# Modeling Agroecosystem Services for Policy Analysis *

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Agriculture is a critically important managed ecosystem, and ecosystem services (ES) are the most important non-market goods provided by agriculture. ES are the benefits accrued from services naturally provided by the environment from which human beings and other organisms benefit. They are "the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life" (Daily, 1997) by "purifying air and water, detoxifying and decomposing waste, renewing soil fertility, regulating climate, mitigating droughts and floods, controlling pests, and pollinating plants" (Salzman, Thompson, and Daily, 2001).

The scientific literature establishes that farmers' land management decisions may affect biological and physical systems through a number of mechanisms. These effects may be limited to the land owned by the farmer, such as a change in soil productivity, or may have off-site effects such as chemical runoff into surface waters. Moreover, some ES such as climate regulation may also be public goods, meaning that they benefit everyone regardless of who pays for them. Consequently, the value of most ES are not embodied in the prices of conventional agricultural products, and therefore these markets do not provide farmers with the economic incentives to supply the amount of ES demanded by society. A socially efficient agricultural policy would provide incentives for farmers to supply the appropriate combination of market and non-market goods demanded by society (Antle and Capalbo, 2002; Antle 2007).

To create incentives for the efficient provision of ES, some form of government intervention will be required, either through direct regulation or through the creation of incentives for farmers to increase the supply of ES. In either case, to support the transition from commodity-based subsidies to policies based on the efficient provision of both market goods and ES, policy decision makers will need to match society's demand for ES with the agricultural sector's ability to supply ES. For example, the 2002 Farm Bill created the Conservation Security Program which pays farmers who adopt environmentally beneficial practices. The 10-year costs of implementing this program were estimated to be $2.1 billion in 2002, but rose to $8.9 billion in 2004. This cost underestimation was due in part to inaccurate estimates of farmer participation (General Accounting Office, 2006). This example suggests that estimates of benefits and costs of these policies with a reasonable degree of accuracy will be needed to facilitate the transition to policies based on provision of ES. Additionally, to be useful for policy decision making, information needs to be provided in a timely manner. Researchers will inevitably need to trade-off cost, timeliness and accuracy in making decisions about the appropriate modeling approach.

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California agriculture has well-known and substantial impacts on ES, ranging from the local ESs such as water and air quality to global ESs such as maintaining biodiversity and regulating global climate. Current legislative initiatives are being discussed that could begin the process of research and policy development needed to create incentives for ES provision. In this paper I discuss one such incentive mechanism known as payments for ES. PES are payments based on the amount of additional ES that are provided by making changes in land management. Clearly, there is a need to develop the capability to evaluate and anticipate the ecological and economic impacts of such incentive mechanisms to support the design of policies to increase the supply of ES.

In this paper I outline basic concepts and measurement issues that arise in estimating the supply side of ES for policy design. By the supply of ES, I mean the additional ES that can be obtained by providing farmers with incentives to change their land use and management practices in ways that increase ES above and beyond the level that would have been obtained without such incentives. I present an example that illustrates the use of a "minimum-data" approach to ES service supply modeling that can provide information on a timely basis to support policy decision making.

## Basic Components of Agroecosystem Service Supply

There are a number of ways to create incentives for an increase in ES supply, including direct regulatory intervention and the use of incentives such as a tax or a cap-and-trade system. We now know that incentive-based approaches to environmental regulation are generally more efficient than command-and-control regulation. One reason for the efficiency advantage of incentive-based regulation is that the cost of reducing emissions often varies substantially from one emissions source to another. There are two key reasons why this cost heterogeneity is particularly important for ES.

First, the technical potential for ES varies from place to place. Technical potential is the maximum amount of ES that can be provided at a site regardless of cost. Technical potential varies across locations because of differences in site-specific conditions, such as soils and climate that impact the biological processes involved. When we sum up the quantities of ES that could be supplied across all sites in a region, we obtain the technical potential for the region.

