

Fueling the Future:

**Impacts of Climate Change and Land Use
on the Economic Viability of Second Generation Biofuels**

A Comprehensive Exercise

By

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1. Executive Summary

Climate change mitigation is becoming an increasingly important factor in energy policy, as atmospheric carbon dioxide levels have grown by 37% since the year 1850 (ESRL 2012). The United States has looked towards biofuels as a possible source of low-carbon energy. The focus in the past decade has been on corn ethanol, but this feedstock has been questioned in terms of both its actual environmental impact and its effect on food prices (Sohl and Sleeter 2012). Low-input high-diversity (LIHD) prairie is an alternative feedstock that is proposed to create cellulosic biofuels while avoiding these negative factors. Currently, costs for growing LIHD prairie are prohibitively high to allow for market penetration in corn-dominated Southern Minnesota.

The present study uses geospatial modeling to compare prices of corn and LIHD feedstocks in the year 2050 when ecosystem services, in addition to direct market value, are included. The study makes use of two land cover scenarios: one in which all marginal land currently planted in corn remains in corn, and another in which corn currently planted on marginal land is converted to LIHD prairie. To determine the possible effects of climate change on feedstock production, we ran each land cover scenario for two different climate scenarios. The first climate scenario corresponds to a world in which carbon emissions remain essentially unregulated, so climate change effects are relatively extreme. The second climate scenario corresponds to a world where nations cooperate to minimize emissions, resulting in only modest climate change effects.

The four combinations of climate and land use scenarios were used as inputs for the InVEST ecosystem service model to quantify each of four services: carbon sequestration, phosphorus retention, sediment retention, and biodiversity. With the exception of biodiversity, the outputs of the InVEST models were then converted into dollar values per hectare per year. This calculation was done within InVEST for carbon sequestration, and done by hand using established methods in the literature for phosphorus and sediment retention. Low, medium, and high values of direct harvest price for both corn and LIHD prairie were found, and total per hectare value for each crop was calculated for each climate scenario.

For the B1 (optimistic) scenario, we found that the annual per hectare value of LIHD prairie exceeded that of corn by between \$391 and \$642 for low future prices of corn. For the A2 (business as usual) scenario, the value of LIHD prairie exceeded that of corn by between \$161 and \$412 for low corn prices. However, the per hectare value of corn exceeded that of LIHD prairie for medium or high future corn prices. These results suggest that LIHD prairie may be viable on the biofuels market in 2050 depending both on market supply and demand, and government incentive structures.

The ocean of grass... is no longer what it was...

- John Graves, Introduction to Texas Sky

2. Introduction: Impacts of Climate Change and Land Use on the Economic Viability of Second Generation Biofuels

Energy production and climate change are perhaps the two most significant environmental challenges that society faces on both global and regional scales. As fossil fuel harvesting continues and emissions build in the atmospheric carbon pool, which has grown to 395 parts per million from 288 in 1850, it is clear that these issues will become even more pressing in the coming years and decades (ESRL 2013).

Through the 2007 Revised Renewable Fuel Standards, the U.S. has ruled that biofuels must become a part of the solution to these problems, mandating 36 billion gallons of biofuels by 2022, about triple current production levels (EIA 2012). Biofuels are becoming an increasingly large share of land use in agricultural areas of the Upper Midwest, with about 40% of the Minnesota corn crop going towards ethanol in 2012 (EIA 2012), but the politics and environmental implications associated with the current push for corn ethanol in the U.S. remain highly controversial (Sohl and Sleeter 2012). Thus, scholars are beginning to investigate the ecologic and economic implications of growing alternative cellulosic feedstocks such as low-input/high-diversity prairie, rather than corn. For example, Gelfand et al. (2013) found that successional prairie vegetation grown on marginal lands can provide triple the carbon sequestration rates of corn, while maintaining similar rates of energy production. This study does not convert these processes into dollar values, making it difficult to say whether the noted advantages to prairie make it an economically viable competitor to corn.

Polasky et al. (2010) incorporated dollar values into their study and modeled a range of possible land cover scenarios using a conservative carbon cost estimate of \$42.32 per ton. They found that significantly expanding corn agriculture resulted in high direct returns to farmers, but this scenario had the lowest total net benefits of any of the six scenarios run in their model. This study highlights the need for refined estimates of ecosystem services, so that social benefits will not be sacrificed when policy makers and land managers decide how to utilize landscapes.

Heavenrich et al. (2012) found that converting corn grown on marginal lands to prairie in Southern Minnesota generated positive net benefits when the cost of corn was low (\$1.85 per bushel) and the price of carbon was high (>\$81 per ton). This analysis was conducted for present day conditions, which is problematic since cellulosic biofuels are not likely to reach widespread production for several years. For this reason, this analysis should be reconsidered in the context of future climate and land cover patterns.

The present study seeks to refine the methodologies employed by Polasky and Heavenrich by estimating total value of different land cover types by using future predicted scenarios of climate change and land use for the year 2050 in attempt to answer the following question: How does future uncertainty impact the economic competitiveness of different biofuel feedstocks when ecosystems services are valued in dollars?

The remainder of the introduction section outlines the research that has been completed relating to the effects of climate change on various biofuel feedstocks. We will first review the

broad-scale issues of climate change and energy security and place biofuels into the context of these discussions. Next, we will discuss the ecosystem services relevant to biofuel production. Finally, we will examine incorporating climate change into our valuation of ecosystem services.

2.A. Problems: Climate Change and Energy

Since the Industrial Revolution, society's entrenched dependence on fossil fuels such as coal and petroleum has had many far-reaching consequences, not only ecological and economic, but also related to human health and energy security. There is extensive evidence that Earth's climate has warmed in the past 100 years (Walther et al. 2002, Lindzen 1997, Baum et al. 2012). As of 2002, the Earth had warmed by an average of approximately 0.6 degrees Celsius, leading to ecological problems including species invasion, decreases in snow cover and ice extent, and earlier spring activities of organisms (Walther et al. 2002).

In addition to climate change, concerns about national energy security have fueled the debate on decreasing the use of petroleum-based fuel sources. High energy prices, increasing energy imports, peak oil predicted by 2050, and greater understanding of the environmental consequences of fossil fuel use have peaked interest in possible biofuel alternatives to gasoline (Hill et al. 2006, Bardi 2009). Corn-based ethanol in particular has become a growing sector of the U.S. energy economy, following standards set in the US EPA's Clean Air Act, Energy Policy Act of 2005, and the Revised Renewable Fuel Standard.

Researchers have raised concerns about multiple aspects of corn ethanol compared to other biofuels, including: Net Energy Balance (NEB), environmental impacts, greenhouse gas (GHG) emissions, economic competitiveness, and land available for production (Hill et al. 2006). Hill et al. (2006) found that corn ethanol production resulted in a NEB of 25%, meaning that the amount of energy contained in corn ethanol is greater than the amount of energy required for production. However, this NEB is lower than that of other biofuels such as soybean biodiesel which had a NEB of 0.73 Megajoules (MJ) whereas corn grain ethanol had a NEB of 0.20 MJ (Hill et al. 2006). This is because the refining process requires high fossil fuel energy input to separate the ethanol from the water solution (~8% ethanol/92% water) created in production (Pimental & Patzek 2005).

In addition, corn ethanol has a host of associated negative environmental impacts. These include the leaching and surface flow of agricultural chemicals such as nitrogen, phosphorus, and pesticides, leading to eutrophication (an excess of nutrients), loss of biodiversity, and elevated nitrate and nitrite levels in drinking water (Hill et al. 2006, Pimental & Patzek 2005). Corn ethanol requires greater quantities of these chemicals compared to other types of biofuel. Additionally, even though corn ethanol technologies are much less petroleum-intensive than gasoline, they have similar GHG emissions (Farrell et al. 2006), releasing 88% of net GHG emissions compared to an energy equivalent amount of gasoline (Hill et al. 2006). Policies reducing environmental externalities in the agricultural sector have the potential to significantly improve environmental performance of corn ethanol (Farrell et al. 2006).

Along with these environmental concerns, Hill et al. (2006) determined that corn ethanol was not economically competitive with petroleum fuels without subsidies. In 2005, ethanol net

production cost was \$0.46 per energy equivalent liter (EEL) of gasoline while wholesale gasoline prices averaged \$0.44/liter (Hill et al. 2006). Hill et al. (2006), however, noted that although not cost-competitive, biofuel production might be profitable because the U.S. federal government subsidized \$0.20 per EEL of ethanol. Though these direct-payments expired in 2011, ethanol producers now receive indirect subsidies through mandates in the Revised Renewable Fuel Standards. This legislation, renewed by President Obama in 2010, artificially increases demand for corn ethanol, which has contributed to a near tripling of corn prices compared to pre-mandate levels (Smith 2012). In 2010, 40% of corn harvested in the US was used to make ethanol, amounting to 9.3% of gasoline usage (EIA 2012). Greater rates of corn ethanol production may be difficult to attain, because corn is a major contributor to human food supplies via direct consumption and livestock feed. It is clear that due to these problems, the pursuit of ethanol production on a large scale will require other technologies, such as cellulosic ethanol (Farrell et al. 2006). In the United States, switchgrass, crop residue, and woody biomass from forest residue are being considered as second-generation biofuel feedstocks (USDA 2010).

The United States Department of Energy has identified switchgrass, in particular, as a promising second-generation biofuel candidate, given its ability to produce high yields in a range of geographic regions (Hartman et al. 2011; Wang et al. 2010), including marginal lands. However, drawbacks of switchgrass are that it is not currently economically competitive with corn-based ethanol (when ecosystem services are not taken into account) and that it produces highest yields when grown in monoculture, which is susceptible to invasion by non-native species, disease and pest outbreaks (Hartman et al. 2011). Researchers have also proposed the possibility of growing high diversity warm season grass mixes as a biofuel feedstock (Tilman et al. 2006, Wang et al. 2010, Webster et al. 2010). As defined by Tillman et al. (2006) Like switchgrass, these low-input high-diversity (LIHD) prairie mixes have environmental benefits such as carbon sequestration, providing wildlife habitat, and ability to be grown on marginal, erodible lands (Tilman et al. 2006). In addition, LIHD prairie can increase soil fertility, maintain biodiversity and provide habitat to a greater variety of wildlife, and provide aesthetic and recreational value (Hartman et al. 2011, Tilman et al. 2006). LIHD prairie has the added advantage of requiring fewer fertilizer and pesticide inputs (Tilman et al. 2006; Webster et al. 2010).

