The Economics of Carbon Sequestration in Agricultural Soils

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Science has established conclusively that concentrations of greenhouse gases (GHG) in the Earth's atmosphere have been rising rapidly since the Industrial Revolution (e.g., see IPCC, 1996). While these increasing concentrations are associated primarily with fossil fuel consumption, a significant share (estimated in the range of 12 to 42 per cent) is believed to be caused by changes in land use, including deforestation and the expansion of agriculture (Watson et al., 2000, p. 5). While the consequences of increasing atmospheric GHG concentrations remains the subject of intensive scientific study and debate (see the US National Assessment [US Global Change Research Program, 2000] and [IPPC, 1996, 2001] for current literature reviews), there is growing national and international momentum to implement policies to reduce GHG emissions. The most obvious -- but not necessarily least costly -- way to do that is to reduce fossil fuel consumption. However GHGs can also be removed from the atmosphere by reversing some of the processes associated with land use changes.

International negotiations to reduce and offset emissions are being led by the United Nations Framework Convention on Climate Change (UNFCCC). The first quantitative goals for GHG emission reductions were set in the Kyoto Protocol (KP) of the UNFCCC (1998), where it was agreed that the industrialized countries would reduce emissions six to eight per cent below 1990 levels by the period 2008-2012. The KP recognized carbon (C) sequestration as a means by which countries could offset emissions of GHGs. The United States insisted that GHG emissions trading be included in the KP mechanisms to reduce GHG emissions, based on the successful SO_2 emissions trading system developed in the United States (Joskow et al., 1998). However, in the KP it was recognized that technical details needed to be resolved before C sequestration could provide measurable, verifiable offsets. In addition to these international activities, individual countries and non-governmental organizations have undertaken policies and programs to manage emissions some of which encourage C sequestration. Some firms in GHG-emitting industries such as electricity generation have begun to enter into contracts for afforestation and soil C sequestration even before they are required to meet GHG emission standards, presumably in anticipation of such standards (CAST, 2000). Agriculturalists in some countries, notably in Canada and the United States, have entered into contracts to adopt management practices that would increase soil C, and at least twenty bills related to C sequestration were introduced for consideration by the U.S. Congress in 2001. Yet ratification of the Kyoto Protocol by the United States has always been in doubt, and is even more so with the position against the Protocol taken by the Bush Administration in early 2001. However the administration ran on a platform that declared carbon dioxide to be a pollutant, and the fossil fuel and electric utility industries continue to express support for reducing emissions through sequestration. Sequestration thus is a topic of considerable current and future interest.

Scientists estimate that about 80 per cent of global C is stored in soils (Watson et al., 2000, p. 4), and that a substantial proportion of C that was originally in soils has been released due to human land use, implying that there is a large technical potential to sequester C in soils (Lal et al., 1998). However, it is less clear what the economic potential is for increasing soil C. Important questions need to be addressed: What economic incentives would be required to induce farmers to undertake the necessary actions to increase soil C? Could C be sequestered in soil at a cost that would be competitive with other sources of GHG emissions reductions?

The goal of this chapter is to survey the emerging economics literature that is beginning to answer these questions. We begin with a brief review of the literature on agriculture and GHG emissions, including the technical feasibility to sequester C in agricultural soils, a topic that has received considerable attention in the recent literature related to climate change mitigation. The following sections address various issues related to assessing the economic feasibility and competitiveness of soil C sequestration, including: the on-farm economics of the adoption of practices that increase soil C; the design of incentive mechanisms and contracts for soil C sequestration; the issue of soil C sequestration in developing countries as a mechanism to fund sustainable agricultural development; recent empirical evidence on the cost of sequestering C in agricultural soils; and a comparison to the costs of competing sinks such as industrial emissions reductions and afforestation.

GHG mitigation and agricultural soil C

Agriculture and forests are mentioned as both emitters of GHGs and as a sink for GHGs in the KP. Annex A of the KP lists agriculture as an emission source in terms of enteric fermentation, manure management, rice cultivation, soil management, field burning, and deforestation. The KP also lists agriculturally related sinks of afforestation and reforestation. Additional sources and sinks are under consideration including agricultural soil C and were important topics of discussion at the November 2000 Conference of the Parties in the Hague. McCarl and Schneider (1999, 2000) provide surveys of the ways agriculture may participate in or be influenced by greenhouse gas mitigation efforts. Here we briefly review the principal interactions between agriculture and climate change mitigation.

First, agriculture is a significant emitter of GHGs, particularly methane, nitrous oxide, and CO_2. Thus policies designed to reduce emissions may be targeted at these agricultural sources. The IPCC (1996) estimates that globally agriculture emits about 50 per cent of total methane, 70 per cent of nitrous oxide, and 20 per cent of CO_2. Sources of methane emissions include rice, ruminants and manure. Nitrous oxide emissions come from manure, legumes and fertilizer use. CO_2 emissions arise from fossil fuel usage, soil tillage, deforestation, biomass burning and land degradation. Contributions across countries vary substantially, with the greatest differences between developing and developed countries. Deforestation and land degradation mainly occurs in developing countries. Agriculture in developed countries uses more energy, more intensive tillage systems and more fertilizer, resulting in fossil-fuel based emissions, reductions in soil C and emissions of nitrous oxides. In addition animal herds emit methane from ruminants and manure (IPCC, 1996; McCarl and Schneider, 1999, 2000).

Second, agriculture may enhance its absorption of GHGs by creating or expanding sinks. This may be achieved through a variety of changes in land use and management practices. The Kyoto Protocol allows credits for emission sinks through afforestation and reforestation. Until recently most studies of C sequestration focussed on the conversion of crop land to forest land (Adams et al., 1993; Parks and Hardie, 1995; Platinga et al., 1999; Stavins, 1999). In these studies producers are assumed to 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 land. The costs of C sequestration are based on converting the forested land to C storage units using a representative time path of C sequestration for a given forest management regime. The KP also allows for consideration of additional sources and sinks for C, including agricultural soils, the focus of this review.

Third, agriculture may provide products that substitute for GHG-emission-intensive products, thus displacing emissions. Agriculture may provide substitute products that replace fossil fuel intensive products. One such product is biomass for fuel usage or production. Biomass can directly be used in fuelling electrical power plants or maybe processed into liquid fuels. Burning biomass reduces net CO_2 emissions because the photosynthetic process of biomass growth removes about 95 per cent of CO_2 emitted when burning the biomass. Fossil fuel use, on the other hand, releases 100 per cent of the CO_2 in it. Substitute building products can be drawn from forestry reducing fossil fuel intensive use of steel and concrete (Marland and Schmalinger, 1997).

Finally, agriculture may find itself operating in a world where commodity and input prices have been altered by GHG emission policies. The need to reduce emissions and the implementation of emissions trading will likely affect fossil fuel prices. For example, diesel fuel distributors might need to purchase an emissions permit, effectively raising fuel prices. Similarly the United States might implement a fuel tax. The tax and corresponding transportation cost increases might influence the cost of petrol-based agricultural chemicals and fertilizers as well as on-farm fuel prices and off-farm commodity prices (McCarl et al., 1999; USDA, 1999; Antle et al., 1999; Konyar and Howitt, 2000).

Technical feasibility of sequestering C in agricultural soils

When undisturbed soil is put into cultivation, the associated biological and physical processes result in a release of 20 to 50 per cent of the soil C over a period of about 50 years, with the amount varying by soil type, agricultural practices and other site-specific conditions (Tiessen et al., 1982; Mann, 1986; Rasmussen and Parton, 1994). Therefore in most cultivated soils there is a potential to rebuild the soil C stock up to the level that existed before the land was used for agricultural purposes. Some scientists believe that with improved management, the stock of soil C could be brought to a higher level than existed before the soil was first disturbed by agriculture.

