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Economic Analysis of Agricultural Soil Carbon Sequestration: An Integrated Assessment Approach

John M. Antle, Susan M. Capalbo, Siân Mooney, Edward T. Elliott, and Keith H. Paustian

This study develops an integrated assessment approach for analysis of the economic potential for carbon sequestration in agricultural soils. By linking a site-specific economic simulation model of agricultural production to a crop ecosystem model, the approach shows the economic efficiency of soil carbon (C) sequestration depends on site-specific opportunity costs of changing production practices and rates of soil C sequestration. An application is made to the dryland grain production systems of the U.S. Northern Plains which illustrates the sensitivity of the sequestration costs to policy design. The marginal cost of soil C ranges from \$12 to \$500 per metric ton depending upon the type of contract or payment mechanism used, the amount of carbon sequestered, and the site-specific characteristics of the areas.

Key words: carbon sequestration, Century ecosystem model, econometric process model, economic efficiency, integrated assessment approach, marginal cost of soil C

Introduction

Over the last decade there has been a high level of interest in carbon (C) sequestration as an efficient means for offsetting greenhouse gas (GHG) emissions. Starting with Sedjo and Solomon and a report from the National Academy of Sciences, carbon sequestration has been suggested as a potentially low-cost means to reduce atmospheric concentrations of GHGs. The Kyoto Protocol to the United Nations Framework Convention on Climate Change added further impetus to carbon sequestration. If ratified, this agreement would require the United States and many other industrialized countries to reduce net emissions of GHGs 6–8% below 1990 levels by 2008–2012.

Agricultural soil C sequestration can be enhanced through changes in land use or changes in production practices. Changes in agricultural land use and management practices alone could potentially sequester between 75 and 208 million metric tons (MMT) of C per year in agricultural soils (Lal et al.). This represents approximately 5–12% of U.S. annual emissions of all GHGs. A single land use or management practice,

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John M. Antle is professor, Susan M. Capalbo is associate professor, and Siân Mooney is research assistant professor, all in the Department of Agricultural Economics and Economics, Montana State University; Edward T. Elliott is director, School of Natural Resource Sciences, University of Nebraska; and Keith H. Paustian is senior research scientist, Natural Resource Ecology Laboratory, Colorado State University.

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however, will not be effective at sequestering C in all regions. Lal et al. estimate approximately 49% of agricultural C sequestration can be achieved by adopting conservation tillage and residue management, 25% by changing cropping practices, 13% by land restoration efforts, 7% through land-use change, and 6% by better water management. Furthermore, if and when a producer reverts to conventional management, stored C is released.[1] These assessments are based on the technical potential for soils to sequester C; additional research is needed to demonstrate whether agricultural producers could sequester C at a cost competitive with emissions reductions or the use of other C sinks such as forests.

Previous economic studies of C sequestration in the United States have focused primarily on the conversion of cropland to forest land (Adams et al.; Parks and Hardie; Alig et al.; Pfaff et al.). Producers convert land to trees if they are compensated for the agricultural rents of the land, where the rents reflect regional or county-level estimates of net returns to agricultural lands. Plantinga, Mauldin, and Miller, and Stavins utilize county-level econometric land-use models to derive the costs of sequestering soil C associated with conversion of cropland to forested uses and vice versa. In these studies, the marginal cost of C sequestration increases as higher quality agricultural lands are converted to forested use. Within the agricultural sector, Pautsch et al., and Feng, Zhao, and Kling have addressed carbon sequestration costs associated with changes in tillage practices, notably the adoption of conservation tillage.

This study extends the economic literature on C sequestration by showing how the integrated assessment approach to analysis of agricultural production systems (Antle and Capalbo) can be used to estimate the marginal cost of sequestering C in soil. This approach to the analysis of soil C links biophysical data and models with economic data and models on a site-specific basis. In this way, the analysis can account for the spatial heterogeneity of biophysical conditions (soil C sequestration rates) and economic decisions (land use), and assess how these conditions interact to determine the marginal cost of sequestering C in soil.

We apply the integrated assessment approach to quantify the costs of sequestering C from changes in land use and management practices in the dryland grain production systems of the Northern Plains region of the United States. In this region, changes in land use such as conversion of cropland to permanent grass, and changes in management practices such as use of reduced fallow, may be economically feasible where afforestation—the conversion of nonforest land to forest—is not.

We compare the relative efficiency of sequestering soil C for two alternative policies relevant to the Northern Plains region: one that provides producers with payments for converting cropland to permanent grass (similar to the Conservation Reserve Program in the United States), and one that provides payments to farmers to switch from a crop-fallow rotation or permanent grass to a continuous cropping system. These policies are similar to ones proposed in recent U.S. legislation. Our analysis shows that the economic efficiency of C sequestration depends on site-specific opportunity costs of changing practices, the rates of soil C sequestration associated with changing practices, and the policy design. We conclude with an assessment of the competitiveness of agricultural soil C sequestration in the Northern Great Plains with industrial emissions reductions and forestry sinks in other parts of the United States.

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[1] Tiessen, Stewart, and Bettany; Mann; and Rasmussen and Parton estimate that 20–50% of soil C is lost from the soil during the initial 20 to 50 years of cultivation.

Measuring the Cost of Agricultural Soil C Sequestration

Assuming agricultural producers are initially employing those land use and management practices expected to yield the highest economic return, it follows that producers will adopt different practices for increasing soil C if and only if there is a perceived economic incentive to do so. While there are many possible ways to design policies to sequester soil C, we have adopted the basic structure of a soil C contract program, where soil C can be purchased by either the government or a private entity.[2]

Within a given region, let a contract pay the farmer g^{is} dollars per hectare per year for T years to change from management practice i to management practice s that sequesters additional soil C. Letting the total increase in soil C over the time period $t = 0$ to T from switching from i to s be $\Delta c^{is} = c_T^s - c_0^i$, the average increase is $\Delta c^{is}/T = c^{is}$ (metric tons per hectare per year). Although the time path for the increase in the stock of soil C in response to the adoption of improved practices is nonlinear, the path is often approximated linearly with the annual average rate of soil C increase (e.g., see the soil C rates discussed in Watson et al.). Furthermore, because it is not practical to measure soil C rates accurately on an annual basis, we assume these average annual rates are what would be actually measured and used in soil C contracts.

The per hectare capitalized value of the contract to the farmer to switch from i to s is specified as

$$(1)\qquad \sum_{t=1}^{T} g^{is}(1+r)^{-t} = g^{is}D(r, T),$$

where $D(r, T)$ denotes the present value of \$1 at interest rate r for T periods. The value of a C contract to the government or other purchaser of carbon depends on the soil C rate parameter (c^{is}) and the time period over which the practices are adopted. If the buyer of the carbon can sell the C for p dollars per metric ton, it follows that the value of the contract to the buyer is

$$(2)\qquad \sum_{t=1}^{T} pc^{is}(1+r)^{-t} = pc^{is}D(r, T).$$

The equivalence of (1) and (2) implies $g^{is} = pc^{is}$. If a program pays farmers g^{is} dollars per hectare per year for soil C sequestration, then the implicit price per metric ton being paid by the government or any other buyer of soil C is equal to g^{is}/c^{is}. Under the assumption of static price expectations for carbon, the payment per hectare per year to the farmer is equal to the value of the C sequestered per hectare per year. More generally, if prices are constant but the rate of increase in soil C varies with time, then it follows that $pk = g^{is}$, where $k = \Sigma_t c_t^{is}(1+r)^{-t}/D(r, T)$.

Producers will switch production practices if and only if the profits per hectare of their profit-maximizing practices are less than the alternative practices plus the payment per hectare. Let the total amount of agricultural land in a region be A hectares, and let the share of land in a given region that is entered into C contracts for switches from i to s be $z^{is}(g)$, where we have assumed $g^{is} = g\ \forall\ i, s$ which result in a positive amount of soil C accumulation, and $g^{is} = 0$ otherwise. This region would sequester $C(g) = T\,\Sigma_i\,\Sigma_s\,c^{is}z^{is}(g)A$

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[2] This is similar to schemes proposed in other C sequestration research (see Pautsch et al.; Feng, Zhao, and Kling; Plantinga, Mauldin, and Miller).

metric tons of C, or $C(g)/T$ metric tons of C per year. The region's marginal cost function for sequestering soil C, $M(C)$, can then be defined as the correspondence between p and $C(g)$.

When a producer switches to alternative practices as part of the program, the reduction in profitability, net of the payment, is the opportunity cost of entering into the contract. Given site-specific data on net returns, the opportunity costs differ across regions, and thus an economic production model of land-use choices is needed to determine the share of land to be entered into a specific type of contract as payment levels increase. An upward-sloping marginal cost curve for soil C in a region reflects the fact that different land units have different opportunity costs.