Tilman et al. (2006) and Wang et al. (2010) found that LIHD prairie feedstock is able to overcome nutrient limitations without high levels of fertilization inputs, through the inclusion of nitrogen-fixing legumes in the mixture of species. Additionally, Wang et al. (2010) found that while water availability impacted yields in switchgrass monoculture, no yield decline was observed when switchgrass was grown in a mixture with other species, suggesting that LIHD prairie mixes may be more resistant to fluctuating precipitation conditions relative to switchgrass. When grown on marginal lands, LIHD prairie produced higher yields than switchgrass monoculture (Tilman et al. 2006). However, like switchgrass, it is not currently economically competitive with corn ethanol. The studies cited here have only taken into account direct

production costs, failing to recognize ecosystem services and functions LIHD prairie or switchgrass provide to society.

2. B. The valuation of ecosystem services

Ecosystem services refer to the benefits that humans derive from natural processes, such as water purification, climate regulation, or recreation opportunities (Millenium Ecosystem Assessment, 2005). Thus, ecosystem services valuation is a useful tool to integrate ecological and economic dynamics although it is subject to various complications.

One such problem is that benefits from ecosystem services may be overlooked in cost-benefit analyses conducted by governments or industry. While these ecosystem services have a social benefit, it is difficult to directly value them, leading to the inefficient degradation of natural resources and processes. Although certain ecosystem services are difficult to quantify, their valuation provides an avenue to incorporate conservation into economic decisions. For example, arthropod predators provide pest control for crops by reducing the need for pesticide application and genetic resistance research. It has been estimated that these natural services are worth \$54 billion annually worldwide, and thus decisions affecting the habitats of these predators must consider how land use change might affect the value of this service (Naylor and Ehrlich, 1997). Researchers have achieved varying success rates in valuing ecosystem services through methods such as contingent valuation, travel costs, and hedonic modeling (Chee, 2004). These specific ideas have been catalogued by several sources, and monetary values have been assigned to many ecosystem services through these techniques.

The following sections outline four of the ecosystem services relevant to the production of first- or second-generation biofuels, including carbon sequestration, biodiversity, and soil/nutrient retention.

2.B.i. Carbon Sequestration

In the context of climate change, carbon sequestration is perhaps the most economically valuable ecosystem service, and the one that is most relevant in the fight against climate change. Several studies have attempted to quantify the amount of carbon being sequestered, and a potential value for that carbon using different geospatial models such as InVEST and FLUX (Johnson et al. 2011, Johnson et al. 2012, Nelson et al. 2009) which combine the calculations of carbon sequestration and valuation. By quantifying carbon sequestration with a tool like InVEST, one can make more effective, efficient, and defensible resource decisions, by taking land use change into account. In order to value carbon sequestration, the price of carbon must be defined, so different ecosystems and land use changes can be compared under the same methodology. One way to do this is to compute the social cost of carbon.

When considering the economic efficacy of different biofuel feedstocks, it is important to consider the varying carbon outputs from each feedstock. One of the ways to integrate all social and economic externalities associated with CO₂ (and other GHG) emissions is using the Social Cost of Carbon (SCC): the discounted value of the additional social costs to current and future society for each ton of carbon released into the atmosphere (Hope and Newbery, 2007). Johnson

and Hope (2012), along with Stern (2007) and Greenstone et al. (2007) have all indicated that SCC is a well-supported method of valuing carbon emissions, although there remains an ongoing debate about exactly what SCC value to use. This confusion occurs because of different calculation methodologies for SCC, which are highly influenced by a large range of factors including assumptions about future economic development, the range and likelihood of social and economic damages from climate change, and the discount rate to apply to such damages (Hope and Newbery, 2007). SCC is heavily influenced by climate sensitivity, the temperature rise from a doubling of CO₂ concentration in the atmosphere, and pure time preference rate (the discount rate that a person experiences assuming no expectation of changing circumstances) (Hope and Newbery, 2007).

2.B.ii. Sediment and nutrient retention

It is harder to determine the value of nutrient or soil retention than many other ecosystem services, because each unit of improvement may only affect a specific area, and these effects vary widely within a spatial context (Keeler et al. 2012). Often water quality improvements are used as indicators of soil and nutrient retention, because as more soil and nutrients are washed away from cropland or grassland the quality of water decreases dramatically. Because nutrient and sediment retention drastically affects local areas, there are different frameworks established to value it.

One method of valuing sediment and nutrient retention is to value the end-use services, utilized directly by humans. To measure the value of sediment retention, the researcher may examine avoided costs (such as avoided flood damages), travel costs expended to get to a recreational site, replacement costs (such as the costs of dredging reservoirs), or damage sustained by a certain industry (such as the effect of sedimentation on freshwater fisheries) (Hansen and Ribaudo 2008). One study reviewed costs (such as for dredging, or treating water) avoided by natural water filtration (Wilson and Carpenter 1999) and found values of \$730,000 and \$25,000,000, indicating that large monetary benefits could be derived from nutrient and sediment retention.

Contingent valuation is another commonly utilized method of valuing water quality-related ecosystem services. Generally when looking at freshwater ecosystems, phosphorus is the nutrient that is of most concern to policy makers as it leads to eutrophication. Mathews (2002) estimated household WTP for 40% reductions in phosphorus in the Minnesota River Basin to be \$154, \$131, and \$60, which they then multiplied by the number of people living in the area, to determine community WTP as an indication of nutrient retention value.

2.B.iii. Biodiversity

Biodiversity is a measure of biological heterogeneity that can be examined at scales ranging from individuals to regions. It includes characteristics such as species richness (number and abundance of species), and genetic variation within a population and between populations of the same species (Walker 1992). In addition to carbon sequestration, sediment and nutrient retention, examination of biodiversity is integral to an assessment of first and second generation

biofuels although biodiversity is difficult to capture in economic terms. Therefore, it is particularly vulnerable to threats posed by biofuel expansion (Fargione et al. 2009). In addition, many researchers are concerned that corn-based biofuel expansion will cause conversion of native ecosystems to cropland, thus threatening landscape biodiversity (Gasparatos et al. 2011). Biodiversity is vastly important because it provides a number of benefits. For example, insect species diversity ensures yearly pollination of crops (Werling et al. 2011). The presence of diverse grassland ecosystems near agricultural fields has been shown to decrease the number of crop pests because they provide habitat for predatory insects that prey on pest species (Meehan 2011). Plant species diversity can also reduce the risk of disease or invasion of non-native species (Hartman et al. 2011).

Much of the research surrounding the impact of biofuels on biodiversity focuses on birds because they are a management priority in grassland ecosystems (Fargione et al. 2009). Land use change has already caused a substantial decrease in bird populations over the past century (Meehan et al. 2010). Furthermore, declining enrollment in the USDA Conservation Reserve Program (CRP) and agricultural expansion continue to threaten bird habitat (Fargione et al. 2009). After peaking at 14.9 million hectares (ha) in September 2007, the amount of land enrolled in the CRP dropped by 931,000 ha by October 2007 while demand for land to grow corn ethanol increased by 4.9 million ha between 2005 and 2008 (Fargione et al. 2009). LIHD prairie feedstock has the potential to provide habitats for rare and declining birds (Fargione et al. 2009; Meehan et al. 2010; Tilman et al. 2006) in addition to increasing overall species richness (Meehan et al. 2010). The time, method of harvest, and species grown will impact the type of birds that may utilize LIHD prairie as habitat. This is because some birds prefer short stubbly grasses (or a LIHD field which has been harvested) and others prefer tall, long grasses (Fargione et al. 2009).

2.C. Incorporating Future Climate into the Valuation of Ecosystem Services

In order to accurately assess whether cellulosic feedstocks can be competitive with corn or other biofuel feedstocks, climate change must be taken into account. Regional impacts of climate change will vary throughout the United States. Thus, the impact of climate change on biofuel feedstock yields will differ depending on the crop and what region it is grown in. Under the current emissions scenario, temperatures in the Midwest are expected to rise 3-8 degrees Celsius by the end of the century (USGCRP 2009). Additionally, precipitation is projected to increase in the winter and spring, and drought frequency is projected to increase in the summer, leading to both flooding and water scarcity (USGCRP 2009). While longer growing season and carbon fertilization may increase yields of certain crops such as soybeans, increased temperatures may shorten maturation time of crops, a factor that may decrease yields in corn (Hatfield et al. 2008, USGCRP 2009), although this will vary by region. Warmer temperatures also exacerbate pest and weed problems, creating the need for additional fertilizer and pesticide inputs.

Although much research has been done on the impacts of climate change on food crops, studies on the impacts of climate change on cellulosic biofuel crops remain scarce. Research

finds that corn may be more susceptible to climate change than switchgrass and prairie. Hatfield et al. (2008) considered the effects of increased temperature and increased atmospheric CO₂ on irrigated corn, and projected a yield decrease of 3% and an evapotranspiration increase of 1.8%, indicating higher water use in the Midwest. When considering the effects of altered temperature, precipitation, and atmospheric CO₂, Brown et al. (2000) projected a maximum decrease of 1.5 Mg/ha in corn yields in Iowa and Missouri. In the same study, Brown et al. (2000) predicted that switchgrass yields will increase by an average of 5 Mg/ha across the Iowa, Kansas, Missouri, Nebraska region. Switchgrass has also been found to be tolerant to drought and flooding conditions for short periods of time (Hartman et al. 2011), and models have predicted that regional climate will become more suitable overall for switchgrass and less suitable for corn (O'neal et al. 2005). However, some researchers (Le et al. 2011, Oliver et al. 2009) caution that prairie ecosystems are limited by water availability, and their water use efficiency may decrease under climate change conditions.

In addition to climate change, future land cover in the United States will further affect which biofuel feedstock will be most economically competitive. Factors such as population growth, increasing urbanization, and agricultural expansion impact future land use (Sohl and Sleeter 2012). The competition between bioenergy crops and food crops for land resources highlights the importance of considering if land resources are available for growing biofuel feedstock (Davis et al. 2011, Gasparatos et al. 2011). Additionally, Searchinger et al. (2008), Gaspartos et al. (2011), and Johnson et al. (2010) find that converting native habitats to agriculture land for biofuel production incur significant environmental, economic, and carbon costs. Performing a spatially explicit analysis allows for the optimization of agricultural resources and an accurate valuation of ecosystem services (Davis et al. 2011). Thus, consideration of future land cover scenarios, in addition to changes in climate, is central to an accurate assessment of the economic competitiveness of different feedstocks.

3. Methods

In order to answer our research question, we valued three ecosystem services for a total of four climate and land-use scenarios. In addition, we calculated a biodiversity habitat quality index but did not include this in our economic analysis. A discussion of the biodiversity model can be found in the appendix. The ecosystem services we considered were carbon sequestration, nutrient retention, and sediment retention. In order to account for different possible futures, we considered two drivers of change: climate and land-use. To account for land-use, we modified 2050 Land-Use Land-Cover (LULC) data based on our two feedstock scenarios and to account for climate change, we used Educational Global Climate Modeling (EdGCM) software to generate future precipitation and evapotranspiration predictions for our two climate scenarios. These climate inputs were then used to account for climate change in our biophysical measure of ecosystem services.

