Analysis

Harvesting benefits from habitat restoration: Influence of landscape position on economic benefits to pheasant hunters

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A B S T R A C T

The provision of ecosystem services via ecological restoration can be affected by the spatially explicit relationships between existing landscape characteristics and the proposed restoration. In such situations, effective economic targeting of restoration is aided by an accounting of the spatially explicit linkages between initial restoration actions, the resulting changes in ecosystem services, and the changes in economic benefits to individuals resulting from changes in the provision of ecosystem services. To this end, we examine the impact of landscape heterogeneity on economic benefits of USDA Conservation Reserve Program (CRP) restoration for ring-necked pheasant (Phasianus colchicus) hunters in Michigan by linking a previously developed ecological production function of ring-necked pheasant sightings to a recreation demand model of hunter site choice. Using proposed pheasant habitat restoration in Michigan as a framework for our analysis, we find that economic benefits generated by restoration depend critically upon the landscape selected for CRP restoration. Poorly targeted restoration sites yield near-zero economic benefits, while well-targeted investments yield about 2.4 times the economic benefits of the median parcel. The results show how managers can use both ecological and hunter behavior information to enhance the return on conservation investments.

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1. Introduction

The precarious state of the world’s ecosystems has been well documented, with the Millennium Ecosystem Assessment (2005) characterizing 15 out of 24 ecosystem services as either in decline or being used unsustainably. Substantial efforts focused on halting and reversing these declines have emerged, as evidenced by the tremendous growth of the ecological restoration and conservation biology research disciplines (Young, 2000; Suding, 2011) and the substantial financial resources dedicated to restoration and conservation efforts. According to Bernhardt et al. (2005), between 14 and 15 billion dollars was spent on over 37,000 river/stream restoration projects in the continental U.S. from 1990 to 2003. The United States Department of Agriculture (USDA) Conservation Reserve Program (CRP) has provided farmers and landowners with rental payments totaling approximately $43 billion between the years 1987 and 2014 (USDA, 2014). Given the high cost of restoration and conservation efforts and the limited budgets available to achieve objectives, it is critical to develop a better understanding of whether and to what extent restoration and conservation efforts impact human well-being. Establishing and accounting for key linkages between basic biophysical structures and processes of the natural environment, the production of ecosystem functions and services, and finally, benefits to humans can allow researchers to characterize and quantify human benefits associated with costly habitat management actions (Haines-Young and Potschin, 2010; de Groot et al., 2002). Characterizing the change in human well-being in monetary terms facilitates a cost–benefit analysis of potential restoration projects, and estimating the economic values of ecosystem services has been identified as a critical research area (Millennium Ecosystem Assessment, 2005; Carpenter et al., 2006, 2009).

Ecological restoration actions have the potential to generate economic benefits to outdoor recreationists such as recreational hunters and anglers through improved catch or harvest rates at hunting or fishing locations. Several research efforts have involved estimating the economic benefits of changes in water quality for recreational anglers by establishing and accounting for the linkages between water quality, angler catch rate, and economic benefits. Lipton and Hicks (1999, 2003) and Massey et al. (2006) estimate the economic benefits resulting from changes in water quality to recreational anglers in the Mid-Atlantic region by linking expected catch models incorporating dissolved oxygen to recreation demand models of angling site choice. Another area of research involves estimating the economic benefits of water temperature mitigation strategies by linking water temperature,
fish population/expected catch, and angler economic benefits. Huang et al. (2012) estimate the net economic benefits to recreational anglers from altered water releases and diversions designed to mitigate stream temperatures, and Johnson and Adams (1988) use a contingent valuation survey in conjunction with a fish production model to estimate the economic benefits to recreational anglers associated with increased streamflow.

The effect of restoration siting decisions and land use change on ecosystem service provision is receiving increasing attention from researchers with respect to a variety of ecosystems, including wetlands (see, e.g., White and Fennessy, 2005; Lesta, 2009; Moreno-Mateos and Comín, 2010), marine reserves (Leslie et al., 2003), alpine environments (Fontana et al., 2013) and floodplain/riparian environments (Oetter et al., 2004; Johnson et al., 2007). Researchers are also beginning to explore how spatial variability across landscapes affects the economic values of ecological outputs (Barbier, 2012; Barbier et al., 2008; Sanchirico and Mumby, 2009), and recreation demand modelers have begun incorporating landscape metrics such as patch shape complexity, path variation, and total edge directly in recreation demand models (see Bestard and Font, 2009). However, there is a lack of research linking such spatially-explicit information regarding the provision of ecosystem services to outdoor recreationist (e.g., hunter, angler) behavior in order to evaluate economic benefits across a range of ecological restoration. This is surprising, given that the increasingly sophisticated spatially-explicit ecological production functions which use landscape metrics to predict species abundance and richness measures for small mammals (Gibson et al., 2004), large mammals (Maier et al., 2005), avian species (Radford et al., 2005; Nielson et al., 2008), and fish and aquatic invertebrates (Pess et al., 2002; Arbuckle and Downing, 2002) could be adapted to generate region-specific species abundance estimates for use in discrete choice recreation demand models of popular outdoor recreation activities such as away-from-home wildlife viewing (22.5 million participants), recreational fishing (33.1 million participants), and recreational hunting (13.7 million participants) (USDOI, 2011). The discrete choice recreation demand approach is well-suited for incorporating spatially-explicit outputs generated by an ecological production function as site quality attributes entering into the individual’s recreation demand decision, thereby allowing a tracing of economic benefits back to the original restoration actions. Accounting for these linkages could help improve targeting of restoration efforts of existing large-scale conservation programs such as the USDA CRP (Claassen et al., 2008) by providing resource managers with a greater understanding of the spatial distribution of landscape characteristics that would generate the largest conservation “bang for the buck”.

