Evaluating the Return in Ecosystem Services from Investment in Public Land Acquisitions

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Abstract:

We evaluate how land use change and the value ecosystem services affect the decision to invest in public land acquisitions. Our application is for the state of Minnesota, and we consider the acquisitions by Department of Natural Resources over the last two decades. We calculate a return on investment (ROI) in conservation showing the increase in the value of ecosystem goods and services from public lands per dollar spent on acquisition. A spatially-explicit modeling tool, the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST), quantifies how changes in land use and land cover (LULC) influence the provision and value of a suite of ecosystem services: carbon sequestration, timber production, water quality, habitat quality, and outdoor recreation. The present value of the difference in the value of ecosystem services from landscapes with and without acquisitions in 1992 and for the econometrically modeled future landscapes in 2022 and 2052 is the return from the investment in the acquisitions. We find a limited number of acquisitions have a ROI above one. Also, we observe the estimated return in the acquisitions is much more influenced by the economic value of ecosystem services than the projected development threat to the acquisitions.

Introduction

Public land acquisition for conservation, including land for parks, wildlife management areas, and scientific and natural areas, reflects a concern for sustaining the benefits provided by the natural environment. The Minnesota Department of Natural Resources (MNDNR) manages over 5.5 million acres in state forests, wildlife management areas, state parks and recreation areas, scientific and natural areas, and other areas. Investment in public land provides a suite of ecosystem services to the public. The ecosystem services evaluated in this paper are carbon storage, water quality improvement, habitat quality improvement for terrestrial species, timber production, and outdoor recreation. The change in the flow of these services due to the public land is quantified, and value is assigned to the change in the flow of the services. The return from the acquisitions can be compared to the cost of investment in the land to assess the rate of return on conservation investment.

The flow of ecosystem services from public land changes over time. Without public protection, agricultural and urban development will continue according to the expected land use change. Protection is more urgent if the land will be developed in the near future because the value of the services from a natural cover will be lost sooner. The change in the flow of services is initially small, but this rises steadily as the expected development occurs. The land use change surrounding the acquired land also influences the flow of services. For example, suppose the land surrounding an acquired parcel gains more natural cover, biodiversity increases with the additional habitat, but the retention of the polluting nutrients by the natural vegetation on the protected parcels has less influence on water quality if the surrounding land already provides this service. The provision of ecosystem services by the acquisitions changes over time and depends on the anticipated land use both inside and outside of the acquisition.

Many studies have examined the selection of reserve sites (Pressey et al., 1993; Church, Stoms and Davis, 1996), and later papers incorporated heterogeneity in land costs (Ando et al. 1998) and the vulnerability to land use conversion (Abbitt, Scott, and Wilcove, 2000; Myers et al. 2000). Newburn et al. (2006) note that the Ando et al. (1998) and the Abbitt, Scott and Wilcove (2000) studies provide contrary site rankings while analyzing similar data sets because land costs and the likelihood of future land use conversions are positively correlated. The cost of acquiring public land and the anticipated land-use land-cover (LULC) change should be jointly considered to target the reserve sites. The mapping of the anticipated LULC conversion is based on the projection of a national plot-level econometric model to determine the amount of each LULC type combined with a grid-cell (30m x 30m) econometric model to spatially arrange the LULC amounts onto the landscape.

Economic models of land-use change also concerned about changes in ecological processes are challenged by issues of scale. Socioeconomic variables are collected for administrative units rather than grid-cells, but ecological processes, such as habitat dispersal or the flows of nutrients, operate at finer scales. Plot-level econometric models are estimated with land use data that includes land cover transitions and inherent soil productivity characteristics, while economic information is incorporated at the coarser scale of county-level net returns to land uses (Lubowski et al. 2006;

Lewis and Plantinga 2007). Parcel-level econometric models can incorporate greater spatial detail by capturing neighborhood effects (Irwin and Bockstael 2002; Newburn et al. 2006; Lewis et al. 2009), but these models cannot incorporate broad-scale factors (such as crop and timber prices) that do not vary within small regions.

The closest prior paper to this analysis for the land use modeling is Lewis and Plantinga (2007) who integrate econometric land use change results with data on the actual landscape collected from the South Carolina Department of Natural Resources. They define parcels of five acres and collect information on land use in 1989, soil characteristics, and urban influence for each parcel to generate simulations of land use change from the fitted transition probabilities of the econometric results. The current paper predicts grid-cell land use change to urban and agriculture by estimating a relationship from the National Land Cover Database (NLCD) Change Product (Fry et al. 2009) and data collected from state agencies in Minnesota. The prediction results create a grid-cell landscape of the most likely places for urban and agriculture. This approach places future land cover at locations on a landscape consistent with fine-scale land use decision-making rather than random assignment.

We use the InVEST model (Integrated Valuation of Ecosystem Services and Tradeoffs; Tallis et al. 2010, http://invest.ecoinformatics.org/) to calculate the provision and economic value of ecosystems services provided from investment in public land. InVEST provides a consistent and transparent methodology for evaluating the tradeoffs across multiple ecosystem services from alternative land-use and land-management scenarios. The InVEST framework uses “ecological production functions” to predict the provision of ecosystem services, then combines these estimates with economic valuation methods to account for the full economic value of the ecosystem services for a given landscape. Because it is based upon a single platform of conforming modules, InVEST is well-suited to address concerns about double-counting. Tradeoffs among multiple ecosystem services and their values have been examined recently with InVEST for actual land-use change and a suite of alternative land use scenarios in Minnesota (Polasky et al. 2011). Nelson et al. (2009) use InVEST to compare biodiversity conservation and ecosystem service outcomes under three alternative land-use trajectories for the Willamette Basin.

A complete framework for reserve site selection considers the benefits of multiple services, public land acquisition costs, and the likelihood of future land-use conversion. This paper advances the study of reserve site selection by considering the joint provision of multiple ecosystem services coupled with an econometric model of fine scale land use change. Previous studies that consider the joint provision of multiple ecosystem services (Polasky et al. 2011; Nelson et al. 2009) have not included the visitor days for outdoor recreation and its value, predicted water quality and its value for an entire state based on nitrogen and phosphorous pollution, and based this provision on an econometrically based grid-cell model of land use change.

Description of the public land acquisitions and the methods for forecasting land-use change

This section begins by describing the data collected on the public land acquisitions by the MNDNR. Next, we describe the methods for the econometric model of land use change followed by a description of the model to spatially distribute urban and agricultural development on the landscape.

<u>Acquisitions of public land</u>

The MNDNR has geographic information systems (GIS) data of fee-title land acquisitions from 1989 to 2008 (Figure 1). Parcels less than forty acres are excluded because the location of the parcels on the landscape cannot be accurately determined. This leaves 123,966 acres of acquired land across six hundred and eighty parcels, and these parcels span nine MNDNR administrative categories (Table 1). Parcels are often additions to pre-existing MNDNR protected areas. Aquatic management areas are established to protect and manage rivers, lakes, and wetlands critical for aquatic life, water quality, and public fishing. Scientific and natural areas preserve natural features and rare resources of exceptional scientific and educational value. Wildlife management areas provide habitat for Minnesota's wildlife species, and outdoor recreation opportunities for hunting and wildlife-watching.

Most of acquired land was for wildlife management areas, but land for state parks, scientific and natural areas, and trails and waterways are also added. The most expensive acquisitions per acre are the aquatic management areas, scientific and natural areas, and state parks. Land purchased in these categories is close to lakes and rivers or has other valuable unique features that make this land more expensive.

<u>Scenarios of future land use</u>

We consider two scenarios of future land-use land-cover (LULC) based on an econometric land use change model. The baseline uses an extrapolation of the land use change trends in the 1990s, and the sensitivity analysis considers land use change assuming a net growth of cropland land based on historically high prices for crops observed from 2009 to 2011.

Baseline econometric analysis of land use change

The baseline projection of LULC change for 2022 and 2052 comes from an econometric model parameterized with USDA Natural Resources Inventory (NRI) data. The NRI data reports land use for 844,000 sampled private land plots throughout the United States (Nusser and Goebel 1997) and the years of 1992 and 1997 are used to parameterize the model. State and county projections reflect what happened there historically, but there is also measurement error at the scale of the county and state. Though the exact location of the NRI plots are not revealed for privacy reasons, county location and plot characteristics are available. This information is sufficient to estimate land use change probabilities for every county and land capability class (an integrated measure of soil quality and agricultural potential; USDA 1973).

The econometric estimation uses a nested logit specification (Lubowski et al. 2006) to identify the parameters (β) in the function $p_{ijkt} = p_{ijst} \cdot p_{ijkt|s} = F(\beta_{jkt} X_{it})$ where p_{ijk} is the probability that plot *i* changes from use *j* to use *k* between 1992 and 1997, β_{jk} is a vector of parameters associated with the *j*-to-*k* transition, X_i is a vector of independent variables (county net returns (Lubowski 2002), land capability measures) for plot *i*. The probability of choosing alternative *k* nested in a subgroup *s* can be expressed as the product of the probability, p_{ijst}, of choosing any of the alternatives grouped within *s*, and the conditional probability, $p_{ijkt|s}$, of choosing *k* given the choice of subgroup *s*. The independence of irrelevant alternatives (IIA) is imposed within but not across the specified subgroups (nests) of choices. No adjustment is made for potential spatial correlation of the model error terms. The NRI data are generated by a stratified sampling routine that ensure the plots are geographically dispersed.

The net returns to forest are measured as annualized revenues from timber production less management costs. Agricultural net returns equal the weighted average of the annual revenues from crop and pasture production less costs and plus government payments. Returns to urban land measure the annualized median value of a recently developed parcel used for a single-family home, less the value of structures. Landowners are assumed to form expectations of future returns based on the average of annual net returns over the preceding five-year period. Dummy variables are constructed for the land capability classes, and these variables are interacted with county average net returns to scale the returns up or down according to the productivity of the plot.

Agricultural expansion

Cropland was in decline in the United States in the 1990s. As a consequence, the baseline projection of land cover for 2022 and 2052 indicates a loss of cropland replaced by pasture and native vegetation. The previous decline in cropland may not persist because of the anticipated global population growth to nine billion people by 2050 and the demand in less developed countries for the diets of the developed world countries.

The real harvest prices for the major crops of Minnesota in 2011 are among the highest in the last twenty-five years (USDA 2011). The prices for corn, wheat, and soybeans are expected to remain high for the next decade (USDA 2010). However, the planted acres of principal crops from 1993 to 2011 have been largely steady throughout the decades of increasing yields and fluctuating prices (NASS 2011). Accordingly, we project a moderate rise in cropland of five percent from 1992 to 2022, and a slight increase in cropland of less than one percent from 2022 to 2052. Urban growth is projected to be the same as the baseline. Pasture and natural land decline according to the proportions observed in the baseline projection.

