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ABSTRACT

When and if the United States chooses to implement a greenhouse gas reduction program, it will be necessary to decide whether carbon sequestration policies — such as those that promote forestation and discourage deforestation — should be part of the domestic portfolio of compliance activities. We investigate the cost of forest-based carbon sequestration. In contrast with previous approaches, we econometrically examine 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. Key commodity prices are treated as endogenous and a carbon sink model is used to predict changes in carbon storage. Our estimated marginal costs of carbon sequestration are greater than 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 inclusion in a cost-effective portfolio of domestic U.S. climate change strategies.

Keywords: abatement; carbon; climate change; costs; forestry; greenhouse gases; land use; land-use change; sequestration.

JEL Classification: Q540, Q230, Q240, Q150

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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.[1] The Kyoto Protocol to the Framework Convention on Climate Change (United Nations General Assembly 1997) states that parties to the agreement (the participating nations) can employ carbon sequestration as part of their portfolios of strategies to achieve their domestic CO_2 targets. Hence, when and if the United States chooses to implement a domestic greenhouse gas reduction program, it will be necessary to decide whether carbon sequestration policies — such as those that promote forestation and discourage deforestation — should be part of the domestic portfolio of compliance activities.

The costs of carbon sequestration will 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 (Marland 1988; and Lashof and Tirpak 1989), and claims have been made that such forestry-based carbon sequestration is a relatively inexpensive means of addressing climate change (Sedjo and Solomon 1989; and Dudek and LeBlanc 1990). 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 U.S. Forest 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.[2]

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

[1] After fossil-fuel combustion, deforestation is the second largest source of carbon dioxide emissions to the atmosphere. Estimates of annual global emissions from deforestation range from 0.6 to 2.4 billion metric tons of carbon for the period from 1989 to 1995, compared with 5.7 to 6.9 billion tons of carbon annually from fossil-fuel combustion and cement manufacturing from 1989 to 1998 (Oak Ridge National Laboratory 2004).

[2] For surveys of carbon sequestration cost analyses, see: Sedjo *et al.* (1995); Richards and Stokes (2004); Stavins and Richards (2005); and Manley, van Kooten, and Smolak (2003).

land, and sorting these in ascending order of cost. In the earliest and simplest of these (Moulton and Richards 1990; Dudek and Leblanc 1990; and New York State 1991), 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.[3] 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 (Richards, Moulton and Birdsey 1993; and Richards 1997).

A carbon sequestration program may increase agricultural land prices, thereby leading landowners to convert unregulated forest lands to agricultural land and offsetting some of the effects of a carbon sequestration program. Alig *et al.* (1997) 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.[4]

A relatively small number of studies have employed 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 (Stavins 1999; Plantinga, Mauldin, and Miller 1999; and Newell and Stavins 2000).[5] 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 (Stavins 1999): (1) land-use changes can involve irreversible investments in the face of uncertainty (Parks 1995), and landowners may want to retain options for future land-use decisions (Pindyck 1991); (2) there may be non-pecuniary returns (for example, esthetics and recreation) to landowners from forest uses of land (Plantinga 1997), as well as from agricultural uses; (3) liquidity constraints or simple "decision-making inertia" may mean that economic

[3] In a variation on this theme, Parks and Hardie (1995) substituted the estimated foregone net revenues from agricultural production for observed sale and rental prices of agricultural land.

[4] Alig *et al.* (1997) employ the Forest and Agricultural Sector Optimization Model (FASOM). Adams *et al*. (1993) use a precursor to FASOM to develop marginal cost estimates for carbon sequestration in the U.S. Later applications of FASOM are found in Adams *et al*. (1999), McCarl and Schneider (2001), Murray *et al*. (2004). A different simulation model, the ERS U.S. Agricultural Sector Model (USMP), is used by Lewandrowski *et al*. (2004) to estimate the costs of afforesting crop and pasture lands and of other agricultural-sector sequestration activities. Our literature review above focuses exclusively on analyses of carbon sequestration in the United States, and does not include studies that provide global or regional estimates of carbon sequestration costs in other parts of the world.

[5] For econometric applications focused on sequestration from changes in agricultural practices, see Antle *et al*. (2003) and Kurkalova *et al*. (2003).

incentives affect landowners only with some delay; and (4) 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 the 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).[6] 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 that are implicit with simpler logit models. The estimated parameters and the 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

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[6] The CRP, established by the Food Security Act of 1985, is the largest U.S. government program targeting land use. The program offers annual rental payments to landowners retiring environmentally-sensitive land from crop production under 10 to 15-year contracts and currently enrolls 34.7 million acres, nearly equal to the total area of Iowa (USDA 2004).

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 analysis; this relationship is reversed, however, at higher levels of sequestration.

We also compare our estimates of the marginal costs of carbon sequestration with estimates of costs of energy-based carbon abatement. Here 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 U.S. climate change strategies. For example, if emission reductions 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) and with the type of policy incentive considered in this paper, our analysis implies that 33% to 44% of the reductions could be met cost-effectively through forest-based sequestration.

In section 2 of the paper, we describe our econometric analysis of land use. In section 3, we turn to the simulation model, and describe the approach we use to introduce incentives for carbon sequestration and to establish a baseline sequence of land-use transitions. In section 4, we present our empirical results, including simulated patterns of land use at the national level under both baseline and policy scenarios, the latter estimated for various hypothetical subsidy/tax rates. We derive estimated carbon flows resulting from the simulated land-use changes, and the associated supply function for carbon sequestration, examine the sensitivity of our results, and compare our results with those from previous studies. In section 5, we conclude.

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.[7] 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.[8] Given these simplifying assumptions, the decision rule that emerges from

[7] If net returns and the costs of converting land between different uses are approximately linear in land quantity, the size of parcels will not affect the relative profitability of land-use options, in which case land-use decisions for a heterogeneous parcel can be treated as the sum of land-use choices on constituent uniform-quality parcels.

[8] Landowners would expect current values to persist over time if land-use net returns follow a random walk. Limited empirical evidence suggests that this is approximately true (Schatzki 2003). For simplicity, we model land-

the related dynamic optimization problem (Lubowski 2002) 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, the owner of a unit of land in use j $(j=1,\dots,J)$ will choose use k at time t if:

$$R_{kt} - rC_{jkt} > R_{jt}, \tag{1}$$

for all alternatives k $(k=1,\dots,K)$, where R_{jt} represents the instantaneous expected net benefits from a unit of land in use j at time t, C_{jkt} is the expected marginal cost of converting one unit of land from use j to use k at time t $\left(C_{jjt} = 0\right)$; and r is the discount rate.