In this paper, we estimate the economic benefits of CRP land to pheasant hunters in Michigan by linking a spatially-explicit model of pheasant sightings as a function of landscape characteristics developed by Nielson et al. (2008) (henceforth referred to as the Nielson Model) to a recreation demand model of pheasant hunting in Michigan. Further, we focus special attention on the extent to which hunter economic benefits are contingent upon the site-selection for CRP conservation actions. Agricultural landscapes have the potential to provide a wide variety of valuable ecosystem services (Swinton et al., 2007; Knoche and Lupi, 2007), and provision of ecosystem services on such landscapes can be aided by targeted CRP conservation efforts on these lands. Researchers have found that the CRP generates benefits for sought-after game species, including white-tailed deer (Gould and Jenkins, 1993) but especially for avian species such as waterfowl (Kantrud, 1993; Reynolds et al., 1994, 2001), sage grouse (Schroeder and Vander Haegen, 2011), northern bobwhite (Riffell et al., 2008) and ring-necked pheasant (Riley, 1995; King and Savidge, 1995; Haroldson et al., 2006, and Nielson et al., 2008). However, despite the cost and documented relationship of CRP land to species populations sought by hunters and wildlife-watchers, efforts to estimate economic benefits to outdoor recreationists have been sparse. One exception is Hansen et al. (1999), who use a recreation demand model to estimate that the economic value of CRP land to pheasant hunters is about $80 million annually. Hansen et al. (1999) use a reduced form approach in which landscape characteristics presumed to influence pheasant population enter directly into the hunter site selection problem. While reduced form approaches are often employed by economists when information is lacking regarding the relationship between the biophysical structure/processes of an ecosystem and ecosystem service outputs (see, e.g., Hicks, 2004), such approaches do not fully characterize the complex nature of the ecological production process in which ecosystem services are generated through the underlying biophysical structure/processes. Furthermore, Newbold and Massey (2010) show that reduced form models can produce biased estimates of willingness to pay and recommend caution when using such models for ecosystem valuation. The availability of the ecological production function via the Nielson model overcomes the limitations associated with the reduced form approach and permits the construction of region-specific pheasant sightings which are a function of the amount and spatial distribution of landscape characteristics such as agricultural land and CRP. This enables us to examine the extent to which changes in hunter economic benefits are contingent upon the selection of a particular geographic area for CRP-related restoration.

2. Methodology

2.1. Ecological Production Function

Eq. (1) illustrates the ecological production function referred to as the Nielson Model, which predicts pheasant sightings on a North American Breeding Bird Survey route \( j \) as a function of a time trend \( \gamma r \), vector of landscape characteristics \( (ch_j) \), a vector of landscape size and distribution metrics \( (m_l) \) within a 1000 m buffer of BBS route \( j \).

\[
Pheasant~Sightings_j = \exp\left[\alpha_0 + \alpha_\gamma \gamma r + \alpha_{ch} ch_j + \alpha_m m_l\right]. \tag{1}
\]

The Nielson Model uses a Bayesian hierarchical modeling approach, fitting the model using Markov-chain Monte Carlo methods to model BBS counts of pheasants as overdispersed Poisson counts, with deviance information criterion (DIC) used as a guide to identify the most parsimonious model. For further details on developing and estimating the Nielson Model, see the Nielson et al. (2008) paper; details relating to our adaptation and replication of the Nielson model for use as a site quality variable in a recreation demand model are presented in Section 3.1.

2.2. Discrete Choice Recreation Demand Model

The discrete choice recreation demand model approach is well-suited for incorporating spatially-explicit outputs generated by an ecological production function as site quality inputs into the individual’s recreation demand decision, thereby allowing a tracing of economic benefits back to the original restoration actions. The behavioral foundation for the discrete choice recreation demand model is random utility theory (McFadden 1974), which posits that the indirect utility individual \( n \) receives from visiting recreation site \( f (v_{nj}) \) is a function of travel costs \( (tc_{nj}) \) which varies by respondent as well as by recreation site, a vector of site quality characteristics which vary by recreation site, and \( e_{nj} \), a random error term capturing unmeasured site characteristics (see Eq. (2)).

