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Land-use change and carbon sinks: Econometric estimation of the carbon sequestration supply function

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Abstract

If the United States chooses to implement a greenhouse gas reduction program, it would be necessary to decide whether to include carbon sequestration policies—such as those that promote forestation and discourage deforestation—as part of the domestic portfolio of compliance activities. We investigate the cost of forest-based carbon sequestration by analyzing econometrically micro-data on revealed landowner preferences, modeling six major private land uses in a comprehensive analysis of the contiguous United States. The econometric estimates are used to simulate landowner responses to sequestration policies. We treat key commodity prices as endogenous and predict carbon storage changes with a carbon sink model. Our estimated sequestration costs exceed those from previous engineering cost analyses and sectoral optimization models. Our estimated sequestration supply function is similar to the carbon abatement supply function from energy-based analyses, suggesting that forest-based carbon sequestration merits consideration in a cost-effective portfolio of domestic US climate change strategies.

Keywords: Abatement; Carbon; Climate change; Costs; Forestry; Greenhouse gases; Land use; Land-use change; Sequestration

1. Introduction

Increased concern about the threat of global climate change has brought with it greater attention to the possibility of encouraging the growth of forests as a means of removing carbon dioxide (CO_2) from the atmosphere. The Kyoto Protocol to the Framework Convention on Climate Change [41] states that the participating nations can employ carbon sequestration as part of their portfolios of strategies to achieve their domestic CO_2 targets. Hence, if the United States chooses to implement a domestic greenhouse gas reduction program, it will be necessary to decide whether to include carbon sequestration policies—such as those that promote forestation and discourage deforestation—as part of its portfolio of compliance activities.

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The costs of carbon sequestration would presumably be one major criterion in that decision. Since the late 1980s, it has been suggested that sufficient lands are available to use the approach to mitigate significant amounts of CO_2 emissions [14,17], and claims have been made that such forestry-based carbon sequestration is a relatively inexpensive means of addressing climate change [8,36]. We investigate the cost of supplying domestic forest-based carbon sequestration using an econometric model of the revealed preferences of landowners who can use their land for alternative purposes.

1.1. Previous studies of the costs of US forest-based carbon sequestration

Three general approaches have been used to estimate the costs of sequestering carbon in the United States: bottom-up engineering cost studies; optimization models that allow for behavioral response in the forest and agricultural sectors; and econometric analyses of the revealed preferences of landowners.[1]

Most previous studies of the costs of sequestering carbon in the United States have employed bottom-up engineering cost analyses or optimization models. In ‘‘bottom-up’’ or ‘‘engineering cost’’ methods, marginal cost schedules are constructed by adopting information on revenues and costs of production of alternative land uses on representative types or locations of land, and sorting these in ascending order of cost. In the earliest and simplest of these [8,21,23], the analysts estimated available land area, forest carbon accumulation rates, and land and planting costs for hypothetical carbon sequestration programs, and thereby derived the total amount of carbon that could be captured and the cost per ton of sequestration.[2] In two other cases, the engineering approach was modified to include an anticipated increase in agricultural land prices as a hypothetical carbon sequestration program expanded, and crop and pastureland were removed from agricultural production. This was done by drawing upon previous (exogenous) estimates of the price elasticity of demand for agricultural land [32,33].

A carbon sequestration program may increase agricultural land prices, leading landowners to convert unregulated forest lands to agricultural land, thereby offsetting some of the effects of a carbon sequestration program. Alig et al. [4] addressed this issue through the use of a two-sector, multi-period simulation model, in which the forest and agricultural sectors were linked, and the welfare of producers and consumers in the two sectors was maximized.[3]

A relatively small number of studies have used a revealed-preference approach, in which actual land-use changes have been analyzed to estimate relationships between land-use choices and relative returns in the forest and agricultural sectors, thereby leading to the simulation of carbon-sequestration cost functions [24,31,39].[4] These studies examined the relationship between observable historic events (changes in timber and agricultural product prices) and landowner responses (conversion of land into and out of forest), and statistically estimated a response function. The models then posited a hypothetical economic stimulus—for example, government subsidies for carbon sequestration—and estimated how landowners would respond.

In theory, there are a number of reasons why landowners' actual behavior regarding the disposition of their lands might not be well predicted by ‘‘engineering’’ or ‘‘least-cost’’ analyses [39]: land-use changes can involve irreversible investments in the face of uncertainty [27,35], and landowners may want to retain options for future land-use decisions [29]; there may be non-pecuniary returns (for example, esthetics, and recreation) to landowners from forest uses of land [30], as well as from agricultural uses; liquidity constraints or simple ‘‘decision-making inertia’’ may mean that economic incentives affect landowners only with some delay; and

[1] For surveys of carbon sequestration cost analyses, see: Sedjo et al. [37]; Richards and Stokes [34]; Stavins and Richards [40]; and van Kooten et al. [47].

[2] In a variation on this theme, Parks and Hardie [28] substituted the estimated foregone net revenues from agricultural production for observed sale and rental prices of agricultural land.

[3] Alig et al. [4] use the Forest and Agricultural Sector Optimization Model (FASOM). Adams et al. [1] use a precursor to this model to estimate marginal costs of carbon sequestration in the US. Later applications of FASOM are in Adams et al. [2], McCarl and Schneider [18], and Murray et al. [22]. Lewandrowski et al. [15] use the US Agricultural Sector Model (USMP), to estimate carbon sequestration costs from afforesting crop and pasture lands and other agricultural-sector activities. Our literature review focuses solely on analyses in the United States.

[4] For econometric applications focused on sequestration from changes in agricultural practices, see Antle et al. [5] and Kurkalova et al. [13].

there may be private, market benefits or costs of alternative land uses of which an analyst is unaware. The econometric cost analyses have sought to address some of these problems.

1.2. Distinguishing features of this analysis

Compared with previous econometric analyses of the costs of carbon sequestration, this study is distinguished by three principal features. First, rather than considering only the movement of land between forestry and agriculture, six major land uses are modeled: forest, crop, pasture, range, urban, and a Federally designated private use, the Conservation Reserve Program (CRP).[5] Accounting for future urbanization is particularly important if it reduces the land base available for conversion to forest.

Second, detailed micro-data are employed that are comprehensive of the contiguous United States. Drawing upon repeated observations of land uses and land characteristics on 844,000 sample points from the National Resources Inventory (NRI), we observe three land-use transitions between 1982 and 1997 for lands encompassing 91% of non-Federal lands in the contiguous United States. Previous econometric analyses have considered relatively small regions of the country and have not incorporated micro-data on land-use changes or land quality, a critical determinant of land-use decisions.

Third, key commodity prices are treated as endogenous in the simulations of the carbon sequestration supply function. A national-scale carbon sequestration policy will likely affect prices for forest and agricultural commodities, in turn changing the incentives of private landowners to convert land to or retain land in forest.

1.3. Preview of the paper

We model probabilities of transitions among land uses as functions of the anticipated economic returns to alternative uses. A nested logit specification is used for the transition probabilities to relax restrictions implicit with simpler logit models. The estimated parameters and respective elasticities are found to be consistent with expectations of economic theory. We find that the probability of a land parcel transitioning to a particular use increases as the county-average net returns to that land use increase, after accounting for the quality of the parcel.