Given $M(C)$, the corresponding total cost can be calculated by integrating under the marginal cost curve adding any fixed transactions costs. Revenue generated by producers selling C contracts is equal to $R = pC(g)$, and the net benefit to producers is the usual producer surplus measure. In the case of a government payment program that pays farmers \$$g$ per hectare, the total cost to the government is revenue R.

Questions about saturation of soil C and the permanence of C stored in soil have been raised in the debate over the implementation of the Kyoto Protocol. Franzluebbers and Arshad suggest a new equilibrium for soil C is reached 20 to 30 years after a switch in production practices occurs. This means that over time the soil becomes saturated with organic C, and thus annual estimates of the amount of soil C that can be sequestered need to be associated with a time limit T. To avoid possible misinterpretation associated with annual measures, we present our results in terms of the quantities of soil C that can be sequestered over the time period T.

One way to address the permanence issue is to view farmers who enter into soil C contracts as providing a service in the form of accumulating and storing soil C, both of which depend on the farmer continuing to maintain the land use or management practices which make the accumulation possible. This implies a higher cost than if farmers only have to be paid during accumulation. Consequently, estimates of the cost of soil C can be sensitive to the assumed duration of the contract or government program, a fact to remember when we evaluate the competitiveness of soil C sequestration. Imposing penalties for failure to comply with the terms of the contract, including penalties for subsequent release of the C after the contract expires, would also have the effect of raising the cost of the soil C.

Integrated Assessment Approach to Modeling Soil C Sequestration Costs

The integrated assessment approach to assess the cost of agricultural soil C sequestration involves linking the output of two disciplinary models—an econometric-process simulation model and a crop ecosystem model—to quantify the responses of farmers to economic incentives to sequester soil C.

The econometric-process model, discussed below, simulates expected returns to alternative production systems on a site-specific basis, in response to incentives provided through a policy that pays farmers to change land use or management practices. These expected returns are used to simulate the farmer's choice of production system for a given land unit. This simulation model utilizes the stochastic properties of the economic production models and sample data, so its output can be interpreted as providing a statistical representation of the population of land units in a given region.

The crop ecosystem model provides estimates of the levels of soil C and productivity (yields) associated with each production system. Following the marginal cost presentation, simulated changes in production systems are combined with simulated changes in soil C to compute the implied marginal costs, government costs, and producer surplus associated with policies in given regions. Thus, the integrated assessment model provides answers to policy questions about the effects of different payment schemes on the quantity of carbon sequestered and the marginal cost of sequestering soil C, and how the costs vary spatially. This approach also provides a basis for estimating the value of using government-based carbon payments as a part of the policy options to offset greenhouse gas emissions.

Econometric-Process Model of Production System Choice

In previous work, an econometric-process model was developed to model a producer's intensive- and extensive-margin production decisions. The motivation for the development of the econometric-process approach was the need to link economic analysis of production systems to site-specific biophysical simulation models to assess the economic and environmental impacts of changes in policies, technologies, or biophysical conditions (Antle et al. 1999; Antle and Capalbo). Site-specific data are used to estimate the economic production models which are then incorporated into a simulation model that represents the decision-making process of the farmer as a sequence of discrete and continuous choices. This discrete/continuous structure of the econometric-process model is able to simulate decision making both within and outside the range of observed data in a manner consistent with economic theory and with site-specific biophysical constraints and processes.

The economic model is specified as follows: the production process of activity i at site j in period t is defined by a nonjoint production function $q_{ijt} = f_i(\mathbf{v}_{ijt}, \mathbf{z}_{ijt}, \mathbf{e}_{jt})$, where $\mathbf{v}$ is a vector of variable inputs, $\mathbf{z}$ is a vector of allocatable quasi-fixed factors of production and other fixed effects, and $\mathbf{e}$ is a vector of biophysical characteristics of the site (soils, topography, climate, etc.) (random terms are suppressed here for notational convenience). For expected output price p_{ijt}, the profit function is $\pi_{ijt} = \pi_i(p_{ijt}, \mathbf{w}_{ijt}, \mathbf{z}_{ijt}, \mathbf{e}_{jt})$. If a crop is not grown, the land is in a conserving use with a return of π_{hjt}. Define $\delta_{ijt} = 1$ if the ith crop is grown at site j at time t, and zero otherwise. The land-use decision on site j at time t is represented by

$$\max_{(\delta_{1jt},\ldots,\delta_{njt})} \sum_{i=1}^{n} \delta_{ijt}\pi_i(p_{ijt}, \mathbf{w}_{ijt}, \mathbf{z}_{ijt}, \mathbf{e}_{jt}) + \left(1 - \sum_{i=1}^{n} \delta_{ijt}\right)\pi_{hjt}. \tag{3}$$

The solution takes the form of a discrete step function:

$$\delta^*_{ijt} = \delta_i(\mathbf{p}_{jt}, \mathbf{w}_{jt}, \mathbf{z}_{jt}, \mathbf{e}_{jt}, \pi_{hjt}), \tag{4}$$

where $\mathbf{p}_{jt}$ is a vector of the p_{ijt}, and likewise for the other vectors. Using Hotelling's lemma, the quantity of the ith output on the jth land unit is given by

$$q^*_{ijt} = \delta^*_{ijt}\,\partial\pi_i(p_{ijt}, \mathbf{w}_{ijt}, \mathbf{z}_{ijt}, \mathbf{e}_{jt})/\partial p_{ijt} = q_{ijt}(\mathbf{p}_{jt}, \mathbf{w}_{jt}, \mathbf{z}_{jt}, \mathbf{e}_{jt}, \pi_{hjt}). \tag{5}$$

Variable input demands are likewise given by

$$v^*_{ijt} = -\delta^*_{ijt}\,\partial\pi_i(p_{ijt}, \mathbf{w}_{ijt}, \mathbf{z}_{ijt}, \mathbf{e}_{jt})/\partial\mathbf{w}_{ijt} = \mathbf{v}_{ijt}(\mathbf{p}_{jt}, \mathbf{w}_{jt}, \mathbf{z}_{jt}, \mathbf{e}_{jt}, \pi_{hjt}). \tag{6}$$

The econometric process approach combines the econometric production model represented by the supply and demand functions given in (5) and (6) with the process-based representation of the discrete land-use decision represented by (3) and (4). The model simulates the producer's crop choice, and the related output and costs of production at the field scale over time and space. This simulation structure utilizes the stochastic properties of the econometric models and the sample data, so its output is interpreted as providing a statistical representation of the population of land units in the region.

By operating at the field scale with site-specific data, the simulation can represent spatial and temporal differences in land use and management, such as crop rotations, giving rise to different economic outcomes across space and time in the region. Moreover, because of the detailed representation of the production system, the econometric-process model can be linked directly to the corresponding simulations of the crop ecosystem model to estimate the impacts of production system choice on soil C.

Each field in the sample is described by area, location, and a set of location-specific prices paid and received by producers, and quantities of inputs. Using sample distributions estimated from the data, draws are made with respect to expected output prices, input prices, and any other site-specific management factors (e.g., previous land use). The econometric production models are simulated to estimate expected output, costs of production, and expected returns. The land-use decision for each site is made by comparing expected returns for each production activity. These spatially and temporally explicit land-use decisions are combined with simulated outputs of the crop ecosystem model to assess changes in soil C.

Empirical Results for the Econometric Production Model

The econometric production model described above was estimated using cross-sectional data from a sample of 425 farms and over 1,200 fields for the 1995 crop year that are statistically representative of the USDA's major land resource areas (MLRAs) in the grain-producing regions of Montana (see figure A1 in the appendix). The major crops include spring wheat, winter wheat, and barley. Data and summary statistics are described in the appendix.

Crop rotations play a critical role in maintenance of soil quality and productivity, and affect soil C accumulation. The effects of crop rotations can be accurately modeled only on a site-specific basis, because their representation requires site-specific data on the history of land use. Aggregation across fields, even at the farm level, prevents the dynamics of soil quality from being accurately represented in both economic and biophysical process models.

The use of rotations and the length of rotations is an economic decision involving a tradeoff between the management and opportunity cost of the fallow and the productivity gains associated with the rotation. The productivity effects of crop rotations are included in the model by incorporating a dummy variable indicating if the previous use was a crop or fallow. If a unit of land was previously cropped, the decision to fallow in the current season with the intent to crop again next season is based on discounted net returns above variable cost for the next period minus the variable cost associated with fallow in the current period. Similar logic is applied to the case where a field was fallowed in the previous period. When the farmer uses continuous cropping (one crop follows another) there is no fallow cost, but productivity is reduced by setting the fallow dummy variable equal to zero.

The MLRAs were stratified into sub-zones (sub-MLRAs) based on high or low precipitation according to historical climate data. Log-linear production models for winter wheat, spring wheat, and barley were estimated using nonlinear three-stage least squares (NL3SLS). The parameter estimates are reported and discussed in Antle and Capalbo, and are also summarized in the appendix to this article.

