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Analysis

Uncertainty in ecosystem services valuation and implications for assessing land use tradeoffs: An agricultural case study in the Minnesota River Basin

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ABSTRACT

Ecosystem services analysis can help recognize the full costs and benefits of land management decisions. Quantification and valuation of services can enhance policies and regulations and, if linked with payments or incentives, properly reward private decisions that yield public benefits. However, the field of ecosystem services research is relatively new and quantification and valuation remains highly uncertain. While there is significant uncertainty about the biophysical production of ecosystem services, there is additional uncertainty about the value of services. This paper explores how uncertainty associated with valuation of ecosystem services in agriculture affects the ranking of land use alternatives in terms of social net benefits. We compare the values of four land use scenarios in the Minnesota River Basin, USA, by combining a range of value estimates for these services with varying estimates for returns from agricultural production. Although variations in ecosystem service values are significant, fluctuations in agricultural returns more significantly determine how scenarios rank with regard to delivery of total value. This analysis suggests that addressing uncertainty in ecosystem service valuation is critical to accurately assessing tradeoffs in land use.

1. Introduction

Human livelihoods depend on a healthy environment and sufficient natural resources (Costanza and Daly, 1992). Yet the critical ways in which ecosystems support and enable human well-being are often undervalued in policy development and land use decision-making (Daily et al., 2009). Recognition of the critical importance of functioning ecosystems to human well-being has prompted efforts to quantify and value a range of the goods and services provided to humanity by ecosystem processes, known collectively as ecosystem services (e.g. Daily, 1997). Although research is advancing rapidly (e.g. Fisher et al., 2011; Nelson et al., 2009), significant uncertainty still limits our ability to generate value estimates rigorous enough to inform policies and land use decisions.

There is considerable uncertainty about both the biophysical production and economic values of ecosystem services. Uncertainty about how biophysical processes and biodiversity generate ecosystem services under varying ecological conditions is important (Daily et al., 2009; Euliss et al., 2010), but this type of uncertainty has received greater prior attention.

In this paper we focus on the considerable uncertainty that exists about the *value* of ecosystem services. Although some ecosystem services support the production of goods with observable market prices such as crops or timber, many ecosystem services have no such explicit signal of value. Measuring non-market values is an important source of uncertainty about ecosystem service values. Valuation techniques often capture only a portion of potential value (Mendelsohn and Olmstead, 2009) or provide hypothetical dollar amounts not connected to actual expenditures (Diamond and Hausman, 1994). Ecosystem service studies often do not value changes in ecosystem services or overlook how human preferences for ecosystem services might change, in potentially non-linear ways, with changing ecological conditions (Turner et al., 2010). Preferences, like biophysical processes, are context and place-specific. In addition to issues related to measuring non-market values, market prices are also subject to wide swings in value due to shifts in preferences or environmental conditions. Agricultural prices in particular are volatile and are affected by changes in weather patterns and market conditions. Corn prices have ranged between $2 per bushel in 2006 to between $6 and $7 per bushel in 2011 (http://future.aae.wisc.edu/data/monthly_values/by_area/2052?area=US). Prices for standardized commodities can also vary across space, again reinforcing that even values for commodities are context and place-specific (e.g., in 2010 corn farmers in Minnesota received on average about $.35 (7%) less per bushel of corn than the average Ohio corn farmer; USDA-NASS, 2011). Tradeoffs between non-market ecosystem services and marketed

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goods are significantly affected by this volatility. Market variability and valuation uncertainty presents considerable challenges to robust analysis of ecosystem services.

Uncertainty in the valuation of ecosystem services is particularly important in agricultural systems, because agriculture occupies nearly 40% of the terrestrial surface of the Earth (Ramankutty et al., 2008). The growth of agriculture in recent decades enabled an unprecedented increase in crop production and supplied food to a rapidly rising world population (Tilman et al., 2011). However, intensive agriculture also significantly impacts global ecosystems, for example consuming more than half of the world's supply of fresh water (Postel et al., 1996) and altering global nitrogen and carbon cycles (Matson et al., 1997). Also, conversion of land for agriculture has destroyed or degraded habitat and poses the single greatest threat to the planet's biological diversity (Green et al., 2005). The dietary and resource demands of a growing human population likely will prompt greater agricultural expansion and intensification and threaten to exacerbate the environmental impacts of agriculture (Tilman et al., 2011).

Efforts to mitigate the environmental costs of intensive agriculture must appropriately value non-market ecosystem services in addition to food and fiber production. Intensive production of one service, such as crops, almost always is associated with lower provision of other services (Bennett et al., 2009). But increasing provision of non-market ecosystem services could potentially reduce returns from agricultural production. Thorough evaluation of tradeoffs among market and non-market agricultural ecosystem services, both now and in the uncertain future, can inform policies and decisions that support “multifunctional” agricultural systems that provide a suite of ecosystem services (Jordan and Warner, 2010).

This paper explores the significance of uncertainty about the value of ecosystem services in analyzing tradeoffs among services in agricultural-ecological systems. We use spatially-explicit ecological production models and context-relevant market return and valuation estimates to calculate the expected change in ecosystem services from potential shifts in agricultural land use. Our analysis suggests that variability in ecosystem service benefit values are significant but that fluctuations in market returns from agriculture are likely to be more important in determining tradeoffs among competing land use activities. This research highlights the significance of incorporating economic uncertainty in ecosystem service valuation efforts and suggests that additional research is necessary to enhance the precision of ecosystem service analysis.

