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# The opportunity cost of land use and the global potential for greenhouse gas mitigation in agriculture and forestry

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ABSTRACT

This paper analyses the role of global land management alternatives in determining potential greenhouse gas mitigation by land-based activities in agriculture and forestry. Land-based activities are responsible for over a third of global greenhouse gas emissions, yet the economics of land-use decisions have not been explicitly modeled in global mitigation studies. In this paper, we develop a new, general equilibrium framework which effectively captures the opportunity costs of land-use decisions in agriculture and forestry, thereby allowing us to analyse competition for heterogeneous land types across and within sectors, as well as input substitution between land and other factors of production. When land-using sectors are confronted with a tax on greenhouse gas emissions, we find significant changes in the global pattern of comparative advantage across sectors, regions, and land types. Globally, we find that forest carbon sequestration is the dominant strategy for GHG emissions mitigation, while agricultural-related mitigation comes predominantly from reduced methane emissions in the ruminant livestock sector, followed by fertilizer and methane emissions from paddy rice. Regionally, agricultural mitigation is a larger share of total land-use emissions abatement in the USA and China, compared to the rest of the world, and, within agriculture, disproportionately from reductions in fertilizer-related emissions. The results also show how analyses that only consider regional mitigation, may bias mitigation potential by ignoring global market interactions. For example, USA-specific analyses likely over-

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estimate the potential for abatement in agriculture. Finally, we note that this general equilibrium framework provides the research community with a practical methodology for explicit modeling of global land competition and land-based mitigation in comprehensive assessments of greenhouse gas mitigation options.

## 1. Introduction

Changes in land use and land cover represent an important driver of net greenhouse gas (GHG) emissions. It has been estimated that roughly a third of the total emissions of carbon into the atmosphere since 1850 has resulted from land use change (and the remainder from fossil-fuel emissions) (Houghton, 2003). For example, in the 1990s, 6.4 billion tonnes of carbon alone per year was emitted to the atmosphere from industrial activities and approximately 2.2 billion per year was emitted from tropical deforestation.[1] In addition, agricultural land related activities are estimated to be responsible for approximately 50% of global methane emissions ($CH_4$) and 75% of global nitrous oxide ($N_2O$) emissions, for a net contribution from non-carbon dioxide (non-$CO_2$) GHGs of approximately 14% of all anthropogenic greenhouse gas emissions (USEPA, 2006a). Because of this potentially important role in the climate change debate, the policymaking community is assessing how agriculture and forestry may enter into future climate policies. These come into play, either as domestic actions (e.g., emission offsets within a cap-and-trade system), or via mechanisms for achieving international commitments for emissions reductions (e.g., the Clean Development Mechanism). In this paper, we develop a new framework for assessing the mitigation potential of land-based emissions that explicitly models the economics of land-use change and management decisions. This framework could be readily combined with existing GHG mitigation studies of industrial and fossil fuel-based GHG emissions.

A number of estimates have been made of the cost of abating greenhouse gas emissions for specific land use change and land management technologies (Richards and Stokes, 2004; USEPA, 2006b, Chapter 5). Recent studies also suggest that land-based mitigation could be cost-effective and assume a sizable share of overall mitigation responsibility in optimal abatement (Sohngen and Mendelsohn, 2003) and stabilization policies (Rose et al., 2008a). However, to date, global economic modeling of land has not been able to fully account for the opportunity costs of land-use and land-based mitigation strategies, nor the heterogeneous and dynamic environmental and economic conditions of land, which have been shown to be important (e.g., Li et al., 2006; Sohngen et al., 2009).

Despite the clear links between the agricultural and forestry sectors through land and other factor markets and international trade, existing studies do not explicitly model the reallocation of inputs (both primary factors and intermediates) within and across these and other sectors and regions in response to climate policies. National and international agricultural and forest climate policies have the potential to redefine the opportunity costs of international land-use in ways that either complement or counteract the attainment of climate change mitigation goals. This paper develops an analytical framework to capture and evaluate these potentially important relationships.

Global economic modeling of land-use is not new. There are global agricultural models (e.g., Darwin et al., 1995, 1996; Ianchovichina et al., 2001; Rosegrant et al., 2001), and global forestry models (e.g., Sohngen and Sedjo, 2006), and some that endogenously model land competition between

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[1] This number was recently revised down to 1.6 BTCE/yr for gross emissions from 1990s tropical deforestation and other land-use change activities, with a range of 0.5 to 2.7 BTCE/yr due to downward revision of estimated tropical deforestation (IPCC, 2007; Houghton, 2008). Throughout this paper, greenhouse gas emissions are measured in metric tons of carbon equivalent, where 1 metric tone = 1000 kg, and one metric tone of carbon (C) equals approximately 3.67 metric tons of carbon dioxide ($CO_2$). Non-carbon dioxide greenhouse gases have been converted to carbon equivalent units using the 100-year global warming potentials currently used for emissions inventories under the United Nations Framework Convention on Climate Change (i.e., the IPCC's Second Assessment Report, IPCC, 1996). Metric tons of carbon equivalent are written as "TCE", million metric tons are written as "MMTCE", and billion metric tons are written as "BTCE."

the two sectors (Sands and Leimbach, 2003). There are other models that employ priority rules that allocate land to food crops first before serving other commodity markets such as timber demand (e.g., Riahi et al., 2007; van Vuuren et al., 2006).

Economic studies of the global supply of forest carbon sequestration have evolved and become relatively mature, with globally consistent dynamic frameworks modeling endogenous movement of land into and out of forestry, explicitly considering multiple forest management alternatives and forest types, and modeling of international trade effects (e.g., Sohngen and Mendelsohn, 2007; Sohngen and Sedjo, 2006; Sathaye et al., 2006; Rokityanskiy et al., 2007). However, the same cannot be said for global studies of agricultural greenhouse gas mitigation potential. A single consistent economic framework does not exist to evaluate global mitigation potential (Chapter 8, IPCC, 2007). Instead, global estimates are derived from a variety of disparate global biophysical and economic modeling results (e.g., Smith et al., 2008) or modeled exogenously, within large sectors, and/or using GHG aggregates (Fawcett and Sands, 2006; van Vuuren et al., 2006; Reilly et al., 2006). Meanwhile, regional studies have illustrated the importance of land-use competition in agricultural and forestry greenhouse gas mitigation (Murray et al., 2005), but have yet to endogenize land input and budget allocation decisions in economy-wide modeling (e.g., McCarl and Sands, 2007).

Results from a variety of modeling teams have found non-$CO_2$ GHG mitigation (across all sectors) to be a cost-effective means of achieving long-term climate stabilization goals—providing substantial cost savings by reducing the mitigation necessary from fossil fuel and industrial $CO_2$ emissions sources (de la Chesnaye and Weyant, 2006). However, these studies do not account for the implications of carbon policies on the net returns of different global land uses and land management decisions, the economy-wide effects of changes in the relative prices of production factors, or for the competition between uses for the same land. Failure to account for land mobility, changing prices, and input substitution, within agriculture and between agriculture and forestry is likely to result in misleading estimates of the marginal cost of mitigation for $CO_2$ and non-$CO_2$ greenhouse gases (e.g., Lee, 2004). Regional abatement potential will be influenced by changes in world and regional prices that are induced by production changes following abatement of emissions. Klepper and Peterson (2006) find such an effect for changes in the price of oil. This paper considers changes in relative prices in regional and global input and output markets driven by GHG abatement in agriculture and forestry.