Second, the economic potential for ES supply is also highly spatially variable. The economic potential is defined as the amount of ES that can be profitably supplied at a site or in a region. Economic potential can be represented using a supply curve which shows the additional quantity of ES that would be provided at each P, as illustrated in Figure 1. Assuming for simplicity that prices are constant over the planning horizon, we can portray the logic of ES supply as follows: the farmer will participate in a contract to supply E units of ecosystem service per unit time at price P if $P > C/E$, where C is the opportunity cost to farmers of increasing the service. The opportunity cost C represents the change in economic returns caused by taking actions to increase the ES. C may include differences in returns to the alternative management practices, additional investments needed to increase ES, and any transaction costs associated with entering into ES contracts. The ratio $C/E$ measures the opportunity cost per unit of ES, so this equation tells us that farmers will increase the supply of ES if the incentive to increase ES is greater than the cost of doing so. Both E and C vary from place to place, so when P is low, only those farmers who can increase ES at low cost will participate in ES contracts. As P increases, so does the number of sites where it pays farmers to increase ES.

In summary, the two key elements that need to be quantified to estimate ES supply are the opportunity cost of changing practices, and the increase in ES that results from changes in practices. In the following sections I discuss each component in turn.

## Quantifying the Opportunity Cost of Changing Practices

The basic component of the opportunity cost of changing practices is the difference in returns between the base practice (the practice that would have been used absent the offer of an ES contract) and the alternative practice that increases ES. Recent research on simulation of ES supply has utilized site-specific data to construct detailed econometric models that can be simulated to estimate returns to practices on a site-specific basis and thereby construct a representation of the spatial distribution of opportunity cost (e.g., Pautsch, et al. 2001; Antle et al., 2003; Feng, Kling and Gassman, 2004; Wu et al., 2004; Lubowski et al., 2005; Antle et al., 2007). This spatial distribution can then be used to derive an ES supply curve as portrayed in figure 1. However, high-resolution bio-physical and economic data with the geographic coverage needed for analysis of agriculture-environment interactions, such as the National Resources Inventory data in the United States, are exceptional and provide limited economic information. In most places including California, site-specific economic data are only available from special-purpose farm surveys, and the time and resources required to undertake special-purpose surveys precludes their use for most policy analysis.

An alternative approach to the use of highly detailed, site-specific data and models is the minimum-data (MD) approach recently proposed by Antle and Valdivia (2006). The MD approach to ES service supply is based on the observation that the structure of the problem allows the spatial distribution of opportunity cost to be estimated in relatively simple terms. The MD approach uses average crop yields and costs of production to estimate the mean of the distribution of opportunity cost, and uses the spatial variation in yields to estimate the spatial variation in returns. This approach has been used successfully to estimate ES supply curves that approximate supply curves derived from more detailed models, such that the policy implications of the analysis would not be affected (Antle and Valdivia, 2006). In California, county-level data collected by USDA as well as the state government are available to implement this type of analysis.

The economic analysis of ES supply based on the opportunity cost of changing practices provides a first approximation to the the incentives that would be needed to induce an increase in supply. However, in addition to the opportunity cost of changing practices, there may be other costs of adjustment associated with changing land use and management. These costs of adjustment may involve capital investments or learning about alternative management practices. In addition to adjustment costs, there may be a variety of other behavioral and institutional factors that influence farmers' willingness to change land use and management practices. There is a sizeable literature on the adoption of conservation practices in agriculture that is closely related to the problem of ecosystem service supply. The literature shows that characteristics of farm decision-makers affect their willingness to adopt conservation tillage, although how they impact decisions appears to depend on their geographic location and other factors (e.g., Fuglie and Kascak, 2001). In addition, the literature on technology adoption shows that risk and uncertainty effectively raise the perceived costs of changing practices (Sunding and Zilberman, 2001). Risk is most likely to impact decision making when there is a substantial difference in risk associated with the land use options, e.g., as would be the case when farmers are choosing between risky crop production and a riskless government payment for idling land.

