IMPACT OF US AND EUROPEAN BIOFUEL POLICIES ON FOREST CARBON.

Selected Paper No. 11359

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Selected Paper prepared for presentation at the Agricultural & Applied Economics Association's 2010 AAEA, CAES & WAEA Joint Annual Meeting, Denver, Colorado, July 25-27, 2010.

Acknowledgement: The authors thank the US Environmental Protection Agency, Climate Change Division; the Climate and Environmental Sciences Division of the Office of Biological and Environmental Research in the US Department of Energy; the Ohio State University program on Climate Water and Carbon, and the Ohio Agricultural Research and Development Center for financial support to conduct this modeling work.

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ABSTRACT

This paper develops a dynamic, regional analysis of the effects of US and European biofuel mandates on land use, forestry stocks, and carbon emissions. The results suggest that these mandates may cause an additional 23-26 million hectares of forestland losses globally, but additional carbon emissions of 1.2 – 1.6 billion t CO_2. The estimates are found to be sensitive to the elasticity parameter on the land supply function in the model, with the higher elasticity estimates associated with larger carbon losses. The regional analysis turns out to be quite important, because some regions end up gaining forestland and increasing carbon stocks. The regional and dynamic effects have been missed by most other noteworthy analyses of the induced land use effects of biofuel policies, potentially leading the authors to overstate the impacts by 3-6 times.

Keywords: Biofuel policy, Indirect land use effects, Forest carbon sequestration

INTRODUCTION

Concerns about global warming and rising energy prices have driven international interest in biofuels as cheaper, sustainable alternatives to traditional fossil fuels. The United States, for example, passed legislation mandating a substantial increase in the volume of biofuels consumed from less than 9 billion to 36 billion gallons by 2022 (Energy Independence and Security Act of 2007). The European Union, the world's largest biodiesel producer, set a target that 10% of transport fuel should be composed of biofuels by 2020 (European Commission, 2007). From an environmental perspective, biofuel policies are to some extent based on the perception that they are renewable and carbon-neutral; that is, they cause no net carbon emissions to the atmosphere. Previous studies of biofuels have supported the position that their use can reduce the net emissions of greenhouse gases, unlike conventional fossil fuels (Farrel et al., 2006, Wang et al., 2007). Recently, however, the debate has grown as to whether ethanol offers carbon emission reduction benefits. In particular, debate has focused on what are now called "indirect" land-use effects. The theory of "indirect" land-use effects suggests that a mandated expansion of ethanol production in the United States and European Union will raise corn and other agricultural prices (e.g., crops and livestock), and these higher prices will cause land conversion both within the United States and elsewhere in the world as producers seek to expand cultivated land.

So far, several estimates of these effects suggest that ethanol production is likely to lead to a net increase in greenhouse gas emissions. Fargione et al. (2008), for instance, analyzed the net emissions of carbon dioxide according to biofuel production and indicate the amount of additionally produced carbon dioxide varies depending on the method used to generate bioenergy.

They argue that when native habitats are converted into cropland to produce more bioenergy, the destruction of the native ecosystem can lead to additional carbon debt, which offsets the benefits of bioenergy. They state that biofuels need to be produced with little habitat destruction and wasted biomass. Further, they also suggest that degraded (or abandoned) agricultural lands could be used for biofuel production as an alternative way to reduce the amount of greenhouse gases. Searchinger et al. (2008) point out that the production of corn-based ethanol should lead to an increase in corn prices, which would result in forests and grazing land being converted into agricultural land. They find that the corn-based ethanol policy of the U.S. government would double the emissions of greenhouse gases, considering the indirect effects of such land-use changes. After assessing the U.S. corn-based ethanol policy and considering the market-mediated changes, Hertel et al. (2010a) found emissions of greenhouse gases would decrease to one-quarter of what Searchinger et al. (2008) found, but the level would still offset biofuel's benefits in reducing greenhouse gases. These studies raise doubts about the effectiveness of bioenergy as a mechanism to reduce greenhouse gas emissions, pointing out that most prior studies did not take into account land-use changes associated with biofuel production. They argue that biofuel production would result in forests and grazing lands being converted into agricultural land and assert that the accurate measurement of the effects of indirect land use is a critical issue for determining the effectiveness of biofuel policy.

When measuring the carbon implications of biofuel policies in forests, it is important to account for adjustments in the timber stock. Existing studies, however, treat forests stock as if they are in a steady-state condition. But because forests are biological stocks that can be managed and adjusted in response to exogenous perturbations, it is important to account for the dynamic adjustment of forests when calculating the implications of biofuel policies on forest carbon. For example, landowners will respond to shifts in incentives by changing their management practices, such as by altering rotation ages, adjusting planting densities, and making other shifts in management. All of these factors will influence the ultimate impact of the policy on the net carbon balance.

Another concern with many of the existing studies is that they have ignored important differences in ecology around the world. Forested ecosystems in different regions of the world have different carbon intensities and sequestration rates, so that land conversions will have different carbon implications depending on the carbon intensity. Further, biofuel policies will shift land rental prices differently in different regions depending the underlying productivity of the land in both agriculture and forestry. It is important to account for these regionally specific factors when measuring the impacts.

This paper addresses these two important issues. First, the study builds a dynamic optimization model to account for the response of forest stocks and forest management to the biofuel policies. Second, the study links accounts for the geographical extent of the impacts of the biofuel policies by using a general equilibrium model that measures the intensity of shifts in land rental functions for different agro-ecosystem zones around the world. The general equilibrium model and the dynamic optimization model use the same set of agro-ecosystem zones and regions around the world so that the economic shifts in land rents impact the correct ecosystems. The integration of the two models has important advantages that will allow for a more accurate evaluation of the indirect effects of biofuel policies on global forest stocks.

