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The Impact of Land-Use Change on Ecosystem Services, Biodiversity and Returns to Landowners: A Case Study in the State of Minnesota

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Abstract Land-use change has a significant impact on the world's ecosystems. Changes in the extent and composition of forests, grasslands, wetlands and other ecosystems have large impacts on the provision of ecosystem services, biodiversity conservation and returns to landowners. While the change in private returns to landowners due to land-use change can often be measured, changes in the supply and value of ecosystem services and the provision of biodiversity conservation have been harder to quantify. In this paper we use a spatially explicit integrated modeling tool (InVEST) to quantify the changes in ecosystem services, habitat for biodiversity, and returns to landowners from land-use change in Minnesota from 1992 to 2001. We evaluate the impact of actual land-use change and a suite of alternative land-use change scenarios. We find a lack of concordance in the ranking of baseline and alternative land-use scenarios in terms of generation of private returns to landowners and net social benefits (private returns plus ecosystem service value). Returns to landowners are highest in a scenario with large-scale agricultural expansion. This scenario, however, generated the lowest net social benefits across all scenarios considered because of large losses in stored carbon and negative impacts on water quality. Further, this scenario resulted in the largest decline in habitat quality for general terrestrial biodiversity and forest songbirds. Our results illustrate the importance of taking ecosystem services into account in land-use and

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land-management decision-making and linking such decisions to incentives that accurately reflect social returns.

Keywords Ecosystem services · Biodiversity · Land use · Private returns to landowners · Net social benefits · Tradeoffs

1 Introduction

Land-use and land-management decisions have major impacts on ecosystems and the goods and services they provide to people ("ecosystem services;" Daily 1997). In this paper, we use the term ecosystem services quite broadly to include any ecosystem process or function that contributes to human well-being. Ecosystem services include carbon sequestration because of its positive impact on climate regulation, nutrient retention because of its positive impact on water quality, water flow timing because of its role in flood and drought mitigation, and inputs to the production of agricultural crops (e.g.,soil productivity, pollination), among others. Changes in land use or land management (agricultural practices, forestry practices, intensity of development) can cause changes in the provision and value of ecosystem services. In general, changes in land use or land management will increase the provision and value of some services but decrease others. Additional tradeoffs are introduced if we are also concerned with other objectives such as biodiversity conservation. Land-use decisions intended to maximize a single output such as agricultural production or timber production are likely to generate an accompanying decline in the provision of other services (MA 2005). Optimal land use and land management requires joint consideration of the value of all objectives.

While the general notion of tradeoffs among objectives in land use and land management is understood in principle, in practice we typically lack the ability to predict how specific land-use or land-management decisions will affect the overall value derived from a landscape (Balmford et al. 2002; MA 2005; NRC 2005). Part of our inability to determine overall value stems from the standard problem in environmental economics of assessing non-market values. Most ecosystem services are not directly traded in markets and lack readily observable signals of value, though economists have made important progress on non-market valuation techniques and have applied these techniques to value a wide range of environmental benefits (e.g., Champ et al. 2003; Freeman 2003). A more novel difficulty in estimating the value of ecosystem services stems from the lack of understanding of how the provision of ecosystem services is affected by changes in land use or land management. In other words, we often lack "ecological production functions" to predict the provision of ecosystem services as a function of ecosystem conditions (NRC 2005; Daily et al. 2009). Understanding how ecosystem service provision and values change as land use and management changes requires joint research among ecologists, economists, and others. Ecologists and other natural scientists study ecosystem processes essential for understanding ecological production functions. Economists analyze decision-making that determines land use and land management, and analyze the value of ecosystem services. Close integration between ecologists and economists is necessary to ensure that choices, ecological production functions, and valuation are linked.

In this paper, we use the InVEST model (Integrated Valuation of Ecosystem Services and Tradeoffs; Tallis et al. 2008, http://invest.ecoinformatics.org/) to calculate the provision and value of ecosystem services and species habitat under alternative land use scenarios. InVEST was developed as part of the Natural Capital Project (www.naturalcapitalproject.org), a partnership between Stanford University, the University of Minnesota, The Nature Conservancy, and World Wildlife Fund, whose aim is to align economic forces with conservation. InVEST uses maps and tabular data of land use and land management in conjunction with environmental information (e.g., soil, topography and climate) to generate spatially explicit predictions of the biophysical supply of ecosystem services. Economic information about demand for ecosystem services can be combined with biophysical supply to generate predictive maps of service use and value (Daily et al. 2009; Nelson et al. 2009). InVEST also analyzes the impact of land use and land management on species habitat provision and quality. InVEST thus provides a powerful tool for simultaneously quantifying and valuing multiple ecosystem services generated by a landscape. By varying land use or land management and evaluating the output from InVEST we can provide information useful to managers and policy-makers weighing the tradeoffs in ecosystem services, biodiversity conservation, and other land-use objectives.

We illustrate the application of InVEST in analyzing tradeoffs among ecosystem services and other land-use objectives using biophysical and economic data from the state of Minnesota, USA. For this application we model carbon sequestration, water quality (phosphorus exports), habitat quality for grassland and forest birds and general terrestrial biodiversity, agricultural and timber production, and the value of land use in urban development. We chose to include these outputs because of their importance in Minnesota as well as availability of data and models. Agriculture makes up a large fraction of land use in the state (44.1% in 1992 and 43.8% in 2001) except in the northeast where forests dominate (Fry et al. 2009). Urban land was the fastest growing land-use in Minnesota from 1992 to 2001 (urban area increased by 2.6% from 1992 to 2001) and is the most highly valued land use in the state (Lubowski 2002). Carbon sequestration in terrestrial ecosystems has emerged as an important issue in international climate negotiations and there is great interest in giving climate change mitigation credit to land use or land management that results in higher levels of carbon storage (e.g., Angelsen 2008; Canadell and Raupach 2008). Further, the state of Minnesota has passed a law that mandates aggressive carbon emissions reductions, to which land-based sequestration can contribute (Minnesota Department of Commerce 2007). Water quality is a major environmental issue in the state and throughout the Mississippi Basin; excess nutrients from agriculture and other sources have significant local impacts (Mathews et al. 2002; Westra et al. 2002) and contribute to the hypoxic zone in the Gulf of Mexico as well (Turner et al. 2008).

Not all of the outputs we track are ecosystem services. Biodiversity is an important determinant of ecosystem processes and may contribute to the provision of many services, but other than existence value for species, we do not consider it to be an ecosystem service in itself. Urban development is not an ecosystem service, although ecosystem service supply can affect its value. We include the value of urban development in the analysis because we are interested in comparing the full value of alternative land-use decisions.

While there have been many studies that quantify the opportunity cost of providing a single ecosystem service (e.g., Kindermann et al. 2008 for carbon sequestration, Swallow et al. 2009 for sediment delivery), and similarly for biodiversity conservation (e.g., Naidoo and Ricketts 2006; Polasky et al. 2008), there have been relatively few studies on the joint provision and value of multiple outcomes (multiple ecosystem services, biodiversity conservation, and returns to landowners) from ecosystems. Several studies have characterized the spatial overlap among ecosystem service provision and biodiversity conservation (Chan et al. 2006; Egoh et al. 2008; Naidoo et al. 2008) but these studies have not analyzed how alternative land-use or land-management decisions affect the provision and value of ecosystem services or biodiversity conservation. Boody et al. (2005) and Santelmann et al. (2004) assess the provision of multiple ecosystem services and returns to agriculture for small agricultural watersheds in the U.S. Midwest while Naidoo and Ricketts (2006) assessed areas where the summed value of ecosystem services with conservation exceeded the opportunity cost of development in the Mbaracayu Biosphere Reserve in Paraguay. Nelson et al. (2008) found that land conservation incentive programs designed for biodiversity conservation do not necessarily do a good job of providing carbon sequestration services and vice versa for an application in the Willamette Basin, Oregon. The closest prior paper to the current paper is Nelson et al. (2009) who compared biodiversity conservation and ecosystem service outcomes under three alternative land-use trajectories for the Willamette Basin. They found that all biodiversity and ecosystem service measures were highest under a conservation scenario, but that returns to landowners were higher under more development oriented scenarios. Compared to Nelson et al. (2009), the current paper uses an improved biophysical model of water supply and nutrients to predict water quality, information on the value of water quality, and actual historical land-use change in addition to scenarios of land-use change.