Projection of future land use to a fine scale landscape

The future land cover amounts by county come from the econometric analysis of land use change, but the spatial distribution of the land cover within the county is addressed using a fine scale land cover change model to allocate where in the county land uses are most likely to go. Finally, simple rules based on the expected ranking of economic rents allocate competing land covers to the grid-cell landscape.

Prioritizing the spatial distribution of land use

We construct a model of land change to urban and agriculture within a county based on fine scale explanatory characteristics of the site and the surrounding region. We model grid-cell conversion to urban and agriculture conditional on being developable in 1992, which excludes protected areas or places incompatible with development for physical or regulatory reasons such as water, urban, and wetlands.

Probit models explain grid-cell land use transition to urban and agriculture as a function of site and regional characteristics. The National Land Cover Database 1992–2001 Land Cover Change Retrofit product (hereafter the NLCD change product; Fry et al. 2009) provides the land use change data at a 30m resolution within a GIS.[1] We extract a subset of fewer than a million cells within four kilometers of where land change occurred from 1992 to 2001. The grid-cell boundaries through extraction and overly with GIS layers obtain many site and regional characteristics on soil quality, accessibility to roads and urban centers, and neighboring land uses. The average percent slope and elevation was calculated for each grid-cell. Proximity measures include the Euclidean distance in meters from U.S. Census-defined communities, which are geographic areas defined by jurisdictional or political boundaries and included in the U.S. Census definitions of places (census-designated place, consolidated city, and incorporated place) and the Euclidean distance in meters from road centerlines of highways within Minnesota.[2] A proximity measure for distance to lakes is created to measure their potential amenity effect on the conversion to urban.

Dummy variables are created for the developable LULC types (barren, forest, grassland, and agriculture) from the NLCD change product. A soil productivity value based on non-irrigated yields within sub-county soil regions is constructed from the U.S. General Soil Map (STATSGO2) Database (USDA National Resources Conservation Service, http://soils.usda.gov/survey/geography/statsgo/).[3] A set of dummy variables are constructed for the land use prior to conversion to urban or agriculture to account for differences in conversion cost based on the land cover. Interaction variables of soil quality and the dummy variable for the forest land type controls for the effect that productive forest has less potential for agriculture. Additionally, Euclidean measures for proximity to the 1992 classification of urban, forest, grassland, and agriculture from the NLCD Change Product are created for an examination of the effect of neighboring LULC types. Variables for the percent of grid-cells with urban classification within half and three-half of a kilometer radius account for the agglomeration of development around communities.

[1] Definitions for the LULC types are provided at http://www.epa.gov/mrlc/change.html.

[2] For detailed definitions, see http://www.census.gov/geo/www/cob/pl_metadata.html and http://www.dot.state.mn.us/maps/gisbase/metadata/road_metadata.htm.

[3] The more spatially detailed Soil Survey Geographic (SSURGO) Database is not currently complete in GIS form for the whole state.

Separate probit models are estimated for urban and agricultural conversion because the developable grid-cells for urban includes agriculture, but the developable grid-cells for agriculture do not include land already in agriculture. We check for spatial autocorrelation by calculating Moran's I but find no evidence of this autocorrelation because the land use change comes from largely isolated grid cells around the state. Hence the estimated models reported in Table 2 are the standard probit models for the full sample.

The change to urban is more likely on flat and low elevation land since steeper slopes raise construction costs and higher elevations are further from commercial and employment centers. Conversion to urban is more likely close to Census-defined communities, highways, and the urban classification from the NLCD Change Product. The percent of urban within half-mile and three-half mile spurs urban development, and the percent of urban within half-mile has a stronger effect than the percent of urban within three-half mile. The dummy variables for the existing LULC type reflect the cost of conversion to urban cover, and the marginal effect suggests grassland is the most likely to convert, followed by forest, and then agriculture. Lakes and forests have a role as amenities and proximity to these increases the likelihood of urban. Since agriculture is a location of human activity, proximity to agriculture increases the construction of sheds, barns, and other man-made structures. The principal drivers of change to urban use are proximity to the urban classification from the NLCD Change Product, the percent of urban use within half-mile and three-half-mile, the slope, and the existing land cover type.

Conversion to agriculture is more likely where there are productive soils. Existing grassland is more likely to convert to agriculture than forest. Steeper slopes and higher elevations deter the conversion to agriculture. Proximity to Census-defined communities and highways encourage the conversion to agriculture, while proximity to the urban and forest classifications from the NLCD change product deter conversion. The principal determinants of the spatial distribution of change to agriculture are proximity to existing agriculture from the NLCD Change Product, soil productivity, the existing land cover type, and the slope.

The coefficient results from the probit models are combined with the explanatory data within a GIS to form a 30m resolution map of the suitability of land use change to urban or agricultural use. The estimated coefficients of the explanatory characteristics are multiplied by the value of the associated data on the landscape, and this is summed to form a single metric that prioritizes urban or agricultural use on the landscape. The two maps, known as the urban and agricultural change suitability layers, allocate the future urban and agricultural use within a county.

Assignment of land use to the grid-cell landscape

The land use prior to the acquisitions by the MNDNR is derived from the 1992 NLCD (Vogelmann et al. 2001). The 1992 NLCD uses 30m resolution satellite image classification based on Landsat Thematic Mapper imagery. Some acquisitions were made as early as 1989, but the land use on the parcels is unlikely to have changed prior to 1992 because the restoration of previously agricultural or urban use takes several years (Steve Merchant, MNDNR). The 1992 NLCD provides information on more land cover classes than the NRI. The NLCD classes are therefore grouped into forest (NLCD classes 41, 42, 43), cropland (61, 82, 83, 84), pasture (81), grassland (51, 71), and urban (21-23, 85). Existing protected areas and urban land, along with the water, barren, and wetland classifications (11, 12, 31-33, 91, 92) in the 1992 NLCD do not transition. A description of each of the NLCD land cover classes is in Table A-1. Figure 1 illustrates the 1992 land cover for the state, a small region near Mankato, and the boundary of a particular acquisition in that region prior to conservation.

The quantity of land in use j from the NRI in 1992 does not match the quantity of land in use j from the 1992 NLCD because the data are collected at different resolutions by different agencies. We add the change in the land cover proportions from the NRI based forecasts to the proportion of corresponding land cover from the 1992 NLCD. This then determines the proportion of the future land cover in use j for the 30m resolution map. Finally, the multiplication of the proportion of the land cover in use j by the total land in the county from the 1992 NLCD determines the quantity of future land cover in use j for each county.

Using the quantity of future land in use *j*, the landscape is filled out in a progressive fashion beginning with urban, followed by cropland, then pasture, and an appropriate natural land cover fills in the rest of the county. The order that the land covers fill the landscape is based on the expected rents from each land use. We assume the bid of urban developers always exceeds or equals the bid of farmers for cropland, the bid of farmers for cropland always exceeds or equals the bid of ranchers and dairymen for pasture, and the bid for pasture always exceeds the bid for natural land. Urban land is allocated to the landscape according to the urban change suitability layer. The agricultural change suitability layer spatially distributes cropland and then pasture. The remaining land is assigned to the natural land cover observed in 1992 or if not previously in a natural land cover then assigned to forest or grassland according to the pre-settlement vegetation.[4]

Figure 2 indicates the land cover within and surrounding a single acquisition for the case of conservation in 1992, the 2052 baseline without conservation, and the 2052 agricultural expansion scenario without conservation. The conserved land cover for the particular parcel shown is mostly grassland and shrub but also some forest. This natural cover in the acquisition will persist on the landscape in 2022 and 2052 although the land cover around the acquisition will change. The anticipated land cover without the acquisition is shown for the case of the baseline agricultural contraction where much of the cropland turns to pasture and in the case of the agricultural expansion where the grassland and pasture turns to cropland. Table A-2 indicates the state acquisition acres by land cover for the landscape 'with acquisitions' that is the same for all years, and the land cover for the landscapes 'without acquisitions' that is different across years and LULC change scenario.

Description of the ecosystem service models

We use InVEST to study the change in the provision and value of carbon storage, water quality, habitat quality and availability, and timber production. The outdoor recreation visitor use and value

[4] Vegetation is based on General Land Office Survey records from the 19th and early 20th century http://deli.dnr.state.mn.us/metadata.html?id=L250000140201.

estimating models developed by the 2006 Wildlife Habitat Policy Research Program (Loomis et al. 2008) evaluate fishing, hunting, and wildlife viewing use and values. We report in 2010 dollars the monetary value of ecosystem services in terms of the value of the annual flow of the services from the public land acquisitions. This annual flow of the services changes over time as the landscape develops further without the acquisitions. The present value of annual services from the acquisition of public land compared to the cost of acquiring the public land is used to assess the return on investment in conservation.

Carbon storage and sequestration

The carbon model accounts for carbon stored in the soil and in above-ground and below-ground biomass. The amount of carbon stored in each of these pools depends primarily on LULC (e.g., agriculture, forest, grassland, wetland) but is also affected by land management (e.g., whether the land is protected or managed for timber harvest, and the forest rotation age for timber land). For carbon storage in the future periods we assume that land use and land management had existed long enough for carbon storage to reach its equilibrium (steady-state) level. We assumed storage equilibrium because we lacked state-wide data on age classes of forests and other LULC that would allow for a more exact estimation of carbon storage values in the future years. To account for the significant differences in carbon storage across wetland types we subdivided the wetland LULC into a peatland and a prairie pothole wetland category based on their occurrence in the state (Glaser 1987). Steady-state carbon levels for all LULC types are listed in Table A-3 and A-4.

We convert a scenario's carbon stock to an annualized flow of carbon sequestration by dividing the change in carbon stock with a change in land use by the time it takes for carbon storage to reach equilibrium for a particular LULC type. This annualized sequestration 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 estimates of the social cost of carbon from a meta-analysis of peer-reviewed studies (Tol 2009). The social cost of carbon represents an estimate of the increase in damages from intensified climate change associated with an additional ton of carbon dioxide emitted to the atmosphere. We use the 33rd and 67th percentile values to generate a lower-end and higher-end estimate from the fitted-weighted distribution reported in Tol (2009) and normalized to 2010$. The lower estimate is **$**21.76 per Mg C and the higher estimate is $91. The European ETS market price for carbon of 14.91€ per Mg CO_2 (as of 12 November 2010) translates to $74.87 per ton C, which falls toward the higher-end of social cost of carbon range.

Water Quality

The InVEST water quality model estimates the annual nutrient retention service provided by land cover. We focus on phosphorus and nitrogen pollution, which are leading causes of surface water impairment in the upper Midwest (Carpenter et al. 1998). The water quality model is run for the eighty-one eight-digit hydrologic unit code (HUC) basins in Minnesota. InVEST applies a two-step process to determine the influence of land cover on water quality. First, the model calculates the average annual water yield in each grid cell using climate data, geomorphological information, and LULC characteristics. Water yield is defined as precipitation minus evapotranspiration. The model assumes that all precipitation not lost to evapotranspiration goes to surface water runoff. There is no modeling of subsurface or ground water flows. The routing of surface water flow across grid cells is defined using a digital elevation model.