Computable general equilibrium (CGE) economic models are well suited to analyze these types of complex market interactions, and have been extensively used in the climate change policy debate (e.g., Fawcett and Sands, 2006; Reilly et al., 2006). Existing CGE frameworks, however, are not currently structured to model land use alternatives and the associated emissions sources and mitigation opportunities. This work has been hindered by a lack of data on key variables, including: consistent and disaggregated global land resources, non-$CO_2$ GHG emissions, and forest carbon stocks. This paper is the first to adopt new global land-use and emissions datasets (Monfreda et al., 2009; Sohngen et al., 2009; Rose et al., 2008b), as well as new engineering mitigation costs estimates (USEPA, 2006b) in global land use modeling.

This analysis develops CGE estimates of global land-use GHG abatement potential in the near term, modeled here as a period of 20 years. Using the newly available land and emissions datasets, we construct a new globally consistent modeling framework to understand how different land-use change and management opportunities for GHG abatement interact with one another and the rest of the economy on both regional and global scales, taking into account global market clearing conditions for commodities.

The overall goal is to better capture land related GHG emissions and sequestration sources and mitigation decisions within an economy-wide framework. To achieve that goal, we have enhanced the standard GTAP global economic model with disaggregated land endowments and land use (Lee et al., 2009), detailed non-$CO_2$ greenhouse gas emissions data linked directly to economic sectors and emissions drivers (Rose and Lee, 2009), and explicit modeling of forestry intensification and extensification. Specifically, our CGE framework has a number of novel features compared to current general equilibrium modeling. We explicitly model different kinds of crop, forest, and pasture land endowments, such that we have intra- and inter-regional land, as well as GHG emissions and sequestration, heterogeneity. We utilize a disaggregated emissions and forest sequestration modeling structure that allows us to capture more refined agricultural and forestry production responses in GHG mitigation. With it we distinguish three types of agricultural mitigation responses: those associated with intermediate input use (e.g., nitrous oxide emissions from fertilizer use in crops), those associated with primary factors (e.g., methane emissions from paddy rice), and those associated with sector outputs (e.g., methane emissions from agricultural residue burning). We also introduce a method for modeling forest intensification decisions (i.e., forest management, such as longer harvest rotation) separately from forest extensification decisions (i.e., land-use change, such as afforestation and avoided deforestation).

We then analyze land allocation decisions and general equilibrium market feedbacks under emissions taxation policies. Special attention is paid to the land-using activities, including forestry, paddy rice, other cereals, other crops and livestock grazing. From this analysis, we derive estimates of the general equilibrium GHG mitigation potential for agriculture and forestry activities, such as livestock production, rice cultivation, nitrogen fertilizer applications, and carbon sequestration in forestry.

## 2. Analytical framework

We develop a modified version of the standard GTAP model (Hertel, 1997)—GTAP-AEZ-GHG—that incorporates different types of land and related GHG emissions and sequestration into the GTAP modeling framework. In so doing, we follow the path-breaking work of Darwin et al. (1995, 1996) which brought climatic and agronomic information to bear—defining different types of productive land via Agro-Ecological Zones (AEZs). Given our interest in elucidating land competition and the opportunity costs of land-based greenhouse gas mitigation technologies, we focus on non-$CO_2$ GHG emissions from agriculture and forest carbon sequestration. Except in our concluding section, we intentionally ignore economy wide fossil fuel combustion $CO_2$ emissions, as the costs of abatement through these channels are well covered in the existing literature (e.g., Hourcade et al., 2001). Furthermore, we modify the standard GTAP model to tie emissions and sequestration more directly to the underlying economic drivers and thereby allow for more refined responses for minimizing the cost of emissions taxes.

We constrain our regional disaggregation to three regions (USA, China, and Rest of World) in order to focus on the implications and insights from our new modeling approaches. Isolating the USA and China allows us to evaluate the responses of regions with different levels of economic development, and different agricultural and forestry sectors. The rest of the world is simply aggregated. We model 24 sectors, including all the major land-based emitting sectors and primary and secondary food and timber markets, as well as other sectors necessary to link these to the economy at large. The main objective of this paper is to develop new methodology. Adding more regions or sectors would simply proliferate numbers and obscure insights from the enhanced modeling structure.

### 2.1. Heterogeneous land

Given our interest in modeling the competition for land, it is important to recognize that land is a heterogeneous endowment. Just as general equilibrium analyses of labor markets should disaggregate labor by skill level, so too should analyses of land markets disaggregate land by productivity. A natural way of doing so is to identify AEZs following Darwin et al. (1995). In this study, we distinguish 18 AEZs, which differ along two dimensions: growing period (6 categories of 60-day growing period intervals), and climatic zones (3 categories: tropical, temperate and boreal). Building on the work of the FAO and IIASA (2000), the length of growing period depends on temperature, precipitation, soil characteristics and topography. The suitability of each AEZ for production of alternative crops and livestock is based on currently observed practices, so that the competition for land within a given AEZ across uses is constrained to include activities that have been observed to take place in that AEZ.

As with virtually all CGE models, inputs in our model are measured in economic value terms. Table 1 reports the annual flow of land rents attributable to the five land-using sectors in our model,

**Table 1**
Land rents at market price by AEZ and sector for the USA and China (million 2001 US$).

| Agro-Ecological Zones | USA | | | | | China | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Paddy rice | Other grains | Other crops | Ruminants | Forestry | Paddy rice | Other grain | Other crops | Ruminants | Forestry |
| Tropical | | | | | | | | | | |
| AEZ1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| AEZ2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| AEZ3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| AEZ4 | 0 | 0 | 0 | 0 | 0 | 4 | 1 | 40 | 1 | 0 |
| AEZ5 | 0 | 0 | 0 | 0 | 0 | 10 | 3 | 92 | 1 | 0 |
| AEZ6 | 0 | 0 | 0 | 0 | 0 | 250 | 60 | 2,464 | 13 | 433 |
| Temperate | | | | | | | | | | |
| AEZ7 | 0 | 1,951 | 3,435 | 3,359 | 0 | 15 | 136 | 551 | 498 | 0 |
| AEZ8 | 0 | 2,087 | 2,877 | 784 | 0 | 95 | 861 | 3,395 | 458 | 100 |
| AEZ9 | 17 | 1,926 | 2,016 | 146 | 0 | 183 | 1,376 | 5,361 | 146 | 220 |
| AEZ10 | 116 | 7,503 | 5,304 | 494 | 209 | 170 | 730 | 3,102 | 83 | 296 |
| AEZ11 | 228 | 4,346 | 2,449 | 309 | 627 | 957 | 972 | 5,175 | 126 | 780 |
| AEZ12 | 158 | 1,069 | 1,951 | 155 | 4,392 | 3,047 | 873 | 13,415 | 234 | 4,941 |
| Boreal | | | | | | | | | | |
| AEZ13 | 0 | 57 | 99 | 88 | 0 | 1 | 29 | 98 | 343 | 0 |
| AEZ14 | 0 | 13 | 28 | 30 | 0 | 0 | 17 | 31 | 267 | 0 |
| AEZ15 | 0 | 2 | 7 | 2 | 0 | 11 | 43 | 65 | 302 | 46 |
| AEZ16 | 0 | 0 | 1 | 0 | 0 | 3 | 11 | 34 | 43 | 3 |
| AEZ17 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 1 | 0 |
| AEZ18 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

