool. In addition, wetlands and their buffers could not intersect roads, bridges, or farmsteads to ensure wetland construction would not affect infrastructure in the watersheds; this final check was done manually using rectified aerial photographs.

Conservation planning scenarios in the Beaver Creek and Lime Creek watersheds included the lowest (largest contributing area) wetland along each stream reach. This was modified in Beaver Creek, where two Iowa CREP wetlands were already installed. These two existing wetlands were included in the Beaver Creek scenarios in lieu of potential wetland sites located lower along the same two stream reaches. In the development of planning scenarios, including existing practices provides a way to recognize the contribution that past conservation practice installations will make toward reducing future NO₃-N losses. Although it is not straightforward to determine exactly which past installations should be credited toward achieving future goals, this approach to assess planning scenarios can consider existing practices. The two Beaver Creek CREP wetlands were digitized, and areas of the wetlands, buffers, and contributing areas were tabulated for inclusion in the assessment of planning scenarios (described below). Fields were assumed to have NO₃-N reduction provided by a wetland if more than half the field was in the contributing watershed above that wetland. This not only simplified the task of determining which fields would be served by a downstream practice; it also represents the reality that each individual field will usually have a single drainage system discharging to one ditch or drainage main.

**Edge-of-Field Practice: Saturated Buffers**

The final practice included in our conservation planning scenario development in these two watersheds relied on a riparian assessment scheme, which maps and classifies the riparian corridor according to distinct and naturally occurring opportunities to intercept runoff, influence shallow groundwater, and stabilize stream banks (Tomer et al., 2015). The approach maps the likely distributions of surface runoff contributions and shallow water tables in a watershed, discretizes and tabulates results along both banks of the channels, and applies a cross classification that conveys recommendations for buffer vegetation and width. Riparian analysis polygons (RAPs) (250 × 180 m) are created along the stream network; the RAPs are centered along the stream then divided by it to evaluate each side of the stream independently. The RAPs provide the spatial frame to collate and classify derived terrain data on runoff-contributing and shallow water table areas. This process is described in detail in a companion paper (Tomer et al., 2015).

This riparian scheme was adapted to identify places where saturated buffers (Jaynes and Isenhart, 2014) could be installed to provide subsurface discharge of tile drainage water. A RAP was a potential saturated buffer site if it (i) had a low or medium runoff contribution (as defined by Tomer et al. [2015]) and (ii) had an average shallow water table width between 25 and 50 m. These criteria were applied to avoid areas where large amounts of surface runoff could challenge installation and maintenance of a saturated buffer and to highlight places with a uniform slope configuration across the riparian zone. A uniform slope will lead to low bank heights adjacent to the stream, minimizing the risk of bank slumping, and provide a gradient toward higher elevations in the adjacent field, minimizing the chance of crop inundation due to saturation of the buffer. A third criterion ensured that soils high in organic C content would provide a substrate for denitrification to occur. A soil organic C content averaging >2% from 0 to 150 cm depth had to extend on average at least 25 m from the stream channel, based on soil survey information. Finally, candidate sites located along roads or under forest cover were eliminated. The drainage area to each RAP was also delineated, and those RAPs intercepting drainage from fields already proposed for treatment by a wetland were removed. The result was that fields whose drainage was identified as being treated with saturated buffers were mutually exclusive from those fields treated with a wetland practice. This would not always be done in an actual planning setting but simplifies the scenarios being evaluated here. The saturated buffer is also an experimental practice that has shown promising early results (Jaynes and Isenhart [2014] reported a 55% load reduction during a single 2-yr study), and this practice is being assessed for inclusion in the INRS.

