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Analysis

# An assessment of the influence of bioenergy and marketed land amenity values on land uses in the Midwestern US

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ABSTRACT

There is substantial concern that bioenergy policies could swamp other considerations, such as environmental values, and lead to large-scale conversions of land from forest to crops. This study examines how bioenergy and marketed environmental rents for forestland potentially influence land use in the Midwestern US. We hypothesize that current land uses reflect market values for environmental benefits of forestland, so that the marketed component of the environmental value of land can be captured as the difference between Census land values and value of land as a timber asset. We use a multinomial logit model to estimate the land use shares of forests, crops and urban in Ohio, Indiana, and Illinois. The results show that marketed environmental rents increase forestland relative to cropland. To examine the effects of biofuels on land use, we conduct policy analysis by altering future land rents. Our baseline scenario projects that urban development uses mostly cropland, but with higher crop rents resulting from increased demand for bioenergy, there will be significant losses of forestland to urban and cropland. On the other hand, if marketed environmental rents grow while crop rents are maintained at their baseline value, urban growth will occur, primarily at the expense of cropland.



## 1. Introduction

Two critical drivers are exerting important influences on land use in the United States: growth in aesthetic, or environmental, values for natural land and open space, and more recently, the development of bioenergy. Many researchers have observed that demand for natural amenities and open space has driven conversion of crop and forestland to housing and other urban uses in the past several decades (e.g., Irwin and Bockstael, 2002; Wu, 2001). Nationally, for example, about one million acres of private forestland were converted to developed uses annually in the 1990s (USDA NRCS, 2001; Stein et al., 2005; US Forest Service, 2007). In addition to urbanization, growth in the demand for bioenergy may exacerbate changes in land use by spurring additional conversion of forestland to cropland (Searchinger et al., 2008; Fargione et al., 2008). Given the importance of both these drivers on future land uses, it is important to build empirical tools that can help analyze the implications on the landscape. This paper builds such a model, which we use to examine the effect these two drivers potentially have on land uses in the Midwestern United States.

The land use change model developed in this paper is based on a multinomial logit area-base model (e.g. Hardie and Parks, 1997; Plantinga et al., 1999; Sohngen and Brown, 2006)[1] that seeks to explain the proportion of land in crops, forests, and urban uses in the counties of three Midwestern States: Ohio, Indiana, and Illinois. Within this region, total forestland and urban land increased over the period 1982 to 1997 while cropland decreased (Choi, 2004). Around 67% of the land was in crops in 1997, and around 23% in forests at the same time period. While there was substantial land conversion to developed uses over the time period, total forestland also increased between 1982 and 1997, by around 1.5 million acres.

In previous applications of similar models, the authors account for rent values in cropland and forestland by calculating the value of land based on crop or forest production potentials, and the current costs of production and prices. Demand for urban land uses are often controlled by population density and distance to urban centers (Kline et al., 2000; Alig et al., 2004). These previous studies, however, have ignored some of the value that urban and suburban users have for the landscape they own. For example, forest cover could provide habitat, hiking trails, shade, fuel wood, or other benefits for land-owners in addition to the traditional values of the timber itself.

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[1] Area base model is based on the Ricardian land rent theory that land use margin depends on land quality. In addition, land owners are assumed to maximize rent by allocating land uses. See Hardie and Parks (1997) for the summary of area base model theory and discussion on multinomial logit application.

Within the literature, it has been recognized that marketed environmental rents have a large influence on management of private forests (Amacher et al., 2003), and that these values influence decisions over the amount of forestland or cropland to hold. Marketed environmental rents, which are associated with the private benefits that landowners obtain from having environmental amenities on their property, are thus likely to have substantial influences on land values, and hence, on land use choices. Indeed, Anas and Chaushie (1984) and Anas (1988) point out that wild or recreational land uses can only be maintained efficiently in private markets if they can generate competitive rents. Thus, if private recreational land is observed, then it must somehow be generating rents. A key issue is whether these rents can also be observed, and then used in empirical models.

Given that rising demand for land in urban uses will cause the value of land in markets to rise above the productive value of land in crops or forests, we assume that this marketed environmental value can be calculated as the difference between the market value of land and the productive value of the land in the best alternative. We then incorporate this additional measure of value, namely marketed environmental value, in our multinomial logit land use model. This builds on earlier approaches that have considered only the productive value of the timberland itself (e.g., Plantinga et al., 1999). Note that we only account for marketed values in this study. Although land markets are likely to have additional externalities, such as loss of open space or changes in water quality, it is unlikely that these true externalities (either social benefits or costs) influence land use change, unless they do so through specific policy actions.

In addition to testing the implications of marketed environmental rents on land use, we account for potential spatial autocorrelation (see Anselin, 1988). With county level data, such as used in this study, it is possible that un-observed policy influences or interactions among neighboring counties could be omitted. If these omitted variables are correlated with the error term, then the statistical results could be incorrect. Nelson and Hellerstein (1997) handled this problem by sampling from a large population. This study has a smaller sample, so we utilize the full sample and develop a spatial autocorrelation model.

The estimated model is then used to project future land uses based on projections of the underlying rents. Because marketed environmental rents are included separately in our model, we can make separate projections for the future value of productive crop and forest land, and the future value of the marketed environmental component. Growth in demand for bioenergy is expected to influence the productive value of crop and forestland, while growth in income is expected to influence marketed environmental rents. The impacts of these different sources of rent changes could have different effects on future land uses, for example, rising environmental values may lead to an increase forestland area in the future, while rising biofuels values may lead to an increase in crop area in the future. In this study we compare and contrast future projections of land use for the three Midwestern States under alternative assumptions about the growth in environmental value or growth in crop and forestland rents resulting from growth in demand for bioenergy.

