Spatial Optimization in Tallgrass Prairie Restoration: Tradeoffs between Ecosystem Services and Restoration Costs

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Senior Comprehensive Exercise

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Environmental Studies

Carleton College

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Abstract

Interest in projects to restore native land cover is increasing as the benefits derived from natural ecosystems and the negative effects of human development are becoming more consistently recognized (Bullock et al. 2011; Palmer and Filoso 2009; Loomis et al. 2000). In the American Midwest, the potential for cost-effective restoration of native grassland is greatest where high ecosystem service potential overlaps with low agricultural suitability. Grassland restoration typically incorporates cattle grazing to restore ecological functions and cattle operations may provide additional benefits in the form of profit on meat production (Fuhlendorf et al. 2010; Chaplin and Van Vleck 2014). Our study sought to answer the question: what are the tradeoffs between restoration costs and ecosystem service benefits when spatially optimizing restoration of grassland from agricultural land in Big Stone County, MN?

We calculated ecosystem services using InVEST and spatially optimized the outcomes with the reserve planning software Marxan, specifically targeting carbon sequestration, nutrient retention, and sediment retention. We generated a series of tradeoff frontiers, curves that plot the maximum ecosystem service benefit against restoration cost, to represent possible restoration plans that can be used by stakeholders to inform decision-making. Our results indicated that carbon can be sequestered cost-effectively regardless of which areas are restored. Conversely, we found that spatial optimization is essential to maximizing the retention of nutrients or sediment. Nutrient and sediment retention services depend on hydrological flow over the landscape, and our results demonstrated that restoring grassland along active waterways increases the benefits of restoration. Optimizing for nutrient or sediment retention effectively optimizes carbon sequestration, maximizing total ecosystem service benefit.

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List of Abbreviations

C	Carbon
CRP	Conservation Reserve Program (US Department of Agriculture)
DEM	Digital Elevation Model
DNR	Department of Natural Resources
InVEST	Integrated Valuation of Ecosystem Services and Tradeoffs (software by the Natural Capital Project)
LULC	Land use/land cover classification
MN	Minnesota
N	Nitrogen
NASS	National Agricultural Statistics Service (US Department of Agriculture)
P	Phosphorous
PAWC	Plant available water content
PET	Potential evapotranspiration
S	Sediment
SDR	Sediment Delivery Ratio
SNA	Scientific and natural areas
USDA	United States Department of Agriculture

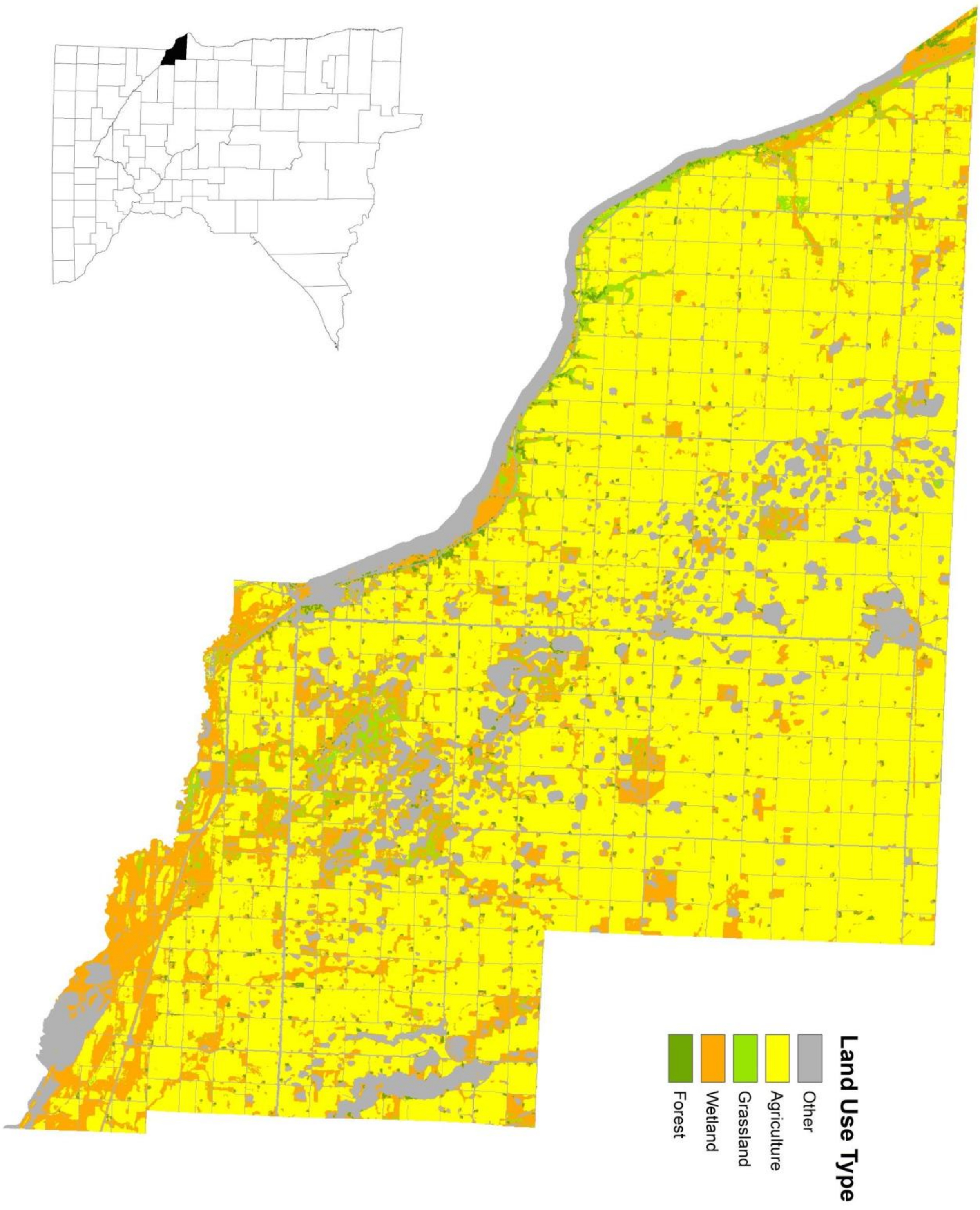


Figure 1. Current land use map of Big Stone County, MN. Land use types were aggregated to fit five major classifications: Agriculture, Grassland, Wetland, Forest, and Other (a combination of barren land, urban areas, and water, which were excluded from ecosystem service analysis) (USDA-NASS CDL 2014).

Introduction

Grassland Restoration

Projects to restore native land cover are undertaken to reestablish ecological processes and to realize the benefits provided by natural ecosystems (Apfelbaum and Chapman 1997; Vaughn et al. 2010). Governments, private organizations, and individual landowners are funding restoration projects more frequently in response to recognition of benefits such as carbon storage and water filtration, as well as increasing interest in mitigating the impacts of human activities such as deforestation and high-intensity farming (Bullock et al. 2011; Palmer and Filoso 2009; Loomis et al. 2000). In Minnesota, agricultural development has eliminated 98% of native tallgrass prairie land cover, also referred to as 'grassland' (Minnesota Prairie Plan Working Group 2011). While crop production is highly profitable, grasslands improve water quality in adjacent watersheds and increase landscape carbon storage, sediment retention, and biodiversity relative to agriculture (Nelson et al. 2009; Loomis et al. 2000; Chaplin and Van Vleck 2014).

The Minnesota Department of Natural Resources has begun to place higher value on tallgrass prairie and has developed conservation plans through modeling the potential benefits of restoration in select areas (Minnesota Prairie Plan Working Group 2011; Chaplin and Van Vleck 2014). Restoration projects, however, require costly conversions of one land use to another, and even the most high-priority conservation areas necessitate cost-effectiveness planning. The costs of restoration for farmers often exceed the individual benefits, even if their land is highly suited to restoration (Parks and Shorr 1997). To incentivize restoration, programs such as the federal Conservation Reserve Program (CRP) offer compensation to farm owners for converting land to conservation uses such as grassland (USDA Farm Service Agency; Parks and Shorr 1997; Ribaudo 1989).

Restoration of grasslands requires cost-benefit analysis on a large time scale to incorporate long-term management strategies and an understanding of the time required for the ecosystem to establish and persist (Apfelbaum and Chapman 1997; Fuhlendorf et al. 2008). Grasslands are dependent on a combination of fire and herbivory to maintain their ecological functionality (Freese et al. 2015; Fuhlendorf et al. 2008, 2010; Steuter and Hidinger 1999; Knapp et al. 1999). On restored lands, artificial burn treatments typically replace natural fire regimes (Fuhlendorf et al. 2008). Burning can be costly but helps prevent invasion of shrubs and other non-native grassland species and preserve insect and animal habitat (Van Dyke et al. 2004; Ansley et al. 2010; Moranz et al. 2012). Grazing in grasslands also supports the survival of native species by increasing the light and nutrients available to young plants (Veen et al. 2008). Raising cattle, whose grazing is functionally equivalent to the extirpated American bison, on restored grassland has been recognized as a complement to restoration goals, and cattle can provide an additional benefit in the form of landowner profit when raised for meat production (Fuhlendorf et al. 2010; Chaplin and Van Vleck 2014).

Ecosystem Services

Proposals for restoration typically weigh the economic costs of restoration against the potential benefits to human inhabitants, which can be measured through the calculation of ecosystem services. Ecosystem services are defined as the human benefits derived from ecological processes (Wong et al. 2015; Millennium Ecosystem Assessment 2005). They can be grouped into four categories: (1) provisioning services, including crops, livestock, water, and timber produced on a landscape; (2) cultural services, including recreational, aesthetic, and spiritual benefits; (3) supporting services, including soil formation, photosynthesis, and nutrient cycling; and (4) regulating services, including climate control, flood mitigation, and water filtration (Millennium Ecosystem Assessment 2005). Ecosystem service calculations are increasingly incorporated into land use planning because they enable the balancing of environmental benefits and other social benefits against costs through economic modeling (Nelson et al. 2009; Bullock et al. 2011; Palmer and Filoso 2009).

In the early 1900s, economists started raising concerns about the externality imposed on future generations by resource depletion, but it wasn’t until the 1960s that economic valuation of environmental resources became common (Gomez-Baggethun 2010). The 1990s brought greater attention to natural capital, most notably in the landmark paper by Costanza et al. (1997) that piqued policy interest in ecosystem services (Gomez-Baggethun 2010). Since then, a range of spatial modeling software packages have been developed to calculate ecosystem services. One such software is Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST), which has informed restoration planning across a variety of ecosystems in the U.S. (Nelson et al. 2009; Polasky et al. 2009; Chaplin and Van Vleck 2014; Palmer and Filoso 2009).

Land use change in agricultural regions tends to have significant ecological impacts on ecosystem services related to nutrient cycling and soil processes (Polasky et al. 2008; Sharp et al. 2015). More anthropocentric services such as recreational hunting and tourism are determined by human habits rather than specific land use change. In this study, we focused on four ecosystem services that rely on ecological processes and respond significantly to changes in vegetation cover: carbon sequestration, nitrogen and phosphorous retention, and sediment retention.

Carbon sequestration is a well-studied ecosystem service that is relatively simple to measure and critical on a global scale (Nelson et al. 2009; Nelson et al. 2010; Millennium Ecosystem Assessment 2005). Carbon sequestration can increase when natural land covers replace agricultural or marginal land (Lal 2004). Sequestering carbon in plant biomass can help mitigate the effects of global climate change including sea level rise, species loss, and collapse of marine food chains (Millennium Ecosystem Assessment 2005; Nagelkerken and Connell 2015). Locally in Minnesota, threats to both industrial and natural systems such as drought, extreme weather, and species invasions may be mitigated by sequestering carbon in vegetation (U.S. Global Change Research Program 2014; Crowl et al. 2008).

Ecosystem services that are based on hydrological processes, such as nitrogen, phosphorous, and sediment retention, respond significantly to land use changes (Meehan et al. 2013; Kovacs et al. 2013; Yang et al. 2003). Nutrients and sediment travel through waterways and pose threats to human and ecosystem health both in local watersheds and far downstream (Carpenter et al. 1998; Sharp et al. 2015; Poudel et al. 2013). Nutrient pollution, including both nitrogen and phosphorous, is high under intensive agricultural land use practices because of overapplication of fertilizers, inattentiveness to slope and retention practices, and inattentiveness to hydrological connectivity with nearby waterways (Carpenter et al. 1998; Hernandez et al. 2010). Nutrient pollution decreases water quality and increases eutrophication which causes die-offs of fish populations locally and as far away as the Gulf of Mexico, where a huge 'dead zone' is attributed to the influx of nutrients from the Mississippi River watershed (Miltner and Rankin 1998; Rabotyagov et al. 2014). Sediment export is an additional threat to water quality: it is the most common non-point source pollution in the U.S. and causes $16 billion in environmental damage annually (Mid-America Regional Council 2012; Yang et al. 2003). Sediment loading increases the cost of treating drinking water, disrupts riparian food chains by destroying habitat for microorganisms, and disrupts recreational uses of waterways (Mid-America Regional Council 2012). Additionally, soil erosion decreases the productivity of arable land, preferentially removing organic carbon and clay contents and, with these, certain essential nutrients and the potential for increasing root depth (Yang et al. 2003). Accelerated erosion from human land use accounts for 70% of all erosion in the U.S. and land cover conservation measures are considered among the most important tools for slowing this degradation (Mid-America Regional Council 2012; Hamel et al. 2015; Yang et al. 2003).

Spatial Optimization

Recent studies on ecosystem services in the context of restoration projects have found optimizing restoration plans based on spatial characteristics helps to maximize the provision of ecosystem services. Nelson et al. (2009) used InVEST to predict changes in ecosystem services, biodiversity conservation, and commodity production levels in response to scenarios of land use change identified by the conservation goals of stakeholders including government agencies, non-governmental organizations, and universities. They concluded that quantifying tradeoffs between ecosystem services in a spatially explicit manner can help make natural resource decisions more effective, efficient, and defensible. Bullock et al. (2011) similarly found that restoration projects can be effective in enhancing ecosystem services, but that schemes to pay landowners for ecosystem service provision need to be further developed in order to optimize multiple services. Polasky et al. (2008) developed a spatially explicit landscape-level model to identify efficient land use patterns that maximize biodiversity conservation objectives for specific levels of economic returns. The study concluded that both biodiversity conservation and the value of economic activity could be increased substantially through spatially optimized land use decisions.

The Minnesota Prairie Conservation Plan is an attempt at spatially optimizing grassland and wetland restoration in regard to ecosystem services. Using spatial analysis, the Plan seeks to optimize ecosystem services against restoration costs and the opportunity cost of agriculture, and examines the cattle-raising potential of grasslands (Chaplin and Van Vleck 2014). The Minnesota Prairie Conservation Plan is a response to Minnesota's shrinking native prairie land, geared towards protecting grassland and wetland habitat. It identifies core conservation areas and creates a vision of a connected prairie landscape from Canada to Iowa. Its main goals are (1) to protect the remaining native prairie in Minnesota, (2) to make prairie core areas and connecting corridor complexes with at least 20% wetland and 40% grassland cover, (3) to make the bordering agricultural landscape more wildlife-friendly by conserving at least 10% of land cover in native perennial vegetation, and (4) to carry out this conservation work utilizing grass-based agriculture in cost-effective ways that are supported by local communities. Chaplin and Van Vleck (2014) studied two landscapes in western Minnesota, the Agassiz Beach Ridges and the Glacial Lake region, to develop the information and techniques needed to achieve the four goals of the Prairie Conservation Plan through spatial optimization.

Where high ecosystem service potential and low agricultural suitability overlap, the potential for cost-effective restoration is high. Spatial optimization of restoration decisions can be extended from ecosystem service calculations to incorporate the costs of restoration through tools such as the reserve planning software Marxan. Marxan is used to generate 'reserve solutions,' a set of areas that, if restored to native land cover, will fulfill conservation goals defined by the user. Marxan selects areas with high conservation potential and low costs to optimize the user’s conservation goals (Nemec and Raudsepp-Hearne 2012; Adame et al. 2014). The Scientific and Natural Areas (SNA) Strategic Land Protection Plan, a program of the Minnesota DNR, used Marxan to prioritize areas for biodiversity and natural resource conservation, but it did not calculate ecosystem service benefits (SNA Strategic Planning Team 2011). Outside Minnesota, Marxan has been employed to optimize restoration planning only in combination with coarsely-estimated ecosystem service benefits (Ball et al. 2009; Adame et al. 2014; Delavenne et al. 2012). Combining Marxan with detailed ecosystem service calculations may significantly improve the accuracy of the reserve solution generated.

Our Study

Big Stone County, MN served as the site for our research because it is a heavily agricultural county with data available on the individual farm level (referred to as ‘parcels’), it was not included in the Minnesota Prairie Conservation Plan, and because its location bordering the Minnesota River makes agricultural runoff particularly relevant to water quality concerns (Minnesota River Basin Data Center; Minnesota Pollution Control Agency). Our study sought to provide information to decision-makers in future restoration decisions by analyzing the costs and spatial arrangements of reserve solutions that prioritize four different ecosystem services: carbon sequestration, nitrogen retention, phosphorous retention, or sediment retention. Using a novel pairing of the InVEST and Marxan models, we asked: what are the tradeoffs between restoration costs and ecosystem service benefits when spatially optimizing restoration of prairie from agriculture in Big Stone County?

Our study is the first in Minnesota to combine InVEST-type ecosystem service calculations with Marxan-enabled reserve planning. The Minnesota Prairie Conservation Plan analyzed ecosystem services but has yet to spatially optimize the results, while the SNA Strategic Land Protection Plan used Marxan to prioritize conservation areas but did not consider ecosystem services (Minnesota Prairie Plan Working Group 2011; SNA Strategic Planning Team 2011). Combining InVEST ecosystem service modeling with the optimization power of Marxan has the potential to inform smarter decisions in Minnesota prairie restorations and beyond.

We identified the economic costs and benefits of achieving a series of ecosystem service goals in order to generate tradeoff frontiers for the provision of ecosystem services. Tradeoff frontiers represent the maximum benefits of restoring ecosystem services for a given cost, optimized through the efficient spatial allocation of land uses. Other researchers have combined economic and biological models to generate tradeoff frontiers for given land use scenarios (Polasky et al. 2009; Calkin et al. 2002; Nalle et al. 2004), but ours was the first to generate optimal scenarios based on cost-benefit analysis of individual land parcels through reserve planning software like Marxan.