Increases in soil C can be achieved through the adoption of various land use and management practices (Lal et al., 1998; CAST, 1992). Management practices that can increase soil C sequestration include land retirement (conversion to native vegetation or reversion to wetlands), afforestation, residue management, less-intensive tillage, changes in crop rotations, conversion of cropland to pasture and restoration of degraded (that is, highly eroded) soils. Lal et al. (1998) estimate that 49 per cent of agricultural C sequestration can be achieved by adopting conservation tillage and residue management, 25 per cent by changing cropping practices, 13 per cent by land restoration efforts, 7 per cent through land use change and 6 per cent by better water management.

It is important to note that practices to enhance soil C storage can change agricultural productivity. Soil scientists argue that improved soil management improves soil quality and ultimately increases crop yields in many cases (Lal et al., 1998). However, such changes are not without their costs, and as we shall discuss further below, farmers face an opportunity cost associated with changing to practices that increase soil C. The literature also acknowledges that these practices may under some conditions actually reduce agricultural productivity or the size of the agricultural cropland base (IPCC, 1996, 2000; Marland et al., 1998; McCarl and Schneider, 1999, 2000).

While some changes in land use and management practices can increase the stock of C in the soil, they can only do so up to a new equilibrium state that depends on management and the biophysical conditions of the site. As the soil C level increases, the rate of soil absorption of C eventually decreases, and the soil's potential to become a future emission source increases because subsequent alteration of the management regime (for example, reversion to conventional tillage after the use of reduced tillage) can lead to a release of C. This latter point is an important one that we will discuss below in the context of what has become known as the permanence issue in the international debate over inclusion of soil C as an allowable sink under the Kyoto Protocol.

As with the process of disinvestment, the rate at which the soil C stock can be rebuilt will be a function of various site-specific biophysical conditions, as well as the types of land use and management practices that are followed. For most changes in agricultural practices, soil scientists estimate that the equilibrium level of soil C will be achieved after a period of 20-30 years following the change in practices. West et al. (2000) reviewing 30+ experiments show this occurs after approximately 20 years. Thus on a land unit where the i^{th} practice has been followed for a long period of time, the stock of soil C reaches an equilibrium level C^i. If the s^{th} practice is then adopted and followed continuously thereafter, the time path of soil C typically follows a logistic path, eventually reaching a new equilibrium level C^s (Figure 1). This time path can be approximated with the annual average rate of soil C increase $\Delta c^{is} \equiv (C^i - C^s)/N$. However it must be emphasized that these rates are an approximation to the generally nonlinear path of soil C accumulation or decumulation, and it also must be emphasized that actual rates vary spatially as a function of biophysical conditions (soil type, depth, climate), land use history and the current practices that are followed, a key point discussed further below.

(Inset Figure 1 here.)

Due to the long periods of time over which soil C stocks change, it is relatively difficult to measure such changes directly in the soil, although such methods do exist and have been applied at long-term study sites (Watson et al., 2000). Therefore in order to estimate such changes over a wide range of locations it is more practical to use models to estimate potential changes in soil C. With available data on site-specific soil and climate conditions, land use history, and other relevant parameters, the changes in the stocks of soil C can be simulated over the long periods of time using bio-physical process models (Parton et al., 1994; Paustian et al., 1996).

The spatial variability exhibited by resource endowments and climate means that a single land use or management practice will not be effective at sequestering C in all regions and thus different management practices will be efficient in different areas. For example in the intensively cropped Cornbelt states, soil C can be increased by reducing tillage intensity. In the Great Plains a large amount of cropland is in a crop-fallow rotation, and soil C can be increased under those conditions by increasing cropping intensity (that is, reducing the frequency of fallow) as well as by reducing tillage. Conversion of crop land to permanent trees or pasture can also increase soil C.

To illustrate, Table 1 presents estimates of the biophysical potential to increase soil C with changes in tillage practices in Iowa and with increased cropping intensity in Montana. A change from conventional tillage to no-till can sequester an additional 0.52 tonnes C per hectare per year in Iowa, compared to the estimated potential in Montana from changing from fallow to continuous cropping of 0.44 tonnes C per hectare per year.[1] The 1997 Census of Agriculture reports 11 million hectares of cropland in Iowa and 7 million hectares of cropland in Montana. The data in Table 1 indicate that if Iowa and Montana producers were to change their practices this could result in about 9 million tonnes or about 1.5 per cent of the US Kyoto accord commitments. Using a nationwide estimate of about 120 million hectares at a rate of 0.5 tonnes per hectare per year would offset about 10 per cent of the US commitment to Kyoto.

(Insert Table 1 here.)

Table 2 presents a wide range of estimates of rates of potential C gain for a variety of practices derived from a number of studies around the world. One can see that the rates in Table 1 are within the bounds of Table 2. Table 2 also indicates the time period over which these rates are applicable. Note that in most cases the periods range from 5 to 30 years, depending on the type of practice.

(Insert Table 2 here.)

The spatial variability evidenced by the data in Tables 1 and 2 has important implications for the design of contracts for soil C sequestration, and for issues such as the monitoring of compliance with contracts. These are topics that we shall address in detail later in this chapter.

Economic feasibility and competitiveness of sequestering C in soil

Soil scientists suggest that the most suitable land for C sequestration is where the potential gain in C is highest (e.g., Lal et al., 1998). However the potential C gain estimates for soils (as in Tables 1 and 2) indicate only the technical potential to increase the stock of C in agricultural soils. These data do not provide the basis to determine at what cost agricultural producers are willing to adopt practices that increase soil C. In the following section we show that the economic feasibility and competitiveness of soil C sequestration depend on the opportunity cost per tonne of C, that is, the opportunity cost per hectare of changing land use or management practices, divided by the rate of soil C accumulation.

We know from various data that there is great spatial variability in the productivity of agricultural land, and thus in the opportunity cost associated with changing land use and management practices. One measure of this economic variability is provided by the different rates at which farmers voluntarily bid to remove land from crop production and place it in conserving uses in the Conservation Reserve Program in the United States. These data show that the highly productive crop land in Iowa is bid into the CRP for about \$36 per hectare, whereas the lower productivity land in Montana bid in for about \$14 per hectare (USDA, 2000). Thus even though the technical potential to sequester C in soil is higher in Iowa than in Montana, it does not follow that soil C can be sequestered at a lower cost per tonne of C. We show below that indeed the opportunity cost per tonne in Montana is lower than in Iowa for some levels of C sequestration.

Economic analysis of agricultural soil C sequestration at farm and regional scales

The atmosphere is a public good -- everyone benefits from it without having to 'pay' for it. This fact means that private individuals and firms, and even individual countries, have little incentive to take actions to prevent the accumulation of GHGs in the atmosphere. Therefore the demand for individuals to reduce emissions must derive primarily from collective action, that is, from government policy. As of this writing, the Kyoto Protocol has not been ratified by the United States or enough other countries to put it into force.

Any perceived incentive to sequester carbon therefore must be based on private entities' anticipation of future policies requiring reductions in GHG emissions.

Two provisions of the Kyoto Protocol illustrate policies that may be implemented by individual governments or through international agreements. One provision is the Clean Development Mechanism, an institution that would channel funds from industrialized countries to pay for investments in cleaner production technologies in developing countries. The other likely future institution is a carbon trading system modeled after the sulfur dioxide emissions trading system now in operation in the United States (Joskow, Schmalensee and Bailey, 1998). A carbon emissions trading system would be used by a country like the United States to meet a *net* emissions target established through its own policies or through an international agreement. This net emissions target could be met either by emissions reductions or by increasing the use of sinks such as forests and cropland. Thus, in contrast to a "cap and trade" system in which the total emissions is fixed, under a system that includes sinks *gross* emissions would not be fixed, and could in fact increase as long as the size of sinks was sufficient to offset emissions increases. The inclusion of sinks in the Kyoto Protocol was controversial because some countries argued that it would allow the United States and other countries with large sink potentials to avoid reducing overall emissions.

In addition to market-based mechanisms, GHG emissions could also be reduced through direct government interventions. There are many examples of government policies designed to reduce environmental impacts of human activity. In the United States, for example, the Conservation Reserve Program provides payments to farmers who take actions that reduce soil erosion. In a similar way policies could be designed to sequester soil C.

