Valuing the ecosystem services of low-input, high-diversity prairie as a biofuel feedstock in southern Minnesota

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Abstract

Biofuels may help to address the United States' dependence on fossil fuels by providing a renewable fuel source (Hill 2006, Tilman 2009). The largest biofuel industry in the United States is currently corn-based ethanol, but the negative environmental and economic impacts of corn agriculture have prompted research into other feedstocks, such as low-input, high-diversity (LIHD) prairie (Tilman 2006). We argue that incorporating the ecosystem service value of LIHD prairie grown on marginal lands in Southern Minnesota would make it an economically competitive biofuel feedstock. Using a spatially explicit model (InVEST) we found that a targeted land-use change of corn to prairie on marginal lands produced a value of $198.89/ha in ecosystem services, $163.34 higher than an all-prairie scenario and $511.28 higher than an all-corn scenario. An economic analysis incorporating the value of ecosystem services found that prairie is only competitive with corn as a feedstock when the prices of carbon and prairie feedstock are high and the price of corn is low. However, improvements in modeling could better quantify prairie's ecosystem service value, making it more competitive with corn. Our results demonstrate the importance of taking ecosystem service value into account when making decisions regarding biofuel policies.

I. Introduction

The United States' consumption of gasoline – nearly 400 million gallons per day – directly contributes to climate change and potentially irreversible impacts on natural ecosystems worldwide (Vitousek 1997, IPCC 2007, EIA 2012). Our reliance on fossil fuels to meet the needs for development, energy, and food production is being called into question due to dependence on unstable foreign parties and negative climate impacts (Nelson 2010). Members of the Organization of Petroleum Exporting Countries (OPEC) hold power over nine out of the ten largest oil companies, which control 79% of the world's petroleum reserves (Greene 2010). Disruptions in trade between the United States and OPEC nations come at a great cost for American citizens. Instability in the past 10 years has driven oil prices to record highs of over $100 a barrel – nearly $30 higher than when prices skyrocketed in the 1980s (Greene 2010, Oil-Price.net 2012). Both our dependence on fossil fuels with few affordable alternatives and foreign countries' monopoly on oil cause the U.S. to lose $250 billion every year (Greene 1998). The instability of relying on imported energy has triggered political action to localize energy production (EPA 2010a).

The desire to reduce climate change-causing greenhouse gas (GHG) emissions from burning fossil fuels has prompted the U.S. to target the transportation sector, as it accounts for 27% of the country's GHG emissions. Transportation fuel consumption results in 1.7 billion metric tons (t) of carbon dioxide (CO_2) emitted annually (IPCC 2007, EPA 2011b). Considering that transportation energy consumption in the United States has increased linearly with GDP per capita since 1946, we are on track to emit even more from this sector (Liddle 2009). Despite advances in fuel economy and regulation, passenger vehicles still emit 5.1 t of CO_2 per car per year (EPA 2011b). Reducing gasoline consumption in the transportation sector would decrease climate change-causing emissions and dependence on foreign oil.

There is citizen support for a shift from foreign oil towards renewable fuels. A survey of Americans in 2006 found an annual household willingness to pay (WTP) of $137 towards a reduction of dependence on foreign oil and development of crop-based fuel in the United States. The survey found additional support for the creation of a national fund to provide grants for advanced energy projects with the intent to reduce oil consumption (Li 2009). The presence of a biofuel-supporting base should encourage continued of such fuels.

A major step in the development of bioenergy has been the implementation of large-scale bioenergy crop production. Worldwide, 50% of land has been changed to grazed land or cultivated crops. In the U.S., a growing amount of this land-use change is due to the expanding biofuel industry (Kareiva 2007), which currently uses corn as its primary feedstock. However, the negative environmental and economic impacts of corn agriculture have prompted research into new biomass sources (Tilman 2006). We argue that the numerous ecosystem services and ecological benefits of low-input, high-diversity (LIHD) prairie suggest that prairie as a potential biofuel feedstock warrants more investigation.

Corn biofuels

The United States has been a leading producer of biofuels since it implemented national policies supporting corn ethanol production and consumption in the 1970s (Solomon 2007). In 2007, 95% of ethanol came from corn grain (*Zea mays L.*), while the other 5% was produced from wheat, barley, and cheese whey (Urbanchuk 2009). In 2009 alone, the United States produced 41 billion liters of ethanol, still mostly from corn grain (Hsu 2010). A large portion of both the corn grown for ethanol and the ethanol processing plants is found in southern Minnesota and northern Iowa, with the rest mostly concentrated in other areas of the Midwest (Figure 1).

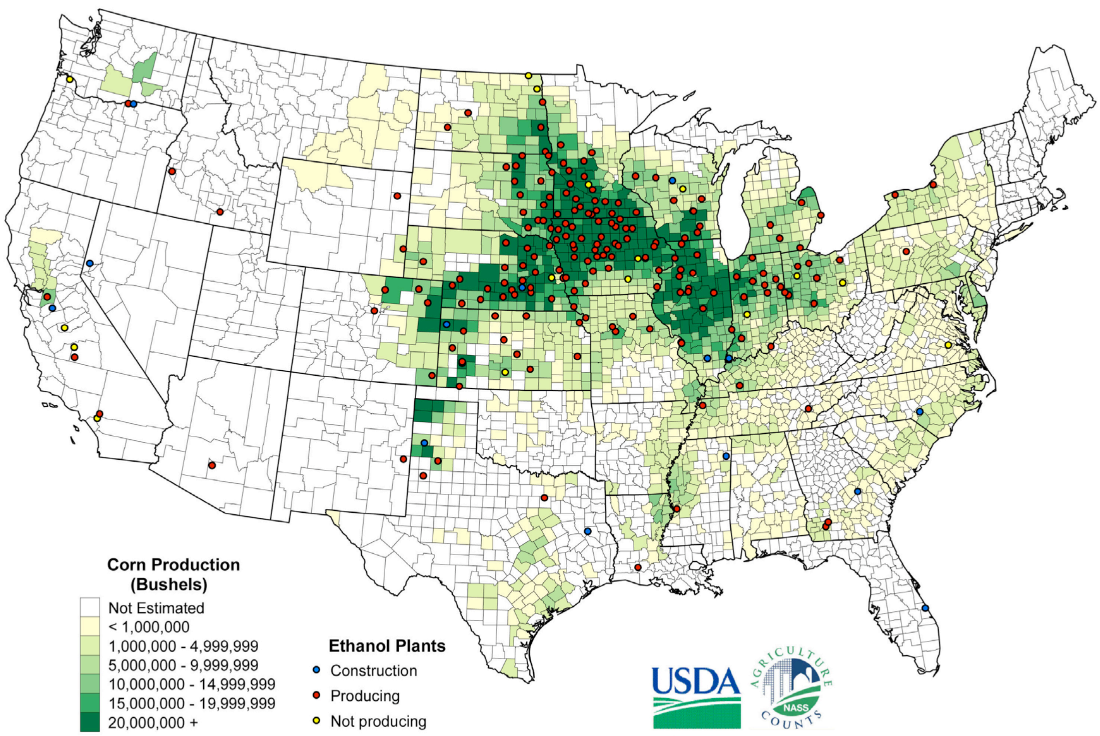

Figure 1. Ethanol and corn production in the U.S. in 2011. Map of corn production by amount per county, shown in green. Distribution of ethanol plants that are currently producing fuel shown as red dots. Production of ethanol and corn are concentrated in the Midwest states, specifically Minnesota, Iowa, Nebraska, and Illinois. Image from National Agricultural Statistics Service (USDA 2011).

Federal subsidies for producers and incentives for consumers have supported the expansion of the ethanol industry since its establishment. The Energy Policy Act of 1978 initiated a \$0.40 per gallon subsidy for ethanol-blended fuels; since then, this subsidy has ranged from \$0.40 to \$0.60 per gallon (Tyner 2007). Consumers who buy E-85 vehicles are eligible to receive additional tax deductions on their purchases (Tyner 2007). The Food, Conservation, and Energy Act of 2008 gave corn farmers, regardless of the end product, a \$0.28 per bushel direct payment to offset costs and incentivize corn cultivation (H.R. 6124 2008).

Concerns about the negative economic impacts of ethanol subsidies provided by the 2008 Farm Bill caused these subsidies to be revoked in 2011 (H.R. 6124 2008, Pear 2012). Subsidies for ethanol have been blamed for causing high corn prices (GAO 2009). While both the ethanol industry and high corn prices increase farmer profits, they also result in increased food prices. Corn grain-fed meat and poultry prices increase for consumers and production costs increase for livestock producers (GAO 2011, Pear 2012). The increase in food commodity prices also impacts federal food programs and large food companies (GAO 2011). Additionally, it has been argued that ethanol costs more to produce than it is worth in energy value (Pimental 2007).

Nevertheless, proponents of corn ethanol tout its environmental benefits over traditional fossil fuel-based fuels. Ethanol blended fuels are perceived as cleaner than gasoline because they emit fewer nitrogen oxides and hydrocarbons (Wang 1998, Park 2010). Additionally, ethanol re-releases carbon (C) into the atmosphere that was recently taken up by plants, whereas burning gasoline adds C that had previously not been part of the C cycle (Wang 2005).

Despite corn ethanol's reputation as a "clean" energy, corn production is the most fertilizer-, insecticide-, and herbicide-intensive crop in the United States (Pimentel 2005) and has been condemned for its significant environmental effects. Some of the most harmful environmental impacts include affected localized water quality, downstream water contamination, decreased soil quality and soil retention (McLaughlin 1998), C emissions as a result of land-use change (Pielke 2002), and loss of biodiversity (Fargione 2009, Gerdiner 2010, Polasky 2011).