In our results we do not find a single land-use scenario that provides higher levels of all ecosystem services and habitat. A land-use scenario that significantly expanded conserved area from 1992 to 2001 is best for the reduction of phosphorus exported into the Mississippi River but a scenario that prevented agricultural expansion from 1992 to 2001 is best for carbon sequestration. The best scenario for habitat provision depends on the group of species being considered. Like Nelson et al. (2009), however, we find a lack of concordance in the ranking of alternatives between net social benefits and the market value of returns to landowners, which suggests that there is a large role for policy in land-use decisions, a point that we return to in the conclusions.

In the next section, we describe the land-use change scenarios, the InVEST model and the data for the Minnesota application. We present results in Sect. 3. Section 4 contains discussion of important results and open questions.

2 Description of the InVEST Model Application in Minnesota

InVEST is a set of Geographic Information Systems models that predict the provision and value of ecosystem services and habitat provision given land use / land cover (LULC) maps and related biophysical, economic, and institutional data for the study region. In part A, we describe creation of the land-use scenarios for Minnesota. In part B, we describe the InVEST modules included in this analysis: carbon storage, water quality, habitat provision, and agricultural production. We also describe the data we use to estimate the value of timber production and urban development.

2.1 Scenarios

We use the National Land Cover Database 1992/2001 Retrofit Land Cover Change Product (Fry et al. 2009) to generate the baseline LULC change map for Minnesota. The map has a spatial grain of 30 m grid cells. Definitions for LULC types are provided in Table 1 (see the on-line appendix for additional details). Land-use change summary statistics for these LULC types are provided in Table 2. We also developed five alternative scenarios of LULC change for the 1992–2001 time period. These alternative scenarios are meant to be illustrative in showing how tradeoffs can be analyzed and are simple and easy to explain rather than attempts at realistic depiction of plausible land-use outcomes. The five alternative land-use scenarios are:

Table 1 Land Use Land Cover (LULC) definitions from the NLCD 1992/2001 Retrofit Land Cover Change Product used in the scenarios for Minnesota

LULC Class	Descriptions
Open water	All areas of open water, generally with less than 25% vegetation or soil cover
Urban	Includes developed open spaces with a mixture of some constructed materials, and lands of low, medium, and high development intensity
Barren	Areas of bedrock, pavement, gravel pits, and other accumulations of earthen material. Generally, vegetation accounts for less than 15% of total cover
Forest	Areas dominated by trees generally taller than 5 meters. Includes deciduous forest, evergreen forest, and mixed forest
Grassland/Shrub	Includes grassland areas dominated by gramminoid or herbaceous vegetation and shrub/scrub areas dominated by shrubs less than 5 meters tall with shrub canopy
Agriculture	Includes cultivated crops, pasture and hayfields
Wetlands	Includes woody wetlands and herbaceous wetlands

Source: http://www.mrlc.gov/faq.php. For more detailed descriptions see the on-line appendix

1. *No agricultural expansion*: no new land is put into agricultural production between 1992 and 2001.
2. *No urban expansion*: no new land is put into urban development between 1992 and 2001.
3. *Agricultural expansion*: all highly productive land for agriculture outside of urban areas is put into agriculture by 2001.
4. *Forestry expansion*: all highly productive forestry land in the northeast portion of the state, and outside urban areas, is put into forestry by 2001.
5. *Conservation*: almost all land within 100 meters of streams in the Minnesota River Basin and agricultural lands with marginal soils throughout the rest of the state are restored to natural vegetative covers by 2001.

See Fig. 1 and the on-line appendix for a summary of land-use change statistics under each alternative scenario.

2.2 Ecosystem Services, Biodiversity and Returns to Landowner Models

We use InVEST to model the change in the provision and value of carbon storage, water quality, and agricultural production across the state from 1992 to 2001. We report the monetary values of ecosystem services in terms of the value of annual flow of services. For carbon sequestration, we divide the change in carbon storage from 1992 to 2001 by the number of years of carbon storage flux from 1992 to 2001 to convert change in stock to an annual flow value. We also use InVEST to model the change in habitat availability and quality. The commercial value of forests and urban development are calculated using estimated average annual returns to these land uses from 1992 and 2002 (Lubowski 2002; Lubowski et al. 2006).

Table 2 LULC Change by LULC Type from 1992 to 2001

	To…							
Change in acres	Agriculture	Barren	Forest	Grassland/shrub	Open water	Urban	Wetlands	1992 Totals
From …								
Agriculture	24,120,804	2,021	97,638	32,365	86,360	43,362	110,464	24,493,015
Barren	120	63,541	1,458	182	3,807	31	897	70,036
Forest	89,799	6,689	14,393,111	19,130	14,044	22,424	112,967	14,658,164
Grassland/shrub	36,275	149	88,885	2,093,448	382	9,278	21,756	2,250,173
Open water	12,487	3,127	21,627	3,287	3,032,070	742	25,059	3,098,400
Urban	6,523	13	2,189	661	3,134	2,656,976	4,360	2,673,857
Wetlands	71,394	557	167,702	23,349	17,577	6,966	6,453,303	6,740,849
2001 Totals	24,337,402	76,098	14,772,610	2,172,424	3,157,374	2,739,779	6,728,806	

2.2.1 Carbon Storage and Sequestration

The carbon model accounts for carbon stored in above-ground and below-ground biomass and in the soil. The amount of carbon stored in each of these pools depends primarily on LULC (e.g., row crops, managed forest, pasture, natural prairie, wetlands, unmanaged conifer forest) but is also affected by land management (e.g., whether the land is protected or managed for timber, and the forest rotation age for timber land). For carbon storage in 1992 we assume that land use and land management had existed long enough in each grid cell for carbon storage in the cell to reach its equilibrium (steady-state) level. We assumed storage equilibrium because we lacked state-wide data on age class of forests and other LULC that would allow for a more exact estimation of carbon storage values in 1992. The one exception to this steady-state rule is for forests in cells set aside for conservation (e.g., state or national parks, wilderness areas, and other lands that restrict economic activity). For these forest cells, we assumed that forests would continue to mature between 1992 and 2001 and sequester carbon. Sequestration dynamics in publically conserved forest grid cells and steady-state levels for all LULC types are listed in the on-line appendix.

For those grid cells that change LULC between 1992 and 2001 we assume the change occurs in 1996, meaning there are 5 years of carbon storage flux on these cells (except for change from publically conserved forests; there are 10 years of biomass carbon flux on these cells). We calculate the change in carbon storage in a grid cell by taking its 2001 land use's equilibrium storage value less its 1992 land use's equilibrium storage value and prorating the difference by how long it takes to transition to the new storage value. For example, if it would take 100 years for full carbon accumulation in going from agricultural land to forest, then we would take 5% of the difference between timber and agricultural land use carbon storage values as the carbon sequestered during this time period. We convert changes in carbon stock to annualized flow of carbon sequestration by dividing by five, the number of years over which carbon stocks change due to LULC change (again, the calculation is slightly different for changes involving carbon biomass stock in publically conserved forests). The on-line appendix has all the details on the sequestration dynamics associated with a LULC change.

The annualized sequestration output from the carbon model can either be reported as tons of carbon sequestered, or it can be converted to a dollar value by using estimates of the social cost of carbon, carbon market prices, or estimates of the cost of carbon capture and storage (Hill et al. 2009). Here we report the value of annualized sequestration using median and mean estimates of the social cost of carbon from peer-reviewed studies (Tol 2009). The social cost of carbon is an estimate of the incremental damage caused by climate change due to the emission of one more ton of carbon into the atmosphere.

2.2.2 Habitat Extent and Quality

The InVEST habitat model maps the extent and quality of habitat for a target conservation objective (e.g., forest birds, amphibians, etc). Maps of LULC are transformed into maps of habitat by defining what LULC counts as habitat for various species. Habitat quality in a grid cell is a function of the LULC in the grid cell, the LULC in surrounding grid cells, and the sensitivity of the habitat in the grid cell to the threats posed by the surrounding LULC.