2. Case Study: the Minnesota River Basin

2.1. Ecosystem Services in the Minnesota River Basin

In this paper we analyze agricultural production and two important non-market ecosystem services, provision of clean water and climate regulation. We analyze how alternative land use patterns in the Minnesota River Basin, a region of nearly 4 million hectares in the Upper Mississippi River Basin dominated by agriculture (Musser et al., 2009), affect provision of these services. The area was primarily prairie, wetlands and deciduous forest in the late 1800's (Marschner, 1974), but was steadily converted to agricultural use in the 20th century. More than 90% of the land is managed for agriculture, predominantly corn and soybeans. Less than 5% of native ecosystems remain (MN DNR, 2006, p. 214) (Fig. 1).

Intense cultivation in this region generates high levels of nutrients and sediment, making the Minnesota River one of the most impaired waterways in Minnesota (Mulla, 1997). Phosphorus loading is of particular concern because excess phosphorus causes freshwater eutrophication harmful to aquatic ecosystems and can stimulate toxic algal blooms dangerous to human health (Carpenter et al., 1998). There is significant local and state interest in managing agricultural runoff and federal Total Maximum Daily Load (TMDL) guidelines could potentially require substantial changes in agricultural land use to reduce phosphorus loading.

Agricultural production also impacts climate regulation through carbon cycling and greenhouse gas emissions. Agriculture is estimated to contribute nearly a quarter of greenhouse gas emissions worldwide, and agriculture could contribute significantly to climate change mitigation through increased carbon storage and reduced greenhouse gas emissions (Burney et al., 2010). Large carbon emissions occur with conversion of native ecosystems for cultivation (Tilman et al., 2001). Agricultural practices also generate emissions of other potent greenhouse gases nitrous oxide (N_2O) and methane (CH_4) (Robertson et al., 2000). Improved management and production practices can simultaneously increase productivity and reduce agriculture's climate footprint (Foley et al., 2011; Smith et al., 2007; Tilman et al., 2011). Retiring less-productive agricultural lands and restoring perennial vegetation sequester carbon in soil and biomass (Johnson et al., 2007).

There are other valuable ecosystem services generated in the Minnesota River Basin that are not analyzed in this paper. In related research, Polasky et al. (2011) analyzed provision of habitat to support biological diversity and timber production. Flood mitigation is also relevant for this heavily-drained and flood-prone landscape. Improved ecosystem services models likely will support more complete analyses of a broader range of services in future research. Here we focus on the values of agricultural production, water quality and climate change mitigation.

2.2. Alternative land use scenarios

We analyze tradeoffs among ecosystem services under different land use scenarios. Since more than 90% of land is in agriculture and the region has low water quality, we focused on scenarios with graduated reductions in the extent of agricultural lands. We used the 2001 National Land Cover Dataset (NLCD) (USGS, 2010), which categorizes fifteen land cover classes for the Minnesota River Basin, to generate land use scenarios. A baseline scenario was created by using multi-year averages of both climatic and agricultural data in conjunction with the 2001 NLCD.

We created alternative scenarios by varying the extent and location of lands allocated to row crops, pasture/hay production, or perennial vegetation not used for agriculture. Retired agricultural land was restored to the native vegetation type for that specific location as identified by a map of pre-settlement vegetation for Minnesota (Marschner, 1974) obtained from the Minnesota Department of Natural Resources (http://deli.dnr.state.mn.us/). We used two selection strategies to identify agricultural lands for retirement. First, we focused on retiring agricultural lands within riparian zones. Stream buffers intercept and capture a portion of sediment and nutrients before they reach adjacent surface waters, and even relatively narrow buffers of 25 m width or less provide significant water quality improvements (Castelle et al., 1994). Larger buffers provide greater sediment and nutrient retention but with decreasing returns to increased width (Mayer et al., 2007). Second, we targeted lower-value agricultural uplands for retirement. Specifically, we used the USDA SSURGO land capability classes (USDA-NRCS, 2009) to identify row crop and pasture/hay lands on less-productive soil types.

We created three alternative land use scenarios: 1) Buffer-25: agricultural lands within 25 m of major rivers and streams were restored to perennial vegetation; 2) Buffer-100: agricultural lands within 100 m of streams and rivers were restored; 3) Buffer + Upland: low productivity agricultural lands were restored in addition to 100-meter stream buffers. The Buffer-25, Buffer-100 and Buffer + Upland scenarios increased the area of perennial vegetation types and reduced the extent of agricultural land in the Minnesota River Basin by approximately 2%, 8%, and 30% respectively (Appendix A Table A-1). Best management practices, such as use of cover crops (Strock et al., 2004), enhanced nutrient management (e.g. Robertson and Vitousek, 2009) and conservation tillage (e.g. Thoma et al., 2005) can potentially increase ecosystem