This paper begins with a description of our empirical model and dataset. Model estimates are then developed and discussed. We utilize the model to generate projections of land use changes for the region across different assumptions about changes in land rents.

## 2. Methodology

This paper assesses how inclusion of data on market values influences existing land use change models. To account for market values, we utilize data from the US Agricultural Census and assume that the values in Census data reflect market values at the time the Census is collected. In collecting the data, the US Census Bureau asks landowners to provide information on the value of their land if sold. Data from the US Agricultural Census should reflect both value of the underlying land in crop production, and any additional value associated with location. For example, when comparing land of similar quality, one would expect Census values for land nearest cities to be larger relative to the underlying production value of crop or forestry than for land further away. If there is no additional value to land, then one would expect Census values to be equal to crop or forest production values.

Rather than incorporating the Census values directly, we therefore incorporate the difference between Census values and forest production values. We call this "marketed environmental rent (ME rent)", since we expect that the additional market value of land above and beyond forest production values will reflect the local environment (including the natural environment and the built environmental associated with development). Our estimates for marketed environmental rent are then incorporated into an area-base model, following Hardie and Parks (1997), Plantinga et al. (1999) and Sohngen and Brown (2006). Although some environmental values clearly fall into the realm of non-market values, which cannot be observed by market transactions, we focus on environmental values that do influence market transactions in this paper.

The area-base model assumes that the probability of a given land use is a function of a number of different variables, including market and environmental rents, population density, soil quality, distance to city center, and per capita income among other variables (see Table 1 for variables). The econometric model estimates the share of land in counties among three different land uses ($j$ = forest, crop, and development). The shares of a particular land use are expressed as a multinomial logistic function as follows:

$$P_j = \frac{e^{\beta_j X}}{1 + \sum_{j=1}^{m-1} e^{\beta_j X}}, j = 1,,,m-1, \text{where } m = 3 \text{ in this study} \quad (1)$$

In Eq. (1), $P_j$ is the proportion of land allocation in usage $j$, $X$ is a vector of explanatory (independent) variables, and $\beta$ the vector of

**Table 1**
Definition of variables.

| Variables | Definition |
|---|---|
| Ln(F/C) | Dependent variable: log of forest proportion to crop proportion |
| Ln(F/U) | Dependent variable: log of forest proportion to urban proportion |
| Constant | Constant term |
| 1982 dummy | Dummy variable for 1982 data (1 = 1982 data; 0 otherwise) |
| 1987 dummy | Dummy variable for 1987 data (1 = 1987 data; 0 otherwise) |
| 1992 dummy | Dummy variable for 1997 data (1 = 1992 data; 0 otherwise) |
| Forest rent | Forest rent (real value in 1984) |
| Crop rent | Crop rent (real value in 1984) |
| ME rent | Marketed environmental rent (real value in 1984) |
| ME rent × density | Interaction variable (marketed environmental rent multiply with density variable) |
| Density | Total population divided by total area in each county |
| Average forest age | Average timber age |
| Average forest age squared | Square of average timber age |
| AVLCC | Average land class in each county |
| *K* factor (erosion) | Soil wind erosion factor (*k*-factor) |
| Slope length | Average degree of slope length in each county |
| Distance to city | Minimum distance from major cities to the center of each county |
| Forest stock rent | Rent for standing timber stock (real value in 1984) |
| F dummy | Dummy for counties that have more than 15% of forestland (1 = county with more than 15% of land in forests; 0 otherwise) |
| F dummy × ME rent | Interaction variable (F dummy multiply with marketed environment rent) |

estimated coefficients. Eq. (1) can be transformed into a linear form with two different land uses ($j$ and $m$) and expressed as follows:

$$\ln\left(\frac{p_j}{p_m}\right) = \beta X + u_i \quad (2)$$

We apply two different assumptions in terms of error term, $u_i$, in Eq. (2). First, it is assumed that the error term is independent and identically distributed error term (Greene, 2000). Second, it is assumed that the error term is spatially auto correlated as in Eq. (3).

$$u = \rho W u + e \quad (3)$$

The error term $u$ follows a spatial autoregressive process with weight matrix $W$, coefficient of autoregressive process $\rho$, and $e$ is an i.i.d. error term (Anselin, 1988). $W$ is an $n$-by-$n$ weight matrix (where $n$ is the number of observations) that defines spatial dependency among observations. After testing a range of alternatives, we use a distance criterion that allows 8 counties as neighbors on average (the weight matrix was obtained by SpaceStat program). The weight matrix is row-standardized so that the sums of each row in the weight matrix sum to 1.

The data used in this analysis consist of county level data obtained from US government resources such as the USDA National Resource Inventory (NRI),[2] Bureau of Census, Census of Agriculture,[3] among others, for the periods 1982, 1987, 1992, and 1997. Dummy variables are incorporated for 1982, 1987, and 1992 to account for factors that changed across the periods, but affected all uses of land similarly. Dummy variables take on the value 1 if the data are from the given year (1982, 1987, and 1992) and a 0 otherwise.

The forest rent variable is the discounted net present value of timber revenue per acre. For each major tree species, the yield functions were weighted by the proportion of the species in each county (USDA Forest Service Forest Inventory and Analysis data). The regional timber prices for Ohio and Illinois are used and state level price data are used for Indiana (OASS, 1999; IASS, 1999). Utilizing the Faustmann formula, the land rents for forestry are derived using a 5% interest rate (Johansson and Lofgren, 1985). All the rent variables are based on the prices deflated by consumer price index base period in year 1984.

Crop rents are measured as the real annual per acre net return to cropland uses. They are estimated using annual revenues above variable costs, using crop yields for the four major crops in the region (corn, wheat, soybean, and oats) from USDA data. The returns for each crop in a given county are weighted by the number of acres of the crop in the county (each period), giving the estimate for crop rents.