\[
v_{nj} = \beta_h t_r c_{nj} + \beta_q q_j + \epsilon_{nj}. \tag{2}
\]

Under the assumption that the error term is iid Generalized Extreme Value, nested logit site choice probabilities are derived. The choice probabilities serve as the expected demand functions for a recreation site in our 2-level nested logit model, described in Eq. (3) as the probability an individual chooses site \( k \) in nest \( l \) from the \( j \) possible sites, where \( \theta_{nm} \) are
distributional parameters to be estimated (Haab and McConnell, 2002). This nested logit approach allows sites within a nest to have error terms that are more correlated with other sites within the nest than with sites in a different nest. Parameters are estimated by maximum likelihood estimation.

$$\text{Prob}(k, l) = \frac{\exp \left( \frac{v_{l}}{\theta} \right) \left( \sum_{j=1}^{l} \exp \left( \frac{v_{j}}{\theta} \right) \right)^{\theta - 1}}{\sum_{m=1}^{L} \left( \sum_{j=1}^{l} \exp \left( \frac{v_{m}}{\theta} \right) \right)^{\theta}}$$

(3)

Given the estimated parameters, Eq. (4) denotes the change in per-trip economic benefits resulting from a quality and/or access change at one or multiple sites; \( v \) are the indirect utilities for sites in the original state and \( (v') \) are the indirect utilities for sites after quality and/or access changes (Haab and McConnell, 2002). This is a “per-trip” measure in the sense that it’s units refer to trips to all sites in the choice set. An estimate of the seasonal economic benefits to recreationists from a quality change is obtained by multiplying WTP by the total number of trips individuals made to the recreational sites within the choice set within the season of interest.

$$\text{WTP} = \frac{1}{\beta_{tc}} \ln \left( \frac{\sum_{m=1}^{L} \left( \sum_{j=1}^{l} \exp \left( \frac{v_{m}}{\theta} \right) \right)^{\theta}}{\sum_{m=1}^{L} \left( \sum_{j=1}^{l} \exp \left( \frac{v_{l}}{\theta} \right) \right)^{\theta}} \right)$$

(4)

3. Survey and Attribute Data

Information on pheasant hunter site choice and visitation frequency was obtained from the 2008–2009 Michigan Upland Game Harvest Report mail survey (MDNR, 2009) administered by the Michigan Department of Natural Resources (MDNR; \( n = 9987; 57\% \) response rate).\(^1\) For our analysis, we included only individuals (\( n = 682 \)) who reported: a) hunting pheasant during the 2008 Michigan pheasant hunting season, b) the county (or counties) in which they hunted pheasant, and c) the number of days spent hunting in each county. Consistent with the spatial resolution of the survey data, hunting sites are defined at the county level. This survey provides site choice information for 682 individuals who reported spending 3265 days hunting pheasant during 2008.

To obtain individual-specific estimates of travel costs for use in estimating this hunting site choice model, the reported number of days hunting needs to be converted to an estimate of trips. To accomplish this, we use a distance-based conversion which uses data for hunting activities in Michigan where both days and trips were known. Knoche (2014) estimated travel cost models for multiple hunting activities and compared welfare estimates using hunter-reported trips with estimates using a variety of plausible relationships between trips, days and distance to hunting sites. That analysis found many distance-based conversions that closely approximated those using hunter-reported trips and found one that only had an approximately 1% difference. The latter conversion (also described and applied in Knoche and Lupi (2013)) was applied to our data generating a total estimate of 3070 pheasant hunting trips for the 682 pheasant hunters in our model.

To account for multiple trips from the same individual when estimating the discrete choice model, we use the frequency weighting procedure in Stata to weight each observed site choice by the estimated number of trips an individual took to the site. Individuals who took trips to multiple sites have a choice set for each unique site visited, weighted by its respective number of visits. In effect, this frequency weighting procedure is an observation replicator, such that \( Y \) trips to county \( X \) is the same as \( Y \) observations of 1 trip to county \( X \). The clustered-robust standard errors procedure is used to account for the correlated error terms across different site choices by the same individual.

The vast majority of pheasant hunting in Michigan takes place in Michigan’s Lower Peninsula. Of the 2008–2009 Upland Game Harvest Survey respondents hunting pheasant during the 2008–2009 Michigan pheasant hunting season (\( n = 871 \)), only 45 (5.2%) lived in the Upper Peninsula. Further, only four of the 45 individuals living in the Upper Peninsula reported hunting a county in the Lower Peninsula (8.9%) and only five of the 826 individuals living in the Lower Peninsula reported hunting a county in the Upper Peninsula (0.6%). Thus, it appears likely that most hunters do not consider counties outside their peninsula of residence when evaluating potential pheasant hunting site options. Finally, the majority of the Upper Peninsula is closed to pheasant hunting – 10 out of 15 counties in the Upper Peninsula have no legal pheasant hunting season, with only one county in the Upper Peninsula having all of its land open to pheasant hunting. Given both the limited pheasant hunting in Michigan’s Upper Peninsula and limited travel between the Upper and Lower Peninsulas for the purpose of pheasant hunting, we restrict our choice model to hunters and counties in Michigan’s Lower Peninsula. Because there are also likely to be regional differences within Michigan’s Lower Peninsula, with similarities between hunting sites within regions and differences between hunting sites across regions, we define two nests in our nested logit model of pheasant hunting. The first nest, which we call the No_EXTENDED_Seaon nest, consists of counties in the Northern Lower Peninsula and counties located in the western portion of the Southern Lower Peninsula which do not have sizeable portions with the extended December pheasant hunting season (MDNR, 2008). The other nest, called the Extended_Seaon nest, contains the other counties in Michigan’s Lower Peninsula. See Fig. 2 for the delineation of nest membership.