Private entities may also be motivated to take individual or collective actions to mitigate GHG emissions. Interest groups may organize concerned individuals to raise funds to purchase C credits, or to enter directly into contracts with individuals or groups to sequester C. Business firms wanting to demonstrate environmental concern also may be motivated to buy C, whether or not their emissions are constrained by government policy. Business firms also may buy C contracts in anticipation of emissions standards or to preempt the possibility of emission standards being imposed. This type of behavior may explain recent transactions involving C even though most countries do not have policies requiring firms to reduce GHG emissions (CAST, 2000).

From an economic perspective, soil C provides value in three dimensions, first as an essential component of soil that affects agricultural productivity, second as a way to offset CO_2 emissions from other sources and third as an indirect source of benefits involving improved environmental quality. The first component of soil C's value is a private benefit to the farmer, hence a farmer who understands the productive value of soil C will make management decisions to optimize the soil C stock. Soil C is not observable visually, and at least some soil scientists argue that many farmers do not understand the role of soil C in production or how to manage it, and thus tend to use practices that result in a smaller-than-optimal soil C stock in terms of private benefits. The second and third dimensions of soil C involve external benefits, hence, farmers will not optimally manage soil C from a social point of view -- and can be expected to maintain a smaller stock than socially optimal -- unless some mechanism exists to induce farmers to equate the marginal social benefit with the marginal social cost.

Our goal is to assess the role that economic incentives play in inducing farmers to adopt practices that would increase the amount of C in the soil. To analyze soil C from an economic perspective, we shall assume that agricultural producers are economically rational and thus utilize those land and management practices that they believe yield the highest economic returns (the analysis can be generalized to account for risk, and other non-economic objectives). Thus economically motivated producers will adopt alternative practices that increase soil C if and only if there is a perceived economic incentive to do so, but we do not assume farmers are necessarily managing the stock of soil C optimally from either a private or a public perspective. If indeed farmers are under-investing in the stock of soil C from either a private or public perspective, then a policy that provides additional incentive to increase the stock of soil C should move them towards a more efficient resource allocation.

Incentives and contract design

Farmers could be provided economic incentives to sequester C in soil through direct government payments or private markets. Direct government programs would include efforts such as the Conservation Reserve Program or other government conservation programs where the government pays farmers to provide environmental services or reduce offsite damages. Alternatively, private markets would arise if the government imposes GHG emissions standards on industry and permits trading of emissions credits from farmers. In either case contracts between buyers (emitters) and sellers (farmers) would specify the payment mechanism and other terms for either a government program to sequester soil C of for sales in a market for C credits. Following Antle et al. (2001a) there are two classes of costs associated with implementing contracts for the provision of an environmental amenity through changes in agricultural practices, farm opportunity costs and contract costs. The first is the opportunity costs of resources expended on the farm to produce the amenity; the second is the costs associated with implementing contracts and involves brokerage fees, and monitoring of compliance with the C accumulation or practice execution terms of the contract, as well as other transactions costs.

For purposes of this discussion we assume that contracts for soil C are designed and implemented using an approach that is similar to existing conservation programs (below we discuss whether this is an efficient way to design contracts). Antle et al. (2001a) describe two ways that these incentive payments could be made. The first type of incentive gives producers a fixed payment per hectare of land switched from a cropping system with a relatively low equilibrium level of soil C to a system that produces a higher equilibrium level of soil C. This per-hectare payment mechanism is similar to existing programs such as the CRP that provide payments on an area basis to producers that adopt land use or management practices designed to reduce environmental damages or enhance environmental quality. As with the CRP, the per-hectare payments could vary by region, although what criteria would be used for this is less clear than in the CRP case where farmers' bids are used to establish payment levels. The second policy is a per-tonne payment mechanism that pays farmers for each tonne of C sequestered when they change land use or management practices. We assume that C rates are established through a combination of field measurements and modeling. For example, agroecozones (areas with relatively homogeneous soils and climate) could be used as the basis for designing a statistically-based sampling strategy for field measurements to establish soil C levels, and models of soil C dynamics could be coupled with those measurements to estimate soil C rates for each combination of land use and management practice. These estimated soil rates could be verified by periodic field measurements.

Per-hectare contracts would specify management practices that the farmer agrees to follow, and the farmer would receive this payment regardless of the amount of C that is sequestered on the contracted land unit as long as the specified practices are followed. A per-tonne contract would be based on the agreed-upon price per tonne of C and the established annual C rate for that agroecozone and the practices the farmer uses. Both types of contracts would involve similar conventional transactions costs for contract negotiation, legal fees and so on. The per-hectare contract would require monitoring to ensure that the farmer follows the practices specified in the contract, whereas the per-tonne contract would require establishing the soil C rates for each type of contract and monitoring C accumulation. Thus assuming that the costs of monitoring compliance with practices under the per-hectare contract is less than the cost of measuring and monitoring C under the per-tonne contract, the total contracting costs required to implement the per-hectare contracts would be lower than for the per-tonne contracts. If the per-hectare contracts were used to satisfy an international agreement such as the Kyoto Protocol or were traded in a carbon market, then some type of measurement and monitoring system would be needed, similar to what would be needed to implement the per-tonne contracts. Consider a contract for *n* periods that pays a farmer g^{is} dollars per time period for each hectare that is switched from system i to system s. We assume that farmers have static price expectations for future real expected returns under each practice. For the producer to switch management practices, the present value of system s must be greater than the present value of system i. Letting D be the present value of 1 at interest rate r for *n* periods, and letting π^i be the returns to the ith system, the present value of expected returns for the the i^{th} system is $V^i = \pi^i D(r,n)$, and the present value of a payment of g^{is} dollars each year is $G^{is} = g^{is} D(r,n)$. Thus the producer will switch from system i to system s for *n* years if $V^i < V^s + G^{is}$, which implies $\pi^i < \pi^s + g^{is}$ or $(\pi^i - \pi^s) < g^{is}$ each period. Defining $h^{is} = \pi^i - \pi^s$ as the opportunity cost of changing from practice i to practice s, it follows that a producer will agree to switch practices if $h^{is} < g^{is}$, otherwise the producer will continue to use practice i.

Now consider a policy that offers producers a payment for each tonne of sequestered soil C. The farmer is offered a price, P ($ per tonne of C), so if the farmer switches from system i to system s and produces an additional Δc^{is} tonnes of C per hectare each period, the payment per hectare will be $P\Delta c^{is}$. Following the logic of the previous paragraph, the farmer will agree to switch from system i to system s if and only if $\pi^i < \pi^s + P\Delta c^{is}$ or if $(\pi^i - \pi^s)/\Delta c^{is} = h^{is}/\Delta c^{is} < P$, that is, if and only if the opportunity cost per tonne $h^{is}/\Delta c^{is}$ is less than or equal to the price per tonne.

As noted earlier the rate of soil C change varies with time. Also price expectations may not be static. Under these conditions one cannot simplify the present value expressions to expressions for payments per time period, but the opportunity cost per tonne per year remains a useful approximation.

Spatial heterogeneity and the marginal cost of soil C at the field and agroecozone scales

Agricultural land is generally spatially heterogeneous with respect to physical, climatic and economic characteristics. This means that each agroecozone has distinct environmental and economic characteristics. To account for the physical and climatic heterogeneity we introduce a site-specific vector of environmental characteristics e_j, for j = 1,…,J land units in the region. To account for economic heterogeneity we index prices and capital services by land unit. Letting p be crop price, w input prices, and z fixed factors, the profit function for each land unit can now be represented as $\pi^i(p_j^i,w_j^i,e_j,z_j^i)$, indicating that profit varies spatially. The opportunity cost for switching the jth land unit from system i to system s is now $h_j^{is} = \pi^i(p_j^i,w_j^i,e_j,z_j^i) - \pi^s(p_j^s,w_j^s,e_j,z_j^s)$ and is also spatially variable. For input quantity x the equilibrium soil C per hectare is expressed as $C_j^i = C^i(x_j^i,e_j,z_j^i)$, and the average rate of C sequestration for a change from practice i to practice s over N years is $\Delta c_j^{is} = [C^i(x_j^i,e_j,z^i) - C^s(x_j^i,e_j,z^i)]/N$. Thus in a spatially heterogeneous region, the opportunity cost per tonne, $h_j^{is} / \Delta c_j^{is}$, varies across land units.