Whether a particular LULC type is considered species habitat depends on the objective of biodiversity conservation. In the Minnesota application, we consider three different conservation objectives: (i) general terrestrial biodiversity that includes all native species, (ii) functional group diversity focusing on breeding forest interior songbirds, and (iii) functional group diversity focusing on breeding grassland songbirds (based on Ehrlich et al. 1988).

Each LULC type is given a habitat suitability or quality score of 0 to 1 for each particular measure of biodiversity with non-habitat scored as 0 and perfectly suitable habitat scored as 1. For example, grassland songbirds may prefer native prairie habitat above all other habitat types (habitat suitability = 1), but will also make use of a managed hayfield (habitat suitability = 0.5). We define habitat suitability or quality across LULC types for general biodiversity and the songbird functional groups in the on-line appendix.

Habitat quality in a grid cell can be modified by LULC in surrounding grid cells. We consider sources of degradation as those human modified LULC types (e.g., urban, agriculture, and roads) that cause edge effects (McKinney 2002; Forman 2003). Edge effects refer to changes in the biological and physical conditions that occur at a patch boundary and within adjacent patches (e.g., facilitating entry of predators, competitors, invasive species, toxic chemicals and other pollutants). The sensitivity of each habitat type to degradation is based on general principles of landscape ecology and conservation biology (e.g., Forman 1995; Lindenmayer et al. 2008) and is specific to each measure of biodiversity. Sensitivity scores are determined from the literature and expert knowledge and specifics are described in the on-line appendix.

We generate a habitat quality score for each scenario by summing across all of the scenario's grid cell-level degradation-adjusted habitat quality scores. Because of the influence of adjacent patches on quality scores, the spatial pattern of land use as well as the overall amount of habitat will matter in determining the landscape habitat quality score. Habitat quality scores should be interpreted as relative scores with higher scores indicating landscapes more favorable for the given conservation objective. The landscape habitat quality score cannot be interpreted as a prediction of species persistence on the landscape or other direct measure of species conservation in the same way that the output of the carbon model is an estimate of the actual carbon stored on the landscape. The InVEST habitat model does not convert habitat quality measures into monetary values.

2.2.3 Water Quality

Land use affects water quality by contributing nutrients to surface and ground waters. The retention of polluting nutrients and filtration of water is an important service provided by functioning ecosystems. The InVEST water quality model evaluates the nutrient retention service provided by a landscape over the course of a year and highlights the impacts of LULC change on water quality. In this study we focused on phosphorus pollution, which is a leading cause of surface water impairment in the upper Midwest (Carpenter et al. 1998).

Due to data limitations we ran the water quality model just for the Minnesota River Basin rather than for the whole state. The Minnesota River Basin drains much of the southwestern and south central part of the state (Fig. 1). The basin contains much of the prime agricultural area in the state (32% of the total agricultural land in the state in 1992 and 2001) and is responsible for roughly 1/5th of the phosphorus exports in Minnesota (MPCA 2004). The majority of pollutants that lead to violations of water quality standards (Total Maximum Daily Loads) for the Minnesota River and Lake Pepin on the Mississippi River originate in southern and central Minnesota (Senjem 2009).

The InVEST water model uses a two-step process to evaluate the water quality impacts of each scenario. First, the model uses climate data, geomorphological information, and LULC characteristics to calculate the average annual water yield in each grid cell. Water yield is defined as precipitation minus evapotranspiration. The water model does not consider connection with deep aquifers, but assumes that all precipitation not lost to evapotranspiration is surface water runoff. The model may thus overestimate the surface water yield. Although

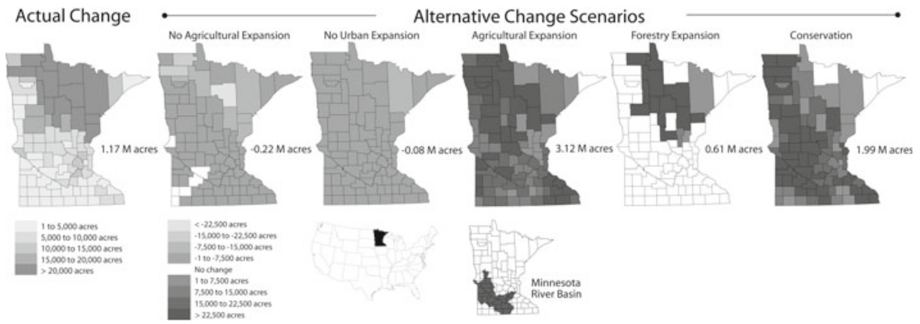

Fig. 1 Number of acres in each county that change LULC between 1992 and 2001. The "Actual Change" map measures observed change in LULC by county. The alternative scenario maps measure change by county relative to the baseline. *Darker shades* indicate that the county experienced greater LULC change under an alternative scenario than under the baseline. The numbers by the side of each map indicate state totals in millions of acres

ground water recharge is typically about 20% of precipitation, the majority of that volume is retained in shallow aquifers and quickly re-connects with surface water bodies (Delin and Falteisek 2007). The Minnesota River Basin has a lower rate of ground water recharge than the eastern and northeastern portions of the state and agricultural drainage likely further reduces flow to deep aquifers. The routing of surface water flow across cells is defined using a digital elevation map.

In the second step, water yield is combined with information about phosphorus loadings and the filtering (nutrient retention) capacities of each LULC type to calculate the annual phosphorus exports from each cell. Phosphorus exports from cells are routed via surface water flow to downstream cells, where some of the phosphorus may be filtered or additional phosphorus added, until it flows into a water body. The spatial pattern of land use can affect phosphorus loadings. In particular, stream buffers of perennial vegetation can effectively filter phosphorus before it reaches a stream. We do not measure changes of phosphorus loadings once it is in a water body but assume that all loadings are delivered to the mouth of the watershed.

We convert the annual loadings of phosphorous at the mouth of the Minnesota River Basin into monetary values using results from Mathews et al. (2002). Mathews et al. (2002) used a contingent valuation survey to estimate how households in the basin would value a 40% reduction in phosphorus loadings into the Minnesota River. They estimated an aggregate annual household willingness-to-pay of $141 million for a 40% reduction in 1997 dollars ($122.7 million in $1992). The water quality benefits (or costs) for each scenario are found by prorating the value of a 40% improvement in water quality to the water quality improvement in the scenario. So for example, a 10% reduction in phosphorus exports would generate an annual value of $30.7 million ($122.7 × 0.25). This method is equivalent to assuming that water quality benefits (costs) are linear in water quality improvement (decline).

2.2.4 Value of Agricultural Production

InVEST uses information on the observed relationships between soil quality and yields, yield trends, prices for agricultural produce, and agricultural production costs to estimate annual net returns to agricultural production on the landscape. Multiplying a grid cell's expected yield by crop prices generates an estimate of annual agriculture revenue in the grid cell. Subtracting costs from revenues generates an estimate of annual economic returns from agricultural activity.

We generate county-level yield functions for corn, corn silage, soybeans, alfalfa hay, pasture, oats, barley, and spring wheat with a dataset that relates yield to land quality for each Minnesota county (USDA-NRSC 2009). In this case, land quality is indesced by land classification category (LCC), where lower LCCs are associated with better land quality. We used observed state-wide yield trends for these crops from 1992 to 2001 to account for technological progress leading to growth in yield through time (USDA-NASS 2009). Crop price and production cost data by agricultural region and year in Minnesota are taken from the Farm Financial Database (http://www.finbin.umn.edu/) and regional censuses of Minnesota agriculture (Farm Business Management 1999–2001).

For the purposes of this paper we assume that agricultural products from Minnesota were traded on a national or international market that was large enough to maintain observed 2001 prices despite any changes from observed 2001 production volumes. Similarly, we assume that agricultural infrastructure in Minnesota was able to respond to alternative production patterns in 2001 such that per unit production costs remain constant.

2.2.5 Value of Timber Production

We use data from Lubowski (2002) and Lubowski et al. (2006, 2008) to estimate annual net returns to forestry for the years 1992 and 2002 (the data are not available for 2001). Timber harvesting is assumed to occur on all forest land not set aside for conservation (see the on-line appendix for details). We multiply working forest acreage in a county in a given year by the county's per acre net return to forestry in that year to calculate the total value of forestry in the county. Estimated returns to forestry in a county are based on the assumption that all non-conservation land forests are managed on an optimal, even-age rotation basis to produce sawtimber (similar to Lubowski 2002).