A major impact of continuously planted corn grown for ethanol is the runoff of nutrients and soil. This runoff pollutes local drinking water sources and degrades marine habitat in nearby water bodies (Duff 2008). Agricultural inputs of phosphorus (P) and nitrogen (N) in the Upper Mississippi River Basin also affect water quality as far away as the Gulf of Mexico. Corn and soybean production accounts for 52% of the nitrogen (N) and 25% of the phosphorus (P) flowing into the Gulf (Alexander 2008, Helsel 2011). Eutrophication from increased nutrient loading negatively impacts marine life through hypoxia, which is the formation of oxygen-limited zones following algal decomposition (Petrolia 2006). Soil runoff from agriculture also negatively affects local lakes and streams; it results in increased sedimentation and turbidity, which in turn leads to rising water temperatures that impact organisms reliant on cool fresh water (Fargione 2009). Soil erosion is particularly high in areas of corn agriculture due to the shallow root system of the corn plant (Howell 1995).

Cellulosic ethanol

Cellulosic feedstocks are currently being studied for possible use as biofuels, because the material is diverse, abundant, and renewable (Somerville 2010). The most common feedstocks under consideration are miscanthus (*Miscanthus giganteus*), switchgrass (*Pansium virgatum*), corn stover, and LIHD prairie. All four of these feedstocks result in fewer life-cycle GHGs than corn and could be feasibly established for the production of biofuels (Tilman 2006).

Due to the physical differences between cellulosic fuel production and corn ethanol production, the cellulosic ethanol industry has the potential to utilize these many feedstocks. Cellulosic ethanol differs from corn ethanol because the processing uses different parts of the feedstock plant. Corn ethanol is predominantly starch-based, meaning the fruit of the corn plant is the feedstock. Cellulosic ethanol uses plant mass made up of the structural component lignocellulose. This material, which is comprised mainly of cellulose, hemicellulose, and lignin, can be used to create fuel by extracting the cellulose and hemi-cellulose to obtain soluble sugars (Somerville 2010). These sugars are then fermented and purified before being used as a fuel (Hahn-Hagerdal 2006).

Government support for cellulosic fuel already exists in the United States. The Food, Conservation, and Energy Act of 2008 allocated $1 billion to incentives, such as tax credits, to both farmers growing cellulosic feedstocks and biofuel producers using cellulosic feedstocks (Khanna 2008). The act also provided assistance to cellulosic biorefineries, as well as support to the research, development, and advancement of biorefinery technology (Khanna 2008).

The government also demonstrated its support of cellulosic ethanol when the Environmental Protection (EPA) created the Renewable Fuel Standards (RFS) program in 2005. This program mandated the production of biofuels, and expanded two years later to the RFS2 program. The mandate specifies that 36 billion gallons of biofuels should be produced by 2022. Although the majority of the mandated fuel is to come from corn-based ethanol, 16 billion gallons are required to be cellulosic ethanol (EPA 2012). And while the current amount is fairly low – 10.45 million ethanol-equivalent gallons or 0.006% of the mandate – the EPA is reviewing the mandate annually and plans to update this requirement as the cellulosic ethanol industry becomes more established in the Midwest (EPA 2010a, 2010b).

Research is currently being done to determine which feedstock option has the least harmful impacts on local environments, sufficient energy yields, and the greatest benefit to society (Tilman 2006, Hill 2009, Somma 2010, etc.). Recently, researchers have highlighted tallgrass prairie as a promising cellulosic feedstock (Tilman 2006). Prairie requires few inputs, provides ecosystem services such as C sequestration and nutrient retention, and can yield three times more energy than unfertilized switchgrass when grown on marginal lands (Tilman 2006).

Tallgrass prairie as a biofuel

Historically, tallgrass prairie flourished in Midwestern North America. From Alberta, Saskatchewan and Manitoba in Canada, prairie spread south through southern Minnesota, the eastern portion of the Dakotas, Nebraska, Kansas, northwestern Missouri, Oklahoma, Iowa, and Texas (Dobrovolny 2003, Forsberg 2009). Now, between one and five percent of the original 142 million acres of tallgrass prairie exists in the Midwest. The rest was lost to agricultural settlement that began in Iowa around 1840 and moved west to Minnesota, Kansas, Nebraska and the Dakotas in the 1860s and 1870s (Samson 1994, Forsberg 2009).

Prairie has the greatest species diversity of all ecosystems in the Midwest, but the loss of these grasslands has significantly altered natural species diversity (Brennan 2005, The Nature Conservancy 2012). Prairies provide much-needed habitat for bird and small mammal populations. Restoration of native prairies would allow the recovery of 15 obligate prairie bird species that have seen significant population losses alongside the loss of this habitat (Brennan 2005). Birds are particularly affected by prairie fragmentation or loss to agriculture or development because they lose habitat for breeding, migrating, and wintering in addition to their major food sources found in prairies (Askins 2007). Insects and small mammals, such as voles, mice, prairie dogs, and shrews, also rely on prairie grasses for food and shelter (French 1976, Fargione 2009, Gerdiner 2010). Increased plantings of prairie for use as a biofuel would also contribute habitat to these prairie species.

Tallgrass prairie cultivated for biofuel, commonly referred to as LIHD prairie, is a more efficient crop than corn. For one, it requires few agricultural inputs beyond the establishment phase (Tilman 2006). Tallgrass prairie is also abundant in perennial C4 and C3 grasses, forbes, and legumes [*see appendix*] (Camill 2004); this diverse mix is intrinsically more efficient in water and nutrient use compared to corn because of niche complementarity (De Deyn 2008, Somerville 2010). Niche complementarity refers to species function differentiation in a system: particular species utilize resources in distinct ways, at different spatial and temporal scales, and in various forms, meaning a system's resources are used more efficiently (De Deyn 2008). Monoculture ecosystems, such as cornfields, lack niche complementarity and therefore do not efficiently utilize available nutrients.

Prairie also retains nutrients well over time due to its seasonality and complex root structure. The resorption of nutrients in perennial grasses (the relocation of nutrients from above-ground biomass to below-ground biomass at the end of the growing season) results in less nutrient removal from the ecosystem when prairie is harvested in the winter (Somerville 2010). This contributes to the differences seen in nutrient retention between corn and prairie (Knapp 1998, Somerville 2010).

LIHD prairie is not only better at retaining nutrients than agricultural landscapes, but it also provides significant advantages in soil quality. The root systems of herbaceous prairie plants increase soil water holding capacity and allow greater infiltration of surface nutrients. This improves soil structure, allows deeply rooted plants access to depth-limited nutrients, and aids in organic matter formation in deep soil, which further improves nutrient availability (Knapp 1998).

Prairies are also better at storing C underground than corn due to their large underground biomass. The high turnover of prairie roots compared to corn roots results in direct inputs of C into the soil (Tilman 2006). In comparison, most corn biomass is aboveground and removed each year. An additional advantage in C storage found in high-diversity prairies is the presence of nitrogen-fixing legumes. The increase of nitrogen to the soil results in increased rates of C storage when compared to grass-only mixes or monocultures (CBO 2009).

The high diversity of tallgrass prairie has other benefits that make it an ideal biofuel crop, such as resistance to extreme variability (climate, fire, and grazing) and increased net primary productivity (NPP) (Knapp 1998, Somerville 2010, CEEIIBP 2011). Diverse plant communities in grassland ecosystems are better able to recover from major drought; as species are lost, ability to resist declines (Tilman 1994). High-diversity ecosystems also show little variation in population size and composition from year to year (Thomas 1985). This is an indicator that cultivating LIHD for biofuels would produce consistent yields year after year, despite harvesting.

Research objective

Our objective was to more accurately value the ecosystem services of LIHD prairie grown on marginal lands, or lands that are not suitable for conventional agricultural practice, in 43 counties in southern Minnesota. Our study builds on the research of Tilman et al. 2006, who performed an ecological analysis of LIHD in the context of biofuel feedstocks, and Polasky et al. 2011, who used the InVEST model to value ecosystem services in Minnesota. We depart from previous research by using InVEST to perform an economic analysis of LIHD prairie's multiple ecosystem services. We focused on the following ecosystem services: C sequestration, nutrient retention, and habitat quality as a proxy for biodiversity. These findings can help decision makers understand and consider the value of LIHD prairie as a cellulosic ethanol feedstock while making policy decisions aimed at meeting the RFS2 mandate.

We argue that LIHD prairie grown on marginal lands in southern Minnesota can be economically competitive with corn as a biofuel feedstock when the value of the ecosystem services is taken into account. Due to under-valuation of these services, we recommend that policy-makers take ecosystem services into account when deciding agricultural subsidies and taxes, and that grants should be put towards the research and development of prairie as a cellulosic ethanol feedstock.

II. Methods

The InVEST model

In order to quantitatively measure the economic value of LIHD prairie ecosystem services, we chose to use the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) tool (Tallis 2011). InVEST was developed as a part of the Natural Capital Project, which is a partnership between the University of Minnesota, Stanford University, The Nature Conservancy, and World Wildlife Fund that aims to better quantify the value of nature. The InVEST model combines maps, tabular data describing biophysical processes, and economic valuations to create spatially-explicit predictions of the change in value of ecosystem services resulting from a defined land-use change scenario.