MODEL

By shifting outputs from traditional food markets to energy markets, biofuel policies in the US and Europe increase prices for agricultural products, and thus the opportunity cost of land. From the perspective of the forestry market, the impacts of a biofuel policy can be modeled as a shift in land rents (figure 1). The specific change in the land rents in each region of course will depend on numerous factors, including the effect of the policy on output, prices, and trade, the productivity of the crop and livestock sectors in each region, and the productivity of forestry in each region. For the purposes of this study, we assume that the shift in land rents is determined exogenously, and we then measure the implications of this shift in the forestry sector, using a dynamic optimization model.

We begin with the model of forests and land use. The model maximizes the present value of welfare in timber markets. Welfare is consumer's surplus minus the costs of managing and maintaining timberlands globally. Annual welfare is given as

(1)

$$W_t = \left\{ \int_0^{Q^*(t)} \left\{ D(Q_t, Z_t) - \sum_{i,j,k} CH^{i,j,k}(\cdot) \right\} dQ(t) - \sum_{i,j,k} p_m^{i,j,k} m_t^{i,j,k} G_t^{i,j,k} - \sum_{i,j,k} \left(R_t^{i,j,k} \right) \left(QFCET_t^{i,j,k} \right) \right\}$$

where $$Q_t = \sum_{i,j,k} \left(\sum_a H_{a,t}^{i,j,k} V_{a,t}^{i,j,k}(m_{t0}) \right)$$

In equation (1), the subscript a is for the age class, and t is for the time period. Superscript i

accounts for the specific country, *j* accounts for the agro-ecological zone within each country, and *k* accounts for the timber type within each agro-ecological zone. Agro-ecological zones capture differences in productivity across the landscape. $D(Q_t,Z_t)$ is a global demand function for industrial wood products given the quantity of wood, Q_t, and income, Z_t. The quantity of wood harvested depends upon $H_{a,t}^{i,j,k}$, the area of land harvested (in million hectares) in each age class, and $V_{a,t}^{i,j,k}(m_{t0})$, the yield per hectare of timber in each age class. The yield per hectare depends upon the species and the management intensity at the time of planting (m_{t0}). CH(▪) is a cost function for harvesting and transporting logs to mills from each of the timber types. Marginal harvest costs for temperate and plantation forest types are constant, while marginal harvest costs for inaccessible forests rise as additional land is accessed.

The costs of planting forests are given as $\sum_{i,j,k} p_m^{i,j,k} m_t^{i,j,k} G_t^{i,j,k}$, where $G_t^{i,j,k}$ is the area of land planted, $m_t^{i,j,k}$ is the management intensity of planting, and $p_m^{i,j,k}$ is the per unit cost of of a unit of management intensity. Units of management intensity enhance yield when the timber is harvested. The yield function has properties typical of ecological species, e.g., $V_a>0$ and $V_{aa}<0$. In addition, the following two conditions hold for trees planted at time t_0 and harvested "a" years later $(a+t_0) = t_{ai}$:

(2) $$\frac{dV^{i,j,k}(t_{a_i}-t_0)}{dm^{i,j,k}(t_0)} \geq 0 \text{ and } \left(\frac{d}{dm}\right)\frac{dV^{i,j,k}(t_{a_i}-t_0)}{dm^{i,j,k}(t_0)} \leq 0$$

The area of land in each forest type in each age class is given as $QFCET_t^{i,j,k}$ and $R_t^{i,j,k}$ is the annual rental value of the land. The area of land is the sum of the area of land in each age class. Competition for land in our model occurs within each agro-ecological zone (although aggregate price changes can also induce land use changes within agro-ecological zones). Within each zone, land rents and forest land areas are related via a constant elasticity of transformation function (CET), such that the total area of land in forest type *i,j,k* is given as:

(3) $$QFCET_t^{i,j,k} = \frac{XE^{i.j.k}\left(\left(\alpha_F^{i,j,k} \middle/ R_t^{i,j,k}\right)\right)^{\tau}}{\left[\alpha_{Cr}^{\tau} R_{Cr}^{1-\tau} + \alpha_{Lv}^{\tau} R_{Lv}^{1-\tau} + \sum_{k=1}^{6}\left(\alpha_F^{i,j,k}\right)^{\tau}\left(R_t^{i,j,k}\right)^{1-\tau}\right]^{\left(\frac{\tau}{\tau-1}\right)}}$$

where α_{Cr}, α_{Lv}, and α_F are CET share parameters for crop, livestock, and forestry land in each agro-ecological zone. These share parameters are calibrated based on initial land areas and rents for the various land uses in each agro-ecological zone in the initial period, as discussed below. The parameters R_{cr} and R_{Lv} are land rents for crop and livestock within each agro-ecological zone. For the purposes of this analysis, we assume that they are given. The parameter τ is the elasticity of land supply.

One concern with using a CET approach to model land supply is that land areas calculated by the CET function are given in value added terms rather than physical terms. Our forestry model, however, manages physical units of land and physical timber products. We need a means to convert between physical and valued added, CET, hectares. Specifically, we would like to have productivity adjustment variable to convert CET forest area to physical hectares.