We build upon the econometric results by simulating landowner responses to carbon sequestration policies, modeled as a combination of a tax on undesirable land-use changes and a subsidy for desirable ones. The simulations are iterative in nature, including feedback effects on commodity prices resulting from induced land-use changes. A carbon sink model is then used to derive changes in carbon storage associated with the sequence of land-use transitions estimated in the land-use simulation model, including forest carbon stored in biomass, litter, and soils, emissions associated with harvesting of merchantable wood, and agricultural carbon linked with soils and harvested biomass. Merging the results of these simulations with marginal costs numbers associated with the various levels of the subsidy/tax leads to the estimated carbon sequestration supply function.

We compare our estimated supply function with those from previous studies, and find that over the range of costs considered in previous studies our marginal cost estimates are considerably greater than those from engineering cost analyses and sectoral optimization models. Likewise, at low levels of sequestration, our marginal cost estimates exceed those from previous econometric analyses; this relationship is reversed, however, at higher levels of sequestration.

We also compare our estimated marginal costs of carbon sequestration with estimates of costs of energy-based carbon abatement. We find that the carbon sequestration supply function is roughly similar to the central tendency of the carbon abatement supply functions, which suggests that forest-based carbon sequestration merits consideration as part of a cost-effective portfolio of domestic US climate change strategies. For example, if emission reductions in the United States on the scale proposed under the Kyoto

[5]The CRP, established by the Food Security Act of 1985, provides annual rental payments to landowners voluntarily retiring environmentally sensitive land from crop production under 10- to 15-year contracts and currently enrolls 34.7 million acres (an area about the size of Iowa) [42].

Protocol were to be achieved entirely through domestic actions (forest-based sequestration and/or energy-based abatement activities) and with the type of policy incentive considered in this paper, our analysis implies that 33–44% of the reductions could be met cost-effectively through forest-based sequestration.

2. Econometric analysis of national land use

2.1. Econometric model

Consider a risk-neutral landowner facing the choice of allocating a parcel of land of uniform quality among a set of alternative uses.[6] We posit that landowners choose uses to maximize the present discounted value of the stream of expected net benefits from the land, and that landowners base their expectations of future land-use profits on current and historic values of relevant variables. Given these simplifying assumptions, the decision rule that emerges from the related dynamic optimization problem [16] is to choose the use with the highest expected one-period return at time t minus the current one-period expected opportunity cost of undertaking conversion. With K potential land uses ($j, k = 1, \ldots, K$), the owner of a unit of land in use j will choose the use k at time t that satisfies:

$$\arg\max_k (R_{kt} - rC_{jkt}) \geqslant R_{jt}, \tag{1}$$

where R_{jt} and R_{kt} represent the instantaneous expected net benefits at time t from a unit of land in use j and k, respectively; C_{jkt} is the expected marginal cost of converting one unit of land from use j to use k at time t ($C_{jjt} = 0$); and r is the discount rate. If the use k satisfying Eq. (1) equals j, then the land unit will remain in its current use at time t; otherwise, the landowner will reallocate the land to the use $k \neq j$ that maximizes expected net returns after conversion costs.

The landowner's profit function may be thought of as including both observed and unobserved components. Specific restrictions on the structure of the unobserved components yield alternative specifications of probabilistic models. An ordinary logit model is one obvious possibility, but the assumption of independent disturbances in the simple logit model implies that the ratio of the probabilities of any two choices is independent of the other alternatives.

We allow for differences in substitutability among alternatives using a *nested* logit specification, which imposes the property of "independence of irrelevant alternatives" within but not across specified subgroups ("nests") of choices. Dividing the choice set into mutually exclusive subgroups K_s ($s = 1, \ldots, S$), each containing elements l ($l = 1, \ldots, J_s$), we specify U_{ijkt}, the landowner's utility from allocating land parcel i in use j to use k at time t, as the sum of a component, V_{ijkt}, that is unique to the alternative k and another component, V_{ijst}, that is common to all the alternatives in K_s including k. Each of these components, in turn, includes an observed component plus an unobserved component characterized as a random error.

Under assumptions analogous to the standard logit model, the probability of choosing alternative k that is grouped in K_s can be expressed as the product of two terms: the probability, P_{ijst}, of choosing any of the alternatives within K_s; and the conditional probability, $P_{ijkt|s}$, of choosing k given the choice of K_s [6]. For land parcel i starting in use j, the probability of choosing land use $k \in K_s$ between time t and $t+1$ is thus:

$$P_{ijkt} = P_{ijst} \cdot P_{ijkt|s} = \frac{\exp(V_{ijst} + \tau_{st} I_{ijst})}{\sum_{s=1}^{S} \exp(V_{ijst} + \tau_{st} I_{ijst})} \frac{\exp(V_{ijkt})}{\sum_{l=1}^{J_s} \exp(V_{ijlt})}, \tag{2}$$

where τ_{st} are parameters, and $I_s = \ln \sum_{l=1}^{J_s} \exp(V_{ijlt})$. This "inclusive value" for nest K_s equals the expected utility for the choice of alternatives within a nest. The expression in Eq. (2) embodies the first-order Markov property since the probability of the parcel changing use depends only on exogenous covariates at time t.

Our chosen nesting structure is based on the premise that land uses with more similar land quality requirements are closer substitutes. We expect land quality to affect land-use net returns principally in terms of agricultural yields, and we measure land quality using the Land Capability Class (LCC), a summary measure

[6] If net returns and land conversion costs are linear in land quantity, the size of parcels will not affect the relative profitability of land-use options, so land-use choices for a heterogeneous parcel can be treated as the sum of land-use choices on constituent uniform-quality parcels.

of the suitability of land for producing crops [43]. Land in crops has the highest average quality as measured by the LCC, while pasture and CRP uses tend to be adopted on higher quality lands relative to forest and range uses. Crops, pasture, and CRP uses appear more similar in terms of the land quality required to generate a competitive level of returns, compared with forests and range. We incorporate these differences in land quality requirements by specifying our nested logit model with three nests: K_1 (crops, CRP, and pasture); K_2 (forest and range); and K_3 (urban). We model urban land use as a unique nest, due to its greater degree of irreversibility, and because land quality, as measured by the LCC, is likely to be a much less important determinant of urban development returns.

Landowners presumably compare net returns to alternative uses on particular parcels. Although we have land-use data at the parcel level, we lack parcel-level observations of net returns. Instead, we observe county-level average returns, and so to allow for parcel-level variation, we interact the average-return variables for each land use with parcel-level indicators of land quality. The magnitude of this land quality adjustment could be different for different land uses depending, for example, on the sensitivity of the different net returns to land quality as well as on the land quality distribution for the land in each use. We thus specify the component of utility that is unique to each alternative k as

$$V_{ijkt} = \alpha^0_{jkt} + \alpha^q_{jkt}\,LCC^q_{it} + \beta^0_{jkt}R_{kc} + \beta^q_{jkt}\,LCC^q_{it}\,R_{kc} + \varepsilon_{ijkt}, \tag{3}$$

where α^0_{jkt} is an alternative-specific intercept, α^q_{jkt}, β^0_{jkt}, and β^q_{jkt} are parameters,[7] R_{kc} is the level of net returns to use k in county c,[8] LCC^q_{it} is a dummy variable indicating whether parcel i is of quality q at time t, and ε_{ijkt} is the error term.[9] We lack data on the costs of changing land use, but we expect these costs to be closely related to land quality. Accordingly, we model conversion costs with the terms $\alpha^0_{jkt} + \alpha^q_{jkt}\,LCC^q_{it}$ in (3), which provide an intercept varying with land quality and initial use.