Adjustment costs and risk premiums can be difficult to estimate, particularly for new practices that have not been observed much in use. Economic-engineering analysis can be used to estimate changes in costs of production, and past studies of technology adoption can be used to estimate risk premiums. Mean values of these components of the opportunity cost can be factored into analysis of ES supply, together with sensitivity analysis to determine whether these factors are likely to have a strong influence on the analysis. The effect of these additional costs is to shift the supply curve upward, meaning that a higher economic incentive is needed to induce any given quantity of ES.

Another factor that is likely to affect farmers' willingness to participate in ecosystem service contracts is transaction costs. These costs include the time and other resources farmers spend learning about the ecosystem service contract, as well as costs of verifying compliance with the contract. In addition, when the processes governing the provision of ecosystem services are spatially dependent, efficient provision may require cooperation among groups of farmers within an agro-ecological zone. These coordination costs are likely to depend on factors such as the number of farms participating, the number of hectares under contract, and the number and frequency of verification measurements required for contracts. If these costs are allocated to participants according to the number of hectares under contract, then the net benefit of contract participation (equation 1) is modified by subtracting transaction costs. If these transaction costs do not vary spatially, they simply shift the mean of the spatial distribution of net benefits in the negative direction. In terms of the ES supply curve, the effect is to shift the curve upward and create a threshold price below which farmers will not enter contracts.

While some analysts have suggested that transaction costs may be high for ES service supply contracts, recent research as well as experience with pilot projects indicate they will not be higher than transaction costs associated with other types of financial instruments. Antle et al. (2003) and Mooney et al. (2004) have shown in a study of carbon sequestration, for example, that transaction costs for implementing efficient contracts would be less than 5 percent of the contract cost. Pilot projects for carbon sequestration implemented brokered by the National Carbon Offset Coalition through the Chicago Climate Exchange have transaction costs on the order of $0.15 per acre, also showing that for carbon prices expected to prevail when the US has a binding carbon emissions cap in the range of $30/MgC or higher, transaction costs will not be prohibitive.

Measuring Ecosystem Services

Ecosystem services are produced through a number of biological and physical processes that depend on site-specific environmental conditions (soils, other physical characteristics such as slope and aspect, climate, proximity to other resources such as surface water) and on human activity (agricultural production and related land use and management activities). In general terms, we can express the provision of ecosystem services as a function ES = f(B,X) where B represents bio-physical conditions at a site (e.g., a farmer's field) and X represents management at the site, such as crop planting, tillage, and fertilizer and pesticide use. Research in agronomy, crop sciences and environmental sciences has greatly advanced our understanding of the processes underlying the function f(B,X) for a number of important ES such as water quality and quantity, carbon, and wildlife habitat. Some other ES such as biodiversity are less well understood. It is clear that these relationships are complex, varying spatially and temporally. This fact leads many people (both scientists and others) to conclude that quantifying ES is a futile exercise, at least in ways that would allow ES to be effectively managed through policies that would create incentives for ES service contracts with farmers and other land managers. However, I will argue here that both practical experience and recent research indicate that ES can be quantified well enough for their management through incentive-based policies.

A first consideration is that many quantitative tools that scientists have developed to simulate ES are highly complex, process-based models that typically require a large amount of detailed data. It is very costly and largely impractical to use these kinds of models to estimate ES for policy analysis, and this fact leads many scientists to conclude that it is not possible to estimate ES for policy analysis. This conclusion is based on several false premises, however. On the one hand, standards of accuracy expected for scientific publication of results is completely different than what may be useful for policy analysis, where often simply knowing the right order of magnitude is very useful and a substantial improvement over qualitative analysis. On the other hand, the spatial and temporal scales useful for policy analysis are usually different than those used for process-based research models. Policy analysis typically deals with populations, not with individuals, and on annual or longer time frames, whereas research models may operate on spatial units of a square meter and daily time steps. What is needed for policy analysis is much simpler models that are designed to achieve a reasonable approximation based on the most important factors determining the outcomes of interest. For example, the Century model, developed to predict soil carbon dynamics, is probably the most widely known research model for anlaysis of soil carbon sequestration. Yet, this model is difficult to use for landscape-scale analysis of the type needed for policy analysis because it has hundreds of parameters and required highly detailed, site-specific land use history, soils and climate data. Research has shown that much simpler models can provide comparable estimates of soil carbon sequestration that are adequate for policy analysis and can be produced at much lower cost in terms of time and data resources (Antle, Stoorvogel and Valdivia 2007).