At the level of the individual land unit (the field), farmers make discrete land use decisions involving tillage system choices, land retirement choices and so on. Antle and Mooney (2001) provide a detailed analysis of these discrete land use decisions and how they are affected by spatial heterogeneity. They define the per hectare and per tonne switch prices as the marginal opportunity cost at which farmers enter into a contract to sequester soil C. In the case in which there is one alternative management practice, the marginal cost curve for C production is perfectly inelastic at a zero quantity for all payment levels or C prices below the switch price, and then is perfectly inelastic at the quantity of soil C produced on that land unit for all payments or C prices greater than the switch price. Antle et al. (2000, 2001a) show that when individual land units are aggregated, the resultant regional marginal cost curve is upward sloping. This mapping can be constructed by ordering all land units according to their switch price or marginal opportunity cost, and then aggregating the quantity of soil C produced at each marginal opportunity cost.

Note that the total quantity of C sequestered is a stock accumulated over time. This raises the issue of how to compare C accumulated at different points in time to a marginal cost at a point in time. Some studies (e.g., Stavins, 1999) discount the flows of C into a stock in the same manner that the financial flows were discounted in the discussion above. This type of procedure can be justified by assuming that P, the real value per tonne of C, is constant. Then the total value can be expressed as $V = \Sigma_t \; P \, \Delta c_t \, (1+r)^{-t} \; = \; P \, \Sigma_t \, \Delta c_t \, (1+r)^{-t}$ $= P \cdot C$, so the term C can be interpreted as the aggregate discounted quantity. However, there is no reason to believe that the real value of C will be constant, calling the logic of this procedure into question. An alternative approach is to simply aggregate C changes undiscounted, recognizing that discounting (if deemed appropriate) would reduce the estimated C stock commensurately.

Spatial heterogeneity and policy efficiency

The literature on design of environmental policies for agriculture has noted that, ignoring contracting costs and market imperfections, existing policies are inefficient in the sense that they pay farmers for the adoption of alternative practices rather than per unit of environmental benefit provided by the practices, and thus do not account for the spatial variability in benefits and costs associated with the adoption of improved management practices (see Babcock et al., 1996; Helfand and House, 1995; Fleming and Adams, 1997). Efficient incentive mechanisms would account for the spatial heterogeneity in the environmental benefits produced and the costs of providing these benefits. Explanations for the use of existing policies include: the cost of information required to implement site-specific policies; information asymmetries between government agencies and farm decision makers; and political considerations (see Wu and Boggess, 1999). In an analysis of C sequestration in forests, Stavins (1999) suggests that a payment mechanism based on tonnes of C sequestered would be prohibitively expensive to implement. Pautsch et al.

(2000) suggest that it is useful to investigate efficient sequestration mechanisms because they provide a lower bound on costs.

Antle et al. (2001a) show that for each quantity of C sequestered, the marginal opportunity cost of the per-hectare payment mechanism (MC_H) is greater than or equal to the marginal opportunity cost of the per-tonne mechanism (MC_T), that is, $MC_H \geq MC_T$, as illustrated in Figure 2, with equality holding at the saturation point C_S. In addition they show that the efficiency of the per-hectare payment mechanism relative to the per-tonne mechanism, (MC_T/MC_H), is a decreasing function of the spatial heterogeneity of the opportunity cost per tonne of C. This result derives from the fact that the opportunity cost per tonne is equal to the ratio of the opportunity cost divided by the C rate. When spatial heterogeneity is high farmers with very low C rates can participate under a per-hectare payment contract, yielding higher cost for each amount of C sequestered than with a per-tonne payment contract. In their analysis of the marginal cost of soil C sequestration in Montana, Antle et al. (2001a) find that a per-hectare payment scheme is as much as four times more costly than the per-tonne payment mechanism. They conclude that there could be high payoffs to implementing C contracts that account for spatial variability in biophysical and economic conditions.

(Insert Figure 2 here.)

Saturation, permanence, contract duration and price equivalence

Soil science has established that there is an upper bound on the amount of soil C that can be stored in the soil (see Figure 1 and West et al., 2000). This saturation of the soil means that the time period over which soil C may be removed from the atmosphere through changes in agricultural land use and other management practices is limited. For this reason agricultural soil C has been described as providing a near-term method to offset GHG emissions while longer-term options are being developed (CAST, 2000).

A related issue is the permanence of C in the soil. Soils research has shown that sequestered C is volatile and it has been found that if practices sequestering soil C (such as reduced tillage) are discontinued the C stored in the soil can be released back in to the atmosphere in a short period of time. 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. During the time period in which C is being accumulated, the farmer is providing both accumulation and storage services. Once the soil C level reaches the saturation point, the farmer is providing only storage services. However, the key point is that both accumulation and storage services depend on the farmer continuing to maintain the land use or management practices that make the accumulation possible. This means that if saturation is reached in N_S years but society wants to store the C for $N_D > N_S$ years, the duration of the contract will have to be for N_D years. Clearly if society wants to sequester soil C and this takes $N_S = 20$ years, but wants this C to remain in the soil for $N_D = 50$ years, farmers will have to be paid for 50 years. This implies a much higher cost than if farmers only have to be paid during accumulation. For example a payment of $1 per hectare over 20 years has a present value (at 5 per cent interest) of about $12.50, whereas the present value of $1 for 50 years is about $18.30, about 50 per cent higher.

A similar result has been derived by McCarl and Murray (2001). They evaluate the price for C that renders the net present value of a stream of C equivalent offsets equal to the costs incurred under a carbon project. They solve the following equation for the breakeven C price P:

$$\sum_{t=0}^{T}(1 + r)^{-t} P \ E_t = \sum_{t=0}^{T}(1 + r)^{-t} C_t$$

where r is the discount rate, T the number of years in the planning horizon, E_t the emissions offset generated by the prospective project in year t, and C_t the cost of operating the emissions offset project in year t including initial investment, operation and maintenance terms as they vary over project life. They compute the price discount for various scenarios, ranging from an emissions offset which costs one dollar per year generating one unit of C per year, to scenarios in which soil saturates in 20 years, at which time the practice may be reversed and the C volatilized. The analysis shows that the saturating/volatile soil C can be worth one-third to one-half of the value of an emissions offset for plausible values of the parameters.

Another way to approach the permanence issue in designing contracts is to impose a penalty for failure to comply with the terms of the contract, including penalties for subsequent release of the C after the contract expires. But this would increase the cost to the farmer of complying with the contract, farmers would demand greater compensation, so this contract provision would also have the effect of raising the cost of the contract.

Some have referred to soil C as a 'commodity' that farmers can produce and sell like conventional agricultural commodities (USDA et al., undated). The issues of saturation and permanence show that contracts between buyers and sellers of environmental services are different from contracts for conventional agricultural commodities. The buyer never actually takes delivery of the commodity, rather, the commodity is stored in the soil that belongs to the land owner. The discussion above shows it is more accurate to describe the farmer as providing a service for a specified period of time.

Other greenhouse gases: global warming potential

As noted earlier 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, 2000; Robertson et al., 2000). Ideally policies to mitigate GHG emissions would reward sinks and tax sources according to some index of their global warming potential (GWP). For example, scientists estimate the 100-year GWP for methane 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 (IPCC, 1996). Indeed, compliance with the Kyoto Protocol was defined in terms of an index of GWP of six greenhouse gases (Tietenberg *et al.* 1998).

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 that accounted 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 simply replace the C rate in the earlier analysis of farm opportunity cost with a measure of GWP and provide a positive payment for a reduction in GWP and impose 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 this does appear to be the direction that policy will move as the needed science and data are developed. **Co-benefits and costs**

Many of the potential soil sequestering practices have been previously encouraged in US government programs designed to achieve environmental improvements and agricultural income support. Conservation titles involving reduced tillage have been included in recent agricultural farm bills in an effort directed toward achieving environmental improvement while at the same time supporting agricultural income. Water quality and soil erosion programs have also been undertaken to encourage practices which change tillage intensity and land use, retire fragile crop lands to grass or trees, preserve or re-establish wetlands, improve wildlife habitats and provide chemical and erosion retaining stream buffers but which also increase soil C. Thus a number of the soil sequestering practices can have positive co-benefits providing net GHG emission reductions along with farm income support and environmental quality improvements. Co-costs of practices that sequester soil C may also arise through increased pesticide use with reduced tillage.