Average Land Capability Class is obtained from the NRI land capability classes (assessed by slope, soil texture, soil depth, effects of past erosion, permeability, water holding capacity, and type of clay minerals). The land quality variable is controlled by the average land capability class[4] (AVLCC, weighted by area) in each county.

Distance to city centers is included to capture a component of land use demand. The larger cities in each state[5] are identified and then for each county the distance from the county's center to the specified cities is calculated. It is expected that land value is inversely related to distance to city centers.

**Table 2a**
Comparison of forestland timber values and marketed environmental rents for forestland (1984 USD).

| | Forestland timber value | | | Marketed environmental value | | |
|---|---|---|---|---|---|---|
| | Ohio | Indiana | Illinois | Ohio | Indiana | Illinois |
| 1982 | \$276 | \$361 | \$151 | \$1266 | \$1244 | \$1605 |
| 1987 | \$355 | \$456 | \$174 | \$775 | \$603 | \$862 |
| 1992 | \$432 | \$430 | \$232 | \$784 | \$672 | \$849 |
| 1997 | \$493 | \$555 | \$336 | \$1052 | \$966 | \$1026 |

Population density has served as proxies for future development rents (Mendelsohn et al., 1994; Plantinga and Miller, 2001). Urban land rent values are approximated with population density variable assuming that the higher the population density the higher the development force, increasing the opportunity cost of maintaining land for other uses (Kline et al., 2000; Alig et al., 2004). From the Bureau of Census we obtained data for county area and population.

Marketed environmental rent, as noted earlier, is the additional value of land above the timber production value. County level average land values are obtained from the United States Census of Agriculture.[6] The average land value is converted to a rent form assuming an annual 5% discount rate, and forest rents are subtracted from this value to estimate marketed environmental rent.

Forest stock rent is incorporated as well to account for the rent on standing natural resources. Forest land rent values described before are bare-land values, and are relevant mainly for the afforestation decision. Forest stock values are likely more important for deforestation decisions. They are calculated by estimating the stock of different types of forest in each county (using USDA Forest Service Forest Inventory and Analysis data), and then valuing the stock with stumpage prices. Rents are calculated by multiplying the stock value to the interest rate ($r = 0.05$), and estimates are provided for the average value per hectare in each county.

Timberland values also increased over time (Tables 2a and 2b), mainly as a result of rising real prices for forest products over the same time period. Environmental values rose over the entire time period, but declined between 1982 and 1987 before rebounding in real terms. These changes resulted from the fairly substantial depression in land values that occurred nationwide as a result of the farm recession of the early 1980s.[7] This adjustment reflects substantial restructuring in land markets, where large amounts of land were available for purchase. The supply of land thus increased, reducing overall values. Although environmental rents have generally trended upward over time, they are, as expected, also influenced by broader economic patterns. The large difference between marketed environmental rents and forest rents suggests that land is substantially more valuable in this region for the non-productive outputs, i.e., the environmental value, than the timber output. Interestingly, marketed environmental value is larger in counties with less forestland, as expected, because scarcity is greatest, and it is smaller in counties with more forestland (Table 2b).

## 3. Results

### 3.1. Estimation

We estimated two different models. First, a base model that ignores the environmental values associated with forest land is

[2] The NRI is a statistical sampling of non-federal land. The data for year 2003 is only available by summary data and detailed sample point data will not be published. (http://www.nrcs.usda.gov/TECHNICAL/NRI/1997/obtain_data.html).

[3] The Census of Agriculture surveys every farm in the United States, and asks the operator how much land he or she has in different agricultural uses.

[4] Land capability classification is a system of grouping soils primarily on the basis of their capability to produce common cultivated crops and pasture plants without deteriorating over a long period. The higher the AVLCC, the lower the land quality because an LCC that falls in a higher number (>4) implies worse land quality.

[5] Ohio: Columbus, Cleveland, Cincinnati, Dayton, Akron, and Pittsburgh. Indiana: Indianapolis, Evansville, Fort Wayne, South Bend, Gary, Chicago, Cincinnati and Louisville. Illinois: Chicago, Springfield, Peoria, Rockford and St Louis.

[6] Estimated market value of land and buildings - average per acre. Nominal values were converted to constant 1984 dollars with the Consumer Price Index.

[7] During the 1970s farmland prices went up substantially and were relatively stable.

**Table 2b**
Comparison of forestland timber values and environmental values for forestland (1984 USD).

| | Forestland timber value | | Marketed environmental rent | |
|---|---|---|---|---|
| | All counties ($N = 270$) | Forest prop. >15% ($n = 130$) | All counties ($N = 270$) | Forest prop. >15% ($n = 130$) |
| 1982 | $261 | $259 | $1441 | $1138 |
| 1987 | $326 | $328 | $768 | $594 |
| 1992 | $362 | $370 | $724 | $557 |
| 1997 | $459 | $459 | $1010 | $705 |

estimated. This model can be used to assess potential influences of rent values on land use. Second, the base model is augmented with environmental values associated with forestry. Several additional variables are incorporated in the model, including the calculated environmental value, the average age of forests, and the annual rent on standing forest stock, and dummy variables for the counties that have at least 15% of their land in forests. Average age of forests controls for quality of forests and recency of forest expansion. Counties with older forest ages on average are those that have not experienced substantial increases in forestland in recent years.

All models are estimated as a seemingly unrelated system of equations (SUR) with the dependent variables being the log of the forest proportion to crop proportion and the log of the forest to urban proportion. We focus on two equations that have forestry in them to assess the influence of the parameters on forestry specifically; however, note that the parameter values are not marginal effects. Marginal effects will be calculated subsequently. Both models are estimated as SUR with spatial error correlation following Anselin (1988, chapter 10). The third equation, the log of the crop to urban proportion can be excluded and the parameter estimates can be inferred from these results. The models are estimated with MATLAB,[8] and the results are shown in Tables 3 and 4.