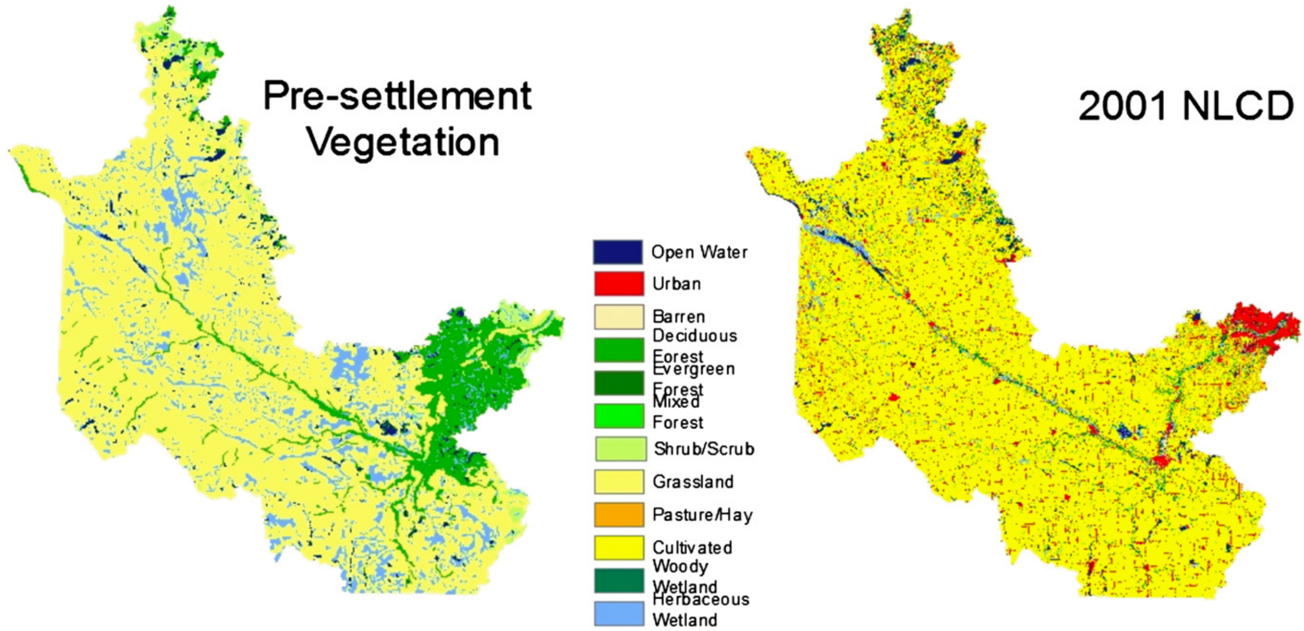

Fig. 1. Pre-settlement and 2001 LULC in the Minnesota River Basin.

services while maintaining agricultural production. In this paper, though, we only analyzed land retirement and restoration.

3. Methods: Ecosystem Service Production and Valuation Models

3.1. Climate Regulation Services

We used the Integrated Valuation of Ecosystem Services and Tradeoff (InVEST, Tallis et al., 2010) model to quantify changes in carbon sequestration and nitrous oxide (N_2O) emissions achieved by shifting from the baseline 2001 landscape to each of the alternative landscape scenarios. We used carbon storage values for each type of land use and land cover to calculate the carbon pools in soils and in above and below-ground biomass for each scenario (Fissore et al., 2009; Pennington, unpublished data). We assumed that baseline scenario in 2001 and each alternative scenario in 2050 would reach carbon storage equilibrium and we assumed a linear rate of sequestration between the baseline and alternative future landscape scenario equilibria.

We also estimated the potential reduction in N_2O emissions from land use changes and associated reductions in the application of nitrogen fertilizers. N_2O is a significant bi-product of fertilizer use and a potent heat-trapping gas, roughly 298 times more potent per molecule than CO_2 (Forster et al., 2007). We assumed an annual rate of nitrogen use of 125 kg/ha (112 lb/acre) for row crops (Rehm et al., 2008) and 25 kg/ha for alfalfa/hay and pasture (Rehm et al., 2000). These rates of nitrogen application include use of synthetic fertilizers, manure and nitrogen fixed by legume crops. We used an emissions factor of 1% to estimate N_2O emissions (IPCC, 2007). Using these assumptions, we estimate that cropland generates 1.25 kg of N_2O per hectare per year and alfalfa/hay and pasture generates 0.25 kg of N_2O per hectare per year. We multiplied the average annual rates of N_2O emissions per hectare by the total land area in each type of agriculture to generate estimates of basin-wide annual N_2O emissions for each scenario.

We monetized the value of carbon sequestration and N_2O emissions reductions using estimates of the social cost of carbon (SCC). SCC estimates assess the increase in damage to society, in terms of diminished economic output, health costs and other impacts, caused by more intense climate change resulting from an additional ton of carbon released into the atmosphere (Tol, 2009). Estimates of the SCC vary widely due to the uncertainties of climate science, damage projections, and treatment of equity issues and discounting (Tol, 2009). We applied a range of SCC estimates to incorporate uncertainty about scientific and economic variables, as well as about values. Due to the long time scales involved in climate change and the delay of significant impacts until far in the future, Nordhaus (2007) suggests an initial value of \$30 per ton of carbon (2005 dollars) rising at an approximate 2% annual rate of increase to \$85 by 2050. The Stern Review (2006) concluded that immediate significant reductions in greenhouse gases were necessary to avoid the risk of potentially catastrophic future damages. The combination of potentially large damages and much lower discounting led to a much higher SCC estimate of \$312 per ton. A review by the IPCC (Yohe et al., 2007) found estimates of the SCC ranging from \$10 to \$350 with a mean of \$43 per ton of carbon. For this analysis we calculated the net present value of carbon sequestration using SCC values (in 2001 dollars) of \$27/ton C (Nordhaus, 2007), \$106/ton (Tol, 2009) and \$171/ton (Anthoff et al., 2009). The \$106 SCC value (Tol, 2009) represents the median value of a recent meta-analysis of SCC estimates. The \$171 SCC (Anthoff et al., 2009) weights the SCC to more directly reflect uncertainty and equity considerations.

To convert the reduction of carbon and N_2O emissions across the landscape between 2001 and 2050 under each alternative scenario s into a monetary value, V_s, we used the following equation:

$$V_s = \sum_{t=0}^{49} SCC_t \left(\frac{C_{st} + CN_2O_{st}}{(1+\delta)^t} \right) \tag{1}$$

where SCC_t is the social cost of carbon in year t, C_{st} is carbon sequestered under alternative scenario s in year t, CN_2O_{st} is the carbon-equivalent avoided emissions of N_2O under alternative scenario s in year t relative to the baseline emissions, and δ is the annual discount rate (Conte et al., 2011). We assume that $SCC_t = SCC_0\,(1+r)^t$, where r is the annual rate of increase in SCC. We used r values of 0.02 and 0.04 (Table 3). We assume that the annual discount rate is 3%.