Under the alternative land-use scenarios in 2001 there is the possibility that the change in working forest coverage could change the average productivity of such forests vis-à-vis the baseline. To account for this effect on net returns to the forestry sector we calculated the average forest productivity index (FPI) of forested grid cells in each county with FPI data under the actual 2001 baseline conditions and each alternative 2001 scenario. We assumed that average forestry net revenues in a county increased by 1% for every 10% increase in the county's mean FPI on working forest land (personal communication, Grant Domke).

Just as with the agricultural sector, we assume that timber is sold into national or international markets so that timber prices do not change with changes in harvest volumes and that inputs to timber harvest are able to respond to alternative production patterns such that per unit production costs remain constant.

2.2.6 Value of Urban Development

We used data from Lubowski (2002) and Lubowski et al. (2006, 2008) to estimate annualized county-level net returns to landowners from urban development for the years 1992 and 2002 under the baseline and each scenario. As with forestry net returns, we use 2002 data as a proxy for 2001 returns. We multiply urban acreage in a county in 1992 by the county's 1992 return per acre of urban area to calculate the total value of urban area in the county in 1992. We follow a similar procedure for calculating county-level values in 2001 under all scenarios.

Table 3 Results for annual changes in ecosystem services, biodiversity, and returns to land from actual land use change and alternative scenarios 1992–2001

Outcome	Change from 1992 to 2001 by scenario					
	1. Actual land use	2. No agricultural expansion	3. No urban expansion	4. Agricultural expansion	5. Forestry expansion	6. Conservation
Carbon sequestration (M Mg)	−1.65	−0.32	−1.51	−16.80	−1.63	−0.49
Estimated value of carbon sequestration (M 1992$ using $42.32 as social cost of carbon)	−$69.65	−$13.46	−$63.81	−$710.97	−$69.04	−$20.58
Estimated value of carbon sequestration (M 1992$ using $83.72 as social cost of carbon)	−$137.80	−$26.62	−$126.23	−$1,406.49	−$136.57	−$40.72
Change in phosphorous export (Mg per year)	−2.90	−6.40	−4.20	181.10	−2.90	−467.40
Estimated value of change in phosphorous export (M 1992 $)	$0.64	$1.42	$0.93	−$40.11	$0.64	$103.53
Change in overall biodiversity measure	0.02%	0.5%	0.2%	−8.4%	1.2%	5.5%
Change in forest breeding birds measure	0.6%	1.3%	0.7%	−12.9%	3.9%	6.2%
Change in grassland breeding birds measure	−1.2%	−1.3%	−1.0%	2.6%	−3.4%	0.7%
Change in agricultural net returns (M 1992$)	−$330.90	−$336.52	−$328.55	−$127.92	−$347.16	−$437.15
Change in agricultural net returns with 1992 prices and costs in 2001 (M 1992 $)	$910.23	$901.24	$914.25	$1,214.68	$887.87	$711.13
Change in forestry net returns (M 1992 $)	$14.74	$14.97	$14.81	$10.37	$15.86	$14.50

Table 3 continued

Outcome	Change from 1992 to 2001 by scenario					
	1. Actual land use	2. No agricultural expansion	3. No urban expansion	4. Agricultural expansion	5. Forestry expansion	6. Conservation
Change in forestry net returns—1992 prices in 2001 (M 1992 $)	$0.01	$0.09	$0.03	−$1.46	$0.54	−$0.06
Change in urban net returns (M 1992 $)	$3,636.65	$3,664.98	$3,340.38	$3,535.61	$3,623.43	$3,643.32
Change in urban net returns—1992 prices in 2001 (M 1992 $)	$136.83	$154.52	−$45.61	$74.70	$130.38	$141.10

'M' stands for millions. 'Mg' stands for mega-grams (metric tons). All dollar values are measured in 1992 constant dollars (1992 $)

We do not adjust returns per acre of urban area in the alternative scenarios even though urban area supply, a determinant of market prices, has changed. We lack the data to determine how urban area values changes as its supply changes.

3 Results

3.1 Actual Land-Use and Land-Cover Change

The actual changes in LULC between 1992 and 2001 across the state led to improvement on many, but not all, objectives (column 1 of Table 3, Figs. 2, 3). Overall, land use shifted modestly toward forest and urban land use and out of agriculture and grasslands. The state's overall and forest bird species habitat measures, water quality in the Minnesota River Basin, and the state's value of timber production and urban land all increased between 1992 and 2001 under the baseline. The grassland bird habitat measure, the state's stored carbon, and the state's value of agricultural production declined between 1992 and 2001. The spatial pattern of changes in the value of the ecosystem services and marketed returns and the overall species habitat score across the state under the baseline are shown in Fig. 4.

The 16% decline in the value of agricultural production under the baseline occurs because of the significant decline in real prices for agricultural crops from 1992 to 2001. The decline in prices dominates improvements in crop yields. If agricultural prices and production costs in 2001 were equivalent to their 1992 values, then the value of agricultural production would have increased by 43%. Because the net change in agricultural area from 1992 to 2001 across Minnesota was quite small, this 43% increase is almost entirely due to yield improvements over the decade.

The improvement in the overall species habitat score between 1992 and 2001 is quite small (Table 3). However, there are interesting shifts among functional groups. The forest bird diversity measure increased under the baseline while the grassland bird diversity measure decreased. The modest shift toward forest and urban land use and out of agriculture and grasslands across the state explains these changes. Further, there was also a small decrease in carbon sequestration under the baseline. The small shift of land out of agriculture in the Minnesota River Basin also led to a slight decline in phosphorus exports (small improvement in water quality). For the state as a whole, it is possible that increases in urban land use,

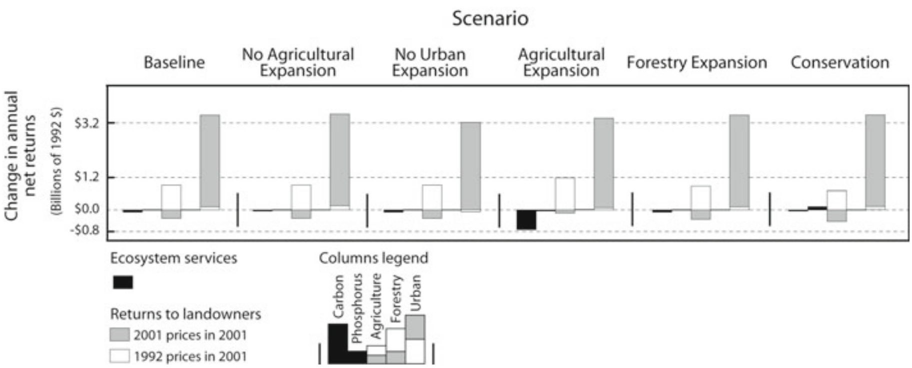

Fig. 2 Change in each modeled ecosystem service benefit and returns to landowner category under the baseline and each alternative scenario

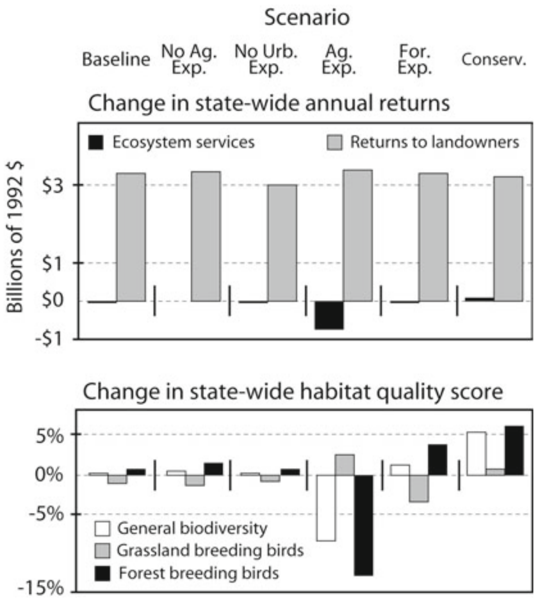

Fig. 3 Change in aggregated ecosystem service benefits, returns to landowners and habitat scores under the baseline and each alternative scenario