The results of the econometric estimation indicate winter wheat, spring wheat, and barley yields are, respectively, about 31%, 23%, and 9% higher when the crop is grown after fallow; the variable costs of production are about 40% lower for all three crops grown on fallow compared to continuously cropped. Thus, while the crop-fallow system provides higher yields and lower average variable costs of production, these higher returns must be traded off against the opportunity cost of a foregone season of returns while the field is fallowed. As a result, the two cropping systems compete closely in terms of net returns averaged over a two-year period.

Biophysical Process Model

The crop ecosystem model known as Century is utilized to represent the processes controlling crop growth, water, nutrient, and organic matter dynamics determining the productivity of agricultural ecosystems (Parton et al.; Paustian, Elliott, and Hahn). Century is a generalized-biogeochemical ecosystem model which simulates C (i.e., biomass), nitrogen, and other nutrient dynamics. It includes submodels for soil biogeochemistry, growth and yield submodels for crop, grass, forest, and savanna vegetation, and simple water and heat balance. For use in agricultural and grassland ecosystems, the model incorporates a large suite of management options including crop type and rotation, fertilization, tillage, irrigation, drainage, manuring, grazing, and burning. The model employs a monthly time step, and the main input requirements (in addition to management variables) include monthly precipitation and temperature, soil physical properties (e.g., texture, soil depth), and atmospheric nitrogen.

For the current application, soils and climate data for each of the sub-MLRAs are used as Century model inputs in addition to management variables such as crop type and rotation, fertilization, and tillage practices. The Parameter-elevation Regressions on Independent Slopes Model (PRISM) data set was used to determine weather-related data. Information on current management systems is from the 1995 Montana Cropping Practices Survey,[3] augmented with the USDA's National Agricultural Statistics Service (NASS) database, the National Resources Inventory (NRI) database, and county-level databases of the National Association of Conservation Districts (NACD).

Soil characteristics are determined using the Advanced Very High Resolution Radiometer (AVHRR) database [from the U.S. Geologic Survey/Earth Resources Observation System (USGS/EROS) Data Center], the State Soil Geographic Database (STATSGO), and the NRI database. Baseline projections of soil C are made using historical climate and land-use records. These projections are compared to NASS records of county-level crop yields and changes in soil C derived from the Century database of native and cultivated soils. The initial land-use allocation from the 1995 Montana Cropping Practices Survey was used to calculate base C levels for each sub-MLRA.

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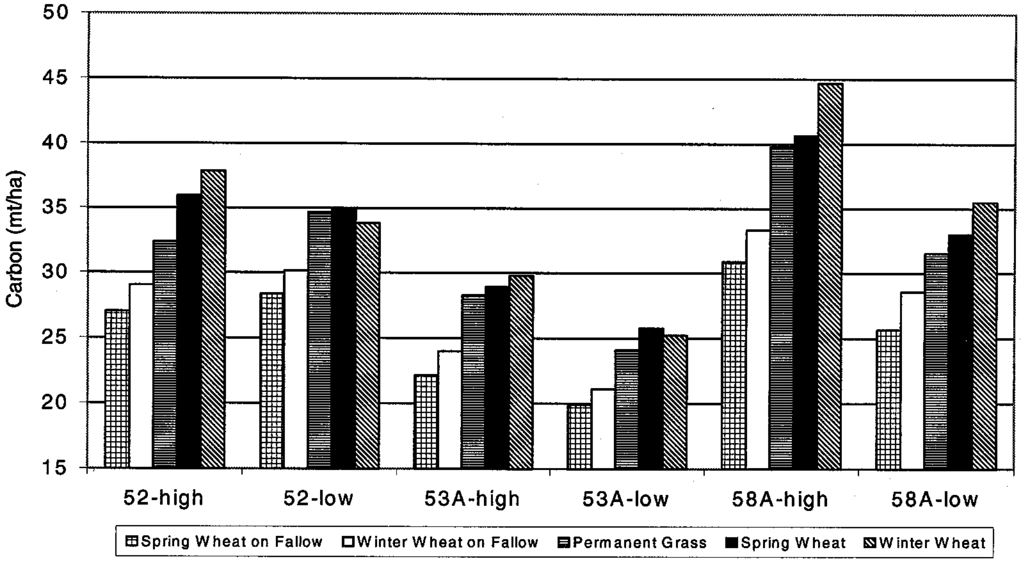
[3] These survey data on field-scale production practices were collected in conjunction with the Montana Agricultural Statistics Service, USDA. The sample included over 450 dryland farm operators in the eastern half of Montana. The data are summarized and reported in Johnson et al. (1997, 1998a, b, c, d); these reports are also available online at www.climate.montana.edu.



Note: Soil C levels for barley are the same as for spring wheat.

Figure 1. Soil C levels predicted by Century model for cropping systems in Montana

The variability in the levels of soil C predicted by the Century model across the six major crop-producing sub-MLRAs and production systems in Montana is shown in figure 1. Simulations of the crop-fallow, continuous cropping, and permanent grass production systems with the Century model show the equilibrium levels of soil C under a crop-fallow rotation range from 3–7 MT per hectare less than continuous grass over a 20-year horizon, and soil C levels under permanent grass range from 1–5 MT per hectare less than under continuous cropping depending upon sub-MLRA. In sub-MLRA 52-low, soil C levels under permanent grass compare favorably with soil C levels under a continuous cropping system. The variability across sub-MLRAs reflects the heterogeneity in bio-physical and climatic conditions, which translates into different equilibrium levels of soil C for the production systems.

Simulation of Soil C Levels and Costs

The economic simulation model selects the land use that maximizes expected returns for each sample field for each policy scenario investigated. When using this model to address soil C sequestration analysis, the net returns are augmented by the per hectare payment (g) to switch to management and land uses that would sequester additional carbon. The economic simulation is executed over a time horizon (approximately 20 years) sufficient to reach an equilibrium for each policy setting g. The land-use patterns are then summarized for each sub-MLRA for each policy setting in the form of proportions $z^{is}(g)$ of land reallocated from activity i to activity s. The Century model is used to simulate the soil C levels and annual average rates for each land use in each sub-MLRA over a given time horizon. Given the land-use changes within each sub-MLRA based on maximizing expected returns, we calculate the levels of soil C sequestered and the resulting C sequestration costs using the procedures discussed earlier.

Simulation Results: Land-Use Changes, Soil C Levels, and C Sequestration Costs

We present the empirical results for changes in land use, changes in soil C levels, and the costs of sequestering soil C for two policy scenarios: (*a*) a policy for conversion of cropland to permanent grass (PG) which gives producers a fixed annual per hectare payment, and (*b*) a policy that pays producers on a per hectare basis for fields switched to continuous cropping (CC). A precedent exists for using compensation schemes to enhance the environmental benefits from use of agricultural land. Existing agricultural policies, such as the Conservation Reserve Program (CRP), compensate producers with per acre payments in return for changes in land use and management that provide environmental benefits. The proposed revisions to the Food Security Act of 1995[4] would offer farmers the option of participating in a voluntary, incentive-based conservation program in exchange for compensation. Alternative policy designs for sequestering carbon, such as per ton payment schemes, are discussed in Antle et al. (2001).

Under the PG policy scenario, the producer could choose to enter a field into permanent grass and receive a payment above and beyond the payment for land in CRP. The level of the CRP payments used in the simulation model is set at the average level of CRP payments in Montana in the mid-1990s ($37.50 per acre or $93.75 per hectare). The PG policy is simulated for *additional* payments ranging from zero (the base case) to $125 per hectare by increments of $12.50 per hectare. Land is enrolled for a period of 20 years, and all cropland and pasture land is eligible for payment. This policy scenario reflects a payment design similar to other land retirement programs (such as the CRP) currently being used in agriculture, and is comparable to payment schemes utilized in other studies of C sequestration (Plantinga, Mauldin, and Miller; Stavins).

The CC policy provides per hectare payments for switching from a crop-fallow or permanent grass system to a continuous cropping system. Producers are offered payments ranging from a low of $5 per hectare per year and increasing by $5 increments to $50 per hectare per year. Clearly, only land that is switched from crop-fallow or grass to continuous cropping results in an increase in soil C attributable to the policy. However, if the policy pays only farmers who switch from crop-fallow or grass to continuous cropping and does not include payments to farmers who already use continuous cropping, it creates an incentive for those farmers to switch temporarily to crop-fallow and then back to continuous cropping.