3.2. Nutrient Retention and Water Purification Services

We used the InVEST Tier 1 water yield and nutrient retention models (Tallis et al., 2010) to estimate loading of phosphorus to the Minnesota River under each alternative landscape scenario. The water yield model uses climate data coupled with soil information and the water use characteristics to calculate the contribution to surface flow from each site. Water flow is routed to streams and rivers using a digital elevation model. The nutrient retention model uses nutrient loading on site and from uphill sites along with filtration capacity to quantify nutrients delivered from the site. Climate data were obtained from the Minnesota State Climatology office. Soils data were downloaded from the SSURGO database (USDA-NRCS, 2009) and a digital elevation model was retrieved from the National Map Seamless Server (USGS, 2010). Information about the water use was generated using Allen et al. (1998) and Schenk and Jackson (2002). Nutrient loadings were derived from Reckhow et al. (1980), Oquist et al. (2007), and through consultation with experts familiar with regional agro-ecosystems and watersheds (D. Ennannay, pers comment; D. Mulla, pers comment). The nutrient filtration capacity to intercept and retain phosphorus was informed by Hanson et al. (2005) (wetlands) and through consultation with experts familiar with regional hydrology (D. Ennannay, pers. comment) (crop lands, forests, wetlands). Phosphorus loading estimates were calibrated by comparing model output for the baseline 2001 scenario with Minnesota Pollution Control Agency (2004) estimates of average annual phosphorus loading for the Minnesota River.

There is considerable uncertainty associated with estimating the value of water quality improvement. There is no single measure of the value of changes in water quality analogous to the social cost of carbon. Water quality is impacted by a variety of nutrients, bacteria, toxic chemicals and sediment. Additionally, humans use and benefit from clean water in a variety of ways including provision of drinking water and recreation, leading to the use of many different valuation approaches. Hedonic pricing methods have been applied to Minnesota lakes to assess the value of water quality (Krysel et al., 2003; Steinnes, 1992). Avoided cost analyses quantify the treatment costs that would be incurred with degraded water. One often cited example involves the Catskills watershed protection plan used by New York City to avoid billions of dollars in construction and operation of a water treatment plant (Chichilnisky and Heal, 1998). Other studies have estimated the cost of averting behaviors such as purchasing bottled water or installing expensive filtration systems (e.g. Poe and Bishop, 1999). Contingent valuation studies have been used to estimate how much people would pay for improved water quality. For example, Carson and Mitchell (1993) estimated that people would pay an average of $242 per person ($29.2 Billion in total for the U.S.) for “drinkable, swimmable and boatable” water (1990 dollars).

To account for uncertainty about the value of improved water quality, we applied a range of estimates. Our ‘high’ value was based on a contingent valuation study in the Minnesota River Basin in which households indicated an annual willingness-to-pay (WTP) for a 40% reduction in phosphorus loading to the Minnesota River (Mathews et al., 2002). For our ‘medium’ value we used a WTP derived by Egan et al. (2009) for improving impaired lakes in Iowa to the same water quality as the cleanest lake in the state. We generated a ‘low’ WTP using the benefit function specified by Johnston and Besedin (2009) and applying variables appropriate to phosphorus reductions in the Minnesota River Basin.

The InVEST water models calculate the total annual phosphorus loads under each alternative landscape scenario. The value estimate used from Mathews et al. (2002) corresponds directly to InVEST model results because it asked respondents about willingness-to-pay for a 40% phosphorus reduction. Assuming diminishing marginal returns to additional phosphorus mitigation decreases, we scaled the WTP for a 40% reduction to WTP for different levels of phosphorus reduction using the function:

$$WTP = \alpha X^{0.5} \tag{2}$$

where X is the percent phosphorus reduction achieved by each scenario and α is a coefficient calculated to match Mathews et al. (2002). For example using this equation, a WTP of $100 for a 40% phosphorus reduction yields α of 158. This coefficient then applied to a reduction of 80% generates an estimated WTP of approximately $141, accounting for the likely decreasing marginal value of additional nutrient reductions.

Johnston and Besedin (2009) and Egan et al. (2009) estimate the value of changes in water quality, fish habitat and recreation benefits. Using values from these studies required translating changes in phosphorus loading from InVEST to changes in water quality. Initial TMDL reports for sub-basins and tributaries within the Basin recommend reductions in phosphorus loading ranging from 32 to 97% (MPCA, 2004). Phosphorus loads for the entire basin likely would need to drop by 50% to achieve sufficient water quality improvement to fully support fishing, boating, swimming and other recreation (B. Dalzell, pers. comm.). For this analysis we assumed that the improvements in water quality and recreational benefits valued by the WTP estimates from Johnston and Besedin (2009) and Egan et al. (2009) would correspond with 50% reductions in phosphorus loading in the Minnesota River Basin. We applied Eq. (2) to the WTP estimates from these studies to derive value estimates for different levels of reductions. There is often a gap between the type of biophysical measures in environmental science (e.g., phosphorus reduction) and measures of benefits used in economic valuation studies (e.g., fishing, boating and swimming). Mapping from biophysical measures to economic measures is a further reason why there can be large uncertainty in estimates of the value of ecosystem services.

This approach yielded estimates of household WTP for 40% reductions in phosphorus of $154, $131, and $60 (2001 dollars). We multiplied these values by the 1,013,123 households included in the Mathews et al. (2002) valuation study to calculate whole-basin estimates of the annual value of phosphorus reductions achieved by each scenario. We calculated the net present values of water quality improvement benefits using a 3% annual discount rate for the period of analysis between 2001 and 2050.

3.3. Agricultural Production

Agricultural production, and in particular cultivation of corn and soybeans, is the dominant land use and primary economic activity in the Minnesota River Basin. Agriculture is on average a profitable endeavor generating high private returns to landowners. Agricultural returns are also highly variable, subject both to unpredictable weather patterns and fluctuations in commodity markets. We calculated several estimates of net present value of agricultural returns between 2001 and 2050 by applying a range of price and cost combinations to agricultural yield estimates.