In our analysis the costs of restoring agriculture to prairie include the costs of plowing, re-seeding, and long-term maintenance including burning and managed grazing, plus the opportunity cost of agriculture, defined as the agricultural revenue given up under restoration, and the cost of cattle management. Benefits include profit on cattle and ecosystem services. Besides increasing carbon sequestration and nutrient and sediment retention, converting agricultural land to prairie increases biodiversity, provide opportunities for recreation such as hunting, and facilitate crop pollination (Minnesota Prairie Plan Working Group 2011; Sharp et al. 2015). Our study did not address these potential services because their data requirements were significantly more complex and extensive (Sharp et al. 2015; Nelson et al. 2009; Polasky et al. 2008). We discuss at the end of this paper the potential for further refining reserve planning by analyzing additional services.

Our study employed a sensitivity analysis to determine how sensitive our ecosystem service modeling results are to differences in parameters and input data. Sensitivity analysis has been employed in other research to improve conservation management by focusing analysis on the data and parameters that matter most. Lautenbach et al. (2013) employed a sensitivity analysis to select an ideal set of parameters for use in calibration in their optimization-based tradeoff analysis of biodiesel crop production,. Lonsdorf et al. (2009) likewise used sensitivity analysis in modeling pollination services across agricultural landscapes, finding that fine-scale resources are important to pollinator service delivery. Our research sought to optimize cost-effective prairie restoration planning in Minnesota by looking at the restoration of individual farms; the unprecedented closeness of this scale necessitates an understanding of where more detailed knowledge of model input variables could improve the reserve solutions identified.

In the following sections, we detail our methods for ecosystem service calculations and spatial optimization, display and explain the implications of our tradeoff frontiers, and discuss the significance of our results and any uncertainty therein. Generally, our study found that carbon sequestration can be efficiently maximized by spatially optimizing any other service (nitrogen, phosphorus, or sediment retention). This is because carbon sequestration does not vary across the landscape, while the others are tied to hydrological flow and see higher benefits relative to costs if restoration is focused along waterways. Stakeholders should therefore focus their attention on these latter services, without regard to carbon sequestration, as carbon will be sequestered efficiently regardless of where in the landscape restoration takes place.

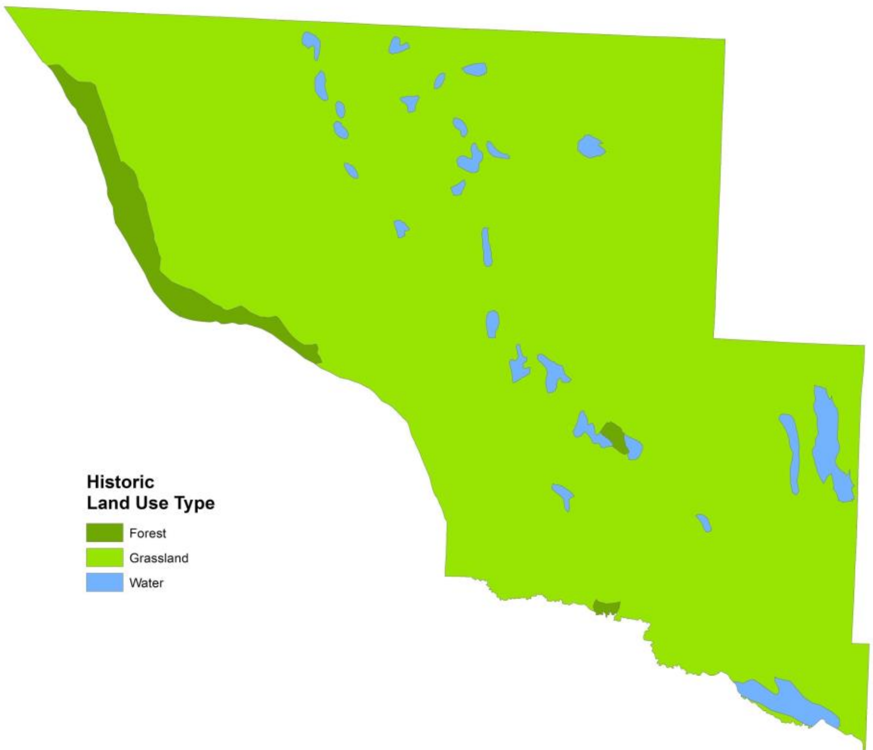


Figure 2. Historic vegetation cover of Big Stone County, MN. Land use types were aggregated to fit three major classifications: Grassland, Forest, and Water (Minnesota DNR 2015).

Methods

Study Area

Big Stone County, located on the western border of Minnesota along the Minnesota River, is a heavily agricultural county that lies within a region historically dominated by prairie (Minnesota Prairie Plan Working Group 2011) (Figure 2). Approximately 75% of the land in Big Stone was devoted to cropland in 2013 (Big Stone County Water Plan Task Force 2013). The County has a shoreland management program and a grant-funding system for projects that improve water quality (Big Stone County 2015). Each of these depends partially on the USDA-sponsored Conservation Reserve Program, in which approximately 2.4% of land is enrolled (Big Stone County Water Plan Task Force 2013).

Roughly 75% of the land in Big Stone County drains into the Upper Minnesota River (Lake Pepin Legacy Alliance 2016). Because of its impact on the Minnesota River watershed, grassland restoration in Big Stone has high potential for ecosystem service provision through the reduction of nutrient export and erosion. Ecosystem service potential combined with the high proportion of agricultural land makes Big Stone County a high-impact candidate for prairie restoration optimization research.

Land Use Scenarios

We compared two scenarios in our study of the restoration potential of Big Stone County. The first scenario is the current or 'agricultural' land use, Big Stone as-is, with all currently agricultural land remaining in agriculture (Figure 1). The second is the 'grassland' land use, a Big Stone County in which all predominantly-agricultural parcels are restored to grassland. Analyzing these two extremes allowed us to determine the total restoration potential in terms of ecosystem service benefits, compare it with the cost of restoring any individual parcel of land, and weigh these costs and benefits to identify which parcels would provide the most cost-effective benefit from restoration depending on the ecosystem services in question. In the following section, we discuss the models used to conduct this analysis.

Choice of Models

Our optimization of different land-use scenarios used two modeling softwares: InVEST, which calculates ecosystem services, and Marxan, a conservation reserve planning program. To produce consistent results, ecosystem service modeling suites such as InVEST use ecological production functions to translate ecological processes into anthropocentric services. Production functions are mathematical expressions that convert changes in ecological functions to ecosystem outputs, such as nutrient export levels or soil erosion (Wong et al. 2015). Two common modeling systems are InVEST (Sharp et al. 2015) and ARtificial Intelligence for Ecosystem Services (ARIES) (Villa et al. 2014). Both InVEST and ARIES are designed to map the spatial distribution of ecosystem services based on parameterized environmental and management factors (Bagstad et al. 2013). Both are well-documented, quantitative approaches to landscape-level solutions that provide biophysical values with economic valuation potential. Other modeling suites (LUCI, MIMES, EcoServ) are under development or in the initial stages of case study evaluation and peer-review, and will not be reviewed here. While ARIES and InVEST produce similar results, InVEST is more extensively peer-reviewed, grounded in ecological production functions, more applicable to data-rich areas, more generalizable, open source, and requires substantially less time to parameterize than ARIES (Bagstad et al. 2013; Nemec and Raudsepp-Hearne 2013; Villa et al. 2014; Wong et al. 2015). Still, InVEST provides only a biophysical and economic output value per designated grid area, and is not designed to select areas for restoration as it does not perform a cost-benefit analysis for restoring a given area of land. For this, it is necessary to turn to reserve planning software.

Reserve planning models select areas for inclusion in a conservation 'reserve' based on the costs and benefits associated with each individual area. They maximize benefits and minimize costs while attempting to either reach a benefit target or stay within a cost budget. The user chooses the specific costs and benefits the model works with, as well as the benefit targets and cost budget constraints. Marxan and Zonation are two comparable, high-profile, open-source reserve planning models. Marxan was developed at the University of Queensland to assist in reserve planning on the Great Barrier Reef and has since been applied in a variety of ecosystems to optimize conservation projects (Ball et al. 2009; Adame et al. 2014; Delavenne et al. 2012). Zonation was developed at the University of Helsinki to help with species connectivity estimates, and expanded into a reserve planning software across a variety of ecosystems and continents (Minin et al. 2014). Both produce similar results, although Marxan provides more efficient reserve outcomes while Zonation prioritizes higher levels of reserve connectivity (Delavenne et al. 2012). Marxan additionally implements simulated annealing and iterative improvement to find an optimal reserve solution. Iterative improvement is an optimization process that conceives an initial reserve plan, adds or removes planning units (Marxan's smallest unit of management) to the reserve system, and eventually lands upon an efficient combination of planning units. Simulated annealing builds upon iterative improvement by making certain that the model does not get stuck prematurely on an inefficient solution. Marxan then generates a 'score' for each of the reserve solutions it finds based on the solution's ability to meet conservation targets and the cost of land conversion. The solution with the best score maximizes conservation targets while minimizing costs (Game and Grantham 2008).

The determination of specific reserve solutions is useful to organizations operating on private lands or governments owning large tracts of land, especially when they have budgets allocated toward conservation (such as in Adame et al. 2014). However, where parcels of interest are controlled by thousands of different individuals, no single solution could provide useful information to stakeholders in Big Stone County. Rather than determining a single reserve solution for Big Stone County, our study sought to provide decision-makers with data to evaluate and use in future restoration projects by comparing the costs of a variety of restoration solutions that prioritize different ecosystem services. In the following sections, we explain how we ran InVEST and Marxan on several different scenarios to show the relationship between different potential conservation priorities using tradeoff frontiers. The conservation priorities considered were carbon sequestration potential, landscape nutrient export, and landscape sediment export.

Table 1. Descriptions of InVEST input variables.

InVEST model	Term	Definition
Carbon Sequestration	**Belowground Carbon**	Tons of carbon stored on one hectare of a land use in belowground organic matter. A mean value derived from studies using several sources including field estimates, values extracted from meta-analyses on specific habitat types or regions, and general tables published by agencies like the IPCC (Sharp et al. 2015).
Carbon Sequestration	**Soil Carbon**	Tons of carbon stored on one hectare of a land use in soil, excluding biomass. A mean value derived from studies using several sources including field estimates, values extracted from meta-analyses on specific habitat types or regions, and general tables published by agencies like the IPCC (Sharp et al. 2015).
Nutrient Retention and SDR	**Threshold Flow Accumulation**	A calibration coefficient used to match the hydrology of the model to the hydrology of the region. The value indicates the threshold for upstream catchment area, in number of pixels (30 m^2 areas of land), at which a pixel is designated as part of a stream network, and no longer retains nutrients or sediment (Sharp et al. 2015). A pixel is considered part of a stream in our analyses when 500 or more pixels flow into it from higher elevations.
Nutrient Retention	**Kc**	The plant evapotranspiration coefficient describes the sum of evaporation and plant transpiration in comparison to a reference crop, which is alfalfa (Wright 1982). Evapotranspiration is calculated from field measurements and calibrated based on climate, management, and crop factors (Allen et al. 1998).
Nutrient Retention	**Root depth**	The soil depth to which roots must proliferate, in mm, in order to extract the water demanded by the land cover (Raes et al. 2012).
Nutrient Retention	**Export coefficient (loading coefficient): load_N, load_P**	The nutrient load for a given land use, expressed in kg/ha/yr (Sharp et al. 2015). This 'load' is to surface waters, and is also described as the loss of a quantity of nutrient from a hectare of land. These values are estimated from data collected on nutrient transport and source factors specific to land uses, such as erosion, streamflow, contributing distance from surface waterbodies, rate of fertilizer application, and tests of soil nutrient levels where available (Mulla et al. 2004).
Nutrient Retention	**Retention efficiency: eff_N, eff_P**	The maximum retention efficiency for a land use class, expressed as a proportion between 0 and 1 of the amount of nutrient from upstream retained by the land use's vegetation (Sharp et al. 2015).
SDR	**Max SDR**	The fraction of soil particles finer than coarse sand. This description of soil texture dictates the maximum sediment load that may leave a parcel. This ratio is used for calibration in advanced studies but the default value of 0.8 is adequately accurate in most environments (Sharp et al. 2015).
SDR	**Borselli iC_0**	The connectivity index, which defines the relationship between the SDR and the degree of hydrological connectivity on the landscape. It is landscape-independent (Vigiak et al. 2012) and the default value of 0.5 is recommended by InVEST's authors (Sharp et al. 2015).
SDR	**Borselli k_b**	The k_b constant is also used to describe the shape of the relationship between the sediment delivery ratio and the degree of hydrological connectivity on the landscape. It is not physically based and may be used for model calibration, but the default value of 2 is widely accepted and the science on sediment modeling is still developing to refine use of this parameter (Sharp et al. 2015).
SDR	**Cover Management (C) Factor**	The C factor is a floating point value between 0 and 1 (Sharp et al. 2015). The ratio represents deviation from a standard scenario of soil loss under clean-tilled, continuous-fallow conditions (Michigan State University 2002). Used to estimate the effect of cropping and management practices on erosion rates (Grigar 2002).
SDR	**Support Practice (P) Factor**	The P factor is a floating point value between 0 and 1 (Sharp et al. 2015). The ratio represents deviation from a standard scenario of straight row farming without contouring to slopes. The factor decreases as more effective erosion control measures, which may include contouring or stripcropping, are used (Michigan State University 2002).

Calculating Carbon Sequestration Potential

Carbon sequestration is the amount of carbon stored in aboveground or belowground living biomass, soil, and detrital mass (Sharp et al. 2015). InVEST's Carbon Sequestration model calculates the potential landscape carbon sequestration based on land-use classification. Our inputs included belowground and soil carbon storage values for our two variable land-use classes, Agriculture and Grassland, neither of which stores significant amounts of carbon in aboveground living or dead mass (Table A2) (Sharp et al. 2015). It was unnecessary to include Other, Forest, or Wetland—our other generalized land-use classes—in this analysis because none of these areas will change as part of agriculture-to-prairie restoration in Big Stone County.

We ran the Carbon Sequestration model separately on both the agricultural land use scenario and the grassland land use scenario and recorded carbon storage values for all land-use parcels. We ran the model at a scale of 30 m^2, and assumed a time scale of 100 years in calculating storage in grassland, long enough to allow the ecosystem to restore exhausted soil carbon stocks through ecological succession (Polasky et al. 2010; McLauchlan 2006; Nelson et al 2009). Each of these parcels can be independently analyzed or selected for a reserve because the carbon sequestration on any parcel of land depends only on the land use and not on conditions in any neighboring parcels.

Calculating Landscape Nutrient Export

Nutrient retention is defined as the ability of a landscape's natural vegetation to filter nutrients out of groundwater that flows through it. The converse, nutrient export, is measured as the amount of nutrients that run off of a land area in water as it flows across a particular land use (Sharp et al. 2015). The Nutrient Retention model calculates export of nitrogen (N) and phosphorous (P) into watersheds, using a combination of spatial and hydrological conditions and the nutrient loading and retention potential of a land use class. Because this model is fundamentally spatial and depends upon the flow of water across neighboring parcels in order to generate an N-export or P-export value for a given raster pixel in ArcGIS (a 30 m^2 area on the land), we included five major land-use classes in our analysis: Other, Agriculture, Grassland, Forest, and Wetland (Figure 1).

For each land use we identified the best-available values in the literature for the plant evapotranspiration constant (Kc), root depth, nutrient (N and P) export or loading coefficients, and nutrient (N and P) filtration efficiency. Table 1 describes these variables; Table A4 identifies our data sources. The biophysical constants were used to determine the amount of nutrient export per pixel.

We ran the model at a scale of 30 m^2. We set the threshold flow accumulation at 500 pixels based on visual analysis of a DEM-generated flow map overlaid on a satellite image of the county (Table 1). InVEST developers recommend this method of visual analysis for study sites like Big Stone County where no verified stream data layer exists (Sharp et al. 2015). In our analysis, a pixel is considered part of a stream when 500 or more pixels flow into it from higher elevations. This threshold generated a stream layer which closely matched streams visible on the satellite imagery.

We ran the Nutrient Retention model on the agricultural and grassland land use scenarios. Our output was generated in terms of tons of nutrient export per pixel, and we subtracted the grassland output from the agricultural output to find the reduction in nutrient export from each parcel if it were restored from agriculture to grassland. Subsequent analyses focused on reducing nutrient export because the effects of agricultural nutrient export on water quality are negative and the retention of those nutrients can be measured as an ecosystem service (Hernandez et al. 2010; Carpenter et al. 1998).

It is important to note that our model assumes a grassland parcel surrounded entirely by grassland will provide the same nutrient retention services as a grassland parcel in a landscape that includes some agricultural land parcels. It was not feasible to run InVEST on all 2^{2531} possible combinations of agricultural and grassland land-use scenarios. To determine the effect of upstream or adjacent land uses on the provision of hydrologically-based ecosystem services, and whether it impacted the validity of our reserve solutions, we developed a test that measured the degree of this effect on the export of N and P from a parcel. This test is described below under 'Quantifying Spatial Assumptions in InVEST.'

Calculating Landscape Sediment Export

The Sediment Delivery Ratio (SDR) InVEST model calculates the proportion of soil loss reaching the outlet of a watershed from erosion within the watershed land area (Sharp et al. 2015). Inputs include the cover-management (C) factor and support-practice (P) factor for each of our five land uses (Table A6), as well as spatial data including a digital elevation model (DEM), precipitation layer, and erosivity and erodibility measures; each of these layers was pulled from government databases (Table A1). We again set the threshold flow accumulation at 500 based on our visual analysis of the watersheds in Big Stone and its correspondence to the DEM (Table A5). Other model coefficients (Borselli k_b, Borselli iC_0, Max SDR) were given default values as prescribed by Sharp et al. (2015) (Table A5).

This model operates under the same assumptions as the Nutrient Retention model, described above, in that we compared sediment export values between parcels from a strictly-agricultural to a strictly-grassland scenario. We applied the same test to our results to test the limitations of this assumption.

Identifying Cost-Effective Reserve Solutions

We transferred our final InVEST outputs to Marxan inputs in the form of 'conservation features.' Each ecosystem service (C, S, N, and P) was entered into the model as a conservation feature which could reach 100% of its potential on the landscape if every agricultural parcel in Big Stone County was restored to grassland. Running Marxan required three major pieces of information: (1) the cost of transforming any parcel from agriculture to prairie, stored in the pu.dat file; (2) the target numbers for each of our conservation features, as derived from InVEST based on maximum landscape potential for ecosystem services (tons of C stored, and tons of N, P, or S prevented from being exported into the watershed) stored in the spec.dat file; and (3) data on how much of a conservation feature is gained by restoring one of the 2,531 potential parcels, also derived from InVEST, stored in the puvpsr2.dat file.