across the 18 AEZs in both China and USA. Several points are immediately apparent. First of all, for any given activity, the distribution of land rents across AEZs is not uniform—nor is it the same for different land-using sectors. For example, the largest land expenditures for paddy rice are in the longest growing period temperate zones (AEZs 11 and 12), whereas for ruminant grazing, they are in the shorter growing period temperate and boreal zones (AEZs 7, 8, and 13–15). Other grains cropping activity (maize and wheat) is more pronounced in AEZ9 in China and AEZs 10 and 11 in USA. The main economic competition between forestry and cropping is in the longest growing period temperate AEZ (AEZ12; e.g., the Southeastern USA).

Our GTAP-AEZ-GHG framework retains a national production function for each land-using commodity, and introduces different AEZs as inputs to this function (see also Darwin et al., 1995; Eickhout et al., 2009). With a sufficiently high elasticity of substitution between AEZ's, we are assured that the return to land across AEZs, but within a given use (sector), will move closely together, as would be the case if we had modeled production of a given homogeneous commodity on each AEZ separately—i.e., corn from AEZ1, . . ., AEZ18 (see the Appendix for a proof of this point).

We constrain land supply across alternative uses, within a given AEZ, via a Constant Elasticity of Transformation (CET) frontier. Thus, for example, the within-AEZ returns to land in forestry and maize production are allowed to differ. This also ensures that the partial equilibrium land supply response to any use is consistent with the econometric literature. The absolute value of the CET parameter represents the upper bound (the case of an infinitesimal rental share for that use) on the elasticity of supply to a given use of land in response to a change in its rental rate. The lower bound on this supply elasticity is zero (the case of a unitary rental share—whereby all land is already devoted to that activity).

We follow the nested CET, land supply approach of Darwin et al. (1995, 1996), as well as Ahammad (2006). In this framework, land owners first decide on the optimal land mix amongst crops. The land owner then decides on the allocation of land between crops and pasture based on the composite return to land in crop production, relative to the return in ruminant livestock production. This also determines the average return to land allocated to agriculture (crops and livestock sectors) in general.

**Table 2**

Non-$CO_2$ GHG emissions by agricultural sector and emissions source (MMTCE).

| | Methane ($CH_4$) | | | | | Nitrous oxide ($N_2O$) | | | | | Total |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | Enteric fermentation | Manure management | Rice cultivation | Biomass burning | Stationary and mobile combustion | Cropland soils | Manure management | Pasture, range, and paddock | Biomass burning | Stationary and mobile combustion | |
| Emissions sources | | | | | | | | | | | |
| USA | | | | | | | | | | | |
| Paddy rice | 0 | 0 | 2.051 | 0.003 | 0.001 | 0.914 | 0 | 0 | 0.002 | 0 | 2.971 |
| Other grain | 0 | 0 | 0 | 0.085 | 0.012 | 37.557 | 0 | 0 | 0.05 | 0.004 | 37.707 |
| Other crops | 0 | 0 | 0 | 0.129 | 0.013 | 23.485 | 0 | 0 | 0.075 | 0.004 | 23.707 |
| Ruminants | 30.869 | 5.058 | 0 | 0 | 0.005 | 0 | 2.913 | 9.932 | 0 | 0.001 | 48.778 |
| Non-ruminants | 0.51 | 5.388 | 0 | 0 | 0.005 | 0 | 1.922 | 0.078 | 0 | 0.002 | 7.904 |
| Total | | | | | | | | | | | 121.067 |
| China | | | | | | | | | | | |
| Paddy rice | 0 | 0 | 59.33 | 0 | 0.005 | 10.822 | 0 | 0 | 0 | 0.004 | 70.16 |
| Other grain | 0 | 0 | 0 | 0 | 0.01 | 20.267 | 0 | 0 | 0 | 0.007 | 20.284 |
| Other crops | 0 | 0 | 0 | 0 | 0.037 | 77.847 | 0 | 0 | 0 | 0.027 | 77.911 |
| Ruminants | 51.901 | 2.089 | 0 | 0.219 | 0.002 | 0 | 5.444 | 9.067 | 0.048 | 0.002 | 68.773 |
| Non-ruminants | 2.981 | 4.577 | 0 | 0 | 0.028 | 0 | 11.928 | 22.351 | 0 | 0.02 | *41.884* |
| Total | | | | | | | | | | | 279.012 |
| ROW | | | | | | | | | | | |
| Paddy rice | 0 | 0 | 107.957 | 0.437 | 0.016 | 28.783 | 0 | 0 | 0.159 | 0.012 | 137.364 |
| Other grain | 0 | 0 | 0 | 1.136 | 0.095 | 99.65 | 0 | 0 | 0.413 | 0.069 | 101.363 |
| Other crops | 0 | 0 | 0 | 2.862 | 0.244 | 162.894 | 0 | 0 | 1.04 | 0.179 | 167.219 |
| Ruminants | 380.44 | 25.968 | 0 | 52.962 | 0.071 | 0 | 17.671 | 141.256 | 11.711 | 0.052 | 630.131 |
| Non-ruminants | 2.985 | 21.337 | 0 | 0 | 0.048 | 0 | 14.115 | 41.07 | 0 | 0.035 | 79.59 |
| Total | | | | | | | | | | | 1115.667 |
| *World total for agricultural sectors* | | | | | | | | | | | *1515.746* |

This return is compared to that in forestry in order to determine the broad allocation of land between forestry and agriculture.

Calibration of the CET land supply functions in the model is based on the available econometric evidence which suggests that the elasticity of transformation between agricultural land and forest land is less than that between grazing and crop land, and both are less than the elasticity of transformation between crop types. The most important elasticity for purposes of this paper is the elasticity of land supply to forestry, as forest sequestration subsidies send a strong signal to expand forest land. This is given a maximum value of 0.25, based on the econometric work of Choi (2004) for the United States. So $-0.25$ becomes the elasticity of transformation between agriculture and forest lands. The other transformation elasticities are set at $-0.5$ (crops vs. livestock) and $-1.0$ (elasticity of transformation amongst crops), respectively.