For the base model in the equation for log-share of forest to crop [Ln(F/C)], the coefficient on forest rent is positive while the coefficient on crop rent is negative, both as expected. Both parameters are also significant at a high degree of confidence. The ratio of forest to crop land by counties in the region appears to be fairly stable over time, as the dummy variables in this first equation are all insignificant. Density has a positive parameter estimate, suggesting that higher density increases the ratio of forests to crops. Average land capability class is positive, and given that higher values for this variable indicate lower quality land, this parameter makes sense (i.e., forests are located on lower quality land on average). The results for $K$ factor and slope length similarly make sense. Neither distance to city nor per capita income is significant in this first equation. The spatial correlation parameter, rho ($\rho$), is statistically significant at 0.01 level and positive sign.

For the equation for the log share of forest to urban land uses [Ln(F/U)] of the base model (Table 3), the dummy variables for time periods are all negative but declining over time, indicating that over time, there has been an increasing proportion of forestland relative to urban land. The parameter on the forest rent variable is positive, but insignificant, suggesting that forest rents have had little to do with conversions between urban and forests. However, the parameter on crop rents is negative, indicating that reductions in crop rents increase the forest to urban proportion. Higher population density reduces the forest to urban proportion, while larger distances from cities increase the forest to urban proportion, both of which are expected effects. The spatial correlation parameter is statistically significant at 0.01 level and shows positive sign.

[8] The code is available upon request.

**Table 3**
Typical land use model (without marketed environmental rents).

| | Ln(F/C) | | Ln(F/U) | |
|---|---|---|---|---|
| | Coeff | Std. err. | Coeff | Std. err. |
| Constant | **−5.623** | **0.277** | **2.088** | **0.410** |
| 1982 dummy | −0.016 | 0.122 | **−1.333** | **0.182** |
| 1987 dummy | 0.022 | 0.091 | **−1.027** | **0.135** |
| 1992 dummy | 0.076 | 0.065 | **−0.411** | **0.096** |
| Forest rent | **0.042** | **0.003** | 0.001 | 0.005 |
| Crop rent | **−0.006** | **0.001** | *−0.003* | *0.001* |
| Density | **0.004** | **0.000** | **−0.013** | **0.001** |
| AVLCC | **1.249** | **0.026** | **0.617** | **0.039** |
| $K$ factor (erosion) | **4.794** | **0.447** | **2.911** | **0.663** |
| Slope length | *−0.039* | *0.018* | *−0.056* | *0.026* |
| Distance to city | 0.001 | 0.001 | **0.007** | **0.001** |
| Per capita income (1000s) | −0.014 | 0.009 | **−0.130** | **0.014** |
| Spatial error coefficient $\rho$ | **0.126** | **0.015** | **0.108** | **0.017** |
| $R^2$ bar | 0.84 | | 0.67 | |
| System $R^2$ | | 0.77 | | |
| Cross equation covariance | | 0.284 | 0.157 | |
| | | 0.157 | 0.617 | |
| Wald statistic ($\rho_{F/C} = \rho_{F/U}$) | | 7.836 | $\sim\chi^2(2)$ | |

Bold = significant at 0.01; italic = significant at 0.05.

The second model incorporates marketed environmental rents and additional variables into the base model (Table 4). The parameter estimates are consistent for variables that also appear in the base model of Table 3, however, several of the parameter estimates change in size. In the equation Ln(F/C), the parameter on forest rents is positive, but smaller than in the base model of Table 3. The parameter on marketed environmental rent is positive, but insignificant. This result suggests, at least in general, that changes in marketed environmental rents have very little impact upon the proportion of forest to cropland (although one must be careful directly interpreting these rather than the marginal effects discussed later). Because of the way that marketed environmental rent is measured (the difference between Census values for land and forest timber values), it is possible that there are strong interactions with population density. We consequently interact marketed environmental rent with density. The interaction term is small, and not significant in the Ln(F/C) equation.

**Table 4**
Land use model with marketed environmental rents.

| | Ln(F/C) | | Ln(F/U) | |
|---|---|---|---|---|
| Variable | Coefficient | Std. err. | Coefficient | Std. err. |
| Constant | **−6.101** | **0.261** | 0.673 | 0.435 |
| 1982 dummy | 0.087 | 0.056 | **0.502** | **0.094** |
| 1987 dummy | *0.119* | *0.049* | **−0.221** | **0.083** |
| 1992 dummy | 0.094 | 0.048 | −0.090 | 0.081 |
| Forest rent | **0.033** | **0.004** | *−0.017* | *0.007* |
| Crop rent | **−0.003** | **0.001** | 0.000 | 0.001 |
| Marketed environmental (ME) rent | 0.000 | 0.001 | **−0.022** | **0.002** |
| ME rent × density | 0.000 | 0.000 | **0.000** | **0.000** |
| Density | **0.004** | **0.001** | **−0.024** | **0.002** |
| Average forest age | **0.026** | **0.007** | 0.014 | 0.011 |
| Average forest age squared | **0.000** | **0.000** | 0.000 | 0.000 |
| AVLCC | **1.043** | **0.028** | **0.402** | **0.047** |
| $K$ factor (erosion) | **2.156** | **0.413** | 0.177 | 0.685 |
| Slope length | −0.020 | 0.015 | −0.051 | 0.025 |
| Distance to city | **0.002** | **0.001** | **0.008** | **0.001** |
| Forest stock rent | −0.000 | 0.000 | 0.000 | 0.000 |
| F dummy | **0.583** | **0.071** | **0.352** | **0.118** |
| F dummy × ME rent | **0.003** | **0.001** | 0.003 | 0.002 |
| Spatial error coefficient $\rho$ | **0.061** | **0.008** | **0.087** | **0.019** |
| $R^2$ bar | 0.84 | | 0.68 | |
| System $R^2$ | | 0.78 | | |
| Cross equation covariance | | 0.208 | 0.117 | |
| | | 0.117 | 0.573 | |
| Wald statistic ($\rho_{F/C} = \rho_{F/U}$) | | 15.214 | $\sim\chi^2(2)$ | |

Bold = significant at 0.01; italic = significant at 0.05.