For example, suppose this parameter is given as A, then the physical area of forests could be given as

(4) $$QFP_t^{i,j,k} = \sum_a X_{a,t}^{i,j,k} = \left(A_t^{i,j}\right) * QFCET_t^{i,j,k}$$

Where $X_{a,t}^{i,j,k}$ are the area of physical hectares in each age class in each timber type, agro-ecological zone, and region, and $QFP_t^{i,j,k}$ is the sum of physical hectares. The parameter $A_t^{i,j}$ is a productivity adjustment that converts CET area, $QFCET_t^{i,j,k}$, to physical hectares $QFP_t^{i,j,k}$. In order to calculate $A_t^{i,j}$ we first assume that the average productivity of crop land decreases if forest and livestock lands are converted to crop land. The productivity parameter is set to 0.66 in this paper, which implies that approximately 1.5 (=1/0.66) new physical hectares are required to produce the same amount of product that one hectare of current crop land produces. Under this assumption, the change in physical crop hectares from one period to the next equals 1.5 times the change in CET calculated crop hectares. Second, we note that the total area of physical land in each agro-ecological zone remains constant. Thus, $A_t^{i,j}$ can be solved by including an additional equation in the model:

(5) $$TL = QCP_t^{i,j} + QLP_t^{i,j} + \sum_k QFP_t^{i,j,k} = QCCET_1^{i,j} + QLCET_1^{i,j} + \sum_k QFCET_1^{i,j,k}$$

$$= QCP_1^{i,j} + \left(\frac{1}{0.66}\right) \times \left(QCCET_t^{i,j} - QCCET_1^{i,j}\right) + A_t^{i,j}\left(QFCET_t^{i,j,k} + QLCET_t^{i,j}\right)$$

In equation (5), TL is the total area of land in each agro-ecological zone, which is fixed, and equivalent to the sum of the physical and CET units of land in the initial period (period 1). $QCP_t^{i,j}$ is the physical cropland area and $QLP_t^{i,j}$ is the physical livestock area in each country and agro-ecological zone. $QCCET_t^{i,j}$, $QLCET_t^{i,j}$, and $QFCET_t^{i,j,k}$ are the CET units of land. Given the relationship in equation (5), and our assumption about changes in the average productivity of land converted from forests or livestock to crops , $A_t^{i,j}$ can be found as follows:

(6) $$A_t^{i,j} = \frac{TL - \left\{QCP_1^{i,j} + (\frac{1}{0.66}) \times \left(QCCET_t^{i,j} - QCCET_1^{i,j}\right)\right\}}{\left(QFCET_t^{i,j,k} + QLCET_t^{i,j}\right)}$$

Over time, physical hectares within our forestry model adjust according to

(7) $$X_{a,t}^{i,j,k} = X_{a-1,t-1}^{i,j,k} - H_{a-1,t-1}^{i,j,k} + G_{a=0,t-1}^{ij,k} \qquad \text{for all } i, j, k$$

This equation shifts forests in from one age class to the next over time, accounting for losses due to harvesting and gains due to regenerating. Note that regeneration can occur naturally if the area replanted is greater than 0, but the intensity of management is 0, e.g., $m_t^{i,j,k}$ =0. The model chooses $H_{a,t}^{i,j,k}$, $G_t^{i,j,k}$, and $m_t^{i,j,k}$ to maximize the present value of welfare,

(8) $$\text{Max} \sum_{t=1}^{T} \rho^t \left(W_t\right)$$

Equation (3), (4), and (7) are constraints for the system. Additional non-negativity constraints are imposed for each of the variables.

To account for the effects of biofuel policies, we assume that the policies will alter the land rental functions shown in equation (3). Specifically, biofuel policies should increase the rents for cropland, and thus shift the rental functions for our forestry model inward. Obviously, the key issue from the perspective of the forestry model is the size of the shift. To calculate the shift, we utilize a computable general equilibrium analysis conducted by Hertel et al. (2010b). In recent years, the general equilibrium community has made great strides including land in the modeling framework (see Hertel et al., 2009), and we build upon this work. Specifically, we use the Global Trade Analysis Project (GTAP) framework so that we can make the land base in the general equilibrium model match the land base in the forestry model. Thus, both models use consistent country and agro-ecological zone definitions for this analysis (see Table 1 for regions). The forestry model does incorporate additional detail on the forestry types within each agro-ecological zone in order to account for potentially important differences in forest productivity and carbon storage across forest types within agro-ecological zones.

To calculate the shift in land rents due to the biofuel policies, we use the following process: First, both models are calibrated with the same initial rental and land area data, as well as CET elasticity parameter (although we do vary this parameter in sensitivity analysis). Second, the GTAP model is perturbed with a biofuel policy scenario (specific details of this scenario are discussed below). As a result of the biofuel policy scenario, additional land will be used for crops and rents will rise. These land use and rental value shifts are used to calculate a shift in the CET forest supply function following Horridge and Zhai (2006). The shift parameter, $B^{i,j,k}$, alters equation (3) to

(3') $$QFCET_t^{i,j,k} = \frac{XE^{i.j.k}\left(B^{i,j,k}\right)\left(\alpha_F^{i,j,k} \big/ R_t^{i,j,k}\right)^{\tau}}{\left[\alpha_{Cr}^{\tau} R_{Cr}^{1-\tau} + \alpha_{Lv}^{\tau} R_{Lv}^{1-\tau} + \sum_{k=1}^{6}\left(\alpha_F^{i,j,k}\right)^{\tau}\left(R_t^{i,j,k}\right)^{1-\tau}\right]^{\left(\frac{\tau}{\tau-1}\right)}}$$

The parameter $B^{i,j,k}$ shifts forest land supply function inward and makes the opportunity costs of holding forest land increase and, therefore, results in additional land conversion from forest land (Figure 1). One limitation of this approach, of course, is that the GTAP model is a static equilibrium approach whereas the forestry model is dynamic. It would be preferable to incorporate dynamic adjustments in the parameter $B^{i,j,k}$, but for this analysis we assume that the parameter is fixed, over time.