### 2.2. Emissions from agriculture

As documented by Rose and Lee (2009), non-$CO_2$ emissions from agriculture (crops and livestock) represent well over 50% of China's total non-$CO_2$ emissions (in carbon equivalent units), just under half of USA non-$CO_2$ emissions, and just over 60% of global non-$CO_2$ emissions. A detailed breakdown for the agricultural sectors is provided in Table 2. In the case of the USA, methane emissions from enteric fermentation and nitrous oxide emissions from crop production are the largest sources of agricultural non-$CO_2$ GHG emissions. These sources are also important in China, as are methane emissions from paddy rice cultivation. China also has significant methane emissions from its production of pigs and other non-ruminants. In the rest of the world (ROW), the largest source of agricultural non-$CO_2$ emissions is ruminant livestock production, followed by paddy rice cultivation and biomass burning, and nitrous oxide emissions associated with nitrogen applications to crops (synthetic and organic) and pasture lands (organic). From the region totals in Table 2 (italicized entries), we see that the USA and China account for 8% and 18% of global agricultural non-$CO_2$ emissions, respectively.

The emissions data in Table 2 were developed from a detailed non-$CO_2$ greenhouse gas emissions database specifically designed for use in global economic models (Rose et al., 2008b). It provides highly disaggregated emissions information that we mapped directly to countries and economic sectors utilizing available input quantity data (Rose and Lee, 2009). The result allows for a more robust and refined representation of non-$CO_2$ emissions sources in economic models and improved modeling of actual emitting activities and abatement strategies.[2] For instance, as shown in Table 2, ruminant non-$CO_2$ emissions come from manure management, enteric fermentation, fossil fuel combustion, and grazing activity, each of which can be managed separately or in combination.

To model and evaluate the general equilibrium input allocation responses to mitigation policies, we tie non-$CO_2$ emissions to explicit input or output levels. More specifically, the methane emissions associated with paddy rice production are tied to acreage cultivated, as the emissions tend to be proportional to the amount of paddy rice land. Nitrous oxide emissions from maize production are tied to fertilizer use. Emissions associated with enteric fermentation and manure management in non-ruminants are tied to livestock capital, whereas in ruminants they are tied to output (discussed below). Emissions from biomass burning, and stationary and mobile combustion are tied to sector output.

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[2] Other global emissions datasets have provided valuable regional and global estimates (e.g., USEPA, 2006a; Olivier, 2002); however, estimated emissions have been developed and presented according to IPCC source categories that aggregate across countries, and more importantly, economic sectors and activities. The Rose et al. (2008b) database provides 2001 emissions for 29 non-$CO_2$ and other $CO_2$ GHG emissions categories with 153 unique emission sources (subcategories) for 226 countries. Most of the categories and subcategories were mapped into GTAP (24 categories and 119 subcategories). The excluded categories/ subcategories include non-$CO_2$ emissions associated with biomass burning not uniquely attributable to anthropogenic activity, tropical forest fire deforestation, biomass combustion, underground storage and geothermal energy, and other $CO_2$ emissions not attributable to fossil fuel combustion. The omitted emissions subcategories will be added to the database in the future as methodologies are developed and activity data becomes available. The new dataset complements the GTAP fossil fuel combustion $CO_2$ emissions database, and the GTAP forest carbon stock dataset (Sohngen et al., 2009).

**Table 3**
Key initial emission intensities (MtC/\$ of input).

| Emission intensities (MtC/\$ of input) | | | | Forest carbon intensities[a] (MtC/\$ of land rent) | | |
|---|---|---|---|---|---|---|
| Input | USA | China | ROW | USA | China | ROW |
| Fertilizer in crops production | 0.0062 | 0.0044 | 0.0044 | 0.058 | 0.017 | 0.134 |
| Ruminant livestock capital | 0.0096 | 0.1072 | 0.0149 | | | |
| Non-ruminant livestock capital | 0.0021 | 0.0058 | 0.0036 | | | |
| Land in paddy rice | 0.0040 | 0.0125 | 0.0049 | | | |

[a] Adjusted forest carbon intensities to calibrate to Sohngen and Mendelsohn (2007) forest carbon response curves.

Any given emissions entry in Table 2 may be large because the economic activity in the sector is large (e.g., a large dairy sector), or it may be large due to a high level of emissions per dollar of input, i.e., the "emissions intensity" of a given activity. The emissions intensity is critical in determining the extent of the impact of a carbon-equivalent emissions tax on a given sector. The *ad valorem* impact of the carbon equivalent tax depends on the product of the per unit tax and the emissions intensity of the taxed activity. Table 3 reports some key emissions intensities from the model for each region. USA has the highest emissions intensity in fertilizer, but China has the highest emissions intensity for ruminants and paddy rice. These are the regions/activities where we would expect to see relatively stronger reductions in emissions following a uniform global GHG tax.[3]

As with most CGE analyses, our model represents technology via a set of production functions in which the key parameters are elasticities of substitution amongst groups of inputs. These may be viewed as smooth approximations to dozens – even hundreds – of underlying technologies, each with their own factor intensities. As the price of one input, say fertilizer, rises, firms are expected to adopt less fertilizer-intensive practices. In our framework, the scope for conservation of fertilizer is captured by the elasticity of substitution between fertilizer and other inputs. If the elasticity is large (ceteris paribus), then a small tax on fertilizer use will induce a large reduction in fertilizer use. If the elasticity is small, then it will take a large tax to induce a significant reduction in fertilizer usage at a given level of crop output. These elasticities are central to the determination of marginal abatement costs for emissions from various activities in our model.[4]

To calibrate the general equilibrium model, we constructed mitigation cost curves that correspond to the GTAP-AEZ-GHG region and sector structure using mitigation cost data from the U.S. Environmental Protection Agency (USEPA, 2006). USEPA (2006) has estimated the engineering costs and emissions implications of alternati management strategies for mitigating key agricultural non-$CO_2$ emissions sources—paddy rice, other croplands (wheat, maize, soybean), and livestock enteric and manure emissions (Chapter 5, USEPA, 2006).[5]

Since we have tied the emissions to explicit economic drivers, we are able to employ a more refined approach than that used by Hyman et al. (2003) for industrial emissions, which simply ties all emissions and mitigation to output levels. Specifically, we introduce an additional layer of substitution elasticities into the production structure that allows for substitution between input-related emissions and specific inputs. Thus, for example, we allow paddy rice producers to respond to a methane emissions tax not only by using less land, but also by *changing the emissions intensity* of land (e.g., by changing irrigation and amendment practices). This additional flexibility allows us to consider alternative calibrations to the USEPA abatement cost curves (USEPA, 2006b).

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[3] Note that emissions intensity is sometimes measured in terms of emissions per unit output. However, a high input emissions intensity does not necessarily translate into a high output emissions intensity.

[4] There is a problem in calibrating the CES production function since emissions are not initially priced. A zero price suggests zero marginal productivity in the cost-minimizing equilibrium. Therefore, we impose a very small price for emissions in the initial database, with revenue flowing to the regional household.

[5] USEPA (2006) cropland and rice paddy GHG abatement estimates includes changes in soil carbon, $N_2O$, and $CH_4$ fluxes.