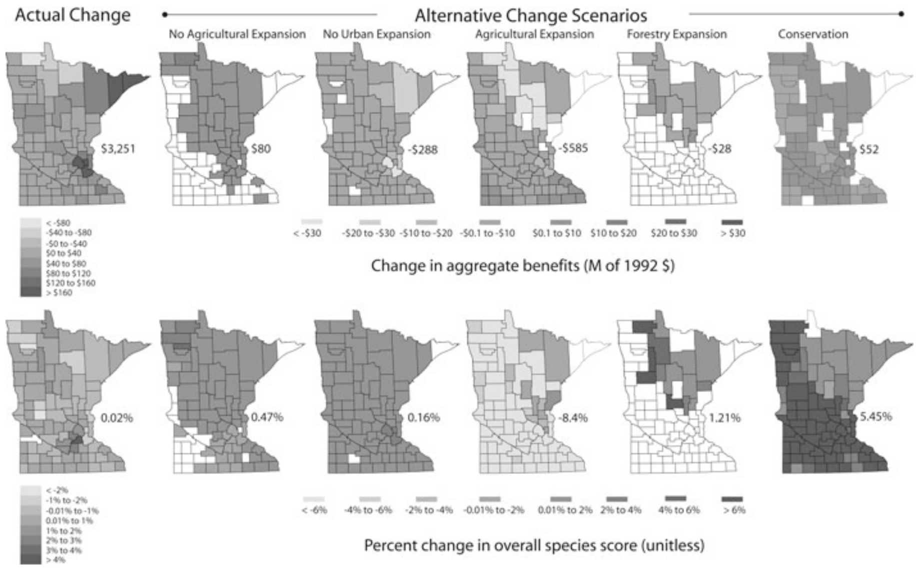

Fig. 4 Change in aggregated ecosystem service benefits and returns to landowners (top row of maps) and overall species habitat scores (bottom row of maps) under the baseline and each alternative scenario. The "Actual Change" maps measure change by county under the baseline. The numbers by each baseline map give state-level change. The alternative scenario maps measure change by county relative to the baseline. *Darker shades* indicate that the county experienced greater positive change under an alternative scenario than under the baseline. The numbers by the side of each alternative scenario map indicate state-level change relative to the baseline. We use a social cost of carbon of $42.32/ton for these maps

which happened mostly outside of the Minnesota River Basin, increased total phosphorus loading and could have led to an overall deterioration in water quality.

The value of forest production and the value of urban land use both show large increases between 1992 and 2001 under the baseline, 161% for forestry and 70% for urban. Virtually all of the increase in value, however, is due to price effects with only a small positive effect due to land-use change. The value of forestry production increases by 0.10% from 1992 to 2001 when prices are kept constant at 1992 levels. The value of urban land increases by 2.64% from 1992 to 2001 when prices are kept constant at 1992 levels.

3.2 Alternative Land-Use Change Scenarios

Columns 2 through 6 of Table 3 show the effects of the alternative LULC change scenarios on Minnesota's production of ecosystem services, species habitat, and returns to landowners. With no new land allowed to transition into agriculture from 1992 to 2001 (Table 3, column 2), Minnesota's agricultural land base shrinks and there is slightly more land in other land uses. This scenario, as compared to actual land use change, significantly increases carbon sequestration in biomass and soil carbon (0.32 million metric tons lost versus 1.65 million metric tons lost). A restriction on agricultural expansion also results in slight improvement in water quality in the Minnesota River Basin (decreased phosphorus exports) and increases in the overall species and forest bird habitat measures. There is, however, a decline in the measure for grassland birds. Grassland birds prefer agricultural land, especially pasture and hayfields, to forest or urban land uses. There are also slightly higher values of forestry production and urban land use as compared to the baseline due to greater amount of land remaining in these categories. Not surprisingly, the value of agricultural production declines in this scenario versus the actual land use case.

Allowing no urban expansion from 1992 to 2001 in Minnesota (Table 3, column 3) generates slightly higher carbon sequestration and biodiversity measures than the baseline. The effect of preventing urban expansion is less pronounced than preventing agricultural expansion largely because the amount of land-use change blocked relative to the baseline is far less. There is, however, a large decline in phosphorus export from the Minnesota River Basin. Urban areas generate high exports per unit area. Urban phosphorus exports, unlike agricultural phosphorus exports, tend to be captured in sewage treatment plants. Although phosphorus exports from urban areas impact surface water quality downstream, much of the cost associated with an increase in urban phosphorus is in the form of higher wastewater treatment costs. There are very small increases in the value of forestry and agricultural production values under the no urban expansion scenario relative to the baseline due to greater amounts of these land-uses remaining on the landscape. There is a larger decline in the overall returns to landowners as urban land use tends to be of much greater per unit value than any other land use.

Expansion of agriculture into all areas with high quality soil results in the most dramatic statewide changes (Table 3, column 4). The modeled agricultural expansion causes a large decrease in carbon sequestration. Phosphorus exports in the Minnesota River Basin increase by 13.1% compared to the 1992 level (in contrast, phosphorus exports declined by 0.2% from 1992 to 2001 in the baseline). Species habitat measures also show significant declines from 1992 to 2001 (−8.4% for the overall species habitat measure and −12.9% for the forest birds measure) with the exception of the grassland bird habitat measure, which shows a slight improvement (2.3%). The grassland habitat measure improves because the increase in agricultural land means greater pasture and hayfield area, marginal habitat for grassland birds. (if, for some reason, agricultural expansion only involves row crop agriculture then grassland birds would not show an improvement). The value of forestry and urban land declines relative to the baseline. The expansion of agriculture area under this scenario means that state-wide returns to agriculture are 11.38% greater than they are under the baseline in 2001.

The results of the forestry expansion scenario (Table 3, column 5) are generally opposite to the agricultural expansion case. Carbon sequestration and phosphorus exports are virtually unchanged from the base case instead of showing large deteriorations as in the agricultural expansion case. Extending the model to a longer time horizon would show more of an improvement in carbon sequestration as forests mature through time. Carbon sequestration gains in this scenario are also limited by the fact that some forest expansion occurs on peatlands in the northern part of the state. Converting peatlands into timber lands results in large releases of soil carbon. Forest bird habitat increases with forest expansion while grassland bird habitat declines. The value of forest land increases relative to the base case while the value of agricultural and urban lands show slight declines relative to the base case.

The conservation scenario scores well on the species habitat, carbon sequestration and water quality metrics (Table 3, column 6). Water quality in the Minnesota River Basin improves dramatically under the conservation scenario with a decline in annual phosphorus exports of 467.4 million tons by 2001, a 33.8% decrease relative to 1992 levels. Most of the water quality improvement comes from putting in 100 m buffer strips along all streams in the Basin. The conservation scenario is the only scenario that resulted in improvements in habitat scores for both forest and grassland birds. Grassland birds benefited from buffer strips that increased grassland habitat. Even though a significant amount of agricultural land was converted to grasslands, most of this land was located along narrow riparian corridors. Given the negative impact of surrounding agricultural lands on the habitat quality of grasslands, the increase in the grassland bird measure was not as high as it would have been had we added the same habitat area in large blocks. This result highlights the importance of considering the spatial arrangement of conservation efforts on the landscape (i.e., habitat connectivity, core area) for biodiversity conservation. The conservation scenario also generates the largest state-wide increase in the overall species and forest bird habitat measures. Not surprisingly, the conservation scenario did relatively poorly on returns to landowners, scoring the lowest of any of the alternatives on agricultural production value, and second lowest (to agricultural expansion) on value of forest production.

A number of the objectives in Table 3 are reported in monetary terms or can be readily converted to dollar terms. Agricultural, forestry and urban land values are reported in monetary terms to begin with. We also convert carbon sequestration and phosphorus exports, initially reported in biophysical terms, to monetary terms using methods discussed in Sect. 2. The only objectives we do not attempt to report in monetary terms are the species habitat measures.