In the second step, water yield is combined with information about nutrient loading and the filtering (nutrient retention) capacities of each LULC type (see Table A-8) to calculate the annual nutrient exports from each cell. Nutrient exports from cells are routed via surface water flow to other cells, where some of the nutrients may be filtered or additional nutrients added, until it flows into a water body. Once nutrients reach a water body we assume no additional retention or removal before delivery to the mouth of the watershed. To estimate the change in water quality with versus without conservation acquisitions, the InVEST water quality model for nitrogen and phosphorous by eight-digit HUC basins is run for each scenario.

Several recent meta-analyses summarize the WTP value of improved water quality for lakes and rivers in the United States (Van Houtven et al. 2007, Johnston and Besedin 2009, Johnston and Thomassin 2010). Johnston and Besedin (2009) express household WTP for water quality improvements as a function of study attributes, geographical area, water body type, population attributes, and changes in water quality. We use the modeled WTP values for nitrogen and phosphorus based on the national meta-analysis conducted by Johnston and Besedin (2009) as a low-end value for water quality. In that study, household WTP studies are mapped to the Resources for the Future (RFF) water quality ladder, where the water quality ladder examines changes in aquatic uses (drinking, boating, swimming, and fishing) in response to variations in biophysical characteristics (dissolved oxygen, turbidity, pH).

Using this approach, baseline water quality is defined using data on lake water clarity (Olmanson et al. 2008) and stream and lake impairment. The stream and lake impairment is classified based on the ability to support aquatic life and recreation, which facilitates the mapping of baseline water quality to water quality ladder metrics such as swimmable and drinkable water. Based on Carlson (1977) and consultation with local water quality experts, a 50% reduction in nutrient loading is assumed to relate to a two-point increase along the water quality ladder. We then used the Johnston and Besedin (2009) benefits transfer model to estimate an annual household WTP value for changes in water quality benefits for a 50% improvement in nutrient loadings given the baseline water quality. The estimates of annual household WTP for the 50% improvement vary across the basins from \$5.94 - \$19.30 per household for nitrogen and \$24.97 - \$44.72 per household for phosphorous.

We use as a high-end value for water quality a Minnesota-specific WTP study that focused on phosphorus mitigation in the Minnesota River Basin (Mathews et al. 2002). Their study estimates an annual WTP at $141 per household for a 40% reduction in phosphorus for the Minnesota River Basin. Next, the values are prorated to the percent changes in water quality modeled from InVEST. For example, acquisitions are modeled to be responsible for a 0.07% reduction in nitrogen loadings in the Rainy River Basin; therefore a WTP value of $4.17 for a 50% reduction was prorated to $0.01. Finally, the prorated WTP values for nitrogen (focusing on river endpoints) and phosphorus (focusing on lake endpoints) in each basin are multiplied by the number of households per basin for the appropriate year (Minnesota Demographic Center (2007) projections for 2022 and 2052) to estimate a state-wide annual household WTP for improvements in water quality.

Habitat extent and quality

The InVEST habitat model accounts for the spatial extent and quality of habitat for a targeted conservation objective (e.g., forest birds, amphibians). Maps of LULC are transformed into maps of habitat based on the habitat suitability of a LULC 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.

In this application, we consider a broad conservation objective that focuses on general terrestrial biodiversity that includes all native species. Each LULC type is given a habitat suitability score of 0 to 1 for each 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). See Table A-5 for the definition of habitat suitability across LULC types for this conservation objective.

The habitat quality score 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. See Table A-5 and A-6 for the sensitivity scores and the influence of threats determined from the literature and expert knowledge.

We generate a landscape habitat quality score for each scenario by summing the grid cell habitat quality scores for a scenario. Because of the influence of adjacent patches on habitat scores, the spatial pattern of land use as well as the overall amount of habitat will matter in determining the landscape habitat quality score. Higher habitat quality scores indicate landscapes with more favorable conditions (e.g., resources available for survival, reproduction, and population persistence) for the given conservation objective.

Timber production

We use 2002 county-level data from Lubowski (2002) and Lubowski et al. (2006, 2008) to estimate annual net returns to forestry (Table A-7). For the landscapes with conservation, timber harvest is assumed to occur on all acquisition of forest land designated as State Forest by the MNDNR but not on other land. For the landscape without conservation, the value of timber harvest is capitalized into the purchase price of the land, which is reflected in the acquisition costs. To avoid double counting, we do not include timber harvest value with the values of the landscapes without conservation. To calculate the total value of forestry, we multiply forest acreage by the per acre net return to forestry in the county. Estimated returns to forestry in a county are based on the assumption that all harvested forests are managed on an optimal, even-age rotation basis to produce saw timber (similar to Lubowski 2002).

Outdoor recreation

Making land publicly accessible will tend to increase activity and increase the value of recreation. The recreation model, based on the visitor use models developed for the 2006 Wildlife Habitat Policy Research Program (Loomis et al. 2008), evaluates fishing, hunting, or wildlife viewing uses associated with wildlife areas over the course of a year. As of 2006, 91% of Minnesota residents at least 16 years old fished, hunted, or viewed wildlife in Minnesota (U.S. Fish & Wildlife Service 2008).

The visitation models are originally estimated from a sample of National Wildlife Refuges with available data on visits per activity, acres of the refuge, natural features (lakes, rivers, and oceans), per-capita income, and the county population within a 60-mile radius of the particular refuge (Caudill and Henderson 2005). We assume for the visitation transfer that the acquired Minnesota land is similar to the visitation to the National Wildlife Refuges, i.e., that a site with similar characteristics will receive a similar number of visitors regardless of state or federal ownership.

The models used to explain visitation rates at the National Wildlife Refuges generally include explanatory variables that are statistically significant at the 10% or higher level. The fishing visitation model includes total acres, per-capita income, and surrounding county population for explanatory variables (Table A-11). Populations for 2022 and 2052 are based on projections of the Minnesota Demographic Center (2007). The hunting visitation model includes the presence of water and total acres for explanatory variables (Table A-10). The wildlife-watching visitation model includes the presence of an ocean (not applicable for Minnesota), per-capita income, total acres, and the surrounding county population for explanatory variables (Table A-9). Scientific and Natural Areas do not allow fishing, hunting, and wildlife-watching visitation so we do not estimate visitor days for these areas. Also, hunting is not allowed at State Parks.

The number of visiting days to a new wildlife area depends on the existing amount of publically available land. A new wildlife area will attract more visitors if there are few existing wildlife areas in the region. On average, the acreage of publicly available land within 60 miles of an acquisition is 152,720 acres. The acreage of pre-existing public land is obtained from the protected areas database (PAD 2009). We use high-end and low-end estimates for visitation based on variation in the amount of nearby public land that could act as alternative visitation sites. The low-end estimate for visitation assumes the acquired land is an extension of already existing public land. The high-end estimate for visitation uses a lower amount of publicly available land to embody the possibility of newly created site. This lower amount of publicly available land is a one standard deviation lower amount of publically available land based on the variation of public land in the Minnesota counties.

In the second step, annual visiting days per activity is multiplied by value of the activity per day to arrive at an annual value per activity. The annual value per activity for fishing, hunting, and wildlife-viewing is summed to calculate the total annual value of recreation for Minnesota. Using the benefits transfer model developed for the 2006 Wildlife Habitat Policy Research Program (Loomis et al. 2008), we used values of $40, $42, and $47 for the value of a day of fishing, hunting and wildlife-viewing, respectively (Table A-12). The value per day comes from the compilation of databases assembled by Loomis (2005) that is up to date in terms of the studies available as of the beginning of 2007. The hunting value per day is the average of 192 estimates from twenty-one studies of big game, small game, and migratory bird hunting value per day in the Northeast. The fishing value per day is the average of fifty-eight estimates from fourteen studies of cold water fishing value per day in the Northeast. The wildlife-viewing value per day is the average of eighty-eight estimates from nine studies of wildlife-viewing value per day in the Northeast. The annual value assumes that the total value of visitor days increases linearly in the number of visitor days.

Results

In this section we report the results of applying the InVEST model for carbon sequestration, water quality, habitat quality, timber production, and outdoor recreation (fishing, hunting, and wildlife-viewing) to baseline and agricultural expansion LULC change scenarios for landscapes with and without acquisitions. Comparing the difference between the landscapes with and without acquisitions for 1992, 2022, and 2052 allows us to compute the present value of the increase in the provision and value of ecosystem services from conservation. For timber and outdoor recreation, values are calculated only for landscapes with the acquisitions because the values of the services on private land are assumed to be incorporated into the purchase price of the land. At the end of this section, we use the results of the analysis to calculate a return on conservation investment for the state.

Land acquisition for conservation in results in an increase in the provision of ecosystem services (Table 3) compared to a landscape without conservation. Carbon sequestration increases from between 7 to 16 thousand metric tons, depending on the LULC change, primarily from the increase and retention of grassland and non-harvested forest cover on acquired lands. Nitrogen and phosphorous loadings to water bodies decline with the acquisition of conservation lands. Some basins exhibit greater increases in water quality due to a larger reduction of development in those basins and the adjoining upstream basins, relative to the baseline water quality. Habitat quality for terrestrial biodiversity also improves. Results for the water and habitat quality models reflect the inherently spatial nature of ecological processes and the importance of considering surrounding landscape-level processes when managing for these services, which explain why the percent change of state-wide services, rather than just the acquisition, is reported. For timber production and outdoor recreation, we report the harvested acres and visitor days on the acquired lands only and not statewide totals. As noted above, we assume the values for timber and outdoor recreation on the private landscape is included in the value of land at the time of purchase.

Table 4 shows the annual present value of the ecosystem services from the acquisitions for 2022 and 2052, and the present value of the complete stream of annual ecosystem services for the baseline and the agricultural expansion scenarios. We use a real discount rate of 2% from the market rate of return for risk-free financial assets (Howarth 2009). The present value of the complete stream of annual ecosystem services is computed by calculating the present value of annual stream of ecosystem services from the year of the acquisition using the 1992 landscape until 2010, and then the 2022 landscape from 2010 to 2040, and finally the 2052 landscape from 2040 onward. In the baseline, the annual benefit per acre is higher in 2022 than 2052 because of a decline in the agriculture and the role of discounting. The same trend is observed for the agricultural expansion scenario but only because of the affect of discounting.

The largest values are associated with carbon sequestration, followed by outdoor recreation and water quality improvement from phosphorus run-off reductions. At the state level, each of these services generates annual values of over one million on the high-end and over two-hundred thousand on the low-end. Timber production and nitrogen run-off reductions collectively generate an annual value of less than fifty thousand. Nitrogen reduction values are low because the value per household is small, and timber values are low because there is little timber harvesting allowed on acquired lands. Outdoor recreation ranks highly because it is assumed to occur on all acres of acquired land except SNAs or hunting in State Parks. Increases in carbon sequestration and water quality improvement only occur when there are expected land changes on the acquisitions.