Experience has shown that understanding the main factors that determine ES, i.e., simply knowing the key arguments in the relationship ES = f(X,B), allows policies to target land that has a potentially high impact on ES supply and thereby substantially increase ES supply cost-effectively. One example is the Conservation Reserve Program (CRP). When the CRP was first implemented, criteria for participation were based on simple "erodibility" factors that ignored another key factor for water quality protection, namely proximity to surface water. Proximity to surface water was incorporated into more recent contracts, allowing the CRP to have much more impact on water quality while reducing the cost of the program. Likewise, it is clear that information on the amounts of chemicals applied can substantially increase the efficiency of policies designed to reduce water contamination.

Recent research has further confirmed the potential for relatively simple ES service indices to facilitate policies that are much more economically efficient. One example is the Environmental Benefits Index created by USDA to facilitate targeting conservation payments. A recent study by the Economic Research Service of USDA showed that policies based on paying farmers based on the amount of services they provide (i.e., for performance), rather than paying simply based on the adoption of certain practices regardless of the amount of ES provided, substantially increases the amount of ES provided per dollar spent (USDA, 2005). Similarly, a recent study of soil carbon sequestration showed that paying farmers per ton of carbon sequestered was up to five times more efficient that paying farmers based on their management practices regardless of how much carbon is sequestered (Antle et al., 2003).

Despite the site-specificity and complexity of processes governing ES supply, in some cases these spatial variations average out when we add up effects across land units. The recent study of carbon sequestration by Antle et al. (2007) shows this result, where using a single average rate of carbon accumulation over a large part of the central US produced almost the same estimate of C sequestration as an analysis based on C accumulation rates estimated county-by-county.