**Development and Comparison of Multipractice Planning Scenarios**

For each watershed, a spreadsheet was constructed to evaluate scenarios comprised of different combinations of practices selected from those described and placed as detailed above. Every cropped field was represented in the spreadsheet as a record (row), and columns were used to represent the relative size of each field and the relative impact of crop rotation in each field on nutrient losses. There is a column for each conservation practice, in which the cells are populated to indicate the presence or absence of that practice within (or below the edge of) each field. A rotation weight was assigned to each field according to the land cover description determined by the sequence of crops identified from 6 yr of cropland data layer data (Table 2). The rotation weight
is based on general evidence in the literature that crop rotation does affect NO$_3^-$-N losses because N fertilizer application rates are typically increased in a continuous corn rotation (Helmers et al., 2012b) and are decreased in rotations including soybean and small grains. We assigned a value of 1.0 to fields with a strict corn–soybean rotation and assigned values for other rotations that varied between 0.9 and 1.1 (Table 2). This range is modest, but the purpose was not to defend any particular set of values for given crop rotation classes. Rather, this provides a way for changes in crop rotation to contribute to NO$_3^-$-N reduction as part of a planning scenario on a field-specific basis. To represent how conservation practices could affect NO$_3^-$-N reduction in the spreadsheet, our approach was based on the assumption that stacked conservation practices will have a multiplicative effect on NO$_3^-$-N loss, as discussed by Lazarus et al. (2014). This would mean, for example, that if a hydrologic pathway carries a given nutrient load through two practices placed in succession along that flow path and if each practice has a 50% removal efficiency, then the fraction of the load expected below the two paired practices would be 25% of the original incoming load. Therefore, the spreadsheet for each watershed was constructed so that the absence of a conservation practice within or below any given field was represented as a value of 1.0, whereas the presence of that conservation practice within (or below) any other given field was represented as a value of $(1 - E)$, with $E$ representing the NO$_3^-$-N removal efficiency of the practice. For each conservation scenario tested, the entire column of values was 1 for each practice not included anywhere in the scenario, and a mix of 1 and $(1 - E)$ values for practices that were included in the scenario, distributed among fields according to the spatial distribution of that practice within and/or below fields across the watershed. The question of the value of $E$ (NO$_3^-$-N removal efficiency) is pivotal in all conservation planning efforts aimed at N reduction in watersheds. Here, we used a single value of $E$ into every cell in the spreadsheet in which it appeared; that is, all values of $E$ came from a single cell in a separate area of the spreadsheet. The calculation process involved summing, among all fields, the products of relative field area times the crop rotation weight for each field; this sum provided a denominator representing the base (or current) nutrient load condition in the watershed. These same products for each field (i.e., field size times rotation weight) were then multiplied across all the columns providing a series of 1 or $(1 - E)$ multipliers, and these products were also summed among fields providing a numerator, which when divided by the denominator represented the fraction of nutrient load realized under the scenario. A “solver” utility in the spreadsheet software was used to identify the value of $E$ that resulted in a calculated numerator that achieves a targeted nutrient loss. We used a target of 40%, which is close to the target suggested by the INRS (Nitrogen Science Team, 2013). The value of $E$ represents the average NO$_3^-$-N reduction efficiency required among all the practices included in the scenario if the 40% NO$_3^-$-N reduction goal were to be met under the scenario. The value of $E$ returned by the solver for each scenario was then compared with the NO$_3^-$-N removal efficiencies reported in the literature for practices included in the scenario. Efficiencies of NO$_3^-$-N removal were recently reviewed for a number of conservation practices as part of Iowa’s nutrient reduction strategy (Nitrogen Science Team, 2013) as noted above for cover crops, drainage water management, and nutrient removal wetlands. However, this approach enabled us to include two practices that are considered experimental because their potential capacity for N removal has not yet been well documented by research. If all practices included are well documented, the evaluation of $E$ would be a simple matter of comparison to a weighted mean (i.e., weighted according to the relative extent of each practice in the scenario). Finally, for each scenario, the value of $E$ was plotted against the area of cropland taken out of production in the watershed as a representation of the long-term cost of the scenario. This information could be replaced by practice cost information, if available, for all practices.