Although other tools assess land-use change with a focus on bioenergy (McCormick 2008), InVEST is unique among these models for a variety of reasons. First, it is spatially explicit, allowing us to tailor our analysis to a specific location. Some models, such as the GHGs, Regulated Emissions, and Energy Use in Transportation (GREET) Model, perform more complete life-cycle assessments of bioenergy fuel production but cannot account for spatially-explicit factors, such as varying nutrient retention due to topology (Wang 1999).

Second, InVEST is a family of models that allowed us to analyze different services using the same general baseline data and methods. Though a variety of other tools exist that can analyze each specific service with greater accuracy (McCormick 2008), we wanted to compare ecosystem service values across multiple services. The use of a variety of models would possibly create confounding variables and would make it difficult to combine our results for an economic analysis.

Third, InVEST was created explicitly to help decision-makers create policies regarding land-use change. Some models, such as Payments for Ecosystem Services (PES) models, are instead aimed at developers looking to maximize both their profits and compliance with environmental regulations (McCormick 2008). However, a goal of our study was to formulate potential options for policy-makers, making InVEST a more ideal tool.

Finally, InVEST expresses the final output in dollars, a valuation step that moves the analysis from one of ecological benefits to one of ecosystem services. The Food and Agriculture Organization (FAO) created Bioenergy Environmental Impact Analysis (BIAS), a comprehensive analysis tool aimed at examining all possible ecological implications of land-use change for biofuel production (Fritsche 2010). Though the main goal of this tool is to provide information to decision-makers, the final results of BIAS are presented as a variety of environmental quality indicators instead of monetary values.

InVEST has previously been used by researchers to demonstrate that biodiversity conservation and increased ecosystem services are not mutually exclusive in the Willamette Basin, Oregon (Nelson 2009). InVEST has also been used to quantify changes in ecosystem services from land-use change in Minnesota (Polasky 2011). Due to the similarities between the Polasky et al. study and our own, we followed their methodology, while also drawing on methodology presented by Tallis et al. 2011 in the InVEST documentation.

Study area

We focused our study on 377,450.55 hectares (ha) of marginal lands in 43 counties in southern Minnesota (Figure 2). By targeting marginal lands, we hoped to select the most realistic areas where prairie would have competitive profits, as well as maximize ecosystem services. This is important because high prices of corn may motivate farmers to grow corn on less ideal land, and undermine any ecosystem services provided by other land cover.

We defined marginal lands using the United States Department of Agriculture's (USDA) National Soil Survey Handbook's Land Capability Classification (LCC). According to the USDA, LCC is defined as "a system of grouping soils primarily on the basis of their capability to produce common cultivated crops and pasture plants without deteriorating over a long period of time" (USDA 2012). We chose soils classified as either Class III or Class IV soils because they have "severe" or "very severe" constraints that affect either the type of crop that can grow there, or that require special conservation practices or careful management. We eliminated more fertile Class I and Class II soils on the basis that corn would be a stronger economic competitor on these lands, and we eliminated Class V soils and above on the basis that their limitations would restrict LIHD prairie farmers' abilities to harvest the crop.

Scenarios

We used 2010 land cover data obtained from the USDA National Agricultural Statistics Service Cropland Data Layer. These data were used to create both a baseline 2010 land use/land cover (LULC) map, as well as three alternative 2010 scenarios. In preparing the LULC dataset, we aggregated the original 255 land cover classifications into one of seven broad classifications based on similarity of land cover (Table 1). Note that Corn/Soy is assumed to be in rotation because farmers in southern Minnesota typically grow these crops on a rotating basis (Wilhelm 2004). Hereafter, all references to corn are assumed to be corn/soy rotation. In choosing values to represent each of the aggregated land covers, we averaged multiple locally applicable values from the literature. Though this represents an assumption in all our models, it would have been nearly impossible to find accurate inputs for each specific land cover.

We chose to model alternative 2010 scenarios in order to examine the changes that could have been achieved in Minnesota had different land-use decisions been made in the past. The four scenarios we used are:

1) *Current Scenario*: 2010 LULC map pooled into seven LULC types (Figure 3).
2) *All Corn (AC) Scenario*: all class 3 and class 4 lands in the study area converted to corn/soy rotation.
3) *All Prairie (AP) Scenario*: all class 3 and class 4 lands in the study area converted to LIHD prairie.
4) *Corn to Prairie (CP) Scenario*: land under corn/soy rotation on class 3 and class 4 lands in the study area converted to LIHD prairie (Figure 4).

Table 1. Land Use Land Cover (LULC) definitions adapted from the USDA-National Agricultural Statistics Service used in the InVEST models.

LULC classification	Descriptions
Corn/Soy	Corn and soy (assumed to be in rotation)
Wetlands	Herbaceous wetlands, woody wetlands, wetlands
Forests	Deciduous forest, Evergreen forest, mixed forest, woodland
Grasslands	Pasture/hay, pasture/grass, grassland herbaceous, other hay, fallow/ idle cropland
Agriculture	All types of agriculture not listed under corn/soy, including fruits, vegetables, etc.
LIHD Prairie	Hypothetical cellulosic ethanol feedstock
Other	Urban, roads, barren, open water

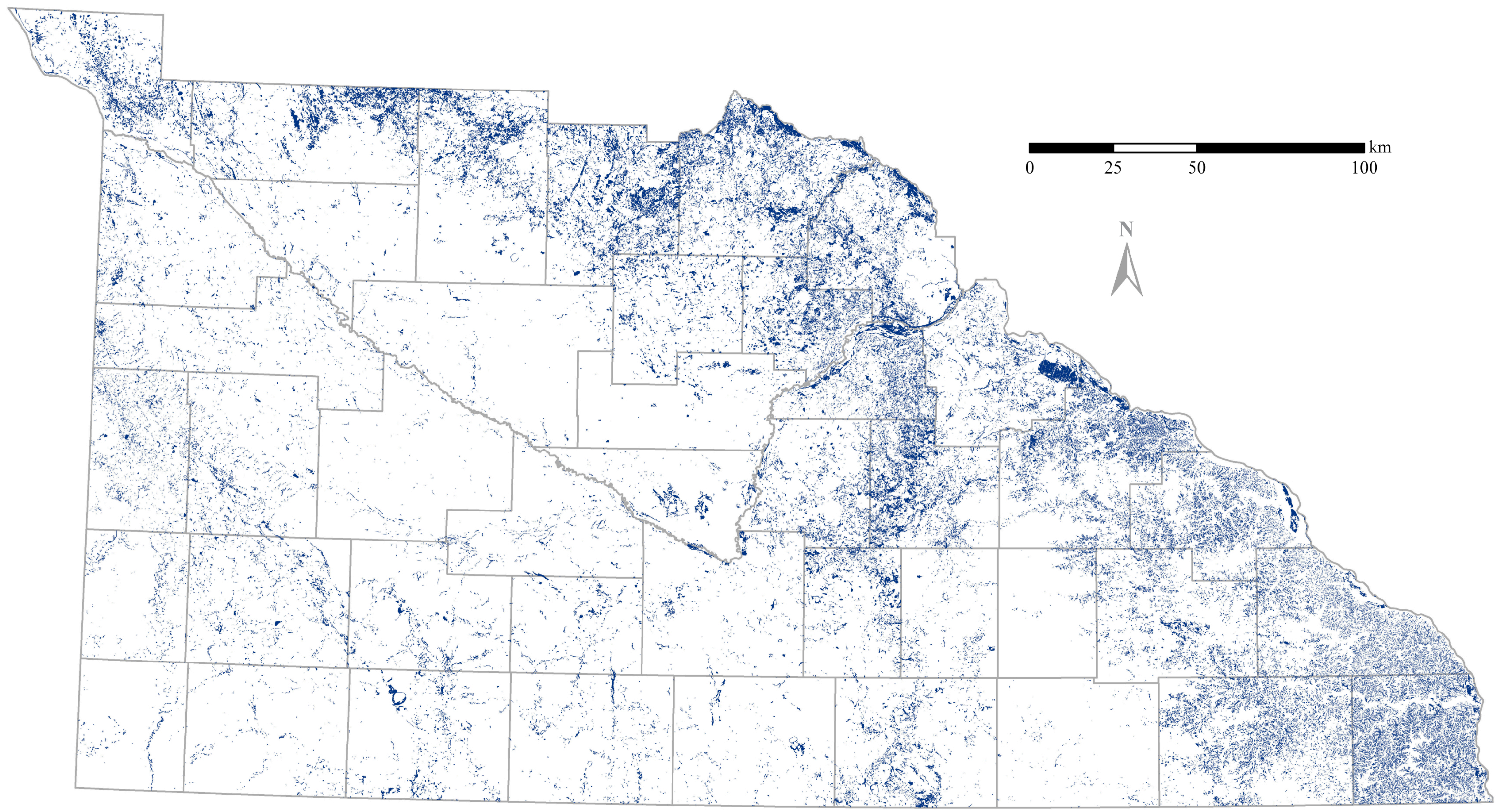

Figure 2. Map of marginal land in southern Minnesota. Navy-colored pixels correspond to class 3 and 4 lands in our study site, which consists of 377,450 ha in 43 counties. LULC data were from 2010 at 30 m resolution.