We estimated yields for the two primary crops, corn and soybeans, as well as for alfalfa hay and pasture. Although other field crops such as sugar beets, wheat, peas and barley are cultivated in this region, corn and soybean cultivation typically occupies 90–95% of lands used for row crop agriculture (USDA-NASS, 2007). For this analysis we assumed that corn and soybeans each constitute 50% of row crop production in a typical year. We similarly assumed that lands in pasture/hay were equally divided between alfalfa cultivation and pasture production. We calculated yields for these four crops for each land use scenario by combining soil productivity maps generated from USDA SSURGO land capability classes (USDA-NRCS, 2009) with regional data about crop yield for each soil class. We assumed future yields would remain similar to current average yields.

Annual returns from these agricultural activities were calculated using regional crop price and production cost data from the Farm Financial Database (http:www.finbin.umn.edu). To incorporate crop price and production cost variability into our analysis we used data from 1993 to 2009 (converted to 2001 dollars) and identified the high, median and low commodity prices and production costs per hectare. We then created combinations of prices and costs to generate low, medium and high estimates for annual agricultural returns and calculated net present values for agricultural production for each landscape scenario assuming that each of these price/cost combinations would persist for the entire period of analysis. We did not include estimates of transition costs that would be incurred for converting existing agricultural land to pre-settlement vegetation types. Such one-time expenditures would reduce the net present value of land restoration to some degree.

4. Results

Each of the alternative land use scenarios realized substantial gains in climate regulation and provision of clean water relative to the baseline scenario. Conversion to perennial vegetation types with higher capacities to store carbon in soils and biomass significantly increased carbon sequestration. The Buffer-25 scenario increased carbon storage in the soil and biomass pools approximately 1% from 304 million Mg C to 307 million Mg carbon. The Buffer-100 scenario increased storage by 5% to 318 million Mg C. Conversion to the Buffer + Upland scenario from the baseline 2001 scenario increased the total carbon pool by 17% to 356 million Mg C (Table 1). The alternative scenarios also achieved significant reductions in N_2O emissions by decreasing the area of agricultural lands receiving nutrient applications. The Buffer-25 scenario reduced N_2O emissions by the carbon-equivalent of approximately 36,734 Mg C each year. The Buffer-100 scenario reduced annual N_2O emissions equivalent to 91,760 Mg C. The Buffer + Upland scenario achieved N_2O emissions reductions comparable to approximately 307,826 Mg C per year (Table 1).

Significant reductions in phosphorus loading to the Minnesota River Basin were also achieved by the alternative land use scenarios. We calculated annual phosphorus loadings under the baseline scenario to be 1339 Mg P, which is roughly equivalent to a loading of 1467 Mg P estimated for the basin in years of average rainfall (MPCA, 2004) (Table 1). The Buffer-25 scenario reduced annual phosphorus loading by 13.8% to 1154 Mg P. The wider stream buffers and upland restoration of the Buffer-100 and Buffer + Upland scenarios achieved much greater reductions of 792 Mg (59%) and 999 Mg P (75%), respectively.

The increases in water quality and climate regulation services accomplished by the alternative scenarios generated significant economic value. Including both increased carbon storage and reduced N_2O emissions, the present value of climate regulation services for the Buffer + Upland scenario ranged from \$1.5 billion to \$15.3 billion (Table 2). The climate regulation values for the other scenarios were substantially lower, ranging from a present value of \$405 million to \$4.2 billion for the Buffer-100 scenario and between \$115 million and \$1.2 billion for the Buffer-25 scenario (Table 2). The values of reductions in phosphorus loading achieved by the alternative scenarios were also substantial. The 14% reduction in phosphorus loading achieved by the Buffer-25 scenario generated net present values of resulting water quality benefits ranging from \$885 million to \$2.8 billion (Table 2). The Buffer-100 scenario reduced phosphorus loading by almost 60% and provided benefits ranging from \$1.8 to \$4.8 billion. The extensive agricultural land retirement in the Buffer + Upland scenario lowered estimated phosphorus loadings by almost 75% generating between \$2.1 and \$5.6 billion worth of increased water quality benefits. Although the Buffer + Upland scenario provided six times the reduction in phosphorus loadings compared to the Buffer-25 scenario, declining marginal returns generated only twice as much monetary value (see Eq. (2)).

Table 1
Biophysical results of ecosystem service models for alternative landscape scenarios.

Scenarios	Carbon storage (Million Mg C)	N_2O emissions (Million Mg C – equivalent yr^{-1})	Total P loading (Mg P yr^{-1})
2001	303.75	1.09	1339.2
Buffer-25	306.98	1.05	1153.8
Buffer-100	317.80	1.00	546.7
Buffer + Upland	356.20	.78	340.3

The reduction in agricultural land areas resulted in significant losses in estimated returns from agricultural production. The estimated net present value of agricultural returns from the 2001 baseline scenario ranged from \$12.2 to \$45.5 billion. The Buffer-25 scenario retired approximately 2% of agricultural land and reduced the net present values of agricultural production by \$228 to \$919 million. The significantly wider stream buffers of the Buffer-100 scenario reduced agricultural lands by nearly 9% and decreased the net present value of agricultural returns by \$958 million to \$3.7 billion. The additional retirement of upland agricultural areas in the Buffer + Upland scenario reduced the agricultural extent by approximately 30% or more than 916,000 ha as compared to the baseline 2001 scenario. Although this substantial conversion targeted lower-productivity soils to minimize lost agricultural revenue per hectare, this scenario nonetheless reduced the net present value of agricultural returns by \$1.7 to nearly \$10 billion.