**Results and Discussion**

**In-Field Practices and Field-Scale Assessments**

**Cover Crops**

In both watersheds, two thirds and one third of the cropped fields were selected at random to generate scenarios that included partial deployment of cover crops (Fig. 3). This was done without geographic prioritization to reflect the general importance of soil improvement practices for watershed health and agricultural productivity. Research on winter rye cover crops in this region has shown that this practice, on average, reduced NO$_3^-$-N losses by 31% on a concentration basis (Nitrogen Science Team, 2013) estimated extent of tile drainage and drainage management opportunities.

The extent of tile-drained fields in these two watersheds was estimated based on the presence of low slopes, soil map units of all hydric soils, and/or soil map units with a dual hydrologic group. Results suggest that 137 of 147 cropped fields in the Beaver Creek watershed and 201 of 243 cropped fields in the Lime Creek watershed are tile drained.

The opportunity to manage tile drainage water using water table control gates was evaluated by identifying those fields where at least half the field area was within a 1-m contour interval. Only four fields in the Beaver Creek watershed met this criterion, and this practice was not further considered in the conservation planning scenarios for the Beaver Creek watershed. In the Lime Creek watershed, however, we identified 51 fields that met this criterion (Fig. 4). Drainage water management was reported (Nitrogen Science Team, 2013) to reduce NO$_3^-$-N loads by reducing tile drainage volumes (rather than NO$_3^-$-N concentration), resulting in an average load reduction of 33%.

**Runoff Risk Assessment and Placement of Grassed Waterways**

Application of the runoff risk assessment to cropped fields in the Beaver Creek and Lime Creek watersheds indicated differences between the two watersheds that reflect the presence of steeper slopes and greater drainage density in Beaver Creek (Table 3). In both watersheds, a relatively small number of fields had slopes >5% on at least a quarter of the field and were close enough to a stream to have an SDR >0.4. This "critical" risk setting occurred in 26 out of 147 fields in the Beaver Creek watershed but in only 9 of 243 fields in the Lime Creek watershed (Table 3). The relative risk of runoff delivery was addressed in our conservation planning scenarios through inclusion of grassed
The distributions of grassed waterways included under scenarios in each watershed are illustrated in Fig. 5.

Multiple locations were found to be suitable for installation of nutrient removal wetlands in both watersheds. We focus here on those potential wetland locations that were furthest downstream along tributary channels. In the Beaver Creek watershed, 11 sites were identified (Fig. 6), including two prior-existing Iowa CREP wetlands, which would intercept flows from 1929 ha (43% of the watershed area) if all 11 were constructed. In the Lime Creek watershed, seven potential wetland locations found in lower tributary reaches could intercept flows from 2026 ha (29% of the watershed area; Fig. 6). Tomer et al. (2013a) provide detailed discussion on locating potential wetland sites in the Lime Creek watershed and anticipated variations in nutrient removal among wetland sites. On average, nutrient removal wetlands have been found to decrease nitrate loads by 52% (Nitrogen Science Team, 2013).

Saturated Buffers

We found 89 RAPs in the Beaver Creek watershed qualified as potential saturated buffer sites based on criteria intended to identify uniform riparian slopes and soils high in C deep in the profile. In contrast, no sites were identified for Lime Creek because soil survey information indicated riparian soils had low (<2%) C contents to 1.5 m depth. This criterion may be too restrictive; additional research is required to determine how to qualify sites for possible installation of saturated buffers. After removing those that did not receive drainage from cropped, tile-drained fields and those adjacent to roads, the final number of potential saturated buffers in Beaver Creek was 40 RAP sites, which received drainage from 27 cropped fields (Fig. 7).
Saturated buffers are an experimental N reduction practice that, based on early experimental data, could remove nearly all nitrate that could be diverted from tile drains to riparian soils (Jaynes and Isenhart, 2014). Research conducted on nonsaturated riparian buffers in the region averages a removal efficiency of 91% (Nitrogen Science Team, 2013), and it makes sense that using tile drainage water to raise the water table in a riparian buffer would enhance opportunities for N removal in many settings. The actual fraction of tile water diverted to a saturated buffer would depend on the cropland area being drained for diversion to riparian soils, the length of riparian zone receiving diverted drainage, and soil-site-vegetation characteristics.