To quantify biophysical magnitude of ecosystem services, we used the Integrated Valuation of Environmental Services and Tradeoff (InVEST) model. We choose to use the InVEST model because it produces spatially-explicit biophysical measures of a variety of ecosystem services that are quantified in units to which an economic value can be applied. We hope that using InVEST will bridge the gap between the complex ecological patterns and processes traditionally studied by ecologists and end-use service values required by economists. The InVEST model for each ecosystem service was run four times, corresponding to four climate-land use scenarios (A2 corn, A2 prairie, B1 corn, B1 prairie). The biophysical outputs of the InVEST model were modified based on various valuation techniques, described in the economic valuation section. The net profits of corn and prairie were both computed for high, medium, and low price scenarios. Finally, the net profits of corn and prairie and the total ecosystem service values were summed to produce a per hectare estimate of the net profits of corn and prairie feedstocks, when ecosystem services are considered. This methodology is described in more detail below (Figure 1).

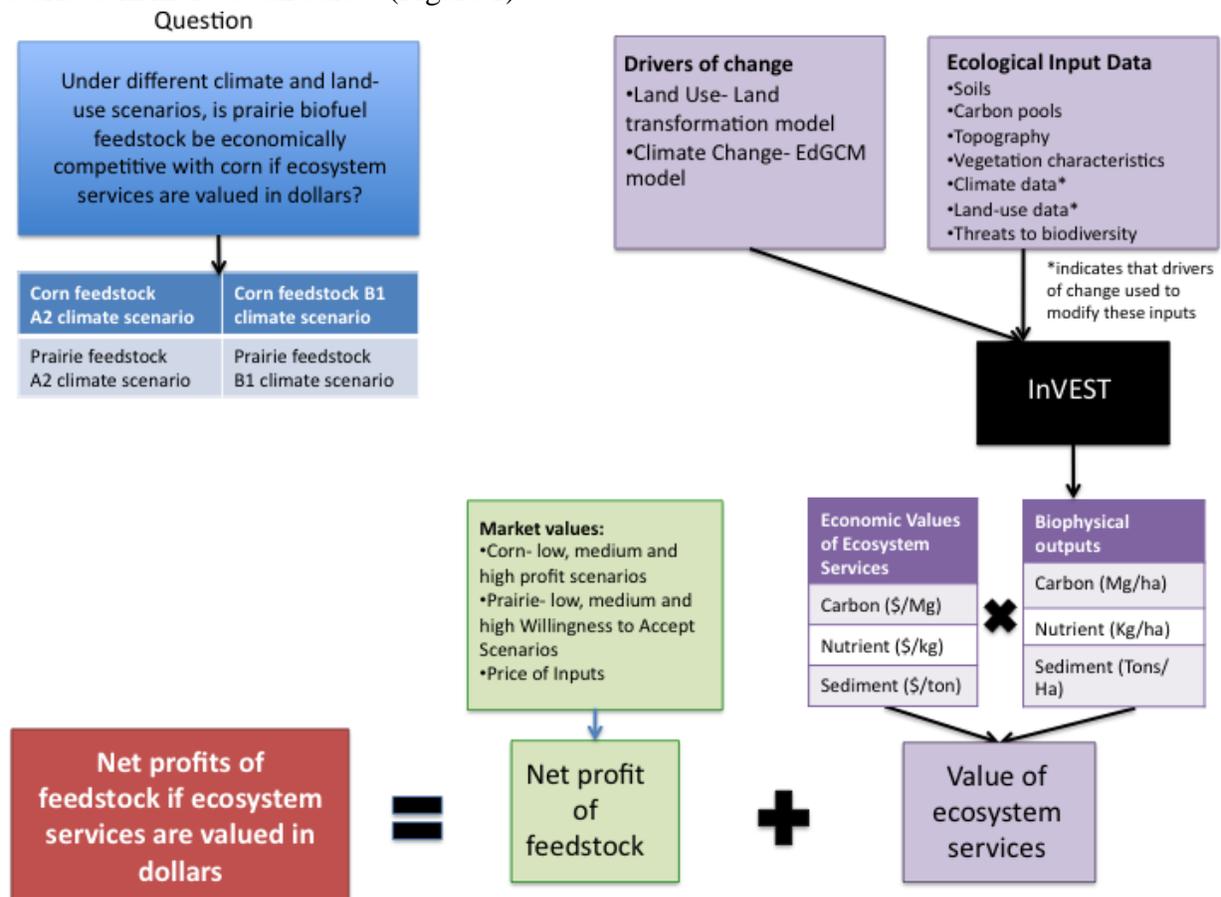


Figure 1. A graphical representation of our methods. This figure displays the steps that we took to calculate the value of three ecosystem services (displayed in purple) and the steps we took to calculate net profits of corn and prairie feedstock (displayed in green). These two values were summed to calculate the net profits of corn and prairie stock if ecosystem services are valued in dollar. This helps answer our research question (displayed in blue).

3.A. Study Area

Our study focused on 48 counties in southern Minnesota (Figure 2). This region was selected because all counties in this region were designated as major corn-growing areas from 2006-2010 (EIA 2012b). Seventy-five percent of the corn grown in the United States is grown within areas designated as major corn growing areas (Figure 3). Because biofuels in the United States are produced predominantly from corn ethanol, these counties are most likely to be impacted by policy decisions regarding biofuel feedstocks. We examined major corn growing regions in Minnesota specifically, because Minnesota is the fifth largest producer of corn ethanol in the nation (Ye 2012). Minnesota also has the highest sales of E85 (fuel that is 85% ethanol, 15% gasoline) in the country (EIA 2012a). Furthermore, Minnesota biofuel production is only projected to increase, with Minnesota being the first state to mandate 20% ethanol content in ethanol-gasoline blends by 2015 (EIA 2012a). This area also includes Carleton College.



Figure 2. Our 48 county study area in Southern MN

U.S. Corn Areas Experiencing Drought

Reflects August 28, 2012
U.S. Drought Monitor data

Approximately 85% of the corn grown in the U.S.
is within an area experiencing drought, based on
historical NASS crop production data.

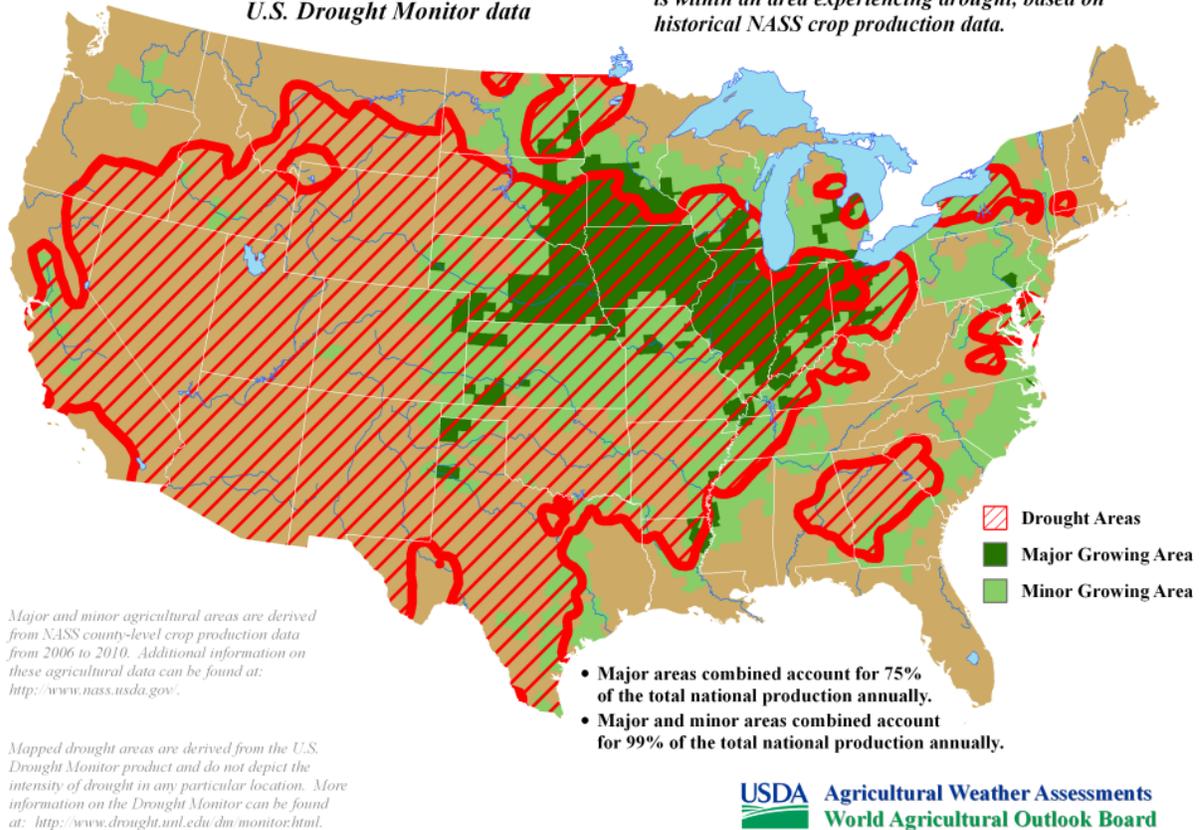


Figure 3. Major US corn growing regions, represented in dark green (EIA 2012).

3.B. Scenarios

For the two land-use scenarios, corn and prairie, we assumed that biofuel feedstock is grown on marginal lands in agriculture. We defined marginal lands using Nonirrigated Land Capability Classifications (LCC), a metric developed by the National Resources Conservation Service (NRCS) to rate soil suitability for agriculture and grazing. We chose to designate LCC classes 3 and 4 as marginal lands, because these LCCs are considered arable farmland, but have severe to very severe limitations for agricultural use (Gelfand et al. 2013). In the first land-use scenario, marginal lands in corn were assumed to remain in corn for biofuel production. In the second land-use scenario, marginal lands in corn, an area of 1,135,990 hectares, were converted to LIHD prairie.

Each land cover scenario was run for two climate scenarios, corresponding to the IPCC A2 and B1 climate scenarios. The various IPCC scenarios are based principally on two factors. The first factor is the focus of future development, which may either be premised solely on economic growth or may also incorporate elements of environmental health. The second factor is the degree to which development will be a globally integrated process with international cooperation, versus a regional process with nations independently pursuing different strategies.