Thus, two variations on the CC scenario could be considered: all fields continuously cropped could be eligible for payments, regardless of their previous cropping history (*nontargeted CC payments*); or only fields with a history of crop-fallow or grass could be eligible for continuous cropping payments (*targeted CC payments*). Both the targeted and nontargeted policies would result in the same *net* increase in soil C, and the same changes in land use and opportunity costs of sequestering C, but the costs of the policy borne by the government and the resulting producer surplus would be greater under the nontargeted program as a result of the additional fields eligible for payments. In this

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[4] Legislation introduced in 2000 would amend the Food Security Act of 1995 to establish a Conservation Security Program which would compensate farmers through the use of conservation security contracts for adopting practices judged to conserve natural resources and to be environmentally enhancing (S.3223 and HR.5511). While details about contract design are not specified, the intent of the legislation is to link payment levels with the natural resource and environmental benefits that would result from the modified production practices and with the opportunity costs of such land-use changes.

analysis we report only the results for the targeted CC payment scheme (the results for the nontargeted payment scheme are available online at www.climate.montana.edu).

A simulation of the model with the payments set equal to zero generates a baseline estimate of the land use and soil C levels for each sub-MLRA for both policies. The economic simulation model was executed for each field in the data set using observed initial conditions for land use and prices set at mean levels to reflect long-run averages over the past decade. The land-use alternatives simulated in the model were winter wheat, spring wheat, and barley in either a continuous cropping or crop-fallow rotation, and permanent grass.

The baseline land-use patterns indicate permanent grass is a more attractive alternative relative to continuous cropping in sub-MLRAs 58A-high, 58A-low, and 53A-high. These areas in the eastern and southeastern part of Montana have lower levels of moisture relative to the more productive sub-MLRA areas 52-high and 52-low. In these latter two areas, continuous cropping accounts for approximately 50% more land acreage than permanent grass.

Simulated Changes in Land Use and Soil C Levels

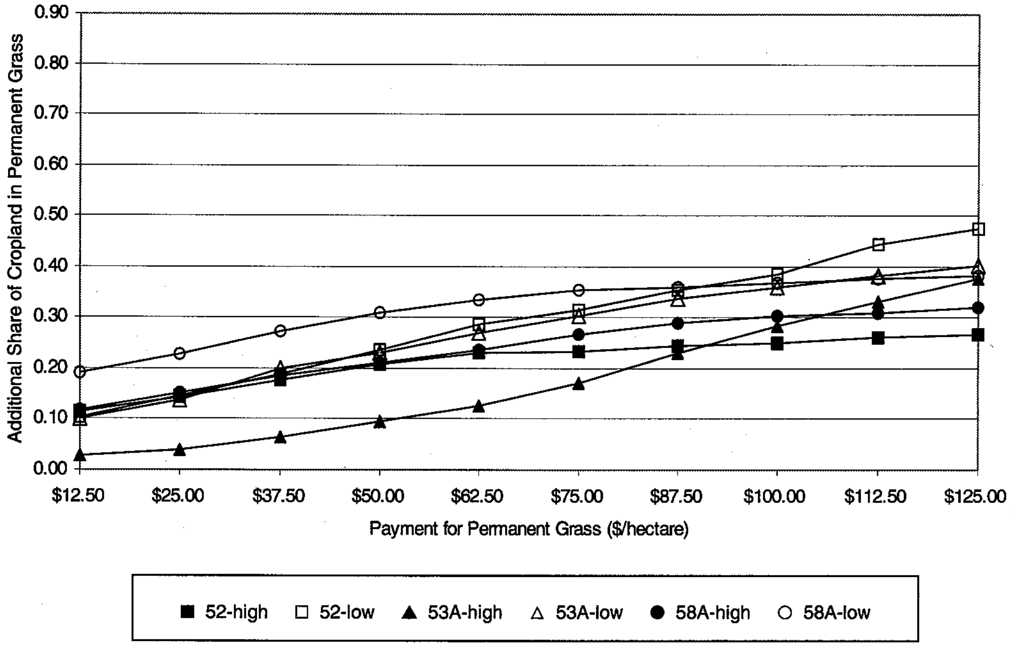
Figure 2 shows the changes in land use under each policy for each sub-MLRA as payment levels increase. For the PG policy, as payment levels increase, the *additional* share of land in permanent grass increases from less than 20% to approximately 25–45% within each sub-MLRA (figure 2a). The baseline shares of land in permanent grass range from a high of 33–35% in sub-MLRA 58A-high and 58A-low to under 7% in sub-MLRA 53A-high. The differences in land use in permanent grass across the sub-MLRAs reflect the effects of spatial heterogeneity on the opportunity cost of grain production.

Recall the crop-fallow and continuous systems yield similar net returns on average; thus the baseline allocation of land in crops is about evenly divided between the two in the sample. This implies a relatively small payment could induce farmers to switch land from crop-fallow to continuous cropping. Baseline shares of total acreage in continuous cropping range from 13% in sub-MLRA 58A-high to approximately 18% in the other four areas. Figure 2b shows the response of land-use changes to payment levels under the CC policy. All sub-MLRAs exhibit a similar pattern of land-use change under the CC policy, reflecting the fact that the opportunity cost of switching from crop-fallow or grass to a continuous cropping system is fairly similar within each sub-MLRA.

The effects of these changes in land use on the changes in the equilibrium levels of soil C after 20 years are shown in figure 3 for each sub-MLRA for each payment level. The amount of soil C sequestered varies depending upon the land area, land use, and the relative productivity of each cropping system to sequester soil C. Under both policies, the largest change in soil C sequestered in response to changes in payment levels occurs within sub-MLRAs 52-high and 52-low which comprise an average of 50% more acreage than the other areas. Comparing across policies, a greater amount of soil C is sequestered under the CC policy relative to the PG policy within each sub-MLRA. The increases in soil C become smaller as payment levels increase, reflecting the diminishing rates of land-use changes shown in figure 2.

On a *per hectare basis*, the average amount of carbon sequestered under the highest PG policy payment is fairly constant across the sub-MLRAs at about 0.4 MT/hectare. For the CC policy, the highest payment level results in average levels of C sequestration per

(a) Permanent grass payment policy



(b) Continuous cropping payment policy

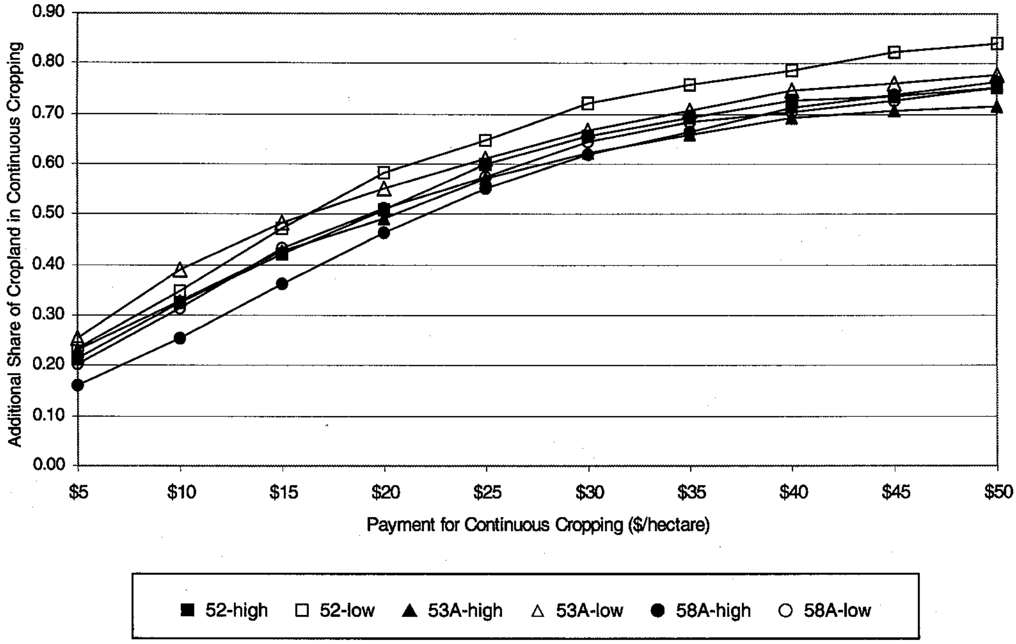
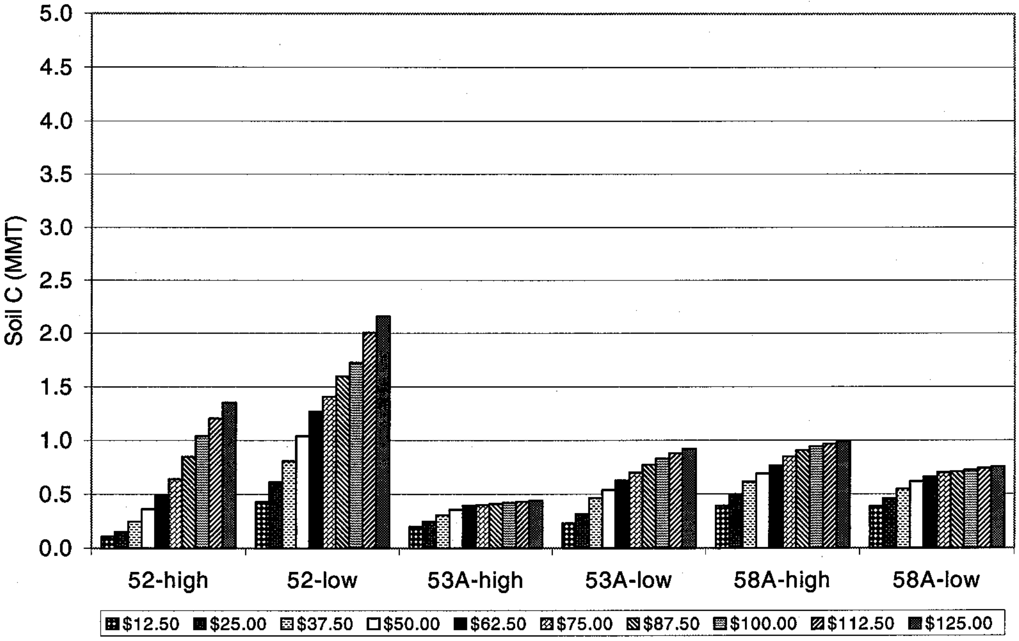