We compare the change in total value for those objectives reported into monetary terms under the various scenarios in Table 4. In Table 4, we use the median social cost of carbon ($42.32) as reported by Tol (2009). When we include all values (carbon sequestration, change in water quality, agricultural, timber and urban land values) and use actual prices in 1992 and 2001, all scenarios result in an increase in total value or net social benefits from 1992 to 2001 (row 1, Table 4). Four of the six alternatives (actual land use, no agricultural expansion, forestry expansion and conservation) generate similar values. The highest total value is generated by the no agricultural expansion scenario (column 2). This scenario scores well because it has the second highest returns to landowners (row 2) but also avoids large declines in stored carbon. Both the actual land use (column 1) and forestry expansion scenarios (column 5) have similar returns to landowners as does the no agricultural expansion scenario, but these two scenarios do not score as well on the carbon sequestration or water quality metrics. The conservation scenario (column 6) generates lower returns to landowners but generates the largest sum of water quality improvements and carbon sequestration values. The full agricultural expansion scenario (column 4) generates the lowest total value despite it having the highest returns to landowners because of the large decline in the value of carbon sequestration as well as a negative impact on water quality. No urban expansion (column 3) scores second to the lowest in total value. It does relatively poorly because it restricts expansion of high-value urban land, thereby generating the lowest returns to landowners.

We also report summary results in both total value and returns to landowners when we take out the effect of price changes between 1992 and 2001 (rows 3 and 4 in Table 4). In this case, all changes in value are due to land use change (and yield increases in the case of agriculture). The main difference in the results in rows 3 and 4 vs. those in rows 1 and 2 are the magnitude of changes between 1992 and 2001. There were large increases in prices for urban land and forestry and large declines in agricultural prices over this time period. The relative ranking of the scenarios remains largely the same regardless of which prices are used. There is, however, one exception. The conservation scenario (column 6) takes more agricultural land out of production than all other scenarios and the opportunity cost of doing so is higher when using 1992 prices rather than 2001 prices. The conservation scenario ranks fourth out of the six scenarios using 1992 prices while it ranks second using 2001 prices.

4 Discussion

In this paper we applied the InVEST model to compare the effects of alternative land use change scenarios on the joint provision of ecosystem services, species habitat, and returns to landowners. Our results illustrate the importance of taking ecosystem services into account in land use decisions. In our analysis, the scenario that generated the highest private returns to landowners (agricultural expansion) also generated the lowest net social benefit of any scenario analyzed. The agricultural expansion scenario resulted in the largest increase in land devoted to the production of marketed goods, and this led to large declines in carbon storage and water quality and their associated values. The agricultural expansion scenario also generated declines in habitat quality for biodiversity, with the exception of grassland birds where an expansion of hay and pasture land slightly improved the picture.

In general, because landowners are financially rewarded for commodity production, but not for the provision of non-market ecosystem services, private land use decisions will tend to over-emphasize the former and under-provide the latter. In other words, land-use patterns like that in the agricultural expansion scenario are more likely to emerge than more conservation-oriented landscapes despite their potential for generating higher net social values. Such a pattern is consistent with overall recent global trends as summarized in the Millennium Ecosystem Assessment, which shows an increase in many provisioning services related to marketed commodities simultaneous with declines in many cultural, regulating and supporting services (MA 2005). The results in this paper, along with similar results in Nelson et al. (2009), provide quantitative evidence that the change in the value of non-marketed ecosystem services are of sufficient magnitude to change the ranking of land use scenarios when evaluated using net social benefits as compared to private returns no landowners.

Understanding the effect of LULC choices on the provision and value of ecosystem services and returns to landowners, the focus of this paper, is an essential precondition for finding efficient land use patterns that maximize social net benefits. However, simply understanding the provision and value of ecosystem services, while necessary, is not sufficient for efficient outcomes to occur. The divergence of private and net social benefits demonstrates the need for instituting policies that encourage choices that enhance the provision of non-market eco-

Table 4 Summary of changes in annual value from actual land use change and alternative land use change scenarios 1992–2001

Outcome	Change from 1992 to 2001 by scenario					
	Actual land use	No agricultural expansion	No urban expansion	Agricultural expansion	Forestry expansion	Conservation
1. Change in total value: carbon sequestration, water quality, agricultural production, forest production, and urban using actual prices (M 1992 $)	$3,251	$3,331	$2,964	$2,667	$3,224	$3,304
2. Change in returns to landowners: agricultural production, forest production, urban using actual prices (M 1992 $)	$3,320	$3,343	$3,027	$3,418	$3,292	$3,221
3. Change in total value : carbon sequestration, water quality, Agricultural production, forest production, and urban using 1992 prices in 2001 (M 1992 $)	$978	$1,044	$806	$537	$950	$935
4. Change in returns to landownrs: agricultural production, forest production, urban using 1992 prices in 2001 (M 1992 $)	$1,047	$1,056	$869	$1,288	$1,019	$852

Results use $42.32 per Mg for the social cost of carbon. All dollar values are measured in 1992 constant dollars (1992 $)

system services, or discourage choices that reduce the provision of non-market ecosystem services. There is an extensive literature in environmental economics on policy mechanisms to correct problems caused by the divergence between private and net social benefits, including tax and subsidy programs, cap-and-trade policies, and payments for ecosystems services. Some existing policies partially align private and social returns by paying landowners for conservation that increases ecosystem services (e.g., the Conservation Reserve Program of the U.S. Department of Agriculture). Further work evaluating both the theory and practice of payments for ecosystem services (e.g., Pagiola and Platais 2007; Jack et al. 2008), responses to policy interventions and the resulting spatial pattern of land use (e.g., Andam et al. 2008; Sánchez-Azofeifa et al. 2007), and analysis of the impact of alternative policy design on the provision of ecosystem services at landscape scales (e.g., Lewis et al. 2010; Nelson et al. 2008) will help improve policy aimed at increasing the net social value of services from ecosystems.

Designing policy to improve the efficiency of outcomes is not a simple task. Two particular aspects of ecosystem services raise special concerns. First, the decisions of many separate landowners generate the spatial patterns of land use and land cover that determines the provision of many ecosystem benefits. Finding policy mechanisms that provide incentives to independent landowners so that they choose an efficient spatial pattern is an on-going area of research (e.g., Parkhurst et al. 2002; Parkhurst and Shogren 2007). Second, landscapes jointly provide bundles of ecosystem services and other benefits. Land-use choices that maximize the provision of one set of outputs will not, in general, maximize the provision of other outputs. In the results shown above, the scenario that maximizes the value of marketed commodities (the agricultural expansion scenario) resulted in the minimum value of non-marketed ecosystem services of any scenarios considered. Even among various non-marketed ecosystem services and various forms of habitat provision there are tradeoffs. What is good for grassland birds is not necessarily good for forest birds, and vice-versa. Putting in buffers along streams is good for water quality but not particularly good for carbon sequestration or biodiversity conservation. Jackson et al. (2005) highlight potential tradeoffs between carbon sequestration and surface water flows and Nelson et al. (2008) show tradeoffs between carbon sequestration and biodiversity conservation.

The tradeoffs among various objectives highlight the importance of relative prices, in addition to ecological production functions, in ranking alternatives. That changes in relative prices can change the rankings of alternatives is obvious to most economists but makes many natural scientists, who want objective answers to ecosystem management, uneasy. However, it is the combination of societal values along with biophysical analysis of the consequences of alternative choices that determines the socially-preferred outcome. The importance of relative prices in our results is shown by the different ranking of scenarios obtained using 1992 versus 2001 prices. The conservation scenario ranked second out of six alternatives using 2001 prices but ranked fourth out of six using 1992 prices.

In this paper we used the market value of returns to landowners to represent the net private benefits of land-use. However, the actual value that landowners derive from their land-use choices are based on their utility functions, which may include some appreciation for nature and other non-marketed goods. Therefore, the net private returns to land use value for each scenario may overstate the actual net private benefits of land-use. However, given our inability to observe landowners' utility functions and the difficulty of converting utility into money metrics, we use the market value of returns to landowners as a proxy for the net private benefits of land-use.