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 this 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 $(s=1,\dots,S)$, we specify U_{ijkt}, the landowner's utility from converting land parcel i from 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. 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.[9] 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:

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use choices as if landowners ignore the value of information to be gained by delaying irreversible decisions. More generally, we assume that landowners do not account for future conversion possibilities when evaluating alternative land uses in each time period. Given the national scope of our study, it would be impossible to model explicitly the stochastic dynamic optimization problem faced by landowners (Rust 1987). However, our econometric approach incorporates option values indirectly. If fixed costs and other irreversible choices introduce rigidities in land use, these will be reflected in the estimated coefficients.

[9] Assumptions of the nested logit model imply that the two disturbances are independent and that their sum has the Gumbel distribution (Ben-Akiva and Lerman 2000).

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

where τ_{st} are parameters, and $I_s = \ln \sum_{l=1}^{J_s} \exp(V_{ijkt})$. This "inclusive value" for nest K_s equals the expected utility for the choice of alternatives within a nest. The expression in equation (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.[10] 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 of the suitability of land for producing crops (USDA 1973). 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.[11] 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.[12] 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. We specify the component of utility that is unique to each alternative as:

$$V_{ijkt} = \alpha^0_{jkt} + \alpha^q_{jkt} LCC^q_{it} + \beta^0_{jkt} R_{kc} + \beta^q_{jkt} LCC^q_{it} R_{kc} + \varepsilon_{ijkt}, \tag{4}$$

where α^0_{jkt} is an alternative-specific intercept, $\alpha^q_{jkt}, \beta^0_{jkt}$, and β^q_{jkt} are parameters, R_{kc} are net returns to use *k* in county *c*, and LCC^q_{it} is a dummy variable indicating whether parcel *i* is of

[10] Land quality is only one potential determinant of substitutability among land uses. To the extent that farmers operate joint crop and livestock operations, farmers may have skills for pasture and range uses — rather than forestry, for example — so crops, pasture, and range uses may be closer substitutes for each other than for other uses. On the other hand, forest and pasture are similar in terms of lower labor requirements.

[11] Assigning values 1 through 4 to LCCs I-II, III-IV, V-VI, and VII-VIII, respectively, the NRI indicates that the average qualities in the contiguous U.S. over 1982-1997 were 1.5 for cropland, 1.9 for CRP, 2.0 for pasture, 2.2 for urban land, 2.7 for forests, and 3.0 for range, with lower values indicating higher quality.

[12] In the case of CRP, program rules restrict which lands are eligible to participate. Notably, eligibility is limited to lands that were planted to an agricultural commodity for four of the previous six crop years.

quality q at time t.[13] 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 (4), 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. Equation (4) thus becomes $V_{ijkt} = \alpha^0_{jkt} + \alpha^q_{jkt} LCC^q_{it} + \varepsilon_{ijkt}$ for *k=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:

$$V_{ijst} = \gamma^0_{jst} + \gamma^q_{jst} LCC^q_{it} + \tau_{st} I_{ijst}\,, \tag{5}$$

where γ^0_{jst} is a constant specific to nest s and γ^q_{jst} is a coefficient on the land quality indicators. Substituting equations (4) and (5) into (2) yields a complete nested logit model for estimation.[14]

2.2 *Econometric Estimation and Results*

We estimate the model using repeated observations of non-Federal land use from the National Resources Inventory (NRI) of the U.S. 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 five-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 (Nusser and Goebel 1997). 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.[15] 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

[13] The eight LCCs are merged into four groups: I-II, III-IV, V-VI, and VII-VIII.

[14] For each set of dummy variables (land quality, alternatives, and nests), one category is omitted to permit identification. Another alternative-specific dummy in each nest is dropped in equation (4) to identify the nest-specific dummies in equation (5). The dummy for the urban choice enters only in equation (5).

[15] Public lands and transportation infrastructure are excluded from the analysis, as changes in these uses are not affected directly by utility maximization by private landowners. 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," because data to measure net returns to these uses are not available.

contiguous United States. Further details on the NRI data are provided in Appendix A.

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 that affect 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 five-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 3,014 counties in the 48 contiguous states (see Appendix A for details). 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).[16] We estimate versions of the econometric model using data from two different time periods and then compare the results. First, we estimate our 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 our 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

[16] 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.

observation of each land unit.[17] 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 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.[18] For brevity, Table 2 reports elasticities, rather than estimated parameters for the twelve equations.[19] For each starting use, we report the estimated elasticity for 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.[20]

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.

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

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17 In the estimation, we weight the observations to account for whether a particular point remains in the same land use for one, two and three transition periods. We also weight observations using the NRI's sampling weights and, to avoid artificially shrinking the standard errors, scale all weights to sum to the total number of observations.

18 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 significant estimates of the inclusive value parameters together with tests (Hausman and McFadden 1984) of a simpler, 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 logit specification.

19 Parameter estimates are reported in Lubowski (2002), and are available from the authors upon request.

20 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.

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 (McMillen 1992). 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 (1999), 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.[21]

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:

$$P_{j1} = f(\hat{\beta}_{j1}, NR_1 + Z, \mathbf{NR}_{-1}) \tag{6a}$$

$$P_{11} = f(\hat{\beta}_{11}, NR_1, \mathbf{NR}_{-1} - \mathbf{Z}) \tag{6b}$$

$$P_{1j} = f(\hat{\beta}_{1j}, NR_1, \mathbf{NR}_{-1} - \mathbf{Z}) \tag{6c}$$

$$P_{jk} = f(\hat{\beta}_{jk}, NR_1 + Z, \mathbf{NR}_{-1}) \tag{6d}$$

for all *j* and *k* not equal to 1, where $\hat{\beta}$ 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

[21] 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 ten to fifteen years.

Z is a conformable vector with each element equal to *Z*. In general, *Z* has the effect of increasing the probability that land transitions into forest and diminishing the probability that it transitions 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 (Stavins and Richards 2005).

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 (Nordhaus and Boyer 2000; and Goulder and Mathai 2000). 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 (van‘t Veldt and Plantinga 2005). 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 equation (6) 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 (6a) and all transitions between non-forest uses (6d). 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 (6b and 6c).

Fourth and finally, the modified transition probabilities in (6) 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 (6c) to zero. We also compute a baseline sequence of land-use transitions by setting *Z* in (6) equal to zero. Naturally, when we compute 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. When we conduct the policy simulations, we assume that 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.

The carbon sequestration supply function is derived using the procedure discussed by Stavins (1999). 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 midpoint of each five-year interval. Likewise, we compute cumulative carbon flows relative to the baseline and the present value of carbon sequestered.[22]

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 (moving 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 to 1997, and the other for the period, 1992 to 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.

[22] See Richards and Stokes (2004) for an assessment of alternative approaches to intertemporal carbon accounting in carbon sequestration cost studies.