The benefits of a number of these activities arise external to the farm producer who is employing them. This raises an important economic issue of inventorying and valuing benefits that accrue above and beyond net GHG reductions as well as the economic issue of determining how the incentive system might be designed to reflect the value of these other effects. An efficient policy designed to encourage beneficial agricultural practices would need to account not only for C sequestration but also for all other co-benefits and costs. Concepts such as an environmental benefits index have been developed for use with existing programs and could be generalized to include soil C sequestration or greenhouse warming potential (Feather et al., 1999).

Contracting costs, incentive compatibility, and program eligibility

Above we noted that if a buyer were to enter into a contract to pay a farmer to use specified practices, the farmer's compliance with the terms of the contract would have to be monitored. When the practices are easily observable, for example changing from a crop-fallow rotation to continuous cropping, compliance can be monitored at relatively low cost, as shown by experience with the Conservation Reserve Program (the US Department of Agriculture has substantial human resources already monitoring compliance with other programs). Alternatively, changes in land use over large areas can be monitored at relatively low cost using remote sensing technology (CAST, 2000). However, if contracts specify other changes in management, such as tillage practices, and use of fertilizer and pesticides, the cost of monitoring compliance may be substantially higher.

The other component of contracting costs is measuring the quantity of soil C sequestered. A significant part of this measurement involves determining soil and climatic variables on a site-specific basis. Farmers know their management practices, and also typically have considerable knowledge of their soil quality and climatic conditions, but buyers of the contracts can only obtain that information at a cost, so there is an information asymmetry between sellers (farmers) and the buyers. One solution to the monitoring and measurement problems is to design contracts so that farmers voluntarily participate and have an incentive to follow the practices designed for their circumstances.

Wu and Babcock (1996) address this issue in a simple model in which use of an environmentally damaging input and an index of land quality determine environmental outcomes, and input use is a function of land quality. They show that farmers will self-select a suitable contract (one that is designed for their land quality) when environmentally damaging input use is non-decreasing in land quality, and when per-acre payments to reduce environmental damages are non-increasing in land quality. In the case of payments for soil C sequestration, the potential to sequester soil C is highest on degraded lands, hence there is the required inverse relationship between payments (either per hectare or per tonne) and land quality. However, note that farmers with high-quality (non-degraded) land are those farmers who are using the management practices that maintain soil C (for example reduced tillage), whereas farmers with low-quality land are those using practices that cause soil C to be lost (for example conventional tillage). This means that the first part of the Wu-Babcock incentive compatibility conditions are not likely to be satisfied in the case of soil C sequestration contracts. Farmers who have been using soil C-preserving practices such as reduced tillage before a soil C contract is offered will have an incentive to claim their land is degraded so that they can receive payments even though their soil may be already saturated so that its potential to sequester additional C may be limited or zero.

This incentive compatibility issue is closely related to who would be eligible for payments under a government program to sequester soil C. A program that pays farmers to increase soil C on land that has been degraded can be viewed as penalizing those farmers who had adopted environmentally beneficial land use and management practices before the program began. Some observers have argued that programs with this type of design would create the perverse incentive for producers using reduced tillage to plow their land, release the C, and then enter the program. A solution in the case of a government program would be to provide payments to those who could document that they had changed practices previously and sequestered soil C. In the case of a private market for C that was driven by an international agreement requiring increases in soil C from a fixed baseline, credit for prior actions might not be acceptable. In that case, governments could prevent the perverse incentive problem by purchasing and holding credits from those farmers who had already adopted soil C-increasing practices.

Another way to look at the perverse incentive problem is suggested by the above discussion of incentive compatibility. Those farmers who adopted improved practices presumably did so because it was in their economic interest. Therefore, their land must exhibit the properties that violate the incentive compatibility conditions. By this logic, those producers should not receive payments for doing what they already had deemed to be in their economic self-interest. Some may counter this argument by claiming that producers are motivated by more than economic self interest. That may well be true; it is then a public policy decision whether the taxpaying public should reward those individuals for their voluntary private acts that yield social benefits.

Incentives for adoption of sustainable practices in developing countries

Sustainable agricultural development is widely acknowledged as a critical component in a strategy to combat both poverty and environmental degradation. Yet sustainable agricultural development remains an elusive goal, particularly in many of the poorest regions of the world (World Bank, 2000). While there have been successes, more typically investments in soil conservation and other sustainable practices have been adopted as part of a development project, and then abandoned after the project is completed.

There are various possible explanations for the failed attempts to promote the long-term adoption of more sustainable land use and management practices. While more research is no doubt needed this situation does not appear to be due to a lack of either research or development efforts, since in many cases technically feasible, appropriate sustainable practices are available.

The other principal explanations for the continuing trend towards land degradation are related to the lack of economic incentives to prevent degradation and encourage conservation. On-farm benefits from investments in soil conservation or other sustainable practices often accrue too far in the future to be valuable to farmers who strongly discount future benefits due to low and uncertain incomes and imperfect capital markets. These on-farm disincentives may be magnified by high transport costs and other market imperfections, adverse government policies, insecure property rights, limited availability of fodder for grazing or fuel for cooking and heating. In many cases the benefits from more sustainable management practices are external to the farm. We should not be surprised if poor farmers in the developing world, often existing at the margin of subsistence, are not willing to forgo some of their income to mitigate environmental problems they may not even realize exist or which will occur well beyond their planning horizon. (Heerink, van Keulen and Kuiper, 2001).

As we noted earlier, an important provision of the Kyoto Protocol was the Clean Development Mechanism (CDM), a process whereby the developing countries would be encouraged through subsidies from industrialized (Annex I) countries to adopt 'clean' development strategies that would emit lower rates of GHGs than they might otherwise.

These considerations have generated interest among the development community in the potential to use payments for soil C sequestration -- either through a policy such as the CDM or through a C emissions credit market -- as a mechanism to provide farmers in developing countries with the economic incentives needed to adopt more sustainable land use and management practices (Antle, 2000). Due to the fact that farmers in degraded environments are some of the world's poorest people, this type of mechanism could simultaneously contribute to the goals of alleviating rural poverty, enhancing agricultural sustainability, and mitigating GHG emissions.

A number of the issues discussed above provide insight into the economic potential to use soil C sequestration as a mechanism to help fund sustainable agricultural development. First, we noted earlier that the buyer of a C contract never actually takes delivery of the commodity, rather, the commodity is embodied in an asset (the soil) that belongs to the land owner. The creation of C contracts with farmers in developing countries may be impeded by a number of property rights issues, including uncertain tenure and land ownership, and weak legal institutions that may limit enforceability of contracts, and widespread political and legal corruption. Second, we noted that the total cost of providing soil C services is equal to the farm opportunity cost plus contracting costs (measurement, monitoring, and transactions costs). The contracting costs per tonne of C associated with negotiating contracts will decline with the size of contract, and a market for C credits is likely to operate for large, standardized contracts (for example, 100 000 tonnes). Considering that a typical farmer may be able to sequester 0.5 tonne per hectare per year, it follows that transactions costs per tonne will be high for individual small farms in developing countries. For small farms to enter into C contracts, some form of institution (such as a cooperative or marketing intermediary) will be needed to organize farmers into groups so as to lower transactions costs, or governments will need to subsidize contracting costs.

A significant component of the estimated global technical potential for C sequestration is severely degraded lands, especially in densely populated tropical regions. In some cases severe degradation is caused by mismanagement, but in many cases lands are degraded because of economic incentives to 'mine' soil fertility. In such cases the required level of investment to both alter management and cause actions to be undertaken to reverse the degradation or cause cessation of current degrading practices in favor of more sequestration friendly practices is likely to be high. Time lags and lost productivity while the land is recovering are also likely to be substantial. Thus sequestering soil C on severely degraded lands is likely to be more costly than on more productive lands where more marginal, less expensive changes are required. There are likely to be additional benefits to land restoration, including food security and other environmental benefits (see the above discussion of co-benefits).