The best-case scenario, which would minimize emissions and thus the adverse effects of climate change, is the B1 scenario, depicting a world where nations coordinate development strategies while placing great importance on environmental health. A2 is the worst-case scenario, portraying a world where nations do not work together, and economic growth is the only goal of the development process. These two extreme scenarios will be run for our models in order to provide upper and lower bounds to the model outputs we can expect in 2050 (IPCC 2001).

3.C. Drivers of change: climate and land-use

In order to accurately assess the future economic viability of corn and prairie biofuel feedstocks, we considered the impacts of two drivers of change: climate and land-use.

3.C.i. Land Use -- Land Transformation Model

The 2050 land-cover data for our study area was generated by Human-Environment Modeling and Analysis Laboratory (HEMA 2013) using the Land Transformation Model (LTM). The data predicts expansion of corn agriculture in response to projected ethanol demand, using the National Land Cover Data from 2001 as a baseline. Cells are converted to corn agriculture based on slope and proximity to ethanol plants. Areas predicted to be impacted by urban expansion (based on US Census populations, slope and distance to existing urban areas, roads, highways, and streams) are excluded from potential agricultural land.

3.C.ii. Climate Change -- Educational Global Climate Model (EdGCM)

In order to generate future climate predictions, we used the NASA EdGCM (Educational Global Climate Model) climate modeling software for each of our climate scenarios. EdGCM is a coarse-resolution general circulation model that predicts climate variables based on a variety of inputs, ranging from insolation to fluxes into the atmosphere of different greenhouse gases. We ran EdGCM two times through the year 2050, with annual emissions patterns (CO₂, CH₄, N₂O) corresponding to the predictions of the A2 and B1 scenarios. For each scenario, we gathered precipitation and evapotranspiration data for use in our InVEST sediment and retention models.

In order to incorporate the climate scenarios into the nutrient retention model, the -1.15 mm/day reduction in potential evapotranspiration was subtracted from our current evapotranspiration layer, obtained from the MODIS satellite. A change in precipitation was not predicted for either scenario, so this variable was not modified.

In the sediment model, the climate scenarios are captured by the rainfall erosivity factor (the R-factor), which is calculated based on the storm energy (the amount of rain that falls) multiplied by the maximum 30-minute intensity (how fast the rain falls) (Renard et al. 1997). Increases in intense precipitation events are expected to contribute to increased soil erosion to a greater degree than an increase in the number of wet days (Berc et al. 2003), so the R-factor is an appropriate factor to modify in order to account for climate change. However, climate model projections do not predict these detailed parameters. In order to overcome this limitation, we used a statistical relationship between average annual precipitation and the R-factor developed based on erosivity 155 locations in the United States (Renard and Freimund 1994).

The EdGCM climate model projections did not predict any increase in average annual precipitation for either IPCC scenario. Despite this apparent lack of change, the distribution of rainfall over the course of the year is expected to change. Precipitation in the Midwest is expected to increase in amount and intensity in the winter and spring, and decrease in the summer (USGCRP 2009). Given that climate change model predictions are less certain for precipitation than they are for temperature (NCADAC 2013) we decided to perform a sensitivity analysis on the R-factor input of our model. We modified our R-factors assuming a -10%, 0%, and 10% increase in average annual precipitation. However, given that neither of our scenarios predicted a change in precipitation, we used the scenario assuming a 0% increase in precipitation in our final calculation of ecosystem service provision of corn and prairie feedstocks.

3.D. InVEST: Biophysical Models

We used InVEST to quantify provision of carbon sequestration, nutrient retention and sediment retention by corn and prairie feedstocks. Two of these models (nutrient and sediment retention) were modified to account for the impacts of climate change. All models were impacted by changing land-use. The models were run at 30 meter (m) resolution.

3.D.i. Carbon Sequestration

In the InVEST carbon model, carbon sequestration was a measure of the amount of carbon removed from the atmosphere and stored in four carbon pools, aboveground biomass, belowground biomass, soil carbon, and litter carbon, for each land cover type (Table 1). Pool values were determined based on carbon pool data from Southern Minnesota (Polasky 2011). This pool data was aggregated from other literature, ensuring that the values are the same relative to each other. Because carbon is released in the creation of fertilizer, which InVEST did not take into account, we modified the carbon stored by corn/agriculture by subtracting the amount of carbon generated per hectare of fertilizer, using fertilizer recommendations from Rehm et al. (2006) and the amount of carbon released in its production as determined from Wood and Cowie (2004), in total this amounts to a .5 Mg release of carbon, which was subtracted from the value of the corn agriculture pool.

Table 1. Carbon sequestered (Mg/ha) by a stand of average age sorted by LUCL classification

LULC	Carbon Sequestered (Mg/ha)
Water	0
Developed/open space/etc	100
Forest	248
Grassland	112.3
Pasture	84.3
Agriculture/Corn	67.2
Wetlands	123.8
LIHD Prairie	100

All Values from Polasky except for Agriculture/Corn which was modified to address fertilizer

The InVEST model used the carbon pool data to calculate the total amount of carbon stored by each land-use scenario. Carbon sequestration was then measured as the difference in total carbon stored by the two land-use scenarios over a certain period of time. We chose to use a 40-year time step because prairie sequesters carbon for 40 years before the ecosystem reaches a state of equilibrium, where no additional biomass is gained (McLauchlan et al. 2006). Thus, the output of this model represents the additional carbon stored by prairie compared to corn over the course of 40 years.

3.D.ii. Nutrient Retention Model

The InVEST nutrient retention model was used to calculate the total phosphorous runoff to surface water bodies across the landscape. The model first calculated annual average runoff from each 30m x 30m cell by taking the difference between precipitation and evapotranspiration (the sum of evaporation and plant transpiration from the Earth's land surface to atmosphere) and modifying it by water that can be taken up and released by plants. Values of phosphorus loading and phosphorus retention coefficients for each land-cover type were then used to calculate the sediment exported or retained by each map cell. Water flow paths were then mapped for each cell and phosphorous export was then summed for all downstream cells. The result was total phosphorus export (Px) per watershed, which was then summed across the 39 watersheds in our study area.

3.D.iii. Sediment Retention Model

The InVEST sedimentation model utilized the Universal Soil Loss Equation (USLE) (Wischmeier and Smith 1978) to calculate soil loss from sheet wash erosion. USLE multiplied amount and intensity of precipitation (the R factor), the ability of different soil types to be eroded

(K factor), and the length and steepness of the slope that the water is running down (LS factor) to produce an estimate of soil erosion on bare land. The equation also accounted for the ability of vegetation of each land cover type to retain soil, by multiplying the crop grown and tillage method (C factor) by the extent to which practices such as contour farming and strip cropping are utilized (P factor) (Renard et al. 1997). USLE is written as follows:

$$\text{USLE} = R \times K \times \text{LS} \times C \times P$$

InVEST used USLE to calculate the amount of soil eroded and retained for each pixel. Then the amount of sediment eroded or retained is summed for each downstream pixel. The result is the total sediment exported to surface water bodies per watershed. These values were summed across the landscape for the 39 watersheds in our study area (Tallis et al. 2011).

3.E. Economic valuation of ecosystem services using InVEST outputs

After using InVEST to generate values of carbon sequestration, nutrient retention, and sediment retention, we multiplied each biophysical output by an economic value to derive the value of ecosystem service provision. The methodologies for choosing each economic value are described below.

3.E.i. Carbon Sequestration

Currently there is no standardized value of carbon for the United States, so we instead used the social cost of carbon (SCC) to capture the value that society assigns to carbon. We will more fully develop our rationale for these values in the discussion. The social costs of carbon values that we selected (\$62 for A2 and \$266 at B1) were modified by the discount rate (7% and 3% respectively) and multiplied by the tons of carbon sequestered to provide us with a dollar value difference between the present and the future land cover scenarios. These values were chosen to be in line with what we would expect policy to be under A2 and B1 scenarios. In B1, we expect governments to be working together to solve the problems associated with global climate change, and therefore we expect them to use a high social cost of carbon. In A2, they are inclined to continue as much as possible under the paradigm of business as usual, we would expect to see a much lower value for carbon. In our optimistic scenario we also took into account the fact that the amount of carbon in the atmosphere would increase the marginal cost of each additional ton of carbon and therefore we used a value of a 4% increase in cost of carbon annually. Greenstone et al. (2011) suggested that the value of each new ton of carbon might increase, but failed to give a specific value.

3.E.ii. Nutrient Retention

We valued phosphorous retention based on percent reduction in phosphorous export for the prairie scenario compared to the corn scenario. Mathews (2002) found an average household willingness to pay (WTP) of \$140 (\$180 in 2013 dollars) per household for a 40% reduction in phosphorous pollution in Southern Minnesota watersheds. From this data, we created a linear

model, similar to that in Heavenrich et al. (2012), to predict WTP from our percent reduction in phosphorous values. We multiplied the WTP values we obtained from this model by 754,120, the number of households in our study area (MN Dept. of Administration, 2011), for an estimate of the total phosphorus retention value provided by the prairie scenario compared to the corn scenario for each climate scenario.

3.E.iii. Sediment Retention

We used an end-use services model to assign an economic value to sediment retention. This model was predicated on the idea that sediment retention itself should not be valued because it does not provide any direct benefit to humans. Rather, sediment retention acts as an intermediate service to improve water quality related ecosystem services, which are directly utilized by humans. Thus, we value the water quality related services instead of valuing sediment retention to avoid double-counting the intermediate services (Griffiths et al. 2012; Keeler et al. 2012).

The USDA Economic Research Service (Hansen and Ribaudo 2008) calculated per-ton estimates of the economic value of ten water-quality related ecosystem services influenced by sediment retention at the resolution of HUC-2 watersheds and USDA-defined Farm Productivity Regions (Table 2). These estimates were reported on a county-by-county basis. The values for these ecosystem services were derived based on travel costs (calculated based on expenditure and cost of travel to a recreational site where environmental quality is a factor), damage functions (calculated based on increased costs/decreased revenue to a business as a result of the decline in quality of an environmental input, such as water), replacement functions (calculated using the value to repair or replace a damaged ecological good or service), and averting expenditures (calculated based on the in change expenditures made to counteract the decline in welfare as a result of degraded environmental quality). We averaged per-ton estimates across the 48 counties in our study area for each ecosystem service. We then summed the values of the ten ecosystem services to calculate a sediment retention value of \$6.84/ton.