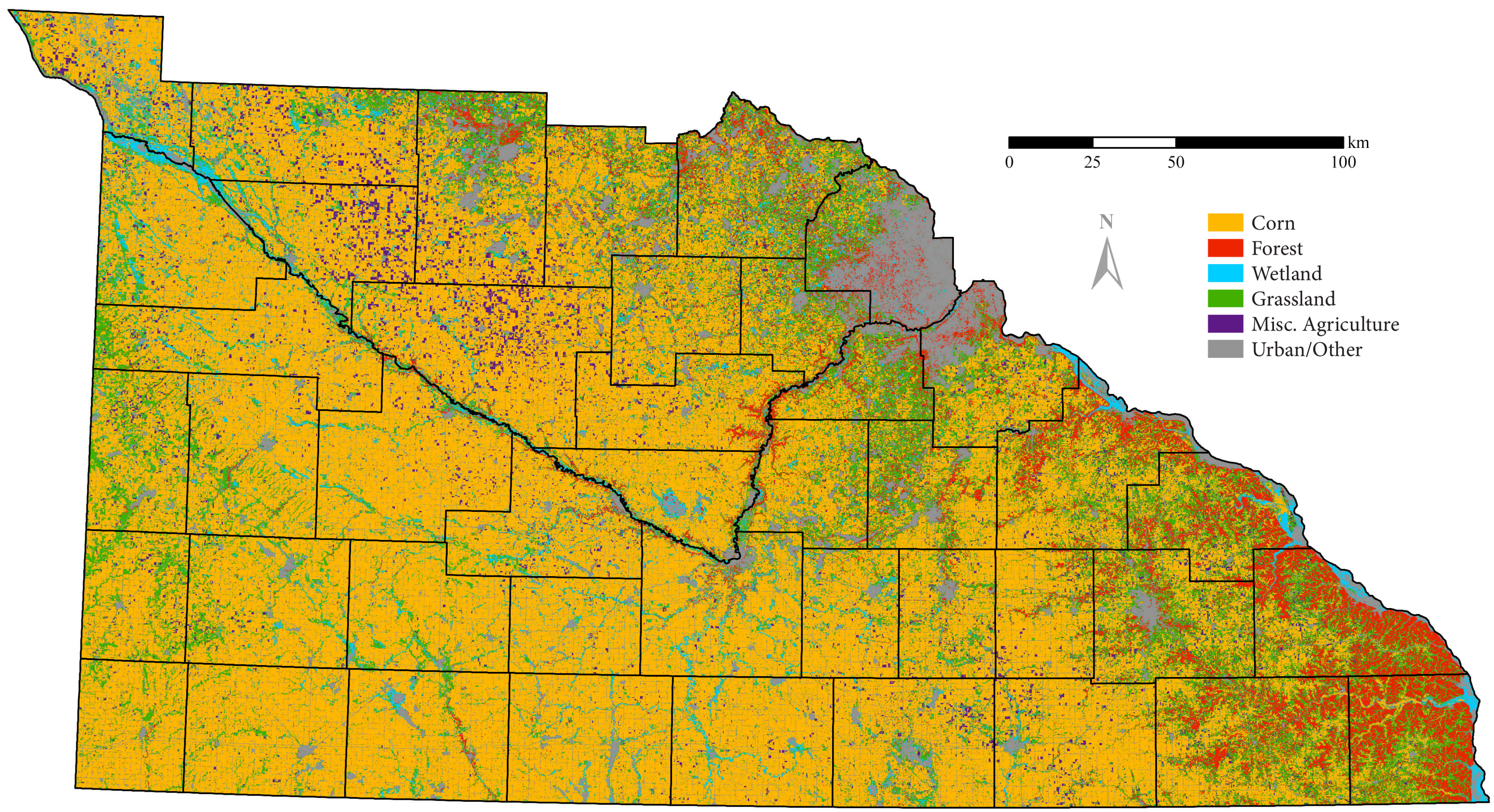

Figure 3. Map of Current scenario. Shows pooled 2010 LULC classifications. This scenario served as the baseline to which all other scenarios were compared. LULC data were at 30 m resolution.

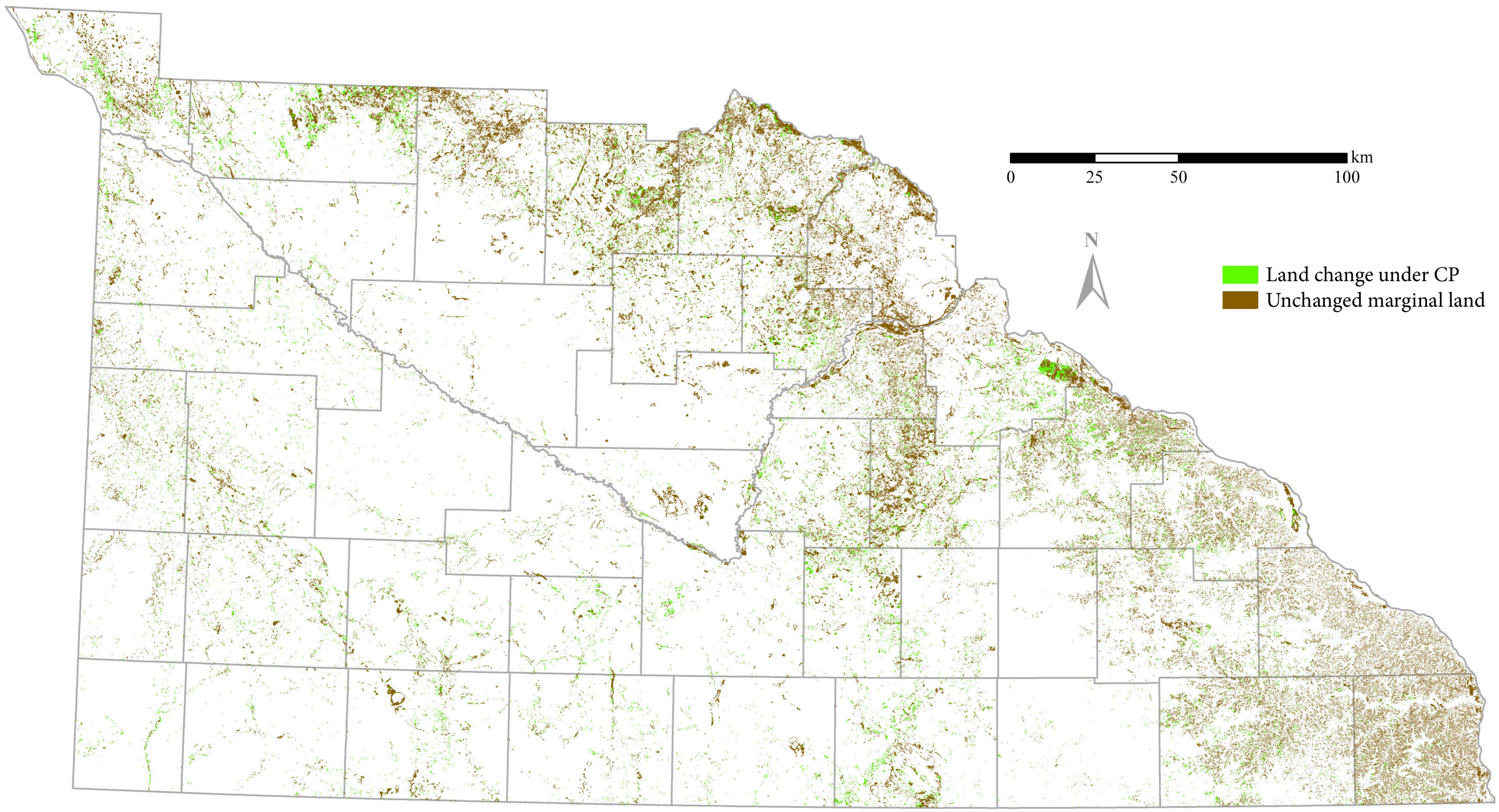

Figure 4. Map of land changed in CP scenario. A total of 81,090 ha out of 377,450 ha of marginal land was changed from corn to prairie in the CP scenario. In the AP scenario, all of the marginal land was converted to prairie. In the AC scenario, only 296,359 ha of marginal land was changed to corn because some areas were already in corn production. LULC data were from 2010 at 30 m resolution.

Economic analysis

The economic analysis was done to determine what price combinations, if any, would allow for LIHD prairie to be economically competitive with corn when the ecosystem services we modeled were included in the value. We chose prices for C, corn, and prairie feedstock within realistic ranges to create potential market scenarios. The comparison included corn net profit, prairie net profit, and modeled values of the C sequestration and nutrient retention ecosystem services.

We established three plausible prices for corn, based off of annual average Minnesota prices from 2001-2011. The low price was set to the 2001 price of $1.85/bushel (bu) (NASS 2002); the high price was set to the 2011 price of $11.50/bu (NASS 2011). For a mid-range value, we chose $3.85/bu, the mid-point between the low and high prices. We selected the mid-point price as opposed to the decadal average because it was $0.63/bu (NASS 2002, 2004, 2008, 2011, 2012) higher, and thus more similar to prices seen today. Though the low price of corn is three times lower than the current price of corn, we believe that it is realistic because ethanol demand has been raising the price of corn (Fornara 2008). The increased availability of a cellulosic ethanol feedstock could provide an alternative that would drive corn prices down. To determine the net profit per hectare, we utilized average Minnesota corn yields and average fertilizer, chemical, and seed input costs from 2010 (USDA 2011b). The same process was repeated for soy, with net profits averaged together.

As there is a limited cellulosic feedstock market, we used willingness to accept (WTA) values of 75, 98, and 133 $/Mg from the Committee on Economic and Environmental Impacts of Increasing Biofuels Production (CEEIIBP 2011). This roughly corresponded to the price of $70-$78 per dry Mg of corn stover we received from POET Biorefining (Johnson 2012). We assumed a yield per ha of 5 Mg based on long-term study of annual NPP in Konza Prairie (Knapp 1998). Though prairie does not have annual input costs such as chemicals or fertilizer, it does have a one-time establishment cost of $3,264/ha (Prairie Moon Nursery 2012). To create an annual cost, we divided establishment costs by a 40-year time horizon to be consistent with assumptions made in the C model.