CRP participation depends on a different set of decisions than other land-use choices, because enrollment depends on both the landowner's bid, which includes a proposed rental rate, and the government's choice of whether to accept the bid, which depends on the environmental characteristics of a parcel as well as the cost. Because the program targets cropland, CRP rental rates are highly correlated with the profitability of cropping in a given locality. We account for the effect of crop net returns on the incentive to remain in cropland. Incentives to enroll in CRP are specified as a function of LCC, as lower land quality as measured by LCC has always been strongly associated with program eligibility. We would thus expect greater enrollment on lower quality lands. Eq. (3) thus becomes $V_{ijkt} = \alpha^0_{jkt} + \alpha^q_{jkt}\,LCC^q_{it} + \varepsilon_{ijkt}$ for $k = \text{CRP}$.

For the component of utility that is constant across the alternatives within each nest, we include constant terms for the nest and interactions with the land quality indicator variables. For land parcel i in use j, the component of utility that is constant within each nest is thus:

$$V_{ijst} = \gamma^0_{jst} + \gamma^q_{jst}\,LCC^q_{it} + \tau_{st}I_{ijst}, \tag{4}$$

where γ^0_{jst} is a constant specific to nest s and γ^q_{jst} is a coefficient on the land quality indicators. Substituting Eqs. (3) and (4) into (2) yields a complete nested logit model for estimation.

[7]If the net return variables perfectly captured the net returns for each land use at the level of the individual parcel, then the coefficients on net returns should simply reflect the marginal utility of income and would not be expected to vary across the alternatives. We do not expect the coefficients to be equal in our specification, as the net return variables are county averages and the LCC interaction terms scale these averages based on parcel-level land quality differences. The magnitude of this adjustment is expected to vary across uses based on the sensitivity of the net returns to land quality and the quality distribution of the land initially in each use.

[8]Due to the logit structure of the model, this specification based on net return levels is identical to one where the net return variables are all rescaled as differences from the net returns to one base land-use category. Any component of utility that is constant across alternatives will drop out of the probabilities in (2), so only relative differences will determine the land-use decisions.

[9]The eight LCCs were merged into four groups (I–II, III–IV, V–VI, VII–VIII) to ensure enough observations in each land use and quality to estimate the nested logit model. An analysis based on a non-nested (conditional logit) version of the model indicated that our land-use simulations (see the next section) were not sensitive to this aggregation of the land quality classes.

2.2. Econometric estimation and results

We estimate the model using repeated observations of non-Federal land use from the NRI of the US Department of Agriculture's Natural Resources Conservation Service (NRCS). The NRI is a panel survey of land use and land characteristics on non-Federal lands conducted at 5-year intervals from 1982 to 1997 over the 48 contiguous United States. Data include approximately 844,000 point-level observations, each representing a land area given by a sampling weight [26]. We observe land use at each sample point in four survey years, providing information on land-use changes over three transitions: 1982–1987, 1987–1992, and 1992–1997. We focus on six land uses: crops, pasture, forest, urban, range, and CRP.[10] The land base in our analysis comprises 1.4 billion acres, representing about 74% of the total land area and about 91% of non-Federal land in the contiguous United States. Further details on the NRI data are provided in Appendix A.[11]

The NRI data reveal the disposition of land units across major uses over time. Land units generally remain in the same use. For example, of land parcels cropped in 1982, 84.2% remained in crops in 1997, while 7.3% had been converted to CRP, 4.6% to pasture, 1.6% to urban use, 1.3% to forest, and 0.9% to range. Of land parcels forested in 1982, 95.4% remained in forests in 1997, while 2.5% had been converted to urban, 1% to pasture, 0.5% to crops, another 0.5% to range, and 0% to CRP. The urban land-use category appears to be an absorbing state, with lands almost never converted from urban to non-urban uses.

Our dependent variable is the choice of land use in year, $t+5$, at each NRI point, and our covariates are the land use in year t, the land quality rating of the point, and proxies for the expected net returns from the land-use alternatives as of year t. To smooth temporary shocks from weather and other factors affecting net returns in particular years, we assume that landowners use an average of the annual net returns per acre to each land use over the most recent 5-year period as the basis for their expectations of future net returns. Denoting each year as t, we specify land-use choices observed at time $t+5$ as a function of the average land-use net returns between years $t-4$ and t, inclusive.

By assembling data from a variety of private and public sources, we constructed county-level estimates of annual net returns (per acre) for crops, pasture, forest, range, and urban uses for all 3014 counties in the 48 contiguous states (Appendix A). Net return estimates are thus constructed for each of our land-use categories, except for CRP, which is treated differently, as discussed above. The estimates for cropland include net returns from market sales as well as direct farm program payments. We report summary statistics on land-use net returns in Table 1.

We employ maximum likelihood methods to estimate the parameters of the model for the transition probabilities from each of four starting land uses (crops, pasture, forest, and range) to each of our six ending uses (crops, pasture, forest, range, urban, and CRP).[12] We estimate versions of the econometric model using data from two different time periods and then compare the results. First, we estimate the model using all observations from the 1992–1997 period, the latest transition for which NRI data are available. These estimates draw solely on cross-sectional variation in the independent variables. If there are variations over time in landowners' responses to economic incentives, possibly the result of permanent changes in factors not modeled by our analysis, then parameter estimates based on the most recent data will provide the best foundation for predicting future behavior.

Second, we estimate the model using a sample of pooled data from each of the three available transitions (1982–1987, 1987–1992, and 1992–1997). For each starting use, we stratify the pooled dataset by NRI point and select a random sample that includes only a single observation of each land unit. In this way, we include observations from each time period, but purge the sample of serial correlation due to parcel-specific unobserved effects, which would violate the assumption of independent disturbances in the logit model and potentially bias the estimates. The estimated coefficients reflect the average behavior of parcels in each land

[10]Public lands and transportation infrastructure are excluded from the analysis, as changes in these uses are not affected directly by private landowner decisions. We omit water bodies and barren lands as these uses are unlikely to vary over time. Finally, we exclude lands classified as marshlands and "miscellaneous," as data to measure net returns to these uses are not available.

[11]All the appendices are available as an online supplement to this article at http://www.aere.org/journal/index.html

[12]Because land virtually never transitions out of urban uses, we do not model these potential land-use changes and, for our simulations, assume land parcels remain urbanized with 100% probability. We estimated the model for lands starting in CRP over 1992–1997 only. Some parcels became eligible to exit the program in 1996, and changed to other uses by 1997.