DATA AND ANALYSIS

Several types of data are needed for this analysis. First, we must obtain data on forest inventories, including information on the age class distribution of forests. Second, we must obtain information on the productivity of forests (e.g., yield functions) and the costs of extracting and transporting timber to mills. The data are obtained from Sohngen et al. (2009). In the dataset described in Sohngen et al. (2009), forest inventories from various sources are allocated to agro-ecological zones. For this analysis, we rely on a definition for agro-ecological zones that builds on the work of the FAO and IIASA (2000), and is described in Monfreda et al. (2009) and Lee et al. (2009). In each region of our model, there are up to 18 agro-ecological zones. The agro-ecological zones differ by growing period (6 categories of 60 day growing period intervals), and climatic zone (tropical, temperate and boreal). The length of growing period depends on temperature, precipitation, soil characteristics and topography, and the suitability of each agro-ecological zone for production of alternative crops and livestock is based on currently observed practices.

In addition to information on forest stocks, we also need information on crop and livestock stocks and rental values to calibrate the value shares in the forest-land supply functions described above in equation (3). The land rental function we use is formally derived from the following CET relationship:

$$(10) \qquad XE = \left(\alpha_{Cr} X_{Cr}^{\frac{(\tau-1)}{\tau}} + \alpha_{Lv} X_{Lv}^{\frac{(\tau-1)}{\tau}} + \sum_{k=1}^{6} \alpha_{F,k} X_{F,k}^{\frac{(\tau-1)}{\tau}}\right)^{\frac{\tau}{(\tau-1)}}.$$

Given a value for τ, and data on land and rents in each agro-ecological zone, the value shares, α's, can be calibrated. Data on land rents and land areas in each agro-ecological zone are obtained from Lee et al. (2009).

To make projections in the forestry sector, we utilize a demand function of the form

$$(11) \qquad Q_t = A_t \left[\left(Y_t / N_t\right)\right]^h \left(P_t\right)^e$$

Where Y_t is global gross domestic product, N_t is global population, P_t is the global price of timber, *h* is income elasticity, and *e* is the price elasticity. Gross domestic product per capita (Y_t/N_t) is assumed to grow at 2.3% per year. Income elasticity for forestry products is calculated from the AIDADS (An Implicitly Additive Demand System) modeling system described in Rimmer and Powell (1996). Initially, it is 0.87, and it rises to 0.93 over the century. Price elasticity for forestry products is assumed to be -1.0 based on work by Simangunsong and Buongiorno (2001), who report elasticity estimates from for end products ranging from -0.62 for sawnwood to -1.33 for plywood, and Uusivuori and Kuuluvainen (2001), who suggest that the demand for industrial roundwood imports ranges from -0.69 for non-tropical hardwoods to -0.92 for softwood, and -0.95 for wood chips.

For policy analysis, we compare a baseline case, in which $B^{i,j,k}$ in equation (3') is set to 0, to a policy scenario, where $B^{i,j,k} > 0$. The parameter $B^{i,j,k}$ is obtained from analysis of biofuel policy using the GTAP model simulation from 2006 to 2015. Specifically, we examine both U.S. and European biofuel mandates, and analyze those within the GTAP model to calculate the change in land used for crops in all agro-ecological zones, following the approach of Hertel et al. (2010b). The biofuel mandates targets for 2015 are from Taheripour et al. (2010). In the United States, the Energy Independence and Security Act of 2007 revised the original Renewable Fuel Standard program (RFS1) and increased the required volume of renewable fuel up to 36 billion gallons by 2022. Under the revised Renewable Fuel Standard program (RFS2), corn ethanol share is capped at 15 billion gallons by 2015. The European Union set target of a ten percentage of biofuel in transport fuel by 2020 (European Commission, 2007). However, we set target of 6.25% by 2015 following Taheripour et al. (2010). The forestry model and the GTAP model in this case have been harmonized so that the regions, agro-ecological zones, current land allocation, and initial rents are the same.

One of the key parameters likely to affect the carbon implications of the biofuel policy is the elasticity parameter on the CET function. Recent estimates by Lubowski et al. (2006) indicate a price (rent) elasticity of land supply of approximately 0.3. Sohngen and Brown (2006), however, find higher land supply elasticity estimates for the US South. Ahmed et al. (2009) suggest that in the short run, land supply elasticity may be less than 0.2, rising to higher levels only over time. For this analysis, we consider several estimates for τ, ranging from 0.2 to 0.5.

When measuring the impacts of biofuels, the standard approach in the literature is to calculate the total hectares of forestland lost and multiply that by assumed average tons of carbon per hectare (e.g., Searchinger et al. 2008). This approach assumes that all hectares are the same and that they are all converted to agriculture at the same time. Our approach assumes that changes happen over time, and this creates some difficulties for creating a carbon measure. Stavins (1999) has argued that for benefit cost analysis, carbon flows should be discounted in order to calculate "present value" carbon associated with land use changes, and a number of studies have now adopted this. Discounting, of course, is appropriate when the marginal value of carbon is constant, but can be incorrect if carbon values are changing over time (see Richards and Stokes, 2004 and van Kooten and Sohngen, 2007). For the purposes of this study, we present both annual flows for carbon changes, and we present discounted present value and annual equivalent amounts for comparison.

RESULTS

While the analysis is conducted for 18 separate regions, we presents the results for six regions, plus a rest-of-world (ROW) region that contains the other 11 regions. Our estimates for $B^{i,j,k}$ by region range from around 4% in Southeast Asia to 44% in the European Union (Table 2). T he largest effects are in the United States and the European Union where the mandate policies we examine are centralized. South America, however, also has a relatively large impact from the biofuel policy, likely due to the close linkage in agricultural commodity markets between that market and the United States. On average, the GTAP model projects that the rental functions shift inwards by around 19%.