**Table 4**
Elasticities of substitution: shaded boxes denote elasticities calibrated for emissions mitigation and sequestration.

| | | Sectors | | | | | |
|---|---|---|---|---|---|---|---|
| | | Paddy rice | Other grain | Other crops | Ruminants | Non-ruminants | Forest |
| Intermediate inputs | USA | 0.80 | 0.80[a] | 0.80 | 0.80 | 0.80 | 1.26[b] |
| | China | 0.50 | 0.50[a] | 0.50 | 0.50 | 0.50 | 1.80[b] |
| | ROW | 0.51 | 0.73[a] | 0.70 | 0.75 | 0.73 | 0.33[b] |
| Value-added | USA | 0.48 | 0.50 | 0.46 | 0.66 | 0.69 | 0.2 |
| | China | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.2 |
| | ROW | 0.45 | 0.48 | 0.53 | 0.60 | 0.63 | 0.2 |
| Capital and capital related emissions | USA | | | | | 0.043 | |
| | China | | | | | 0.001 | |
| | ROW | | | | | 0.030 | |
| Land and land related emissions | USA | n/a | | | | | |
| | China | 0.005 | | | | | |
| | ROW | 0.026 | | | | | |
| Output and output related emissions | USA | | | | 0.023 | | |
| | China | | | | 0.015 | | |
| | ROW | | | | 0.012 | | |

[a] In the GE model, these econometrically estimated elasticities of substitution (OECD, 2001) provide abatement very close to those from USEPA (2006).

[b] Elasticity of substitution between own-use of forest products and land, $\sigma_{\text{carbon}}$, is calibrated to reproduce the intensive sequestration response in forestry.

Before calibrating the CGE GHG abatement responses, we calibrated the base input elasticities of substitution in production, both amongst intermediate inputs and value-added and between elements of value-added. Following the approach suggested by Keeney and Hertel (2005), these parameters were calibrated to econometric estimates reported in a literature survey by the OECD (2001). For the GHG abatement response calibration, we begin by fixing output levels in the sectors, as well as input prices to match the partial equilibrium assumptions of the engineering cost estimates. We then proceed to vary the carbon equivalent price to map out a partial equilibrium abatement response from the CGE model for the relevant sector in each region. This response is compared to that estimated by EPA at $50/MtC. For $N_2O$ emissions from fertilizer use in the crops sectors, the two abatement cost estimates are in remarkably good agreement, so no further adjustment is required. However, for methane emissions from paddy rice production, the abatement response of our model (in partial equilibrium) is too low — the OECD (2001) calibrated production function suggests less scope for abatement than the USEPA estimates. In this case, we introduce the possibility of changing the input emissions intensity (recall Table 3). Specifically, we introduce scope for substitution between land and methane emissions in paddy rice production. This captures the fact that, by altering the type of cultivation practices, emissions may be reduced—but at the expense of utilizing more land for paddy rice production. Calibrated elasticities of substitution for China and ROW are given in the shaded entries of the first column of Table 4. These elasticities are very small, suggesting that this element of overall abatement is less important than the substitution of other inputs for land in paddy production. (The USA is a minor rice producing region, and regional abatement cost schedules are not available, so this elasticity is left at zero.)

A similar situation arises with methane emissions from non-ruminant production, where we add the possibility of changes in the emissions intensity per dollar of livestock capital. (See the shaded entries in that column of Table 4.) However, in the case of methane emissions from ruminant livestock production, the OECD (2001) calibrated production function gives a much larger abatement response than suggested by USEPA. In this case, we simply tie emissions to output and calibrate the substitution elasticity between emissions and output in order to replicate the USEPA abatement estimate.

### 2.3. Forest carbon sequestration

Forest carbon stocks can be increased by increasing the biomass on existing forest acreage (the intensive margin) or by expanding forest land (the extensive margin). The former can increase carbon storage per hectare by delaying harvests or modifying management (via changes in tree species or intensity of management). The latter can increase carbon stocks by afforesting non-forested lands or preventing conversion of current forest lands (i.e., avoided deforestation). We first develop regional forest carbon supply curves using the partial equilibrium, dynamic optimization model of global timber markets and forest carbon stocks described in Sohngen and Mendelsohn (2007).[6] We refer to this model as the "global timber model." We then calibrate the CGE model's regional forest carbon responses to the global timber model's supply curves.

We map out the carbon supply curves by introducing a range of carbon prices to the global timber model. The endogenous variables (e.g., harvest age, harvest area, land use change, and timberland management) adjust to maximize net surplus in the timber market and the benefits from carbon sequestration. Cumulative carbon sequestration in each period is calculated as the difference in total carbon stored between the carbon price scenario and the baseline scenario where there is no carbon tax.

The global timber model simulates long-run carbon sequestration potential by decade for 100 years. Because of our "near term" focus, we consider the potential for sequestration for a "representative" year within the first 20 years.[7] Specifically, we calculate the present value of cumulative sequestration over the first 20 years, and then calculate the annual equivalent amount, using a 5% discount rate, the rate assumed by the global timber model.[8] The results are reported in Table 5 for the three GTAP-AEZ-GHG regions.

Carbon sequestration in each region is decomposed into the amount derived from land use change, aging of timber, and modified management of existing forests. The land use change component is what we refer to as the "extensive" margin, and it is reported in the first column of Table 5.[9] These entries are determined by assessing the annual change in forestland area, tracking new hectares in forests compared to the baseline due to afforestation and avoided deforestation, and tracking the carbon on those hectares. For regions that undergo afforestation in response to carbon policies (predominantly temperate regions), carbon on new hectares are tracked by age class so that the accumulation of carbon on new hectares occurs only as fast as the forests grow. There is little gain from reductions in deforestation in the temperate forests of the USA and China, so smaller benefits from land use change are expected in initial periods in these two countries. In contrast, larger early benefits are expected in tropical regions where reductions in deforestation are the primary response to the climate policy. Reduced deforestation yields an instantaneous effect by maintaining a carbon stock that would otherwise be lost. We see evidence of this in Table 5, where the carbon storage at $5/TCE due to land use change is very small in USA and China, but the sequestration potential is quite large (143 MMTCE on an annualized basis) in the ROW region. As a consequence, the extensive margin portion of the forest sequestration supply curve for ROW is initially quite flat, indicating considerable sequestration potential for a modest cost.

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[6] The model maximizes the net present value of consumers' surplus in timber markets less costs of managing, harvesting, and holding forests. In so doing, it determines the optimal age of harvesting trees (and thus the quantity harvested) in accessible regions, the area of inaccessible forest harvested, the area of land converted to agriculture, and timber management. More expansive details are available in Sohngen and Mendelsohn (2007).

[7] Extending this horizon further would increase the potential for sequestration as longer term adjustments would be taken into account.

[8] Discounting physical carbon is common practice in the forest carbon sequestration literature (e.g., Murray et al., 2005); however, it is a topic beyond the scope of this paper. See Richards (1997) for a good conceptual discussion of issues.

[9] Table 5 does not include estimated changes in carbon storage in forests set aside at the accessible/inaccessible margin in temperate and boreal regions and in wood products in general. The global timber model estimates both, but we have chosen to exclude these factors from our CGE modeling since they account for only a very small portion of total carbon sequestration. Their omission therefore is unlikely to change our findings. Wood products could be accounted for in our framework, since we do follow wood products through markets and eventually to consumers. However, we have not yet estimated the carbon content of these flows and the associated stocks in our model.