The spatial variation of the development threat and the annual value of the ecosystem services per acre on acquired land by watershed are shown in Figure 3. The development threat based on the expected land use change on the acquisitions is largest in the southern and central regions of the state where the agricultural and population centers are. An acquisition distant from major population and agricultural centers could also face development threat if the acquisition happens to be close to the limited population and agriculture of that watershed. The carbon value per acre is the highest in the Southeast because the development threat is high and losses of forest cover along the stream gullies common to the area. The value of the phosphorous reductions per acre is largest in watersheds with a large population, a high development threat, and a low level of existing phosphorous exports (i.e. a small reduction in phosphorous can mean a large percentage decrease in phosphorous export). The outdoor recreation value per acre is highest where population and per capita income is large, and the existing amount of public land is low.

Table 5 indicates the present value of the benefits per acre of all the ecosystem services for the baseline and the agricultural expansion scenarios, from $950 per acre for the low end baseline to $3,310 per acre for the high end agricultural expansion. The scenario for the LULC change has a smaller influence on the benefits per acre than the low and high-end estimates based on the value per unit for the service. The average cost per acre of the acquisition land is $1,720 per acre. The ROI is defined as the ratio of the present value of the benefits per acre to the acquisition cost per acre. A value of one or above indicates the state receives a return from the land equivalent or greater than the investment in the land. Comparing present value estimates, the low-end of the ROI ranges from 0.55 to 0.62 and the high-end ranges from 1.76 to 1.92. The difference in the low and high-end of the ROI stems from the value per unit for carbon sequestration, water pollution reduction, and the existing amount of public land for the outdoor recreation. Using the high-end of the ROI, the state pays back the cost of investment in the acquired land in 64 to 71 years.

The annual benefit per acre for all services, the cost per acre, and the ROI by watershed in 2052 are shown in Figure 4 for low end and high end and the baseline and agricultural expansion scenarios. Annual benefits per acre for the state in 2052 at the low end is $14-15 per acre and at the high end is $46-48 per acre. The annual benefits are highest in the South and the West where the population and agricultural centers are, and costs per acre are the highest in Southeast around the population centers. By incorporating benefit and costs per acre together, the ROI is seen to be the highest in the East and North suggesting the costs per acre are proportionally higher than the benefits per acre in the South. The ROI for the state in 2052 at the low end is 0.57-0.59 and at the high end is 1.78-1.88. The frequency of annual benefits per acre and the ROI for each watershed is shown in Figure 5. The annual benefits per acre are clustered near zero and between ten and twenty, and only a few watersheds have annual benefits per acre greater than forty. The ROI is slightly more diffuse, but the bulk of values are between zero and one. The watersheds with the highest annual benefits per acre do not correspond to the watersheds with the highest ROIs.

Three strategies that could guide purchase decisions by state agencies include the least cost per acre, highest benefit per acre, and return on investment. Figure 6 indicates three scatter plots to examine if these strategies match the acquisition purchases from 1989 to 2008. The amount of acres purchased is plotted against the cost per acre, the benefits per acre, and the return on investment in each watershed. The observed relationship with the cost per acre is negative, as expected, but not statistically significant, and the relationship with benefits per acre is unexpectedly negative but also not statistically significant. The return on investment relationship with the amount of acres purchased in a watershed is positive and statistically significant.

Conclusion

In this paper we applied the InVEST model to evaluate the return on investment of land acquisitions by the MNDNR by considering the joint provision of ecosystem services, species habitat, and the cost of the land acquisitions. Our results indicate the return from ecosystem services exceed the cost of investment in the acquisitions when the high-end values for the ecosystem services are used, while the scenarios for land-use change that affect the provision of ecosystem services have less influence on the returns. A statistically significant positive relationship is found between the amount of acquired land in a watershed and the corresponding return on investment which indicates the targeting strategy observed by the MNDNR is consistent with return on investment. The observed distribution of return on investment of the acquisition is centered at 0.6 with the low-end values and 1.8 at the high-end values. Acquisitions with higher returns on investment have acquisition costs lower than the average and benefits close to the average. In other words, acquisitions with low development pressure and distant from population centers are good deals for the state where the acquisition costs are much lower.

The decisions of many separate landowners generate the spatial patterns of land use and land cover that determines the provision of ecosystem benefits for species habitat and water quality. This makes the modeling of land use within and surrounding the acquisitions important for the assessment of ecosystem services. Our land use change approach models pixel level changes to urban and agriculture to capture the fine scale differences important for ecological processes, but land use is also determined by parcel boundaries. Accounting for ecosystem service flows differentially through time also requires careful thought. For example, a change from annual crops to perennial grassland or forests may yield water quality improvements and habitat benefits and result in a build-up of carbon stocks through time. Eventually, carbon sequestration will cease as a new equilibrium level of carbon storage is reached but water quality improvements and habitat benefits will continue to flow as long as the natural cover persist.

The range of uncertainty around many non-market values can make the ROI rankings among acquisitions ambiguous. For example, if recreation is less valuable further from a population center, then acquisitions closer to the urban areas would be higher in the rankings. Larger non-market values for water quality in the Mississippi River would make acquisitions closer to the river and upstream of the Twin Cities metropolitan area have a higher ranking. 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. By including the water quality improvements in downstream states along the Mississippi River or the province of Manitoba along the Red 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.

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. Values for non-market goods and services are functions that depend on the levels of provision of various goods and services rather than constants in a given year as we assumed. While low and high-end estimates of ecosystem service provision and valuation are used, an exploration of uncertainty with probability distributions would identify the most likely possibilities rather than just the lowest and highest, as well as help to evaluate the robustness of the return on investment to such uncertainty.

The distributional consequences of acquisitions matter because the highest return on investment is associated with the purchases distant from population centers. The beneficiaries of the acquisitions distant from cities are those who have the means to travel long distances for recreation or those most likely to be adversely affected by future climate change. Alternative values for ecosystem services based on the preferences of other segments of society could change the ranking among the acquisitions.

We consider a broad set of ecosystem services based on the availability of applicable statewide data for Minnesota. However, we did not examine an exhaustive set of ecosystem services, principally because of the lack of relevant data. For example, we do not estimate flood damage reduction, pollination potential, air quality improvements, and proximity to open space. To the extent that all of these, if examined in a more detailed study, would have positive values, our numbers are likely to underestimate the full economic value of the investment decisions examined in this study.

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Figure 1. Location of the 1989-2008 acquisitions and the 1992 land cover without conservation

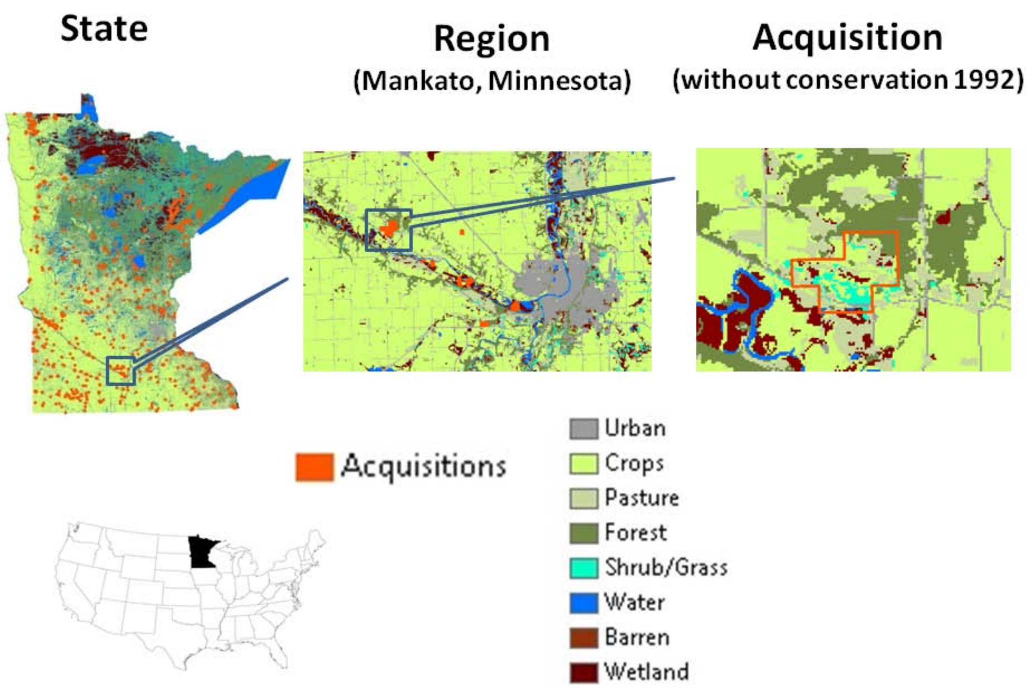


Figure 2. Land use cover of an acquisition and the surrounding landscape near Mankato, Minnesota is shown in 1992 and 2052 for with and without conservation scenarios. The baseline and agricultural expansion projections of land cover are shown for 2052.

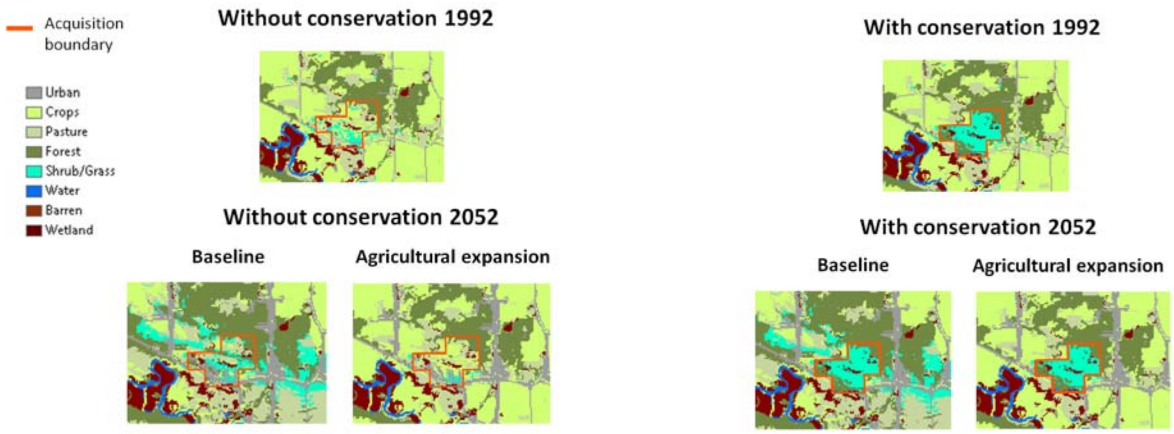


Figure 3. Development threat represented by the percent of acquisition developed from 1992 to 2052, the annual benefit (2010$) per acre of acquisition in 2052 for carbon, reduction of phosphorous export, and recreation, and the percent change in the overall species score in 2052 for the baseline and agricultural expansion scenarios by 8-digit watershed. The numbers by the side of each map indicate the state average.