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 (Figure 1). Under the baseline scenario, the largest change is in the area of urban land, which increases from 76 million acres to 495 million acres during the 250-year simulation period (Figure 1). The areas of land in agricultural uses (crops, range, pasture, and CRP) decline throughout the simulation period, with cropland experiencing the greatest absolute losses. 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 (Figure 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.[23] 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.[24] Figure 3 portrays carbon flows relative to the baseline for selected

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[23] This is true except at low levels of the subsidy/tax, where the increments get larger as the subsidy/tax increases. As discussed below, this results in our supply function having a convex portion.

[24] In the contiguous United States, the equilibrium level of carbon in cropland soils ranges from a minimum of 17 tons per acre to a maximum of 36 tons per acre (the mean is 24 tons per acre). Pasture, range, and CRP lands store more carbon than cropland. The mean equilibrium storage on these lands is 34 tons per acre, with a range from 22 to 48 tons per acre. Forests achieve much higher equilibrium levels of carbon storage. If never harvested, mature forests in the contiguous United States reach levels of carbon storage ranging from 86 to 355 tons per acre, with a mean of 140 tons per acre. Harvested forests store less carbon. The average forest stand harvested on a 30-year rotation will hold 61 tons of carbon at the start of the rotation and 77 tons by the end of the rotation. Harvesting releases only some of the carbon in the forest. Between 20% and 45% of the carbon in the merchantable portion of trees is sequestered long term in wood products and landfills.

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 pasture and rangeland, producing an immediate and negative flow of carbon (see Appendix C).

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, and so there is a more even pattern of harvest and regrowth.[25]

The land-use change and carbon flow results are combined to produce a marginal cost function for forest 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 need to be carried out over 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.[26] This is portrayed in figure 4, where the solid line indicates the supply function for our basic scenario (where the harvesting of wood products is allowed).

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), the logistic is a convex function of its arguments — net returns and land quality. As forest net returns increase in the policy scenarios,

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[25] For most of the simulation, the flows trend upward, due largely to the addition of new forests over time, but after about 200 years, this trend is reversed. There are two reasons for this. First, throughout the simulation a large amount of carbon is stored in solid wood products and gradually released over time. This negative flow is offset for most of the simulation by the addition of new forest lands which generate positive flows, but once forest area stabilizes, the negative carbon flows from the product pools become apparent. The second explanation is associated with forest lands that are never harvested (see Appendix C). The carbon in unharvested forests reaches a maximum level after 12 decades. The zero flows in unharvested forests result in a diminished aggregate carbon flow. At the end of the simulation the flows drop to lower levels for the highest subsidy/tax rates. At higher rates, the area of forest more quickly reaches its highest level. As a result, the negative flows from the product pools become apparent sooner, and zero flows in unharvested forests occur earlier.

[26] It should be noted, however, that the results from both sets of econometric estimations and related simulations are similar within the range of the data, that is, with marginal costs of carbon sequestration that are within the range of historical experience of variance in timber returns.

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.[27]

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.[28] 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 (Figure 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 social rates of discount. 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.

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[27] The convexity of the marginal cost curve at higher marginal costs is not due to passing the inflection point in the logistic probability curve. At higher subsidy/tax levels, most of the land gets converted to forest in the earliest periods, leaving progressively less land to be converted in subsequent periods. The constraint on available land leads to decreasing incremental increases in total cumulative net forestation (and carbon sequestration), despite the fact that net forestation in the first period increases at an increasing rate for all of the subsidy/tax levels considered.

[28] With harvesting, some carbon is fixed in solid wood products and landfills for long periods of time. As a result, it is possible for the total amount of carbon sequestered to be greater under harvesting than no-harvesting regimes (Stavins 1999).

4.3 *Comparison with Results from Other Studies*

We compare our marginal cost estimates with those from previous studies of carbon sequestration in forests (Figure 5) and carbon abatement (Figure 6).[29] Over most of the range of carbon prices considered in previous forest carbon sequestration studies, our cost estimates are higher than those obtained using optimization models (Adams *et al*. 1993; and Callaway and McCarl 1996) and bottom-up engineering cost methods (Richards *et al*. 1993). Comparing our estimates with those from the earlier econometric study by Stavins (1999), 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 U.S. farm acreage totaling 551 million acres, including cropland and pasture (422 and 129 million acres, respectively). In the present study, crop and pasture acreage are somewhat lower, but rangeland is included, yielding a total of 933 million acres of privately-owned agricultural land (cropland, pasture, rangeland, and CRP) available for conversion to forest. As indicated in figures 1 and 2, a substantial amount of the increase in forest area in our simulations is due to conversion of rangeland to forest. Stavins' (1999) econometric model was fitted to data exclusively of transitions between cropland and forest.[30]

Finally, we compare the marginal costs of forest carbon sequestration with estimates of energy-based carbon abatement.[31] 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.

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 U.S. climate change strategies. Consider, for example, a domestic program that would bring U.S. annual greenhouse gas emissions 7% below 1990 levels over the period, 2008 to 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.[32] 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

[29] The marginal cost estimates from the previous studies are reported and discussed in Stavins (1999). For a synthesis of forest carbon sequestration marginal cost estimates, see: Stavins and Richards (2005).

[30] An additional explanation for the difference is that the 36 counties examined in the Stavins (1999) study may simply not be representative of the United States along relevant dimensions.

[31] The abatement cost estimates are derived from estimates in the Energy Modeling Forum (1995). See Stavins (1999) for details.

[32] Estimated U.S. net greenhouse gas emissions in 1990 were 1,555 million short tons of carbon equivalent (US EPA 2004). Projected 2010 energy-based carbon emissions are about 1,972 million short tons (US DOE 2004) and projected other greenhouse emissions minus sequestration removals are 47 million short tons of carbon equivalent (US DOS 2002).

Figure 6).[33] Our analysis indicates that if cost-effective emission reductions[34] 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 Figure 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 the estimated carbon sequestration supply function that resulted 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 such factors as option values associated with delaying irreversible land conversion, liquidity constraints, and unobserved benefits and costs of alternative land uses. Our results are consistent with previous econometric estimates that were based on more limited data sets.

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 U.S. target under the Kyoto Protocol would be cost-effectively achieved by employing forest-based sequestration policies, in addition to energy-based carbon abatement strategies. At a minimum, forest-based carbon sequestration merits consideration as part of a cost-effective portfolio of domestic U.S. climate change policies.

[33] A quadratic form was chosen so as to minimize the Akaike (1974) information criterion (AIC).

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

Table 1. Summary Statistics

Variables	Level	Mean, 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)	0.153) (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	1,946 (1,946)	2,389 (2,389)	2,349 (2,349)	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 five-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 (Greene 2000). * and ** denote significance at the 5% and 1% levels.