Table 2. Value of ecosystem services gained by the removal of 1 ton of sediment from the watershed of counties in Southern Minnesota. Services defined by USDA

Ecosystem Service	Method of Valuation	Average dollar value per ton of sediment (in 2013 dollars)
Reservoir Services	Replacement function	\$0.19
Navigation	Averting expenditures	\$0.09
Water-based recreation	Travel cost	\$2.07
Freshwater Commercial Fisheries	Damage function	\$0.15
Steam-Electric Powerplants	Replacement function	\$1.22
Municipal and Electric Water Use	Damage function	\$1.76
Flood Damages	Damage function	\$0.65
Irrigation Ditches and Canals	Replacement Cost	\$0.04
Roadside Ditches	Averting expenditures	\$0.26
Municipal Water Treatment	Damage function	\$0.41

3.F. Market Value of Corn and Prairie

In order to compare the degree of economic competitiveness between corn and prairie, we had to factor in profits and costs incurred by growing each of these feedstocks. Specific values were chosen for inputs and prices associated with biofuels on the open market. These values were then added to the values for ecosystem services in order to perform a more complete multidisciplinary analysis.

The costs of producing one hectare of corn were collected from enterprise budgets (Lazarus 2010) which took into consideration fertilizer, chemical, and seed costs inputs, along with a number of other minor costs. The gross profits for corn were based on estimates of corn yields in 2050 taken from Rosegrant et al. (2008) survey of future yield estimates, which were partially derived from increased technology and desire for biofuels. From this a yield of 662 bushels per hectare was selected, which is a little over a 60% increase in output. There is some disagreement as to how climate will impact this value, while changing precipitation values might make it harder to grow corn, the corn belt will also shift north with increased temperatures, which could increase yields for Minnesota Farmers (Mendelsohn et al. 1994, Roseburg and

Miranowski 2012). Under our assumption of business as usual we used the corn prices derived from USDA tables (Wisner, 2013). High, medium, and low values per bushel of corn (\$3.55, \$5.45, and \$8.50 respectively) were selected by analyzing the profits for ethanol feedstocks over the last five years, along with future predictions supplied by the Agricultural Marketing Resource Center. The high and low values represent the high and low from that period in 2012 values and the middle is based on the average for this period.

Prairie profits were harder to calculate than corn profits as there is currently no market for prairie as a feedstock. LIHD prairie does not require fertilizers or chemicals, but it does require seed and other costs associated with planting. The cost was based off of an estimate of the onetime cost of \$3088.81 per hectare from Prairie Restorations Inc, divided by a 40 year time horizon in keeping with our carbon sequestration valuation. The price per unit of prairie was based off of a yield of 4.3 tons per hectare from biofuels digest and a high, medium, and low Willingness to Accept (WTA) cost of \$75, \$100, and \$133 per ton respectively for biofuel refineries (CEEIIBP 2011). These values take into account miscellaneous costs such as transport and land rent that enterprise budgets also calculated, which means that while these prices were added separately to the corn costs, they are already accounted for in prairie costs. It is possible that these values could begin to rise as prairie grass becomes more competitive as a feedstock for biofuels, and as more people begin to produce it and process it, which could bring up the market value of each hectare of prairie. Also, because this project is presenting a static analysis of the value of corn and prairie there is no discussion of discount rate aside from the discount rate expressed in the social cost of carbon. We are not advocating investment in prairie or corn biofuels by any organization, merely discussing their competitiveness with each other given a valuation of economic and ecosystem services (Table 3).

Table 3. Input costs and profits for low, medium and high prices of corn and prairie

	Corn	Prairie
Total Costs	(\$1,489.86)	(\$77.22)
Profit High	\$5,629.05	\$575.14
Profit Mid	\$3,609.22	\$432.43
Profit Low	\$2,350.96	\$324.33
Net Profit High	\$4,139.19	\$497.92
Net Profit Mid	\$2,119.35	\$355.21
Net Profit Low	\$861.10	\$247.11

4. Results

4.A. Carbon Sequestration Model

After running the carbon sequestration model, we found that prairies stored 51,120,000 tons of carbon in comparison to corn over the 40 year time step. In the A2 scenario, the value of additional carbon stored by prairie was \$3,169,402,880 which is equivalent to \$69.75 per hectare per year. In the B1 scenario, the value of carbon sequestered by prairie over our time step was \$13,597,999,104, which is \$299.25 per hectare per year.

4.B. Nutrient Retention Model

Under the A2 climate scenario, the prairie scenario decreased phosphorus export (Px) by 13.17% compared to the corn scenario. When inserted into our WTP model, this reduction yielded a value of \$59.27 per household, equivalent to \$44,696,692 over our study area. Given that the conversion of corn to prairie on marginal lands occurred on 1,135,990 hectares within our study area, the value in phosphorus retention services is \$39.35 per hectare under the A2 climate scenario.

For the B1 scenario, a 13.25% reduction in Px in the prairie scenario yielded a similar per household value of \$59.63, corresponding to a total value of \$44,968,176. This is equivalent to a value of \$39.59 per hectare converted. The difference between two total values under the two climate scenarios is just \$271,484, meaning that uncertainty in future climate change will not have a large effect on the phosphorus retention services provisioned by the prairie scenario (Table 4).

Table 4. Decreases in phosphorus export in prairie in each scenario, and value per Ha

	B1 Climate Scenario	A2 Climate Scenario
Percent decrease in phosphorus export	13.25	13.17
Total value (\$)	44,968,176	44,696,692
Value/area converted (\$/Ha)	39.59	39.35

4.C. Sediment Retention Model

In both climate scenarios, the prairie feedstock scenarios had 28.8% less soil erosion than the corn feedstock scenarios. In the baseline climate scenario (0% precipitation change), the prairie scenario lost 103 fewer tons per hectare of total soil than the corn scenario. This resulted in an economic value of \$666.34 per hectare of retained sediment when prairie feedstock is grown. The relationship between precipitation and soil erosion was linear with an increase of 3.7 million tons of soil erosion for every percent increase in precipitation for the corn scenario, and an increase of 2.6 million tons of soil erosion for every percent increase in precipitation in the prairie scenario, meaning that uncertainty of future climate change could potentially drastically alter the amount of sediment retention (Table 5).

Table 5. Value of sediment retention by prairie for different precipitation scenarios. Values in this table were computed by subtracting soil erosion rates of corn from soil erosion rates of prairie. The difference in soil erosion per hectare was then valued in dollars.

Percent change in precipitation from baseline scenario	Difference in total sediment exported (millions of tons)	Difference in sediment exported/hectare (tons/ha)	Difference/hectare in economic value (dollars/ha)
10	138	121	\$785.86
0	117	103	\$666.34
-10	95	84	\$543.67

4.D. Summary

In the B1 scenario, the per hectare value of prairie (ecosystem services plus market value) exceeded that of corn when corn prices were low. Prairie generated profits \$391-\$642 greater than corn when the price of corn was low and the price of prairie was low, medium, or high. However, when the price of corn was medium or high, corn profits exceeded prairie profits by \$616 -\$2,866 (Table 6). In the A2 scenario, the per hectare value of prairie exceeds that of corn when the price of corn was low and price of prairie was low, medium or high. The profits of prairie in this scenario exceeded the profits of corn by \$161-\$412 If the price of corn is medium or high, corn profits exceed prairie profits by \$845 -\$3,166 (Table 7). Although climate change affected the magnitude by which prairie was competitive with corn, it did not affect the overall result that prairie is competitive only when the price of corn is low.

Table 6. This table displays, for both climate scenarios, the economic value of each ecosystem service for both feedstocks and the market value of each feedstock. These values are summed to obtain the total value or the value of each feedstock if ecosystem services are considered. Note that all prairie scenarios are competitive with corn when the price of corn is low, but none are competitive when the price of corn is medium or high.

	Corn High	Corn Mid	Corn Low	A2 Prairie High	B1 Prairie High	A2 Prairie Mid	B1 Prairie Mid	A2 Prairie Low	B1 Prairie Low
Carbon	-	-	-	\$69.75	299.25	\$69.75	299.25	\$69.75	299.25
Nutrient	-	-	-	39.35	39.59	39.35	39.59	39.35	39.59
Sediment	-	-	-	666.34	666.34	666.34	666.34	666.34	666.34
Market Value	\$4,139.19	\$2,119.35	\$861.10	\$497.92	\$497.92	\$355.21	\$355.21	\$247.11	\$247.11
Total Value	4139.19	2119.35	861.1	1273.36	1503.1	1130.65	1360.39	1022.55	1252.29

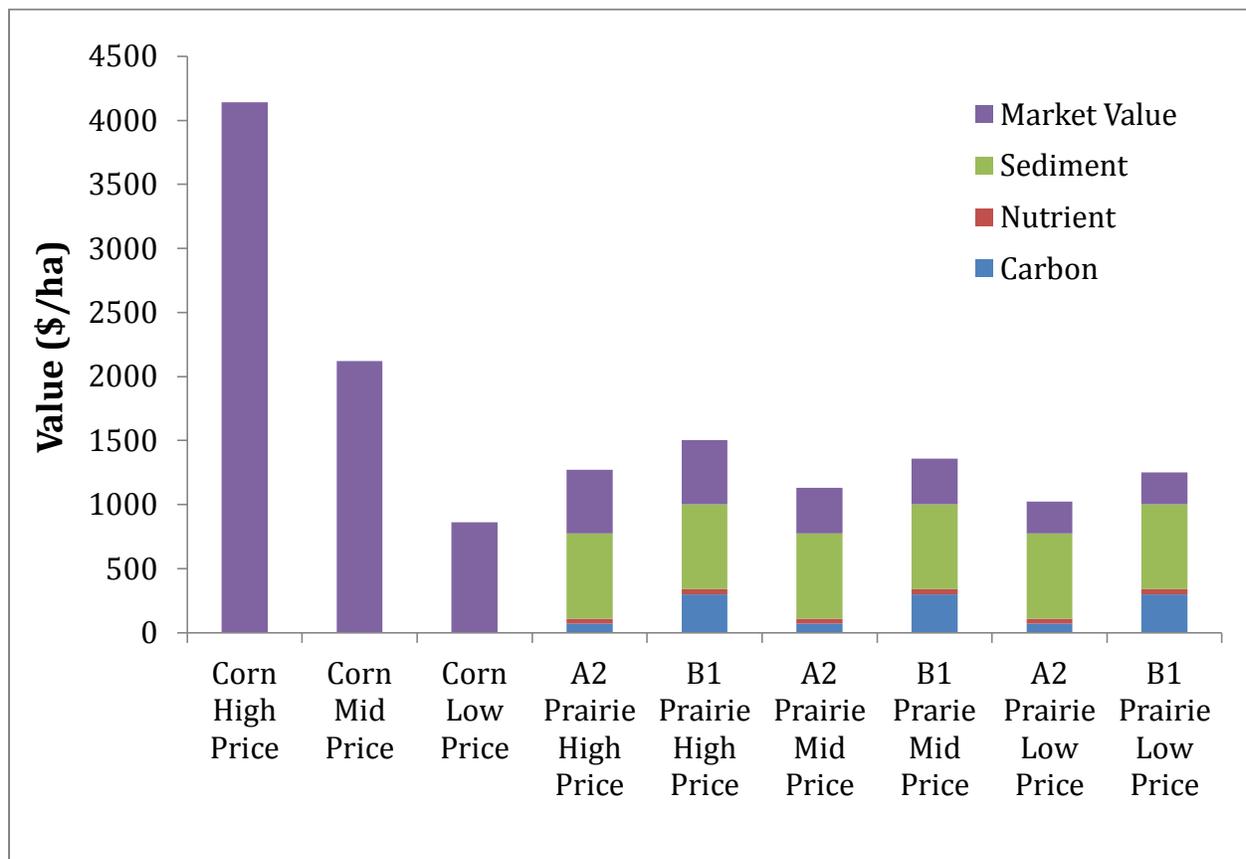


Figure 4. Value of each feedstock by price and scenario. Prices further broken down by ecosystem services obtained from InVEST and by market value derived from price per unit of feedstock and input costs.