### Comparison of Conservation Planning Scenarios

Conservation planning scenarios in the two watersheds were comprised of combinations of practices, with locations and spatial extents (i.e., areas treated) specified for each practice (Table 4; Fig. 3–7). Based on a spreadsheet approach, for many scenarios, we calculated the average nitrate removal efficiency ($E$) that would be required among all the practices in the scenario in order for that scenario to meet a 40% reduction goal (Fig. 8). The simplest scenario comprised only one practice (cover crops) on all fields and returned the obvious result of $E = 0.4$. In some scenarios, the spreadsheet returned a value of $E > 1.0$, indicating that the practices in the scenario treated <40% of the watershed; we did not plot scenarios for which $E > 0.8$ (data not shown). We calculated $E$ for many combinations of practices (each combination comprising a scenario) and plotted each result against the land area that would be converted from crop production to install the conservation practices included under that scenario. After plotting $E$ versus “cropland removed” for many scenarios, it was possible to identify a group of scenarios that spread the required nutrient reduction across the watershed and among multiple practices while removing relatively little

![Fig. 5. Distribution of suggested grassed waterway locations included in conservation planning scenarios in the Beaver Creek and Lime Creek watersheds. Fields are classified according to runoff risk assessment based on slope steepness and sediment delivery ratio.](image-url)
The practices included in these scenarios that were reviewed under the INRS (including wetlands, cover crops, drainage water management, and buffers) (Nitrogen Science Team, 2013) were reported to reduce NO$_3$-N concentrations (or loads) by >30%. Therefore, scenarios with $E < 0.3$ should have the greatest chance of meeting a 40% reduction goal. In both the Beaver Creek and Lime Creek watersheds, scenarios that included >66% implementation of cover crops and wetlands were associated with values of $E < 0.3$ (Fig. 8). This result may have been constrained by the practices and implementation extents (Table 4) considered for our demonstration.

The example we present (Fig. 8) was generalized to allow alternative or little-assessed practices to be included in planning scenarios. For scenarios that only include practices that have been well researched, an alternative approach is to populate the practice-specific columns in the spreadsheet with average reductions determined through a literature review (e.g., Nitrogen Science Team, 2013). We considered this alternative by mapping one scenario in each watershed (Fig. 9) that comprised practices reviewed by Iowa’s Nitrogen Science Team (2013), including cover crops (average NO$_3$-N removal efficiency, 0.31), drainage water management (average NO$_3$-N removal efficiency, 0.33), nutrient removal wetlands (average NO$_3$-N removal efficiency, 0.52), plus saturated buffers, which we assigned an average NO$_3$-N removal efficiency of 0.43, which is the expected NO$_3$-N removal efficiency for woodchip bioreactors. The scenarios comprising these practices would achieve a 34 to 39% NO$_3$-N reduction if 66% deployment of cover crops were to be combined with other practices as shown in Fig. 9 and if the land from production (Fig. 8).
practice-specific average nutrient reduction efficiencies listed above were to be exactly achieved. Based on our more generalized spreadsheet approach, the practices shown in Beaver Creek (Fig. 9) would need an average $E$ of 0.39 and those shown in Lime Creek (Fig. 9) would need an average $E$ of 0.43 in order for these scenarios (shown in Fig. 9) to provide a 40% N reduction. These results were generated without the use of watershed simulation models. However, testing of planning scenarios such as these with simulation models is possible and encouraged in future research. None of the scenarios presented here is viewed as optimal for

![Table 4. Summary of practice information used in developing conservation planning scenarios.](image)