The choice of elasticity parameter does affect the estimate of $B^{i,j,k}$, however, the direction of the change is not the same for all regions. For the United States, the European Union, and ROW, a higher elasticity parameter increases $B^{i,j,k}$, whereas for other regions $B^{i,j,k}$ declines with a higher elasticity. The results for the United States and the European Union make sense because more land will convert to agricultural for any given change in land rents, thus, the biofuel policies are likely to have larger implications for the area of land used for crops in those regions under the larger elasticity. This in turn implies a larger shift in the rental function for higher elasticities from the perspective of the forestry sector modeled here. Alternatively for other regions, the larger shift in land use changes in the United States and the European Union causes a larger increase in biofuel output, and thus a smaller change in global commodity prices. In turn, other regions experience smaller changes in their land rents under the higher elasticity estimate. Thus, higher elasticity estimates tend to increase the impacts in the countries where policies are implemented and reduce the impacts elsewhere.

The land-use changes induced by the biofuel policies lead to carbon emissions over the next 30 years on the order of 50-200 t CO2 per year, depending on the elasticity parameter in the CET function (Figures 2a and 2b). The emission profile suggests that emissions don't occur instantly when the policy is introduced. In fact, our results indicate that the effects on carbon start relatively small, but continue to play out well into the future. From the perspective of calculating the carbon impacts, this dynamic path can have important implications. For example, if one is using life cycle analysis (LCA), the typical assumption is to assume that all of the land use changes and carbon changes occur initially, and last for a given time period. This may dramatically overstate impacts.

In total, we find that over the next 30 years, the biofuel policy could lead to a 0.8-0.9% reduction in forestland, or 24-26 million hectares. The largest proportional changes in land use occur in the United States and European Union (Table 3), which is not surprising given that the policies are centralized in these regions and the shifts in the rental functions are greatest there. The area changes tend to get larger moving from a lower to a higher τ parameter. This makes sense as a more elastic land supply function implies larger shifts in land area for any particular price change.

The proportional changes in land use projected by the forestry model for the bioenergy policies are smaller than the shifts in the rental functions discussed above. There are two important reasons for this. First, the forestry model includes both accessible and inaccessible lands, and for the purposes of this analysis, we have focused the shifts in the rental functions in the accessible forest areas. Since a large portion (2/3rds) of the world's forests in our forestry model is "economically" inaccessible[1], the proportional impacts are relatively small. Second, the forestry model allows for rents to shift, and in this case rents for forestland tend to increase. Rising forest rents will reduce the impact of the inward and upward shift in the rental function (figure 3).

What is surprising about the area changes in Table (3) is that forest area in Southeast Asia increases under the higher two elasticity parameter estimates. The key drivers for this surprising result are the rental function shift (B), and the change in timber price. For Southeast Asia, the shift in the rental function is relatively small (4-5%). Timber prices rise with the overall reduction in forestland, and they increase more under the larger land use changes associated with the higher elasticity parameters. For example, timber prices rise from 0.5% on average in the first 30 years when τ =-0.2 to 1.2% when τ =-0.5. In other regions, higher timber prices and

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[1] Note that our inaccessible forests are those forestlands where the marginal benefits of accessing the land (timber and potential agricultural value) are less than the marginal access costs.

timber returns do mitigate the loss of forestland caused by the shifting rental functions, but the rental function shifts are fairly large in most regions. In Southeast Asia, however, the timber price and revenue increases are large enough to overcome the rental function shift.

To calculate the carbon implications of these changes, we discount the present value effects of the biofuel policies on carbon and the resulting annual equivalent amounts. Present value calculations are done for 30 years since the largest effects are likely to occur in that time period, and furthermore, new policies will probably be implemented as the current policies expire. The present value loss of carbon is found to be 1233-1606 Million t CO_2, with annual equivalent amounts ranging from 80.4 to 104.5 Million t CO_2/yr induced carbon loss. The annual equivalent amounts, of course, are much less than the maximum annual losses shown in figure 2 because the initial losses and the longer-term losses both are smaller than the maximum loss observed.

Aside from the fairly large "ROW" category, the biggest share of carbon losses occur in Central and South America. These regions are heavily forested, and have close trade linkages to North America and Europe, so it is perhaps not surprising that the effects in Central and South America are so large. Interestingly, the carbon losses are much larger on a per hectare basis in Central and South America than the United States or European Union because the lands that are affected have bigger carbon stocks there. Alternatively, Southeast Asia shows a gain in carbon stocks. As noted above, this results from the increase in land area devoted to forests, and suggests the importance of modeling price responses across regions.

To get a sense for the benefits of conducting dynamic analysis versus static analysis, we develop a static estimate for comparison purposes. For the static analysis, we simply take the land area changes projected in Table 3 and multiply them by the average tons of CO_2 per hectare for each of the regions. This is consistent with most of the analyses conducted so far. Not surprisingly, the net carbon effects of the policy when calculated this way are up to three times larger than we estimate with our approach (Table 4). Thus, even though current studies may approximate the land use changes reasonably well, without conducting detailed dynamic analysis on the forestry stocks, their results may substantially over-state the true impacts. A final alternative way to conduct the analysis would be to multiply our land area change by the global average carbon stock estimates used by Searchinger et al. (2008). They use an estimate of 355 t CO_2 per hectare, implying induced emissions of 8.2 – 9.3 billion t CO_2 from the biofuel policies. Using a gross global average for carbon stocks per hectare dramatically overstates the implications of biofuel policies because it ignores any regional differences in important economic and ecological factors (the size of $B^{i,j,k}$, stock adjustments to perturbations, etc.)