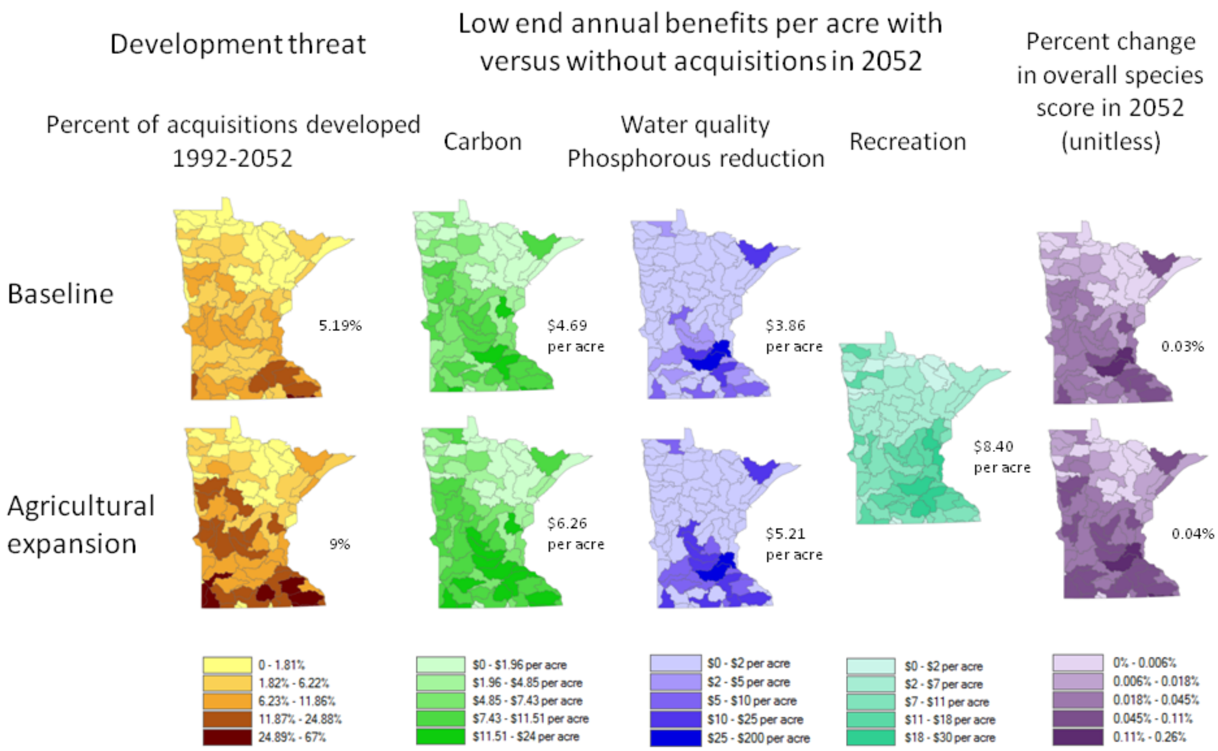


Figure 4. The low and high end annual benefits per acre (2010$) for all the ecosystem services (species score not included), the costs per acre, and the return on investment from the acquisition for the baseline and agricultural expansion scenarios by 8-digit watershed. The numbers by the side of each map indicate the state average.

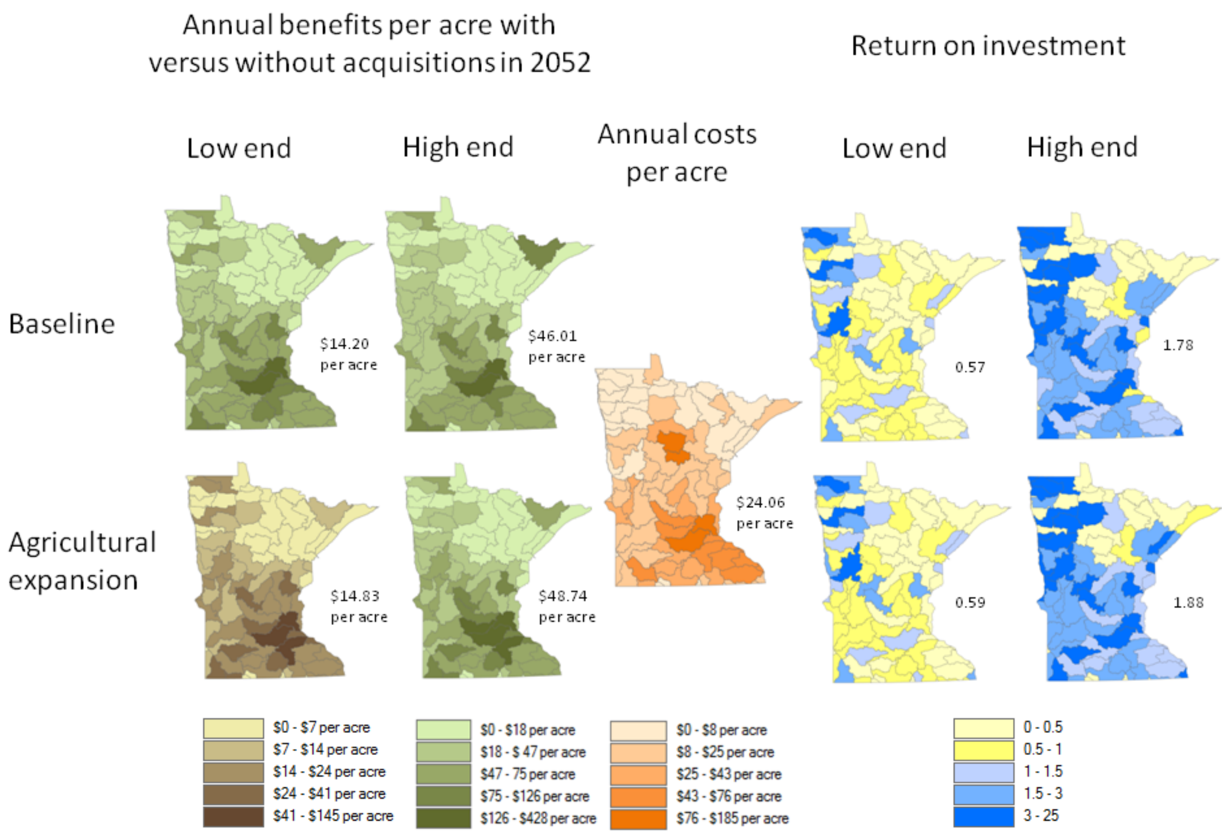


Figure 5. Frequency of the low end annual benefits per acre and the return on investment by watershed for the baseline and agricultural expansion scenarios. Letters indicate particular watersheds on each of the frequency graphs.

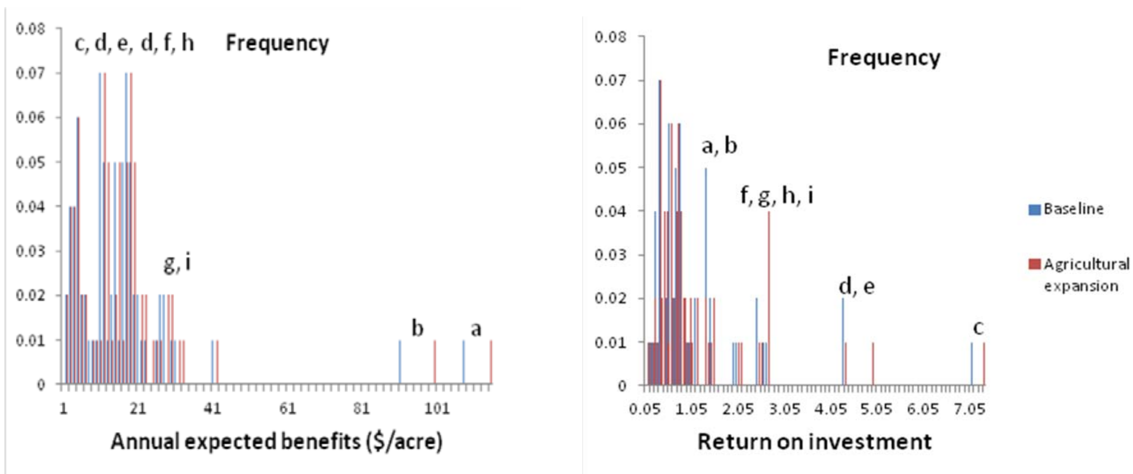
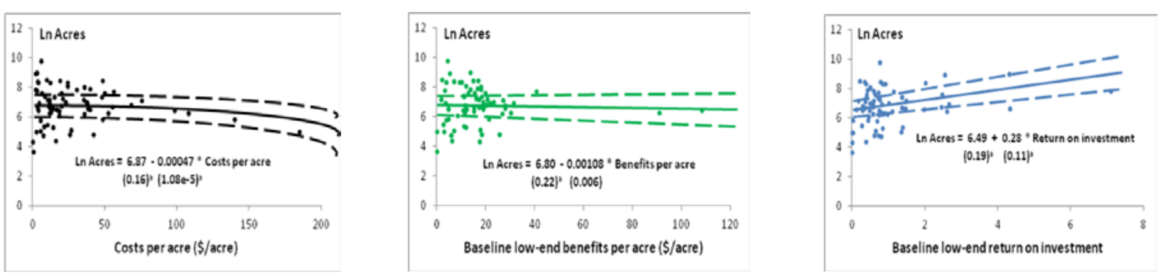


Figure 6. Natural log of the acquired acres in relation to the (a) cost per acre, and the baseline low-end (b) benefits per acre and (c) return on investment. Solid line represents the best fitting model, dashed lines represent ± 1 standard error (SE).

(a) **(b)** **(c)**



Note: Robust standard errors shown in parentheses. [a] indicates indicates significance at the 10% level. The best fit in (a) based on the residual sum of squares criterion is a non-linear quadratic.

Table 1. Acquisitions by MNDNR administrative classification from 1989 to 2008

Administrative classification	Acres	Real expenditure (2010$) per acre
Aquatic management area	1,945	5,472
Trails and waterways	16,688	406
Off-highway vehicle	1,710	366
Scientific and natural area	13,654	2,818
State forest	4,455	1,355
State park	13,131	2,815
State recreation area	1,214	1,265
Wild and scenic river	121	770
Wildlife management area	71,047	1,684
Total	123,966	--

Table 2. Probit Models for Land-Use Change in Minnesota (Baseline Land-Use Category = Developable* 30-meter grid-cells)

Variable	Urban			Agriculture		
	Marginal Effect	Robust Std. Error	Pr (>|z|)	Marginal Effect	Robust Std. Error	Pr (>|z|)
Slope	-422e-06	2.39e-06	0.07	-7.12e-04	9.39e-05	0
Elevation	-1.13e-07	4.26e-08	0.01	-4.24e-05	1.47e-06	0
Distance to Census-defined Community	-9.26e-09	2.10e-09	0	-2.09e-07	3.46e-08	0
Distance to highways	-1.59e-09	2.43e-09	0.51	-4.21e-07	4.80e-08	0
Distance to lakes	-3.32e-08	4.39e-09	0		--	
Distance to urban	-4.12e-06	3.73e-07	0	3.03e-06	5.91e-07	0
Distance to forest	-4.77e-07	6.99e-08	0	2.81e-06	2.36e-06	0.23
Distance to agriculture	-6.25e-07	1.29e-07	0	-1.61e-04	1.75e-06	0
Percent urban						
Half kilometer	2.26e-05	3.02e-06	0		--	
Three-half kilometer	1.46e-05	1.94e-06	0		--	
Dummy variables for existing LULC (Barren omitted dummy variable)						
Forest	0.04	3.19e-03	0	0.07	2.28e-03	0
Grassland	0.49	0.02	0	0.98	1.21e-03	0
Agriculture	0.02	1.67e-03	0		--	
Soil productivity (Grassland omitted dummy variable)		--		1.37e-03	1.71e-04	0
Soil productivity*Forest		--		-6.31e-03	2.10e-04	0.23
Constant						
Number of observations		661,738			310,565	
Log likelihood		-27,056			-53,694	

* We estimate separate land-use change models for urban and agriculture because developable grid-cells for urban includes agriculture, though this is not the case for agriculture.