County attributes used as independent variables in our nested logit model include the travel cost (time cost + vehicle operating cost) associated with traveling to and from a county, the amount of publicly accessible hunting land in the county, the size of the county, and the pheasant sightings estimate from the Nielson Model summed across all landscape parcels in the county. We computed the variable Price to reflect the round-trip travel costs from each hunter’s residence to each of the 68 counties in Michigan’s Lower Peninsula. Price per mile operating costs were calculated using 15,000 miles driven annually as a reference point, with 10,000 and 20,000 miles driven resulting in increased depreciation and increased depreciation of about 3.5 cents per mile and 3.8 cents per mile (AAA, 2008), respectively. We add the midpoint of these two estimates (3.65 cents) to per-mile depreciation costs of 17.0 cents to obtain a per-mile vehicle operating cost for the distance traveled. Per-mile vehicle operating costs, which includes average per-mile costs for gas, maintenance and tires, was 17.0 cents in 2008 (AAA, 2008). Per-mile depreciation rates were calculated using 15,000 miles driven annually as a reference point, with 10,000 and 20,000 miles driven resulting in decreased depreciation and increased depreciation of about 3.5 cents per mile and 3.8 cents per mile (AAA, 2008), respectively. We add the midpoint of these two estimates (3.65 cents) to per-mile operating costs of 17.0 cents to obtain a per-mile vehicle operating cost of 20.65 cents. The other component of travel costs is time cost, which is the opportunity cost of the time required for round-trip travel to the site. To estimate an opportunity cost of time for each individual, we used the individual’s address, in conjunction with ARCGIS software, to assign each individual the United States Census Bureau median household income for their census tract of residence.\(^2\) For

\(^1\) Even though the survey achieved a reasonably high response rate, we do not have access to the data on nonrespondents and a nonresponse study was not conducted.

\(^2\) When lacking survey data on individual income, recreation demand modelers often assign individuals a proxy for income via aggregate measures which characterize income at a geographic scale, such as median household income for census areas or zip codes (see, e.g., Heberling and Templeton (2009), Whitehead et al. (2009), Henderson et al. (1999), and Jakus et al. (1997)). This approach is based on the tendency of individuals to cluster geographically by socioeconomic status. If membership in the sub-group of interest (in this case, pheasant hunters) is correlated with higher (or lower) median incomes compared to others in their census tract, then welfare estimates could be biased downwards (upwards).
individuals whose addresses were not recoverable using ARCGIS software (4.3% of the population) we assigned the individual the median household income for their zip code of residency. To value the time component of travel cost, we follow the many studies in the recreation literature by using one-third of the wage rate (Parsons, 2003). Wage rate was computed by dividing median income by the number of work hours per year (2080). To estimate travel time, we assumed an average trip speed of approximately 64 km (40 miles) per hour.

The variable Public_Access consists of federally owned land, state-owned land, and privately-owned land that is publicly accessible to hunters through Michigan’s Commercial Forest Act program. The variable Size reflects the geographic area of each county. In models using aggregate sites such as the 68 counties in our choice set, many studies include a measure of group size to mitigate potential differences in site heterogeneity (Haener et al., 2004). However, unlike some recreation such as fishing at lakes (Lupi and Feather, 1998), pheasant hunting takes place at myriad places across the landscape, and therefore does not have easily defined access points. Never the less, we control for differences in the amount of potential hunting locations across counties by including variables for publicly accessible land and geographic size of a county.

3.1. Computing Pheasant Sightings via the Nielson Model

To construct the Pheasant_Sightings variable, we apply the Nielson Model ecological production function which predicts pheasant sightings obtained from the BBS as a function of landscape characteristics obtained from the National Land Cover Dataset (NLCD) and the USDA. To develop this model, Nielson et al. (2008) estimated pheasant sightings associated with landscape characteristics of 3888 Breeding Bird Survey (BBS) routes. Eq. (5) is Nielson et al.’s (2008) study area-wide model (which we refer to as the Nielson Model), which estimates the number of pheasants counted along a survey route in year $i$.

$$T_i = \exp[1.5451 - 0.0059(year_i - 1996) + 0.2748(\text{NLCD Woody Vegetation}) + 0.7040(\text{NLCD Herbsceous Vegetation}) + 1.4949(\text{NLCD Agricultural Field}) - 0.6584(\text{NLCD Agricultural Field})^2 - 0.1901(\text{CRP Herbsceous Vegetation}) - 0.0526(\text{Mean Patch Size}) - 0.1702(\text{Interspersion and Juxtaposition Index})]$$