Figure 2. Changes in land-use shares by sub-MLRA and policy scenario

(a) Permanent grass payment policy



(b) Continuous cropping payment policy

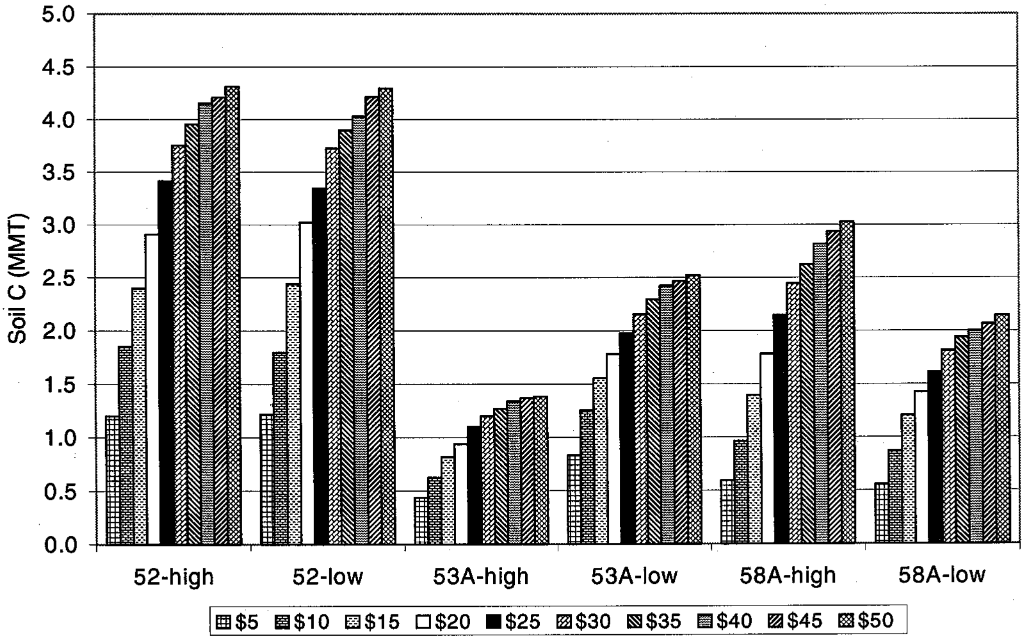


Figure 3. Changes in soil C by sub-MLRA and policy scenario

hectare per sub-MLRA ranging from 0.8 to 1.1 MT/hectare. Over the six sub-MLRAs considered, the total C sequestered ranges from 1.75 to 4.84 MMT under the PG policy, and from 4.80 MMT to 17.7 MMT under the CC policy.

Costs of Sequestering Soil C

To compare the relative efficiency of the two policies, the marginal cost curves for each sub-MLRA are constructed as discussed above. The per hectare payment levels are divided by the area-specific and activity-specific carbon sequestration rates to obtain the implicit price per metric ton of carbon. This is arrayed with the amount of carbon sequestered over the 20-year time period, where the amount of carbon sequestered is a function of the opportunity cost and site-specific land-use decisions.

Alternative ways of displaying the marginal costs would be to array the costs per metric ton and the *annual* carbon sequestration or to use a *discounted* carbon quantity. Use of annual carbon sequestration quantities could be misleading because there is an upper bound on the total amount of carbon that can be sequestered in each sub-MLRA (saturation), and thus the resulting annual amounts would depend upon how many years one wants to consider. Likewise, discounting the carbon levels assumes we know the relevant social rate of discount and time horizon. Moreover, for comparisons of our results to the biophysical estimates of soil C potential in the literature cited above, it is necessary to use undiscounted measures of soil C.

The simulated marginal cost curves for both the PG and CC policies embody the combined effects of site-specific land-use changes, soil C productivity differences, and differences in the payment levels (figure 4). For the PG policy, the spatial differences in land area, opportunity cost of alternative land uses, and carbon sequestration rates cause a corresponding heterogeneity among the marginal cost curves. For the CC policy, the relative homogeneity of changes in land-use patterns shown in figure 2 suggests the observed differences in marginal costs of C sequestration are explained largely by the spatial differences in the productivity of the soils to produce soil C and by the size of the sub-MLRA.

The relative efficiency of the PG and CC policies can be seen by comparing the marginal cost of producing a given level of soil C. As an example, to sequester an additional 0.75 MMT of C in each sub-MLRA under a PG policy, the marginal costs start at $150/MT and increase to over $500/MT of C. Under the CC policy, 0.75 MMT of C could be sequestered for less than $50/MT even in the less efficient production areas.

In general, our results show that for each sub-MLRA and for all C levels, the PG policy is far less efficient than the CC policy. Furthermore, the patterns of land-use change under the CC policy reveal the marginal cost curves under the CC policy are more elastic relative to the PG cost curves. Above $150/MT, these CC marginal cost curves turn steeply upward in response to the limitations on the quantity of soil C that can be sequestered when all acreage is in continuous cropping.

Table 1 presents a comparison of the quantity of soil C sequestered over the 20-year time horizon and undiscounted government costs and estimates of producer surplus, aggregated across all sub-MLRAs.[5] In order to sequester approximately 7 MMT of C

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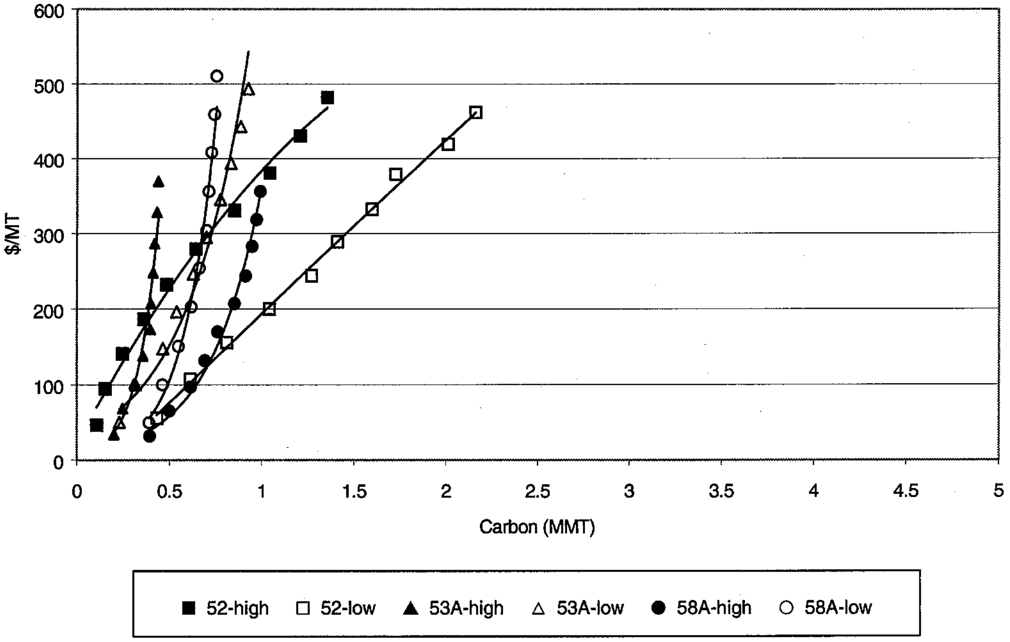
[5] Reporting the undiscounted government costs and producer surplus is consistent with our decision to not discount the quantities of soil C. These undiscounted measures correspond directly with the relevant areas on the marginal cost graphs.