CONCLUSION

This paper examines the implications of United States and European biofuel mandates on global land use change, forest stocks, and carbon emissions. We begin by developing a dynamic model of forests and land use that allows us to measure the implications of perturbations to the forest stock via land rental functions. We then use a detailed computable general equilibrium analysis of the biofuel mandates to determine the specific scale of the perturbations to the land rental functions. This computable general equilibrium analysis models land in the same regions and agro-ecological zones as used by the dynamic forestry model. By building the two models off the same land database, by using a consistent CET structure to model competition across land uses within agro-ecological zones, and by harmonizing the parameter choices for the CET function, we are able to closely integrate the two models. The computable general equilibrium model is thus used to provide shifts in the rental functions for land used by the forestry model. These shifts in the rental functions are then used in the forestry model to calculate new scenarios of future forestland management and use. The scenarios are compared to the baseline to determine induced forest area changes and carbon adjustments.

The results indicate that 23-26 million hectares of forestland would be lost globally in the nexst 30 years as a result of the US and European biofuel mandates. These forestland losses result in 1.2 to 1.6 billion additional tons of present value CO2 emissions. The largest proportional losses in forestland occur in the United States and European Union. The largest physical losses of land occur in Central and South America. Central and South America also have the largest carbon losses associated with the land use changes. Surprisingly, some regions, like Southeast Asia, actually experience gains in forestland. The reason for this is that the land rental function perturbations projected by the computable general equilibrium model are small for Southeast Asia relative to other regions, and rising timber prices raise the returns to forestry. The increased returns to forestry are large enough to avoid the losses typically considered in the modeling of induced land use changes from biofuels. Studies that do not account for the full response of markets, including forestry markets, to policies will not account for these types of adjustments.

The results indicate the importance of dynamic modeling whenever adjustments to forest stocks are analyzed. Forestry is a dynamic stock that requires dynamic analysis. Static analysis may over-state the impacts – in our case by up to 3 times if region specific carbon factors are used and up to 6 times if gross global average carbon emissions are used. It is thus vitally important to utilize dynamic analysis to get the timing of the carbon changes correct, and to account for potential management adjustments, and it is important to conduct the analysis with region specific economic and ecological effects. Our results show that the $B^{i,j,k}$ parameter shfit depends on economic differences across regions (e.g., trade relationships, types of products produced and consumed), and the scale of the change in that parameter ultimately has a large impact on the size of the effect.

While this analysis improves on several earlier analyses, there are still some shortcomings. First, we use dynamic analysis only in the forestry model. We do not conduct dynamic analysis with the computable general equilibrium model. We thus ignore future potential adjustments in the biofuel policies or other policies that could have important implications for near term carbon management. Second, we have only examined a traditional biofuel policy based on starch ethanol. Cellulosic ethanol could be important in the future, depending on technology development, and almost certainly would have important implications for land use. Third, we have ignored other types of policies that may affect land use in the future, such as incentives for carbon sequestration. Such incentives would raise the value of land in forests and future land uses, and could have important implications for the scale of the results. Future analysis should address these and other shortcomings.

REFERENCES

Ahmed, A.S., Hertel, T.W., & Lubowski, R. (2009). *Calibration of a land cover supply function using transition probabilities*. GTAP Research Memorandum, No 14. Center for Global Trade Analysis, Purdue University, West Lafayette, IN, USA.

Congress U.S. (2007). *Energy Independence and Security Act*. Washington, DC.

European Commission (2007). *Impact assessment of the renewable energy roadmap – March 2007:* Directorate-General for Agriculture and Rural Development, European Commission, AGRI G-2/WMD.

FAO and IIASA. (2000). *Global agro-ecological zones - 2000*. Rome, Italy & Laxenburg, Austria: Food and Agriculture Organization (FAO) of the United Nations, and International Institute forApplied Systems Analysis (IIASA).

Fargione J. et al. (2008). Land clearing and the biofuel carbon debt. *Science, 319*: 1235-1238.

Farrell, A. E., Plevin, R.J., Turner, B.T., Jones, A. D., O'Hare, M., & Kammen, D.M. (2006). Ethanol can contibute to energy and environmental goals. *Science 311*: 506-508.

Hertel, T.W., Lee, H., Rose, S., & Sohngen, B. (2009). Modeling land-use related greenhouse gas sources and sinks and their mitigation potential. In T. Hertel, S. Rose and R. Tol (Eds.), *Economic analysis of land use in global climate change policy* (chapter 6). New York: Routledge Press.

Hertel T. W., Golub A., Jones A. D., O'Hare M., Plevin, R.J. & Kammen, D.M. (2010a). Effects of US maize ethanol on global land use and greenhouse gas emissions: Estimating market-mediated responses. *BioScience, 60* (3).

Hertel, T.W., Tyner, W.E., & Birir, D.K. (2010b). The global impacts of biofuels mandates. *The Energy Journal, 31*(1):75-100.

Horridge, M. and Zhai, F. (2006). Shocking a single-country CGE model with export prices and quantities from a global model. In T. Hertel and L. Winters (Eds.), *Poverty and the WTO: Impacts of the Doha Development Agenda* (Appendix to Chapter 3). New York: Palgrave-MacMillan.

Lee, H.-L., T.W. Hertel, S. Rose and M. Avetisyan (2009). An integrated global land use data base for CGE analysis of climate change policy options. In T. Hertel, S. Rose & R. Tol (Eds.), *Economic Analysis of Land Use in Global Climate Change Policy* (Chapter 4). New York: Routledge Press.

Lubowski, R.N., Plantinga, A.J., and Stavins, R.N.(2006). Land-use change and carbon sinks: econometric estimation of the carbon sequestration supply function. *J Environ Econ Manag, 51*:135–52.

Monfreda, C., N., Ramankutty, & Hertel, T.W. (2009). Global agricultural land use data for climate change analysis. In T. Hertel, S. Rose, & R. Tol (Eds.), *Economic analysis of land use in global climate change policy* (Chapter 2). New York: Routledge.