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

Figure 1. The Area of Land by Use in the Baseline Scenario

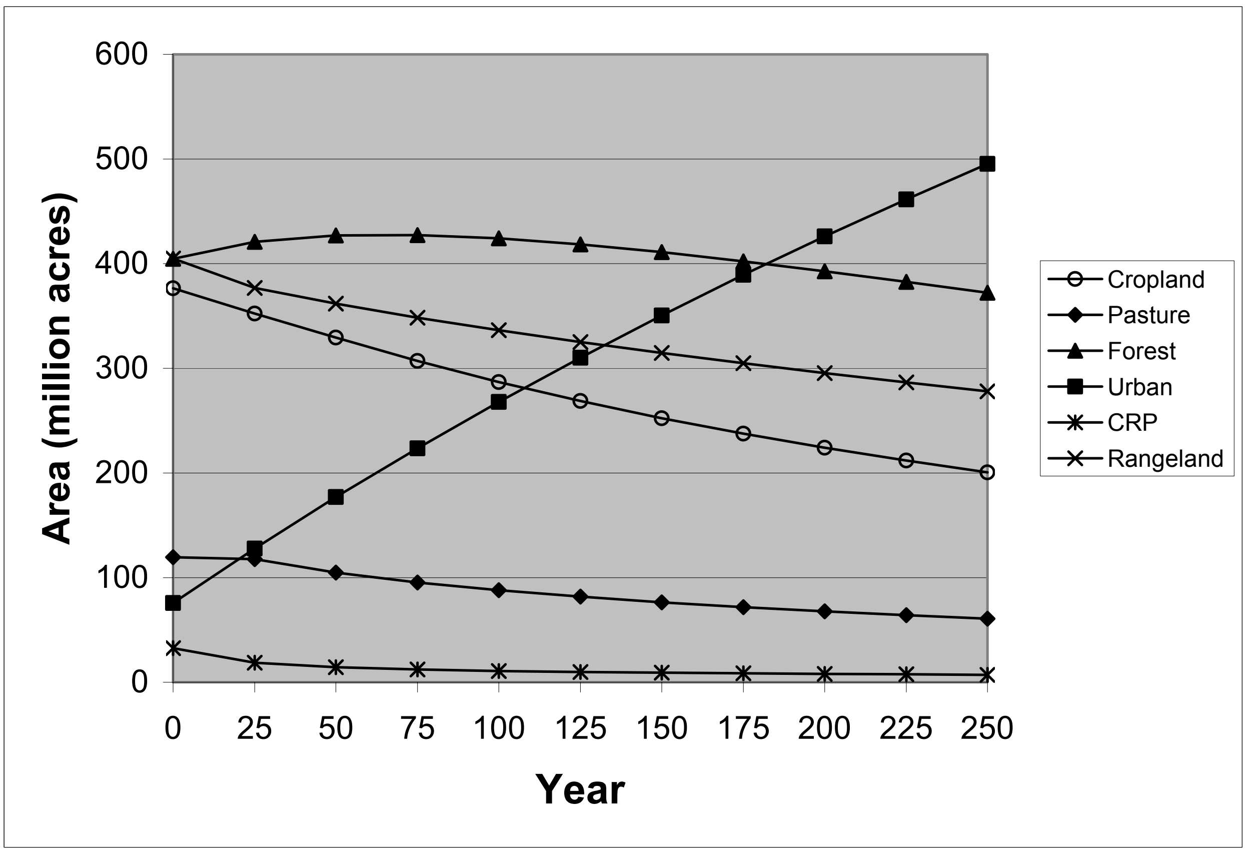


Figure 2. The Area of Land by Use with a $100 Per Acre Subsidy/Tax

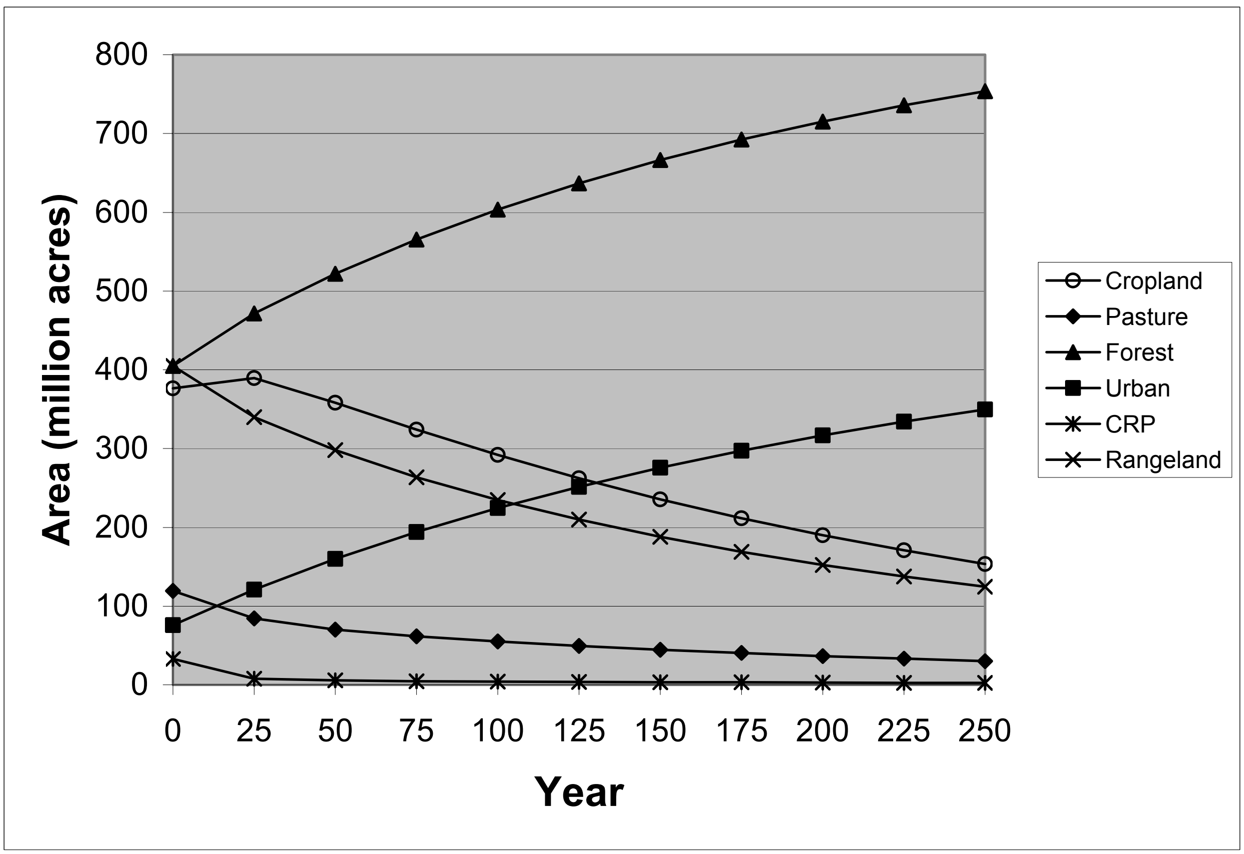


Figure 3. The Flow of Carbon Relative to the Baseline with Different Subsidy/Tax Rates