5. Discussion

This study finds that prairie feedstock is economically competitive in both climate scenarios when the price of corn is low and the price of prairie is low, medium, or high. These results are similar to those of Heavenrich et al. (2012) who found that prairie was economically competitive with corn when the price of corn was low and the price of prairie and carbon were low, medium, or high. This is unexpected given that Heavenrich et al. (2012) only examined class 3 marginal lands and did not value sediment retention. Furthermore, future land cover change examined in this study contributed to a 1% increase in corn grown on marginal lands, meaning more land was converted to LIHD feedstock. The future climate scenario also caused a \$4.25-\$4.49 per hectare increase in the value of phosphorous retention and our high SCC value was \$266 compared to their high carbon price of \$135.

These results may be explained by the predictions that corn yield will dramatically increase between now and 2050 (Roseburg and Miranowski 2012). Improved technology and increased growing season length are expected to increase corn yields to 266 bu/ha. Furthermore, our estimate of the economic value of ecosystem service provision in Minnesota may be considered highly conservative. We were unable to value biodiversity, which caused us to underestimate total ecosystem service provision. Furthermore, the economic values assigned to the sediment and nutrient retention models do not consider non-use values, thus underestimate the total value provided by water-based ecosystem services. Addressing these factors would lead to a more accurate representation of the actual ecosystem service values derived by people in our study area. Furthermore, it would contribute to the development of policies that are less likely to favor environmental degradation.

One potentially confounding factor in our ecosystem service valuation is the biophysical measure of sediment retention. With no change in precipitation, we found prairie had 102 tons/ha less soil erosion than corn feedstocks. This is similar to a finding by Brown et al. (2000), who found that switchgrass feedstock had 119 Mg/ha/year less soil erosion than corn feedstock for a study site Kansas. However, Brown et al. (2000) also found that for an Iowa site, switchgrass feedstock had 26-36 Mg/ha/year less soil erosion than corn. Given that Southern Minnesota is geographically closer to Iowa than Kansas, we would expect to see values closer to those found in Iowa. One potential reason for the high sediment values obtained in this study could be the use of a modified R factor. While InVEST provides R-factor data for the entire United States, this data is based on precipitation data recorded from 1960-1999 and does not take into account increases in precipitation occurring over the last decade. We attempted to capture current and future climate trends using the statistical relationship between average annual precipitation and rainfall erosivity developed by Renard and Freimund (1994) using current and future precipitation data. However, this led to higher R-factor values for our study area, even in the scenario with a 10% reduction in precipitation, indicating that the calculated R-factors may be contributing to an overestimation of soil erosion across our study area. Our inability to accurately account for current, and future, climate conditions using the statistical relationship between

average annual precipitation and rainfall erosivity highlights the need for an improved method of capturing changing climate conditions in R-factor calculations.

5.A. Ecological limitations of models

Our biophysical models were subject to many ecological limitations. One limitation impacting all four models was sensitivity to grouping of LULC classes. Within a broad land cover class, there may be differences in the extent to which different land cover subclasses provide ecosystem services (Tallis et al. 2011). For instance, different types of forests (mixed, deciduous, coniferous) may have differently sized carbon pools or may retain different amounts of sediment, but may be grouped into a single “Forest” category. Due to the inability of LULC maps to capture the true heterogeneity of the landscape without becoming overly complex, some land cover subclasses had to be grouped into larger land cover classes, despite differences in ecosystem provision between the subclasses. In these instances, the abundance of each land cover subclasses was used to calculate a weighted average of ecosystem service provision for the larger land cover class.

Many of our models also faced limitations in trying to capture the effects of climate change. For instance, we were unable to account for climate change in the biophysical portion of our carbon model. It would have been possible to incorporate changes in carbon sequestration by ecosystems by modifying carbon pool size to reflect the impacts of climate-change on carbon stored by an ecosystem. However, there is no consensus in the literature regarding possible impacts of climate change on carbon storage capacity of different ecosystems. Ge et al. (2012) even found that one geographical area may show either positive or negative sequestration values depending on which climate scenario is run.

We also faced difficulty incorporating climate change into the nutrient and sediment retention models. Both of these models were limited by the inability to input monthly precipitation data into InVEST. This prevented us from capturing alterations in the distribution and intensity of precipitation throughout the year. Furthermore, the effects of winter hydrology, such as snow accumulation, snowmelt, soil freeze, were ignored (Savabi et al. 1995). A lack of temperature input into the nutrient model may have hindered our ability to capture the effects of climate change. Future variations in temperature could affect length of growing season and productivity, causing differences in rates of nutrient uptake. However, the absence of temperature data may have been mitigated by the inclusion of evapotranspiration, which is partially determined by temperature.

Similarly, the sediment model only considered the independent effects of each individual variable on soil erosion, instead of accounting for possible interactions between variables, thus limiting its sensitivity to climate change. A complex web of interrelated factors influences soil erosion rates. For example, amount of precipitation may influence soil moisture, which may in turn influence vegetation cover (Nearing et al. 2004). Furthermore, amount and intensity of precipitation may influence infiltration rates of water into soil. Models such as the Water Erosion Prediction Project (WEPP) are better than InVEST for capturing these interacting factors.

5.B. Economic limitations of models

A major challenge facing the valuation of ecosystem services is the lack of unified methodology and units of valuation (Boyd and Banzhaf 2007, Griffiths et al. 2012, Keeler et al. 2012). This is likely a result of the disparity between ecologists' study of ecosystem processes and functions and economists measures of end-use value (Boyd and Banzhaf 2007). Our own economic model required a variety of valuation techniques to choose a dollar value for our diverse suite of ecosystem service. In our carbon model, carbon sequestration itself is treated as an ecosystem service, even though people do not directly benefit from less carbon dioxide in the atmosphere. Instead they benefit indirectly, by avoiding the damages caused by impacts of climate change, such as homes destroyed by severe weather events or decreased agricultural productivity. By contrast, our sediment model is based on end-use services utilized by people, treating sedimentation as an intermediate process influencing ecosystem services. Similarly, our nutrient model is based on stated preference for an improvement in water quality. There are several advantages and disadvantages to each of these methods of valuation. In the following paragraphs, we will discuss some advantages and disadvantages to the two valuation methods utilized by the water quality models. We will then cover why we chose not to value biodiversity. The carbon model valuation is discussed in further detail in the ethics section.

The two models used to value sediment and nutrient retention were the end-use services model and a contingent valuation model, respectively. Some advantages to the end-use services model, which we used for sediment valuation, are that it provides measurable units to which dollar amounts may be assigned (such as commercial losses in the fishing industry), and that it prevents double-counting of ecosystem services (Boyd and Banzhaf 2007). One weakness of this model is that it does not incorporate non-use values such as vicarious enjoyment, stewardship, and option of future generations to access the ecosystem (Wilson and Carpenter 1999). In order to calculate the true-value of water quality based ecosystem services, both use and non-use values must be considered, particularly because non-use values have been shown to constitute a large portion of freshwater value (Griffiths et al. 2012; Hansen and Ribaudo 2008; Wilson and Carpenter 1999). Valuation methods which do not capture non-use values may bias policy decisions in favor of environmental degradation.

In contrast to our sediment retention model, our nutrient retention model used Willingness to Pay (WTP) studies to choose an economic value. The WTP methodology is often used to capture the value of non-use market services, meaning that ecosystem services are less likely to be undervalued using this methodology. However WTP is based on how much people report they are willing to pay for a non-market good, not how much they actually do pay (Wilson and Carpenter 1999). Furthermore, WTP outcomes may be influenced by wording of survey questions, although recent improvements in survey design and administration methodology have minimized these effects (Griffiths et al. 2012).

Both the nutrient and sediment valuation methodologies were further limited by failing to account for spatial heterogeneity of valuing water-based ecosystem services. Choosing an economic value to reflect benefits of improved water quality at a state or federal level is

challenging as the effects of nutrient or sediment retention often have local effects (Boyd and Banzhaf 2007, Lovejoy et al. 1997). For example, sediment or nutrient runoff upstream of a reservoir or water treatment plant will have a greater effect on replacement costs that need to be paid by those facilities than pollutants released downstream (Tallis et al. 2011). If a watershed contains no reservoirs, then the value of avoided reservoir sedimentation is 0 within that watershed. Yet state and federal regulations may mandate that members of this watershed must pay for those ecosystem services. Spatial heterogeneity also exists for measures of water-related ecosystem services, such as (WTP), which may be influenced by local water quality, with people living near degraded streams willing to pay more for an improvement in water quality (Tait et al. 2012).

Stemming from the lack of a unified methodology in valuing ecosystem services, we elected to refrain from valuing biodiversity in economic terms. This is a contentious issue in the scientific community and there remains no universal method to value biodiversity and thus no universal figure for its value in dollar terms. The most commonly used method is contingent valuation, which uses a hypothetical market in which people express their willingness to pay for the protection of biodiversity (Berta et al. 2008). Although a well supported, yet not unanimous, method for valuing biodiversity, contingent valuation data are incompatible with InVEST's output for biodiversity. InVEST only produces a habitat quality score, which is an abstract synthesis of numerous input factors, not suitable for use in the construction of a contingent valuation model.

We also refrained from valuing biodiversity because biodiversity estimates typically under-value its services. In addition, the literature fails to apply economic valuation to the entire range of biodiversity benefits (Nunes 2001). Therefore, any valuation estimates give an incomplete perspective, or at best an underestimate, to the unknown value of changes in biodiversity. We thus determined that comparing biofuel feedstocks would be more robust if we did not use incomplete underestimates of biodiversity's economic value.