Carbon sequestration model

C sequestration is the amount of C stored in above- or below-ground living biomass, soil, and detrital mass (Baer 2002, Feng 2005, Tallis 2011). Though there is no set price of C, society accrues the cost of emitting an additional ton of C through economic impacts of climate change (Tol 2011). As a result, sequestering C has a value to society of the marginal cost of admitting an additional ton of C (Tol 2009, 2011).

The C model utilized the aggregate of four C storage pools: above-ground biomass, below-ground biomass, soil, and dead organic matter in each land use (Table 2). The pool numbers were found using literature with field-level data that most closely reflected the land uses in southern Minnesota. For most land-use types, several numbers from different studies were averaged together [*see appendix*].

The model calculated differences in the amount of C stored in a land parcel over a given time horizon between two land-use scenarios. This is because InVEST uses changes in C pool size, as opposed to C sequestration rates. It is necessary to include a hypothetical time horizon in order to value the change in C storage on a yearly basis. We assumed a time horizon of 40 years for three reasons. First, prairie continues to sequester carbon at a constant rate for at least 40 years (McLauchlan 2006). Second, we assumed that biofuels will continue to be in demand 40 years from now based on historically increasing ethanol production, as well as possible oil shortages (Solomon 2007). Finally, the relatively small contribution (<10%) of ethanol to the nation's fuel supply leaves ample room for expansion (EPA 2010a).

C values were derived from Tol (2011) who performed a meta study of papers from 1994 to 2010 on the social cost of one additional ton of C emitted (SCC). The social cost is the damage done by emissions of CO_2 compared to a baseline scenario in which CO_2 emissions do not increase (Pearce 2003). We used C prices of 36, 81, and 134 $/t. These values represent the 33rd, 50th, and 67th percentile of the range of values from papers published between 2000 and 2010.

Because InVEST does not account for emissions released when creating the synthetic fertilizer used in almost all corn production, we compensated by calculating the emissions separately. We utilized current recommended N application rate for corn production (Rehm 2006) and multiplied it by both a coefficient that represents C emissions from each unit of fertilizer production (Wood 2004) and the number of ha which change from corn to another land use in our scenarios. This amount represents C emissions that would be avoided in scenarios that shift land use away from fertilizer-intensive corn production. We then valued the quantity of emissions utilizing the same prices per tC utilized in the C sequestration model.

TABLE 2. Size of C pools for LULC classifications.

LULC classification	C pool (Mg/ha^{-1})
Corn/Soy	122.2
Wetlands	180.75
Forests	127.7
Grasslands	122.2
Agriculture	60
LIHD Prairie	122.2
Other	0

Nutrient retention model

Nutrient retention is defined as the ability of a landscape's natural vegetation to filter nutrients out of groundwater that flows through. It is measured as the amount of nutrients in the water when it leaves a land-use type (Tallis 2011).

To calculate nutrient retention, the InVEST model first calculated the difference between water input from precipitation and loss to evaporation. The difference is the water runoff for each cell. Next, values of nutrient loading, root depth, and nutrient retention coefficients were determined for each land-cover type by using averaged values from the literature [*see appendix*] (Keeler 2012). The biophysical constants were used to determine the amount of nutrient export per cell, which was then aggregated and summed at the subwatershed and watershed levels. Due to time constraints and the intense processing requirement of the nutrient retention model we ran it at a resolution of 200 meters (m).

Though InVEST has a mechanism for valuing nutrient retention based on the treatment of water, we chose not to use it because of the scale of our study site. Water treatment facilities have differing treatment costs, so we would have to determine water treatment costs for every facility in southern Minnesota. As a more feasible alternative, we opted to utilize a method employed by Polasky et al. (2011) in a similar study on ecosystem services resulting from different land-use change scenarios. Polasky et al. utilized a study by Matthews (2002) performed in the Minnesota River Valley. The study surveyed households and determined that a household would be willing to pay $140 per year for a 40% reduction in P loading in the Minnesota River. We adjusted the household WTP for inflation, yielding $190.20, and extrapolated the total WTP for all of southern Minnesota by multiplying the household WTP by the number of households in our study area according to the 2010 census. This yielded a total of $218,740,841.

The valuation per ha was determined by first calculating the percent change in nutrient export from the Current scenario to each of the alternatives. We assumed a linear relationship between WTP and percent reduction up to the values determined by Matthews (2002) (Polasky 2011). The total WTP, adjusted to reflect the modeled percent change in nutrient export, was then divided by the number of ha that underwent land change to determine the change in ecosystem service value per ha per year between scenarios.

Biodiversity and habitat quality model

Biodiversity is the number and abundance of all species in an ecosystem (Angermeier 1994). Although biodiversity provides many established services to society, there is no consensus on how to give biodiversity a monetary value. For this reason, we followed the methodology of Tallis et al. 2011 and evaluated the benefits of biodiversity by looking at relative habitat quality scores.

Biodiversity was modeled using habitat quality as a proxy for biodiversity. Birds were chosen as our proxy for biodiversity because they are often a measure of ecosystem health (Naugle 1999) and it is not uncommon to use one species as an indicator species for ecosystem health (Caro 1999). The model creates an index value of habitat quality in order to compare biodiversity across scenarios instead of assigning a monetary value. Habitat quality was determined by labeling each grid cell as habitat, then adjusting the quality by the LULC in surrounding grid cells, and the sensitivity of each LULC in a grid cell to surrounding threats.

The model required the input of a table on threats to biodiversity and a table of each habitat type's sensitivity to each threat. The threat table (Table 3) was derived from causes of endangerment to American species found in Czech et al. (2000). Threats unrelated to our study (e.g. logging, military activity, vandalism, disease, etc.), redundant to other threats (e.g. pollution of water, air or soil is implied in agriculture, industrial activity is implied in urbanization, etc.), or unavailable within our LULC data (e.g. mineral, gas, oil, and geothermal extraction or exploration, etc.) were removed. The maximum distance over which the threat would be in effect and the relative weight of each threat to birds living in LIHD prairie habitat were determined from relevant literature (Trombulak 2000, Donald 2001, McKinney 2002, Brotons 2005).

The sensitivity table [*see appendix*] labels each LULC type as habitat or not, and then supplies the sensitivity of each LULC type as bird habitat to each threat relative to its sensitivity to other threats. Prairie, forest, and wetlands were labeled as habitat (1) because birds use these natural ecosystems for different habitat uses, such as breeding or foraging. Pasture grasslands were labeled as partial habitat (0.5) because prairie birds could use these lands if no natural prairie was available to them, but prefer prairie. Agricultural lands were labeled as poor habitat (0.1) because birds may use them in rare cases where no other land is available (Polasky 2011). Sensitivity score assignment was based on the methods of Polasky et al. (2011) and literature (Trombulak 2000, Donald 2001, McKinney 2002, Brotons 2005).

The output of the model is a quality sum index, which must be compared to another quality sum index derived from a different scenario to have meaning. For this reason, the biodiversity model was run using the same parameters at a resolution of 100 m each time, only changing the input scenarios.

Table 3. Biodiversity model threats input. A higher weight means that a threat classification is a greater threat to habitat quality, while maximum distance indicates the distance at which the threat ends.

Threat	Relative weight	Maximum distance (km)
Agriculture	1	10
Roads	0.6	2
Urbanization	1	10
Ranching	0.2	5

III. Results

Economic analysis

All models demonstrated that the CP scenario maximized both the overall value and the per-ha value of land-use change. Regardless of assumptions about the price of C, AC always resulted in a loss of ecosystem service value and AP always resulted in a gain. The CP scenario also always resulted in a gain, but its gains per ha were nearly six times greater than those of AP. The models indicated that the scenario with the strongest potential to compete with corn economically was CP (Table 4).

Economic analysis of CP without ecosystem services showed that prairie is less than half as profitable as corn (Table 5). With the ecosystem service values we modeled, CP still cannot compete with the current record high corn prices (Table 6). However, when the high price of C and high prairie feedstock price are compared to a mid-range (3.85 \$/bu) corn price from 2001 to 2011, prairie and corn have similar net profits per ha (Table 6). Furthermore, if a lower (1.85 \$/bu) price of corn is used, such as the <2 \$/bu prices seen in 2001, 2004, and 2005 (USDA 2002, 2005, 2006), prairie is more profitable under all C and cellulosic feedstock price scenarios we examined, even without ecosystem services taken into account. When a prairie feedstock price that is more representative of what a refinery is willing to pay is used – around \$30/Mg (Committee on... 2011) – prairie is still competitive in scenarios with the low corn price and C prices of \$81 or more.

TABLE 4. Modeled annual ecosystem service value per hectare.

Ecosystem Service	All Corn	All Prairie	Corn to Prairie
C sequestration*	-\$229.02	\$28.20	\$162.22
C emissions from N production*	-\$1.57	\$1.57	\$1.57
Nutrient retention	-\$81.80	\$5.78	\$35.10
Total value	-\$312.39	\$35.55	\$198.89

**C calculations used a price of \$81/tC.*

Table 5. Comparison of input costs and crop values of corn and prairie.