We converted our InVEST data into Marxan input files using a series of ArcMap tools. We clipped raster output files to our land parcel layer using Zonal Statistics and converted ecosystem service benefits for each parcel into the format of the puvpsr2.dat file. We aggregated these ecosystem service benefits to the total potential landscape value to create the spec.dat files.

We aggregated economic data on the cost of land conversion to generate our pu.dat file. Cost, provided to the model on a per-area basis, included restoration cost (one-time seeding costs and net present value of management costs), opportunity cost of agriculture (net present value of the production of different crops), cost of cattle management (net present value), and benefit of raising cattle for profit (net present value). These inputs and their sources are documented in Table B1. We were unable to account for relative decreases in fixed costs if working with a conservation area larger than a single parcel, but because most adjacent parcels are held by different landowners who will likely make decisions independently, we made the assumption that the cost of entry for any parcel into a reserve solution would be independant of the status of other parcels in the solution.

For each of our four ecosystem services, we ran Marxan ten times to find reserve solutions that optimized each service individually. We varied the target amount of ecosystem services from 10% to 100% of the landscape potential in Big Stone County, using 10% increments. We ran Marxan using the Simulated Annealing and Iterative Improvement features, with 10 million 'iterations per run' and 100 'repeat runs' for each of our forty sets of ecosystem service conservation targets.

Generating Tradeoff Frontiers and Calculating Gini Coefficients

We generated tradeoff frontiers for each of the optimized ecosystem services against the costs of the associated reserve solutions (Figure 3). Each of the tradeoff frontiers is a visualization of the output of ten Marxan runs, with an individual ecosystem service target increasing from 10% to 100% of its total landscape potential in Big Stone County. In each tradeoff frontier we included 'residual' ecosystem services that were not optimized in each model: for example, when we optimized C sequestration by cost, the parcels selected for restoration also increased the ecosystem services of N, P, and S retention. These 'residual' benefits contribute to the overall benefit of the reserve solution without being accounted for during optimization. We included each residual ecosystem service in our tradeoff frontiers to capture the unintended benefits of possible reserve solutions that were optimized based on only one ecosystem service (Figure 4).

For each optimized ecosystem service and the curves describing the residual services that accompany them, we calculated Gini coefficients. Gini coefficients describe the difference between expected production of an ecosystem service, based on cost input into restoration on randomized land areas (a straight line from cost of 0 to maximum cost), and the amount of the service that was actually achieved through spatial optimization. We linearly interpolated and calculated the areas between each curve and its corresponding line of 'expected' provision. We divided these areas by the area under the 'expected' line to find the Gini coefficients. Gini coefficients range from 0 to 1; a number nearer to 1 indicates that a reserve solution provides greater benefits than what a spatially random solution would provide.

Sensitivity Analyses on InVEST Variables

We performed sensitivity analyses on all of the non-spatial InVEST input variables in order to determine which values—many of which have only been published by two to three sources—are the most urgent candidates for further research based on their importance to the model. Sensitivity analysis was performed by running InVEST several times, changing the input of a single variable each time between low, mean, and high values. Where multiple values were reported in the literature, the low and high represented the reported extremes; where two or fewer were reported, the low and high were generated by halving and doubling the best available data point. We compared the model output from each parameter's low, mean, and high InVEST runs to see if the effect on output was proportional to the change in inputs.

Quantifying Spatial Assumptions in InVEST

We ran InVEST on landscapes that either had a fully intact agricultural landscape (agricultural land use) or had all agricultural parcels restored to grassland (grassland land use). In the C Sequestration model each parcel provides an ecosystem service based solely on land cover, but in the Nutrient Retention and SDR models parcels are connected to each other through the flow of water across the landscape, and so the type of land cover next to or upstream of a parcel can affect the amount of retention services provided by that parcel. Other studies have suggested the relationship between a restored parcel and other land uses is significant, especially when the services are tied to landscape-level factors such as hydrologic processes (Gleason et al. 2011; Wang et al. 2014). Addressing all possible impacts of spatial heterogeneity would have required running InVEST 2^{2531} times, once for each possible combination of parcels restored. To make our modeling process feasible we ignored these potential effects and studied the change in ecosystem services provided when Big Stone's agricultural parcels were restored entirely to grassland.

To ascertain whether restored grassland parcels would provide significantly different levels of N, P, or S retention when situated near agricultural parcels, we ran the Nutrient Retention and SDR models on four 'test landscapes' in which only one parcel was selected for restoration, leaving the remainder in agriculture. The four parcels we used in these tests had been consistently chosen for restoration in our Marxan-generated reserve solutions, and represented some of the best land available for restoration in Big Stone. Each parcel was adjacent to a waterway and showed high restoration potential in our original InVEST runs. We recorded the percentage difference in ecosystem services provided on these parcels between the original and 'test' runs to analyze whether the difference between a grassland parcel receiving high agricultural runoff and a grassland parcel receiving reduced runoff from an all-grassland landscape was significant.

Results

Optimization

We optimized each ecosystem service—Carbon (C), Nitrogen (N), Phosphorus (P), and Sediment (S)—with regard to the total cost of restoration. While all can be optimized in cost-effective restoration solutions, the relationship between cost and restoration potential varied greatly depending on which ecosystem service was chosen. C exhibited the least efficient relationship between cost and restoration potential: the amount of C sequestration was directly proportional to the cost of a restoration solution, with a Gini coefficient of 0.055 (Figure 3). The other three services had more efficient curves. S retention was the most efficient (Gini coefficient of 0.55), followed by N retention (Gini coefficient of 0.42) and P retention (Gini coefficient of 0.37). Initially the marginal cost of N, P, and S retention was low, and small increases in cost had disproportionately large increases in restoration potential (Figure 3). The threshold value, the point at which the marginal cost of restoration became greater than the marginal ecosystem service benefit, varied between models. The S threshold occurred at a cost of 0.71 billion USD and a benefit of 70% of the total possible S retention (Figure 4b). The N threshold was slightly more expensive (1.08 billion USD), but with slightly more benefit as well (80% of the total possible N retention) (Figure 4c). The P threshold was even more expensive (1.33 billion USD) without an increase in benefit (80% of the total possible P retention) (Figure 4d). There was no discernable C threshold, as the ratio of marginal costs to benefits fluctuated around 1:1 throughout its entire restoration potential (Figure 4a). Because we did not convert ecosystem services to economic values, the thresholds were calculated from the proportion of total benefit and the proportion of total cost.

The tradeoff frontiers reflect the relationship between ecosystem services and the landscape. C sequestration potential does not vary across an agricultural landscape, as any parcel of agricultural land restored to prairie will sequester C proportional to its area (Figure D1). Figure 5a shows this lack of spatial variability, as most agricultural parcels were selected by our optimization model at a medium to high frequency; there was no repeated spatial pattern in which parcels were selected. Conversely, N, P, and S retention potentials are dependent on water flow across the landscape, with areas immediately adjacent to active waterways exhibiting high potential relative to areas further away. Parcels optimized for N show a disjointed visual spatial pattern, but a few specific parcels were selected in the majority of optimization runs, while the rest were selected infrequently (Figure 5c). Relative to N (Figure D2a), P retention potential was more diffuse across the landscape (Figure D3a). This meant more parcels were chosen at mid- to high-frequencies in the P-optimized solutions than in the N-optimized solutions, as they offered a higher amount of P retention than of N retention. Still, similar spatial patterns hold for both: a few important parcels were selected in nearly every optimization run, while the rest of the landscape is selected infrequently (Figure 5d). Spatial optimization had the most visible effect on S retention patterns, as most of the landscape had low potential save for a few select areas that had extremely high levels of potential retention (Figure D4a). Figure 5d shows how closely S

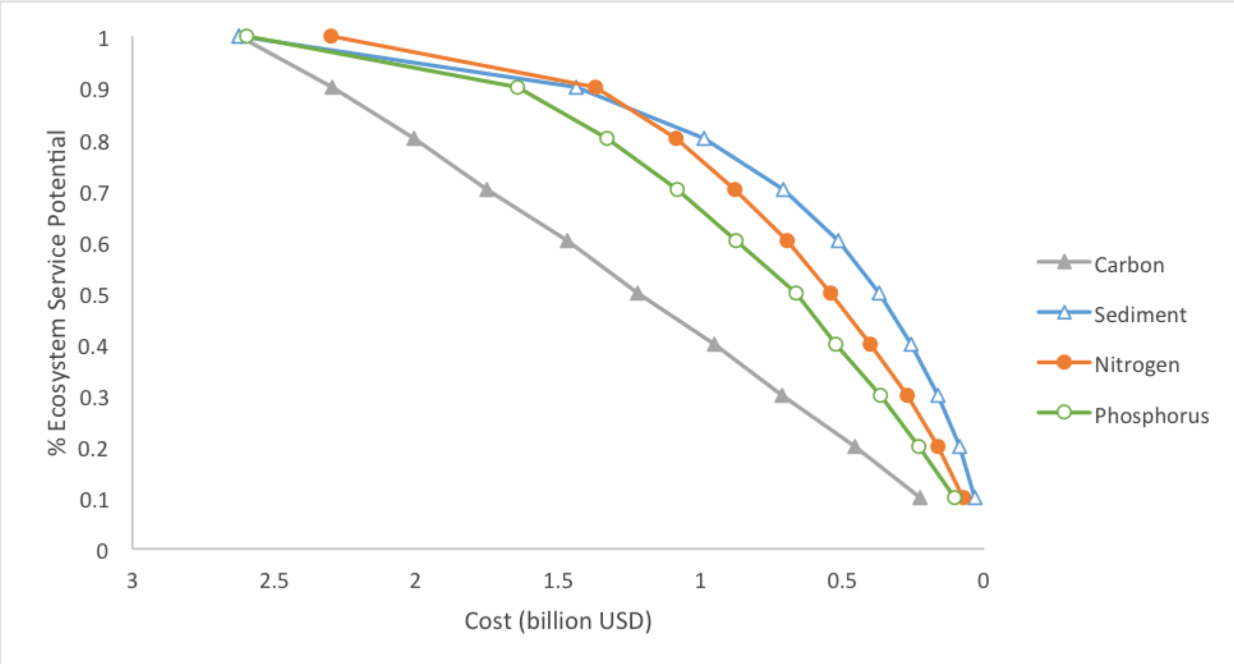


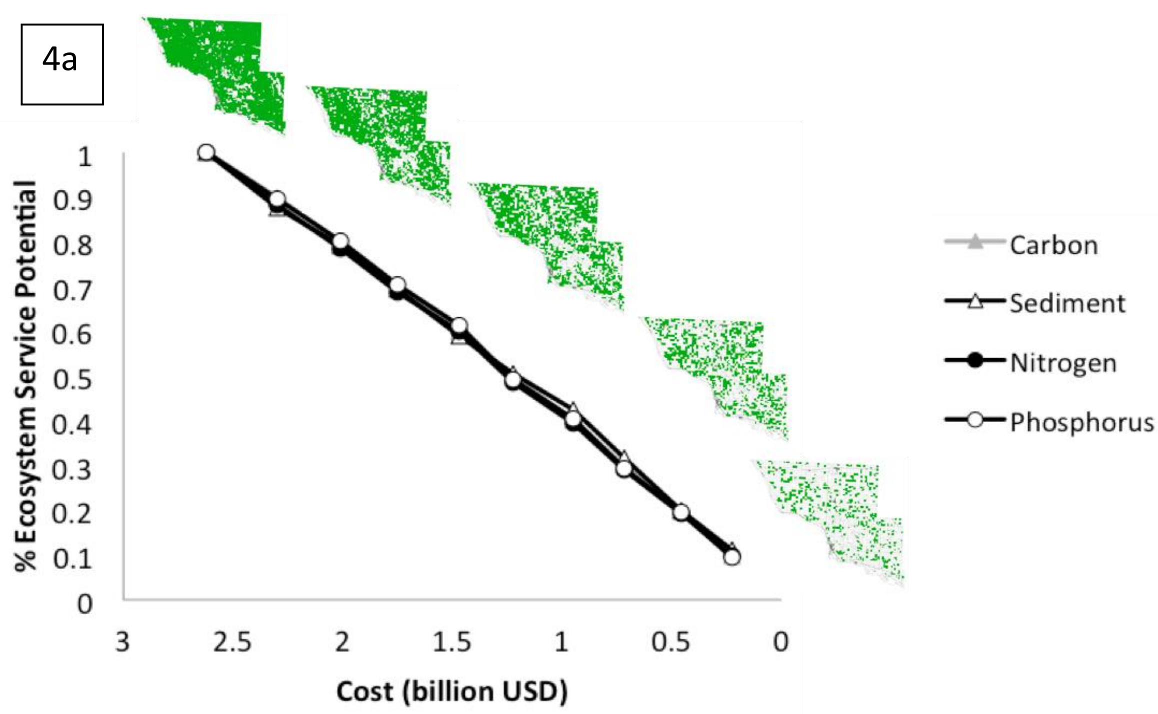
Figure 3. Optimized tradeoff frontiers between the proportion of total landscape potential for each ecosystem service and the total cost of restoration. Each dataset represents a scenario that optimizes that particular ecosystem service alone in regard to cost. Cost is decreasing along the x-axis to visualize the relationship as a tradeoff of benefits by showing 'cost reduction.' C was the least cost-efficient (Gini = 0.055), followed by P (Gini = 0.37), N (Gini = 0.42), and S as the most efficient (Gini = 0.55). Total costs at 100% of ecosystem service potential do not align as certain models can achieve a complete solution without restoring every possible parcel due to the spatial heterogeneity of particulate export based on elevation.

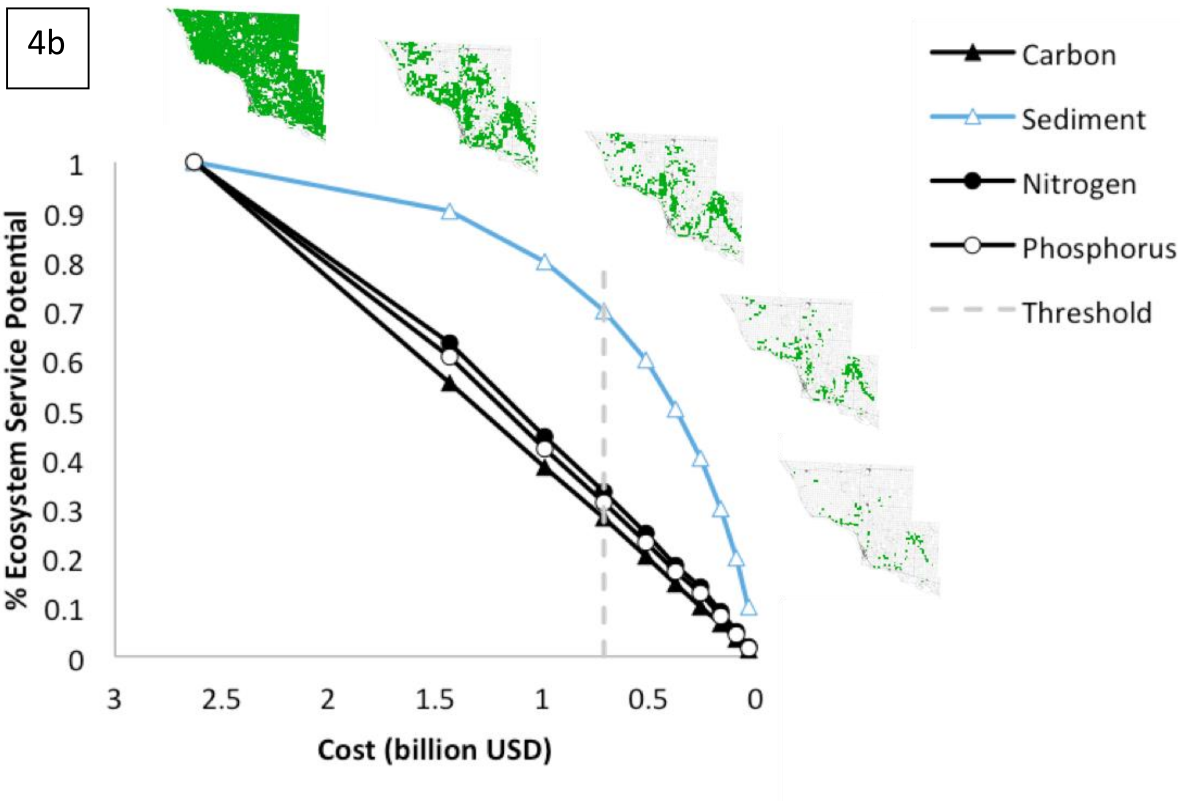
retention was tied to the landscape: most parcels were chosen less than 20% of the time, while select parcels along major waterways were selected in nearly every solution. Overall, this heterogeneity enabled more cost-effective restoration of N, P, and S through spatially targeted restoration solutions, hence the higher Gini coefficients.

Considering a single ecosystem service in optimization can obscure the potential for convergent restoration of multiple ecosystem services. The optimized solution for a single ecosystem service facilitates the non-optimized restoration of the other three ecosystem services, which we refer to as 'residual' services. The relationship between the optimized service and its residuals depends upon their spatial correlation. N and P retention, which are derived from the same InVEST model, are inherently correlated: optimizing for N retention results in high P residual (Gini = 0.26) (Figure 4c) and optimizing for P retention results in high N residual (Gini = 0.43) (Figure 4d). While S retention is calculated using a different model, this model is run according to similar hydrological processes and relies upon the DEM to identify areas of high S retention, much like the Nutrient Retention model. The spatial correlation between S and the nutrient model is not overwhelmingly strong (Figures 5b and c compared to Figure 5d), but enough of a correlation exists that optimizing S retention creates some small N (Gini = 0.10) and P (Gini = 0.068) residuals (Figure 4d). This same relationship holds true when optimizing N or P retention, as each creates a small residual in S retention (Gini = 0.023, Gini = 0.11, respectively) (Figure 4c,d).