Estimating the economic feasibility of soil C sequestration in agriculture

We now survey recent empirical studies that provide evidence of the economic feasibility of agricultural soil C sequestration in two regions of the United States and for the United States as a whole.

<u>A comparison of two recent regional studies</u>

Antle et al. (2000, 2001a) examine the regional economic potential for C sequestration on approximately 3 million hectares of cropland soils in Montana using a site-specific econometric-process model coupled with output from Century, an ecosystem model of soil C dynamics (Parton et al., 1987; Paustian et al., 1992; Paustian et al., 1996). The econometric-process model approach uses site-specific data from a production survey to estimate econometric production models for each crop, and then uses these data to parameterize a simulation model that represents the farmer's discrete land use and continuous management decisions (Antle and Capalbo, 2001). This simulation model is executed using the site-specific production data, with C rate data obtained from simulations of the Century ecosystem model parameterized for agroecozones used in the policy analysis. Simulations were conducted for per-hectare and per-tonne payments for six agroecozones in Montana. The marginal cost curves for these agroecozones show that there is substantial spatial variation in the amount and cost of C that can be produced by changing from crop-fallow rotations to continuous cropping in the dryland grain production system in that region.

Pautsch et al. (2001) use a discrete-choice econometric model in conjunction with soil C estimates from the EPIC model (Williams et al., 1989) to estimate the costs of soil C sequestration in Iowa. This model estimates the probability of adoption of conservation tillage in Iowa, using data from the National Resources Inventory, combined with county-level yield data and state-specific price data. This model is used to estimate the average annual cost of purchasing soil C for a government program that would provide payments targeted to those producers adopting conservation tillage. They consider a policy that "perfectly discriminates" across land units – this is simply a per-hectare policy based on soil C rates for each individual land units rather than for an agroecozone. They justify this type of analysis by noting that it provides a lower bound on the cost of sequestering soil C.

To compare the results of these two studies, Antle et al. (2001c) showed how the cost estimates from these two studies can be converted into common units. Figure 3 shows this comparison of the marginal cost per tonne C in Montana (MC_M) and Iowa (MC_I) for C prices ranging from $10 to $700 per tonne C. The location of these marginal cost curves illustrates both the regional differences in the marginal costs of sequestering soil C and the differences in the crop acreages and C rates in the two regions. At prices greater than $340 per tonne MC_M is almost vertical implying that any further price increases will not result in appreciably more soil C since all the cropland has been switched to the most productive means for sequestering soil C. Iowa producers can sequester a greater total amount of soil C at higher prices, due to the larger area of cropland available (9 million hectares in Iowa vs 3million hectares in Montana), the higher soil C rates in Iowa (Table 1), and the possibility of inducing more farmers to adopt reduced tillage with a sufficiently large economic incentive.

(Insert Figure 3 here.)

These analyses – and others like them that will no doubt be forthcoming in the future -- impose a number of assumptions and parameter estimates that should be subjected to sensitivity analysis. Both studies measure only farm opportunity costs, and assume efficient per-tonne payments are made and are targeted only to those farmers changing practices; no payments are made to those who previously adopted C-storing practices. Also these studies ignore the permanence issue in the sense that they do not include payments required to store C after saturation occurs. Antle et al. (2000) compute costs for less efficient per-hectare payments for increasing cropping intensity and for conversion of crops to permanent grass, and find the costs may be several times higher due to spatial heterogeneity and due to the higher opportunity costs of converting crops to grass. Antle et al. (2001b) assess the effects of uncertainty in the measured C rates used in the Antle (2000) and Antle (2001a) studies. They find that errors in measured C rates are magnified in estimates of marginal costs, due to the non-linear relationship between C rates and marginal cost. Further research should address the effects of uncertainties in other key parameters that affect farm opportunity cost, such as the possible effects of C sequestration practices on crop productivity.

An analysis of C sequestration in US Agriculture

McCarl and Schneider (2001) and McCarl et al. (2001b) investigated C sequestration for the United States as a whole using a mathematical programming, sector wide modeling approach to develop information on the marginal abatement cost curve describing how much GHG emissions can be offset for different effective C prices considering a number of emission and sequestration possibilities. Their analysis considered changes in US agriculture and forestry (AF) including afforestation, biofuel production, changes in crop management (mix, input use, tillage, irrigation), livestock management and manure management. They examined the relative role of AF sequestration efforts in the total portfolio of potential agricultural responses at alternative C price levels.

The McCarl and Schneider (2001) modeling approach involves simulation of the amount of GHG net emission reduction produced in the AF sectors and the choice of strategies under alternative C prices. They used the Agricultural Sector Model (ASM) (McCarl et al., 2001a) modified by Schneider (2000) to include GHGs (hereafter called

ASMGHG) coupled with data from a forestry sector model (Adams et al., 1996). ASMGHG depicts production, consumption and international trade in 63 US regions of 22 traditional and 3 biofuel crops, 29 animal products and more than 60 processed agricultural products. Environmental impacts such as levels of greenhouse gas emission or absorption for CO_2, methane and nitrous oxide plus chemical use, and soil erosion were included. ASMGHG simulates the market and trade equilibrium in agricultural markets of the United States and 28 major foreign trading partners. The model incorporates domestic and foreign supply and demand conditions, as well as resource endowments. The market equilibrium provides: commodity and factor prices; levels of domestic production; export and import quantities; management adoption; resource usage; and environmental impact indicators. ASMGHG was subjected to C prices from $0 per tonne to $500 per tonne, using 100-year global warming potentials of 21 for methane and 310 for nitrous oxide to put them on a tonne C equivalent basis.

Analysis using ASMGHG shows that AF provides cost effective emissions offsets particularly through soil sequestration. Figure 4 shows the amount of C offsets gained at C prices ranging from $0 to $500 by broad category of strategy. Low cost strategies involve afforestation, soil C sequestration, fertilization and manure management. Another finding is that a portfolio of AF strategies seems to be desirable including biofuels, forests, agricultural soils, methane and nitrous oxide based strategies. Figure 4 also shows that different strategies take on different degrees of relative importance depending on the C price level. Reliance on individual strategies appears to be cost increasing. For example reliance solely on agricultural soil C shows it would cost about $30 per tonne to achieve 60 MMT, whereas by using the total portfolio leads to a cost below $15 (adding all the contributions from Figure 4).

(Insert Figure 4 here.)

The analysis also shows that there are important differences between technical, economic and competitive economic potential. As argued earlier estimates of the technical potential to sequester C in soils ignore the factors of cost and resource competition. Lal et al. (1998) for example compute a total agricultural soil C potential but do not specify the cost of achieving such a potential level of sequestration. The total technical potential in the ASMGHG model if only C is maximized and other resource constraints are ignored is 75 MMT annually, but relying only on soils a maximum of 70 MMT can be produced at a price approaching $300 per tonne. When sequestration strategies are considered simultaneously along with other strategies the maximum rate of sequestration is 94 per cent of the single strategy amount and declines at prices above $60 per tonne due to resource competition.

The ASMGHG model also shows that mitigation based offsets are competitive with food and fiber production. Achieving net GHG emission offsets requires changes in the AF sector. For example, the results show that corn prices more than double at C prices above $300 as land competition rises due to afforestation and biofuel usages of croplands.

Finally this model shows that there can be substantial co-benefits. McCarl and Schneider (2001) found as much as 25 per cent reductions in nitrogen and phosphorous runoff and in erosion. At higher prices, environmental co-benefits stabilize as a result of increased biofuel production.

Markets for C: can agriculture compete?