(a) Permanent grass payment policy



(b) Continuous cropping payment policy

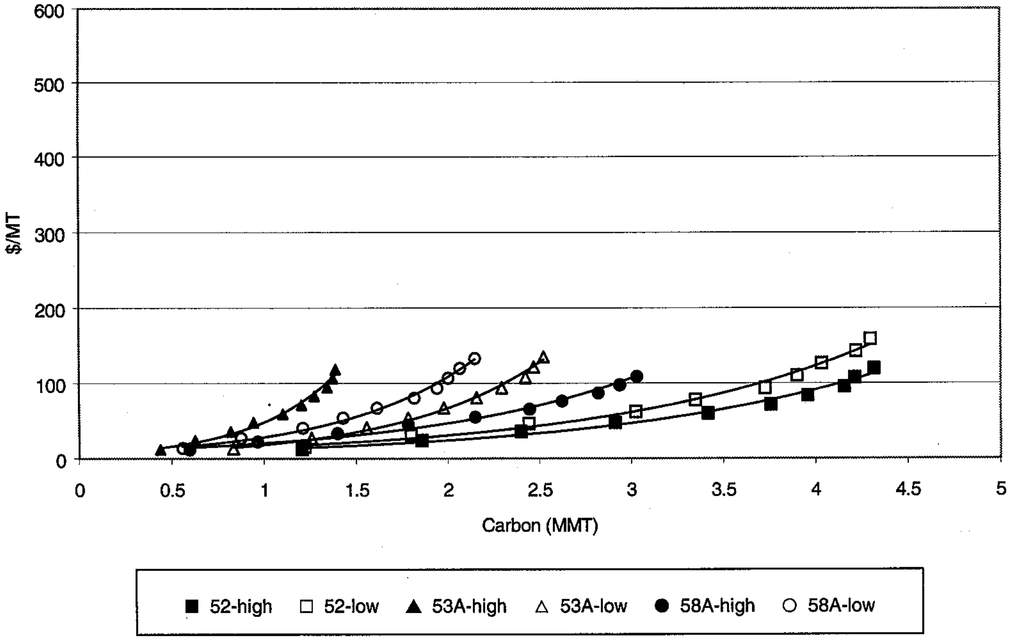


Figure 4. Marginal cost for soil C by sub-MLRA and policy scenario

Table 1. Levels of Carbon Sequestered, Costs to Government, and Producer Surplus, by Policy Scenario for All Sub-MLRAs

A. Permanent Grass Payment Policy

Payment Level ($/hectare/year)	Quantity of Soil C Sequestered (MMT)	Cost to Government ($ million)	Producer Surplus ($ million)
25	2.37	216.9	81.3
50	3.71	670.2	325.1
75	4.82	1,305.3	673.0
100	5.82	2,121.5	1,135.4
125	6.76	3,084.0	1,674.4

B. Continuous Cropping Payment Policy

Payment Level ($/hectare/year)	Quantity of Soil C Sequestered (MMT)	Cost to Government ($ million)	Producer Surplus ($ million)
10	7.61	201.7	66.4
20	12.22	647.1	303.4
30	15.54	1,226.3	639.6
40	17.28	1,818.6	1,063.5
50	18.25	2,404.9	1,531.2

(more precisely, 6.76 MMT in the PG scenario and 7.61 MMT in the CC scenario), the PG policy would involve government outlays that are more than ten-fold larger than the CC policy, and total costs that are nearly twice as high. From the taxpayers' point of view, the CC policy is far superior to the PG policy, providing much more soil C sequestered for a given government cost. From the producers' point of view, the PG policy provides much larger income transfers per metric ton of soil C sequestered. These differences in the efficiency of the two policies can be measured at either the aggregate level or on a sub-MLRA basis. Over all sub-MLRAs, the efficiency gains associated with sequestering approximately 7 MMT of C using the CC policy rather than the PG policy amount to over $430/MT of C at the margin.

The effects of spatial heterogeneity on government costs and benefits to producers are illustrated in table 2 which compares similar data for sub-MLRAs 52-high and 58A-low. Within the payment levels considered in the simulation model, the CC policy always sequesters more C than the PG policy, and the marginal costs per MT of C are lower. As payment levels are raised beyond $125/hectare under the PG policy, the increases in soil C are minimal, as less productive land is switched into grass at a decreasing rate. Such an intensive switch to permanent grass may actually cause a decline in the overall soil C levels if the acreage is taken from the land that was continuously cropped. For the CC policy, payments in excess of $50/hectare do not add appreciably more soil C because the share of land in continuous cropping at payment levels of $50/hectare is at least 90% of the cropland acreage. For a given marginal cost of producing soil C, the PG policy provides a higher producer benefit in sub-MLRA 58A-low, where the opportunity cost of

Table 2. Simulation of Land-Use Changes, Carbon Sequestration Levels, and Costs for Sub-MLRAs 52-high and 58A-low

A. Permanent Grass Payment Policy

Area	Payment Level ($/hectare/ year)	Change in Share of Land in Permanent Grass	Quantity of Soil C Sequestered (MMT)	Marginal Cost of Carbon Sequestered ($/MT)	Average Cost of Carbon Sequestered ($/MT)	Government Costs ($ mil.)	Producer Surplus ($ mil.)
MLRA 52-high	25	0.04	0.15	95	67	14.6	4.2
	50	0.10	0.37	186	123	67.6	22.9
	75	0.17	0.64	279	185	179.7	60.7
	100	0.28	1.04	381	247	396.7	138.8
	125	0.37	1.35	482	294	653.4	255.4
MLRA 58A-low	25	0.23	0.46	100	55	46.2	20.4
	50	0.31	0.62	203	76	125.3	78.0
	75	0.35	0.70	304	95	213.0	146.2
	100	0.37	0.73	408	105	296.3	219.4
	125	0.38	0.75	510	117	384.7	294.8

B. Continuous Cropping Payment Policy

Area	Payment Level ($/hectare/ year)	Change in Share of Land in Continuous Cropping	Quantity of Soil C Sequestered (MMT)	Marginal Cost of Carbon Sequestered ($/MT)	Average Cost of Carbon Sequestered ($/MT)	Government Costs ($ mil.)	Producer Surplus ($ mil.)
MLRA 52-high	10	0.33	1.86	24	16	44.7	14.3
	20	0.51	2.91	48	24	139.6	67.0
	30	0.66	3.76	72	34	271.0	143.1
	40	0.73	4.15	96	40	399.3	235.5
	50	0.75	4.32	120	42	518.1	337.6
MLRA 58A-low	10	0.32	0.88	27	18	23.7	7.2
	20	0.51	1.43	54	28	77.2	37.0
	30	0.65	1.82	81	39	146.4	74.7
	40	0.70	2.00	107	44	213.8	124.9
	50	0.75	2.15	133	50	285.9	177.8

Notes: Total hectares: MLRA 52-high = 0.68 million, MLRA 58A-low = 0.36 million. Baseline share of land in permanent grass: MLRA 52-high = 0.07, MLRA 58A-low = 0.36. Baseline share of land in continuous cropping: MLRA 52-high = 0.15, MLRA 58A-low = 0.13.

switching to permanent grass is relatively low, as compared to sub-MLRA 52-high. However, the CC policy provides producer benefits roughly in proportion to cropped area due to the similar opportunity cost of switching from crop-fallow to continuous cropping in the two areas.

Competitiveness of Agricultural Soil C Sequestration

Our analysis shows soil C sequestered by grain producers in the Northern Great Plains region under the CC policy could be competitive with C sequestered from afforestation or through industrial emissions reductions. A recent study of afforestation in Maine, South Carolina, and Wisconsin reports that the average cost estimates are in the range of \$45–\$60 per MT of C (Plantinga, Mauldin, and Miller; Plantinga and Mauldin). Stavins estimates the average cost per MT of C sequestered through afforestation to be in the range of \$38 for the Delta states to approximately \$70 for the United States. These findings are comparable to the range of the average costs reported in table 2 for the CC policy, although the quantities of soil C that can be sequestered at these costs differ.

However, as we noted earlier in the discussion of soil C permanence and contract duration, the costs of sequestering soil C would be higher if contracts were extended over a longer time period to ensure C sequestered would remain in the soil, or if payments were not targeted to land uses that changed. How much higher the costs would be depends on the duration of the contracts and the share of land currently in production systems generating the greatest levels of soil carbon.

Studies of the cost of reducing C emissions through C taxes in the United States found compliance with the Kyoto Protocol would require C to be priced in the range of \$100 per metric ton (Wiese and Tierney). When assuming C emissions credits could be traded internationally, a U.S. government study determined C could be priced as low as \$14 to \$23 per metric ton (Council of Economic Advisers). Experience with the SO_2 trading system in the United States showed large-scale energy models are likely to overestimate the costs of attaining emissions reductions (Joskow, Schmalensee, and Bailey). Partly based on this experience, and based on evidence about the cost of reducing industrial CO_2 emissions, Sandor and Skees conclude a market in tradeable C emissions credits could price C in the range of \$20 to \$30 per MT. Kopp and Anderson contend these low values for C are not likely given the various practical considerations which may limit the effectiveness of a global emissions trading system. They argue that with trading only among the developed countries as would be allowed by the Kyoto Protocol, C emissions costs would be at least \$72 per ton. Under the assumption the United States would meet a larger share of its emissions reductions commitments through reductions in energy consumption, the higher estimated costs of compliance obtained in earlier studies become relevant.