The C sequestration model, as stated previously, did not vary across the landscape and relies solely on land-use classification to determine sequestration potential. Thus it was decoupled from N, P, and S retention, and C optimization depended on least-cost optimization within the agricultural landscape, rather than the combination of least-cost and spatial ecosystem service optimization. The spatial decoupling of the C solutions was demonstrated in all of our tradeoff frontiers: the relationship between optimized C and cost was linear (Gini = 0.055) as was the residual N (Gini = 0.038), P (Gini = 0.055), and S (Gini = 0.054) services (Figure 4a). When we optimized for N, P, or S, C residuals maintained that same linear relationship with cost (Gini = -0.12, Gini = 0.0011, Gini = 0.009, respectively) (Figure 4b,c,d). This suggests that C sequestration can be cost-optimized under the spatial optimization of N, P, and S, as any parcel that is important to a these services is approximately as effective as any other parcel in terms of C sequestration.

Combined, these results form a cohesive picture of ecosystem service potential in grassland restorations. C sequestration is an important conservation goal, but focusing restoration efforts on that alone will reduce the likelihood that a restoration project will promote maximum total ecosystem service benefits. Stakeholders should consider N, P, and S retention, as C will be optimized regardless. Optimizing S on its own will be most cost-effective, but optimizing either N or P will cost-effectively provide greater ecosystem service totals because N and P are more spatially correlated to each other than to S.




4b
Carbon
Sediment
Nitrogen
Phosphorus
Threshold
% Ecosystem Service Potential
1
0.9
0.8
0.7
0.6
0.5
0.4
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0
3
2.5
2
1.5
1
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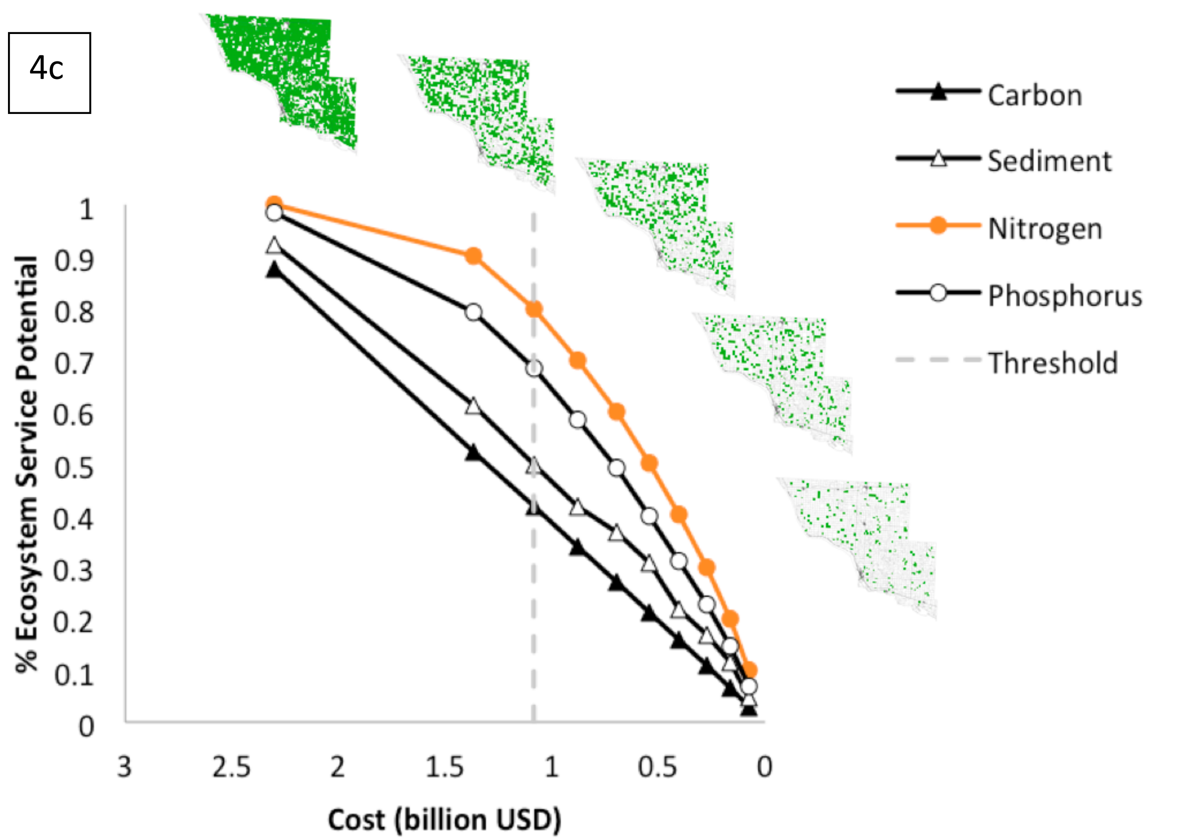
Cost (billion USD)


4c
Carbon
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Cost (billion USD)

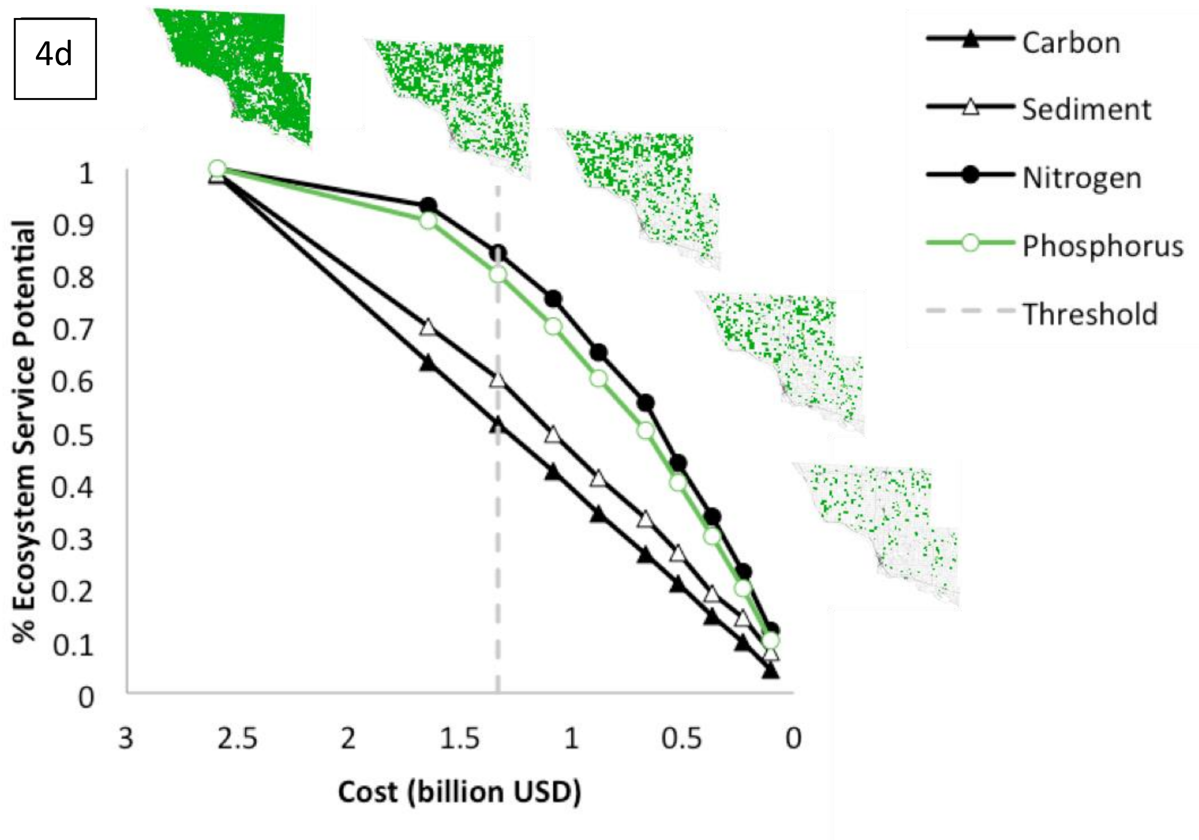

Figure 4. Optimized tradeoff frontiers between the proportion of total landscape potential for each optimized ecosystem service (and its residuals) and the total cost of restoration. Cost is decreasing along the x-axis to visualize the relationship as a tradeoff of 'benefits' via cost reduction. The threshold value at which the cost:benefit ratio exceeds 1 (i.e. higher costs for fewer benefits) is marked vertically as a dashed gray line. Maps of reserve solutions are included, corresponding to every other point along the curve (20%, 40%, 60%, 80%, and 100% of the target ecosystem service optimized on the landscape). These maps represent the reserve solutions for the particular ecosystem service optimized in each map. (a) C optimized by cost (Gini = 0.055), with residual N (Gini = 0.038), P (Gini = 0.054), and S (Gini = 0.055). Threshold value nonexistent. (b) S optimized by cost (Gini = 0.55), with residual C (Gini = 0.009), N (Gini = 0.10), and P (Gini = 0.068). Threshold value at 70% of landscape S potential and a cost of 0.71 billion USD. (c) N optimized by cost (Gini = 0.42) with residual C (Gini = -0.12), P (Gini = 0.26), and S (Gini = 0.023). Threshold value at 80% of landscape N potential and a cost of 1.08 billion USD. (d) P optimized by cost (Gini = 0.37), with residual C (Gini = 0.0011), N (Gini = 0.43), and S (Gini = 0.11). Threshold value at 80% of landscape P potential and a cost of 1.33 billion USD.

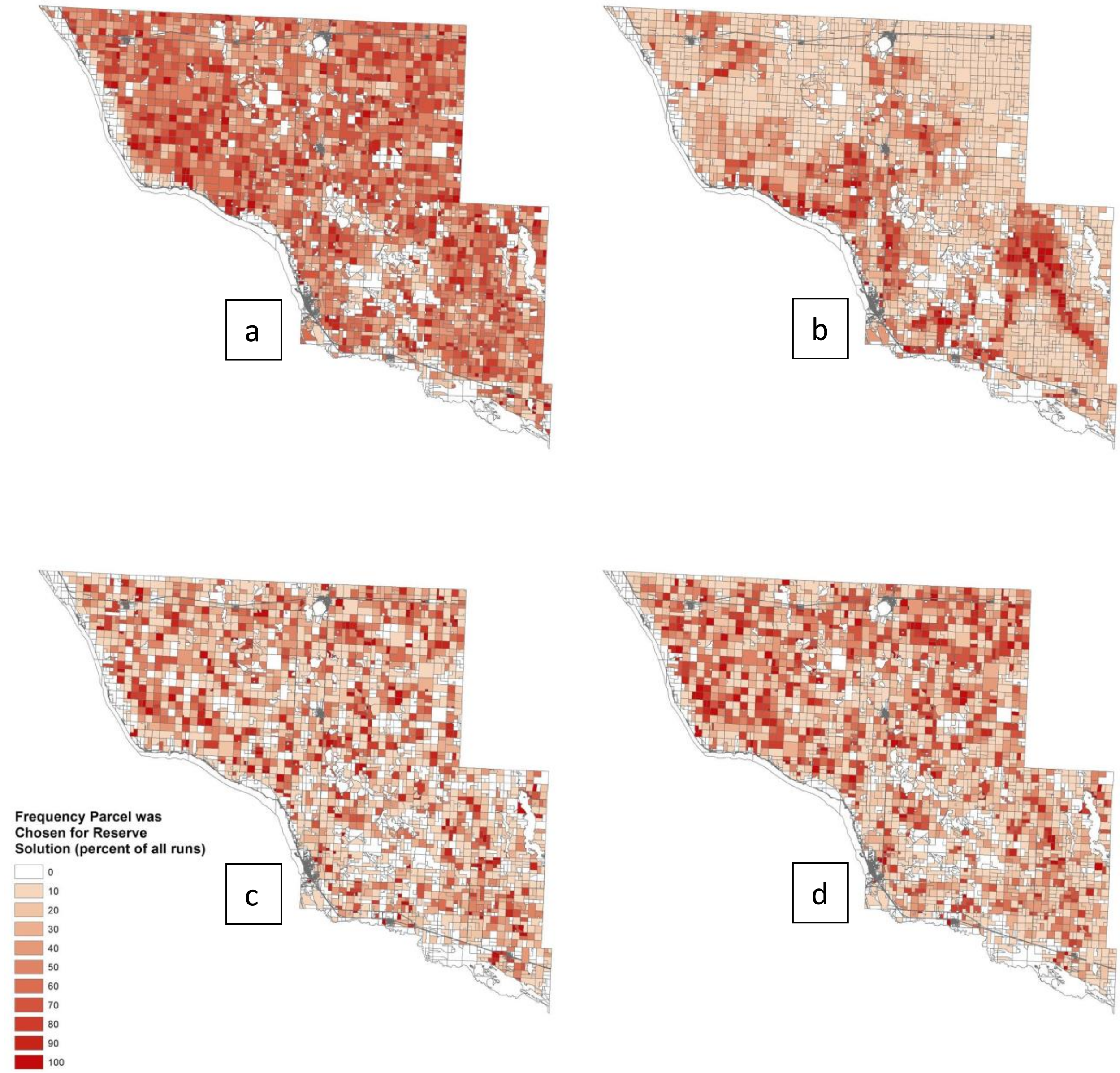

Figure 5. Maps of how frequently individual parcels were chosen when optimizing for a given ecosystem service. Darker red indicates a parcel was chosen in most or all of the ten optimization runs of the given ecosystem service (10%, 20%, 30%, all the way to 100% of landscape ecosystem service potential). Lighter red indicates a parcel was chosen only once or twice, while white indicates the parcel was never chosen. (a)C sequestration. Most of the landscape was chosen in a majority of optimization runs (most parcels are darker red), indicating a lack of spatial patterns in C sequestration. (b)S retention. Most of the landscape was only chosen in one or two optimization runs, but areas close to the Minnesota River along the southern border and in the southeastern part of the state were selected in nearly all runs, indicating high spatial variability and high optimization potential. (c)N retention. Most of the landscape was chosen in one or two optimization runs. Those areas selected in nearly all runs are along waterways, but not spatially aggregated like S retention. (d)P retention. Most of the landscape was chosen in one or two optimization runs, but there is less spatial variability than N retention, as seen by the larger number of parcels selected in a majority of runs.

Sensitivity Analysis

Carbon Sequestration

Our sensitivity analyses identified several parameters (see Table 1 for parameter descriptions) that had influenced our InVEST model outputs. A 'proportionally sensitive' parameter increased output proportional to an increase in input, a 'highly sensitive' parameter increased output more than the corresponding increase in input, and a 'mildly sensitive' parameter increased output less than the corresponding increase in input.

In the C sequestration model the total landscape sequestration potential from converting agriculture to prairie ranged from a loss of 6.5 million tons to a gain of 16 million (Table C2) depending on the values we gave agriculture and grassland carbon pools (Table C1). These values are user-defined via literature review, making them subject to uncertainty and error and thus critical in refining a pertinent model. The total landscape C in each land-use scenario varied proportionally to the input C data (e.g. when we doubled the agricultural C value, the agricultural landscape C sequestration doubled as well) (Figure C1). This parameter was the only parameter subject to variability within the C model.

Nutrient Retention

The Nutrient Retention model had three variable parameters: the loading coefficient, retention efficiency, and threshold flow accumulation (Table C3). The loading coefficient was determined through an extensive literature review (see Table 1 and Appendix A for more information). Increasing the loading coefficient increased nutrient export proportionally (Figure C2), with no marked difference in the spatial distribution of nutrient export, marking it as an important parameter to accurately define, but only a proportionally sensitive one. The retention efficiency parameter was also determined through literature review. It had an inverse effect on the outcome: increasing its value slightly decreased nutrient export, with a marked visual difference in the spatial distribution of nutrient export. It is an important factor with a low landscape sensitivity but high local sensitivity. Although retention efficiency is only mildly sensitive on the landscape scale its value is subject to uncertainty within the literature and must be carefully considered (Figure C2). Threshold flow accumulation (TFA) had an inverse relationship with the model output, with nutrient export slightly decreasing as TFA increased (Figure C2). There was a marked difference in the spatial distribution of nutrient export with different TFA values, indicating low landscape sensitivity but high local sensitivity. It is important to accurately define this value because it must be tailored to each individual study area and DEM, despite its mild sensitivity (Sharp et al. 2015).

Sediment Retention

The SDR model contained several parameters that strongly impacted model results (Table C4). The P and C factors were defined via literature review (see Table 1 and Appendix A). While the P factor increased SDR model output proportionally, the C factor was slightly more sensitive and an increase in input increased caused a disproportionately high increase in output (Figure C3). Both were important factors, but the C factor had the potential to disproportionately affect the results due to its high sensitivity. The effect of the TFA parameter on sediment retention was similar to its effect in the Nutrient Retention model: increasing its input value led to slightly decreasing output values (Figure C3) and a marked difference in the spatial distribution of sediment retention. While still an important parameter because of its specificity to an individual study area and its effect on the distribution of sediment retention, TFA was only mildly sensitive.

The last three parameters, Borselli iC_0, Borselli k_b, and Max SDR, were mildly sensitive, highly sensitive, and proportionately sensitive respectively (Figure C3). These parameters were not defined through literature review but instead set as the default values suggested by the InVEST user guide, which states that are variable only in advanced studies that interact actively with data that is collected in the field (Sharp et al. 2015). The Borselli iC_0 factor is potentially landscape independent (Vigiak et al. 2012), and while the Borselli k_b factor is highly sensitive and affected the spatial distribution of sediment retention significantly, previous studies recommend maintaining the default value unless calibrating the model to actual data (Vigiak et al. 2012; Jamshidi et al. 2013; Sharp et al. 2015). The Max SDR is another parameter only calibrated in studies that rely on comparison to real-world data (Sharp et al. 2015), and so it is not important to adjust in this study despite its proportional relationship with S export. Neither iC_0, k_b, nor Max SDR were important parameters to the model function in our study.

Overall

Our results indicated that most model parameters are not highly sensitive to uncertainty in input data. Because our model looked at the ecosystem services generated in the transition from agriculture to grassland, we were interested less in how a parameter affected the output of a single run than in how a parameter could affect the difference between the outputs for the two scenarios. Some of the parameters were identical in both the agricultural and grassland scenarios (Borselli iC_0, Borselli k_b, Max SDR, and TFA). Other parameters derived from literature reviews differed significantly between the the two (carbon values, nutrient loading coefficients, nutrient retention efficiencies, and C and P factors) (Tables A2, A4, A6). While most of these parameters were mildly to proportionally sensitive, the inconsistencies in the literature reviews from which they were derived may have resulted in inaccuracies in the outputs of our models. Future research should refine the input parameter data through further literature review, and attempt to estimate the probable range of outputs based on the uncertainty in the inputs.