5.B.i. Carbon Sequestration

With regards to our carbon sequestration model, the decision surrounding what value for the social cost of carbon (SCC) to use was important and significantly impacted our results. In 2009, the Interagency Working Group on the Social Cost of Carbon, initiated by the EPA, aimed to calculate SCC (EPA, 2009). The main alterations the Working Group implemented into their model were changes in net agricultural productivity, human health, property damages from increased flood risk, and the value of ecosystem services due to climate change. The Working Group's 2010 average estimates of SCC were \$5, \$35, \$65 (in 2007 dollars) at the discount rates of 5, 3, and 2.5%, respectively (EPA, 2009). This estimate, however, has recently come under scrutiny in the literature. Johnson and Hope (2012) found that the Working Group's SCC calculation methodologies chose discount rates that were too high because they did not use equity weighting, the notion that the cost of a dollar is more valuable in poorer areas than in rich areas, making the cost of pollution more expensive in poor regions (Johnson and Hope, 2012). They further took issues with the fact that the discount rates that the working group used

undervalued duties to future generations. Using modified discount rates, Johnson and Hope (2012) determined that using their methodology, average SCC was \$62, \$122, and \$266 (in 2007 dollars) at the discount rates of 2, 1.5, and 1%, respectively. In addition, Johnson and Hope note that there is a significant distinction between a global SCC and a local SCC and that based on equity weighting, SCC could change drastically.

These social costs were considered with duty to future generations in mind. This is generally expressed through our use of discounting. Because the effects of climate change will be around for hundreds of years we have to find a way of expressing impacts on future generations. These future generations will suffer as a direct result of the carbon that is being emitted right now, and we address that through the use of discounting. In pure economic terms discounting is based on the idea that money in the present is worth more money than money in the future. The higher the discount rate, the less we value future money, so while a discount rate of 0 would mean that money in every time period is worth the same amount a discount rate of 10% would indicate that you would have to get \$110 next year to be just as satisfied as you would be to get \$100 this year (Johnson and Hope 2012). Generally, environmentalists, economists, and ethicists differ on what value a discount rate should take, especially for the issue of climate change. Environmentalists, often try to use lower values for discount rates, because these lower discount rates assign a greater value to future generations, and illustrate the greater need for action in the present. In the Stern Report on the Economics of Climate Change (2007) they said that discounting must in the case of climate change must be able to assess several diverging paths that have long term and intergenerational impacts, meaning that conventional discounting may not be as effective. They challenge the prospect of pure time discounting (discounting because we prefer to have goods now rather than later) because of the length of the time horizon associated with climate change. They instead only support discounting in terms of increased future consumption, the idea that future generations will be better off than we are due to increases in consumption and technology. The report goes on to suggest that using discount rates that are substantially higher than zero is significantly neglecting our duties to those future generations, and proposes a discount rate of close to zero to account merely for future uncertainty. This rate, however, is more indicative of what an environmentalist would propose and is therefore much lower than what we would expect to see an economists or policy maker choose. They would probably opt for higher discount rates, which could be because they failed to take into account social costs along with the specific accounting costs, leading to to an even higher discount rate.

In addressing the social cost of carbon we had to choose discount rates that would be acceptable for all three of these lenses, so for our analysis we used a discount rate of 3% for our optimistic (B1) scenario and 7% for our business as usual (A2) scenario. There are several key pieces of United States legislation, both environmental and economical that use similar discount rates. In their analysis of clean air legislation, the EPA uses discount rates of 3% and 7%, further in the working group on climate change surrounding the social cost of carbon selected a discount rate of 3% for their median value (Johnson and Hope 2012). While the rate of 3% is higher than

some of the lower estimates for discount rate that we could find, we feel that 3% is a good value because of the uncertainty present in the future of biofuel production it was also a median selection in the research done by Greenstone et al. (2011), who were also concerned with the social cost of carbon. Also, because our carbon analysis goes out to 2090 we wanted to chose a value that seemed reasonable over that time span Johnson and Hope take their analysis a step further to suggest that higher discount rates should be applied to poorer areas, because every dollar they spend now is one that they cannot spend on necessities, rather than in more affluent areas where they can better afford to deal with the consequences of climate change, we see our B1 scenario as being better able to capture this lower income group, although we realize that there still may be inequalities that this does not fully capture, however further manipulation of discount rate is difficult using our model. A discount rate of 7% is also within the proposed values from the EPA, and is also the cost benefit analysis discount rate cited in a white house circular from the Office of Management and Budget, and while it is higher than most environmentalists might suggest we feel that it adequately captures the spirit of business as usual, existing at the upper level of acceptable discount rates (Administration, 1992).

5.C. Ethics

So far our work has focused very little on the ethics of biofuels, the production of which has been questioned, given food and resource shortages. The issues of food vs. fuel is an integral consideration for accurate assessment of biofuel production potential and were incorporated into our analysis and results.

There is significant disagreement in the intellectual community about the ethics of using biofuels as a substitute for fossil fuels to decrease our carbon emissions. Pacala and Socolow suggest that an increased use of biofuels could potentially help us in our transition away from fossil fuels, as a low carbon alternative, but they also warn that increased production of biofuels could compromise agricultural productivity (Pacala and Socolow 2004). Pimentel et al. (2009) suggest that “Increased use of biofuels further damages the global environment and especially the world food system,” citing an increased need for agricultural land and resources in order to produce biofuels on a large scale, and the potentially negative environmental ramifications of biofuel production. While many of these detractors tend to operate on the global scale, there are others who are cautiously optimistic that a biofuel transition can occur as long as biofuel strategies are location specific (Rosegrant et al. 2008) They claim that while there is a tradeoff between food production and biofuel production, this tradeoff can be managed effectively to increase employment for small farmers and not dramatically harm the local or global economy. Robert et al. (2006) study found that a complete biofuel transition could occur in Stockholm by 2030 without exceeding their proportional share of national biofuel outputs, as long as there are appropriate supply and demand side changes made to the transportation sector. In this paper they also emphasize the importance of biofuels eventually being replaced by more sustainable renewable fuels such as solar or wind, pointing to biofuels as a method to ease the transition from fossil fuels to renewables. Tillman et al. (2006) suggest that concentrating on non food based biofuels would have a yield a dramatic decrease in carbon emissions (41% for biodiesel)

and require fewer inputs than corn ethanol, making them both environmentally and agriculturally good alternatives to fossil fuels.

Furthermore, our study focuses on biofuel production on marginal lands that would not be otherwise used to produce food crops. It is important to note that there is no unified definition for marginal lands. Given that the definition of marginal lands dramatically impacts the availability of land to grow biofuels versus the availability of land to grow food, the definition chosen will impact not only how much energy may be produced, but how much potential agricultural land may be displaced by biofuel production. We chose LCCs 3 and 4 to define marginal lands based on the “severe” to “very severe” designation placed on these land cover types by the NRCS. However, other studies may define only LCC 3 as marginal lands (Heavenrich et al. 2012), providing a more conservative estimate of land available for biofuel production.

5.D. Policy implications

There are several approaches policy makers may take to enact policies to better value ecosystem services provided by biofuel feedstocks. Implementing subsidies equivalent to the ecosystem services provided by a given feedstock could incentivize these more ecologically friendly feedstocks (Lant et al. 2008, Turner and Daily 2008). Subsidies could be paid directly to the farmer in the form of some dollar amount per unit of prairie feedstock produced (Chamberlain and Miller 2012) or per hectare of land converted to prairie feedstock.

Such subsidies to support the establishment of an LIHD biofuel industry would be justified by the strong precedent of price supports for corn grown in the U.S. Between 2006 and 2011, the federal government funded about \$6 billion per year for the growth and production of corn ethanol (Koplow 2006). Due to the environmental and food security risks associated with the expansion of corn monoculture, these subsidies were not renewed in 2011. However, corn remains the most heavily subsidized crop in the country, in the form of direct payment to producers, with about \$5 billion being paid out in 2012 (Scientific American 2012). Since much of this corn is destined for biofuel refineries, these subsidy programs constitute continued federal support of the corn ethanol industry. Moving some of this subsidy money into LIHD prairie will not only support the development of cellulosic biofuels, but also increase the levels of ecosystem services provided by America's agricultural land. Alternatively, an emissions or pollution tax may be applied (Lant et al. 2008, Turner and Daily 2008). Chamberlain and Miller (2012) propose that a nitrogen tax be implemented by taxing fertilizer, instead of by taxing direct runoff. Researchers have also suggested implementing a system of tradable pollution permits, similar to proposed systems for carbon dioxide emissions (Lant et al. 2008, Turner and Daily 2008). Certain voluntary carbon and nitrogen trading programs already exist at the state level (Chamberlain and Miller 2012).

Policy makers face several challenges when designing policy incentives to value ecosystem services. One of the most important is the scale at which policy incentives are implemented. The biophysical reality of ecosystem service provision does not correspond to political boundaries, and different stakeholders in different locales may perceive different costs

and benefits (Turner and Daily 2008). Policy makers must also work within the bounds of private landowner rights. Lant et al. (2008) proposes the creation of ecosystem service districts within federal and state governments to better account for the heterogeneous nature of ecosystem service provision.

6. Conclusion

If we had additional time to research this topic, one potential avenue of research would be to compare the magnitude of the impacts of climate and land-use change on ecosystem service provision. Projections of future land-cover and climate conditions are uncertain, and therefore, use of scenarios is necessary. Determining the relative impact of these two factors would allow researchers to better determine sources of uncertainty in their results. Furthermore, if one of these factors is not found to significantly impact ecosystem services, it could reduce the number of scenarios, thus the amount of uncertainty. This research can also aid in selecting models based on whether they are better equipped for addressing the effects of climate change or land use change.

LIHD prairie used for the production of biofuels provides an environmentally and ethically sound alternative to the use of corn as a biofuel feedstock and our study found that prairie feedstock is economically competitive with corn, but only when corn prices are low. Policy makers could incorporate our findings into more environmentally and ethically sound agricultural policy legislation. Because carbon sequestration was one of the most economically valuable of the ecosystem services, the price assigned to social carbon costs will have a large impact on the economic competitiveness of prairie feedstock. Furthermore, climate change will likely increase the difference in ecosystem services provided by prairie as compared to corn, thus it is an important factor to consider when comparing the economic competitiveness of these two feedstocks. Finally, agricultural expansion due to increased biofuel demand has the potential to cause conversion of native ecosystems to agricultural land, thus dramatically reducing future ecosystem service provision. Prairie feedstock has the potential to offset the negative impacts of agricultural expansion. More research needs to be done to better incorporate the effects of changing climate and land-use into measures of ecosystem provision, and to better capture the complete value of ecosystem service provision by prairie and corn feedstocks. Even given our conservative estimate of prairie ecosystem services, our results indicate that prairie feedstock has the potential to be economically competitive with corn in the future.