We used combinations of price, cost and benefit estimates to evaluate net benefits of each alternative scenario and assess the tradeoffs between agricultural production and ecosystem service values. We find that the estimated social value of provision of additional ecosystem services generally exceed revenue lost from agricultural land retirement, except when estimated agricultural returns are high and non-market service values are low (Table 3; Figs. 2–4). When using a low estimate of future agricultural returns, the values of increased climate change mitigation and water quality improvement services were always sufficient to more than compensate lost agricultural revenues (Table 3). Using a medium estimate of future agricultural returns, increased ecosystem service benefits almost always exceeds agricultural revenue lost. Only for the lowest value estimates for both carbon and water quality does the change in agricultural revenue lost outweigh gains in ecosystem service benefits and this occurs only in the Buffer + Upland scenario (Table 3). However, using high agricultural return estimates, ecosystem services generated by both the Buffer-100 and Buffer + Upland scenario compensate for lost agricultural revenue under medium and high ecosystem service values, but not under low value estimates.

Our analysis indicates that the Buffer-25 scenario ranks higher than the baseline 2001 scenario for all combinations of water quality value and SCC estimates. The Buffer-100 scenario ranks higher than the baseline 2001 in 25 out of 27 agricultural return and ecosystem service value combinations. The most extensive agricultural land retirement of the Buffer + Upland scenario generates higher total net

Table 2
Change in high, medium and low values of agricultural returns and non-market ecosystem services for alternative scenarios compared to 2001 baseline.

		Net present value 2001 to 2050 ($ millions, 2001 dollars)		
		Buffer-25	Buffer-100	Buffer + Upland
Agricultural returns (price/cost 1993–2009)				
Low		−228	−958	−1,713
Median		−442	−1809	−4274
High		−919	−3721	−9955
Value of water quality improvements (WTP for 40% P reduction)				
\$60		885	1.833	2149
\$131		2386	4083	4750
\$154		2796	4785	5566
Value of climate change mitigation (SCC = \$/ton C)				
2% annual SCC increase	\$27	115	405	1468
	\$106	450	1591	5763
	\$171	726	2568	9297
4% annual SCC increase	\$27	188	666	2411
	\$106	737	2613	9463
	\$171	1190	4215	15,266

present values for 22 out of 27 agricultural return and ecosystem service value combinations.

5. Discussion

This study of agricultural ecosystem services highlights the importance of incorporating uncertainty into assessments of tradeoffs among ecosystem services. The enhanced provision of clean water and climate regulation services achieved by agricultural land retirement in the Minnesota River Basin are certainly valuable. However, the variability of agricultural returns is more significant than the uncertainty in non-market ecosystem service valuation in determining the ranking of alternative land use scenarios. If future agricultural returns from this region are low or moderate, then conversion of the 2001 landscape to any of the alternative landscape scenarios would realize significant gains in total economic value under almost every combination of ecosystem service values. If agriculture from 2001 to 2050 generates profits similar to the unusually high returns realized in 2006 and 2007, then modest agricultural land retirement would be justified, unless ecosystem service values are fairly high. Our analysis indicates that if future agricultural returns from the Minnesota River Basin are comparable in real dollars to the returns experienced from 1993 to 2009, then the 2001 landscape configuration will under-provide total economic returns under all combinations of ecosystem service values.

Our analysis constitutes a first step in confronting economic uncertainty in ecosystem service valuation, and subsequent research could address additional critical uncertainties. More thorough assessments could quantify all relevant ecosystem services to more accurately compare the total potential value of alternative land use scenarios. For example, retirement of agricultural land and conversion to perennials could reduce loading of other pollutants besides phosphorus (e.g., nitrogen), mitigate downstream flooding, and increase habitat for species of value to hunters, bird watchers or other recreationists. Inclusion of estimated values for these and other relevant ecosystem services would increase the total non-market value generated by alternative landscape scenarios.

Agricultural ecosystem services analysis would also be enhanced by consideration of best management practices in addition to land retirement. Best management practices offer considerable opportunities to increase provision of some important ecosystem services. Effective use of these practices could provide valuable ecosystem service benefits with potentially little loss of revenue from agricultural production.

Table 3
Change in total value for alternative scenarios compared to 2001 baseline. Ecosystem service values are: Low = SCC of \$27 w/2% increase & WTP of \$60; Medium = SCC of \$106 w/2% increase & WTP of \$131; High = SCC of \$171 w/4% increase & WTP of \$154. (BOLD denotes value combinations for which alternative scenarios generate estimated total value lower than the 2001 baseline.)

Scenarios	Total net present value of non-market ecosystem services and agricultural returns (\$ millions, 2001 dollars)			
	Combined value of non-market ecosystem services	Value of agricultural returns		
		Low	Medium	High
Buffer-25	Low	772	558	81
	Medium	2608	2394	1917
	High	3758	3544	3067
Buffer-100	Low	1281	429	**−1482**
	Medium	4716	3865	1953
	High	8042	7190	5279
Buffer + Upland	Low	1904	**−657**	**−6339**
	Medium	8800	6239	558
	High	19,119	16,559	10,877

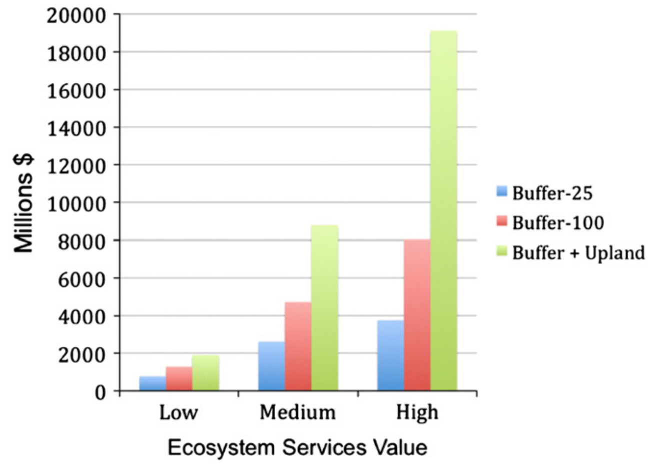

Fig. 2. Change from 2001 scenario in total net present value of alternative scenarios, assuming LOW agricultural returns.Ecosystem services values are: low = SCC of \$27 w/2% increase & WTP of \$60; medium = SCC of \$106 w/2% increase & WTP of \$131; high = SCC of \$171 w/4% increase & WTP of \$154.