The parameter on average forest age is positive, and the squared term for average forest age has a positive parameter as well. Increasing forest age in general implies more forestland, but the effect is non linear. We suspect that this non-linearity is driven by a number of counties with fairly small proportions of forestland, but whose trees are old on average (likely because that land was never cleared in the first place). The forest stock rent parameter is negative but insignificant and the parameter is very small. High forest stock rents indicate either more productive timber species or high forest standing volumes. Harvesting is more likely to occur in areas with more valuable forest. The spatial correlation parameter, rho ($\rho$), is statistically significant at 0.01 level and shows positive sign.

In the Ln(F/U) equation, the parameter on forest rent is negative, and significant at the 5% level. This is somewhat surprising, but indicates that forest rents have little influence on decisions over urbanization. The parameter on environmental rent is also negative, suggesting that higher environmental values reduce the area of forestland. Although density is positively correlated with the ratio of forest to crop land, increasing density leads to more urbanization and fewer forests relative to urban areas. The interaction term between environmental rent and density is positive and significant (although small) in the Ln(F/U) equation, suggesting that for higher population density levels, an increase in environmental rents can increase the forestland proportion. The spatial correlation ($\rho$) in this equation also shows a positive and significant result.

When considering the two equations in Table 4, the results imply that the forest and crop ratio is influenced by marketed environmental rents. Where the difference between land sales values and timber rents are larger, the ratio of forest to crops increases. Environmental rents do not appear to be large enough to offset the influence of urbanizing pressures on forestland levels overall. However, at the margin, for higher population density, an increase in environmental rent can increase the proportion of forest to urban land (although this effect is fairly small).

### 3.2. Marginal Effects

To assess land use choices more closely, we examine marginal effect of land usages influenced by rents and population density. Marginal effects are defined in this analysis as the change in land in each use for a small change in one of the rent values. Thus, we postulate a 1% change in one of the rent values, and then calculate the effect of this change in land rent in terms of the change in land in each of the categories (forest, crop, and urban). For this analysis, we use the estimated results from the second model (Table 4), and we use the 1997 dataset. All other variables are held constant while forest rents, crop rents, marketed environmental rents, and population density are changed independently. The marginal effects are shown in Table 5. The numbers in parentheses are the differences of each land usage in each case from the 1997 data.

The model estimates in Table 4 indicate that a 1% increase in forest rents leads to an additional 53 thousand acres (0.37%) and 43 thousand acres (0.75%) in forestland and urban land, respectively, and this is offset by a 96 thousand acre (0.2%) loss in cropland. If forest rents are rising relative to other rents, the forest sector expands over cropland. However, higher forest rents also lead to more urban land. This results from the negative sign on the forest rent parameter in the LN(F/U) equation. Increases in forest rents are thus unlikely to slow down expansion of urban land as population density increases.

Raising crop rents by 1% increases cropland by 30 thousand acres. For the most part, this increase in land area is derived from forestland (21 thousand acres), while some is also derived from urban land (8 thousand acres). Although it may be unrealistic to think of urban land reversing itself and going back into crop or forest uses, this marginal effect implies that increases in crop rents may be able to offset future increases in urban area.

Increasing marketed environmental rents by 1% reduces cropland by 27 thousand acres, and increases urban land by 27 thousand acres, with the remaining acres shifting to forests. Rising environmental rents imply additional uses of cropland, but fairly small changes in forestland. With higher marketed environmental rents, forestland is largely preserved, and development occurs mostly on cropland. Changes in population density behave similarly: An additional 1% of population density increases urban land by 31 thousand acres, while reducing cropland by 28 thousand acres. Urban development via population increases has stronger implications for cropland, at least at the margin. Urban development could prefer cropland because of development cost and values from keeping nearby forest stands.

**Table 5**
Marginal effect of land uses by 1% increase in rents and population density. Total acres in study region shown for each change in rent value, with the difference from the 1997 baseline value shown in parentheses.

| | Urban | Forest | Crop |
|---|---|---|---|
| | 1000 acres | | |
| 1997 data* | 5750 | 14,376 | 48,251 |
| Forest rent +1% | 5793 (43) | 14,429 (53) | 48,157 (−96) |
| Crop rent +1% | 5742 (−8) | 14,355 (−21) | 48,281 (30) |
| Marketed environmental rent +1% | 5777 (27) | 14,377 (1) | 48,224 (−27) |
| Population density +1% | 5781 (31) | 14,374 (−2) | 48,223 (−28) |

*: Expected value from the 1997 sample data and coefficients estimates from model in Table 4.

### 3.3. Land Use Projections

The models estimated earlier can be used to project land use change by adjusting important variables, such as rents and population density. For this analysis, we project land rents for cropland and forestland through 2045. We also use US Census projections of population change over this same time period to project future population growth.[9] These projections are used to estimate potential land use proportions in 2005 and 2045.