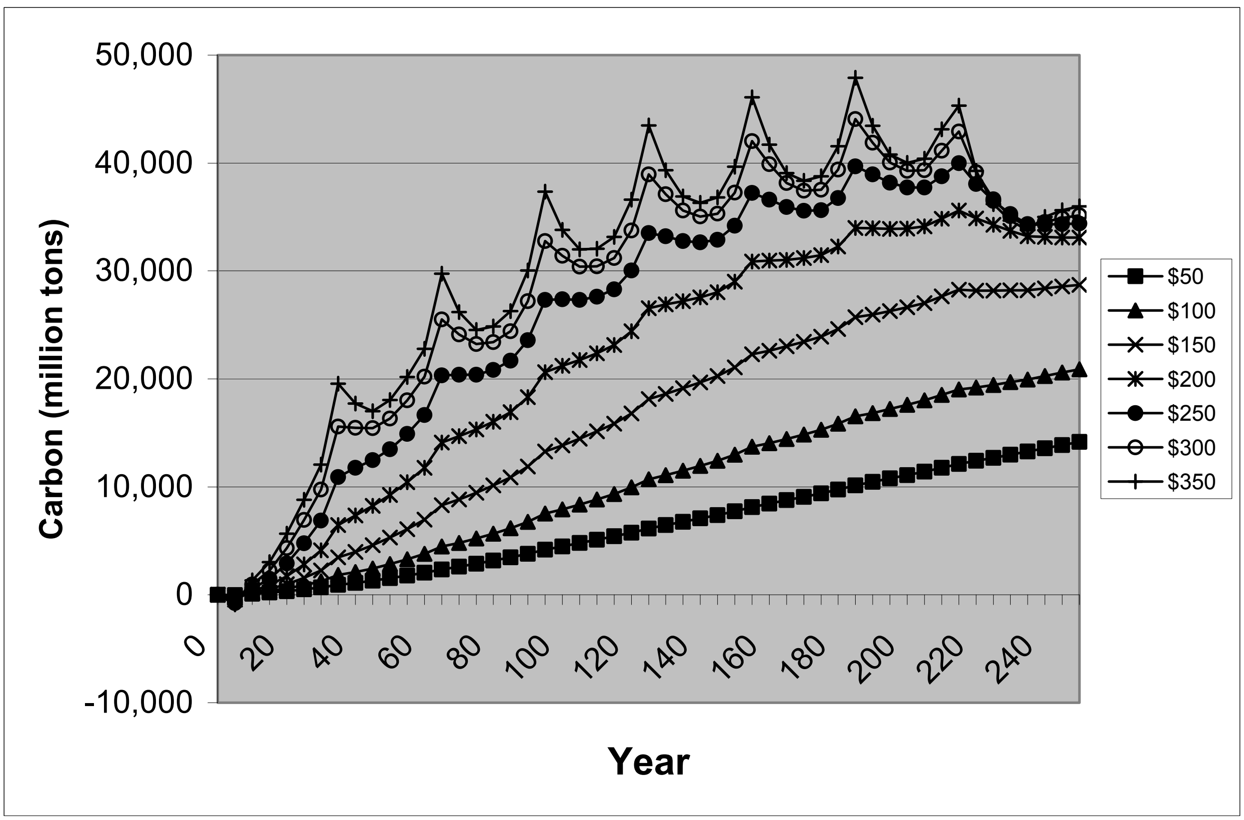


Figure 4. The Marginal Costs of Forest Carbon Sequestration in the United States

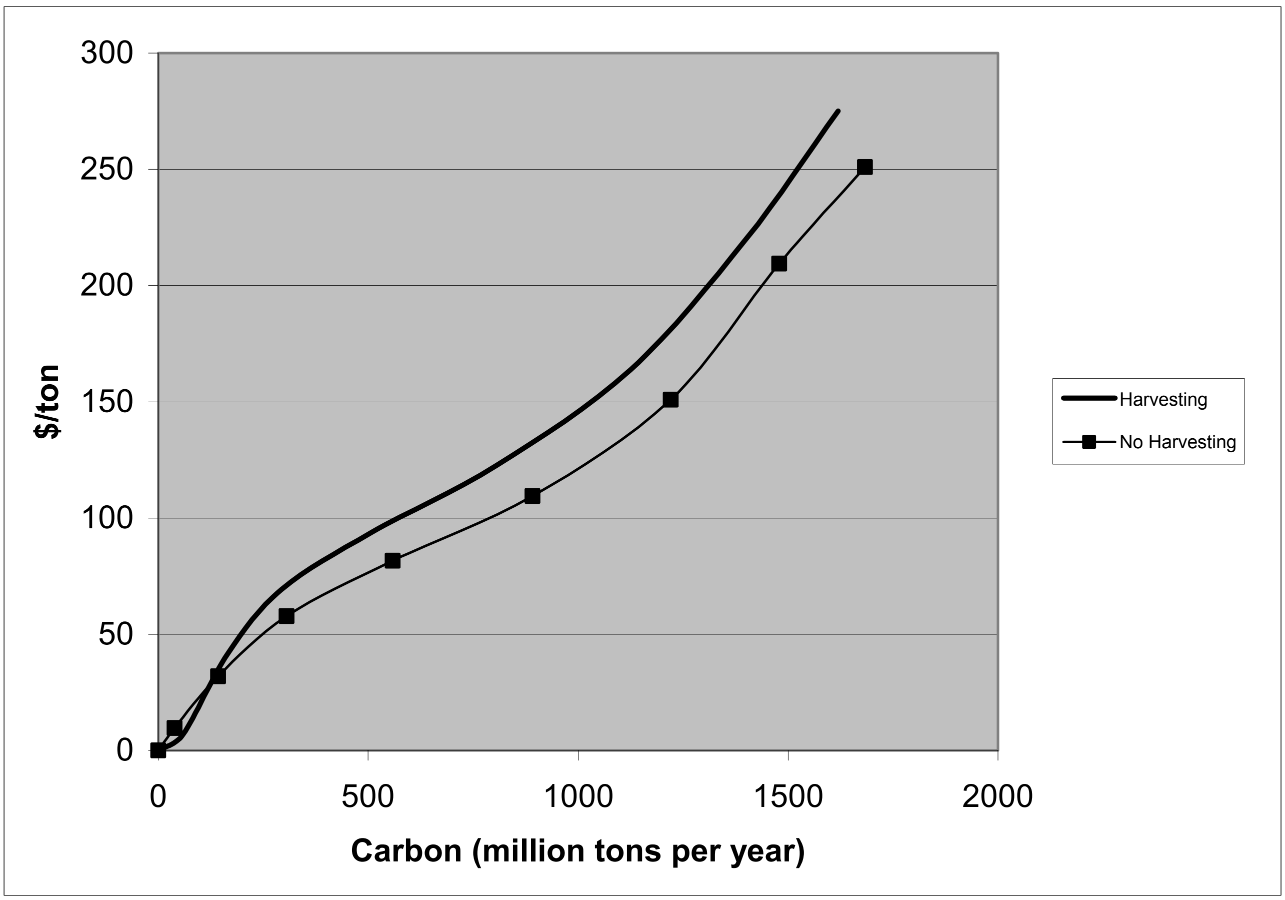


Figure 5. Alternative Estimates of the Marginal Costs of Forest Carbon Sequestration in the United States