The range of uncertainty around many non-market values can make rankings among alternatives ambiguous. For example, if water quality was deemed to be roughly twice as valuable as we assumed in this study, the conservation scenario would have been ranked first under either 1992 or 2001 prices. There is also a considerable range of estimates in the social cost of carbon (Tol 2009). Relative rankings can also be influenced by the geographic or temporal scope of the analysis. Had we been able to evaluate water quality benefits for the entire state, rather than just for the Minnesota River Basin (constituting about 20% of phosphorus exports for the state), or had we included water quality improvements in downstream states along the Mississippi River, we would have generated higher water quality benefits. Just how much higher, and whether these would be enough to change the rankings of alternative, is not clear.

We often lack good signals of the relative value of non-market ecosystem services and habitat conservation. In light of this, it is not always clear the best way to proceed. Should controversial or imprecise estimates of marginal value be used? Or should ecosystem services and habitat conservation measures that lack robust estimates of marginal value be excluded from the accounting of net benefits? Either choice presents difficulty. Not providing monetary estimates for some objectives risks overlooking the importance of these objectives. But inclusion of imprecise numbers risks providing flawed analysis and may call into question the entire exercise. In this study, we choose to include estimates of the value of carbon sequestration from integrated assessment models of climate change and the value of water quality improvements from a contingent valuation survey, but not the value of habitat provision. Other researchers can draw the line differently. No matter where the line is drawn, researchers should be clear about what they have done and what the implications of their decisions are for the results.

In this paper we assumed that observed commodity prices would not be affected by deviations from observed land-use patterns. In reality, prices for goods and services are functions of their supply and the provision of various related goods and services. An important avenue for future research involves the integration of detailed biophysical models capable of predicting the provision of ecosystem services, with general equilibrium models capable of predicting relative prices based on supply and demand. Work along these lines has advanced recently with interest in the effects of biofuels on land use, carbon sequestration, energy and food prices (e.g., Keeney and Hertel 2009; Searchinger et al. 2008).

In this paper we ranked a small set of scenarios on the returns to landowners and the social value of ecosystem services. An alternative to scenario ranking, and one that most economists will find more appealing, is to use optimization tools to find land-use and land-management patterns that maximize a given objective function. We have taken an optimization approach in earlier work where we use tools from operations research to find the frontier of maximum feasible landscape biodiversity scores for various levels of landscape of commodity production values and vice-versa (Polasky et al. 2005, 2008), and the efficiency frontier between carbon sequestration and biodiversity conservation for fixed conservation budgets (Nelson et al. 2008). Finding land-use and land-management patterns that maximize net social benefits can be difficult for several reasons. First, the objective function may be non-linear and could be could be explained by the whole land-use pattern. Second, discrete choices are typically made over a large number of land parcels (land is either in one land use or another). At present, Invest does not have the capability of finding optimal land-use and land-management patterns. We hope to undertake an optimization analysis in the near future.

Evaluating the joint provision and value of multiple ecosystem services from landscapes is still in its relative infancy. We have only imperfect understanding of socio-economic-ecological systems and we have limited ability to predict consequences of human actions on ecological processes, the provision of ecosystem services, and ultimately on the effect on human well-being. Part of the difficulty in understanding the links in the chain from human actions through ecological processes and back to human well-being lies in the natural sciences. Improving understanding of ecosystem services requires gaining better understanding of ecological production functions. For example, our understanding of nutrient cycling and the hydrological system is imperfect, and predictions of water quality models often come with large error bounds. Furthermore, ecological systems may have threshold effects that cause radical changes in provision of ecosystem services and such shifts may be difficult to predict prior to their occurrence (e.g., Biggs et al. 2009; Scheffer et al. 2001). Another part of the difficulty in predicting the value of ecosystem services comes from imperfect economic models and data, which affects both the ability to predict human behavior that affects the environment as well as the ability to value non-market goods and services.

Both ecological and economic uncertainty can make evaluation of the net present value of the long-term flow of ecosystem services problematic. Effects that occur through time raise related issues of what is the proper discount rate to use in such analysis, what might be the long-term consequences of current action on ecosystem processes and the flow of ecosystem services, and what values will various ecosystem services have for future generations. Taking proper account of land-use changes that affect ecosystem service flows differentially through time also requires careful thought. For example, switching from annual crops to perennial crops or forests may yield water quality improvements and habitat benefits and result in a build-up of carbon stocks through time. Eventually, however, carbon sequestration will cease as a new equilibrium level of carbon storage is reached but water quality improvements and habitat benefits will likely continue to flow as long as the perennial crops or forests persist. The ideal way to evaluate these different benefit streams is by taking the present value of the flow of services in an infinite horizon model, but this places a premium on understanding the future and properly weighing present versus future benefits.

Further improvements in modeling and data are needed to increase the reliability of estimates of the value of ecosystem services. Even without improvements in models or data, sensitivity analysis could be used to analyze whether the ranking of alternatives is robust and to determine the variables that have an inordinate affect on results. We have not used sensitivity analysis in this paper to systematically explore the effect of uncertainty in ecosystem service provision and valuation, nor to evaluate the robustness of ranking of alternatives to such uncertainty. Such analysis is clearly worth pursuing in the future.

Additional work will also be needed to incorporate other ecosystem services (e.g., flood mitigation, pollination, and disease control) and to more fully integrate biodiversity conservation objectives. Despite all of this, we are already in position to improve upon current decision-making that largely ignores ecosystem services. As illustrated in this case study, the value of ecosystem services, particularly related to carbon sequestration and water quality, are likely to be large; large enough to change the social ranking of the desirability of land-use and land-management decisions.

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References

Andam KS, Ferraro PJ, Pfaff A, Sanchez-Azofeifa GA, Robalino J (2008) Measuring the effectiveness of protected area networks in reducing deforestation. Proc Natl Acad Sci 105(42):16089–16094
Angelsen A (ed) (2008) Moving ahead with REDD: issues, options and implications. CIFOR, Bogor
Balmford A, Bruner A, Cooper P, Costanza R, Farber S, Green R, Jenkins M, Jefferiss P, Jessamay V, Madden J, Munro K, Myers N, Naeem S, Paavola J, Rayment M, Rosendo S, Roughgarden J, Trumper K, Turner RK (2002) Economic reasons for conserving wild nature. Science 297:950–953
Biggs R, Carpenter SR, Brock WA (2009) Turning back from the brink: detecting an impending regime shift in time to avert it. Proc Natl Acad Sci 106:826–831
Boody G, Vondracek B, Andow DA, Krinke M, Westra J, Zimmerman J, Welle P (2005) Multifunctional agriculture in the United States. BioScience 55:27–38
Canadell JG, Raupach MR (2008) Managing forests for climate change mitigation. Science 320:1456–1457
Carpenter SR, Caraco NF, Correll DL, Howarth RW, Sharpley AN, Smith VH (1998) Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecol Appl 8(3):559–568
Champ P, Boyle KJ, Brown TC (2003) A primer on nonmarket valuation. Kluwer, Boston
Chan KMA, Shaw MR, Cameron DR, Underwood EC, Daily GC (2006) Conservation planning for ecosystem services. PLoS Biol 4:e379
Daily G (1997) Nature's services: societal dependence on natural ecosystems. Island Press, Washington
Daily G, Polasky S, Goldstein J, Kareiva PM, Mooney HA, Pejchar L, Ricketts TH, Salzman J, Shallenberger R (2009) Ecosystem services in decision-making: time to deliver. Front Ecol Environ 7(1):21–28
Delin GN, Falteisek JD (2007) Ground-water recharge in Minnesota: U.S. Geological Survey Fact Sheet 2007–3002, 6 p
Egoh B, Reyers B, Rouget M, Richardson DM, Le Maitre DC, Jaarsveld ASvan (2008) Mapping ecosystem services for planning and management. Agric Ecosyst Environ 127(1–2):135–140
Ehrlich PR, Dobkin DS, Wheye D (1988) Birder's handbook: a field guide to the natural history of North American birds. Simon and Schuster, New York
Farm Business Management, Annual Report for the Southwest, Southcentral, Southeast, West, Northwest, and Northeast Regions. 1992–2001. Minnesota State Colleges and Universities, Minnesota Riverland Community College, University of Minnesota, Division of Agricultural Education
Forman R (1995) Land mosaics: the ecology of landscapes and regions. Cambridge University Press, New York
Forman R (2003) Road ecology: science and solutions. Island Press, Washington
Freeman AM III (2003) The measurement of environmental and resource values: theory and methods, 2nd edn. Resources for the Future, Washington
Fry JA, Coan MJ, Homer CG, Meyer DK, Wickham JD (2009) Completion of the National Land Cover Database (NLCD) 1992–2001 Land Cover Change Retrofit product. U.S. Geological Survey Open-File Report 2008–1379
Hill J, Polasky S, Nelson E, Tilman D, Huo H, Ludwig L, Neumann J, Zheng H, Bonta D (2009) Climate change and health costs of air emissions from biofuels and gasoline. Proc Natl Acad Sci 106(6):2077–2082
Jack BK, Kousky C, Sims KRE (2008) Designing payments for ecosystem services: lessons from previous experience with incentive-based mechanisms. Proc Natl Acad Sci USA 105:9465–9470
Jackson RB, Jobbagy EG, Avissar R, Roy SB, Barrett DJ, Cook CW, Farley KA, le Maitre DC, McCarl BA, Murray BC (2005) Trading water for carbon with biological sequestration. Science 310:1944–1947
Keeney R, Hertel TW (2009) The indirect land use impacts of United States biofuel policies: the importance of acreage, yield, and bilateral trade responses. Am J Agric Econ 91(4):895–909
Kindermann GE, Obersteiner M, Sohngen B, Sathaye J, Andrasko K, Rametsteiner E, Schlamadinger B, Wunder S, Beach R (2008) Global cost estimates of reducing carbon emissions through avoided deforestation. Proc Natl Acad Sci 105(30):10302–10307
Lewis DJ, Plantinga AJ, Nelson E, Polasky S (2010) The efficiency of voluntary incentive policies for preventing biodiversity loss. Resource and Energy Economics (in press)
Lindenmayer D, Hobbs R, Montague-Drake R, Alexandra J, Bennett A, Burgman M, Cae P, Calhoun A, Cramer V, Cullen P (2008) A checklist for ecological management of landscapes for conservation. Ecol Lett 11:78–91
Lubowski RN (2002) Determinants of land-use transitions in the United States: econometric analysis of changes among the major land-use categories. Ph.D. Dissertation, Harvard University, Cambridge, MA
Lubowski RN, Plantinga AJ, Stavins RN (2006) Land-use change and carbon sinks: econometric estimation of the carbon sequestration supply function. J Environ Econ Manage 51:135–152
Lubowski RN, Plantinga AJ, Stavins RN (2008) What drives land-use change in the United States? A national analysis of landowner decisions. Land Econ 84(4):529–550