One of the most important recent influences on land use in the United States has been the increasing use of corn-based ethanol in fuel supplies. In the future, the development of technologies to produce cellulosic ethanol could further spur land use changes. As a result of these trends, high energy prices have fueled substantial growth in ethanol production and demand for corn, as evidenced by relatively high crop prices. We consequently consider three different future scenarios: (1) baseline scenario, (2) bioenergy scenario, and (3) high market environment rent scenario (HMER scenario), as explained in more detail later.

For projecting crop and forest rents, this region is assumed to be a price taker on international markets for agricultural and forestry products. The same changes in forest and agricultural prices and rents are assumed for each county. Although pulpwood and saw timber prices have historically increased more than 3% per year in this region, future projections suggest that these prices will stabilize or decline slightly in the next 30–40 years (Haynes, 2003). We thus assume that forest rent rates and forest stock rents rise at 1% per year throughout the baseline scenario. Cropland rent rates are assumed to rise at 2% per year, following the studies by FAPRI (2010) and USDA (2010). These studies predict 2% to 4% real annual increases rate in crop rents, and we use the lower value as our baseline assumption for crop rents. Marketed environment rent is assumed to rise at 1.5% per year for baseline scenario. Other variables such as distance and soil quality are expected to be constant over the years across scenarios.

[9] County level population growth projection was obtained from each state office web page, Office of Strategic Research in Ohio Department of Development, Indiana Business Research Center, and Illinois Department of Commerce and Economic Opportunity.

**Table 6**
Land use projection (000 acres).

| | 2005 | 2045 | Changes |
|---|---|---|---|
| *Baseline scenario* | | | |
| Forest | 14,379 | 13,428 | −951 |
| Crop | 47,685 | 44,192 | −3493 |
| Urban | 6314 | 10,758 | 4444 |
| *Bioenergy scenario* | | | |
| Forest | 14,379 | 11,862 | −2517 |
| Crop | 47,685 | 46,762 | −923 |
| Urban | 6314 | 9753 | 3439 |
| *High marketed environment rent scenario* | | | |
| Forest | 14,379 | 13,171 | −1208 |
| Crop | 47,685 | 40,164 | −7521 |
| Urban | 6314 | 15,043 | 8729 |

For the bioenergy scenario, we assume that there is higher demand on bioenergy, higher demand on crop product, in turn higher demand on cropland. Recent work by Taheripour et al. (2010) and Hertel et al. (2010) calculates that recent legislation in the US would increase crop prices by 17–20% by 2022 relative to the 2022 baseline. This would translate into a 3.5% per year increase in crop rents compared to our baseline assumption of 2.0% per year. For the bioenergy scenario, we thus assume crop rents rise at 3.5% per year over the time horizon of our study. Similarly, we assume forestland rent rises at 1.7% per year following the study by Sedjo and Sohngen (2009), who showed that higher demand on bioenergy would also increase the demand on feedstock from forestland. In the bioenergy scenario, marketed environmental rent is assumed to be the same as in the Baseline. For the HMER scenario, we assume marketed environmental rent rises at 2.5%[10] while the other rents growth rate stays at the same rate as in the baseline scenario.

For the projections, we use the model in Table 4. The resulting land use projections are shown in Table 6. In the baseline, both forest and cropland are projected to decrease over the next 40 years. The flows are relatively large, suggesting that urban land could expand by 4.4 million acres, or by 67% relative to the current area. Most of the urban land is derived from cropland, although about a quarter is derived from forests. These results reverse recent trends where forestland area has increased around 0.2% per year for the past several decades (Smith et al., 2003).

For the bioenergy scenario with higher crop rents, the loss of cropland slows dramatically compared to the baseline. There is less urbanization than in the baseline, but there are still 3.4 million new acres of urban land projected by 2045. The big change is that most of this new urban land is converted from forests rather than cropland. The higher crop rents associated with bioenergy trends appear to allow the area of cropland to remain steady, but expansion in urban uses occurs at the expense of forestland.

The HMER scenario projects a much stronger increase in urban land—more than 8.7 million acres of urban development over the 40 year projection period. For the most part, cropland is lost, although 1.2 million acres of forestland are also converted to urban uses. Marketed environment rent is the gap between the Census land sale value and forest rent that includes both land value associated with location such as near high development area and environment values. As this gap gets bigger, it spurs much more urbanization. Interestingly, this represents about the same amount of land removed from forestland uses as the baseline. For the most part, new land used for urban uses is derived from cropland.