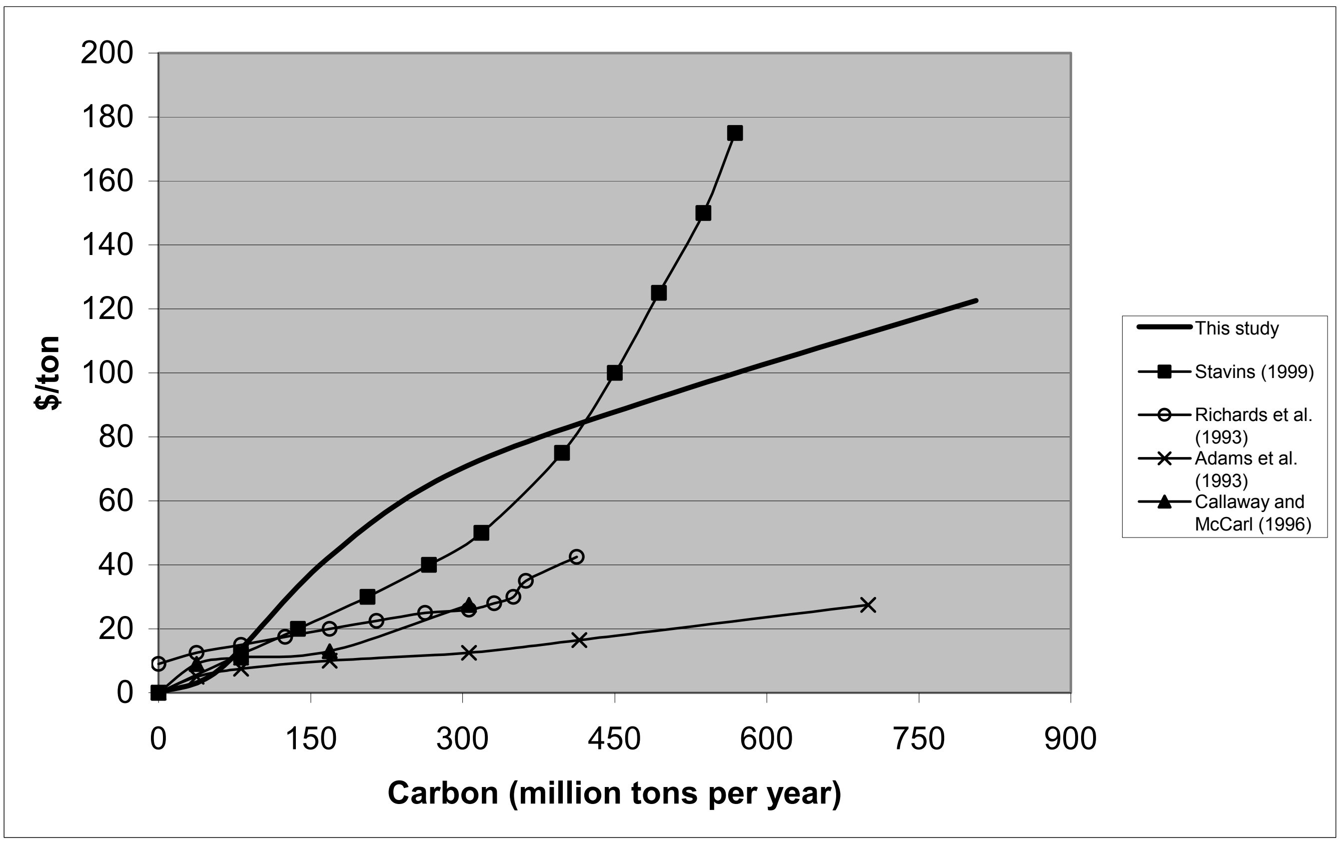
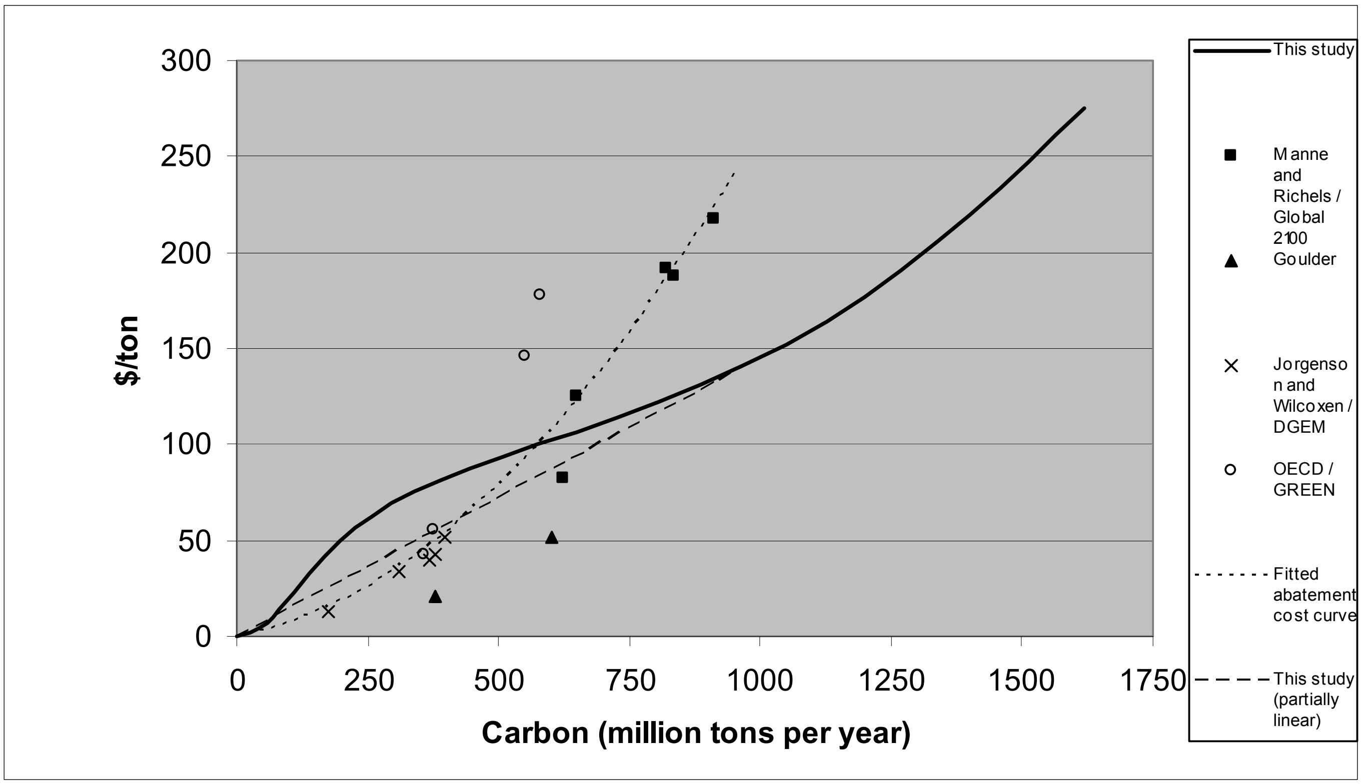