Table 1
Summary statistics

Variables	Level	Mean and standard deviation by year[a]			
		1982	1987	1992	1997
Land use is crops (1 = yes, 0 = no)	NRI point	0.296 (0.456)	0.286 (0.452)	0.269 (0.443)	0.266 (0.441)
Land use is pasture (1 = yes, 0 = no)	NRI point	0.092 (0.289)	0.089 (0.285)	0.089 (0.284)	0.084 (0.278)
Land use is forest (1 = yes, 0 = no)	NRI point	0.282 (0.450)	0.284 (0.451)	0.284 (0.451)	0.286 (0.451)
Land in urban use (1 = yes, 0 = no)	NRI point	0.036 (0.187)	0.041 (0.197)	0.046 (0.209)	0.054 (0.225)
Land use is range (1 = yes, 0 = no)	NRI point	0.293 (0.455)	0.289 (0.453)	0.287 (0.452)	0.286 (0.452)
Land use is CRP (1 = yes, 0 = no)	NRI point	0 (0)	0.010 (0.098)	0.024 (0.153)	0.023 (0.024)
Land Capability Class 1–2 (1 = yes, 0 = no)	NRI point	0.232 (0.422)	0.232 (0.422)	0.231 (0.422)	0.231 (0.421)
Land Capability Class 3–4 (1 = yes, 0 = no)	NRI point	0.347 (0.476)	0.347 (0.476)	0.347 (0.476)	0.347 (0.476)
Land Capability Class 5–6 (1 = yes, 0 = no)	NRI point	0.215 (0.410)	0.214 (0.410)	0.215 (0.410)	0.215 (0.410)
Land Capability Class 7–8 (1 = yes, 0 = no)	NRI point	0.206 (0.404)	0.207 (0.404)	0.207 (0.405)	0.207 (0.405)
Crop net returns ($/acre/year)[b]	County	58.4 (38.4)	73.8 (45.0)	82.8 (48.2)	n/a
Pasture net returns ($/acre/year)	County	16.1 (11.3)	7.9 (8.0)	12.7 (9.0)	n/a
Forest net returns ($/acre/year)	County	6.0 (5.9)	9.0 (9.0)	17.2 (17.1)	n/a
Urban net returns ($/acre/year)	County	3302 (3120)	4651 (5591)	4475 (4418)	n/a
Range net returns ($/acre/year)	County	11.2 (10.1)	10.3 (8.9)	10.4 (9.3)	n/a

[a]Point-level variables are indicator variables weighted by NRI-point acreage weights. Net returns are lagged 5-year averages in 1990 dollars (deflated by the producer price index for all commodities) weighted by the county acreage in each land use given by the NRI.
[b]Equals the sum of the market-component of crop net returns plus direct government payments (excluding the Conservation and Wetlands Reserve Programs).

Table 2
Own-return land-use choice elasticities[a]

Initial land use	Final land use and time period[b]									
	Crops		Forest		Pasture		Range		Urban	
	1982–97	1992–97	1982–97	1992–97	1982–97	1992–97	1982–97	1992–97	1982–97	1992–97
Crops	0.192**	0.011**	0.332**	0.310**	0.090**	0.183**	0.477**	0.376**	0.156**	0.342**
	(0.005)	(0.001)	(0.024)	(0.043)	(0.017)	(0.031)	(0.036)	(0.048)	(0.005)	(0.016)
Forest	0.178**	0.295**	−0.000	0.001	0.091	−0.000	0.235**	0.232	0.511**	0.792**
	(0.039)	(0.064)	(0.000)	(0.055)	(0.179)	(0.000)	(0.033)	(0.330)	(0.010)	(0.058)
Pasture	0.306**	0.341**	0.023*	0.005	−0.005	−0.012	1.373**	1.042**	0.314**	0.331**
	(0.012)	(0.022)	(0.011)	(0.027)	(0.003)	(0.008)	(0.033)	(0.050)	(0.014)	(0.026)
Range	0.072	0.065	0.064**	0.127	0.159	0.399	−0.002	−0.001	0.385**	0.419**
	(0.069)	(0.229)	(0.023)	(0.906)	(0.700)	(0.417)	(0.001)	(0.971)	(0.036)	(0.031)

[a]Elasticities are evaluated at the means of the data and are the percentage change in the probability of choosing the final land use, conditional on being in the initial use, for a 1% change in the net returns to the final use. Standard errors, in parentheses, are estimated using the Delta Method [11]. * and ** denote significance at the 5% and 1% levels, respectively.

[b]There are no own-return elasticities for CRP as net returns from CRP are not directly specified in the econometric model.

use from 1982 to 1997, weighted by the acreage in each period. If changes in estimated parameters over time are due to transient shocks, then estimates based on a longer time period are potentially superior for predictive purposes.

The results for both the 1992–1997 and 1982–1997 periods indicate good fit of the model, and are consistent with the expected economic relationships.[13] Estimated parameters for the eight equations are reported in Appendix D. For each starting use, Table 2 reports the estimated elasticity of the probability of choosing each land-use alternative with respect to the net returns to that alternative (‘‘own return elasticities’’). These elasticities indicate the percentage change in the probability of a particular land-use change for a 1% change in the corresponding net returns. In 11 out of 20 cases for 1992–1997 and 14 out of 20 cases for 1982–1997, the own-return elasticities are positive and significantly different from zero at the 0.01 level. In the three cases for each set of estimates where the own-return elasticities are negative, they are never significantly different from zero, even at the 0.05 level. The cross-elasticities (not reported) are always opposite in sign to the own-return elasticities.[14]

The elasticities indicate that landowners starting with lands in either crops or pasture responded as anticipated to net returns from alternative land uses. For land starting in range, the own-return elasticities with respect to urban net returns are positive and significant, as well as the forest net return elasticity for 1982–1997. None of the other own-return elasticities (for land starting in range use) are significantly different from zero, suggesting that rangeland owners are relatively insensitive to the profitability of alternative uses, with the exception of urban development. This is reasonable, given that range lands tend to be of the lowest quality and thus unsuitable for other agricultural uses.

[13]For the transition-specific estimates, pseudo R^2 values (McFadden's likelihood ratio index) range from 0.68 to 0.95 and 0.71 to 0.95 for the 1992–1997 and 1982–1997 estimates, respectively. The positive and generally significant estimates of the inclusive value parameters together with tests [12] of a non-nested logit model using the same variables are consistent with violations of the ‘‘independence of irrelevant alternatives’’ hypothesis, supporting the use of the less restrictive nested specification. For land initially in pasture, one inclusive value parameter in both the 1982–97 and 1992–97 estimates lies outside the unit interval with 95% confidence. This suggests these estimates may not be globally consistent with random utility maximization [19]. Possible explanations are the crude data on pasture returns, as well as difficulties in distinguishing permanent pasture versus pasture in rotation with crops. Any misspecification in the pastureland equation, however, is expected to have small impacts on the overall results given that pasture accounted for a relatively small share of the total land base in 1997 (see Fig. 1).

[14]The cross-elasticities are the elasticities of the probability of choosing a particular use j with respect to the profits of a different use k. In the nested logit model, these can be of the same sign as the own-return elasticities when the inclusive value parameters are negative.