Important methodological advancements also could strengthen analysis of ecosystem services and bolster the utility of such research for policy development and land use decision-making. In particular, better linkages between biophysical models of service production and economic models of value of services could close the large gap between natural science research and economic valuation. Clear focus on “final” ecosystem services of direct benefit to society rather than “intermediate” services could establish explicit connections between biophysical changes and changes in human welfare and help avoid incorrectly double-counting ecosystem service benefits (Boyd and Banzhaf, 2007; Fisher and Turner, 2008).

Additionally, engagement of farmers or other stakeholders could reduce uncertainty related to individual land management decisions by linking on-the-ground practices with the quantification and valuation of relevant ecosystem services. Use of participatory processes also could better incorporate ethical considerations, issues of social justice, and notions of uncertainty and irreversibility into ecosystem service analysis (Wilson and Howarth, 2002).

Inherent uncertainty in the production and valuation of ecosystem services complicates efforts to assess tradeoffs among alternative land use practices. We presented an initial effort to cope with economic

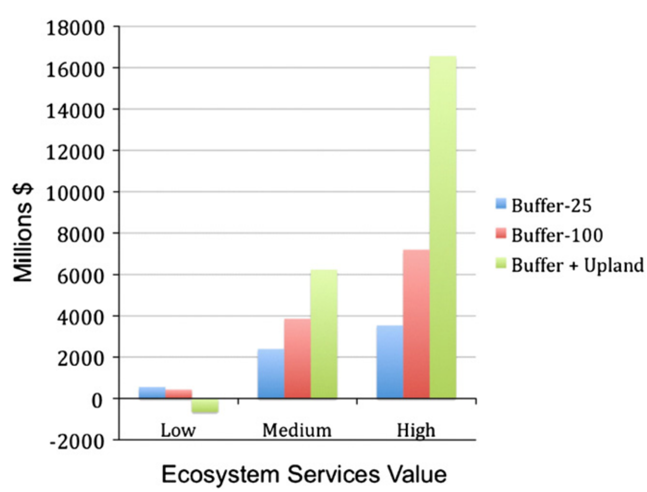

Fig. 3. Change from 2001 scenario in total net present value of alternative scenarios, assuming MEDIUM agricultural returns.Ecosystem services values are: low = SCC of \$27 w/2% increase & WTP of \$60; medium = SCC of \$106 w/2% increase & WTP of \$131; high = SCC of \$171 w/4% increase & WTP of \$154.

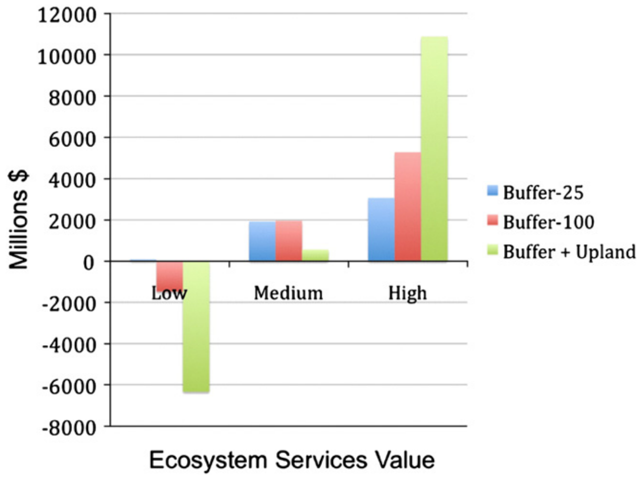

Fig. 4. Change from 2001 scenario in total net present value of alternative scenarios, assuming HIGH agricultural returns.Ecosystem services values are: low = SCC of $27 w/2% increase & WTP of $60; medium = SCC of $106 w/2% increase & WTP of $131; high = SCC of $171 w/4% increase & WTP of $154.

uncertainty, but incorporation of the above additional elements in future research could further reduce critical uncertainties. Efforts to enhance the rigor of ecosystem service valuation in agricultural systems are greatly needed, to highlight agricultural landscape configurations that might be socially preferable and to provide guidance regarding how to balance demands for provision of both food and fiber and non-market ecosystem services.

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Appendix A

Table A-1
Area in each LULC category for 2001 baseline scenario and alternative BMP and Restoration scenarios.

LULC class	Area in hectares				
	Pre-settlement	2001	Buffer-25	Buffer-100	Buffer + Upland
Open water	84,424	119,574	120,027	120,955	128,813
Urban		266,642	266,594	266,594	265,998
Barren	124	3983	3984	3989	4008
Deciduous forest	418,808	113,386	119,838	137,901	217,873
Evergreen forest		3456	3456	3456	3456
Mixed forest		1201	1201	1201	1200
Shrub/scrub	135,746	10,679	11,367	14,298	44,241
Grassland	2,785,830	90,752	140,362	293,351	743,897
Pasture/hay		188,761	180,598	162,773	76,165
Cultivated		2,888,297	2,827,434	2,647,157	2,084,438
Woody wetland	456	26,128	26,112	26,129	26,078
Herbaceous wetland	448,506	163,184	170,939	193,906	274,563

Note: total areas of scenarios differ very slightly (by less than .2%) due to imperfect alignment of different soil and land use input maps.

Table A-2
LULC class descriptions for the National Land Cover Dataset (http://www.mrlc.gov/faq.php).