Figure 6. Estimates of the Marginal Costs of Forest Carbon Sequestration and Carbon Abatement in the United States [a]



[a] The abatement cost estimates are derived from Energy Modeling Forum (1995). See Stavins (1999) for details.

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Appendix A: Data Description [35]

Land Use: Land-use data is from USDA's National Resources Inventory (NRI). "Crops" include row and close-grown crops, fallow, haylands, pasture in rotation with crops, vineyards, orchards, and nurseries. "Pasture" includes land managed for introduced forage. "Range" includes land with native or introduced forage suitable for grazing that receives only limited management. "Forests" are lands at least one acre in size and 100 feet in width that are least 10% stocked with trees with the potential to reach 13 feet at maturity (equivalent to a canopy cover of at least 25%). "Urban" areas include land in residential, industrial, commercial, or institutional uses, as well as parcels below ten acres, such as small parks, that are also completely surrounded by urban lands. This definition excludes roads and other lands used for transportation in non-metropolitan areas, as these are separately identified by the NRI.

Cropland Net Returns: Estimated annual cropland net returns per acre consist of two components: a weighted average of the net returns for 21 major crops and total Federal farm program payments, excluding payments from the Conservation and Wetlands Reserve Programs. We used state-level marketing-year-average prices and county-level yields from the National Agricultural Statistics Service (NASS) for 21 major crops. Cash costs at the state and regional levels, respectively, are from the Census of Agriculture and Economic Research Service (ERS). County acreages from NASS and the Census of Agriculture provided weights. County-level estimates of total Federal direct farm program payments per acre are from the Census of Agriculture and include receipts from deficiency payments, support price payments, indemnity programs, disaster payments, and soil and water conservation projects.

Pasture Net Returns: Annual net returns per acre for pasture were estimated using yields from the National Cooperative Soil Survey (NCSS) times the state price for "other hay" from NASS minus costs for hay and other field crops from the Census of Agriculture.

Range Net Returns: Annual net returns per acre for rangeland were estimated with forage yields from NCSS multiplied by state-level grazing rates reported by ERS.

Forest Net Returns: Annual net returns per acre from forestry were constructed by annualizing at a 5% interest rate the estimated net present value of a weighted average of sawtimber revenues from different forest types. State-level stumpage prices were gathered from state and Federal agencies and private data services. Regional timber yields for different forest types were obtained from Richard Birdsey of the U.S. Forest Service. Regional replanting and management costs were derived from Moulton and Richards (1990). An infinite stream of timber revenues for each forest type was estimated using the optimal rotation age from the Faustmann formula. County acreage and sawtimber output data from the Forest Inventory and Analysis (FIA) and Timber Product Output (TPO) surveys of the U.S. Forest Service provided weights for averaging across individual forest types and species, respectively.

Urban Net Returns: Annual urban net returns were estimated as the median value of a recently-developed one-acre parcel used for a single-family home, less the value of structures,

[35] Further descriptions of data sources are provided in Lubowski (2002) and are available upon request.

annualized at a 5% interest rate. Median county-level prices for single-family homes were constructed from the Census of Population and Housing Public Use Microdata Samples and the Office of Federal Housing Enterprise Oversight (OFHEO) House Price Index. Regional data on lot sizes and land value relative to structures were from the Characteristics of New Housing Reports (C-25 series) and the Survey of Construction (SOC) micro-data from the Census Bureau.

Appendix B: Policy Simulation Model

The policy simulation model is used to estimate land-use changes that occur in response to incentives for carbon sequestration. The simulation is conducted at the NRI point level, indexed by *i*, beginning in 1997, where $t = \{0,1,...\}$ references the start times of five-year intervals. Based on the sampling design, each point is associated with a certain number of acres. We define A_{ijt} as the number of acres associated with point *i* in use *j* at time *t*. In the initial period, each point is in one of the six uses as indicated in the 1997 NRI data. Thus, A_{ij0} equals the acres represented by point *i* if the point is in use *j* in time 0, and equals 0 otherwise. Given a sequence of transition probabilities, we can estimate how the land will be distributed across the six use categories at each time in the future. As per equation (2), the probability that land represented by point *i* transitions from use *j* to *k* during the interval beginning at *t* is denoted P_{ijkt}.

We express the land at point *i* that converts from use *j* to *k* during the interval beginning in *t* as $TA_{ijkt} = P_{ijkt} \cdot A_{ijt}$. The acres represented by point *i* in use *j* at time *t*+1 are given by,

$$A_{ijt+1} = \sum_k TA_{ikjt} = \sum_k P_{ikjt} \cdot A_{ikt} \ , \tag{B.1}$$

reflecting the first-order Markov structure of the model.

The transition probabilities in the initial period (*t*=0) are estimated with the net returns observed in 1997 and a given value of *Z*. With the initial acres A_{ij0}, we can estimate the acres in each use in period 1 using (B.1). The induced change in land use implies a change in the supply of land-based commodities and services and, hence, changes in related prices and net returns. We model these endogenous price effects for forest and cropland, and assume that net returns to pasture, range, urban, and the CRP remain constant in the simulation. We are justified in ignoring price effects for pasture and CRP, because these uses represent small and declining shares of the land base in the baseline scenario. This implies that price adjustments would have small effects on the total flow of land into forest under the policy. Range is a major component of the land base (about 30% initially), but we were unable to find any data on markets (specifically, demand elasticities) for forage, the principal rangeland output. The probabilities for transitions into urban uses were found to be insensitive to changes in urban net returns,[36] indicating that these probabilities would tend to remain the same with endogenous price effects.

Consistent with the model of landowner behavior underlying the econometric analysis, crop and forest commodities are supplied inelastically. Thus, we can use crop and timber yields

[36] While urban net returns had a small impact on the probabilities of conversions to urban uses, this impact far exceeded the effects from changes in net returns to the other land uses.