The Nielson Model was originally fit using landscape characteristic data from the 1992 NLCD and USDA CRP data, and landscape metrics Mean Patch Size (MPS) and Interspersion and Juxtaposition Index (JI). Four NLCD landscape characteristic categories are included in the model: NLCD Woody Vegetation (includes NLCD classifications of shrubland and orchards/gyardens), NLCD Herbaceous Vegetation (grassland/herbaceous and pasture/hay), NLCD Agricultural Field (row crops, small grains, and fallow) and CRP Herbaceous Vegetation (14 types of herbaceous CRP enrollment classifications). Variables in Eq. (5) were calculated using a 1000 m buffer around the BBS route. Assuming a straight BBS route, this corresponds to an area of 19,487 acres. Nielson et al. (2008) also estimated the model using buffers of 400 m and 700 m, however, 1000 m was used for the final model as this approach yielded the lowest deviance information criterion and the fewest variables.

In replicating the Nielson Model to estimate site-specific pheasant sightings estimates in Michigan, we closely follow the approach described in Nielson et al. (2008) while taking advantage of recently available landscape characteristic data. We use 2006 NLCD and Michigan USDA CRP geolocation data and employ an ARCGIS fishnet procedure to construct a grid consisting of 19,487 acre landscape parcels which overlays the Michigan’s Lower Peninsula (see Fig. 1). For each of these parcels, landscape characteristic data and landscape metrics MPS and JLI (computed using FRAGSTATS v4 [McGarigal et al., 2012]) were used in Eq. (5) to produce a pheasant sightings estimate for that parcel. Pheasant sightings estimates for each parcel were used to construct total pheasant sightings for each county. For landscape parcels which overlap two or more counties, the pheasant sightings estimate for that landscape parcel was proportionally allocated to each county.

In Fig. 2 we compare the county-level pheasant sightings predicted by the Nielson Model to county-level pheasant brood observations through MDNR mail carrier surveys. The MDNR mail carrier survey is an annual survey conducted during the summer months in which cooperating rural mail carriers within 43 southern Michigan counties record the number of pheasant broods observed during the survey period. The 43 counties were selected for the mail carrier survey based on the potential of habitat in that county to support wild pheasant populations — 41 of these 43 counties were identified in the 2013 National Wild Pheasant Conservation Plan (Association of Fish and Wildlife Agencies, 2013) as having wild pheasant range. Further, MDNR has found the mail carrier data to be a good predictor of pheasant abundance and fall harvest (Luuikkonen, 1998). MDNR mail carrier survey estimates in Fig. 2 reflect the total brood observations in each county for the previous ten surveys (2004–2013). To facilitate comparison of MDNR mail carrier survey brood observations and the pheasant sightings estimates from the Nielson Model, we only include Nielson Model estimates for the counties surveyed via the MDNR mail carrier survey. We segment county-level estimates for the MDNR mail carrier survey and the Nielson Model into quartiles to further aid comparison.

Fig. 2 illustrates similarities between the Nielson Model pheasant sightings estimates and pheasant sightings estimated via the MDNR mail carrier surveys. We use the Pearson correlation coefficient to characterize the relationship between the Nielson Model sightings predictions and MDNR mail carrier brood observations. The estimated Pearson correlation coefficient of $+0.62$ indicates a positive correlation between MDNR mail carrier pheasant brood observations over a ten year period and our county-level pheasant sightings derived from application of the Nielson Model.

4. Results

The nested logit model results, estimated using Stata 13 software (StataCorp, 2013) are displayed in Table 1. Price is statistically significant and negative, meaning that as the cost of traveling to a hunting site increases, the indirect utility associated with that site (and the probability of selecting that hunting site) decreases. As the number of pheasant sightings and the amount of publicly accessible hunting land increases at a hunting site, the indirect utility associated with that hunting site increases. Dissimilarity parameters represent the interdependence of unobserved indirect utility in each nest; a dissimilarity parameter of one indicates no correlation in unobserved indirect utility between sites within a given nest. In our model, Extended_Season and No_Extended_Season are statistically significant and different than one, meaning that counties within each respective nest are similar in unobserved utility to other sites in that nest.