**Table 5**
Forest carbon sequestration supply schedule: by category, annual equivalent abatement over 20 years (MMTCE)[a].

| Global Carbon price | Extensive Margin | Intensive Margin | Total |
|---|---|---|---|
| USA | | | |
| 5 | 1.672 | −1.663 | 0.009 |
| 10 | 3.509 | 6.802 | 10.311 |
| 20 | 7.023 | 24.585 | 31.608 |
| 50 | 17.811 | 73.503 | 91.314 |
| 100 | 43.069 | 102.749 | 145.818 |
| 200 | 118.287 | 119.006 | 237.293 |
| 500 | 270.741 | 286.616 | 557.357 |
| CHINA | | | |
| 5 | 0.44 | 3.018 | 3.458 |
| 10 | 0.612 | 14.865 | 15.477 |
| 20 | 1.21 | 26.899 | 28.109 |
| 50 | 4.154 | 73.928 | 78.082 |
| 100 | 12.797 | 98.522 | 111.319 |
| 200 | 73.532 | 97.503 | 171.035 |
| 500 | 108.663 | 202.142 | 310.805 |
| ROW | | | |
| 5 | 143.218 | 31.572 | 174.79 |
| 10 | 281.67 | 78.626 | 360.296 |
| 20 | 539.266 | 114.936 | 654.202 |
| 50 | 1203.164 | 250.691 | 1453.855 |
| 100 | 1672.509 | 387.619 | 2060.128 |
| 200 | 2189.741 | 366.732 | 2556.473 |
| 500 | 2885.44 | 868.723 | 3754.163 |

[a] The table does not include estimated changes in forest carbon storage in wood products and forest set asides at the inaccessible margin in temperate and boreal regions.

The combined effect of management and aging represent the "intensive" margin for sequestration, as they reflect the stock of carbon per unit of forestland.[10] The global timber model's projections for annualized sequestration at the intensive margin are reported in the second column of Table 5. Overall, there is substantial potential for increasing the forest carbon stock at the intensive margin—particularly for carbon prices up to $100/TCE.

According to Table 5, the USA and China could provide about 13% of global potential sequestration over the next 20 years. This, at first glance, is a surprisingly large proportion of the total carbon given that these countries contain only about 10% of the world's total forestland. However, the bulk of this additional sequestration in USA and China comes at the intensive margin and is attributable to changes in forest management aimed at increasing carbon stocks. The USA estimates in Table 5 are consistent with a recent detailed national assessment of USA mitigation potential in forestry and agriculture, which suggests that for $55/TCE, 88.8 MMTCE per year could be sequestered in USA forests via afforestation and forest management (Murray et al., 2005).

We calibrate the GTAP-AEZ-GHG model to Table 5 by implementing a forest sequestration subsidy with the CGE model running in partial equilibrium mode (i.e., output levels and input prices fixed). The subsidy is applied to an augmented regional land input that includes two components: composite

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[10] The aging component is estimated by comparing the carbon that accrues in forests under the particular carbon price scenario examined vs. the carbon that would have accrued in the carbon price scenario forest area (and management intensity) if managed with the baseline age classes. The algorithm used to calculate carbon due to aging does not distinguish between old and new hectares. Thus, if hectares newly forested in the mitigation scenario are eventually harvested in an age class older than the baseline age class, the carbon associated with longer rotations are counted as aging rather than as part of the afforestation component. This type of interaction between the extensive and intensive margins can give rise to negative contributions to sequestration at very low carbon prices (see the USA entry for $5/TCE in Table 5). The management component is estimated by comparing the carbon sequestered under the carbon price scenario to the carbon sequestered assuming the carbon price scenario forest area and age classes are managed with the baseline management intensities.

forest land (aggregated land from all AEZs used in the country's forestry production) and the own-use of forestry products in the forestry sector, which can be thought of as representing the volume of forest biomass on a given amount of forest land. Forest land area and forest biomass volume are allowed to substitute in production with an elasticity of substitution denoted by $\sigma_{\text{carbon}}$. While such a grouping of inputs may not appear intuitive at first glance, it works well to mimic the two margins along which forest carbon can be increased—the intensive margin (modified management and aging) and the extensive margin (more forest land).

We perform two calibrations. First, we assume that $\sigma_{\text{carbon}} = 0$. In this case, the effect of the sequestration subsidy will be to increase the profitability of forestry with current management practices, thereby leading to an expansion of forest land with constant carbon intensity. This is the extensive margin and we calibrate to it by adjusting the incremental annual carbon intensity of forests. The calibrated values of these intensities are reported in Table 3. The higher the forest carbon intensity, the greater the profitability of expanding land area in response to the sequestration subsidy. The carbon intensity is larger in ROW than in China and the USA; thereby, ROW has a comparative advantage in forest carbon sequestration, i.e., a given per unit sequestration subsidy will have a greater *ad valorem* impact in the ROW region.

Next, we calibrate the intensive margin. To do so, we fix the total land in forestry, which eliminates the extensive margin altogether, and introduce $\sigma_{\text{carbon}} > 0$ (once again running the model in partial equilibrium mode to mimic the assumptions made in Sohngen and Mendelsohn (2007). In this case, the subsidy encourages an increase in the carbon intensity of forestry. In our model, this is reflected as a substitution of own-use of forest products, in the forestry sector, for forest land. This *reduces net forestry output* (net output is gross output produced *less* own-use), thereby increasing the carbon intensity of production per unit of output. In effect, the forestry sector chooses to sacrifice some sales of commercial timber by adopting production practices that increase the carbon content on existing forest land. This intensive margin is calibrated by adjusting $\sigma_{\text{carbon}}$ until the GTAP-AEZ-GHG model replicates the intensive margin carbon sequestration response from the global timber model. The fitted values of $\sigma_{\text{carbon}}$ are reported in the final column of Table 4. We find that this formulation of the GTAP-AEZ-GHG model permits us to replicate abatement costs from the dynamic global timber model quite well for subsidies up to \$100/TCE.

## 3. Results

Having calibrated the GTAP-AEZ-GHG model to a suite of partial equilibrium GHG abatement cost curves, we now deploy our CGE model to investigate the market interactions between these different abatement opportunities. We summarize these interactions with general equilibrium global GHG abatement supply schedules, and then analyse the details within and between sectors regionally and globally. We also consider unilateral versus global carbon policies.

### 3.1. Impacts of a global carbon tax

Fig. 1a portrays the global abatement supply, including both emissions from agriculture and forest carbon sequestration, taking into account full general equilibrium adjustments.[11] Here, we see that forestry and agriculture could provide emission reductions of up to 3.0 billion metric tons of carbon equivalent (BTCE) per year in the near term. The largest share of global abatement is from the extensive margin of forestry, which may be seen as the difference between the forestry total abatement curve in Fig. 1a and the intensification curve. Most of this abatement is due to reduced emissions through avoided deforestation in tropical regions. Avoiding deforestation has a relatively large immediate impact on carbon emissions as large quantities of *in situ* carbon are preserved. Fig. 1b offers a closer look at the results for the global agricultural sectors, where the ruminants sector offers the greatest abatement potential, followed by other crops.

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[11] The general equilibrium supply schedules are derived by varying the per unit carbon tax incrementally from \$1/TCE to \$100/TCE.