Effects of Surrounding Land Use on Ecosystem Service Output

Our spatial variability analysis showed that a restored parcel's ecosystem service potential is minimally affected by its spatial context, and the degree of sensitivity to surrounding parcels was model-dependent. S retention in restored parcels surrounded by agricultural land decreased by up to 8.5% of those surrounded by grassland, a fairly large uncertainty. N retention fluctuated around the original retention value with a margin of 0.2%, while P retention increased by a maximum of 0.6% (Table C5). These results indicated that the Nutrient Retention model is less dependent on spatial context than the SDR model, and the effect of spatial context can be ignored for N and P retention outputs. Conversely, the S retention restoration solutions that selected only a few parcels to restore to grassland (e.g., the one which restored only 20% of total potential S retention) were subject to a nearly 10% drop in ecosystem service benefits due to spatial uncertainty. At 20% restoration of S retention, this amounted to a total of 408 tons of sediment export to the watershed. Spatial variability not accounted for in our study method could dampen the cost-effectiveness of the SDR model, as ecosystem service potential may be less than indicated at low-level costs.

Discussion

Implications of Our Results

We created tradeoff frontiers to express the relationship between the costs of restoring grassland from agriculture and the ecosystem service benefits achieved through restoration in Big Stone County, MN. These frontiers provide stakeholders with cost estimates for spatially optimized reserve scenarios at several levels of ecosystem service provision. Our novel method of combining InVEST and Marxan models enables landscape-specific analysis of ecosystem service potential to be paired with least-cost selection of land parcels. This method provides a tool for use in future projects to maximize the impact of investment in restoration wherever the potential project area is large enough to enable spatial selectivity.

Our results remain generalizable to the interests of different stakeholders because they do not prescribe particular restoration action. Potential stakeholders (landowners, governments, private organizations) in future restoration action in Big Stone County or in ecologically similar regions may value certain ecosystem services over others, and our tradeoff curves and optimization maps can inform where they might allocate their money. In addition to ecosystem service valuation, future restoration decisions may need to be optimized according to limiting factors such as budget and the unpredictable ‘willingness to pay’ among family farmers, for whom the cost of land conversion may exceed what is predicted in our cost-benefit calculations. Even though costs and valuation of benefits are likely to vary, our results demonstrate the usefulness of spatially optimizing selection of parcels for restoration. The Gini coefficients of the curves which maximize provision of benefits (nutrient export reduction or sediment export reduction) show that our solutions provide benefits far beyond what a spatially random solution would provide. We offer InVEST and Marxan in combination as effective tools for identifying spatially optimized restoration solutions both in Big Stone County and in similar scenarios.

Spatial Independence of Carbon Sequestration

Our results indicate that certain ecosystem services, when prioritized in a reserve solution, generate a greater proportion of ‘residual’ benefits than others. Our tradeoff frontiers show that, if a stakeholder is mainly interested is carbon sequestration, optimizing for any of the four ecosystem services (C, S, N, or P) would produce a similar, linear relationship between cost of restoration and carbon sequestered. As C sequestration potential does not vary across the landscape, its optimization is influenced only by variation in agricultural opportunity cost, which in Big Stone County has limited variability (Figure D5b). The optimization of S, N, or P has the potential to provide more total benefit to the stakeholder than C optimization because the residual benefits of these are higher and include nearly as much carbon sequestration for a given cost as would optimizing C itself.

Assumptions in the InVEST Models

C sequestration is one of the best-studied ecosystem services, but predicting it still depends on assumptions about certain ecological factors. Our C Sequestration model assumes that a restored prairie has 100 years to grow. This is a time frame supported by other research that has used the InVEST model (Polasky et al. 2010; Nelson et al. 2009). Future research could refine this assumption by observing the progression of carbon storage over time in prairies near Big Stone County or under similar conditions (e.g., taking into account the duration of time previously in agriculture or surrounding landscape characteristics). Time since cessation of agriculture has been shown in other studies to correlate positively with the size of soil carbon pools (McLauchlan 2006), but the importance of local soil conditions to carbon storage potential is not yet clear (McLauchlan 2006; Baer et al. 2005; Bach et al. 2010).

The Nutrient Retention and SDR models rely on the hydrological characteristics of a landscape and information on land-use types in order to produce accurate results because they are fundamentally watershed-scale models (Sharp et al. 2015). The DEM and other spatial layers we used to describe hydrology (precipitation, PET, PAWC, erosivity, erodibility) originated from well-established government sources (Table A1), but our data on characteristics of individual land uses could be further refined. We aggregated our land-use classes into agriculture, prairie, wetland, forest, and barren/other, both for simplicity and because of the difficulty of finding data biophysical constants in the literature describing regions near Minnesota even for these land-use classes (Tables A4 and A6). More precise data on the effect of land uses and land use patterns on runoff and retention would enable more accurate assessment for a reserve solution.

Effects of Adjacent Land Uses on Ecosystem Services

One of this study's assumptions was that the benefits provided through converting a parcel of agricultural land to grassland would be the same whether that parcel was surrounded by a landscape of all grassland parcels or one that included some agricultural parcels. We knew that the spatial distribution of restored land affects the provisioning of watershed-based services (N, P, and S retention) (Rabotyagov et al. 2014; Kovacs et al. 2013; Poudel et al. 2013), but we ran our Nutrient Retention and SDR models as if the surrounding landscape would not influence the amount of ecosystem service provided by an individual parcel. Our Marxan-optimized solutions were based on the changes measured between strictly-agricultural and strictly-prairie landscapes.

When we tested this assumption by running InVEST on landscapes where only one parcel was restored to prairie, we found that there was little variation in the amount of N or P retention services provided (<1%). S retention, however, was reduced by up to almost 10% in the test landscape compared to the strictly-prairie landscape (Table C5). Although this issue diminishes solutions where the ecosystem service target is high enough to promote grassland parcel connectivity, these results indicate that we must temper expectations for the S retention model's apparently efficient restoration solutions (Figure 4b). For the N and P models this test demonstrates that our initial results are highly reliable in regards to spatial context.

Sensitivity Analyses and Implications for Future Research on InVEST Parameters

When an InVEST input parameter that is disputed in the literature is also one to which a model is proportionally or highly sensitive, there is a potential for error. The models are highly sensitive to the C factor, proportionally sensitive to carbon storage values, nutrient export coefficients, and P factor. All of these parameters are variable within the literature. Field research in or near Big Stone County could improve the accuracy of these values and be an important investment in improving a reserve solution.

Values for belowground and soil carbon storage were based on a broad literature review of carbon pools in and around Minnesota (Table A2), but there was high variability in the data, with standard deviations ranging from 30-80% of the reported value. This variability suggests that carbon pools are not uniform across the Midwestern prairie or agricultural landscapes. To increase the accuracy of our results, more site-specific studies need to be undertaken.

The Nutrient Retention analysis could be improved by obtaining more detailed information on conditions of nitrogen export on Minnesota landscapes. Nitrogen export values (load_N and eff_N) have been reported significantly less in the literature than phosphorous values, and we were required us to translate nitrogen input values from areas as far away as Maryland into our model (Table A4). Retention efficiency values (eff_N) have limited impact on model results, but the model is proportionately sensitive to the export coefficient (load_N) (Figure C2). Research into regionally-specific nitrogen loads for both agricultural uses and restored prairie would be especially beneficial to stakeholders with an interest in limiting nutrient runoff.

The responsiveness of the SDR model to C and P factors puts a premium on site-specific values. We employed commonly accepted C factors for agriculture and grassland (Table A6), but Big Stone-specific C values would take into account local variables such as crop canopy, incorporated residues, residue mulch, and tillage. The P factor is much less variable across agricultural regions (Michigan State University 2002). Because the SDR model is highly sensitive to C and P factor input (Figure C3), the accuracy of these variables is important to valuing the sediment retention capacity of parcels. Research into the locally variable C factor could especially improve model accuracy.

The threshold flow accumulation is not a literature-derived parameter, but rather a calibration coefficient used to match the hydrology of the model to the hydrology of the region. We calibrated this parameter using satellite imagery because there is no existing verified stream data for Big Stone County (Tables A3 and A5). Although our models are only mildly sensitive to threshold flow accumulation, acquiring a stream layer based on a waterway map generated from field observation would assist in calibrating the value and mitigate possible user error.

Understanding Cattle Operations

There is some uncertainty in our data on beef cattle that contributes to the inability of our results to precisely reflect the economic cost of restoring agriculture to prairie. We included cattle in our study because they complement restoration goals and their grazing behavior is essential to long-term grassland management; in addition, we predicted that the profit from beef production could incentivize restoration (Fuhlendorf et al. 2010; Chaplin and Van Vleck 2014). However, incorporating cattle costs and benefits into our analysis required making several assumptions. Cost per head of cattle, benefit per head of cattle, and amount of cattle per acre are values that each vary widely from farm to farm. In order to determine how many cattle should be on a given acre, one must know the length of their grazing season, the average weight of cattle on the farm, the average yield of their pasture, and the daily utilization rate for livestock (what percent of its weight each animal needs in forage each day), all of which vary based on cattle breed (Tony Lourey, Personal Communication, February 3, 2016). In order for our model to more accurately reflect the influence of ecologically sustainable cattle ranching on prairie restoration, data on cattle operations specific to Big Stone County would be useful.

Expanding Analysis of Restoration Benefits

Future studies may increase the value of reserve solutions by incorporating additional ecosystem services in their analyses. We analyzed only carbon, nutrient, and sediment conditions on the landscape because these services have been observed to vary greatly between agricultural and prairie land and are relatively simple to model based on land use data (Polasky et al. 2010; Nelson et al. 2010). However, considering the impacts of restoration on other services such as biodiversity preservation and crop pollination, which are addressed in InVEST's Habitat Quality and Crop Pollination models, we could further refine the process of maximizing the total ecosystem services provided by a reserve solution.

Marxan has optional advanced settings which could be employed to address valuation of biodiversity improvements through restoration. One such factor is 'Minimum Clump Size' which specifies a minimum size for a group of parcels which provide particular ecosystem service or conservation feature. If the amount of a conservation feature found in a clump is smaller than this minimum value, then it does not advance the conservation target. This factor would be useful in future research that places higher conservation value on larger patches of prairie that enable habitat connectivity for the purpose of biodiversity conservation (Mitchell et al. 2013) and lower value on small or isolated patches. Another factor, 'Separation Distance,' specifies the minimum distance at which planning units are considered separate. This factor may be useful in research which is concerned with the dispersal capacity of invasive species.

It is not difficult to convert InVEST outputs to Marxan inputs, and incorporating additional InVEST analyses could improve the ability of our method to optimize restoration for cost-effectiveness. Through further research, it would ultimately be possible to provide stakeholders with tradeoff frontiers for a suite of ecosystem services and more fully inform their decision-making on restoration actions.

Conclusion

Our study found that more ecosystem services are maximized when a reserve solution is spatially optimized for nutrient or sediment retention. Each of our four ecosystem services had a distinct spatial pattern on the landscape of Big Stone County. Carbon sequestration did not vary over the landscape. Nitrogen and phosphorus retention were spatially correlated, but dispersed across the landscape. Sediment retention was aggregated in a few key areas of the County, and was moderately correlated with nitrogen and phosphorus retention. Because carbon sequestration did not vary spatially, it was implicitly optimized in all solutions. Therefore, optimizing grassland restoration for nutrient or sediment retention will also optimize carbon, leading to higher overall benefits. Ultimately, stakeholders in grassland restoration decide which ecosystem services to prioritize, but our results indicate that it is essential to apply spatial optimization to determine which areas should be targeted for cost-effective restoration.

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Appendix A: InVEST inputs and sources

Table A1. Spatial Data Layers

Layer	Model			Source
	Carbon	Nutrient	SDR	
Current LULC	✓	✓	✓	USDA-NASS CDL 2014.
Grassland LULC	✓	✓	✓	Minnesota DNR 2015.
Root restricting layer depth		✓		Soil Survey Staff 2015.
Precipitation		✓		PRISM Climate Group 2004.
PAWC		✓		Soil Survey Staff 2015.
PET		✓		Soil Survey Staff 2015.
DEM		✓	✓	U.S. Geological Survey 2015.
Watersheds		✓	✓	Minnesota DNR 2014.
Erosivity			✓	NOAA Office for Coastal Management 2013.
Erodibility			✓	Soil Survey Staff 2015.

Table A2. Carbon Sequestration biophysical table.

LULC	Aboveground Carbon (tons/ha)	Belowground Carbon (tons/ha)	Soil Carbon (tons/ha)	Dead Carbon (tons/ha)
Agriculture	0[1]	2.797142857[2]	58.95131579[3]	0
Grassland	0[1]	15.78111111[4]	79.23642857[5]	0

1. Sharp et al. 2015.
2. Rees et al. 2005; Qiu and Turner 2013; Polasky et al. 2011; Slobodian et al. 2002.
3. Qiu and Turner 2013; Tol 2011; Al-Kaisi et al. 2005; Kaye et al. 2005; David et al. 2009; Pérez-Suárez et al. 2014; Slobodian et al. 2002.
4. Johnston 1996; Qiu and Turner 2013; Verburg et al. 2004; Frank et al. 2004; Slobodian et al. 2002.
5. McLauchlan 2006; David et al. 2009; Kaye et al. 2005; Verburg et al. 2004; Frank et al. 1995; Slobodian et al. 2002.

Table A3. Nutrient Retention model parameters.

Parameter	Value	Source
Threshold Flow Accumulation	500	Sharp et al. 2015 and visual verification

Table A4. Nutrient Retention biophysical table.

LULC	Kc	Root Depth (mm)	Nitrogen		Phosphorus	
			Export Coefficient	Retention Efficiency	Export Coefficient	Retention Efficiency
Other	0[1]	1[1]	0	0	0	0
Agriculture	1.15[2]	2000[5]	7.42[6]	0.77[9]	0.54[11]	0.15[12]
Grassland	1.5[2]	3000[5]	0.151[6]	0.8[10]	0.169[11]	0.45[13]
Wetland	1.3[3]	3000[5]	0.55[7]	0.8[10]	0.01[7]	0.8[14]
Forest	0.75[4]	3000[5]	2.5[8]	0.95[9]	0.2[8]	0.6[1]

1. Kovacs et al. 2013.
2. Allen et al. 1998b.
3. Allen 1998a.
4. Bandaragoda et al. 2003.
5. Meehan et al. 2013
6. MNPCA 2013.
7. Cadmus Group 1988.
8. USEPA 1976.
9. Keller et al. 2015.
10. Sharp et al. 2015.
11. Mulla et al. 2004.
12. Johnson et al. 2012.
13. Johnson et al. 2012; Kovacs et al. 2013.
14. Kovacs et al. 2013; Meehan et al. 2013.

Table A5. Sediment Delivery Ratio model parameters.

Parameter	Value	Source
Threshold Flow Accumulation	500	Sharp et al. 2015 and visual verification
Max SDR	0.8	Sharp et al. 2015
Borselli Kb	2	Sharp et al. 2015
Borselli iC_0	0.5	Sharp et al. 2015

Table A6. Sediment Delivery Ratio biophysical table

LULC	Cover-Management (C) Factor	Support Practice (P) Factor
Other	0	0
Agriculture	0.155[1]	1[2]
Grassland	0.045[1]	1[3]
Wetland	0.125[3]	1[3]
Forest	0.0044[3]	1[3]

1. Kuehner 2001.
2. Sharp et al. 2015.
3. Yang et al. 2003.

Appendix B: Marxan inputs and sources

Table B1. Economic cost inputs to the Marxan model.

	Cost	Units	Source
Restoration Cost Net Present Value (NPV)	0.35212486	$/m^2	Minnesota Prairie Plan Working Group 2011 (Table 37).
Cattle Cost NPV	0.2769803	$/m^2	USDA-NASS 2010-2011.
Cattle Benefit NPV	-0.1364448	$/m^2	Hancock 2006.
Crop NPV - Corn	2418	$/30m^2	USDA-NASS 2010-2014a.
Crop NPV - Soybeans	1838	$/30m^2	USDA-NASS 2010-2014a.
Crop NPV - Alfalfa	1449	$/30m^2	USDA-NASS 2010-2014b.
Crop NPV - Other Hay/Non-Alfalfa	1015	$/30m^2	USDA-NASS 2010-2014c.
Crop NPV - Sugarbeets	3164	$/30m^2	USDA-NASS 2005 and 2007.
Crop NPV - Spring Wheat	1209	$/30m^2	USDA-NASS 2010-2014a.
Crop NPV - Other	890	$/30m^2	USDA-NASS 2010-2014a; USDA-NASS 2009-2010.

Appendix C: Sensitivity analysis and spatial assumption test

Table C1. Carbon Sequestration sensitivity analysis parameters

Parameter (units)	Input Value		
	Minimum	**Average**	**Maximum**
Agriculture Carbon (tons C/ha)	20.81	61.74845865	104.3
Grassland Carbon (tons C/ha)	31.72	95.01753968	198.35

Table C2. Carbon Sequestration sensitivity analysis outputs. The InVEST outputs of landscape C sequestration potential, comparing the interaction between different levels of the C input parameters. For instance, the potential sequestration calculated if we used the minimum literature value for Grassland Carbon and the maximum literature value for Agriculture Carbon was -6,540,215.10 tons (intersection of **Grassland Min** and **Agriculture Max** scenarios).