Trading emissions allowances can be an efficient mechanism to implement an emissions reduction policy, because the regulated firms have an incentive to allocate emissions allowances so that the marginal costs of emissions reductions are equated across all sources (Tietenberg, 1985). Market trading schemes also are desirable because they encourage innovation and faster adoption of more efficient technology as compared to command and control approaches to regulation (see Joskow et al., 1998). A trading scheme for sulfur dioxide emissions was authorized in the Clean Air Act of 1990, with the first trades taking place in 1992 (see Petsonk et al., 1998; Joskow et al., 1998). US farmers are participating in emerging markets for other environmental goods such as water and for nutrient emissions (see Colby et al., 1993; Landry, 1996; Schleich and White, 1997). GHG allowance trading was incorporated into the Kyoto Protocol, and could provide the mechanism for the implementation of a soil C sequestration policy in which farmers would sell C credits to industries that emit GHGs.

In a C market C credits would be supplied by firms that could reduce emissions at relatively low cost, or by entities that sequester C if they can do so at a cost that is competitive with emissions reductions. There are a variety of estimates of the cost at which emissions reductions could be obtained and thus of the probable price of C if a market were to emerge. These as well as other complications associated with large sectoral models are involved in estimating the likely price of a tonne of C. Wiese and Tierney (1996) used a model of the US economy to estimate the effects of meeting C emissions goals of the KP through taxes on the C content of energy. They considered two scenarios, a \$110 (1987 dollars) per tonne C tax that stabilized emissions at 1990 levels by 2020, and a \$162 (1987 dollars) per tonne C tax that reduced emissions by 20 per cent from their 1990 levels by 2020. These taxes were levied on the production of coal and on the production and imports of crude oil and natural gas. In another model-based analysis the Clinton Administration estimated that under a global emissions trading system, US compliance with the Kyoto Protocol would require a 3 to 5 per cent increase in energy prices for US households in the period 2008-2012, corresponding to a C emissions tax in the \$14 to \$23 per tonne range (Yellen, 1998; Council of Economic Advisers, 1998). Also Weyant and Hill's report (1999) that a multi model, Energy Modeling Forum sponsored study of non agricultural Kyoto compliance found costs that averaged from \$44 - \$89 per tonne depending on the trading assumption and in the individual model results as high as \$227.

Experience with the SO_2 trading program that was initiated under the 1990 clean air legislation suggests that the cost of reducing C emissions may be less than what economic models predict. Before SO_2 trading began economists estimated that an emissions credit would trade for over \$300 per tonne. After the market was implemented the price has been in the range of \$100 per tonne on the Chicago Board of Trade (Joskow et al., 1998). It seems likely that the input-output models and econometric models over-estimated the cost of abating SO_2 emissions because they use aggregate historical data and because they fail to account for the abatement opportunities that can be made when economic agents are rewarded for reducing emissions efficiently. Sandor and Skees (1999) used this logic and evidence on the cost of reducing industrial CO_2 emissions to estimate that a market in tradeable C emissions credits could price C in the range of \$20 to \$30 per tonne.

Kopp and Anderson (1998) argue that C prices as low as those estimated by the Clinton Administration and by Sandor and Skees (1999) are not likely given the various practical considerations that may limit the effectiveness of a global emissions trading system. They also note that, if such trading were to take place under the Kyoto Protocol, only emissions trading among the developed countries whose emissions are restricted would be allowed. A similar analysis to the Clinton Administration's with trading only among the developed countries showed that C emissions costs would be at least \$72 per tonne. Under the assumption that the United States would meet a larger share of its emissions reductions commitments through reductions in energy consumption, the estimated costs of compliance are even higher, as indicated by the Wiese and Tierny (1996) study discussed above.

These data suggest that at least the lowest-cost producers of soil C could compete with forestry and with industrial emissions reductions in a market for C credits. However, the marginal cost curves in Figure 2 also indicate that for agriculture to be a major player in the provision of C credits, the price per tonne of C would probably have to be at least \$50 or higher. Alternatively if farmers received credit for the various co-benefits of environmentally positive practices, including increasing soil C, then the amount paid for the C component alone would not have to support the adoption of the improved practices. Under these conditions soil C sequestration appears to have considerable potential in a broader green payment program such as has been proposed in recent US legislation.

Conclusions

This paper reviews the literature on the technical feasibility of soil C sequestration and the emerging economics literature on the economic analysis of soil C sequestration. The key economic question to be answered is whether C can be sequestered in agricultural soils at a cost that is competitive with industrial emissions reductions and other sinks for C such as forests. In this paper we survey the issues that arise in designing contracts for soil C sequestration, and review the empirical evidence to date.

The economic cost of sequestering soil C depends on both the on-farm opportunity costs of sequestering soil C and contracting costs. The farm opportunity costs are highly spatially variable, raising issues of how to design contracts that take this spatial variability into account in an efficient manner. Soil C is not easily observable like trees, therefore the cost of measuring soil C stocks and changes over time is a key issue that needs to be addressed. Similarly, monitoring compliance with land use and management practices specified in contracts raises issues of incentive compatibility. The fact that C stored in soil can be released if practices are not maintained over time raises the issue of permanence of soil C. By viewing soil C contracts as contracts for accumulating and storing C in soil for a specified period of time, the permanence issue can be resolved. Practices that sequester soil C may produce important environmental co-benefits, such as reduced soil erosion, that should also be counted if farmers are to be provided appropriate incentives for adopting practices that sequester soil C and produce other benefits.

Empirical studies for Iowa, Montana and the United States as a whole provide estimates of the marginal cost of sequestering C in soil that range from near zero to hundreds of dollars per tonne C, depending on how much C is to be sequestered. The US wide analysis indicates that the most efficient policy would utilize a mix of greenhouse gas mitigation methods, including soil C sequestration. All of these studies show substantial potential for agriculture to play a role at low carbon prices (below $50 per ton). Further research to better develop the marginal abatement curves for agricultural sources is needed to assess the best greenhouse gas strategy for each region of the United States and for other regions of the world.

Payments for soil C sequestration has been promoted as a way to help provide farmers in the developing countries to adopt more sustainable land use and management practices. While many regions in the developing world do appear to have a high technical feasibility to sequester soil C, the costs of rehabilitating highly-degraded lands may be high, and it remains to be seen where soil C sequestration would be economically feasible. Lack of well-defined property rights, fragmented land use, and related incentive problems may limit the feasibility of soil sequestration in many regions.

Notes

1. We use the word tonne to mean 1000 kilograms or one metric ton.

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Table 1. Annual increase in soil C from management changes: Iowa and Montana.

Change in management	Additional carbon[1] (tonnes/ha/yr)
Iowa – Change in tillage practice	
Intensive tillage to moderate tillage	0.19
Intensive tillage to no-till	0.52
Montana – Change in cropping system	
Spring wheat fallow to continuous spring wheat	0.38
Spring wheat fallow to continuous winter wheat	0.44
Permanent grass to continuous spring wheat	0.06
Permanent grass to continuous winter wheat	0.13

[1]Estimates were calculated using Century (Parton et al., 1987; Paustian et al., 1992, 1996) and provided by the Natural Resource Ecology Laboratory, Colorado State University
Source: Antle et al. (2001c).

Table 2. Rates of potential carbon gain under selected practices for cropland (including rice land) in various regions of the world.

Practice	Country/ Region	Rate of Carbon Gain (t C ha^{-1} y^{-1})	Time[1] (yr)	Other GHGs and Impacts
Improved crop production and erosion control	Global	0.05–0.76	25	$+N_2O$
– Partial elimination of bare fallow	Canada	0.17–0.76	15–25	
	USA	0.25–0.37	8	$\pm N_2O$
– Irrigation water management	USA	0.1–0.3		
– Fertilization, crop rotation, organic amendments	USA	0.1–0.3		$+N_2O$
– Yield enhancement, reduced bare fallow	Tropical and subtropical China	0.02	10	
– Amendments (biosolids, manure, or straw)	Europe	0.2–1.0	50–100	
– Forages in rotation	Norway	0.3	37	
– Ley-arable farming	Europe	0.54	100	
Conservation tillage	Global	0.1–1.3	25	$\pm N_2O$
	UK	0.15	5–10	
	Australia	0.3	10–13	
	USA	0.3	6–20	
		0.24–0.4		
	Canada	0.2	8–12	
	USA and Canada	0.2–0.4	20	
	Europe	0.34	50–100	
	Southern USA	0.5	10	
		0.2	10–15	
Riceland management				
– Organic amendments (straw, manure)		0.25–0.5		$++CH_4$
– Chemical amendments				$--CH_4$, $+N_2O$
– Irrigation-based strategies				$--CH_4$

[1]Time interval to which estimated rate applies. This interval may or may not be time required for ecosystem to reach new equilibrium.
Source: Watson et al. (2000), Table 4-5.