(per acre), to translate land-use changes into output changes. After aggregating output changes appropriately, corresponding price changes are computed using own-price demand elasticities estimated in previous econometric studies.[37] Changes in cropland area result in immediate changes in crop output, since crops are assumed to be harvested in the year they are planted. In the case of forests, timber harvests will be delayed for a period of years while the forest stand matures. We assume harvests on afforested lands are delayed for one optimal rotation period, after which time the forest is "fully regulated" and provides a constant annual timber flow.[38] All land originally in forest (at t=0) is also assumed to be fully regulated.[39] When these lands are converted to non-forest uses, we assume that 20% of the timber is merchantable.

After computing the price changes resulting from the land-use changes between periods 0 and 1, we form new measures of net returns in period 1. We apply the national or regional percentage price change to the county-level prices used to compute net returns. With the period 1 net returns, we recalculate the transition probabilities and repeat the procedure. This stage of the simulation ends when the crop and forest net returns have converged (that is, period-to-period changes in prices are near zero). The converged net returns are equilibrium values that reflect all of the anticipated supply adjustments in agricultural and forest commodity markets. This process is atemporal, since it represents an instantaneous adjustment to the new market equilibrium after the introduction of the subsidy/tax. For this reason, we hold urban land constant during this stage of the simulation. Urban land will increase over time with factors such as population growth — as it does in the second stage of the simulation discussed below — but should not affect the immediate adjustment in net returns to cropland and forests.

In the second stage of the simulation, we compute the time path of land-use changes. Specifically, we recalculate the transition probabilities for the initial period using the converged net returns for cropland and forests, the associated value of the subsidy/tax, and the observed net returns for the other uses. Beginning with the initial acres in each use (A_{ij0}), we use (B.1) to compute the sequence of land-use transitions (TA_{ijkt}) through time. Unlike in the first stage of the simulation, the net returns remain at their equilibrium values throughout this stage.

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[37] For crop commodities, we use a national-level demand elasticity for raw food inputs by food processors (Goodwin and Brester 1995). We apply this elasticity (-0.661) separately to each of the twenty-five crop commodities in our model. For timber, we use demand elasticities for seven timber production regions — Pacific Northwest (-0.300), Pacific Southwest (-0.497), Rocky Mountains (-0.054), North Central (-0.141), Northeast (-0.029), South Central (-0.193), Southeast (-0.285) — from the Timber Assessment Market Model (Adams and Haynes 1980). These elasticities apply to a composite timber type representative of the species found within the region. In general, we would expect the mixes of crop and timber types to change in response to price changes. However, we assume for simplicity that the crop and timber type shares remain constant over time.

[38] Specifically, if t^* is the optimal rotation length, then there is an equal area of forest in each age category in the interval $[0, t^*]$. Each year, timber of age t^*, or $1/t^*$ of the forest area, is harvested.

[39] The fully regulated assumption overstates the timber supply from private forests, since non-industrial landowners, who own almost 80% of private forest in the U.S., frequently manage their lands for non-timber outputs such as recreation (Birch 1996). To account for these alternative objectives, we assume that a fixed percentage of forest land area in each timber production region is never harvested, while the remaining forest is harvested as described above. The no-harvest percentages are determined by calibrating the model to regional timber harvest data for 1997. The no-harvest percentages range from 6 to 62% in the South Central region and Pacific Southwest, respectively.

Appendix C: Carbon Sink Model

In a forest, carbon is stored in biomass (living components of trees and plants) and in the floor litter and soils. Timber harvests convert some of this carbon back to CO_2, but much of the carbon in the merchantable biomass is captured in wood products for decades after harvest. On agricultural lands, soils are the principal carbon sink, because little carbon in the harvested biomass is permanently sequestered. We assume that no carbon is stored in urban lands, though urban lands have trees, and some residual carbon may be stored in urban soils.

Carbon flows when land remains in the same use.

Agricultural soils become saturated with carbon if the land remains in the same use for long periods. For all land that begins and remains in agricultural use (cropland, pasture, rangeland, and CRP), we assume the soil carbon is at this equilibrium level, equal to the initial soil carbon levels for cropland and pasture converted to forest. We assume that range and CRP lands have the same equilibrium carbon levels as pasture.[40]

As discussed above, land that begins in forests is assumed to have a fully regulated structure.[41] For forests that are never harvested, carbon accumulates in the stand until it reaches an equilibrium level (assumed to be at 120 years). For harvested forests, carbon in the merchantable biomass is processed into primary wood products (e.g., paper, lumber, and plywood), then transformed into end-use products (e.g., newspapers, housing), and remains sequestered in end-use products or is disposed in landfills. Plantinga and Birdsey (1993) provide estimates (by regions and timber types) of the remaining share of merchantable carbon at the start of each decade after harvest. We assume the last share reported (at 50 years) applies in subsequent years. Except for soil carbon, all carbon in non-merchantable biomass (e.g., branches), understory vegetation, and floor litter is released as emissions at the harvest time. The soil carbon reverts to the level specified by Birdsey (1992) for a forest after a clearcut harvest. From there, carbon accumulates with new forest growth until the rotation age is reached again.

Carbon flows when land changes uses.

When land moves from cropland into another agricultural use (pasture, rangeland, and CRP), there is a gradual transition from the lower equilibrium soil carbon level to a higher level. Based on results from the Century model (Conant, Paustian, and Elliot 2000), the transition takes 40 years and is described by:

$$\Delta C_n^{CP} = \sum_{s=0}^{n} \left(0.125PER - 0.016\right) e^{-0.1s} \ , \qquad \text{(C.1)}$$

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[40] Estimates of equilibrium soil carbon for cropland and pasture are from Birdsey (1992), who provides estimates of carbon stored in forest stands for different regions, forest types, and previous land uses.

[41] We match the Birdsey (1992) forest carbon data to our land-use estimates, based on state-level forest composition. Thus, the carbon yield from each acre of forest land within a state is a weighted average of type-specific yields, where the weights equal the state-level forest type shares.

where ΔC_n^{CP} is the cumulative change in carbon *n* years following the land-use change and *PER* is the precipitation to evaporation ratio. State-specific values of *PER* were taken from the Century model database. When land moves from pasture, rangeland, or CRP to cropland, it is usually tilled in preparation for planting. This releases soil carbon, and we assume an instantaneous drop from the first soil carbon level to the next. Likewise, we assume an immediate loss of soil carbon when agricultural land is converted to urban use.

When agricultural land is converted to forest, carbon accumulates in the soils, floor litter, and biomass. To model these flows, we use the corresponding carbon yield tables from Birdsey (1992) for forests established on cropland and pasture. Timber harvests are handled as described above. When a forest is converted to cropland, pasture, or urban use, there is an immediate adjustment to the new equilibrium soil carbon level, again reflecting disturbances to the soils. Consistent with the policy simulation model, 20% of the merchantable biomass is assumed to be manufactured into wood products, while the carbon in the remaining merchantable biomass, non-merchantable biomass, and floor litter is immediately released as emissions.