	Landscape Carbon Sequestration Potential (tons)		
Scenarios	Grassland Min	Grassland Avg	Grassland Max
Agriculture Min	983,104.77	6,686,873.25	15,998,205.27
Agriculture Avg	-2,705,877.18	2,997,891.30	12,309,223.32
Agriculture Max	-6,540,215.10	-836,446.62	8,474,885.40

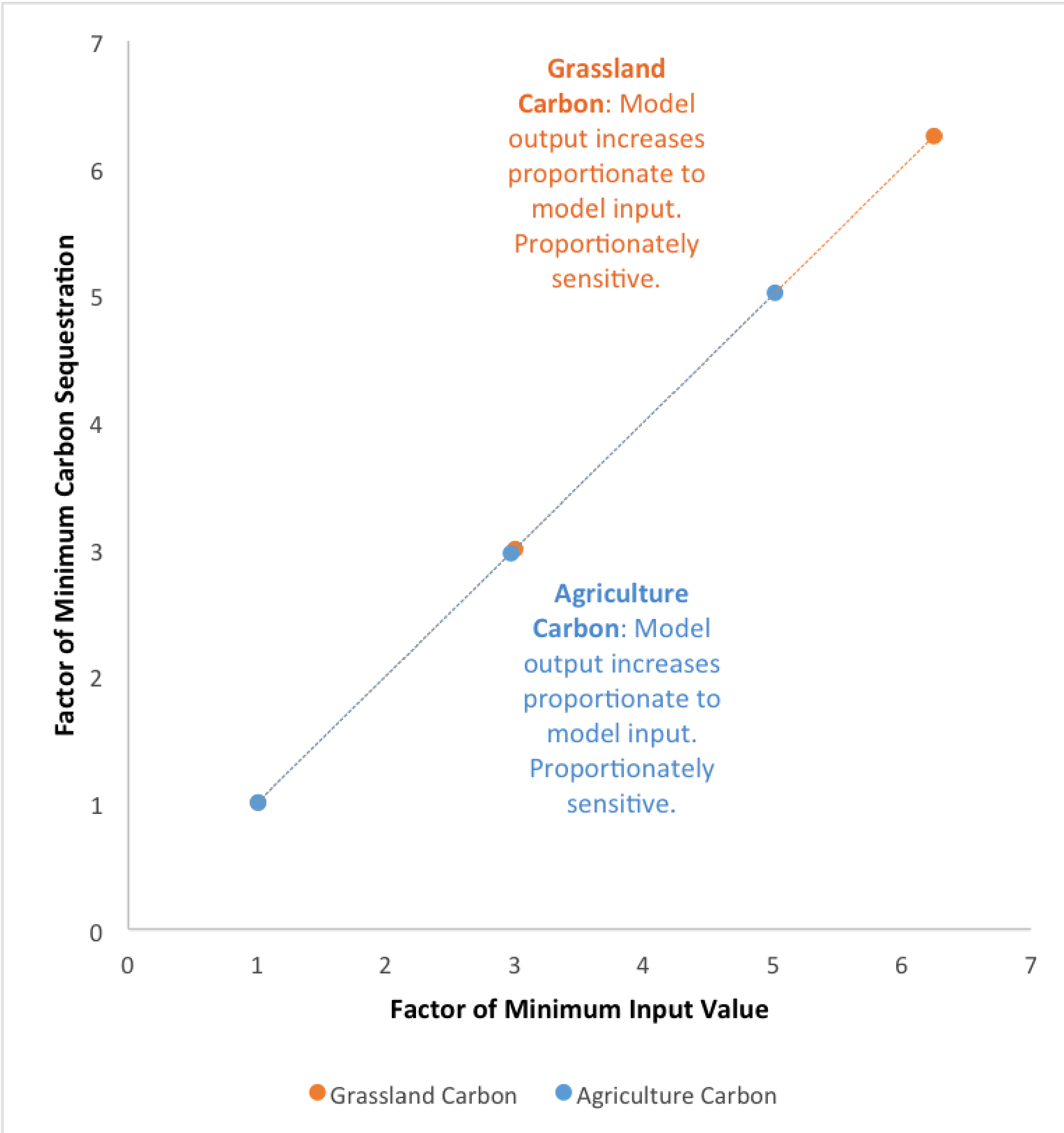


Figure C1. Carbon Sequestration sensitivity analysis. A comparison of the relationship between a Carbon model input parameter and the model output. Each input value (Minimum, Average, and Maximum) for each parameter was divided by the Minimum input value to convert variable inputs into a simpler Factor of Minimum Input Value scale. Model outputs were similarly converted to a factor of the minimum output of each input parameter. These were graphed to see if the relationship between input parameter and model output was proportional or not. Both the Grassland Carbon and Agricultural Carbon inputs were tested, and each had a proportional effect on model outputs. That is, if the input was doubled, the output was doubled as well (proportionally sensitive).

Table C3. Nutrient Retention sensitivity analysis parameters

Parameter (units)	Input Value		
	Minimum	**Average**	**Maximum**
Loading Coefficient	0.28	0.54	0.95
Retention Efficiency	0.05	0.2	0.4
Threshold Flow Accumulation (pixels)	500	1000	2000

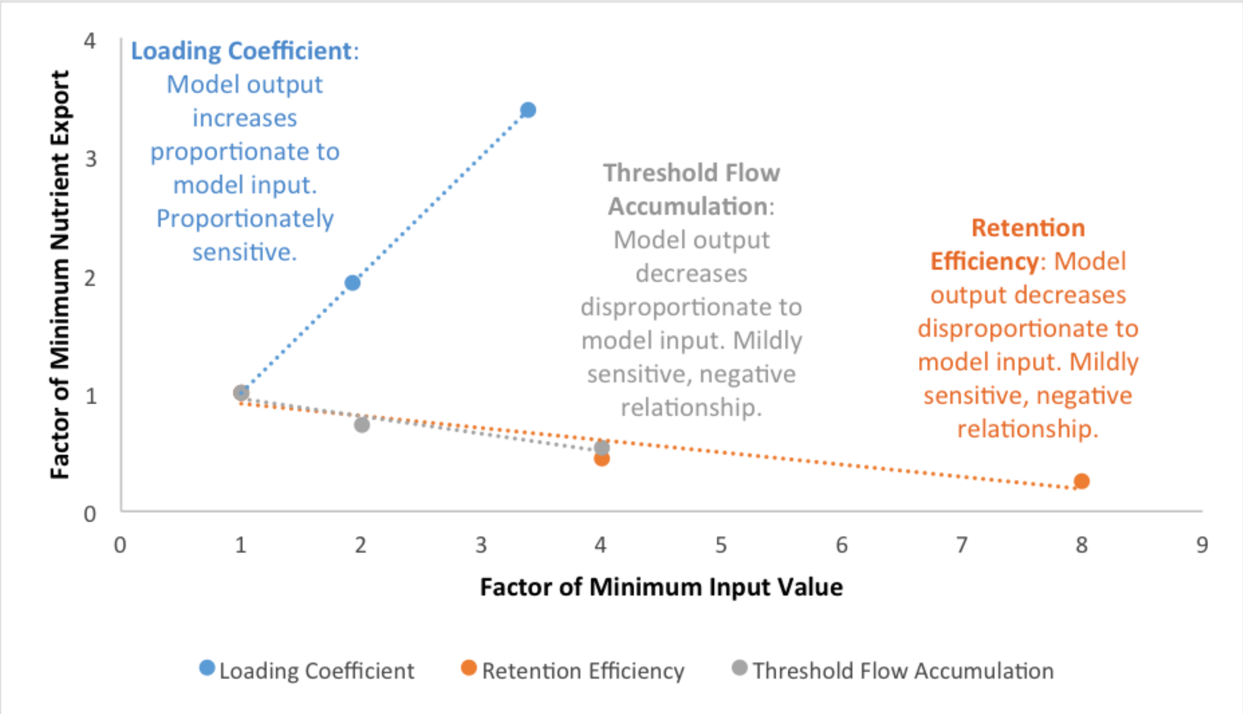


Figure C2. Nutrient Retention sensitivity analysis. A comparison of the relationship between a Nutrient Retention model input parameter and the model output. Each input value (Minimum, Average, and Maximum) for each parameter was divided by the Minimum input value to convert variable inputs into a simpler Factor of Minimum Input Value scale. Model outputs were similarly converted to a factor of the minimum output of each input parameter. These were graphed to see if the relationship between input parameter and model output was proportional or not. The loading coefficient was proportionally sensitive (the output doubled if the input doubled), while both the retention efficiency and threshold flow accumulation factors were mildly sensitive (the doubling the input did not double the output, which was less than doubled), with a negative effect on output.

Table C4. Sediment Delivery Ratio sensitivity analysis parameters

Parameter (units)	**Input Value**		
	Minimum	**Average**	**Maximum**
P Factor	0.28	0.54	0.95
C Factor	0.05	0.2	0.4
Borselli iC_0	0.25	0.5	1
Borselli k_b	1	2	4
Max SDR	0.4	0.8	1
Threshold Flow Accumulation (pixels)	500	1000	2000

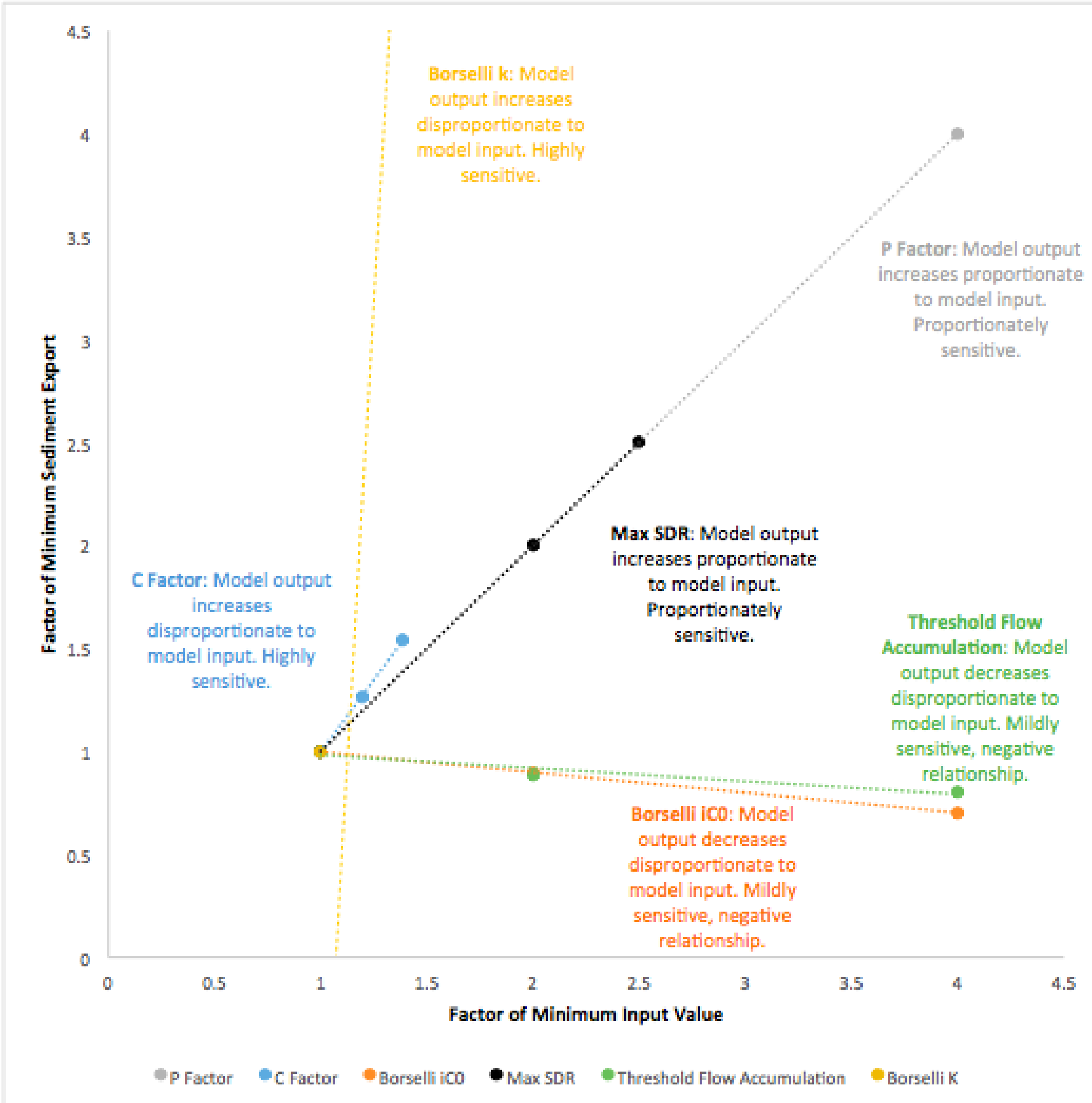
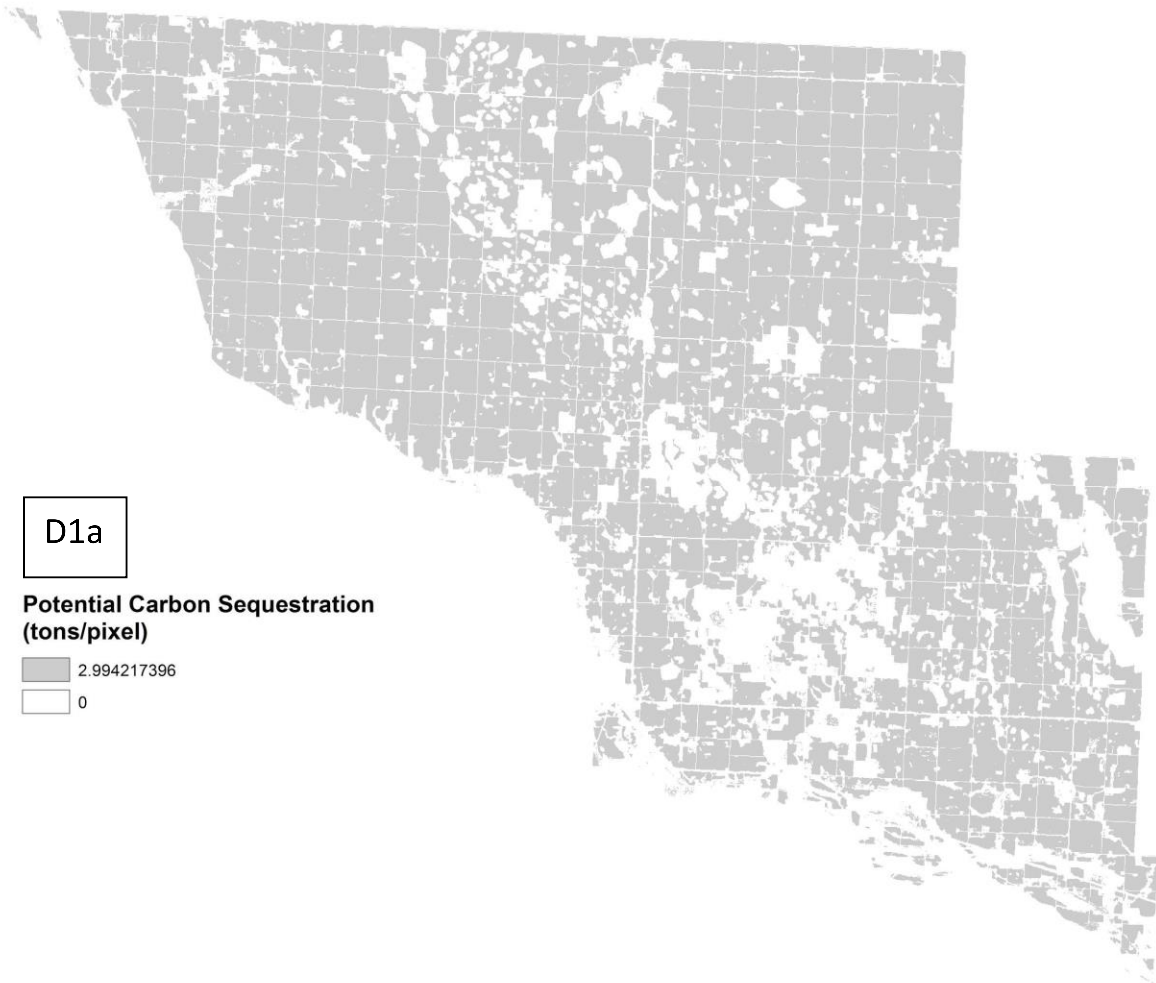


Figure C3. Sediment Delivery Ratio sensitivity analysis. A comparison of the relationship between a SDR model input parameter and the model output. Each input value (Minimum, Average, and Maximum) for each parameter was divided by the Minimum input value to convert variable inputs into a simpler Factor of Minimum Input Value scale. Model outputs were similarly converted to a factor of the minimum output of each input parameter. These were graphed to see if the relationship between input parameter and model output was proportional or not. The Max SDR and P Factor were proportional sensitive, while the Borselli iC_0 and threshold flow accumulation factors were mildly sensitive and reduced output. The C factor was highly sensitive (doubling the input more than doubled the output, slightly), and the Borselli k_b factor was highly sensitive (doubling the input more than doubled the output, extremely).

Table C5. Changes in ecosystem service potential from surrounding land use context. Ecosystem service differences between a grassland parcel surrounded by grassland and one surrounded by agriculture. Indicates that S retention may be overestimated in our model by up to 10%, while the effect on others is negligible.