	Corn/Soy rotation	Prairie
Fertilizer input ($/ha/yr)	-$151.60	$0
Chemical input ($/ha/yr)	-$57.73	$0
Seed input ($/ha/yr)	-$176.14	-$81.60*
Gross profit ($/ha/yr)	$1,459.71	$490.00**
Net profit ($/ha/yr)	$1,074.25	$408.04

Notes: All dollar values are in 2010 constant dollars (2010 $). Prices of corn and soy from USDA 2011.
**Prairie establishment has a one-time cost of $3,264/ha, based on current commercial prairie establishment costs from Prairie Moon Nursery 2012. We divided this cost by a 40-year time horizon.*
***Profit per ha assumes an annual yield of 5 Mg/ha (Briggs 1995) and a willingness to accept feedstock price of $98/Mg (CEEIIBP 2011).*

Table 6. Comparison of corn net profit and LIHD prairie profit with ecosystem service values under varied cellulosic feedstock, C, and corn prices. Positive differences in profitability (DP)* between corn and prairie indicate price scenarios under which prairie could potentially be economically competitive.

Price per Mg prairie feedstock	Price per tC	Prairie ecosystem service value per hectare	Prairie net value per ha/ yr**	DP using high corn price	DP using mid-range corn price	DP using low corn price
$75.00	$36.00	$91.60	$385.00	-$1,067.23	-$455.45	$156.96
$98.00	$36.00	$91.60	$500.00	-$952.23	-$340.45	$271.96
$133.00	$36.00	$91.60	$675.00	-$777.23	-$165.45	$446.96
$75.00	$81.00	$162.22	$455.62	-$996.61	-$384.83	$227.96
$98.00	$81.00	$162.22	$570.62	-$881.61	-$269.83	$342.59
$133.00	$81.00	$162.22	$745.62	-$706.61	-$94.83	$517.59
$75.00	$134.00	$245.34	$538.80	-$913.43	-$301.65	$310.76
$98.00	$134.00	$245.34	$653.80	-$798.43	-$186.65	$425.76
$133.00	$134.00	$245.34	$828.80	-$623.43	-$11.65	$600.76

**Difference in profitability is the net value of prairie minus the net profit of corn for high ($1,452.23), mid-range ($840.45), and low ($228.04) prices.*
***Net value includes ecosystem services value and feedstock value.*

All Corn

The AC scenario was the only scenario that resulted in negative changes for all three models (Tables 7-9). The AC scenario would release 17.9 million tC, which is nearly half of the C currently stored in southern Minnesota. It would also result in an additional 4,800 tC emissions from fertilizer production emissions and a 23,000 t increase in N application.

All Prairie

Though prairie sequesters C and retains nutrients efficiently, the AP scenario did not maximize ecosystem service value in any of the three models (Tables 7-9). The AP scenario resulted in half the overall value of ecosystem services as CP. Additionally, because the AP scenario called for changing over 329,000 ha of land compared to 81,000 ha under CP, the per-ha value of this scenario was even less competitive.

Corn to Prairie

The CP scenario resulted in the highest overall ecosystem service value, and due to its relatively small land change, produced the highest per-ha values as well (Tables 7-9). It stored 5,027,590 tC, 13% more C stored than the Current scenario. The CP scenario achieved nutrient retention values seven times higher than the broader land-use change in AP because it minimized the number of hectares changed and targeted those likely to contribute to nutrient export.

TABLE 7. Annual C storage value per hectare of land changed from Current.

Price per tC	All Corn	All Prairie	Corn to Prairie
$36	-$48.81	$9.27	$55.80
$81	-$109.82	$20.86	$125.55
$134	-$181.68	$34.50	$207.70

TABLE 8. Change in N export from Current per hectare. Negative changes indicate increased ecosystem service value, with the exception of dollar values.

Land-use scenario	Change in N export (kg/ha/yr)	% change in export	Change in value of N export from per ha
All Corn	0.537	4.433	-$81.80
All Prairie	-0.038	-0.398	$5.78
Corn to Prairie	-0.230	-0.521	$35.10

Note: Change is the difference in export from Current and the alternative scenario divided by the number of hectares that underwent land use change.

Table 9. Biodiversity and habitat quality model results. A larger habitat quality score indicates a lower overall habitat quality in a given scenario.

Land-use scenario	Habitat quality score	% change in score from Current
Current	149,264	N/A
All Corn	153,758	2.93
All Prairie	143,267	-4.02
Corn to Prairie	141,564	-5.16

IV. Discussion

Using the InVEST model, we estimated the value of the ecosystem services of LIHD prairie grown as a biofuel feedstock and examined what price conditions would make it competitive with corn production. Out of our three scenarios, the CP scenario maximized all modeled ecosystem services. This was due to the combined ecosystem services of increased prairie area and the maintenance of multiple services already provided by existing land cover, such as forests and wetlands. This suggests that a total conversion of all marginal lands to prairie is not the best choice. When making land-use decisions in favor of biofuels, current LULC must be assessed in order to avoid the reduction of existing ecosystem services.

Our economic analysis comparing prairie's competitiveness with corn while taking into account ecosystem services revealed that prairie is only competitive with corn when the price of corn is low. However, our models only account for some of the value that these services provide because we do not monetize biodiversity, nor do we comprehensively take into account all ecological and economic aspects of LIHD prairie. Additionally, there are other ecosystem services, such as recreation, that the models do not consider. Prairie may therefore be more competitive with corn as a biofuel feedstock in other economic scenarios, such as those with a high price of prairie and a mid-range price of corn, if the comprehensive values of all ecosystem services are included.

Ecological uncertainty of ecosystem services

One reason our valuation of ecosystem services does not represent the true value of LIHD prairie is because of the ecological uncertainty in our values. An assumption that we made in all of the services we modeled was combining the original LULC classes into more generalized land-use types (i.e. *carrots*, *tomatoes*, and *oats* were all combined as "agriculture"). These pooled land-use types take on values that may not represent the true value for all original LULC classes present in these combined land-use types. For example, in the C sequestration model, we found one C storage value for each of the seven different pooled land-use types. This single value did not take into account the variation in C storage values among the original LULC classes that were combined into these pooled land-use classes.

Another inherent assumption in the InVEST model is the oversimplification of C and N cycles (Tallis 2011). In order to determine the rate of C sequestration, the model does not use actual sequestration rates, but rather divides the difference between the C pool sizes of two scenarios by the time horizon and assumes equal change each year. This could lead to an incorrect valuation of C if the pool accumulation is non-linear. For example, if C is sequestered at a rate that increases rapidly at first but then becomes saturated over time and we assume instead a linear rate of sequestration, then the C storage value will be underestimated in the short term. If C sequestration rates have a sigmoidal distribution and we assume linear, the C storage value will be overestimated in the short term. The N cycle is also simplified in the nutrient retention model because the model does not take into account any biogeochemical transformations (such as denitrification) that may occur while nutrients travel downstream.

The model also does not account for potential variation in prairie harvesting practices that may affect biodiversity (Fargione 2009). Considerations regarding harvest method include timing of harvest, prairie height at harvest time, and the amount of available vegetation that is harvested. For example, entire fields of prairie can be harvested at once, or prairie can be harvested in a patchwork mosaic that leaves some areas with taller grasses. Some bird species, such as the grasshopper sparrow (*Amodramus savannarum*) and Savannah sparrow (*Passerculus sandwichensis*), benefit most from shorter grasses while others, such as the sedge wren (*Cistothorus platensis*) and Henslow's sparrow (*Ammodramus henslowii*), prefer taller and denser vegetation (Fargione 2009). It is important to determine the harvest management practice that maximizes both yield and biodiversity and conservation aims. These oversights in the model could have had significant impacts on the biodiversity and habitat quality results.

Economic uncertainty of ecosystem services

We found that even when including ecosystem service value, prairie is not economically competitive with corn under current high corn prices. However, our economic analysis is likely an underestimate of prairie ecosystem service value due to assumptions made about the value of each service or omission of some services altogether. For example, we assumed three different prices for C, but it is possible that none of these prices equitably reflect the environmental costs associated with the impacts of increased C in the atmosphere. Estimates of the social cost of C have run as high as $1,602 per ton of C emitted (Tol 2011), and may be that high to account for the fact that different locations on Earth are predicted to see different impacts from climate change (Anthoff 2009). A small island nation has much more land at risk of inundation from sea level rise than an inland Midwestern state such as Minnesota. These small countries, often also poorer counties, may be less equipped to address these costs because $1 of damage is worth more to a poor nation than $1 of damage is worth to a wealthier nation (Pearce 2003).

Economic benefits due to reduced health care costs are not accurately accounted for in the models. Less biodiversity reduces opportunity for new medicinal gains, thus reducing overall healthcare costs (Alves 2007). Though the majority of medicinal discoveries have been in tropical areas (Gentry 1993, Balick 1996), it is possible that new medical treatments and drugs can be formulated out of prairie species. Diverse microbial communities may enhance the ability of crops to suppress pathogens, which could lead to discoveries of disease prevention in humans (Ghorbani 2008). Nitrate contamination of water is also a threat to human health, even at concentrations below the standards set by the EPA and the World Health Organized (Puckett 2011). This contamination is linked to diseases such as methaemoglobinaemia (Blue Baby syndrome), colon cancer, neural tube defects, and other reproductive problems (Wolfe 2002, Ward 2005).

Water contamination by sediment and nutrients affects the economic livelihoods of fishermen both locally and as far away as the Gulf of Mexico. Agricultural runoff into lakes and rivers causes hypoxic conditions that disrupt ecosystem functioning and negatively affect fish populations (Alexander 2008, Helsel 2011). Hypoxia has economic impacts on commercial fishing, the food service industry, the tourism industry, and recreational fishing (Downing 1999), because high nutrient concentrations cause reduced species diversity and smaller fish populations (Welle 2001). Commercial and recreational fishing in the Gulf generate a revenue of over $2.8 billion dollars annually, but declines in both fish and shrimp abundance and fishing efficiency since hypoxia has increased suggest that hypoxia impacts this revenue (Downing 1999).