Code	Class	Descriptions
11	Open water	All areas of open water, generally with less than 25% vegetation or soil cover
21	Developed, open space	Includes areas with a mixture of some constructed materials, but mostly vegetation in the form of lawn grasses. Impervious surfaces account for less than 20% of total cover. These areas most commonly include large-lot single-family housing units, parks, golf courses, and vegetation planted in developed settings for recreation, erosion control, or aesthetic purposes.
22	Developed, low intensity	Includes areas with a mixture of constructed materials and vegetation. Impervious surfaces account for 20–49% of total cover. These areas most commonly include single-family housing units.
23	Developed, medium intensity	Includes areas with a mixture of constructed materials and vegetation. Impervious surfaces account for 50-79% of the total cover. These areas most commonly include single-family housing units.
24	Developed, high intensity	Includes highly developed areas where people reside or work in high numbers. Examples include apartment complexes, row houses and commercial/industrial. Impervious surfaces account for 80 to100% of the total cover.
31	Barren land (rock/sand/clay)	Barren areas of bedrock, desert pavement, scarps, talus, slides, volcanic material, glacial debris, sand dunes, strip mines, gravel pits and other accumulations of earthen material. Generally, vegetation accounts for less than 15% of total cover.
41	Deciduous forest	Areas dominated by trees generally greater than 5 m tall, and greater than 20% of total vegetation cover. More than 75% of the tree species shed foliage simultaneously in response to seasonal change.
42	Evergreen forest	Areas dominated by trees generally greater than 5 m tall, and greater than 20% of total vegetation cover. More than 75% of the tree species maintain their leaves all year. Canopy is never without green foliage.
43	Mixed forest	Areas dominated by trees generally greater than 5 m tall, and greater than 20% of total vegetation cover. Neither deciduous nor evergreen species are greater than 75% of total tree cover.
52	Shrub/scrub	Areas dominated by shrubs; less than 5 m tall with shrub canopy typically greater than 20% of total vegetation. This class includes true shrubs, young trees in an early successional stage or trees stunted from environmental conditions.
71	Grassland	Areas dominated by grammanoid or herbaceous vegetation, generally greater than 80% of total vegetation. These areas are not subject to intensive management such as tilling, but can be utilized for grazing.
81	Pasture/hay	Areas of grasses, legumes, or grass–legume mixtures planted for livestock grazing or the production of seed or hay crops, typically on a perennial cycle. Pasture/hay vegetation accounts for greater than 20% of total vegetation.
82	Cultivated crops	Areas used for the production of annual crops, such as corn, soybeans, vegetables, tobacco, and cotton, and also perennial woody crops such as orchards and vineyards. Crop vegetation accounts for greater than 20% of total vegetation. This class also includes all land being actively tilled.
90	Woody wetlands	Areas where forest or shrub land vegetation accounts for greater than 20% of vegetative cover and the soil or substrate is periodically saturated with or covered with water.
95	Emergent herbaceous wetlands	Areas where perennial herbaceous vegetation accounts for greater than 80% of vegetative cover and the soil or substrate is periodically saturated with or covered with water.

Table A-3
Prices and costs of major agricultural commodities analyzed (2001 $). Yield data obtained from USDA-NRCS (2009). Average price and cost derived from 1993 to 2009 data for southwest and west regions of Minnesota from the Farm Financial Database, http://www.finbin.umn.edu/.

1993–2009	Price per			Production costs/ha		
	Low price	Median price	High price	Low costs	Median costs	High costs
Corn for grain (bushels)	1.75	2.16	3.22	530.28	586.80	967.95
Soybeans (bushels)	5.10	6.03	7.85	305.08	355.18	444.83
Pasture (aum)	1.10	8.87	10.35	13.53	24.13	35.15
Alfalfa (short ton)	72.94	82.42	101.20	317.43	395.15	535.53

Table A-4
Carbon model input table. Metric tons of soil organic carbon (SOC) and biomass carbon per hectare by LULC type. Frank et al., 1995; Slobodian et al., 2002; Al-Kaisi et al., 2005; Liebig et al., 2005; Bedard-Haughn et al., 2006; Euliss et al., 2006; Omonode et al., 2007; Fissore et al., 2009.

LULC	Mean SOC	Mean Biomass C
Open water	0	0
Developed, open space	82.7	17.3
Developed, low intensity	82.7	17.3
Developed, medium intensity	82.7	17.3
Developed, high intensity	82.7	17.3
Barren	0	0
Deciduous forest	138.8	108.8
Evergreen forest	120.4	144.8
Mixed forest	135.1	116
Shrub/scrub	118.8	59.5
Grassland	98.75	10.05
Pasture/hay	77.55	6.7
Cultivated	66.6	1.1
Woody wetland	123.8	0
Herbaceous wetland	123.8	0

Table A-5
Water model input table. Rooting depth (Schenk and Jackson, 2002), evapotranspiration coefficient (ETK) (Allen et al., 1998), phosphorus loading (Reckhow et al., 1980) and filtration (Ennaanay personal comment), and nitrogen loading (Reckhow et al., 1980; Oquist et al., 2007; Ennaanay personal comment) and filtration (Ennaanay personal comment) by LULC type.

LULC	Rooting depth (cm)	ETK	P loading (g ha^{-1})	P filtration (%)
Open water	1	1	1	0
Developed, open space	1	1	1100	0
Developed, low intensity	1	1	1100	0
Developed, medium intensity	1	1	1100	0
Developed, high intensity	1	1	1100	0
Barren	1	1	50	5
Deciduous forest	104	1000	90	60
Evergreen forest	104	1000	60	60
Mixed forest	104	1000	90	60
Shrub/scrub	91	750	90	50
Grassland	91	750	100	50
Pasture/hay	91	750	640	30
Cultivated	64	500	950	5
Woody wetland	64	500	1	60
Herbaceous wetland	1	400	1	60

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