Parcel	Difference (tons)			Difference (%)		
	Sediment	Nitrogen	Phosphorus	Sediment	Nitrogen	Phosphorus
1	-0.9	0.02	0.02	-3.8	0.1	0.6
2	-1.8	0.02	0.01	-4.3	0.1	0.3
3	-1.0	0.02	0.03	-2.2	0.06	0.7
4	-1.7	-0.02	0.003	-8.5	-0.2	0.2

Appendix D: Supplementary figures



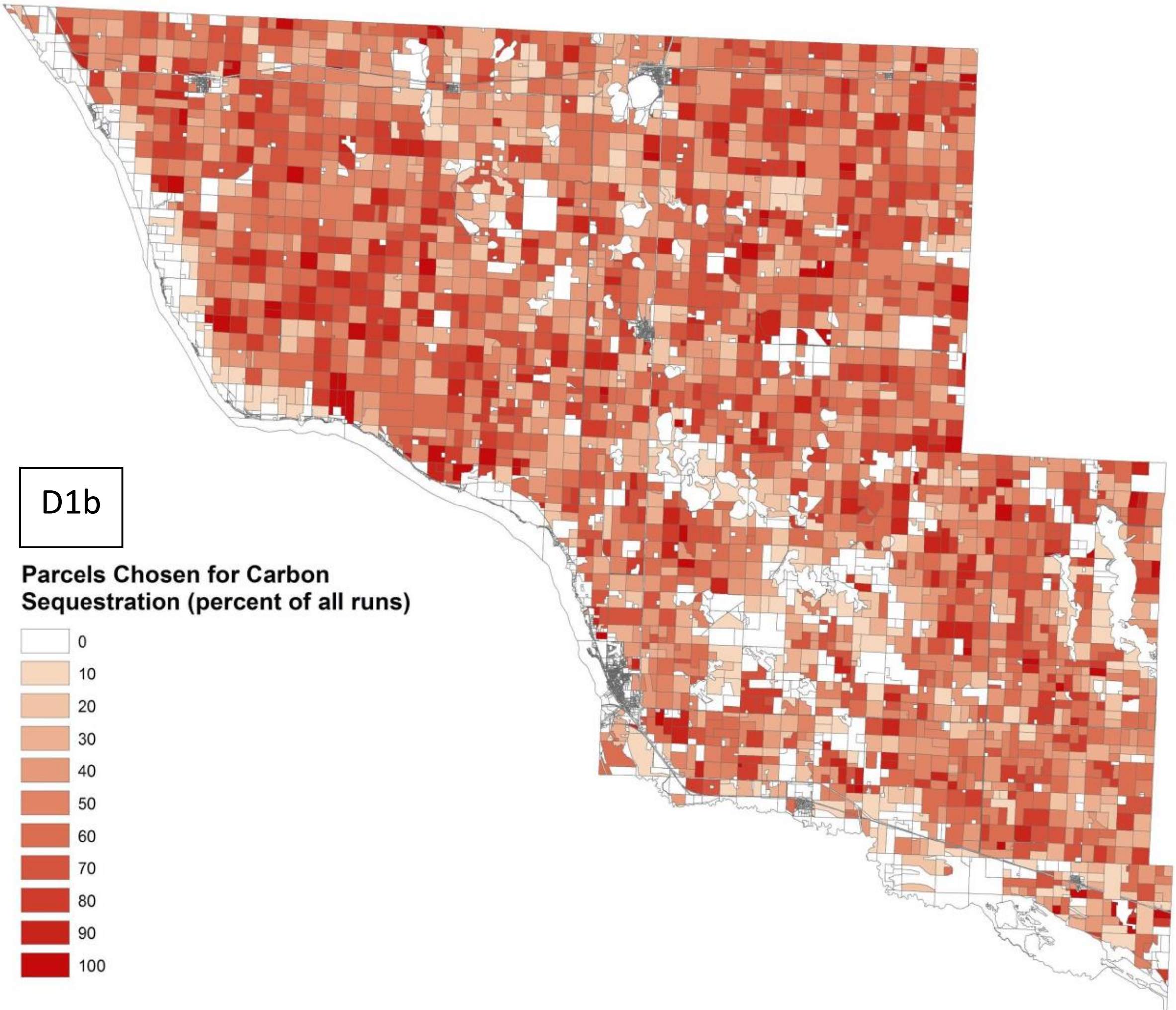
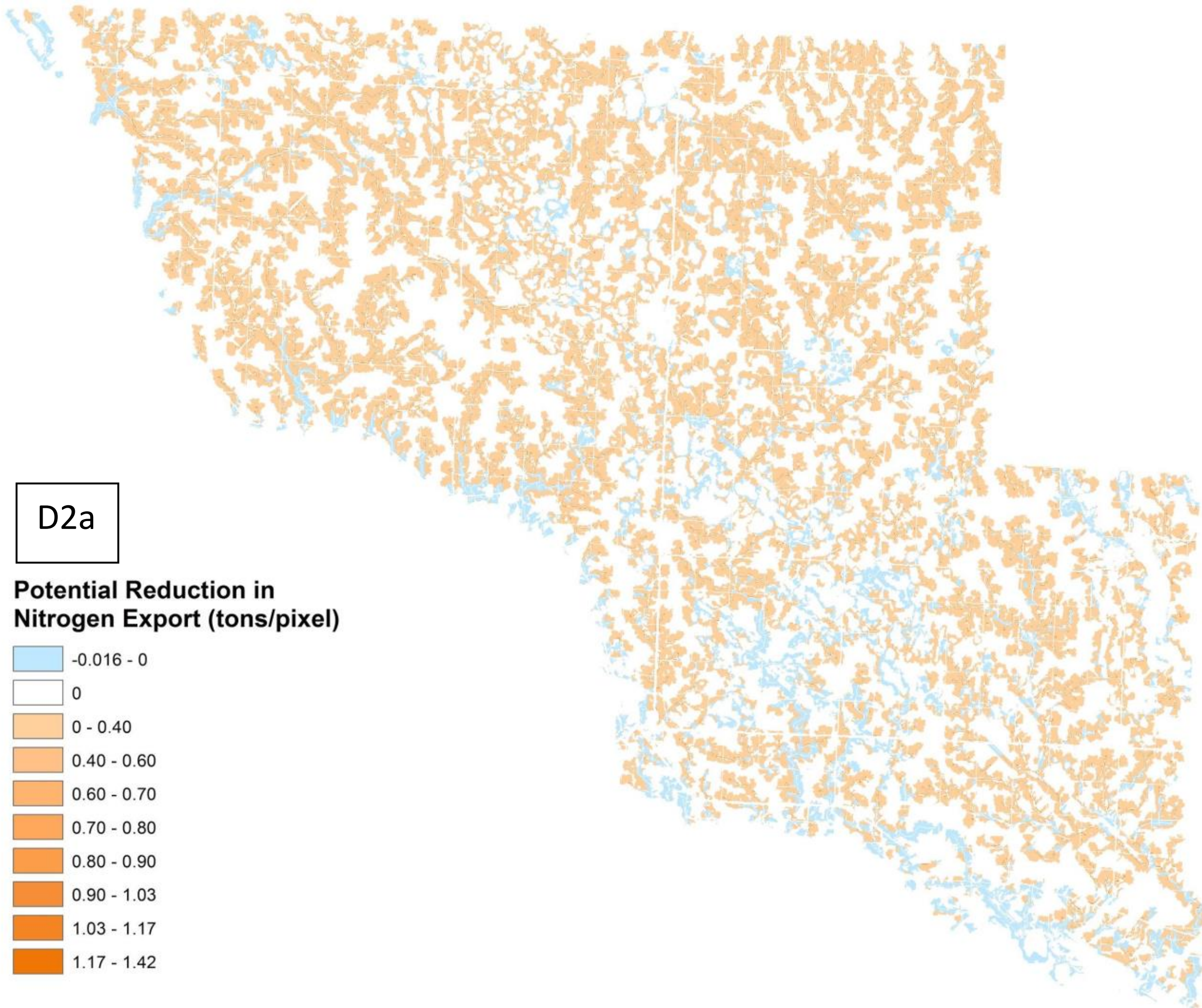


Figure D1. Carbon Sequestration Potential. (a)Map of C sequestration potential (tons) per pixel for the transition between agricultural land and grassland. There is no spatial variety across the landscape: either agricultural land can be restored, which provides the same potential per pixel, or the land is not in agriculture and offers no restoration potential. (b)Map of land parcels chosen for restoration, optimizing C sequestration. This combined all ten optimization runs, from 10% of potential to full restoration, and reports the frequency each parcel was chosen in the best restoration solution. There is no discernable spatial pattern, as parcels are chosen in a more scattered way to reflect the spatially homogenous nature of C sequestration.


D2a
Potential Reduction in Nitrogen Export (tons/pixel)
-0.016 - 0
0
0 - 0.40
0.40 - 0.60
0.60 - 0.70
0.70 - 0.80
0.80 - 0.90
0.90 - 1.03
1.03 - 1.17
1.17 - 1.42

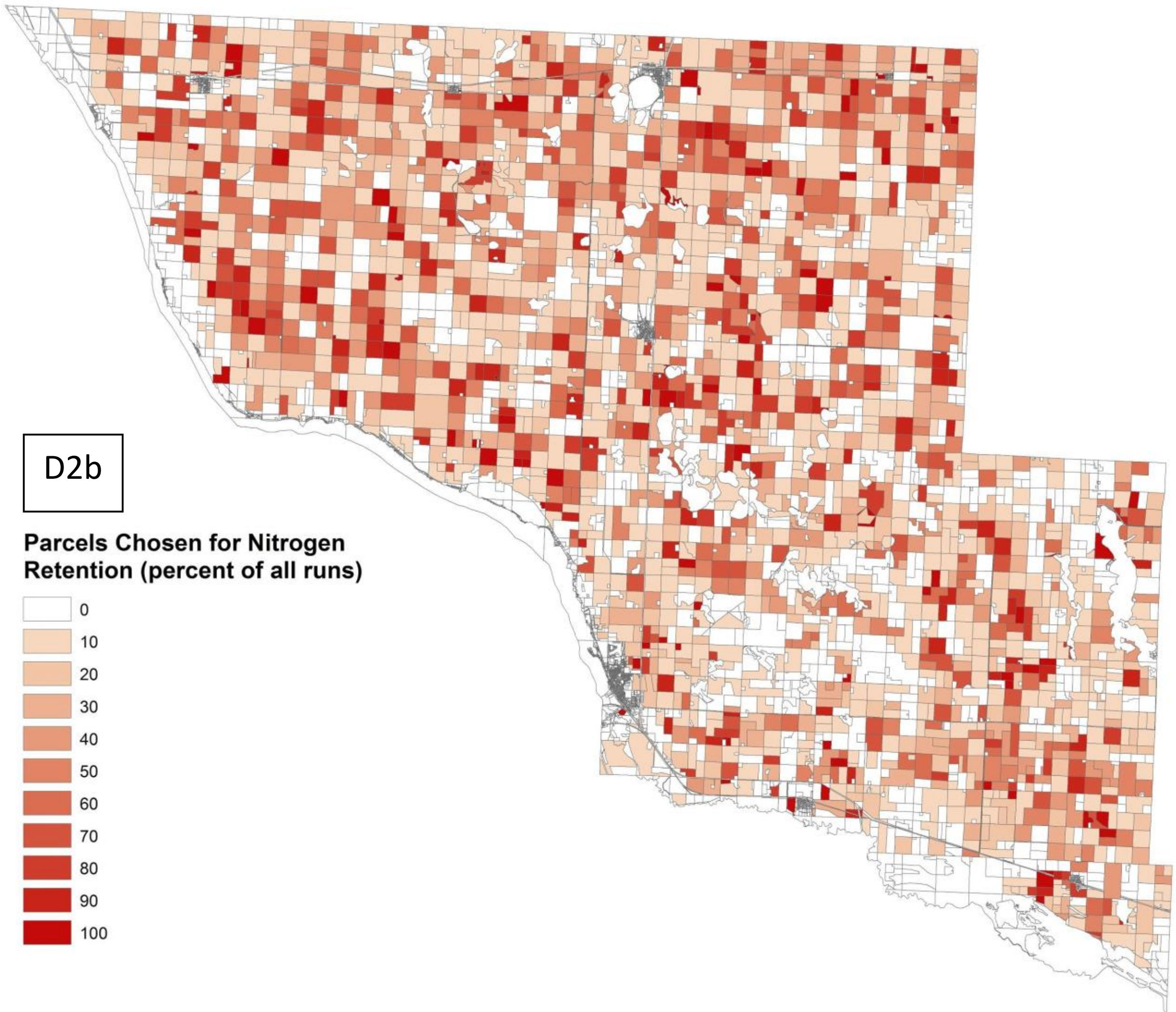
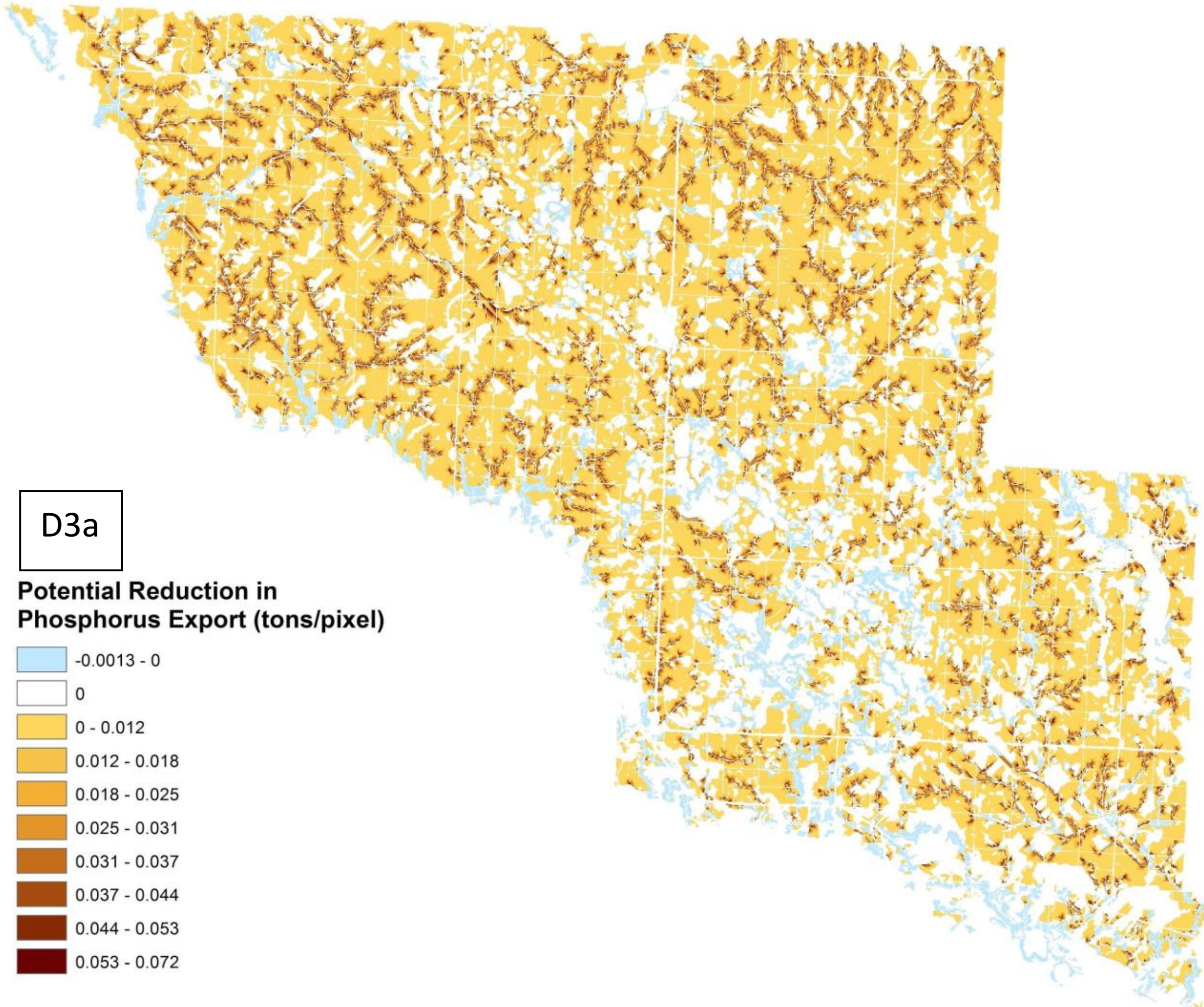


Figure D2. Nitrogen Retention Potential. (a)Map of N retention potential (tons) per pixel for the transition between agricultural land and grassland. N retention potential is centered along waterways within agricultural lands, with the highest potentials directly adjacent to the water. The negative values are located exclusively on non-agricultural land, reflecting very marginal changes in the N retention capability of surrounding ecosystems (e.g. Forests and Wetlands) as restoration reduces overall nutrient load. (b)Map of land parcels chosen for restoration, optimizing N retention. This combined all ten optimization runs, from 10% of potential to full restoration, and reports the frequency each parcel was chosen in the best restoration solution. There is no explicitly visual spatial pattern, although there are several specific parcels that are chosen in nearly all optimization runs. This indicates that optimizing N retention focuses on parcels containing or adjacent to active waterways.


D3a
Potential Reduction in
Phosphorus Export (tons/pixel)
-0.0013 - 0
0
0 - 0.012
0.012 - 0.018
0.018 - 0.025
0.025 - 0.031
0.031 - 0.037
0.037 - 0.044
0.044 - 0.053
0.053 - 0.072