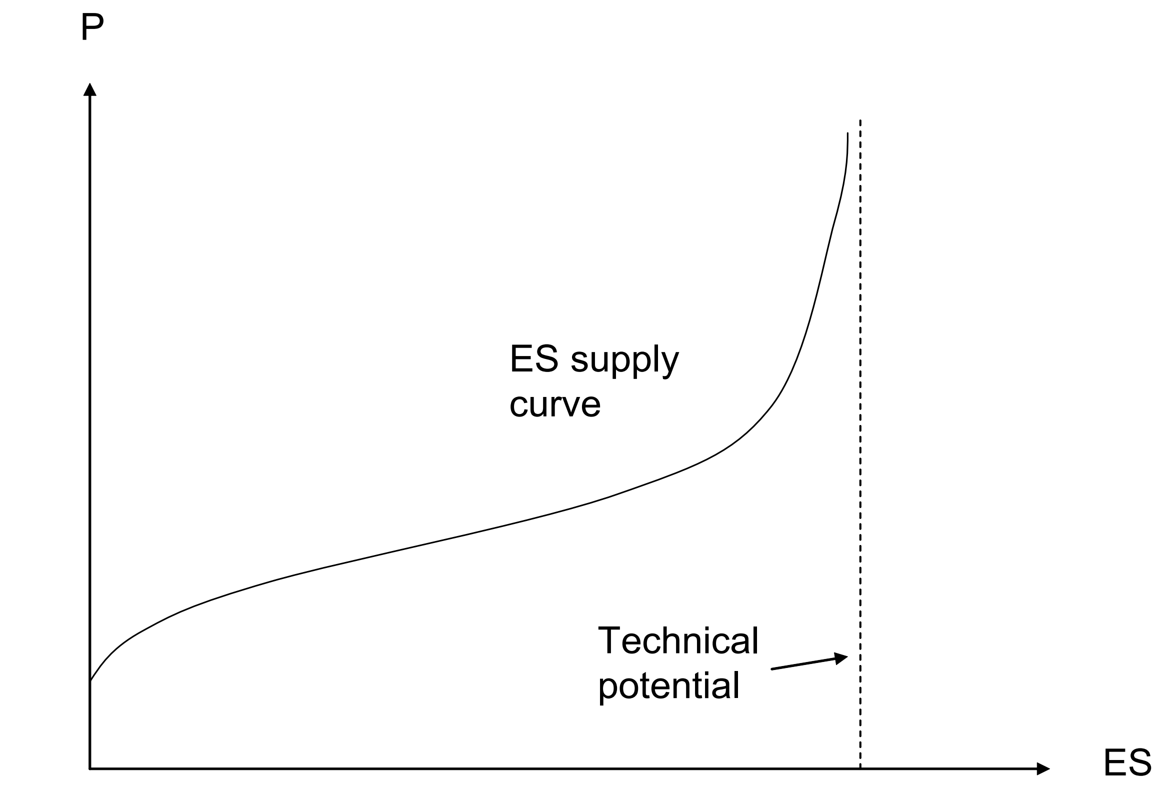
One challenge that will need to be addressed, particularly in places like California, is how to deal with multiple ES services. In many places in the Central Valley, for example, agriculture may sequester carbon, provide wildlife habitat in wetland areas, and affect water and air quality. This is a topic that will require further research, but a few observations can be made here. First, in some cases one indicator such as soil carbon may be a good proxy for a larger set of ES that are highly correlated with it. In that case, a simple index of multiple services may also work well. Second, when there are multiple ES that are not highly correlated, then economic theory shows that multiple policy instruments may be required to effectively deal with multiple ES, and inevitably there will be tradeoffs between different types of services because they do not all move together.

## An Example: Introduction of Switchgrass for Biofuels and C Sequestration

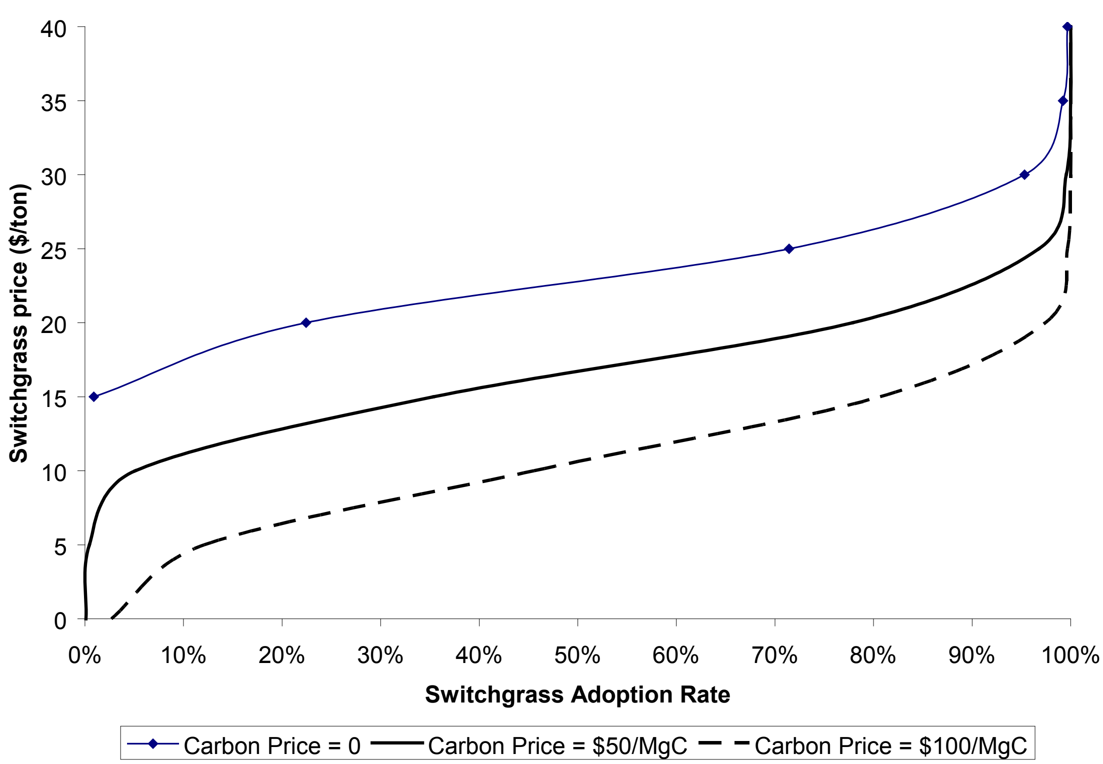
To illustrate some of the above concepts, I conclude with a recent MD analysis to estimate the potential for biofuels production and ES supply in the form of soil C sequestration (Antle, Archer and Hanson, 2007). Currently, policy decision makers in USDA and other federal agencies and Congress are seeking information regarding policies that could encourage alternative sources of energy, particularly to reduce dependence on oil, motivated by trade, national security and environmental concerns. A key element in this strategy appears to be biofuels, particularly the use of corn to produce ethanol to blend with or replace gasoline. There is also interest in the potential for alternative crops such as switchgrass that could be used for ethanol production and that would have a more positive impact on net greenhouse gas emissions than corn-based ethanol.