4.1. Pheasant Hunter Benefits of CRP Land

Economic benefits of CRP land to Michigan pheasant hunters are estimated by examining how hunter welfare changes when all

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3 Access to otherwise protected USDA CRP data was granted through a USDA Section 1619 Cooperator Memorandum of Understanding issued by the Farm Service Agency. This agreement prohibits the disclosure of the geospatial USDA CRP data, and as such, we do not map Michigan-level USDA CRP data in our figures or otherwise provide information on USDA CRP location within this paper.
213,674 acres of CRP land is converted to the 2006 NLCD cultivated crops landscape characteristic. First, we estimate the per-trip economic benefits of CRP land for pheasant hunters in Michigan (Eq. (4)). We find that the conversion of all CRP land to NLCD cultivated crops results in a per-trip welfare loss of about $1.82. To obtain an estimate of the reduction of seasonal economic welfare from this conversion, we multiply this per-trip welfare loss by the estimated number of trips taken in the season for all pheasant hunters represented in our model. The estimate of the total number of trips (183,081) is obtained by multiplying the total number of pheasant hunters (43,144) by the percentage of these hunters both living in the Lower Peninsula and not hunting in the Upper Peninsula (94.3%) and by the estimated number of hunting trips per person (4.5). Thus, the total seasonal welfare loss to hunters from this conversion is estimated to be about $332,600, which yields a mean per acre CRP value for pheasant hunters of $1.56/acre.


The MDNR has partnered with the U.S. Fish and Wildlife Service, Michigan Department of Agriculture, and non-governmental organizations such as Pheasants Forever and Michigan United Conservation Clubs to implement the Michigan Pheasant Restoration Initiative (MPRI). The stated objective of the MPRI is to establish 10 cooperatives (each roughly 10,000 acres in size) in Michigan throughout 3 pilot focus areas (see Fig. 3) and restore 1200 acres of pheasant habitat in each of these cooperatives. To ensure that our policy analysis is consistent with the Nielson Model, we assume that each cooperative is 19,487 acres. We closely replicate the MPRI objective in our analysis such as Pheasants Forever and Michigan United Conservation Clubs. The Nielson et al. (2008) model includes orchards/vineyards in the NLCD woody vegetation category. We felt the benefits from using 2006 NLCD data vastly outweighed this minor discrepancy in how orchards/vineyards were categorized within the 1992 and 2006 NLCD datasets.

We select the midpoint of the MPRI cooperative level restoration goal and assume that 1600 acres of herbaceous CRP land is restored within each cooperative, for a total restoration of 14,400 herbaceous CRP acres across the nine counties. For each landscape parcel which is completely within the borders of an MPRI pilot focus area county, we generate a new pheasant sightings estimate for that landscape by converting 1600 acres of cultivated crops within that landscape to herbaceous CRP land. We use these new pheasant sightings estimates to estimate the economic benefits associated with multiple policy scenarios in which different landscape parcels are selected for location of the MPRI cooperatives. To estimate the economic benefits to pheasant hunters associated with achieving the MPRI restoration goal, we assume that the high-quality pheasant habitat restored via the MPRI will have the same characteristics as land enrolled under the USDA herbaceous CRP category. These policy scenarios provide important insights regarding the extent to which economic benefits accruing to hunters vary depending on ecological restoration siting decisions.

For our previous analysis regarding the seasonal economic benefits of CRP land to pheasant hunters in Michigan, information on the size and location of CRP patches allows us to account for changes in landscape metrics MPS and IJI resulting from the conversion of these patches to cultivated crops. However, such precision with MPS and IJI is not possible in our MPRI analysis due to uncertainties regarding the size and location of the cultivated crop patches converted to herbaceous CRP land. For insights into how this conversion might impact MPS (and hence impact pheasant sightings measures and hunter welfare), we use three assumptions on patch size of the cultivated crops land area converted to CRP. First, we estimate the change in hunter welfare under the assumption that the 1600 acres of restored CRP land in each landscape parcel has a mean patch size equal to the mean patch size of all herbaceous CRP patches throughout Michigan (6.3 acres), and that these patches are not directly adjacent to other CRP patches. This results in 254 patches of herbaceous CRP added to the landscape parcel. Second, given the potential of the cultivated crop conversion to occur on land currently adjacent to herbaceous CRP land, we estimate hunter welfare assuming that 50% of the herbaceous CRP patches are placed adjacent to existing patches. This results in an addition of 127 patches to the landscape. Finally, we explore the welfare implications of the restoration if all restored herbaceous CRP is adjacent to existing herbaceous CRP, thus creating zero new patches and resulting in no change in MPS in that landscape parcel.

It is more challenging, however, to develop plausible scenarios for how the conversion of cultivated crops to herbaceous CRP land would

Fig. 1. Michigan Lower Peninsula counties and landscape parcels used to estimate pheasant sightings.
affect the IJI variable. As IJI depends on the relative proportion of patch adjacencies, the direction and the extent that IJI would be impacted as a result of the conversion of cultivated crops is ambiguous. Welfare estimates in Table 2 can thus be taken to be representative of an “average” landscape parcel in which IJI does not change with the conversion of the cultivated crops to herbaceous CRP. Additional examination revealed that a +/− one standard deviation change in IJI results in a change to hunter welfare (relative to the results in Table 2) of approximately +/− 20%.