Conclusions

Previous published studies of C sequestration have considered the conversion of agricultural land to forests. There are important reasons to consider the economic feasibility of using cropland to sequester C.

- First, there are large areas of agriculture not suitable for afforestation with substantial technical potential to sequester C in soil.
- Second, a change in agricultural practices to sequester soil C is likely to bring subsidiary environmental benefits associated with reduced soil erosion and enhanced productivity.
- Third, a change in agricultural practices does not have the potentially large, and often negative, regional economic impacts that are associated with land retirement programs.

In this study we have developed a conceptual framework for analysis of the economic potential for C sequestration in agricultural soils which shows the economic efficiency of soil C sequestration depends on site-specific opportunity costs of changing practices and on the rates of soil C sequestration associated with changing practices. We then demonstrated how an integrated assessment approach to simulation modeling can be used to implement this analytical framework and to derive estimates of the costs of agricultural soil C sequestration. Linking a site-specific econometric-process model of production system choice with a crop ecosystem model designed to simulate soil C dynamics, we obtained estimates of the marginal costs of sequestering C which account for the spatial heterogeneity in agricultural land use and in rates of soil C sequestration.

Our analysis of dryland grain production systems in the Northern Plains shows how site-specific land-use decisions change in response to policy incentives, and how this induces changes in soil C within a given region. Based on our findings, a policy providing payments for converting cropland to permanent grass is a relatively inefficient means to increase soil C, with marginal costs per MT of C ranging from \$50/MT to over \$500/MT. In contrast, payments to adopt continuous cropping were found to produce increases in soil C at a marginal cost ranging from \$12 to \$140 per MT of C even in the less productive regions of the Northern Great Plains. For this policy, the average costs do not exceed \$50 per MT of C.

In concluding, several caveats should be mentioned which may affect the costs of soil C. First, if the duration of contracts for soil C sequestration were extended beyond the time period T needed to reach the saturation of soil C, the estimated costs would increase. Second, in this analysis the entire opportunity cost associated with changing agricultural practices was attributed to a single environmental benefit—sequestering C. In many cases, changes in land use and management practices produce multiple environmental benefits, such as reduced soil erosion, improved water quality and wildlife habitat, and visual amenities. If additional environmental benefits were incorporated into an analysis of soil C, the relative economic efficiency of alternative land use and management options could be different, and other options to sequester soil C may become more competitive with nonagricultural reductions in GHG emissions.

Finally, it is important to note that agriculture is both a sink for C as well as a major emitter of CO_2 and two other potent greenhouse gases, nitrous oxide and methane (McCarl and Schneider; Robertson, Paul, and Harwood). Ideally, policies to mitigate GHG emissions would reward sinks and tax sources according to their global warming potential (GWP), wherein methane is estimated to be about 21 times more potent than a unit of CO_2, and nitrous oxide is estimated to be about 310 times more potent [Intergovernmental Panel on Climate Change (IPCC)]. Both methane and nitrous oxide are also likely to be influenced by land use and other management practices.

An efficient GHG policy would provide incentives according to GWP to account for the total mixture of emission and sequestration fluxes of GHG caused by a farmer's altered land use and management practices. To do so, one could replace the C rate in our analytical framework with a measure of GWP, and introduce a policy which would provide a positive payment for a reduction in GWP and a tax on actions that increase GWP. While this generalization is straightforward in principle, implementing it poses formidable measurement problems because methods and models to quantify nitrous oxide and methane emissions are not as well developed as those for C. Nevertheless, as the needed science and data are developed, this does appear to be the direction policy will take.

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Appendix:

Data Summary and Empirical Results for the Production Models

This appendix describes the data and the empirical results for the econometric production application for the dryland grain production system of Montana. The farm- and field-level production data were collected in a survey designed to be statistically representative of the grain-producing areas of the state, stratified by the USDA's Major Land Resource Areas (MLRAs) (illustrated in figure A1 below).

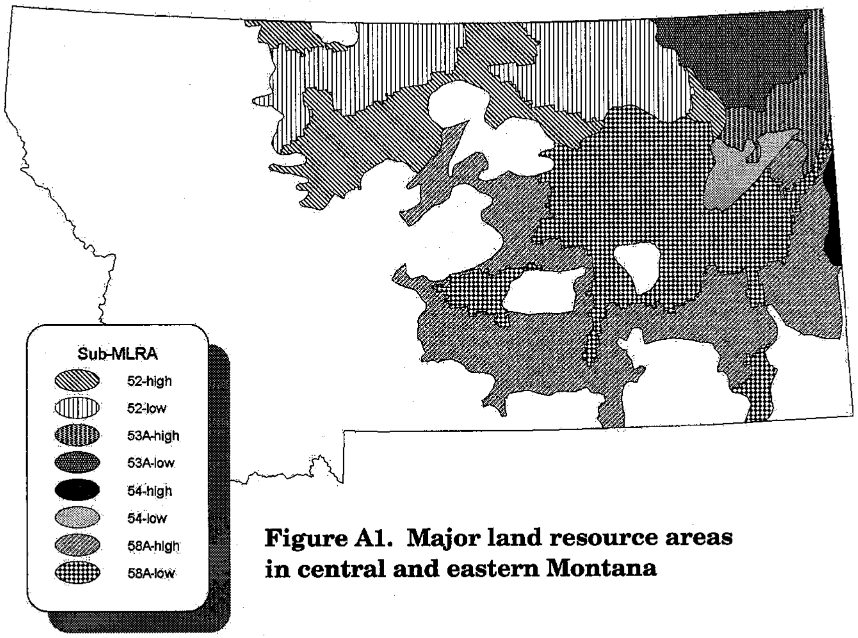

Figure A1. Major land resource areas in central and eastern Montana

Detailed production practice data were collected for up to five fields on each farm. The survey provided useable data for 425 farms. Table A1 presents summary statistics by crop, by cropping system, and by MLRA.

The practice of planting crops on land that was in crop the prior year is referred to as recropping in much of the Northern Plains region. For purposes of this study, we have denoted recropping as continuous cropping (CC) for consistency with the terminology in the agricultural economics literature. The data in table A1 show spatial variation in yields and costs of production between cropping systems and among MLRAs. Within an MLRA, the per acre yield for a crop produced on fallow is generally three to five bushels greater per acre than the same crop produced under continuous cropping conditions. The short-term variable costs of production per acre are usually \$5 to \$8 greater for a crop produced under CC conditions than for the same crop produced after fallow. Most of this per acre difference in short-term variable costs is associated with higher per acre costs for fertilizer for crops produced under CC conditions. When the variable production costs associated with fallowing the land in the prior year are added to the current year's cost of production, the per bushel variable costs for crops produced after fallow are similar to the variable costs for the crop produced under CC conditions.

The MLRAs are delimited by the U.S. Department of Agriculture on such criteria as prevailing land use, elevation and topography, climate, water, soils, and natural vegetation, and so yields, production costs, and returns vary by MLRA. The total acres in farms devoted to annually planted crops ranged from a low of 40% in MLRA 58A to a high of 67% in MLRA 52 (Johnson et al. 1997). At the individual crop level, 25.7% of the annually planted cropland in MLRA 52 was in spring wheat after fallow as compared to 29.4% in MLRA 53A, 28.7% in MLRA 54, and only 14.4% in MLRA 58A (Johnson et al. 1998a, b, c, d).

Table A1. Summary Statistics of the Montana Farm Management Survey, by Crop and MLRA

	CROP					
	Winter Wheat		Spring Wheat		Barley	
MLRA / Item	Fallow	CC	Fallow	CC	Fallow	CC
MLRA 52:						
Average Yield (bushels/acre)	46.60	37.30	39.10	35.40	61.40	59.20
Average STVC ($1998/acre)	40.41	47.17	32.84	38.35	34.55	40.78
Pesticides	5.48	5.43	3.73	5.57	3.44	5.65
Fertilizer	10.12	15.72	6.36	10.60	7.55	11.47
Machinery Operating Cost	18.41	19.96	16.60	18.23	17.27	18.77
MLRA 53A:						
Average Yield (bushels/acre)	N/A	N/A	28.40	19.90	34.90	29.90
Average STVC ($1998/acre)	N/A	N/A	27.64	31.79	25.47	30.69
Pesticides	N/A	N/A	3.60	2.88	1.22	2.05
Fertilizer	N/A	N/A	4.06	8.68	2.24	7.65
Machinery Operating Cost	N/A	N/A	13.79	14.54	15.23	15.33
MLRA 54:						
Average Yield (bushels/acre)	22.60	N/A	19.30	13.90	26.10	23.60
Average STVC ($1998/acre)	26.38	N/A	27.56	29.14	25.32	25.74
Pesticides	2.43	N/A	2.65	2.64	2.22	2.12
Fertilizer	6.05	N/A	5.08	8.08	3.29	5.01
Machinery Operating Cost	11.89	N/A	6.17	12.79	12.75	12.09
MLRA 58A:						
Average Yield (bushels/acre)	35.60	34.70	27.20	22.70	42.90	39.30
Average STVC ($1998/acre)	40.10	48.18	32.78	40.79	34.15	42.38
Pesticides	5.17	2.10	5.79	9.10	6.37	10.18
Fertilizer	12.29	20.38	6.27	10.96	7.09	10.43
Machinery Operating Cost	16.70	21.38	15.12	15.35	14.73	16.94

Source: Antle et al. (1999, p. 100), using data from the 1995 Montana Cropping Practices Survey.
Notes: CC denotes continuous cropping; STVC denotes short-term variable costs.