The MD analysis uses data obtained from existing sources to estimate the mean crop yields as well as yield variability (NASS, 2007; Berdahl et al., 2005; Vogel et al., 2005), and mean costs of production (Swenson and Haugen, 2006), for agro-ecozones in North Dakota. The rate of adoption of switchgrass was simulated as a function of the price of switchgrass, taking as given the price of wheat and the price that farmers could receive for sequestering soil carbon, receiving credit for 0.33 MgC/ha/season. Figures 2 and 3 present results of the analysis. Figure 2 shows switchgrass adoption curves over a range of switchgrass prices from zero to $40/ton, with each curve simulated assuming an alternative carbon price (zero, $50/MgC, and $100/MgC). Figure 3 shows the adoption curve that results as the carbon price is varied, while holding the switchgrass price fixed at $20/ton. These prices represent the payments farmers would need to receive at the farm gate. The figures show that switchgrass adoption is likely to be highly sensitive to the prices of switchgrass and carbon. A positive price for carbon sequestration also would substantially encourage conversion of wheat to switchgrass. While it is not clear what price would be paid for use of switchgrass for ethanol production, the U.S. department of energy has identified a target feedstock cost of $30 per dry ton delivered to a biorefinery (INEEL, 2003). With transportation costs from the farm of $5-15 per ton, this results in a farm-gate price of $15-25 per ton. Figure 2 shows that the amount supplied would respond greatly to prices in that price range. Without payments for carbon, a $30 price would generate substantial switching out of grain production into switchgrass. At a lower price such as $20/ton, the willingness to produce switchgrass would be quite low without carbon credits, but would increase substantially in response to a positive carbon price, particularly if it were in the range of $50/MgC or higher (a price similar to what is being paid currently for carbon credits in the European Union, where the price is currently about $15 per metric ton of $CO_2$, and also the price specified in recent US legislation to create a carbon trading system). Likewise, Figure 3 shows that given a switchgrass price in the range of $20, there would be a rapid increase in adoption of switchgrass as the carbon price increased from zero to $50/MgC, from about only 20% participation with a zero carbon price to about 80% with a $50 carbon price. Figure 3 also shows that a transaction cost of $1/acre would have little impact on participation, contrary to the claims by