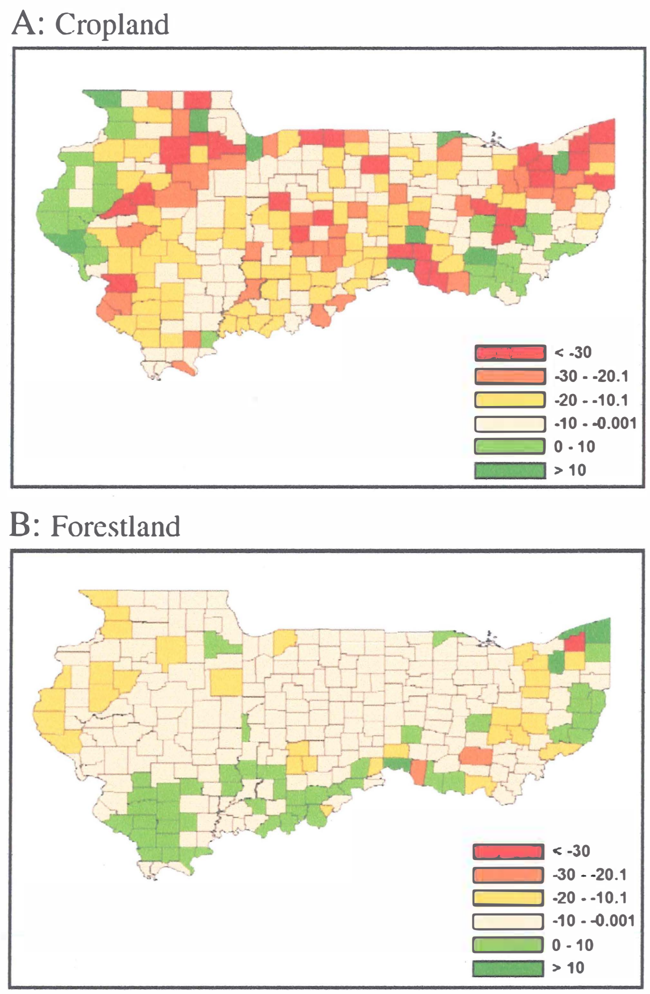


**Fig. 1.** Baseline scenario land use changes by between 2005 and 2045.

Figs. 1–3 show the projected changes in crop and forestland uses for the three scenarios geographically. In the baseline (Fig. 1), total forests in the region decline by 6.7%, or around 1.0 million acres, and total cropland declines by about 7.3%, or 3.5 million acres. One can see that this does not happen uniformly across the region. Most cropland losses are located in counties surrounding urban areas, and there are some counties with cropland gains, particularly in the southern part of Ohio and the western edge of Illinois. Forestland losses are fairly uniform across a large section of the region, but forestland gains occur in the areas that are already heavily forested, e.g. the eastern Ohio and the southern parts of the region.

The bioenergy scenario (Fig. 2) reverses much of this picture as expected. Cropland area increases in most counties, although there are cropland losses in some regions. The biggest cropland increases occur in south central Ohio and western Illinois. Interestingly, in northeastern Ohio, cropland area declines. Forestland area declines in most areas, and the losses tend to be heaviest in the regions with the strongest cropland increases. In the HMER scenario (Fig. 3), cropland losses are very large, and they occur throughout the region. There are some cropland gains in counties around urban centers and in urban counties themselves in Ohio. Forestland losses also occur over most of the region, although some regions experience gains in forestland, particularly those regions far from urban centers, where forests already dominate the landscape.

[10] The High Market Environmental Rent scenario is assumed as we do not have additional information or citations on similar studies; however, we have chosen a conservative number for the increase in value.