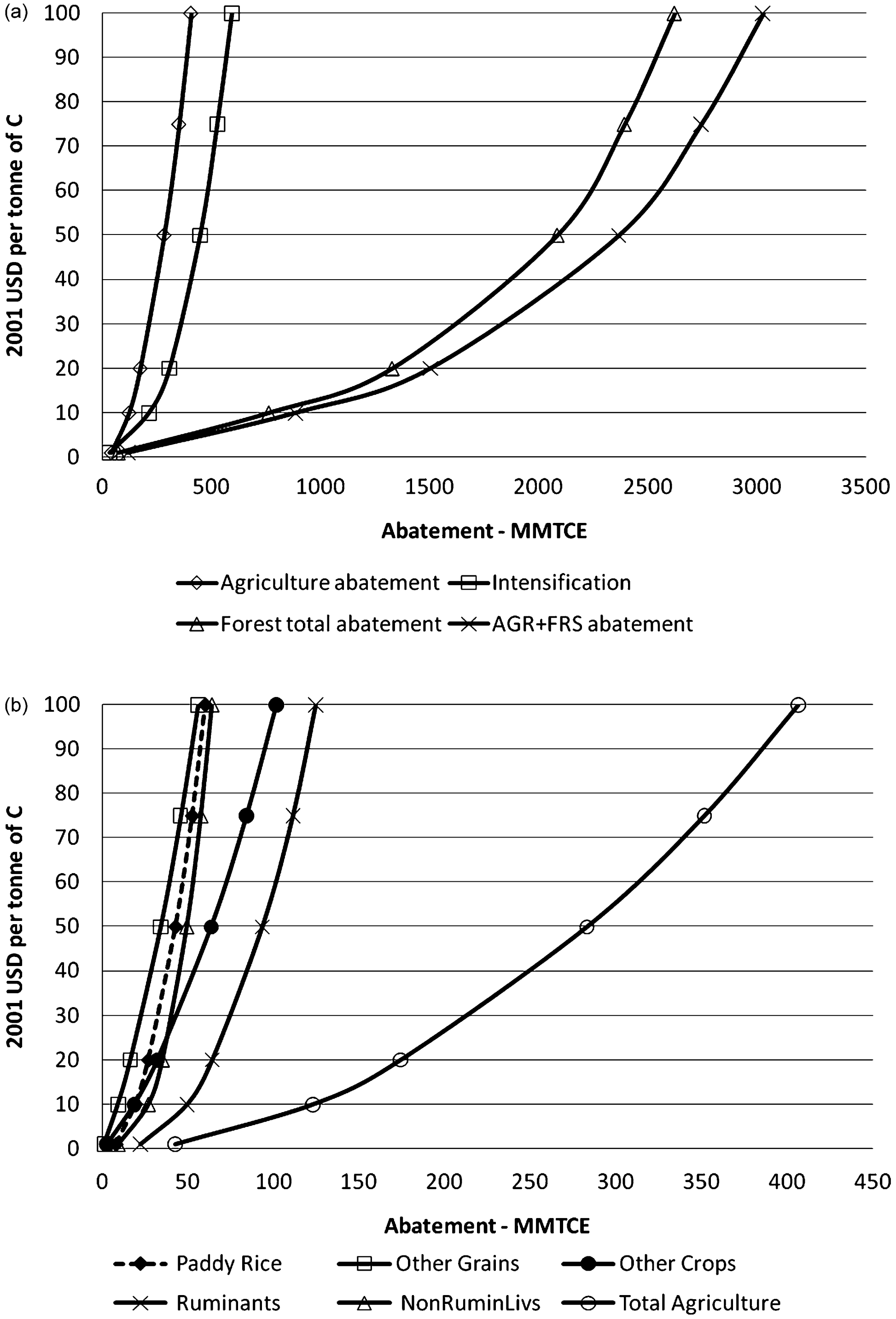

**Fig. 1.** (a) Global general equilibrium GHG abatement supply schedules. (b) Global agriculture subsector GHG abatement supply schedules.

**Table 6**
General equilibrium impact of emissions taxes on net emissions in each region following a global tax of $100/TCE in agricultural sectors and forestry (MMTCE).

| Abatement type/region | | Emissions change by region | | |
|---|---|---|---|---|
| | Global | USA | CHN | ROW |
| Fertilizer | −81 | −16 | −14 | −50 |
| Land use related emissions in paddy rice (methane) | −53 | 0 | −17 | −36 |
| Land and capital use related emissions in ruminant livestock | −111 | −6 | −11 | −93 |
| Miscellaneous | −52 | −4 | −16 | −32 |
| Forest sequestration | −2624 | −183 | −169 | −2272 |
| Total impact | −2920 | −210 | −228 | −2483 |

For ease of exposition, we focus our discussion on the highest tax level shown in Fig. 1a and b: $100/TCE. Table 6 decomposes the global abatement at this price by region (columns) and type of abatement—fertilizer, paddy rice, ruminant livestock, miscellaneous agriculture and forest sequestration (rows).[12] The reduction in total emissions for a $100/TCE global tax is largest in ROW, followed by China and USA—where abatement levels are quite similar. In all regions, forest sequestration provides the largest proportion of the total emissions reductions. Reductions in emissions from fertilizer use in USA and from paddy rice in China are the second largest abatement activities, whereas ruminant livestock related emissions are the second largest individual source of abatement in ROW. Reducing emissions from rice paddies is also important in ROW.

These results indicate that forest carbon sequestration could play an important role in the global land use emissions abatement. As noted previously, in our modeling approach, the forest margin is broken into intensification and extensification. The latter dominates in the near term in the ROW region, accounting for most of the abatement (Fig. 1a). However, sequestration attainable from intensification efforts in ROW is still quite large, around 400 MMTCE per year for $50–$100/TCE. Nonetheless, this is only about 20% of total potential abatement in ROW (Table 5). The results for ROW are heavily driven by efforts to reduce deforestation. Intensification is a more significant part of the forest abatement story in the USA, accounting for nearly all of the abatement at low carbon prices (less than $20/TCE), and more than 67% at carbon prices of $100/TCE (Table 5).

Carbon sequestration through forest extensification has two different effects on emissions from agriculture. On one hand, forest extensification bids land away from agriculture production, thereby reducing output and hence emissions—particularly of those GHG emissions linked to land use. On the other hand, it encourages more intensive production on the remaining land in agriculture, which can drive up GHG emissions from any particular hectare. In a separate simulation of a forest sequestration subsidy alone, we ascertained that the former effect dominates, so that sequestration-driven forest extensification reduces overall agriculture emissions on net.

### *3.2. Changes in agricultural factor intensities and output*

Table 7 reports the percentage changes in agricultural output and input use intensities (land, labor and fertilizer) in the three regions. We see that some of the largest changes in input intensities are for fertilizer use in USA crop production, where the emissions intensity is quite high (recall Table 3), and land use in paddy rice production in all three regions. These responses are directly related to the emissions tax on these inputs. For example, in order to reduce fertilizer usage in USA corn production,

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[12] The decomposition of global emissions by sector or emissions type utilizes the numerical integration technique proposed by Harrison et al. (2000) to apportion the impact of each group of instruments on total emissions in each region or in the world. This has the virtue of producing individual estimates that add to the grand total. This would not be the case if the simulations were conducted separately, due to interaction effects.