Table 3: Biophysical change in ecosystem services in the state with versus without the acquisitions for 1992, 2022 and 2052 in the baseline and agricultural expansion scenarios

Ecosystem services	1992	Baseline		Agricultural expansion	
		2022	2052	2022	2052
Carbon sequestration (metric tons of C)		7,483	6,721	14,589	15,668
Water pollution reduction: phosphorus	0.15%	0.11%	0.11%	0.21%	0.23%
Change in forest/grassland birds biodiversity measure	0.12%	0.11%	0.19%	0.33%	0.49%
Timber production (harvested acres)	3,161	3,161	3,161	3,161	3,161
Outdoor recreation (visitor days)		36,759	40,677	36,759	40,677
		20,322	22,496	20,322	22,496

Table 4: Annual present value of ecosystem services (thousands 2010 $) with versus without the acquisitions for 1992 and the 2022, 2052, and the present value of the stream of annual ecosystem services for the baseline and the agricultural expansion scenarios.

Ecosystem services	Value per unit ($)	1992	Baseline			Agricultural expansion		
			2022	2052	Present value	2022	2052	Present value
Carbon sequestration	$91.10 per tC	2,100	1,971	1,058	145,616	2,482	1,413	165,252
	$21.76 per tC	590	471	253	34,819	593	338	44,251
Water pollution reduction: phosphorus[a]	$0-8.62 per household depending on basin	1,900	1,554	1,103	122,817	1,489	1,491	138,588
	$0-1.57 per household depending on basin	400	295	208	23,329	279	281	27,559
Timber production	-$0.67-5 per acre depending on county	10	7	5	521	7	5	521
Outdoor recreation	$40-47 per visitor day depending on activity	1,700	1,345	821	102,517	1,345	821	102,517
		910	744	454	56,684	744	454	56,684
Sum value of all services	High	5,320	4,916	3,008	374,364	5,358	3,750	410,212
	Low	2,110	1,556	941	118,246	1,658	1,098	132,349

[a] The per household value of a percent reduction in phosphorous is from Matthews et al. (2002) at the high end and from Johnson et al. (2006) at the low-end.

Table 5: Present value of benefits and costs per acre (2010 $), return to investment, and the pay-back period

Return on investment in ecosystem services		**Baseline**	**Agricultural expansion**
Present value of benefits per acre	Upper	3,020	3,310
	Lower	950	1,070
Costs per acre		1,720	1,720
Return on investment	Upper	1.76	1.92
	Lower	0.55	0.62
Years to pay back investment	Upper	71	64
	Lower	--	--

Appendix

1. Land use / land cover (LULC) maps

We create ten maps of land use / land cover (LULC) in Minnesota at the grid cell level (cell size = 30 x 30 m). Two maps have the LULC pattern of the 1992 NLCD outside of the acquisitions, and the LULC on the acquisitions of one 1992 map has only native natural cover (with acquisitions). Four maps have a predicted LULC pattern for 2022 based on the projections of the econometric model, two maps for the baseline scenario (with and without acquisitions) and two maps for the agricultural expansion scenario (with and without acquisitions). There are also four maps for 2052 based on the projections of the econometric model, two maps for the baseline scenario (with and without acquisitions) and two maps for the agricultural expansion scenario (with and without acquisitions). The econometric model for the projection of the LULC change is described in the main text.

On each LULC map each grid cell is assigned a one-digit classification (1 to 8) of LULC. The NLCD classes are grouped into forest (4) (NLCD classes 41, 42, 43), cropland (2) (NLCD classes 61, 82, 83, 84), pasture (3) (NLCD class 81), grassland/shrub (5) (NLCD classes 51, 71), and urban (1) (NLCD classes 21-23, 85). Existing protected areas and urban land, along with the water (6), barren (7), and wetland (8) classifications (NLCD classes 11, 12, 31-33, 91, 92) in the 1992 NLCD do not transition. The definitions of the LULC categories (Anderson land cover classification system) are given in Table A-1.

Table A-1. LULC class definitions from the definitions of the grouped classes of the NLCD 1992 used in the maps for Minnesota (from http://www.mrlc.gov/nlcd92_leg.php).

Code	Class\ Value	Descriptions
1 (NLCD classes 21-23, 85)	Urban	Includes developed open spaces with a mixture of some constructed materials, but mostly vegetation in the form of lawn grasses such as large-lot single-family housing units, parks, golf courses, and vegetation planted in developed settings for recreation, erosion control, or aesthetic purposes. Also included are lands of low, medium, and high intensity with a mixture of constructed materials and vegetation, such as single-family housing units, multifamily housing units, and areas of retail, commercial, and industrial uses.
2 (NLCD classes 61, 82, 83, 84)	Cropland	Includes cultivated crops – Cultivated crops are described as 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. This class also includes all actively tilled land.
3	Pasture/Hay	Pasture/Hay is described as grasses, legumes, or grass-legume mixtures planted for livestock grazing or the production of seed or hay crops,

Code	Class\ Value	Descriptions
(NLCD class 81)		typically on a perennial cycle.
4 (NLCD classes 41, 42, 43)	Forest	All areas characterized by tree cover (natural or semi-natural woody vegetation, generally greater than 6 meters tall); tree canopy accounts for 25% to 100% of the cover.
5 (NLCD classes 51, 71)	Grassland/Shrub	Areas dominated by upland grasses and forbs. In rare cases, herbaceous cover is less than 25%, but exceeds the combined cover of the woody species present. These areas are not subject to intensive management, but they are often utilized for grazing. Also, areas characterized by natural or semi-natural woody vegetation with aerial stems, generally less than 6 meters tall, with individuals or clumps not touching to interlocking. Both evergreen and deciduous species of true shrubs, young trees, and trees or shrubs that are small or stunted because of environmental conditions are included.
6 (NLCD classes 11, 12)	Water	All areas of open water, generally with less than 25% vegetation or soil cover, and all areas characterized by year-long surface cover of ice and/or snow.
7 (NLCD classes 31-33)	Barren	Areas of bedrock, pavement, scarps, talus, slides, glacial debris, strip mines, gravel pits, and other accumulations of earthen material. Generally, vegetation accounts for less than 15% of total cover.
8 (NLCD classes 91, 92)	Wetlands	Includes woody wetlands and herbaceous wetlands – Areas where forest or shrub land vegetation accounts for greater than 20 percent of vegetative cover and the soil or substrate is periodically saturated with or covered with water. This class also includes areas where perennial herbaceous vegetation accounts for greater than 80 percent of vegetative cover and the soil or substrate is periodically saturated with or covered with water.

1A. LULC on the acquisitions

A summary of LULC on the acquisitions for the maps with acquisitions and the maps without acquisitions in 1992, 2022, and 2052 for the baseline and agricultural expansion scenarios is summarized in Table A-2. Area is measured in acres.

Table A-2: LULC on the acquisition for the maps with acquisitions, and the maps without acquisitions in 1992, 2022, and 2052 for the baseline and agricultural expansion scenarios.

Land use/land cover	With acquisitions	Without Acquisitions				
		1992	Baseline		Agricultural expansion	
			2022	2052	2022	2052
Urban	0	1,518	2,629	3,543	2,629	3,543
Cropland	0	23,960	19,180	19,801	28,733	30,793
Pasture	0	15,370	16,362	13,468	13,230	11,817
Forest	56,102	45,085	46,795	46,896	43,748	43,371
Shrub/Grassland	38,231	8,400	9,365	10,623	5,992	4,808
Water	2,605	2,605	2,605	2,605	2,605	2,605
Barren	46	46	46	46	46	46
Wetland	26,569	26,569	26,569	26,569	26,569	26,569
Total Acres	123,552	123,552	123,552	123,552	123,552	123,552

The LULC on the acquisition lands for the 'with acquisiton' maps do not have urban, cropland, or pasture by construction of the maps. The 1992 map without acquisitons includes developed and natural land, and the developed land is mostly in agriculture and pasture, and much less urban while the natural land is mostly forest and wetland with some grassland. Comparsion of the 'with acquisition' LULC and the 1992 map without acquisitons indicates the restored developed land becomes mostly grassland/shrub and a third becomes forest sinc e the developed acquisition land mostly agriculture and pasture built over historically grassland and shrub.

Th e LULC on the 'without acquistion' maps in 2022 and 2052 have alternate trends depending on the baseline and agricultural expansion scenarios. Urban expands equivaltently in both of the LULC change scenarios, but the grassland/shrub and forest land cover steadily rises in the baseline scenario but steadily fall in the agricultural expansion scenario. The rise of natural cover in the baseline scenario or the fall in the agricultural expansion scenario is low in comparison to the increase in the natural cover from the restoration of the developed land. Thus, the difference in the provision of ecosystem services from the acquisiton of these lands by comparing the two LULC change scenarios is unlikely to be large.

2. Carbon storage and sequestration

We assume land-use grid cells, including those in working forests, have attained their LULC biomass and SOC storage steady-state levels or equilibrium as of 1992. Per acre equilibrium levels for all non-working forest LULC types and their sources are listed in Tables A-3 and A-4.