In Table 2, three assumptions on MPS and the assumption of constant IJI are used to examine how economic benefits accruing to pheasant hunters from the MPRI vary across three different restoration siting scenarios. We computed the change in pheasant sightings in each landscape (across the nine MPRI pilot focus areas) which resulted from the conversion of 1600 acres of agricultural land to herbaceous CRP, and identified the landscape with the minimum, median, and maximum pheasant sightings increases in each county. In the first scenario, “Minimum”, we assume that for each of the nine counties, the conversion occurs within the landscape parcel in each county that realizes the smallest increase in pheasant sightings as a result of the conversion. Essentially, the policy-relevant analogy is that the MPRI managers selected the worst possible landscape parcel to locate the cooperative in each of the nine counties. In the second scenario, “Median” we assume that the conversion occurs within the landscape parcel in each county that realizes the median increase in pheasant sightings in that county, and in the third scenario, “Maximum” we assume that conversion occurs within the landscape parcel in each county that realizes the largest increase in pheasant sightings in that county. In this “Maximum” scenario, the policy analogy is that MPRI managers have the information and flexibility necessary to locate the additional 1600 acres of CRP land in the landscape parcel that provides the greatest increase in pheasant sightings. The seasonal welfare per acre in Table 2 is the seasonal welfare estimate in Table 2 divided by the total number of acres of agricultural land converted to herbaceous CRP (14,400 acres) in the MPRI restoration siting scenario.

Table 2 illustrates the critical role restoration siting decisions play in generating economic benefits to pheasant hunters. Choosing the least productive landscape in each county for restoration of 1600 acres of herbaceous CRP essentially results in zero economic benefits to hunters from the 14,400 acres of restored land. For a different perspective, interpret the “Median” scenario as being a situation in which a manager, facing no constraints on restoration site selection but also possessing no information on the benefits of restoration on different landscapes, selects the landscape with the median increase in economic benefits from restoration in each county. Relative to this scenario, a manager able to identify and select for restoration landscapes with the largest increase in economic benefits (the “Maximum” scenario) in each county are able to achieve an increase in economic benefits to hunters of about 140%.

The above analysis demonstrates the importance of restoration site selection in generating economic benefits. To further illustrate, Fig. 4 shows the economic benefits for each landscape parcel within the primary and secondary MPRI restoration areas in Michigan’s Southern Lower Peninsula, using the “Median” method as described above. For each parcel we converted 1600 acres of the agricultural land within that parcel to CRP herbaceous land. We assumed that 50% of the converted patches are adjacent to existing CRP patches and 50% are new patches, and we held all other landscape characteristics within and across parcels constant. Restoration parcel hot spots which produce

### Table 1

<table>
<thead>
<tr>
<th>Nested logit model of pheasant hunting in Michigan.</th>
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<tbody>
<tr>
<td>Independent variables</td>
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<tr>
<td>Price</td>
</tr>
<tr>
<td>Pheasant_Sighting</td>
</tr>
<tr>
<td>Public_Access (10,000 acres)</td>
</tr>
<tr>
<td>Size (100 sq. miles)</td>
</tr>
<tr>
<td>Dissimilarity parameters</td>
</tr>
<tr>
<td>Extended_Season</td>
</tr>
<tr>
<td>No Extended_Season</td>
</tr>
<tr>
<td>Model statistics</td>
</tr>
<tr>
<td>Log-likelihood</td>
</tr>
<tr>
<td>Wald chi² (4)</td>
</tr>
<tr>
<td>Observations</td>
</tr>
</tbody>
</table>

*a* P values for the dissimilarity coefficients are estimated against the null hypothesis that the coefficient is equal to one.
economic benefits are scaled using shades of red — the darker the color, the greater the economic benefit of restoring that parcel. Parcels which lacked sufficient agricultural land for our restoration scenario (less than 1600 acres) are indicated in gray. Some parcels, shaded in blue, had small negative economic benefit estimates (mean for those parcels = −$126; median = −$108) due to the non-linearities and spatial juxtaposition of agricultural lands within the pheasant sightings equation (Eq. (5)).

5. Discussion

The objective of this paper is to explore how restoration site selection affects economic benefits for a user group whose outdoor recreational experience is directly affected by ecosystem services generated through ecological restoration efforts. Examining the linkages between CRP restoration, pheasant abundance, and hunter economic benefits, we find that benefits depend dramatically on restoration site selection. An important factor affecting the levels of economic benefits across different possible restoration sites is that the ecosystem service output desired by pheasant hunters (i.e., increases in pheasant) is affected by a highly non-linear ecological production function. In particular, diminishing returns to agricultural land results in the economic benefits from CRP restoration (via the conversion of agricultural land to CRP land) being highly sensitive to the amount of agricultural land within the landscape selected for restoration. In the extreme, the worst restoration sites generate effectively zero economic benefits. In contrast, an informed resource manager can select restoration sites in a manner that increases the returns on economic benefits by about 140% relative to a baseline scenario in which the manager possesses no spatially-explicit information. Such non-linear relationships between landscapes and human well-being have also been identified in Kopmann and Rehdanz (2013). Additionally, this result conforms with the finding of Newbold and Massey (2010), who showed that poorly targeted environmental improvements can lead to welfare decreases. These analyses suggest that managers should be particularly concerned with landscape heterogeneity and spatial dependencies when the landscape characteristic targeted for restoration provides positive but diminishing returns within the ecological production function.