One prairie ecosystem service that is not valued is recreation value. Although InVEST does not have a tool for modeling the recreational value of prairie, we argue this is an oversight and that prairies also provide the ecosystem service of recreation. Recreational value is derived from aesthetic appeal of the site and activities such as hiking, picnicking, camping, hunting, horseback riding, photography, biking, cross-country skiing, snowshoeing, and wildlife viewing (Sheyenne National Grasslands , Klenosky 2003, Forest Service 2008, Forest Service 2012). Recreational value is also derived indirectly from activities such as recreational boating, fishing, and swimming, which benefit from the water filtration services provided prairies. Although prairie grown as a cellulosic ethanol feedstock would not provide many of these direct recreational opportunities, it does contribute habitat for increased wildlife viewing, as well as aesthetic appeal (Moir 1972, Tews 2004).

Little research has been done on valuing prairie aesthetics (Moir 1972, Chenoweth 1990), however, one way to value recreation is by looking at park visitation rates in association with park use fees (Table 10). In 2009, the Midewin National Tallgrass Prairie in northwest Illinois estimated an annual visitation rate of 16.9 thousand people (Forest Service 2009). Although neither this prairie nor any other national grasslands or prairies charge an entrance fee, the privately-owned Spring Creek Prairie Audubon Center (SCPAC) near Lincoln, Nebraska charges $4 per adult visitor for daily admission. However, using park use fees does not fully account for consumers' WTP for prairie recreational access because they mainly account for park operational expenses. For example, the Lyndon B. Johnson National Grassland Group Campground states that their fee is used for maintenance and improvement of the facilities (Forest Service 2008).

TABLE 10. Fees at national prairies and grasslands.

Park	Entrance fee	Camping fees	Other fees
Tallgrass Prairie National Preserve (KS)	none	N/A	N/A
Midewin National Tallgrass Prairie (IL)	none	N/A	Special areas/events may be charged a fee
Sheyenne National Grassland (ND)	none	$6/night	N/A
Caddo-Lyndon B. Johnson National Grassland (TX)	none	$4/night at lake $150/weekend at LBJ campground	Maps: $9 Lake use: $2/day
Chickasaw National Recreation Area (OK)	none	$14 – $30/night	Boat daily permit: $4 Boat annual pass: $30 Picnic pavilions: $30/day

Sources: NPS 2012a, Forest Service 2012, Sheyenne National Grasslands, Forest Service 2008, NPS 2012b

A better method for valuing recreation is consumers' WTP for recreational activities and continued opportunities through club membership. A study by Klenosky et al. (2003) surveyed residents near Midewin about their WTP for park admittance. Survey respondents indicated a WTP of $5 per daily visit and $20-$25 for an annual pass (Klenosky 2003). When surveyed about their WTP for clean water for boating, fishing, and swimming, over 1,500 Americans reported a WTP of $93, $70, and $78, respectively (Clarson 1993). Organizations that specialize in prairie appreciation, recreation, and/or preservation, such as those listed in Table 11, charge escalating rates for membership, beginning as low as $8-$20 for students and going as high as $300-$1,500 for lifetime memberships. These values may more accurately represent the ecosystem service value of recreation because the direct beneficiaries of recreation – the park users – placed a value on the benefits they receive due to their use of the prairies.

TABLE 11. Example organizations dedicated to prairie appreciation, recreation, or preservation.

Organization name	Organization mission	Membership cost range*
Minnesota Native Plant Society	Conserve and raise awareness of native Minnesotan plants, including those found in prairies	\$8 – \$25
Grassland Heritage Foundation	Land trust organization that preserves, restores and educates about prairies in Kansas	\$15 – \$500
Friends of Konza Prairie	Support Konza Prairie, an 8,600-acre tallgrass prairie preserve in Kansas	\$30 – \$500+
Native Prairies Association of Texas	Land trust organization that protects, restores and increases appreciation of prairies in Texas and the US	\$20 – \$500
Spring Creek Prairie Audubon Center	Improve knowledge, appreciation, and conservation of tallgrass prairies in Nebraska	\$35 – \$60

**Levels of membership are variations of student, individual, family, and organization, and are on an annual basis. Some organizations also offer more expensive "lifetime" memberships.*
Sources: Minnesota Native Prairie Society 2012, Grassland Heritage Foundation, KEEP 2012, NPAT 2012, Audubon 2012

Ethical considerations surrounding ecosystem services

When choosing how to value the ecosystem services modeled by InVEST, we faced several ethical considerations. Our methods of valuing C lack ethical representation of all people and locations. We assume a price of C to be within a certain range, but people of different locations and economic statuses, such as inhabitants of a small island nation, may feel these values to be too conservative. Economists have suggested the use of equity weights to determine the correct social cost of C for a particular country. Equity weights are values derived from first determining the local impacts of increased C in the atmosphere within a country, then aggregating values for an entire country, and finally adjusting this national value based on the GDP of the country (Anthoff 2010). For example, a wealthy country may be equally affected by climate change as a poor country, but the poorer country would receive a higher equity weight to account for the more drastic societal impacts of climate change.

On the global scale, it is difficult to determine the social cost of C. Equity weights only work under a global welfare function, which is when there is one entity making the world's decisions (Anthoff 2010). Instead, Anthoff and Tol (2010) present a framework of four ways that national governments can decide on a social cost of C: 1) Sovereignty, or ignoring international impacts, 2) Altruism, or attempting to aid people abroad through climate policy, 3) Good neighbor, or using victims' discount rates out of guilt, and 4) Liability, or using one's own discount rate in an effort to compensate people abroad. If equity weighting is used within one country, it must be adopted consistently in order to fulfill the idea of liability, or the "polluter pays" principle (Pearce 2003).

Additionally, there is also issue in whether or not current generations have an obligation to future generations. This concept, termed intergenerational equity, means that future generations are entitled to a planet that will provide them with the same access to vital resources as current generations (Weiss 1990). We know that C emissions will have an impact on future generations, but depending on how strongly we value intergenerational equity, we may underestimate the monetary value of damages to future generations. If we take into account intergenerational equity when valuing C, we would have to value C more in order to prompt a reduction in emissions that would allow future generations to have conditions equal to current conditions (Weiss 1990).

The valuation of biodiversity is another ethical debate that may cause the undervaluation of prairie's ecosystem services. We discuss how value can be seen in the ecological services that biodiversity provides, like increased NPP, but value for biodiversity can also be derived through its intrinsic value**.** Therefore, biodiversity is potentially undervalued if its intrinsic value is not recognized. Ehrenfeld (1988) argues that monetizing biodiversity using economic terms undervalues it and even leads to unwise courses of action. He points to a study by Clark (1973) which found that it made more economic sense to kill off an entire blue whale population and reinvest the profits in industry rather than harvest the whales at a sustainable rate. A true cost-benefit analysis, such as that which was used in the Clark study, is the incorrect way to assess biodiversity value.

Ehrenfeld argues, "value is an intrinsic part of diversity" (1988). This value is not dependent on species-specific properties. It does not depend on the possible uses for these species or on their role in global ecosystems. Diversity, he argues, simply *is* value and the two are so intertwined as to be inseparable (Erhrenfeld 1988). Ghilarov (2000) agrees that biodiversity has intrinsic value and argues that ecologists should not have to experimentally find a special justification for biodiversity's scientific usefulness. Scientific efforts to quantitatively prove the value of biodiversity through monetary value and scientific data, he argues, are extraneous because humans already know that biodiversity has value. If society does choose to value biodiversity intrinsically, it would provide further motivations for switching from corn monocultures to high-diversity prairies.

Economic and policy implications

Although our study found that prairie is not an economically competitive ethanol feedstock in most market scenarios, circumstances do exist in which prairie could be economically competitive as a biofuel feedstock. The route taken to incorporate this ecosystem service value into policy depends on the interpretation of the externalities in this scenario. In order for society to benefit from the positive externalities of LIHD prairie, policy makers would need to encourage prairie cellulosic ethanol production. One way to do this is through subsidies to farmers. A subsidy would offset the lower net profits of prairie production, thus making it more profitable and attractive as a crop. The cost of the subsidies would be, at least in part, displaced by avoided costs to society such as avoided water treatment and avoided measures taken in response to climate change. Subsidies have successfully been applied to agriculture for biofuels in the past in the form of corn ethanol subsidies, which made corn ethanol competitive with petroleum-based fuels (Hill 2006). If a subsidy for LIHD prairie is too high, however, it could promote conversion of other ecosystems, such as wetlands and forests, to prairie. This possibility was seen in the AP scenario: though the model showed modest gains in ecosystem services over the Current scenario, it did not provide as many services as the scenario in which forests, wetlands, and other land covers were maintained.

Alternatively, a corn tax could be implemented to account for the negative externalities associated with corn production. This may encourage farmers to search out a crop with fewer taxes, but it would not explicitly promote prairie as an alternative crop. The only way to do this would be to put the tax revenue towards prairie expansion. Instead, farmers may choose to grow soybeans, switchgrass, sugar beets, or another profitable Minnesota crop (USDA 2011b). The best-case scenario, then, would be to take into account the positive externalities of growing prairie and the negative externalities of growing corn. This could take the form of a combination pro-prairie subsidy and anti-corn tax.