Figure 1. Time path and saturation point for soil C in response to a change in land use or management practices.

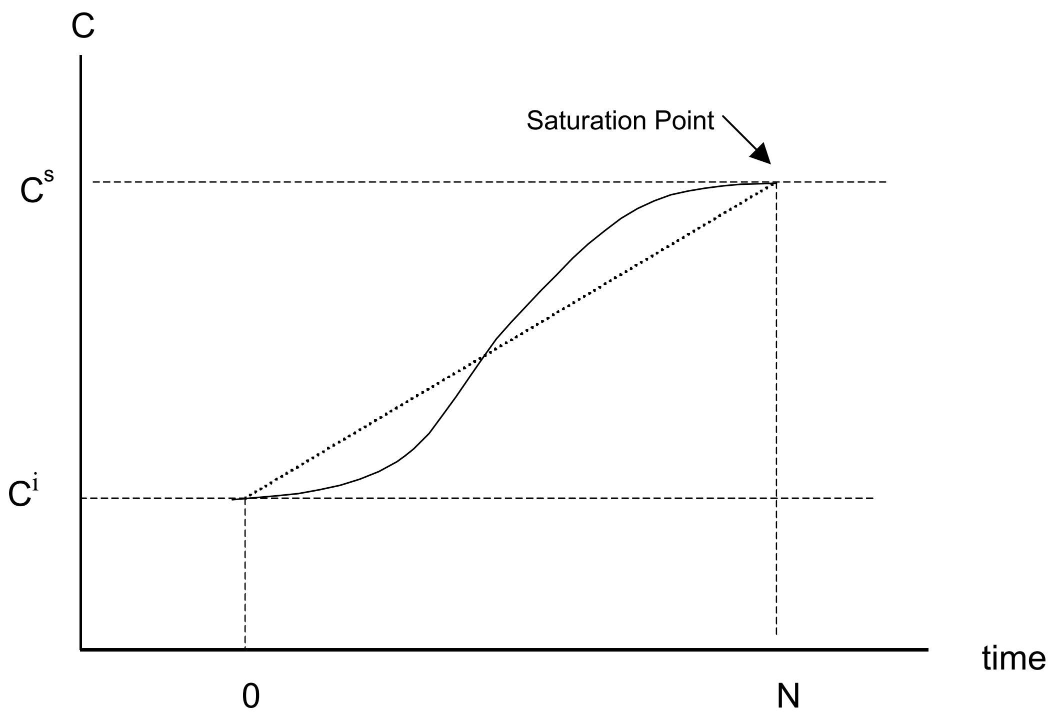

Figure 2. Marginal cost functions for per-hectare and per-tonne payments.

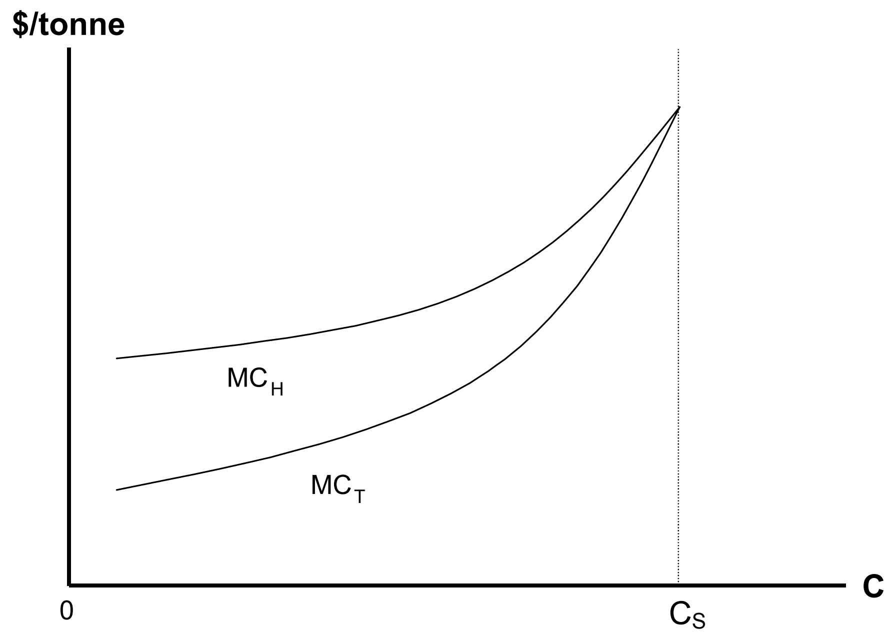

Figure 3. Marginal cost of soil C sequestration under a per tonne payment scheme in Iowa for conservation tillage and Montana for cropping intensification. (Source: Antle et al., 2001c).

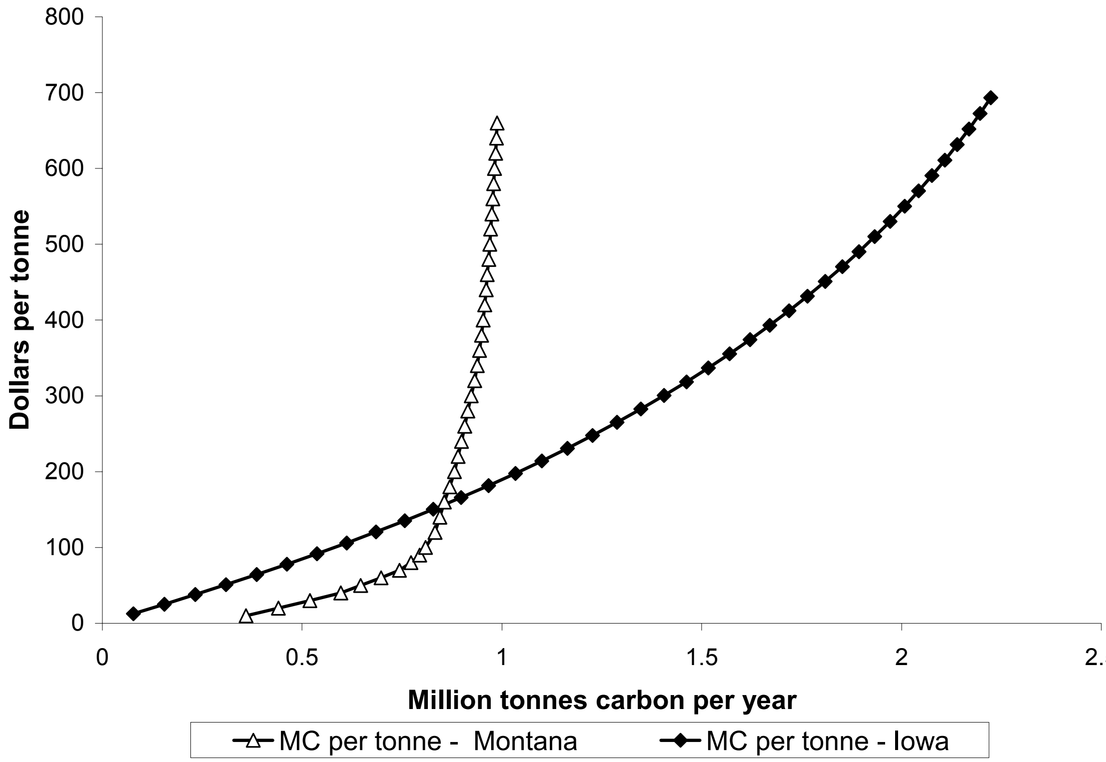

Figure 4. Agricultural mitigation potential at $0 to $500 per ton carbon equivalent prices.