Table A-3. Metric tons of stored soil organic carbon (SOC) per acre within the first meter of the soil profile by LULC type

LULC	**Mean SOC (SD)**	**N of estimates**	**Notes**	**Source**
Wetland – prairie pothole	50.10 (18.25)	3	Equilibrium achieved at 75 years.	Slobodian et al. 2002, Bedard-Haughn et al. 2006, Euliss et al. 2006
Wetland – peatland	530.15	1	Equilibrium achieved at 2000 years.	Gorham 1991
Grassland	39.98 (16.23)	12	Equilibrium achieved at 50 years.	*Frank et al. 1995, Zan et al. 2001, Frank et al. 2002, Coleman et al. 2004, Al-Kaisi et al. 2005, Liebig et al. 2005, McLauchlan et al. 2006, Omonode et al. 2007*
Agriculture	29.18 (8.58)	41	Equilibrium achieved at 20 years. Corn and soybean rotation using conventional agricultural practices and average fertilizer applications.	Bauer et al. 1987, Hansen and Strong 1993, Frank et al. 1995, Biondini et al. 1998, Schuman et al. 1999, Yang and Wander 1999, Yang and Kay 2001, Halvorson et al. 2002, Paul et al. 2003, DeGryze et al. 2004, Al-Kaisi et al. 2005, Liebig et al. 2005, Puget and Lal 2005, Russell et al. 2005, Euliss et al. 2006, Venterea et al. 2006, Gál et al. 2007, Kucharik 2007, Morris et al. 2007, Omonode et al. 2007, Franzluebbers et al. 2009
Urban	33.47	1	Equilibrium achieved at 50 years.	Fissore et al. *in press*
Forest - unmanaged	155.6	6	Assumed all unmanaged forests 95 years old.	Smith et al. 2006 based on afforestation tables for six forest types in northern lakes region.

LULC	Mean SOC (SD)	N of estimates	Notes	Source
Forest - managed	157.0	6	Assumed even-age harvest rotation for forests: 50 years for Aspen-Birch and 75 years for Elm-Ash-Cottonwood, Maple-Beech-Birch, Oak-Hickory, Spruce-Balsam Fir, and White-Red-Jack Pine.	Smith et al. 2006 based on reforestation tables for for six forest types northern lakes region.

Note: Different types of wetlands have different carbon storage potential. In the northern part of the state wetlands are typically peatlands with very high carbon storage in their soils (Gorham 1991). Based on a state map of peatlands from the Minnesota Department of Natural Resources, we assumed that wetlands in Aitkin, Beltrami, Carlton, Cass, Itasca, Koochiching, Lake, Roseau, and St. Louis Counties were peatlands. Wetlands in all other counties were assumed to be regular wetlands or prairie potholes, which have a lower SOC storage value.

Table A-4. Metric tons of stored biomass carbon per hectare by LULC type

LULC	Biomass Mg ha^{-1} Mean (SD)	N of estimates	Notes	Source
Wetland – prairie pothole	n/a	n/a		
Wetland – peatland	n/a	n/a		
Grassland	4.09 (0.77)	10	Equilibrium achieved at 50 years. Belowground biomass is the only source of biomass carbon considered.	Risser et al. 1981, Bransby et al. 1998, Oesterheld et al. 1999, Zan et al. 2001, Baer et al. 2002, Tilman et al. 2006, Nelson et al. 2009
Agriculture	1.94 (0.93)	6	Equilibrium achieved at 20 years. Belowground biomass is the only source of biomass carbon considered. Pastures are continuously grazed at 2 head per hectare. Hayfields assumed to be 50% of natural grassland.	Schuman et al. 1999, IPCC 2006

LULC	Biomass Mg ha^{-1} Mean (SD)	N of estimates	Notes	Source
Urban	7.00	1	Equilibrium achieved at 50 years.	Fissore et al *in press*
Forest - unmanaged	159.0	6	Assumed all forests ~ 95 years old.	Smith et al. 2006 based on afforestation tables for for six forest types in the northern lakes region.
Forest - unmanaged	73.3	6	Assumed even-age harvest rotation for forests: 50 years for Aspen-Birch and 75 years for Elm-Ash-Cottonwood, Maple-Beech-Birch, Oak-Hickory, Spruce-Balsam Fir, and White-Red-Jack Pine.	Smith et al. 2006 based on reforestation tables for for six forest types in the northern lakes region.

3. Biodiversity Conservation Model: Habitat Extent and Quality

For each species group we assign a habitat suitability score to each LULC type ranging from 0 to 1, with non-habitat scored as 0 and the most suitable habitat scored as 1, with marginal habitat scored in between. 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). For this study we scored habitat differently based on its level of state and federal protection. We used the Minnesota Department of Natural Resources GAP data on stewardship for the state: code 1 and 2 are publicly protected lands, code 3 is land under an easement, and code 4 private lands (MN DNR 2000). We assume the habitat quality potential of a LULC increases with the level of protection. See Table A-5 for information on habitat suitability scores of LULC types for general terrestrial biodiversity.

Table A-5. Sensitivity to degradation sources and habitat suitability weights each LULC type for General Terrestrial Biodiversity. Higher numbers indicate more sensitivity or more suitable habitat

LULC	Agriculture area	Urban area	Primary roads	Secondary roads	Light roads	Habitat Suitability
Open water	0.00	0.00	0.00	0.00	0.00	0.00
Urban	0.00	0.00	0.00	0.00	0.00	0.00

LULC	Agriculture area	Urban area	Primary roads	Secondary roads	Light roads	Habitat Suitability
Barren	0.00	0.00	0.00	0.00	0.00	0.00
Forest	0.70	0.80	0.80	0.60	0.40	1.00
Grassland	0.60	0.70	0.70	0.50	0.40	1.00
Agriculture	0.00	0.50	0.50	0.40	0.40	0.20
Wetland	0.60	0.80	0.80	0.60	0.40	1.00

Second, we evaluate the impact of threats, which can degrade and reduce habitat quality in a grid cell either directly (e.g., habitat loss) or indirectly (e.g., edge effects from habitat fragmentation). Designated threats for this study include urban and agricultural areas, and primary, secondary, and tertiary or light roads. Urban and agriculture areas were quantified directly from the scenario LULC map while roads were evaluated using a statewide road layer (MN DOT 2009). The impact of threats is mediated by three factors.

The first factor we determine is the relative impact of each threat on a habitat grid cell. Because some threats are more damaging for all habitats, we assign a relative impact score to all threats (see Table A-6). A threat's weight, w_r, indicates the relative negative impact of a threat. For example, if urban grid cell has a weight of 1 and road cell a weight of 0.5 then the urban area causes twice the degradation, all else equal.

Table A-6. Weights and effective distances for degradation sources used in the habitat quality model

Degradation source	Maximum effective distance of degradation source (km)	Weight
Agriculture area	4.0	0.8
Urban area	5.0	1.0
Primary roads	3.0	0.8
Secondary roads	2.0	0.7
Light roads	1.0	0.5

We assign a threat-mitigating factor represented as the distance between the grid cell and the threat and the impact of the threat across space. If a grid cell is within the assigned impact distance of a particular threat then the grid cell is within the threat's degradation zone. In general, the severity of a threat on habitat quality decreases as distance from the habitat grid cell to the threat increases, so that grid cells that are proximate to a threat will experience higher degradation or lower habitat quality. We use an exponential distance-decay rate to describe how a threat's impact diminishes over space. For example, if the maximum distance of a threat is set at 1 km, the impact of the threat will decline by ~ 50% when a habitat pixel is 200 m from the defined threat. The impact of threat r_y on habitat in grid cell x, given by i_{rxy}, is normalized by the maximum effective distance of threat r, $d_{r\max}$, and is represented by the following equation,

$$i_{rxy} = \exp\left(-\left(\frac{2.99}{d_{r\max}}\right)d_{xy}\right)$$

where, d_{xy} is the distance between grid cell x and the source of threat r, grid cell y.

We determine the relative sensitivity of a habitat type in a grid cell to all threats and is the final input used to generate the total degradation level a grid cell. Let $S_{jr} \in [0,1]$ indicate the sensitivity of habitat type j to degradation source r where values closer to 1 indicate greater sensitivity to a threat. For example, a forest habitat patch may suffer more degradation from an adjacent pasture (more sensitive) than a grassland habitat patch (lower sensitivity). The model assumes the more sensitive a habitat type is to a threat, the more degradation to that habitat will be caused by that degradation source. A habitat's sensitivity to threats is based on general principles from landscape ecology (e.g., Lindenmayer et al. 2008).

Therefore, the total threat level in grid cell x with LULC or habitat type j is given by D_{xj},

$$D_{xj} = \sum_{r=1}^{R}\sum_{y=1}^{Y} w_r r_y i_{rxy} \beta_x S_{jr}$$

where, y indexes all grid cells on the landscape (including x). If $S_{jr} = 0$ then D_{xj} is not a function of threat r.

We calculate the quality of habitat in parcel x of LULC j by Q_{xj} where,

$$Q_{xj} = H_j\left(100 - D_{xj}\right)$$

Therefore, when $Q_{xj} = 100$ the quality of habitat in grid cell x is at its maximum.

We give a habitat quality landscape score for each scenario, which is an aggregate of all grid cell-level habitat quality scores on the landscape under each scenario.

4. Forestry Returns

Estimated returns to forestry development on the landscape were modeled using data from Lubowski (2002) and Lubowski et al. (2006, 2008; see Table 20). Lubowski (2002) and Lubowski et al. (2006, 2008) found average per acre county-level net returns to commercial forestry for 1992. We multiplied county *i*'s 1992 per acre net forestry returns by a scenario's acres of forest on state forest land and then summed across the two values to determine county *i*'s net forestry returns for that scenario.

Table A-7: Average per acre net returns to managed forestry from Lubowski (2002) and Lubowski et al. (2006, 2008) (all values are expressed in 1992 dollars; 1992 = 100).

County FIPS Code	1992 Managed Forestry	County FIPS Code	1992 Managed Forestry	County FIPS Code	1992 Managed Forestry
27001	-$0.31	27031	$0.45	27061	$0.46
27003	$2.36	27033	$0.83	27063	$0.83
27005	$0.89	27035	$1.37	27065	$0.17
27007	$0.79	27037	$1.88	27067	$0.83
27009	$4.71	27039	$2.36	27069	-$0.59
27011	$0.83	27041	$0.83	27071	$0.06
27013	$2.36	27043	$0.33	27073	$0.83
27015	$2.36	27045	$2.03	27075	$0.09
27017	-$0.27	27047	$2.36	27077	$0.66
27019	$0.83	27049	$1.41	27079	$0.83
27021	$1.08	27051	$2.36	27081	$0.83
27023	$0.83	27053	$2.36	27083	$0.83
27025	$1.13	27055	$2.11	27085	$0.83
27027	$1.76	27057	$2.95	27087	-$0.03
27029	$0.36	27059	$2.15	27089	-$0.56

County FIPS Code	1992 Managed Forestry	County FIPS Code	1992 Managed Forestry	County FIPS Code	1992 Managed Forestry
27091	$2.36	27133	$0.83		
27093	$0.83	27135	$0.67		
27095	$0.18	27137	$0.58		
27097	$1.06	27139	$0.80		
27099	$2.36	27141	$2.92		
27101	$0.83	27143	$2.36		
27103	$0.83	27145	$2.16		
27105	$0.83	27147	$2.36		
27107	-$0.41	27149	$0.83		
27109	$2.19	27151	$0.83		
27111	$0.83	27153	$1.02		
27113	-$0.04	27155	$0.83		
27115	$0.10	27157	$2.01		
27117	$0.83	27159	$5.00		
27119	$0.21	27161	$2.36		
27121	$2.36	27163	$0.83		
27123	$0.83	27165	$0.83		
27125	-$0.67	27167	$0.83		
27127	$2.36	27169	$2.32		
27129	$2.36	27171	$1.22		
27131	$2.36	27173	$2.36		

5. Water Quality and Yield Models

The following model descriptions are adapted from Tallis et al. (2010). For each scenario we determined water yield and total phosphorous loadings for the Minnesota 8-digit watershed. First, we model water yield, which approximates the absolute annual water yield across the basin, and is calculated as the difference between precipitation and actual evapotranspiration on each grid cell. We used maps of 30-year mean annual precipitation and reference evapotranspiration (adapted from data provided by the Minnesota State Climatology Office), soil depth and plant available water content (USDA-NRCS 2009), as well as data on the coefficients of rooting depth (Schenk and Jackson, 2002) and evapotranspiration (adapted from Allen et al. 1998) for each LULC type (See Table A-7).