The parcel-level economic benefits estimates illustrated in Fig. 4 produce key information for governmental and non-governmental organizations seeking to maximize return on restoration dollars when a key objective is to increase benefits for pheasant hunters. Restoration hotspots as indicated in Fig. 4 align closely with the nine MPRI pilot focus area counties (see Fig. 3) targeted for initial restoration efforts, suggesting a well-targeted restoration focus. Even within these nine counties there is evidence of hotspots which provide the most economic benefits. Moreover, if restoration activities could target conversion of agricultural lands to habitat types and locations that are even more beneficial to pheasants than generic CRP herbaceous lands, then the economic benefits would be even greater.

Table 2

<table>
<thead>
<tr>
<th>Restoration siting scenarios</th>
<th>% change in pheasant sightings in restoration area</th>
<th>Seasonal welfare (median as baseline)</th>
<th>Seasonal welfare per acre</th>
<th>Welfare difference</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>No patch increase</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Minimum</td>
<td>−2.9%</td>
<td>−$900</td>
<td>−$0.06</td>
<td>−107%</td>
</tr>
<tr>
<td>Median</td>
<td>+24.9%</td>
<td>$12,700</td>
<td>$0.88</td>
<td>Baseline</td>
</tr>
<tr>
<td>Maximum</td>
<td>+52.9%</td>
<td>$30,100</td>
<td>$2.09</td>
<td>+137%</td>
</tr>
<tr>
<td><strong>50% new patches</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Minimum</td>
<td>−1.5%</td>
<td>−$100</td>
<td>−$0.03</td>
<td>−103%</td>
</tr>
<tr>
<td>Median</td>
<td>27.4%</td>
<td>$13,900</td>
<td>$0.97</td>
<td>Baseline</td>
</tr>
<tr>
<td>Maximum</td>
<td>63.0%</td>
<td>$34,200</td>
<td>$2.38</td>
<td>+145%</td>
</tr>
<tr>
<td><strong>All new patches</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Minimum</td>
<td>−0.4%</td>
<td>−$100</td>
<td>−$0.01</td>
<td>−101%</td>
</tr>
<tr>
<td>Median</td>
<td>33.0%</td>
<td>$14,900</td>
<td>$1.04</td>
<td>Baseline</td>
</tr>
<tr>
<td>Maximum</td>
<td>69.7%</td>
<td>$37,000</td>
<td>$2.57</td>
<td>+148%</td>
</tr>
</tbody>
</table>
While site selection for CRP restoration plays an important role in the relative amount of economic benefits accruing to pheasant hunters, both the Michigan CRP as a whole and the MPRI generate modest aggregate economic benefits to these hunters. A limiting and conservative assumption within our MPRI restoration scenario analysis is that restoration occurring through this program is through the conversion of agricultural land to generic herbaceous CRP land. In contrast, restoration actions via the MPRI program would likely focus on restoring distinct herbaceous CRP classification subcategories which are the most beneficial to pheasant (and are not represented in the sightings model), such as upland bird habitat buffers. Restoration efforts targeted at the most beneficial habitat types would result in greater increases in pheasant abundance, and thus greater economic benefits to pheasant hunters. An additional limitation of our survey data and analysis is that the total number of pheasant hunter trips is held constant despite increases in pheasant abundance - this assumption becomes less valid with large changes in pheasant populations. If such quality improvements resulted in latent pheasant hunters rejoining the active hunter population or if active pheasant hunters increased their number of hunting trips, our estimates would underestimate economic benefits. Finally, future analyses may want to consider the potential crowding effects that might occur with large spatially-concentrated improvements in habitat.

Although methodological differences complicate direct comparison, temporal and regional differences likely contribute to the difference between our per-acre economic benefits estimate ($1.56/acre) and the average per-acre estimate across the 13 state study area from Hansen et al. (1999) ($6.48/acre in 2008 dollars). For example, the number of Michigan pheasant hunters has declined by 63% over the previous two decades (Moritz, 1992; Frawley, 2012) — scaling up the aggregate economic benefits estimates within this paper under the assumption of a 1991 level of pheasant hunters yields a per-acre CRP value of about $4.17/acre/year. Finally, opportunities and preferences for different types of hunting in Michigan appear to have changed over the previous two decades, with increasing interest in and availability of deer hunting and sharply declining participation small game hunting (Frawley, 2006). This substitution away from pheasant hunting and towards deer hunting (especially archery hunting, for which the season is open throughout the Michigan pheasant hunting season) likely results in lower willingness to pay for increases in pheasant hunting quality relative to the past.

In summary, while our analysis reveals relatively modest economic benefits to pheasant hunters from CRP land, the level of economic benefits from habitat restoration efforts varies considerably depending on restoration site selection. These results illustrate how managers can leverage spatially-explicit ecological data and the site choices of outdoor recreationists to enhance the return on conservation investments.

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References