Other policy options exist that instead incentivize consumers to use cellulosic ethanol over corn ethanol or gasoline. Tax credits are one way to make cellulosic fuels more affordable for consumers. A tax credit could make the price of cellulosic ethanol equivalent to or even cheaper than its fossil fuel-based or corn-based counterparts. If the credit does not reduce the price of cellulosic fuels to at least equal that of gasoline, it would not achieve its goal of incentivizing biofuels. An indirect way to incentivize cellulosic fuels would be to provide consumer incentives, such as subsidies and tax credits, for purchasing flex-fuel vehicles that can run on cellulosic ethanol. This method is less direct, however, because flex-fuel vehicles designed to run on up to 100 percent cellulosic ethanol can also run on fuels containing any amount of gasoline or corn ethanol (DOE 2010).

Finally, if the U.S. government is going to create policies supporting LIHD prairie-based ethanol, it will need to provide financial support for the building the industry infrastructure. Currently, the federal government provides a corporate tax credit to build production equipment for renewable energy technologies through the Business Energy Investment Tax Credit (IRS 2012). Expanding this tax credit with an emphasis on cellulosic ethanol refineries would greatly benefit the expansion of this industry.

V. Conclusions

LIHD prairie cultivated for cellulosic ethanol has the potential to replace gasoline, provide the United States with the fuel it needs, and contribute important ecosystem services. We found that among three alternative 2010 LULC scenarios, a corn-to-prairie conversion on marginal lands provided the most ecosystem services. A full conversion of marginal lands to prairie may result in net losses of ecosystem services because LIHD prairie would be replacing other important ecosystems such as forests and wetlands. Though the CP scenario yielded the most overall ecosystem service value, it was only competitive in economic scenarios with low corn prices. Nevertheless, we argue that we were unable to fully capture the ecosystem service values we modeled, and that prairie provides other services, such as recreation, we were unable to quantify. Therefore, we conclude LIHD prairie grown on marginal lands in southern Minnesota has the potential to be economically competitive with corn as a biofuel feedstock if the value of all its ecosystem services is considered.

However, in order to fully understand the feasibility of utilizing LIHD prairie as a feedstock, more research is needed regarding its production and feasibility as a cellulosic ethanol feedstock. Large-scale, long-term field studies that directly mimic production-scale plots are necessary in order to determine the true rates of C sequestration over time, nutrient runoff, and yields on typical agricultural soils. These studies would allow for more accurate inputs to InVEST and could also be used to discover the optimal harvest method that maximizes both yield and ecosystem services. In addition to better field data, more realistic scenarios would improve the model results. For example, the LULC scenarios we used were alternative 2010s, not future LULC scenarios. Creating future land-use scenarios could provide more accurate and relevant predictions for policy makers.

Other ways of assessing the accuracy of InVEST processes could be completing a comparison of the model results to field data, or performing a sensitivity analysis. A sensitivity analysis would show the amount of uncertainty in the production and valuation of ecosystem services that is caused by uncertainty in the model and model inputs. Although more accurate model inputs could improve results, improvements in the model are still necessary to account for oversimplification of ecosystem processes.

Finally, more research needs to be done regarding the valuation of these services. A strong indicator of a service's anthropogenic value is consumers' WTP for each service. Surveys administered to southern Minnesota residents asking about their WTP for nutrient retention, biodiversity, recreation, and aesthetics would greatly enhance this study. Improved valuation of ecosystem services could also be achieved by using the current amount that people pay for these services.

Despite the fact that our society undervalues ecosystem services, we continue to take advantage of the environment in which we live and the benefits it provides. If biofuels are to expand in the future in order to meet the energy needs of the United States, this expansion must be done in a way that maximizes ecosystem services.

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Appendix

Table 1. Perennial native prairie species for LIHD prairie harvesting

Functional type	Species
C3 grass	Switchgrass (*Panicum virgatum*), Junegrass (*Koeleria cristata*), Wild Rye (*Elymus canadensi*), Western Wheatgrass (*Agropyron smithi*)
C4 grass	Big Bluestem (*Andropogon gerardi*), Little Bluestem (*Schizachyrium scoparium*), Indian Grass (*Sorghastrum nutans*), Common Yarrow (*Achillea millefolium*)
Legume	Lupine (*Lupinis perennis*), Bush Clover (*Lespedeza capitata*), Bergamot Plant (*Monarda fistulosa*)
Forb	Stiff Goldenrod (*Solidago rigida*), Kentucky Bluegrass (*Poa pratensis*), Purple Prairie Clover (*Petalostemum purpureum*), Blazing Star (*Liatris aspera*)
Woody legume	Leadplant (*Amorpha canescens*)
Woody	Burr Oak (*Quercus macrocarpa*), Pin Oak (*Quercus elipsoidalis*)

Adapted from Tilman 2006

Table 2. Carbon pool sources by LULC groupings

	Author(s)	Year	Location
LIHD prairie and grasslands	Breuer et al.	2006	Lahn-Dill Highlands, Germany
	Baer et al.	2002	CRP lands, Gage and Saline County, Nebraska, USA
	Brye et al.	2002	Arlington, Wisconsin, USA
	Kucharik	2007	CRP lands, Dane County, Wisconsin, USA
	Kucharik et al.	2006	Madison, Wisconsin, USA
	Camill et al.	2004	Northfield, Minnesota, USA
	Alster and Esch	2011	Northfield, Minnesota, USA
	Lal	2002	Midwest, USA
	Lal	2003	Texas, USA
Corn/soy and agriculture	Lal	2003	Global
Forests	Smith et al.	2005	Northern Prairie States, USA
	Weishampel et al.	2009	Northern Minnesota, USA
Wetlands	Bridgham et al.	2006	North America

Nutrient retention model assumptions

1) The model also does not recognize when flow paths are disrupted by tile-drainage and ditches. These alternative paths for agricultural runoff might create a more direct route for runoff into the local water source that is not reflected in the model. Field drainage is often used to improve growing conditions on lands that are not well-drained (Helsel11). The excess soil water is collected and funneled off the land and directly into streams, greatly influencing stream quality (Helsel11).

2) One input to the nutrient retention model that incurs some uncertainty in root depth. We assumed uniform root depth across each land-use type. Root depth plays a large role in determining nutrient retention, so this assumption may both over- and underestimate retention values because root depth varies on the species level, not the ecosystem level. This is not reflected in our numbers especially since we pooled different land-use types together and only used one root depth for each land-use type.

3) A problem in quantifying the impacts of nutrient inputs is not accounting for chemical interactions. The model assumes that the nutrients do not change form downstream through interactions with other chemicals after leaving a cell. Increases or decreases of nutrients caused by chemical reactions would result in over- or underestimation of retention value (Puckett11).

4) In order to use a WTP study to value the results of the nutrient retention model, we had to make several assumptions. First, the Matthews (2002) study was limited to the Minnesota River Valley (MRV). We extrapolated this to all of southern Minnesota. Though residents who live far away from the MRV likely would not value its water quality as highly as those that visit it often, we assumed that a reduction in N loading would be valued in Minnesota waterways other than the Minnesota River. Secondly, the study valued a reduction in P levels. We modeled N retention and assumed it would be valued the same as P, because overall, participants were rating water quality. Third, the study found the total value of a 40% reduction in P levels. Our models did not show a 40% reduction, but we assumed people would value water quality improvements linearly. Finally, the study valued a reduction P loading from 1997 levels, whereas we modeled 2010 N loading levels (Matthews 2002).

Table 3. Values of nutrient loading, root depth, and nutrient retention coefficients for use in nutrient retention model.

LULC classification	Evapotranspiration coefficient	Root depth (mm)*	Nutrient export coefficient	Vegetation filtering value**
Corn/soy	972	1000	16090	40
Forest	1056	2000	2860	60
Grasslands	812	1500	8650	50
Wetland	983	800	1	80
Agriculture	972	1000	16090	40
LIHD	812	1700	8650	50
Pasture	892	1500	12370	50
Open water	542	1	1	0
Barren	50	1	1430	5
Urban	1100	1	9970	0

**Value of 1 indicates non-vegetated LULC classes.*
***Value of 0 indicates LULC class has no filtering capacity.*
Source: Keeler, B. personal communication. University of Minnesota.

Biodiversity model assumptions

1) The habitat quality model has several assumptions in how threats are evaluated. The model assumes each threat to work independently of other threats. To combine the effects of two threats, it simply adds the threat values together. However, in reality, threats could influence each other to have a greater effect than just the sum of their impacts.

2) The model also asks for a value of threat decay – either exponential or linear. It does not allow for other possible decay patterns, such as logarithmic decay or any other non-standard decay, which would affect the severity of a threat in proximity to a habitat.
3) Decisions we made in the use of the model may have also affected our habitat quality model results. Our use of the biodiversity model includes common habitat threats to birds, but disregards effects to other organisms. We chose birds because they are a well-studied proxy of habitat biodiversity, but incorporating the impacts of threats to plant habitat quality may show different results.
4) Additionally, we acknowledge that not all possible threats are included, so we may have incorrectly estimated the value of prairie as a habitat.

Table 4. Biodiversity model sensitivity input values. Sensitivity values are the sensitivity of a given LULC classification to a threat relative to other LULC classes.

LULC classification	Habitat score	Sensitivity to agriculture	Sensitivity to roads	Sensitivity to urbanization	Sensitivity to ranching
Corn/Soy	0.1	0	0.1	0.1	0.1
Forest	1	0.6	0.4	0.7	0.2
Wetlands	1	0.9	0.9	0.9	0.4
Grassland	0.5	0.7	0.5	0.8	0.3
Agriculture	0	0	0	0	0
Other	0	0	0	0	0
LIHD	1	1	1	1	0.8