Richards, K., & Stokes, C. (2004). A review of forest carbon sequestration cost studies: A dozen years of research. *Climatic Change*, *63*(1-2): 1-48.

Rimmer, M.T. & Powell, A.A. (1994) Engel flexibility in household budget studies: Nonparametric evidence vs. standard functional form. *COPS/IMPACT Working Paper No. OP-79*. Clayton, Vic.: Monash University.

Searchinger, T.D. et al. (2008). Use of U.S. croplands for biofuels increases greenhouse gases through emissions from land-use change. *Science, 319*: 1238-1240.

Simangunsong, B.C., & Buongiorno, J. (2001). International demand equations for forest products: A comparison of methods. *Scandinavian Journal of Forest Research*, *16*: 155–172.

Sohngen, B., Tennity, C., Hnytka, M., & Meeusen, K. (2009). Global forestry data for the economic modeling of land use. In T. Hertel, S. Rose, & R. Tol (Eds.), *Economic Analysis of Land Use in Global Climate Policy* (Chapter 3). New York: Routledge Press.

Sohngen, B. & Brown, S. (2006). The influence of conversion of forest types on carbon sequestration and other ecosystem services in the south central United States. *Ecological Economics, 57*: 698-708.

Stavins, R.N. (1999). The costs of carbon sequestration: A revealed-preference approach. *American Economic Review, 89*(4): 994-1009.

Taheripour, F., Hertel, T.W., Tyner, W.E. (2010). Biofuels and their By-Products: Global Economic and Environmental Implications. *Biomass and Bioenergy,* 34:278-289.

Uusivuori, J., & Kuuluvainen, J. (2001). Substitution in global wood imports in the 1990s. *Canadian Journal of Forest Research, 31*: 1148–1155.

van Kooten, G. C., & Sohngen, B. (2007). Economics of forest ecosystem carbon sinks: A review. *Int. Rev. Environ. Resour. Econ. 1*(3): 237-269.

Wang, M, Mu, M., & Huo, H. (2007). Life-cycle energy and greenhouse gas emission impacts of different corn ethanol plant types. *Environmental Research Letters:* 1–13.

Table 1.

Region	GTM	GTAP
1	US	USA
2	CHINA	CHIHKG
3	BRAZIL	BRAZIL
4	CANADA	CAN
5	RUSSIA	Russia
6	EU25	EU27
7	OthEurope	Oth_Europe
8	OthCEE_CIS	Oth_CEE_CIS
9	CENT AMERICA	C_C_Amer
10	REST SOUTH AM	S_o_Amer
11	SUB SAHARAN AF	S_S_AFR
12	SOUTHEAST ASIA	Mala_Indo
13	OCEANIA	Oceania
14	JAPAN	JAPAN
15	AF MIDDLE E	MEAS_NAfr
16	EAST ASIA	E_Asia
17	OthSouthAsia	R_S_Asia
18	India	INDIA

Table 2. Shift parameter calculated by the GTAP model, weighted over the land areas in each agro-ecological zone ($B^{i,j,k}$). The values represent a percentage shift in the rental function.

TAU	-0.2	-0.3
US	28.1%	28.4%
SOUTH AM	26.8%	26.5%
EU25	42.6%	44.3%
CENT AMERICA	11.9%	11.4%
SUB SAHARAN AF	14.9%	14.4%
SOUTHEAST ASIA	4.2%	3.9%
ROW	13.1%	13.2%
Total	19.2%	19.4%

Note: Shift changes area are weighted average over 30 years.

Table 3:

	United States	European Union	Central and South America	Sub-Saharan Africa	Southeast Asia	Rest of World	Total
Panel A: τ = -0.2							
Area Change (Million ha)	(3.1)	(4.7)	(3.9)	(2.6)	(0.8)	(8.5)	(23.6)
Area Change (Percent)	-1.5%	-3.3%	-0.5%	-1.7%	-0.5%	-0.5%	-0.8%
Carbon Change (AEA) Mill. tCO2/yr	(10.1)	(9.8)	(23.2)	(7.0)	3.0	(33.0)	(80.2)
Carbon Change (PV) Mill. tCO2	(155.3)	(150.7)	(357.1)	(108.0)	45.9	(507.7)	(1232.9)
Panel B: τ = -0.3							
Area Change (Million ha)	(5.0)	(5.3)	(6.3)	0.2	2.5	(12.3)	(26.2)
Area Change (Percent)	-2.4%	-3.9%	-0.9%	0.1%	1.7%	-0.7%	-0.9%
Carbon Change (AEA) Mill. tCO2/yr	(16.3)	(16.9)	(42.3)	3.2	6.8	(25.8)	(91.2)
Carbon Change (PV) Mill. tCO2	(250.4)	(259.4)	(650.1)	49.9	104.0	(396.0)	(1402.1)
Panel C: τ = -0.5							
Area Change (Million ha)	(8.6)	(6.1)	(4.2)	1.1	2.0	(10.5)	(26.3)
Area Change (Percent)	-4.0%	-4.6%	-0.6%	0.7%	1.4%	-0.6%	-0.9%
Carbon Change (AEA) Mill. tCO2/yr	(21.7)	(17.0)	(42.2)	(2.2)	39.4	(60.9)	(104.5)
Carbon Change (PV) Mill. t CO2	(334.1)	(260.6)	(648.2)	(33.3)	605.9	(936.1)	(1606.3)

Table 4:

	United States	European Union	Central and South America	Sub-Saharan Africa	Southeast Asia	Rest of World	Total
				Million t CO2			
τ = -0.2	(491.5)	(633.4)	(756.2)	(250.2)	(65.2)	(965.1)	(3,161.6)
τ = -0.3	(781.1)	(716.9)	(1,202.8)	15.1	206.0	(1,447.2)	(3,927.0)
τ = -0.5	(1,346.1)	(802.9)	(914.3)	99.0	156.8	(922.8)	(3,730.4)

Figure 1: Effect of biofuel policy on forestry land rental function.