The econometric production models for Montana grain crops (winter wheat, spring wheat, and barley) were specified as log-linear supply functions and variable cost functions. Since the econometric production models are used to calculate expected net returns in the simulation model of land-use decisions, we needed to account for the fact that *observed* net returns may be negative when *expected* net returns are positive. Therefore, we used a system of supply function, cost function, and corresponding input demand functions rather than a profit function. The revenue component of expected returns is equal to the output price times the supply function derived from the restricted profit function, and cost is computed with the cost function. This system of a supply function and a cost function is theoretically equivalent to a profit function, and joint estimation produces efficient parameter estimates.

A log-linear input demand equation was also included to represent machinery operating costs. The supply and machinery cost equations are functions of prices for fertilizer and herbicide inputs normalized by output price, field size, and a land-use indicator (fallow or crop).

The MLRAs were stratified into high and low precipitation sub-zones according to historical climate data (Paustian et al.). Dummy variables for these zones were included to capture systematic differences in productivity across sites. A variety of herbicides are used by farmers, and a hedonic procedure was employed to quality-adjust these input data (Antle, Capalbo, and Crissman). Expected crop prices were defined as average prices received in a farmer's region net of transportation costs to the nearest grain elevator.

Table A2. NL3SLS Estimates of Supply Function, Machinery Cost, and Cost Function Models

	CROP					
	Winter Wheat		Spring Wheat		Barley	
Description	Estimate	*t*-Statistic	Estimate	*t*-Statistic	Estimate	*t*-Statistic
Supply Function:						
Intercept	2.448	5.32	3.185	11.49	3.615	9.47
Land	1.003	20.73	0.993	38.89	0.910	16.76
Fallow Dummy	0.268	3.12	0.211	6.06	0.089	1.45
Fertilizer Price	-0.350	-2.70	-0.124	-1.56	-0.320	-2.61
Pesticide Price	-0.014	-0.43	-0.015	-0.61	-0.033	-1.10
MLRA 52-high	0.041	0.53	0.004	0.06	0.056	0.69
MLRA 53A-low	-0.024	-0.08	-0.286	-4.90	-0.351	-3.30
MLRA 53A-high	-0.327	-1.70	-0.423	-7.26	-0.544	-5.26
MLRA 54-low	-0.348	-2.63	-0.679	-9.82	-0.506	-3.42
MLRA 54-high	—	—	-0.486	-5.31	-0.318	-1.57
MLRA 58A-low	-0.294	-2.11	-0.428	-5.65	-0.201	-1.26
MLRA 58A-high	-0.168	-2.04	-0.429	-5.75	-0.287	-2.97
Machinery Cost:						
Intercept	-0.304	-0.14	-1.054	-0.45	2.447	2.80
Land	1.109	22.93	1.104	33.44	1.085	20.35
Fallow	0.022	0.26	-0.101	-2.25	-0.013	-0.22
Crop Price	1.716	1.10	2.170	1.36	0.186	0.18
Fertilizer Price	-0.063	-0.47	-0.147	-1.35	0.110	0.84
Pesticide Price	0.040	1.12	0.002	0.05	-0.001	-0.04
MLRA 52-high	0.130	1.59	0.069	0.87	0.151	1.79
MLRA 53A-low	0.613	1.88	0.082	0.81	-0.143	-1.03
MLRA 53A-high	0.272	1.30	-0.148	-1.47	-0.019	-0.14
MLRA 54-low	-0.117	-0.76	-0.294	-2.58	-0.111	-0.64
MLRA 54-high	—	—	-0.031	-0.20	-0.145	-0.61
MLRA 58A-low	0.053	0.33	-0.019	-0.16	-0.325	-1.64
MLRA 58A-high	0.106	1.12	0.133	1.11	-0.005	-0.05
Cost Function:						
Intercept	0.784	1.02	-0.702	-1.17	-1.522	-1.44
Fertilizer Price	0.885	55.06	0.827	46.80	0.868	52.49
Output	1.014	11.10	1.129	15.83	1.200	9.71
MLRA 52-high	-0.356	-2.40	-0.225	-1.36	-0.292	-1.68
MLRA 53A-low	-0.053	-0.09	-0.079	-0.49	-0.200	-0.84
MLRA 53A-high	0.113	0.31	0.000	0.00	-0.181	-0.74
MLRA 54-low	-0.421	-1.60	0.304	1.54	0.111	0.34
MLRA 54-high	—	—	0.570	2.25	-0.117	-0.26
MLRA 58A-low	-0.060	-0.22	0.061	0.28	-0.139	-0.41
MLRA 58A-high	-0.053	-0.33	0.296	1.40	-0.048	-0.23
Fallow Dummy	-0.546	-3.25	-0.575	-5.88	-0.482	-3.71

Source: Antle and Capalbo (2001, p. 396).

The parameter estimates in table A2 show that the quantity supplied and machinery costs are approximately proportional to field size. Supply (and thus yield) and machinery cost also vary significantly by sub-MLRA. Using sub-MLRA 52-low as the baseline, the yields are significantly lower for less productive regions (MLRAs 53A, 54, and 58A) and higher for the more productive sub-MLRA 52-high. Costs differ by sub-MLRA, although not in a systematic way. The fertilizer and pesticide price parameters in the supply functions have the theoretically predicted negative sign. Noting the supply functions are estimated with linear homogeneity in prices imposed, these parameters imply short-run supply elasticities with respect to the output price of about 0.36 for winter wheat, 0.14 for spring wheat, and 0.35 for barley.

Table A2 also shows that winter wheat, spring wheat, and barley yields are about 31%, 23%, and 9% higher, respectively, when the crop is grown after fallow (these percentage changes are calculated as $e^d - 1$, where d is the parameter of the fallow dummy variable). In the cost function, the fallow dummy variable indicates variable costs of production are about 40% lower for all three crops after fallow. These results confirm the hypothesis that fallowed fields are more productive than continuously cropped fields. When fallow costs are considered, the data show the crop/fallow system and the continuous cropping system yield similar net returns on average, explaining the fact that cropland in the region is allocated to each type of system in roughly equal proportions.

The simulation model was calibrated to predict the observed mean frequencies of crops produced in the sample data using three parameters: the expected yield variability, the discount rate, and the expected future crop price. The expected yield variability refers to the variance of yield expectations in the population. Presumably, the variance of yield expectations in the population of farmers is less than the variance of observed yields. The base simulations used an expected yield variance that is 90% of the observed variance. Analysis determined the simulation results were not highly sensitive to this parameter.

The expected present value of returns to a crop/fallow rotation depends on a nominal discount rate. In the base model simulations, this discount rate was set at 7%. The net returns for crops produced after fallow also involve expected future crop prices. To represent the price uncertainty, and to account for the fact that in 1995, prices were above the long-run trend in real crop prices, the expected future crop prices were assumed to be less than the average observed market price in 1995. In the base simulations, it was assumed the future crop prices were random variables with a mean 10% below the 1995 average market price, and with a variance equal to the observed variance of prices. Our results indicate the choice between continuous cropping and a crop/fallow rotation is sensitive to both the discount rate and future expected prices. This finding reflects the fact that the two systems are competitive, i.e., small changes in expected future prices relative to current prices can induce a farmer to modify the choice between continuous cropping and the use of fallow.

To provide a validation of the model, the observed proportion of each land use in each sub-MLRA was computed and compared to the simulated proportions. A plot of observed and simulated mean land use falls along a 45-degree line, an indication the simulation model does reproduce the observed data without a systematic bias. Additional validation of the model can be made by testing its ability to predict observed phenomena not represented in the data used to estimate and calibrate the model. For this purpose, the model was used to simulate the percentage of acreage allocated to conserving uses (as in the Conservation Reserve Program operated by the U.S. Department of Agriculture) as levels of payments for the conserving use are varied. This exercise confirmed the model correctly predicts that larger amounts of land are allocated to conserving uses in the sub-MLRAs where crops are less profitable, and it also correctly predicts that approximately 20% of acreage in the region is allocated to the conserving use when simulated payments are in the range of actual payments for CRP contracts in Montana.