The elasticities with respect to forest net returns are especially important for our simulation model of carbon sequestration. These elasticities are positive, as expected, in seven out of eight cases. While the elasticity is negative for forestlands over 1982–1997, it is not significantly different from zero. The elasticity with respect to forest net returns is positive and significantly different from zero for lands starting in crops for both models. For lands starting in pasture and range, the forest net return elasticities are positive for both periods and significantly different from zero over 1982–1997.

A final econometric consideration is the fact that land parcels located near one another may have unobserved characteristics that are correlated across space. If such characteristics influence land-use decisions or if local land-use choices are interdependent, error terms will be correlated across space, leading to inconsistent and inefficient estimates in a logit model, due to induced heteroskedasticity [20]. We explored the potential importance of spatial dependence by eliminating observations near one another. Estimates with samples that included only one point within each of the NRI's primary sampling units produced results similar to estimates including all points, suggesting that spatial dependence is not a critical concern.

3. Simulation model of carbon sequestration

In order to estimate the carbon sequestration supply function, we conduct policy simulations and compute corresponding flows of carbon in terrestrial sinks. We examine a two-part policy involving a subsidy for the conversion of land to forest and a tax on the conversion of land out of forest. As noted by Stavins [39], a policy that only subsidizes forestation creates incentives for landowners to convert land out of forest and then back into forest in order to receive the subsidy. This entails inefficient expenditures on land conversion, which are discouraged by the tax on deforestation in the two-part policy. A second feature of the policy is a requirement that afforested lands remain in forest for a specified period of time.[15]

For each policy scenario, we specify the level of the subsidy and tax, denoted Z. For land moving into forest, the subsidy increases the annual net return to forest, and, for land moving out of forest, it reduces the annual net return to non-forest uses. Suppressing the parcel and time subscripts, indexing land uses by j and k, forest use by 1, and the relevant set of non-forest uses by -1, Z modifies the estimated transition probabilities P_{jk} in the following manner:

$$P_{j1} = f(\hat{\boldsymbol{\beta}}_{j1}, NR_1 + Z, \mathbf{NR}_{-1}), \tag{5a}$$

$$P_{11} = f(\hat{\boldsymbol{\beta}}_{11}, NR_1, \mathbf{NR}_{-1} - \mathbf{Z}), \tag{5b}$$

$$P_{1j} = f(\hat{\boldsymbol{\beta}}_{1j}, NR_1, \mathbf{NR}_{-1} - \mathbf{Z}), \tag{5c}$$

$$P_{jk} = f(\hat{\boldsymbol{\beta}}_{jk}, NR_1 + Z, \mathbf{NR}_{-1}) \tag{5d}$$

for all j and k not equal to 1, where $\hat{\boldsymbol{\beta}}$ is a vector of estimated parameters, NR_1 is the annual net return to forest land, $\mathbf{NR}_{-1}$ is a vector of annual net returns to the relevant non-forest uses, and $\mathbf{Z}$ is a conformable vector with each element equal to Z. In general, Z has the effect of increasing the probability that land shifts into forest and reducing the probability that it shifts out of forest.

Four issues merit comment. First, the policy mechanism implicit in our cost simulation model provides incentives for land-use changes, rather than for carbon sequestration directly. If the costs of administering these two types of policies are the same, carbon sequestration costs will be lower with a policy directly targeting carbon, because it will tend to convert land with the lowest costs per ton of carbon, rather than the lowest cost per acre of land. It is likely, however, that the administrative costs of the two approaches would differ significantly. In general, performance measures based on inputs to carbon sequestration—such as quantities of particular types of land—would be relatively easy to monitor, compared with the task of monitoring the quantity of carbon sequestered, relative to an assumed reference case [40].

[15]This is a common contractual arrangement under voluntary land conservation programs. For example, CRP contracts stipulate that land parcels must be kept in a specified conservation use for a period of 10–15 years.

Second, the subsidy and tax rates are identical to each other, and remain constant over time. If rates of carbon sequestration differ between land that is afforested and land on which deforestation is deterred, then equal subsidy and tax rates may not be efficient. We assume such equal rates, because it simplifies the policy simulations considerably. Moreover, analyses of efficient and cost-effective climate change mitigation have found that carbon prices should rise over time to delay the costs of stabilizing or reducing greenhouse gas concentrations [10,25]. If incentives for carbon sequestration were to increase over time, landowners with forward-looking price expectations would, in some instances, have incentives to delay carbon sequestration, even if it is profitable to undertake immediately [48]. To be consistent with the econometric model, we assume landowners have static price expectations, which implies that significant incentives do not exist to delay profitable carbon sequestration; and to simplify the simulations, we treat the subsidy/tax rate as constant over time.

Third, the probabilities in Eq. (5) are expressed as functions of R_1, the annualized net returns from timber harvests. Thus, we assume initially that all forested land is eligible for timber harvesting, including those lands converted to forest in response to the subsidy. We also consider policies designed to establish permanent forest stands. In this case, timber harvesting is prohibited on afforested lands, but still allowed on land that is forested at the beginning of the simulation. To model this policy, we set R_1 equal to zero for all transitions of land into forest (5a) and all transitions between non-forest uses (5d). In this case, landowners receive only the subsidy Z for afforesting their land. For land that begins in forest, and hence is ineligible for the forestation subsidy, R_1 is unmodified (5b and 5c).

Fourth and finally, the modified transition probabilities in (5) are used in the policy simulation model to generate estimates of acreage transitions between pairs of land-use categories for each level of Z. Land afforested in response to the subsidy is assumed to remain in forest throughout the policy simulation. Thus, for these forest lands, but not for land originally in forest, we set the probability in (5c) to zero. We also compute a baseline sequence of land-use transitions by setting Z in (5) equal to zero. Naturally, in computing this baseline, we do not require afforested land to remain in forest. The impact of the policy on carbon sequestration is measured relative to the baseline (that is, we net out the change in carbon storage that would have occurred in the absence of the policy). As a practical matter, baseline actions are not observable once the policy is in effect, implying that some landowners are subsidized for forestation they would have undertaken even without subsidies. In the policy simulations, all landowners are subsidized for forestation, and hence are required to retain their land in forest. But to measure the cost of the policy, we consider foregone net returns net of the baseline. In this way, our estimates measure the economic costs to society, rather than the financial costs to the government.

A policy simulation model, described in Appendix B, is used to simulate the response to the incentives for carbon sequestration and to estimate feedback effects on timber and crop prices resulting from induced land-use changes. Aggregating simulated land-use changes at the level of each NRI point, the simulation model generates national estimates of acreage transitions among land-use categories over long time horizons. A carbon sink model, described in Appendix C, is used to translate the estimated land-use transitions into projected carbon flows. This model accounts for changes in carbon stocks in the relevant biomass, soil, and product categories for each of the land uses. It is important to acknowledge that, while we use the best available information on carbon storage, considerable uncertainty surrounds these estimates. In addition, we have ignored some of the practical issues that would arise with a carbon sequestration policy, including monitoring and enforcement.[16]

The carbon sequestration supply function is derived using the procedure discussed by Stavins [39]. For the baseline and each level of the subsidy/tax, we record the areas of land afforested and deforested during each period of the simulation. Net forestation for a given subsidy/tax level is defined as the area afforested minus the area deforested, net of this quantity in the baseline. The product of net forestation and the subsidy/tax gives the opportunity cost of the policy. For example, if a payment of $100 is required to induce an extra acre of forestation, then this represents the foregone profits associated with the original use of the land. The total cost of the policy is computed as the present value of cumulative opportunity costs discounted from the

[16]For more discussion, see the special issue of Choices devoted to climate change [38].

midpoint of each 5-year interval. Likewise, we compute cumulative carbon flows relative to the baseline and the present value of carbon sequestered.[17]

Total costs and total carbon are computed for each subsidy/tax rate and expressed as annualized equivalents. Marginal costs of carbon sequestration equal the change in costs divided by the change in carbon, computed as the ratio of the increment in annualized total costs (from one subsidy/tax level to the next) to the corresponding increment in annualized total carbon. The supply schedule is constructed by arraying marginal costs against annualized total carbon.