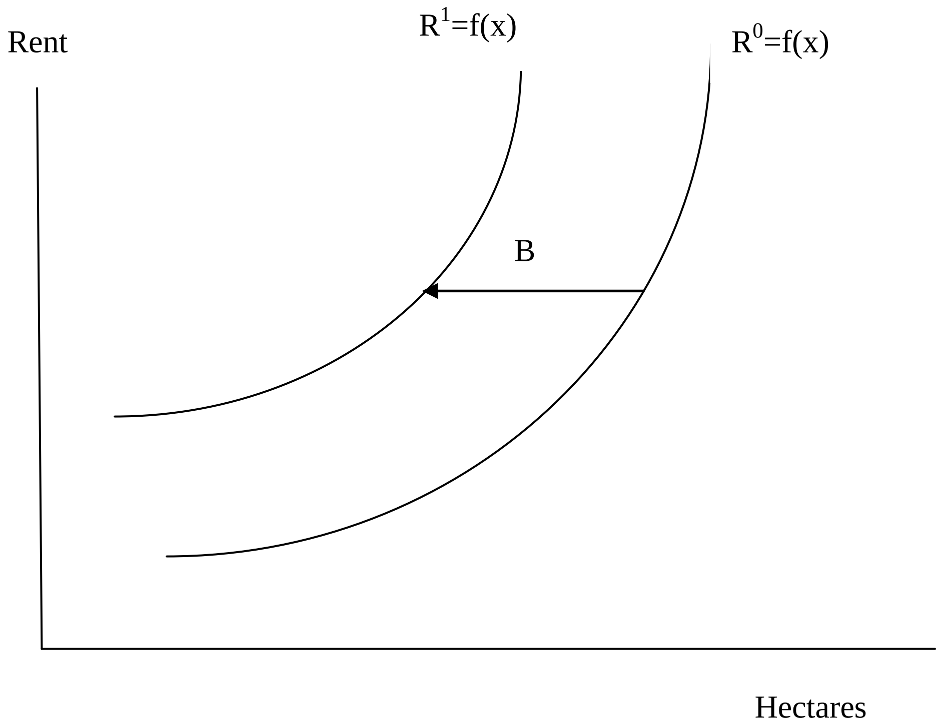
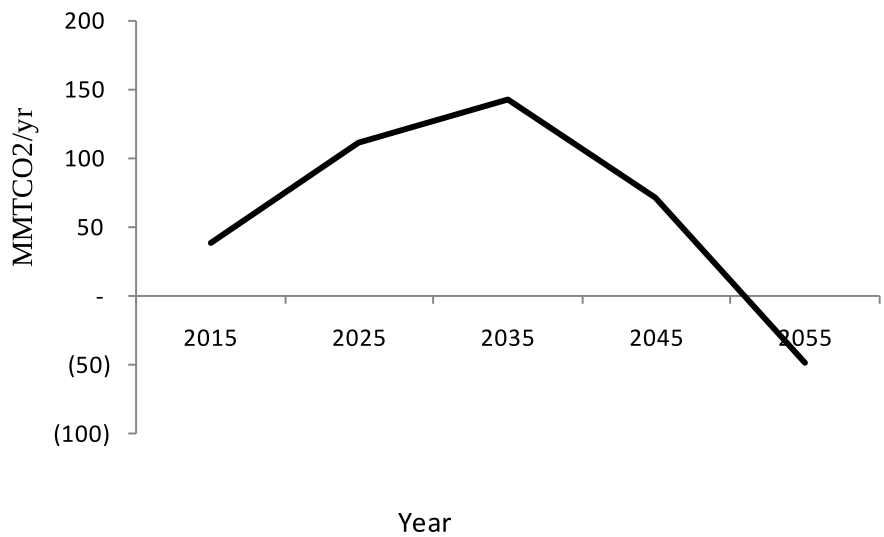


Figure 2: Annual global carbon emission due to induced land-use changes from the US and EU biofuel mandates (million metric tons CO2 per year).

Panel a: Elasticity = -0.2



Panel b: Elasticity = -0.3

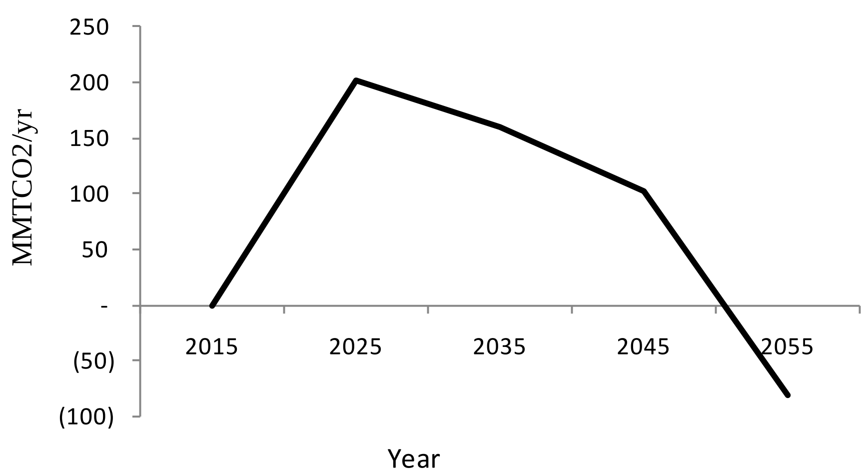


Figure 3. Rental function