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Appendix A

Assumptions for the Carbon Model

The model greatly oversimplifies the carbon cycle, which makes it easier for us to use it having less information, but means that we have to make some simplifications. The first of these is that we model carbon sequestration so no LULC type is either gaining or losing carbon over time, meaning that the only way that carbon can be changed in this model is for the LULC type to change, so the only difference between the corn and prairie scenarios is the difference between the cells that changed from corn to prairie.

The LULC type also relies heavily on how well data on cover type is collected. The results are only as reliable and detailed as the data that we found for each LULC, while the data we used is good, it fails to take into account things like the differing age of forests or prairies that store different amounts of carbon, or differences between different types of wetlands or crops, lumping everything together, which allows us to get a glimpse of what sequestration looks like.

The model also fails to capture carbon that moves from one pool to a different pool, moving from a forest to a prairie, or down a river. It assumes that all carbon that is removed from one space is instantly released into the atmosphere and does not allow for that carbon to either stay where it was or move to a different pool. The model also tends to undervalue the way that carbon is sequestered in different ecosystems, assuming that carbon sequestration progresses in a linear way, when carbon is actually stored at an increasing rate for the first few years of an ecosystem, and after a point in decreasing amounts, until eventually it is stored at a relatively low rate every year. This is especially important in regards to prairie which will sequester more carbon shortly after it is planted and less as the prairie ages.

Table 1. Carbon model biophysical table

Land-use/Land-cover Type	Aboveground Biomass	Belowground Biomass	Soil Carbon	Litter Carbon
Water	0	0	0	0
Developed	17.3	0	82.7	0
Forest	109.54	0	138.42	0
Grassland	12.56	0	99.77	0
Pasture	6.7	0	77.55	0
Agriculture	0.6	0	66.6	0
Wetlands	0	0	123.8	0
LIHD Prairie	12.56	0	99.77	0

Table 2. Carbon model valuation table

	A2	B1
	Business as Usual	Optimistic
Value of sequestered carbon	\$62	\$266
Discount Rate	7%	3%
Annual Rate of Changes in price of Carbon	0	4%

Appendix B

Assumptions and limitations of the water-based models

- 1) The model only considers sheet erosion (erosion from plains with gentle slopes). We did not account for the effects of rill-inter-rill, gully or stream bank erosion because these require location-specific data, and thus are not suitable to be modeled at a landscape-level scale. Although sheet erosion is not the main contributor to soil erosion, it provides a good indicator of relative levels of soil erosion in the two different landscape scenarios (Ziv et al. 2012).
- 2) Effects of tile drainage were not accounted for in this model. Tile drainage is prevalent across the Midwest and would likely increase water infiltration, decreasing soil erosion (Ziv et al. 2012). Because tile drainage is used for virtually all crop types, it should not affect relative rates of soil erosion.
- 3) The model does not account for in-stream sediment deposition, meaning that it systematically overestimates sediment exported from watersheds (Tallis et al. 2011).
- 4) In the LULC data for this model, “Barren” land is separated into its own category.
- 5) This model only accounts for surface water dynamics. Groundwater is not considered.
- 6) The threshold flow accumulation value was set to 600 by matching the flow accumulation to the streams displayed in the 25k Rivers and Streams layer from the Minnesota DNR.
- 7) The slope threshold value was set to 75% based on a slope layer created using the DEM.

Table 1. Biophysical table for the InVEST nutrient retention model

Land-use/Land-cover type	Root Depth (mm)	Evapotranspiration coefficient	Nitrogen Load (g Ha-1 yr-1)	Nitrogen filtering coefficient	Phosphorus loading (g Ha-1 yr-1)	Phosphorus filtering coefficient
Water	0	1	0	0	0	0
Developed/open	0	1	5000	0	1000	0
Forest	4000	1000	3720	80	112	50
Prairie	4000	1000	8650	80	1500	80
Pasture	3700	1000	8650	80	1500	80
Corn	1300	1100	16090	50	406	65
Wetland	0	1	0	0	0	0

*Load values from Lin 2004.

**Root depths from Canadell et al. 1996

***Evapotranspiration and filtering coefficients based on suggested values from InVEST documentation

Table 2. Inputs and sources for the InVEST sediment retention model

Input	Source
Digital Elevation Model	http://deli.dnr.state.mn.us/
Rainfall Erosivity Index (R-factor)	Calculated based on precipitation data: http://www.prism.oregonstate.edu/
Soil Erodibility (k-factor)	http://soildatamart.nrcs.usda.gov/
Land Use/Land Cover	http://ltm.agriculture.purdue.edu/usgs.htm
Watersheds	http://deli.dnr.state.mn.us/
Subwatersheds	http://deli.dnr.state.mn.us/

Table 3. Biophysical table for the InVEST sediment retention model

Land-use Land Cover Type	C Factor	P Factor	Sediment Retention Efficiency
Water	0.001	0.001	80
Developed	0.001	1	5
Forest	0.001	1	60
Grasslands	0.01	0.2	40
Pasture	0.02	0.89	40
Cultivated Crops	0.28	0.89	25
Wetlands	0.003	1	60
LIHD Feedstock	0.01	0.02	40
Barren	1	1	20

*Values from Ziv et al. (2012) and Tallis et al. (2011)

Appendix C-Biodiversity Model

This appendix outlines the methods, results, and limitations of the biodiversity model. We decided to pull the biodiversity model from our analysis because it was difficult to incorporate climate change impacts and also to value biodiversity but it still has some connections to the competitiveness of prairie and corn biofuel feedstocks.

Methods

In order to model biodiversity, we examined relative habitat quality scores for our feedstock scenarios according to the methodology of Tallis et al. (2011). InVEST compared the biodiversity of different land-cover scenarios by using habitat quality as a proxy for biodiversity. We chose birds, specifically grassland bird species, as indicators of habitat quality because they have been used in many studies as a measure of ecosystem health (Naugle 1999). InVEST produced a quality sum index, which indicated habitat quality across each land-cover scenario. The quality sum index is based on a habitat quality score calculated for each pixel in the landscape. This score was based on the land-use of the pixel, the land-use of the surrounding pixels, and the sensitivity of each LULC type in a pixel to threats.

InVEST required tabular inputs outlining the threats to biodiversity and the sensitivity of each habitat type to the threats considered. The threats table (Table x) was derived from a list of threats to species endangerment in the United States (Czech et al. 2000). In determining the relevant threats to biodiversity in Southern Minnesota, we eliminated threats that were unrelated to our study (ex. Logging, military activity, vandalism), redundant to other threats (pollution of water, air, or soil is implied in agriculture; industrial activities are inherent in urbanization), or unavailable in our LULC data (mineral, gas, oil, geothermal extraction and exploration). The maximum distance over which the threat would be in effect, the relative weight of each threat to birds, and the type of decay for each threat were determined from relevant literature and expert opinion (Polasky 2011, Trombulak 2000, Donald 2001, Palomino 2007, McKinney 2002, Brotons 2005).

The sensitivity table assigns a habitat score to each LULC type and then provides the sensitivity of each LULC type (considering each type as bird habitat) to threats relative to the sensitivity of each other threat considered. Prairie, grasslands, and wetlands were considered habitat (1) because birds frequently use these ecosystems for breeding, foraging, and other habitat uses. Pasture was considered partial habitat (0.5) because birds may use these areas as habitat if their primary habitat areas are unavailable. Agriculture was designated as poor habitat (0) because birds rarely use agricultural lands as habitat. Sensitivity scores for each threat were based on the methodology of Polasky et al. (2011) and relevant literature (Trombulak 2000, Donald 2001, Palomino 2007, McKinney 2002, Brotons 2005).

Results

Between the corn scenario and the prairie scenario, biodiversity as reported by habitat quality was vastly different. With quality scores of 584,908 and 17,920.8 for corn and prairie,

respectively, the prairie scenario was superior to the corn scenario in terms of biodiversity as larger habitat quality sum scores indicate a lower overall habitat quality in a given scenario.

Limitations and Assumptions

- 1) The biodiversity model was also subject to many limitations inherent in the assumptions we made in modeling biodiversity. Primarily, InVEST assumes that threats work independently from one another. It calculates the combined effect of two threats (agriculture and urbanization, for example) by aggregating the threats together, rather than accounting for how threats may influence each other, resulting in limitations of accuracy.
- 2) Our biodiversity model was also limited in that it only considered two possible decay patterns representing how each threat degrades in effect as distance from the threat increases. InVEST also asks for a value representing how the influence of each threat degrades as distance from the threat increases. InVEST only considers two possible decay patterns (linear and exponential) and ignores the possibility of alternative forms of decay, such as logarithmic which likely would have influenced our results.
- 3) In addition, the indicator species we used to inform our biodiversity carries associated assumptions and subsequent limitations that could have influenced the outcome of our study. Grassland birds were an appropriate indicator species for our study because their behavior is well documented in our study area but looking at biodiversity through the lens of another animal species or a plant species likely would have yielded different results according to different threat characteristics and associated habitat/sensitivity scores.
- 4) Another limitation of our use of InVEST’s biodiversity model is that we did not include all possible threats as outlined by Czech et al. (2000), excluding outdoor recreation areas, logging, and disease, for example. We also combined other threats with associated threats (reservoirs and running water diversions, Pollution of water, air, soil, and aquifer depletion we assumed were included in urbanization).

Table 1. Threat Table for InVEST Biodiversity model. Table includes information about threats to biodiversity, the maximum distance (km) each threat has an affect on its surroundings, the relative weight of the threat (from 0 to 1) and if each threat decays in a linear (0) or non-linear (1) manner.

Maximum Distance	Weight	Decay	Threat
10	1	0	Agriculture
10	1	0	Urban
5	0.2	0	Ranching

Table 2. Sensitivity Table for the InVEST Biodiversity Model. Includes each LULC type, its habitat score (0-1) and its sensitivity to each threat (0-1).

Land-use Land-cover type	Habitat Score	Sensitivity of habitat to agriculture	Sensitivity of habitat to urban	Sensitivity of habitat to grazing
Water	0	0	0	0
Developed	0	0	0	0
Forest	1	0.6	0.7	0.2
Grassland	0.5	0.7	0.8	0.3
Pasture	0.5	0.7	0.8	0.3
Agriculture	0	0	0	0
Wetlands	1	0.9	0.9	0.4
LIHD Prairie	1	1	1	0.8