4. Empirical results

Simulations are conducted for subsidy/tax rates (values of Z) ranging in \$50 increments from \$0 to \$350 per acre. The two versions of the econometric model—one estimated for the period, 1982–1997, and the other for the period, 1992–1997—produce very similar carbon sequestration supply functions for carbon prices below \$200 per ton. Our discussion below focuses on results generated with the 1992–1997 model.

4.1. Primary results

We simulate national total quantities of land in six categories of use over time for the baseline and policy scenarios. In the initial year (1997), there are approximately equal areas of cropland, rangeland, and forest (about 400 million acres each), while the areas of land in pasture, urban, and CRP vary from 33 to 119 million acres (Fig. 1). Under the baseline scenario, the largest change is in the area of urban land, which increases from 76 to 495 million acres during the 250-year simulation period (Fig. 1). The areas of land in agricultural uses (crops, range, pasture, and CRP) decline throughout the simulation period, with cropland experiencing the greatest absolute loss. Forest land increases initially but begins to decline after about 50 years. After 250 years, there is a relatively small net decline in forest area in the baseline.

When a \$100 per acre subsidy/tax is introduced, forest area almost doubles during the simulation period, from 405 to 754 million acres (Fig. 2). Most of the increase is due to increased forestation and, correspondingly, there are larger declines by the end of the simulation in the areas of cropland, rangeland, pasture, and CRP. The increase in urban land area is also smaller than in the baseline, as more land is converted to or retained in forest. The area of cropland increases for several decades before declining. By encouraging conversion of cropland to forests, the subsidy/tax has positive effects on net returns to cropland, raising the probability that land moves into crops. Initially, there is a net increase in cropland as more land moves into crops from other agricultural uses (pasture, range, and the CRP) than is converted from crops to forest and urban. But as the remaining acres of other agricultural lands decline, flows of land into crops fall, and there is a net decline in cropland.

While the general pattern of land-use changes is similar at other levels of the subsidy/tax, in a given year the increment in forest area declines as the rate of the subsidy/tax increases.[18] In addition, at the highest subsidy/tax rates, forest area increases rapidly in early years of the simulation before quickly leveling off. For example, at subsidy/tax rates above \$250 per acre, the area of forest more than doubles by 25 years into the simulation, but after 25 years, relatively little forest is added, because a large share of the land base has already been afforested.

There is an increase in carbon storage when agricultural lands are converted to forests or land is retained in forest.[19] Fig. 3 portrays carbon flows relative to the baseline for selected subsidy/tax rates. There is a small negative carbon flow in the first period of the simulation for all levels of the subsidy/tax, because cropland area increases initially under the subsidy/tax scenarios due to higher net returns induced by the policy, whereas cropland area declines initially in the baseline. The net increase in cropland results from the conversion of

[17]See Richards and Stokes [34] for an assessment of alternative approaches to intertemporal carbon accounting in carbon sequestration cost studies.

[18]This is true except at low levels of the subsidy/tax, where the increments increase with the subsidy/tax. As discussed below, this results in our supply function having a convex portion.

[19]Appendix C provides details on estimated carbon storage levels under different land uses.

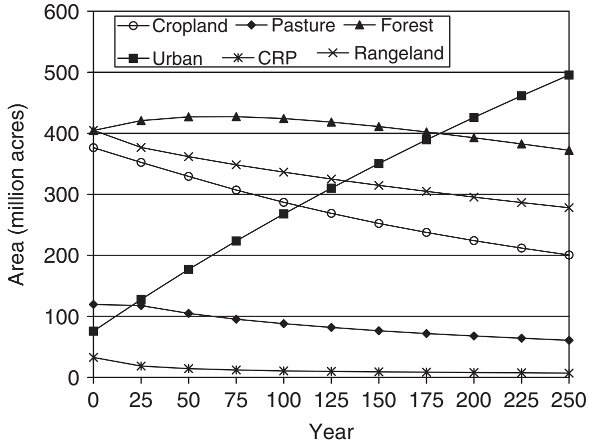

Fig. 1. The area of land by use in the baseline scenario.

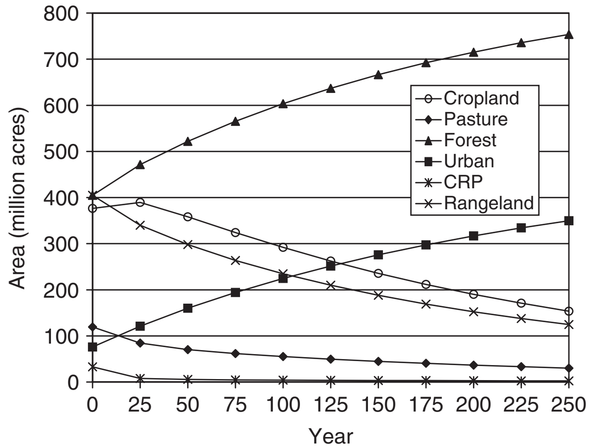

Fig. 2. The area of land by use with a $100 per-acre subsidy/tax.

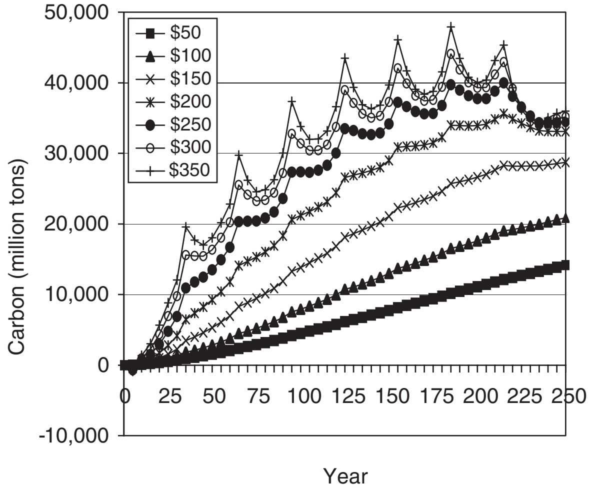

Fig. 3. The flow of carbon relative to the baseline with different subsidy/tax rates.