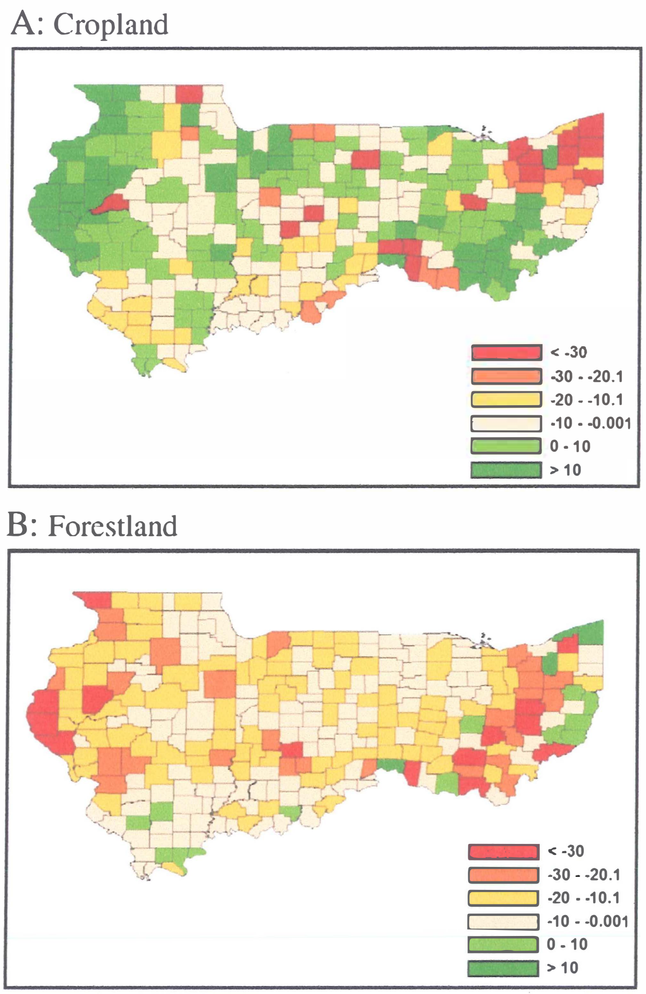


**Fig. 2.** Bioenergy scenario land use changes by between 2005 and 2045.

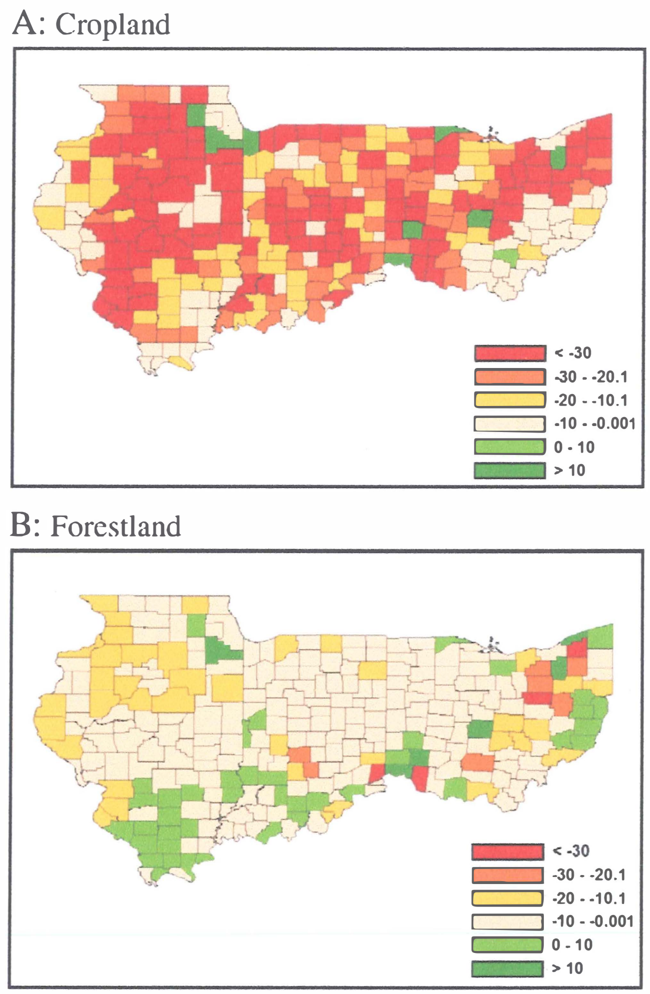


**Fig. 3.** HMER scenario land use changes by between 2005 and 2045.

## 4. Conclusion

This paper examines how marketed environmental rents and rising demand for bioenergy could potentially influence land use in three Midwestern states: Ohio, Indiana, and Illinois. The paper begins by examining the relationship between land use and marketed environmental rents for forestland in the Midwestern states of Ohio, Indiana, and Illinois. Most land use change models focus on estimating the rent value of land based on the productive capacity of the land. Timberland values are estimated as the productive value of land in timber, and cropland values are estimated as the value in crops. However, this method may miss some of the value of forestland, given that there are likely to be non-consumptive uses of forestland. For example, many landowners may hold timberland simply for the aesthetic or environmental values associated with owning forestland.

The relationship between these values is described in the context of a Hartman (1976) forestry model, where forest land values are assumed to depend on timber values and a non-market amenity related to the stock of timber. The environmental, or non-market, component is shown to be the difference between the market value of a land sale value and the timber value component. We hypothesize that we can capture the marketed component of the environmental value of land by utilizing US Agricultural Census data on land values. Marketed environmental value is the difference between US Agricultural Census land values and the value of land as a timber asset. It is important to recognize that these marketed environmental rents are privately held. They do not account for externalities associated with holding timberland.

Two versions of a logit model of land use in three Midwestern states (Ohio, Indiana, and Illinois) are estimated. First, we estimate a version similar to many existing land use models that do not account for effects of marketed environmental rents on land use decisions. Second, we estimate a version that accounts for marketed environmental rents.

Marketed environmental rents are shown to have a significant influence on the results. Higher marketed environmental rents increase the proportion of forest land relative to cropland, although they reduce the proportion of forestland relative to urban land. An interaction term between marketed environmental rents and population density suggests that as population density increases, increases in marketed environmental rents potentially lead to more forestland. Marginal effects calculated for deviations in all of the rent values as well as population density have the expected signs. They suggest that increases in marketed environmental rents will have stronger marginal impacts on cropland. That is, as marketed environmental rents increase, we expect to see stronger conversion of cropland into urban uses rather than land from forest into urban uses.

Projections of future land use changes are developed whereby all rents are changed simultaneously. Our baseline projection assumes that forest rents rise at 1%/year, crop rents rise at 2%/year, and marketed environmental rents rise at 1.5%/year. Under this scenario, around 4.3 million acres shift from crop and forest uses to urban land over a 40-year period. Most of the increase in urban land is derived from crop uses, although forestry uses decline as well. We also investigate a bioenergy scenario, whereby cropland rents rise at 3.5%/

year, timberland rents rise at 1.7%/year, and marketed environmental rents are assumed to be the same as in the baseline. This scenario presents a dramatically different future, whereby large areas of forestland are lost in the next 40 years, and urban/developed land remains fairly stable. Urban land increases, but not quite as much as in the baseline scenario. There are approximately 2.5 million acres more of land in crop in 2045 in this scenario than in the baseline scenario. The final scenario examines higher marketed environmental rents (+2.5%/year) and maintains the crop and forest rents at their baseline levels. This scenario implies strong additional flows of land into urban uses, largely at the expense of cropland. Forests are not strongly affected by this scenario relative to the baseline.

These results imply that it is important to account for the source of the increase in land rents, and that the source could have strong impacts on land use flows. If the primary reason for increasing land values in rural areas relates to developments in the bioenergy sector, then one would expect more cropland and less forestland, with little additional use of land in the urban sector. On the other hand, if there is a strong increase in land values resulting from increases in marketed environmental rents on the landscape, one would expect additional conversion of land from crop to urban land, with some flows from forestry to urban uses.

Although the results of this study suggest that capturing environmental values can be important in land use modeling, it is important to note that environmental values remain difficult to obtain. We have utilized US Agricultural Census data to approximate the value of land transactions in suburbán areas. Unfortunately, these values have some limitations. First, they also account for infrastructure developments on the landscape, such as barns and houses. Second, they account only for environmental values held by landowners themselves, and not for externalities associated with different uses of the landscape. It is unlikely that externalities strongly influence land values at this time, but it would be useful to understand the size of these externalities so as to value the impact of these land use change on the external values.

Third, US Agricultural Census data likely focuses most directly on cropland and not on forestland simply due to the sample (farmers). It would be preferable, of course, to conduct primary research and estimate forestland sales values by county directly, and to use those in the estimation procedures. It is difficult, however, to obtain consistent data across a landscape to estimate forestland values in isolation of cropland values. Hedonic methods or other survey methods may be appropriately applied to specific locations or regions, but these data are not available more broadly for the three state region. As part of this research, we did consider the use of tax valuation data. These data were available at the county level only in Ohio for the four sample periods (1982, 1987, 1992, and 1997). The other two states in the three-state region were unable to provide this information.

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