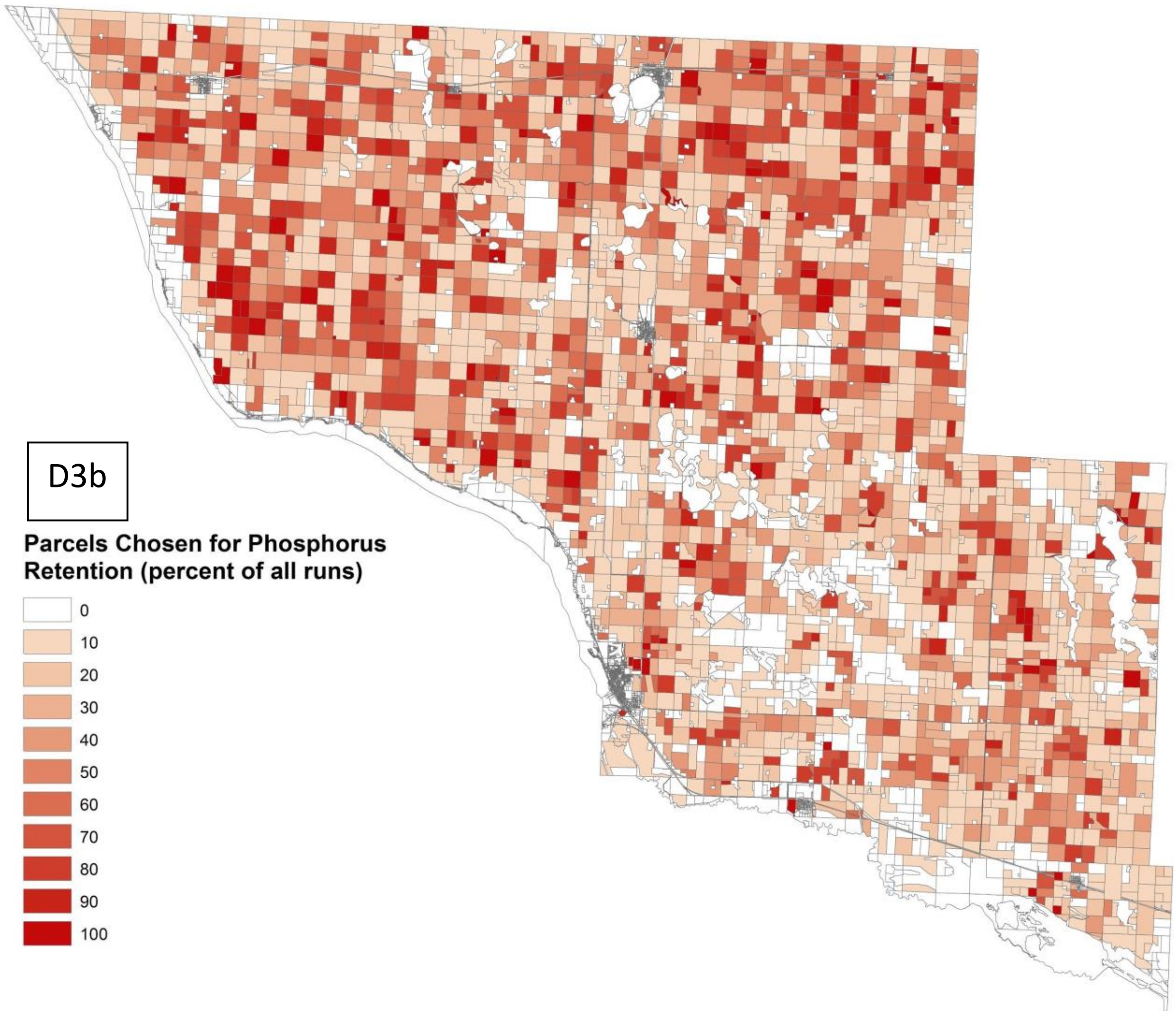
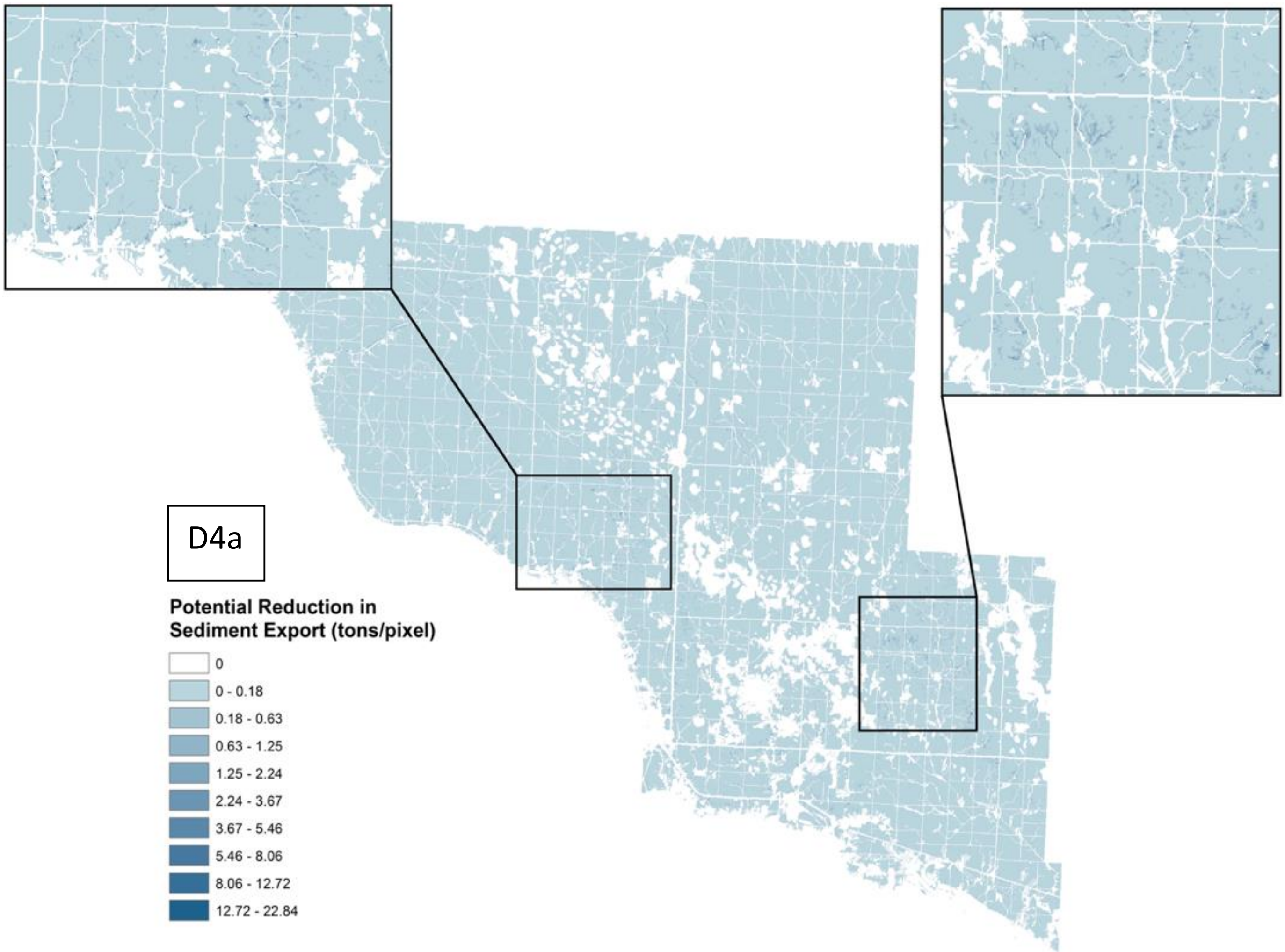


Figure D3. Phosphorus Retention Potential. (a)Map of P retention potential (tons) per pixel for the transition between agricultural land and grassland. P retention potential is centered along waterways within agricultural lands, with the highest potentials directly adjacent to the water; this effect is more pronounced than with N retention, and there is more area suitable for P retention than N retention. The negative values are located exclusively on non-agricultural land, reflecting very marginal changes in the P retention capability of surrounding ecosystems (e.g. Forests and Wetlands) as restoration reduces overall nutrient load. (b)Map of land parcels chosen for restoration, optimizing N retention. This combined all ten optimization runs, from 10% of potential to full restoration, and reports the frequency each parcel was chosen in the best restoration solution. There is no explicitly visual spatial pattern, although there are several specific parcels that are chosen in nearly all optimization runs. However, this effect is less pronounced than in N retention, indicating that optimizing P retention also focuses on parcels containing or adjacent to active waterways, but less so than optimizing N retention.


D4a
Potential Reduction in Sediment Export (tons/pixel)
0
0 - 0.18
0.18 - 0.63
0.63 - 1.25
1.25 - 2.24
2.24 - 3.67
3.67 - 5.46
5.46 - 8.06
8.06 - 12.72
12.72 - 22.84

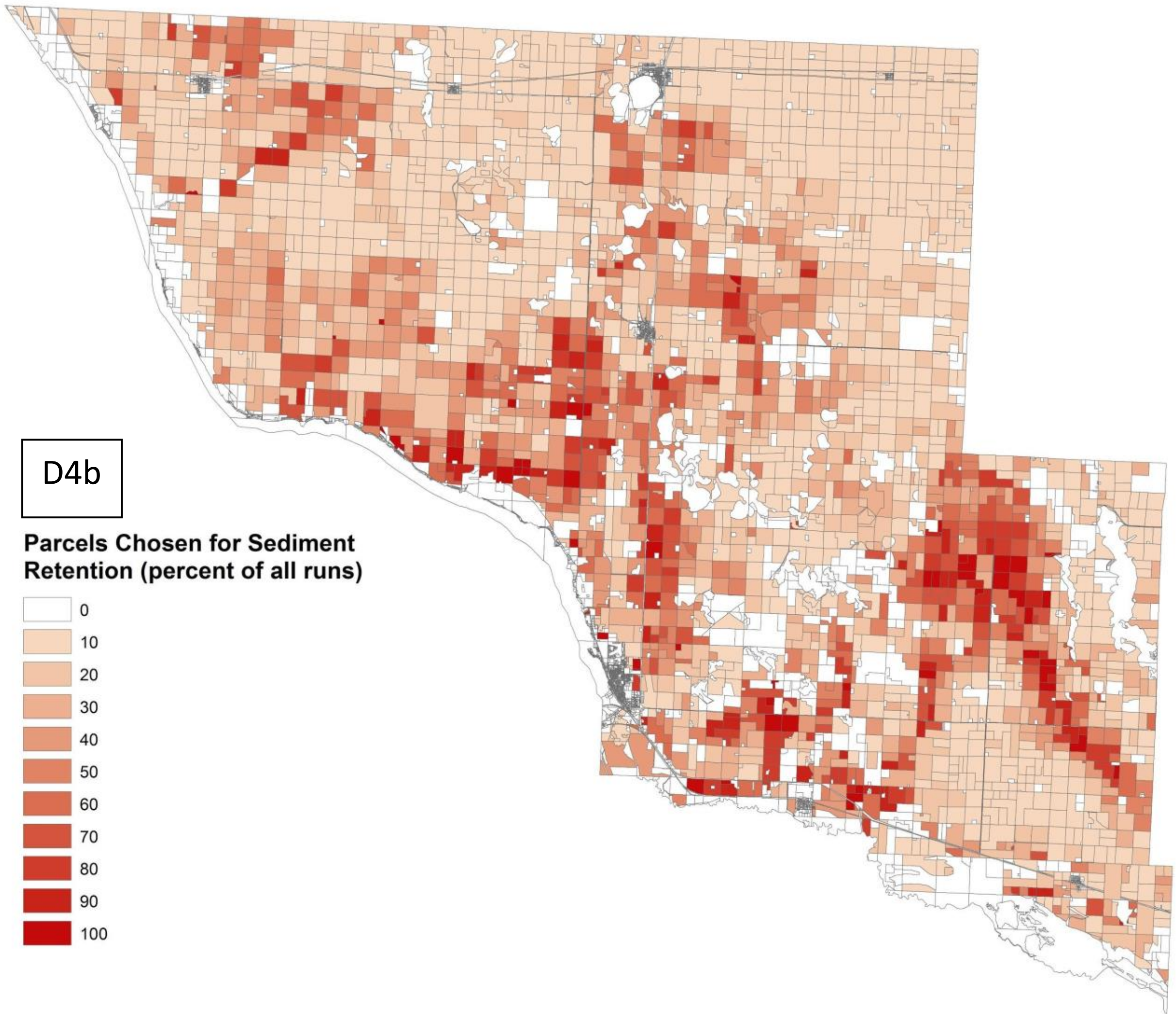
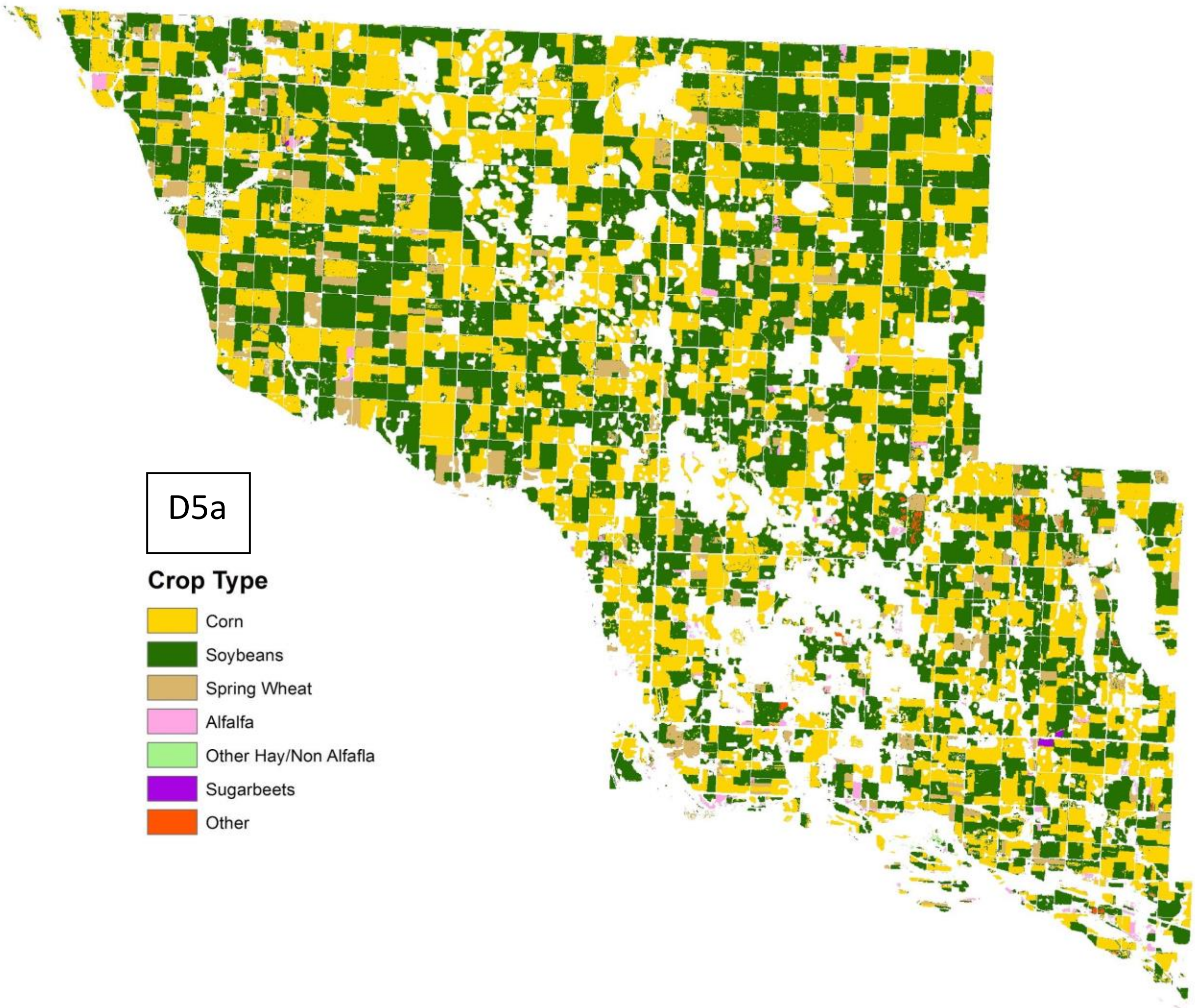


Figure D4. Sediment Retention Potential. (a)Map of S retention potential (tons) per pixel for the transition between agricultural land and grassland. S retention potential is fairly uniform across the landscape, save for a very few select pixels along waterways in the south-eastern quadrant of the county and along the Minnesota River (see cutouts). These pixels exhibit an extremely high S retention potential relative to the background. (b)Map of land parcels chosen for restoration, optimizing S retention. This combined all ten optimization runs, from 10% of potential to full restoration, and reports the frequency each parcel was chosen in the best restoration solution. S retention shows the greatest spatial aggregation of parcels, surrounding the high-priority pixels along watersheds in the southeast and along the Minnesota river along the southern border. This corroborates the S retention model's efficiency in selecting cost-effective restoration solutions, as there are a few parcels that contribute disproportionate amounts of S retention.


D5a
Crop Type
Corn
Soybeans
Spring Wheat
Alfalfa
Other Hay/Non Alfafla
Sugarbeets
Other

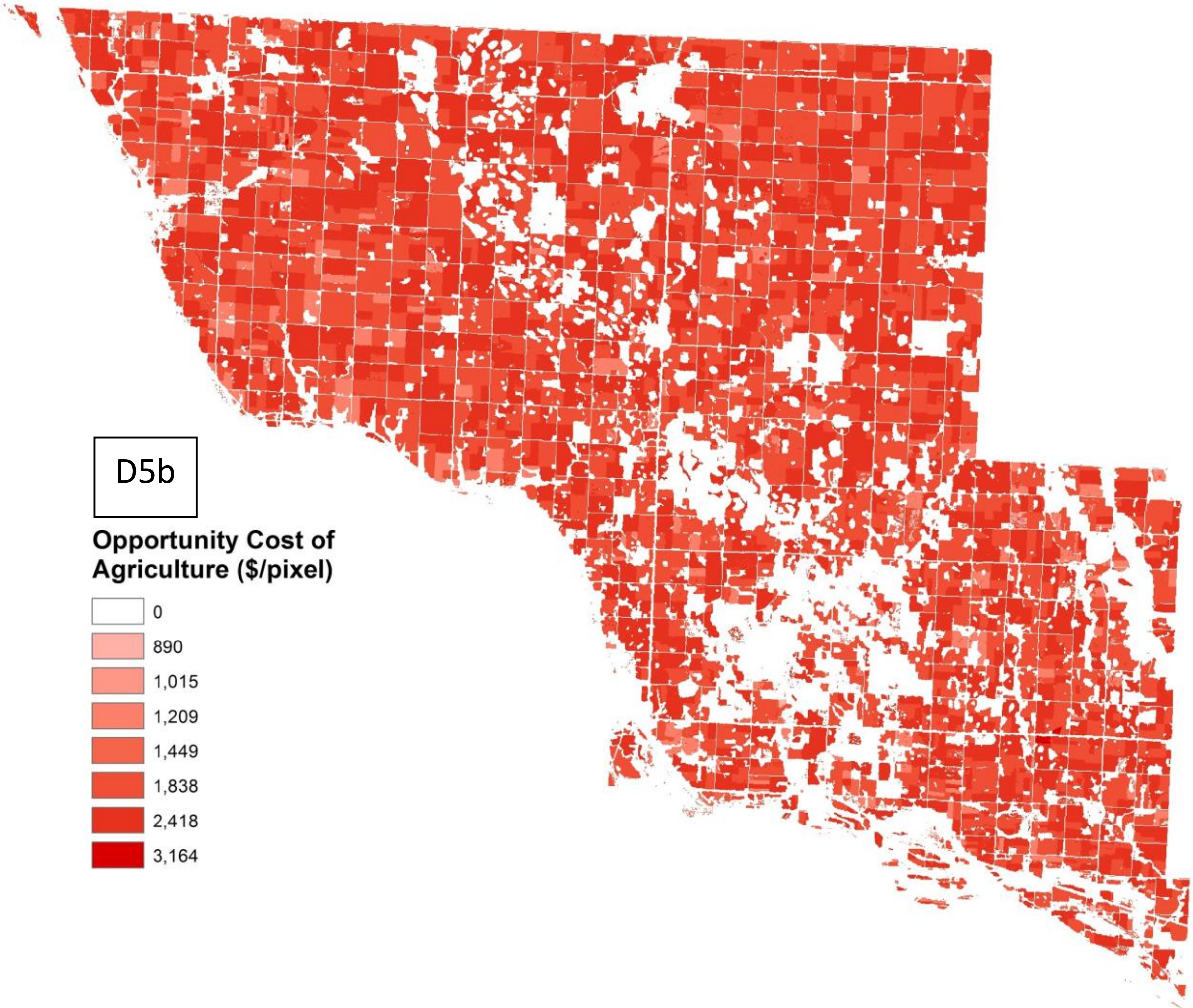


Figure D5. Agricultural Cost. (a)Map of agricultural crops cultivated in Big Stone County during 2014 (USDA-NASS CDL). Most of the county is in a corn-soybean rotation, with some spring wheat and alfalfa fields. Sugarbeets and other agricultural crops are essentially absent. (b)Map of the opportunity cost of agriculture across the landscape. Values are based off the annual profit earned under the crop types in (a) (Table B1). There is some variation, but generally the cost is comparable across most of the landscape.

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