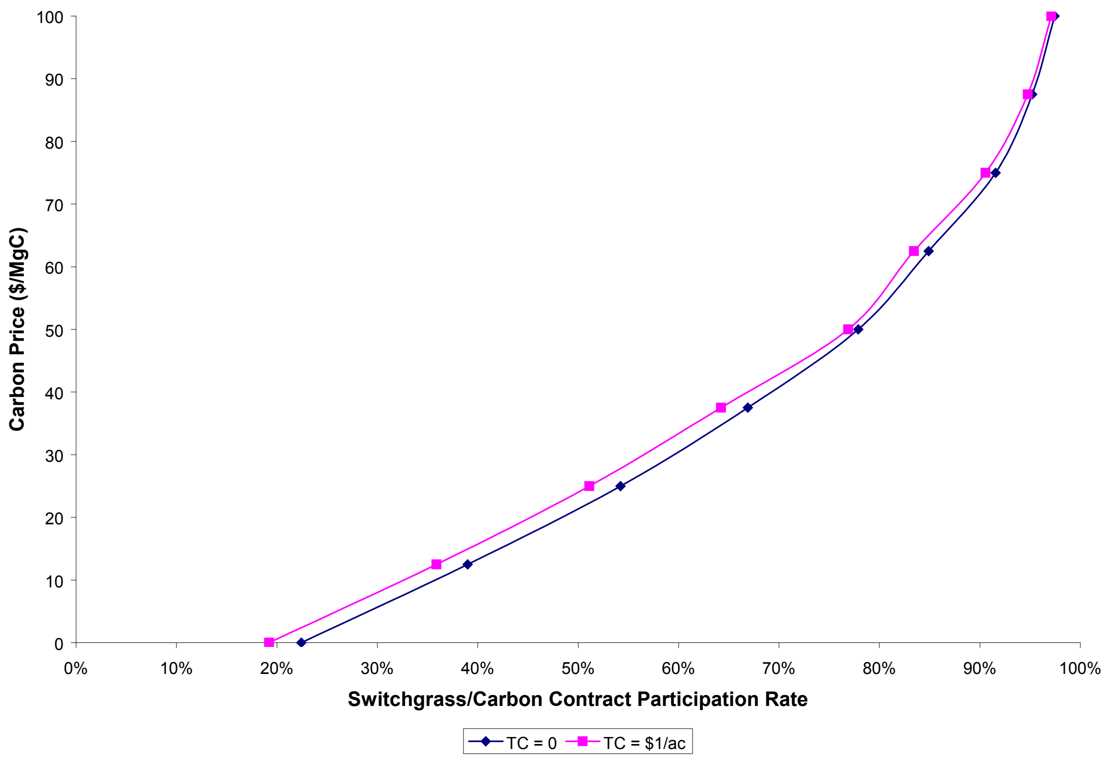
## Figure 1. Technical potential for ES Supply and the ES Supply Curve




## Figure 2. Switchgrass Adoption Curves for North Dakota Wheat Production under Alternative Carbon Price Assumptions



## Figure 3. Switchgrass/Carbon Contract Participation Rates for North Dakota Wheat Production with a $20/ton Switchgrass price, and with zero and $1/acre transaction costs




some observers that transaction costs would be likely to derail farmers' participation in a carbon market. This is the type of information that the USDA needs to design biofuels and related policies. This analysis was implemented with about one week of time from a small team of researchers.

### Conclusions

I conclude with the optimistic view that we have a great deal of relevant knowledge, data and models to quantify ES changes and their costs to be useful for implementation of relatively efficient policies. A great deal of data and many relevant models are available to implement MD-type analysis of ES supply in California. While there will inevitably be a period of learning how to implement ES policies effectively, there do not appear to be major technical or economic barriers to doing so. The most important factor at this point is the implementation of policies that create an incentive for increase ES supply and stimulate the process of learning how to design effective mechanisms to implement ES contracts.

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