Mathews LG, Homans FR, Easter KW (2002) Estimating the benefits of phosphorus pollution reductions: an application in the Minnesota River. J Am Water Res Assoc 38(5):1217–1223

McKinney ML (2002) Urbanization, biodiversity, and conservation. BioScience 52:883–890

Millennium Ecosystem Assessment (MA) (2005) Ecosystems and human well-being synthesis. Island Press, Washington DC

Minnesota Department of Commerce (2007) Minnesota's Next Generation Energy Initiative. Accessed at http://www.state.mn.us/mn/externalDocs/Commerce/SummaryNext_Generation_Energy_Initiative_100807041804_MEP2.pdf on May 17, 2010

Minnesota Pollution Control Agency (MPCA) (2004) Detailed assessment of phosphorus to Minnesota watersheds. Prepared by Barr Engineering Company for the Minnesota Pollution Control Agency

Naidoo R, Balmford A, Costanza R, Fisher B, Green RE, Lehner B, Malcolm TR, Ricketts TH (2008) Global mapping of ecosystem services and conservation priorities. Proc Natl Acad Sci USA 105:9495–9500

Naidoo R, Balmford A, Ferraro PJ, Polasky S, Ricketts TH, Rouget M (2006) Integrating economic cost into conservation planning. Trends Ecol Evol 21(12):681–687

Naidoo R, Ricketts TH (2006) Mapping the economic costs and benefits of conservation. PLoS Biol 4:e360

National Research Council (NRC) (2005) Valuing ecosystem services: toward better environmental decision-making. National Academy Press, Washington

Nelson E, Mendoza G, Regetz J, Polasky S, Tallis H, Cameron DR, Chan KMA, Daily G, Goldstein J, Kareiva P, Lonsdorf E, Naidoo R, Ricketts TH, Shaw MR (2009) Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Front Ecol Environ 7(1):4–11

Nelson E, Polasky S, Lewis DJ, Plantinga AJ, Lonsdorf E, White D, Bael D, Lawler JJ (2008) Efficiency of incentives to jointly increase carbon sequestration and species conservation on a landscape. Proc Natl Acad Sci USA 105(28):9471–9476

Parkhurst GM, Shogren JF (2007) Spatial incentives to coordinate contiguous habitat. Ecol Econ 64(2): 344–355

Parkhurst GM, Shogren JF, Bastian C, Kivi P, Donner J, Smith RBW (2002) Agglomeration bonus: an incentive mechanism to reunite fragmented habitat for biodiversity conservation. Ecol Econ 41:305–328

Pagiola S, Platais G (2007) Payments for environmental services: from theory to practice. World Bank, Washington

Polasky S, Nelson E, Lonsdorf E, Fackler P, Starfield A (2005) Conserving species in a working landscape: land use with biological and economic objectives. Ecol Appl 15(4):1387–1401.

Polasky S, Nelson E, Camm J, Csuti B, Fackler P, Lonsdorf E, Montgomery C, White D, Arthur J, Garber-Yonts B, Haight R, Kagan J, Starfield A, Tobalske C (2008) Where to put things? Spatial land management to sustain biodiversity and economic returns. Biol Conserv 141(6):1505–1524

Sánchez-Azofeifa GA, Pfaff A, Robalino JA, Boomhower J (2007) Costa Rica's payment for environmental services program: intention, implementation, and impact. Conserv Biol 21(5):1165–1173

Santelmann M, White D, Freemark K, Nassauer JI, Eilers JM, Vache KB, Danielson BJ, Corry RC, Clark ME, Polasky S, Cruse RM, Sifneos J, Rustigian H, Coiner C, Wu J, Debinski D (2004) Assessing alternative futures for agriculture in the U.S. Corn Belt. Landsc Ecol 19(4):357–374

Scheffer M, Carpenter SR, Foley JA, Folke C, Walker B (2001) Catastrophic shifts in ecosystems. Nature 413:591–596

Searchinger T, Heimlich R, Houghton RA, Dong F, Elobeid A, Fabiosa J, Tokgoz S, Hayes D, Yu T-H (2008) Use of U.S. croplands for biofuels increases greenhouse gases through emissions from land-use change. Science 319:1238–1240

Senjem N (2009) Overview of the Lake Pepin—Upper Mississippi River TMDL. Proceedings of TMDL 2009 Conference, Water Environment Federation Specialty Conference. Minneapolis, MN. August 9–12, 2009

Swallow BM, Sang JK, Nyabenge M, Bundotich DK, Duraiappah AK, Yatich TB (2009) Tradeoffs, synergies and traps among ecosystem services in the Lake Victoria basin of East Africa. Environ Sci Policy 12:504–519

Tallis HT, Ricketts T, Ennaanay D, Nelson E, Vigerstol K, Mendoza G, Wolny S, Olwero N, Aukema J, Foster J, Forrest J, Cameron D (2008) InVEST 1.003 beta User's Guide. The Natural Capital Project, Stanford

Tol RSJ (2009) The economic effects of climate change. J Econ Perspect 23(2):29–51

Turner RE, Rabalais NN, Justic D (2008) Gulf of Mexico hypoxia: alternate states and a legacy. Environ Sci Technol 42:2323–2327

USDA-NASS (2009) The national agricultural statistics service, United States Department of Agriculture. Quick Stats (ag statistics database). http://www.nass.usda.gov/Data_and_Statistics/Quick_Stats/index.asp

USDA-NRSC (2009) Soil survey staff, natural resources conservation service, United States Department of Agriculture. Soil Survey Geographic (SSURGO) Database for Minnesota. http://soildatamart.nrcs.usda.gov

Westra JV, Easter KW, Olson KD (2002) Targeting non-point source pollution control: phosphorus in the Minnesota River Basin. J Am Water Resour Assoc 38(2):493–505