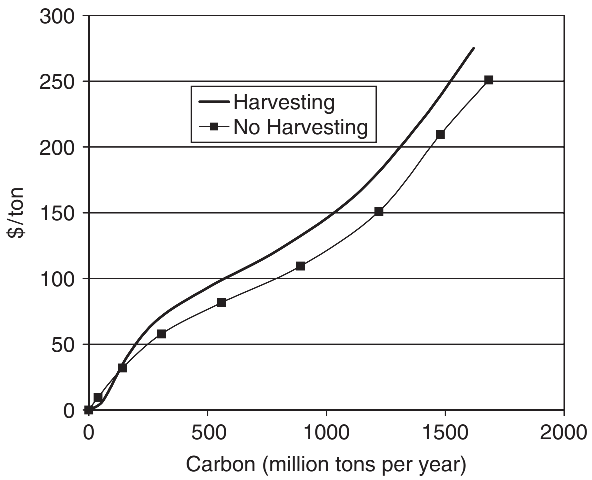

Fig. 4. The marginal costs of forest-based carbon sequestration in the United States.

pasture and rangeland, producing an immediate and negative flow of carbon (see Appendix C). Once new forests become established, transitions between non-forest lands contribute relatively little to the aggregate carbon supply. The $100 per acre subsidy/tax captures an additional 1.2 billion tons of carbon on non-forest land over the 250-year horizon by, for example, deterring conversion of pasture to cropland. However, transitions involving non-forest lands account for less than 1% of the net carbon sequestration induced by the policy.

Moving from lower to higher subsidy/tax rates, carbon flows increase, though eventually at a diminishing rate, due to the declining induced increment in forest area. At the highest subsidy rates, a scalloped pattern emerges, reflecting the periodicity of harvest and regrowth cycles. On afforested lands, there is a positive flow while trees grow to maturity. The flow is negative at the time of harvest and, then, the cycle repeats itself. The pattern is most pronounced at the highest subsidy/tax rates, because much of the land that will be afforested during the simulation is converted during the first years of the simulation. At lower subsidy/tax levels, the increase in forest area is more gradual, creating a more even pattern of harvest and regrowth.[20]

The land-use change and carbon flow results are combined to produce a marginal cost function for forest-based carbon sequestration. Econometric estimation and respective simulations were carried out both with the full set of three transitions over the period 1982–1997 and with the final transition of 1992–1997. As highlighted above, the share of land in urban uses has increased monotonically and dramatically, suggesting significant structural change. Because the simulations are forward-looking and span very long time horizons, we employ the more recent 1992–1997 econometric estimates and related land-use simulations in our calculations of the carbon sequestration marginal cost function.[21] This is portrayed in Fig. 4, where the solid line indicates the supply function for our basic scenario (allowing harvesting of wood products).

Although the marginal cost function is convex throughout most of its range, it is concave at low levels (below $70/ton). This is a consequence of the logistic specification used in the underlying econometric model. Recall that the vast majority of plots in the NRI sample remains in the same use over time; in other words, there is a low probability that land transitions out of its current use. At low marginal costs (low subsidy/tax),

[20]For most of the simulation, the flows trend upward, due largely to the addition of new forests over time, but after about 200 years, the flows drop to lower levels for the highest subsidy/tax rates. There are two reasons for this. First, throughout the simulation a large amount of carbon is stored in solid wood products and gradually released. Once forest area stabilizes, these negative carbon flows are no longer offset by the addition of new forest lands, which generate positive carbon flows. Second, carbon in forests that are never harvested reaches a maximum level after 12 decades (see Appendix C). The zero flows in unharvested forests result in a diminished aggregate carbon flow. At higher subsidy/tax rates, the negative flows from the product pools become apparent sooner, and zero flows in unharvested forests occur earlier.

[21]It should be noted, however, that the results from both sets of econometric estimations and related simulations are similar.

the logistic is a convex function of its arguments. As forest net returns increase in the policy scenarios, there is a range of subsidy/tax rates for which the land moving into forest increases at an increasing rate, implying that marginal costs of carbon sequestration increase at a decreasing rate at low levels of the subsidy/tax.[22]

4.2. Sensitivity analysis

In deriving the main results, we assume all forest lands (including lands converted to or retained in forest in response to policy) are periodically harvested. Another realistic policy scenario involves a contractual stipulation that prohibits timber harvests on lands enrolled in a carbon sequestration program. A prohibition on harvesting may increase the discounted value of carbon sequestered, because negative carbon flows resulting from harvesting are avoided. But harvesting restrictions also reduce or even eliminate revenues received by landowners, increasing the financial incentives needed to increase forest area. Thus, a priori, the net effect on the marginal costs of carbon sequestration of allowing timber harvesting is ambiguous.

Therefore, we also conduct simulations for a no-harvesting scenario. We find that prohibiting harvesting lowers the marginal costs of sequestration (Fig. 4). The key reason is that at each level of the subsidy/tax, total forest area is greater under the no-harvesting scenario. Although harvesting restrictions reduce the amount of land enrolled in the carbon sequestration program, they also result in higher timber prices relative to the corresponding harvesting case, which helps to retain existing (that is, non-program) lands in forest. In addition, at a 5% discount rate, the per-acre discounted carbon flows are greater under the no-harvesting scenario. In sum, more forest land outside the program and more carbon sequestration on program lands combine to produce lower marginal costs when harvesting is prohibited.

We also examined the sensitivity of the carbon sequestration supply function to alternative discount rates. In the econometric and simulation analysis reported above, the private net returns to forest and urban uses are annualized using a 5% rate (the returns to other land uses are measured in annual terms). We compare three alternatives (1%, 3%, and 7%) to the benchmark rate of 5%. Higher discount rates lower the present value of both costs and carbon flows. Hence, in theory, changes in the discount rate have an ambiguous effect on unit costs of the policy (the ratio of annualized dollars to annualized tons of carbon). Empirically, we find that higher discount rates increase unit costs, and decrease annualized carbon flows. Thus, the marginal cost curve for carbon sequestration shifts up as we move from lower to higher discount rates. For sequestration of 500 million tons of carbon per year, the marginal (average) cost per ton rises from \$8 (\$4) to \$41 (\$20), \$93 (\$53), and \$164 (\$102) at discount rates of 1%, 3%, 5%, and 7%, respectively.

4.3. Comparison with results from other studies

We compare our marginal cost estimates with those from previous studies of carbon sequestration in forests (Fig. 5) and carbon abatement (Fig. 6).[23] Over most of the range of carbon prices considered in previous forest-based carbon sequestration studies, our cost estimates are higher than those obtained using optimization models [1,7] and bottom-up engineering cost methods [33]. Comparing our estimates with those from the earlier econometric study by Stavins [39], we find similar costs at low carbon sequestration levels, but lower costs at higher carbon sequestration levels. In particular, the Stavins curve becomes close to vertical by about 600 million tons of carbon per year, whereas our curve is still relatively flat at this point. Part of the reason for this difference is the fact that Stavins extrapolates his econometric results from the Mississippi Delta region to US farm acreage totaling 551 million acres, including cropland and pasture. In the present study, crop and pasture acreage are somewhat lower, but rangeland and CRP are included, yielding a total of 933 million acres of privately owned agricultural land available for conversion to forest. As Figs. 1 and 2 indicate, a substantial amount of the increase in forest area in our simulations is due to conversion of

[22]The convexity of the marginal cost curve at higher marginal costs is due to the fixed size of the land base, rather than the passing the inflection point in the logistic probability curve. Net forestation in the first period increases at an increasing rate for all of the subsidy/tax levels considered. However, at higher subsidy/tax levels, the constraint on available land leads to decreasing incremental increases in total cumulative net forestation (and carbon sequestration).