<table>
<thead>
<tr>
<th>Conservation practice (scale)</th>
<th>Scenario description/criteria</th>
<th>Number of fields/area treated†</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cover crops (in-field)</td>
<td>three scenarios:</td>
<td></td>
</tr>
<tr>
<td></td>
<td>all cropped fields</td>
<td>152 fields; 4637 ha</td>
</tr>
<tr>
<td></td>
<td>2/3 of cropped fields (at random)</td>
<td>101 fields; 2860 ha</td>
</tr>
<tr>
<td></td>
<td>1/3 of cropped fields (at random)</td>
<td>51 fields; 1763 ha</td>
</tr>
<tr>
<td>Controlled drainage (in-field)</td>
<td>any 1-m contour interval occupies &gt;50% of field</td>
<td>omitted (4 fields qualified) 51 fields, 1078 ha</td>
</tr>
<tr>
<td>Grassed waterways (in-field)</td>
<td>runoff contributing area &gt;0.75 ha</td>
<td></td>
</tr>
<tr>
<td></td>
<td>three scenarios:</td>
<td></td>
</tr>
<tr>
<td></td>
<td>critical risk fields</td>
<td>26 fields; 41 ha</td>
</tr>
<tr>
<td></td>
<td>critical and very high-risk fields</td>
<td>85 fields; 140 ha</td>
</tr>
<tr>
<td></td>
<td>critical, very-high-risk, and high-risk fields</td>
<td>117 fields; 203 ha</td>
</tr>
<tr>
<td>Wetlands (below-field)</td>
<td>lowest wetland along each tributary</td>
<td>11 wetlands‡ 7 wetlands</td>
</tr>
<tr>
<td></td>
<td>area of wetlands</td>
<td>15 ha</td>
</tr>
<tr>
<td></td>
<td>area of buffers</td>
<td>38 ha</td>
</tr>
<tr>
<td></td>
<td>area of cropland taken by wetlands</td>
<td>21 ha</td>
</tr>
<tr>
<td>Saturated buffers (below-field)</td>
<td>riparian assessment polygons (riparian zones with uniform slope below 27 tile drained fields)</td>
<td>40 riparian sites; 87 ha; 52 ha cropland</td>
</tr>
</tbody>
</table>

† “Area treated” indicates land areas directly occupied by the practice.
‡ Includes two existing wetlands.
either Beaver Creek or Lime Creek because our purpose is to provide a tool that can be used to engage local landowners in the planning process, which is the only viable approach to identify options suitable not only for a given watershed and its landscape but also for the farm management settings within that watershed. These options can only be identified through local consultation.

Conclusions

We have demonstrated a system to develop and test watershed-scale conservation planning scenarios using high-resolution, LiDAR-derived DEMs. As such DEMs become available, similar map products could be constructed at little cost for many Midwest HUC12 watersheds. Siting criteria for conservation practices can be used flexibly, and new types of practices could be included as their nutrient-reduction capacities become documented. Based on field reviews (Tomer et al., 2015), we believe accuracies for locating candidate sites for practices should be >80% when conditioning of DEMs is conducted accurately. Results indicate that if three or four practices are suitable for adoption in a watershed, then the spreadsheet scheme described here could identify watershed management alternatives most apt to meet reduction goals efficiently. Simulation models could be used to further test/refine scenarios. We are deploying ArcGIS software tools to facilitate use of the ACPF in Midwest watersheds, and an early version is being beta tested. A release version with user manual will be available online by 1 Oct. 2015 (USDA–ARS, 2015). Continued efforts to develop alternative nutrient-reduction practices, develop placement criteria for new/emerging practices, and understand controls on nutrient removal efficiencies among practices will allow this approach to inform local management in more watersheds. The approach could also identify strategies for P reduction if practices included are designed and/or known to reduce P loads.

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Fig. 9. Conservation planning scenarios for the Beaver Creek and Lime Creek watersheds, including implementation of cover crops on two thirds of all agricultural fields and nutrient removal wetlands and a third practice (i.e., saturated buffers in Beaver Creek and drainage water management in Lime Creek). If average nutrient removal rates anticipated using the Iowa Nutrient Reduction Strategy (INRS) were realized, The Beaver Creek scenario should achieve a 39% NO₃–N reduction and the Lime Creek scenario should achieve a 34% NO₃–N reduction based on results from the INRS (Nitrogen Science Team, 2013).
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