The water yield model is based on the Budyko curve, developed by Zhang et al. (2001), and annual average precipitation. We determine annual water yield (Y_{jx}) for each grid cell on the landscape (indexed by $x = 1,2,\ldots,X$) as follows:

$$Y_{jx} = \left(1 - \frac{AET_{xj}}{P_x}\right) \cdot P_x$$

where, AET_{xj} is the annual actual evapotranspiration on grid cell x with LULC j and P_x is the average annual precipitation on grid cell x. The evapotranspiration partition of the water balance, $\frac{AET_{xj}}{P_x}$, is an approximation of the Budyko curve (Zhang et al. 2001).

$$\frac{AET_{xj}}{P_x} = \frac{1 + \omega_x R_{xj}}{1 + \omega_x R_{xj} + \frac{1}{R_{xj}}}$$

where, R_{xj} is the Budyko Dryness index on a grid cell x with LULC j, which is the ratio of potential evapotranspiration to precipitation (Budyko 1974). ω_x is an annualized ratio of plant accessible water storage to expected precipitation.

$$\omega_x = Z \frac{AWC_x}{P_x}$$

where, AWC_x is the volumetric plant available water content measured in mm and is estimated as the difference between field capacity and wilting point. AWC_x is defined by soil texture and effective soil depth, which establishes the amount of water capacity in the soil that is available for use by a plant. Z is the Zhang constant that presents the seasonal rainfall distribution. Finally, with R_{xj} is calculated by the following,

$$R_{xj} = \frac{k_{xj} \cdot ETo_x}{P_x}$$

where, ETo_x is the reference evapotranspiration on grid cell *x* and k_{xj} is the plant evapotranspiration coefficient associated with the LULC *j* on pixel *x*. ETo_x represents an index of climatic demand while k_{xj} is largely determined by a grid cell's vegetative characteristics (Allen et al. 1998).

Second, we determine the quantity of phosphorous retained by each grid cell in the watershed using information on nutrient loadings based on export coefficients and filtering characteristics of each LULC (see Table 21; Reckhow et al. 1980), the water yield output noted above, and a Digital Elevation Model (EROS Center 1996). Adjusted Loading Value for grid cell *x*, ALV_x, is calculated by the following equation:

$$ALV_x = HSS_x \cdot pol_x$$

where, pol_x is the export coefficient at grid cell *x* and HSS_x is the Hydrologic Sensitivity Score for grid cell *x* and is calculated as:

$$HSS_x = \frac{\lambda_x}{\overline{\lambda_W}}$$

where, $\overline{\lambda_W}$ is the mean runoff index for the basin, and λ_x is the runoff index for grid cell *x* and is calculated by the following:

$$\lambda_x = Log\left(\sum_U Y_u\right)$$

where, $\sum_U Y_u$ is the sum water yield of all grid cells along the water flow path above and including grid cell x.

Once we determine ALV_x, we then estimate how much of the load is retained by each grid cell downstream of a neighboring cell, as surface runoff moves phosphorous across the landscape and towards the mouth of the watershed. Using a GIS, we model the route of surface water down flow paths as determined by the slope of a grid cell. Each grid cell downstream is allowed to retain phosphorous based on its land-use type. Finally, the model aggregates the phosphorous loading that reaches the stream from each grid cell to determine the total loading for the entire watershed.

Table A-8. Estimates for nutrient loading, evapotranspiration, rooting depth, available water capacity, and vegetation filtering.

LULC	Evapotranspiration	Rooting	Nitrogen	Nitrogen	Phosphorous	Phosphorous

		depth	loading	efficiency	loading	efficiency
Open Water	542	1	1	0	1	0
Urban	1100	1	9970	0	1910	0
Barren	50	1	1430	5	118	5
Forest	1056	2000	2860	60	236	60
Grassland	812	1500	8650	50	1050	50
Agriculture	972	1000	16090	40	4460	40
Pasture	892	1500	12370	50	2755	50
Wetland	983	800	1	80	1	80

Source: Reckhow et al 1980, Allen et al. 1998, Schenk and Jackson 2002.

6. Recreation visitation and value

The purpose of the visitor use estimating models for wildlife viewing, hunting, and fishing is to relate MNDNR acquisition visitation to acquisition acres, natural features of the acquisitions (e.g., lakes and rivers), population and income in the surrounding area. We adapt the visitor use estimating models from Loomis and Richardson (2008) to determine the visitation to the public lands acquired by MNDNR. We relate the model from Loomis and Richardson (2008) to Minnesota acquisitions by accounting for the presence of surrounding public land and excluding the explanatory variable for the presence of an ocean.

6A. Statistical results of the visitor use estimating models

The following model descriptions are adapted from Loomis and Richardson (2008). The model coefficient estimates come from a model of visits to National Wildlife Refuges with data from Caudill and Henderson (2004). Loomis and Richardson (2008) determine refuge acres and natural features from brochure and planning documents about the refuges. Per capita income is for the counties surrounding the refuge. County population is based on the population of all counties within a 60 mile radius surrounding the particular refuge. Explanatory variables statistically significant at the 10% level or higher are retained for predicting non-consumptive (wildlife viewing) visits, total hunting visits, and fishing visits to the refuges.

The coefficient estimates for the model of non-consumptive visits is shown in Table A-9. Per capita income of the area economy surrounding the protected area, total acreage of the protected area, and the county population surrounding the protected area have a positive effect on non-consumptive visits and is significant at the 10% level. Due to the double log functional form, the per capita income, total acres, and county population coefficients can be interpreted as the percent change in non-consumptive visits. A 1% change in per capita income causes a 1.46% change in non-consumptive visits. A 1% change in total acres causes a 0.46% change in non-consumptive visits. A 1% change in county population causes a 0.26% change in non-consumptive visits. As total acres and county population increase, non-consumptive visits increase at a decreasing rate (diminishing marginal effect).

Table A-9. Coefficient estimates for the model of non-consumptive (wildlife viewing) visits

Variable	Coefficient	Std. Error	t-Statistic	Prob.
Constant	-12.11	8.32	-1.45	0.14
Ln Per Capita Income	1.45	0.80	1.81	0.07
Ln Total Acres	0.46	0.12	3.69	0.00
Ln County Population	0.25	0.14	1.74	0.08

Number of observations: 87. Adjusted R-squared is 0.21.

The coefficient estimates for the model of total hunting visits to the protected areas is shown in Table A-10. The presence of a lake has a positive effect on total hunting visits to the protected area and is significant at the 10% level. Total acreage of the protected area has a positive effect on total hunting visits and is significant at the 5% level. Due to the double log functional form, the total acre coefficient can also be interpreted as the percent change in total hunting visits. A 1% change in total acres causes a 0.3% change in total hunting visits. As total acres increase, total hunting visits increase at a decreasing rate (diminishing marginal effect).

Table A-10. Coefficient estimates for the model of total hunting visits

Variable	Coefficient	Std. Error	t-Statistic	Prob.
Constant	3.96	1.47	2.69	0.00
Lake	0.94	0.51	1.82	0.07
Ln Total Acres	0.30	0.14	2.12	0.03

Number of observations: 73. Adjusted R-squared is 0.08.

The coefficient estimates for the model of total freshwater fishing visits to the protected areas is shown in Table A-11. The total acreage of the protected area and the county population surrounding the protected area has a positive effect on fishing visits and is significant at the 5% level. The per capita income of the area economy surrounding the protected area has a negative effect on fishing visits and is significant at the 1% level. Due to the double log functional form, the per capita income, total acres, and county population coefficients can be interpreted as the percent change in fishing visits. A 1% change in total acres causes a 0.49% change in fishing visits. A 1% change in county population causes a 0.65% change in fishing visits. As total acres and county population increase, fishing visits increase at a decreasing rate (diminishing marginal effect). A 1% change in per capita income causes a 4% change in freshwater fishing visits.

Table A-11. Coefficient estimates for the model of total freshwater fishing visits

Variable	Coefficient	Std. Error	t-Statistic	Prob.
Constant	35.75	15.74	2.27	0.02
Ln Total Acres	0.49	0.21	2.24	0.02
Ln Per Capita Income	-4.04	1.50	-2.67	0.00
Ln County Population	0.65	0.23	2.81	0.00

Number of observations: 62. Adjusted R-squared is 0.23.

A limitation of these visitor use estimating models is that there is no explanatory variable that acknowledges the dependence of the number of visiting days to a protected area on the existing amount of publically available land. A public area attracts more visitors if there are a limited number of existing protected areas. The acreage of pre-existing public land is obtained from the protected areas database (PAD 2009). The amount of publicly available land within 60 miles of the acquisition is added to the total acreage of the acquisition, and this acreage is applied to the coefficient on total acreage in the each of the models to calculate the increase in visits. The low and high-end estimates of visits from each of the models are based on the amount of publicly available land within 60 miles that is a viable alternative to the acquisitions.

6A. Values per trip day for wildlife viewing, total hunting, and freshwater fishing

Values of fishing, hunting and viewing days come from the recent U.S. Forest Service database and publication by Loomis (2005). The completeness of the database for fishing studies is checked by comparing it to the Boyle et al. (1998) Sport Fishing Database believed to have the most complete coverage of fishing valuation studies. Rosenberger provided a listing of very recent studies up to and including January 2007 that had not been entered into the Loomis (2005) database. Studies in the database have the most updated values per hunter day, angler day and viewer day tables by geographic region. In addition, all the database studies were disaggregated into three types of fishing (cold, warm, anadromous - i.e., steelhead and salmon), three types of hunting (big game, small game and waterfowl), and two types of viewing (general wildlife viewing and bird viewing). Table A-12 indicates the average values per day for hunting, fishing, and wildlife viewing.

Table A-12. Average values per day for hunting, fishing, and wildlife viewing.

Species category	Average value per day for the Northeast	Number of estimates	Number of studies
Hunting			
Big game	58.45	142	
Small game	32.40	11	21
Waterfowl	35.99	39	
All game	42.28	192	
Fishing			
Cold water	39.54	58	14
Wildlife viewing	46.48	88	9

Values are reported in 2010$.

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