[23]Stavins [39] reports and discusses the marginal cost estimates from previous studies.

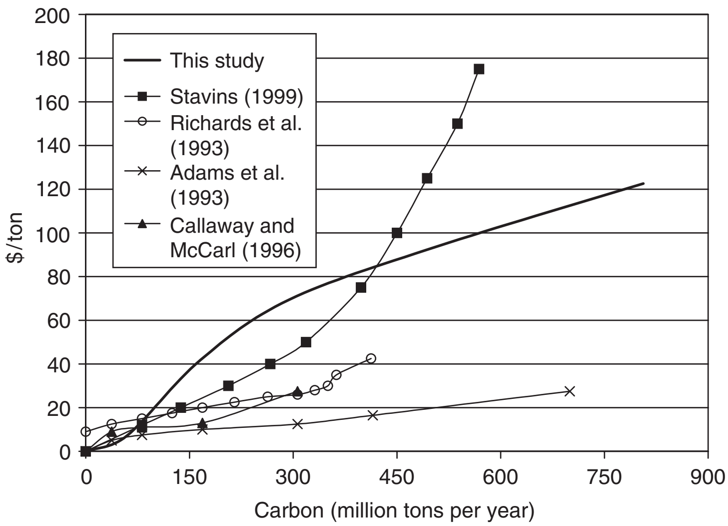

Fig. 5. Alternative estimates of the marginal costs of forest-based carbon sequestration in the United States.

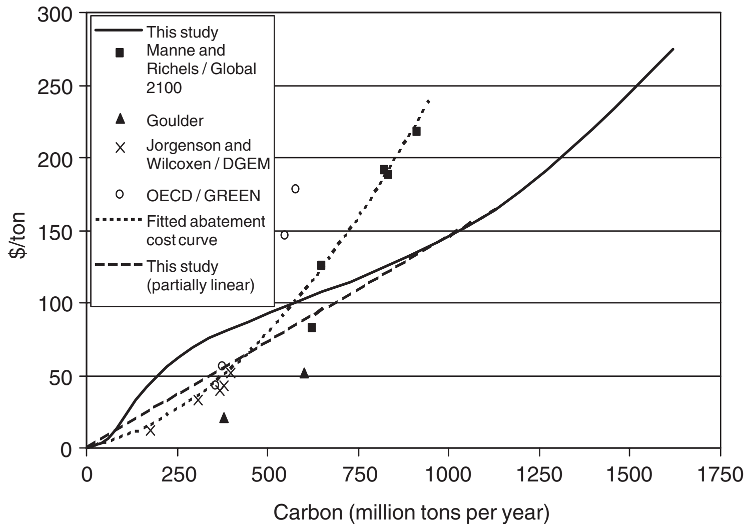

Fig. 6. Estimates of the marginal costs of forest-based carbon sequestration and carbon abatement in the United States. The abatement cost estimates are derived from Energy Modeling Forum [9]. See Stavins [39] for details.

rangeland to forest. Stavins' econometric model was fitted to data exclusively of transitions between cropland and forest.[24]

Finally, we compare the marginal costs of forest-based carbon sequestration with estimates of energy-based carbon abatement.[25] For estimates below $100 per ton, our estimates are comparable to those of Jorgensen and Wilcoxen, Manne and Richels, and OECD/Green—particularly if the initial part of our curve is treated as linear—and higher than those in Goulder. For costs above $100 per ton, our estimates are lower than the abatement studies considering costs in this range, namely OECD/Green and Manne and Richels.

[24]An additional explanation for the difference is that the 36 counties examined in the Stavins [39] study may simply not be representative of the United States along relevant dimensions.

[25]The abatement cost estimates are derived from estimates in the Energy Modeling Forum [9]. See Stavins [39] for details.

Given the relative positions of the supply functions, the results suggest that forest-based carbon sequestration merits consideration as part of a cost-effective portfolio of domestic US climate change strategies. Consider, for example, a domestic program that would bring US annual greenhouse gas emissions 7% below 1990 levels over the period, 2008–2012, as would be required by the Kyoto Protocol. This would entail a reduction of about 573 million short tons of carbon in the year 2010.[26] To compare the abatement and sequestration cost estimates explicitly, we fit a function to the point estimates from the abatement studies (the dotted line in Fig. 6).[27] Our analysis indicates that if cost-effective emission reductions[28] in the United States on the scale proposed under the Kyoto Protocol were to be achieved entirely through domestic actions (forest-based sequestration and/or energy-based abatement activities), 33% of the reduction would be achieved through forest-based sequestration (or up to 44% if the lower portion of our curve were linearized to join the convex section, the dashed line in Fig. 6).

5. Conclusions

In this econometric analysis of carbon sequestration costs, six major land uses were modeled, detailed micro-data of land use and land quality were employed that are comprehensive of the contiguous United States, and key commodity prices were treated as endogenous in the simulations of the carbon sequestration supply function. We compared our estimated carbon sequestration supply function with ones from previous studies, and found that over the range of carbon prices considered in most studies, our marginal cost estimates are greater than those from engineering cost analyses and sectoral optimization models. Because our cost estimates are derived from landowners' actual behavior regarding disposition of their lands, they may reflect factors such as option values associated with delaying irreversible land conversion, liquidity constraints, and unobserved benefits and costs of alternative land uses.

We find lower marginal costs of carbon sequestration when timber harvesting is prohibited on lands enrolled in the carbon sequestration program. Marginal costs fall because the additional present value costs of enrolling lands on which harvesting is prohibited are more than outweighed by the additional present value carbon sequestered. This result is reinforced by endogenous price effects. Restrictions on timber harvesting on enrolled lands raise timber prices, creating incentives for landowners to retain existing (non-program) lands in forest.

The national scope of our study allows us to compare directly our estimates of the marginal costs of carbon sequestration with estimates of costs from energy-based carbon abatement analyses. We find that the estimated carbon sequestration supply function is roughly similar to the central tendency of the carbon abatement supply function, indicating that about a third of the US target under the Kyoto Protocol would be cost-effectively achieved by employing forest-based sequestration policies, in addition to energy-based carbon abatement strategies. This suggests that forest-based carbon sequestration merits consideration as part of a cost-effective portfolio of domestic US climate change policies.

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[26]Estimated US net greenhouse gas emissions in 1990 were 1555 million short tons of carbon equivalent [46]. Projected 2010 energy-based carbon emissions are 1972 million short tons [44] and projected other greenhouse emissions minus sequestration removals are 47 million short tons of carbon equivalent [45].

[27]A quadratic form was chosen so as to minimize the Akaike [3] information criterion (AIC).

[28]The cost-effective portfolio equalizes the marginal costs of sequestration and abatement at the level that achieves the desired total reduction.

Appendix. Supplementary materials

Supplementary information (Appendices A to D) associated with this article can be found in the online version at doi:10.1016/j.jeem.2005.08.001 or in the online archive for supplementary material at http://www.aere.org/journal/index.html

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