**Table 7**
Percentage changes in agricultural output levels and input intensities following a $100/TCE global carbon tax in agricultural sectors and forestry.

| | | Factor of production | | |
|---|---|---|---|---|
| | Output | Land | Labor | Fertilizer |
| USA | | | | |
| Paddy rice | 19 | −22 | 10 | −21 |
| Other grains | 1 | −6 | 8 | −27 |
| Other crops | 4 | −8 | 5 | −29 |
| Ruminants | 2 | −8 | 3 | |
| China | | | | |
| Paddy rice | −7 | −20 | 17 | −3 |
| Other grains | −2 | 0 | 6 | −12 |
| Other crops | −1 | −1 | 5 | −13 |
| Ruminants | −7 | 4 | 6 | |
| ROW | | | | |
| Paddy rice | −4 | −17 | 13 | −6 |
| Other grains | −4 | −16 | 9 | −16 |
| Other crops | −3 | −16 | 8 | −15 |
| Ruminants | −6 | −20 | 6 | |

farmers use more of other inputs. Thus labor usage per bushel of corn rises by 8%, while other variable input intensities rise by even more.

Of course total emissions in any of these sectors depend, not just on input–output emissions intensities, but on total output. The first column in Table 7 reports the changes in output in these land-using agricultural sectors. Agricultural output in ROW falls in all of the land-using sectors, as forestry area expands in response to the sequestration subsidy. Of course, in practice, much of the loss in ROW agricultural area will actually be forgone deforestation. So these should be viewed as changes, relative to baseline. In China, there are declines in all these sectors as well, with the largest percentage output reductions arising in the cases of paddy rice and ruminant livestock production, where China's emissions intensities are relatively large (Table 3). In contrast, agricultural output in USA agricultural sectors rises, as the emissions tax scenario favors USA farming. The largest percentage output increase occurs in USA paddy rice and amounts to 13% in the wake of rising world prices due to declining output in China and ROW. This increase is attained through higher yields, and not increased land area.

### 3.3. Changes in Land Use

Competition for land – particularly between forestry and agriculture – is playing an important role in determining regional abatement and output responses. Therefore, in this section, we investigate the land market in more detail. We do this in Table 8 for the USA and China.

Since the nested CET transformation function does not model a constraint on total physical hectares in a given AEZ, but rather preserves the sum of productivity-share-weighted hectares within each AEZ, where productivity is based on observed land rents, we report the land quantity outcomes as share-weighted percentage changes in land use (Table 8). The weights are the share of total land rents *in a given AEZ*, generated by a particular activity. The entries in Table 8 should be interpreted as percentage changes, relative to the annual flow of economic value associated with total land in a given AEZ.

In the temperate AEZ with the longest growing period in the USA (AEZ12), we find expanding forestry production in response to a $100/TCE forest carbon sequestration subsidy that absorbs more than 7 percent of the land endowment in that AEZ. The majority of the expanded forest land is from other crops. However, all of the agricultural sectors are giving highly productivity land in AEZ12 to forestry in response to the sequestration subsidy. In AEZ11, there is a similar percentage expansion in forestry, but now two-thirds of it comes from grains production. This reflects the differences in current economic activities (and productivity) across AEZs.

**Table 8**
Percentage changes in rental share weighted land use, by AEZ and sector in USA and China due to a $100/TCE global carbon tax in agricultural sectors and forestry.

| | Forestry | Paddy rice | Other grains | Other crops | Ruminants |
|---|---|---|---|---|---|
| USA | | | | | |
| AEZ1–AEZ6 | 0 | 0 | 0 | 0 | 0 |
| AEZ7 | 0 | 0 | 0.33 | 0.86 | −1.18 |
| AEZ8 | 0 | 0 | 0.09 | 0.48 | −0.57 |
| AEZ9 | 0 | 0.03 | −0.10 | 0.23 | −0.16 |
| AEZ10 | 2.17 | 0.04 | −1.28 | −0.66 | −0.23 |
| AEZ11 | 7.37 | −0.07 | −4.24 | −2.26 | −0.42 |
| AEZ12 | 7.32 | −0.29 | −2.27 | −4.04 | −0.36 |
| AEZ13 | 0 | 0 | 0.32 | 0.84 | −1.15 |
| AEZ14 | 0 | 0.00 | 0.29 | 0.95 | −1.23 |
| AEZ15 | 0 | 0 | 0.08 | 0.84 | −0.85 |
| AEZ16 | 0 | 0 | −0.02 | 0.02 | 0 |
| AEZ17, AEZ18 | 0 | 0 | 0.00 | 0 | 0 |
| China | | | | | |
| AEZ1–AEZ3 | 0 | 0 | 0 | 0 | 0 |
| AEZ4 | 0 | −1.60 | 0.03 | 1.63 | −0.03 |
| AEZ5 | 0.10 | −1.85 | 0.04 | 1.77 | −0.02 |
| AEZ6 | 3.65 | −1.66 | −0.05 | −1.81 | −0.03 |
| AEZ7 | 0 | −0.24 | 0.17 | 0.95 | −0.87 |
| AEZ8 | 0.74 | −0.40 | −0.07 | 0.08 | −0.35 |
| AEZ9 | 1.06 | −0.52 | −0.16 | −0.29 | −0.09 |
| AEZ10 | 2.15 | −0.81 | −0.29 | −0.92 | −0.10 |
| AEZ11 | 2.96 | −2.42 | −0.10 | −0.28 | −0.08 |
| AEZ12 | 4.90 | −2.88 | −0.12 | −1.65 | −0.08 |
| AEZ13 | 0 | −0.03 | 0.15 | 0.65 | −0.77 |
| AEZ14 | 0 | −0.02 | 0.16 | 0.36 | −0.50 |
| AEZ15 | 3.15 | −0.46 | 0.00 | 0.05 | −2.65 |
| AEZ16 | 1.11 | −0.57 | 0.13 | 0.61 | −1.26 |
| AEZ17 | 0 | −0.74 | 0.43 | 1.05 | −0.72 |
| AEZ18 | 0 | 0 | 0 | 0 | 0 |

While USA agricultural land is moving into forestry in some AEZs, cropland is moving to other AEZs in response to the net effect of rising prices and the carbon tax. Specifically, land area for production of other crops and other grains expands in those AEZs where forestry is not a significant activity. Meanwhile, land devoted to ruminant livestock production shrinks across the board.

Forestry expands in all AEZs in all regions. The largest rental share-weighted increases are in ROW (not shown), where the expansion is on the order of 12% in the more productive tropical and temperate AEZs. Forestry expansion in China is more modest, when weighted by its share of land rents. In China, land area in paddy rice and ruminants falls in all AEZs, with the land often being absorbed by the other crops and other grains sectors. While across regions there are similarities in that there are broad land conversions from agriculture to forestry, the detailed land-use and GHG changes within regions (within-AEZs and across-AEZs) varies by region.

## *3.4. Changes in global competitiveness*

Table 9 reports the change in regional trade balances due to the global carbon tax of $100/TCE. From these results we see that the carbon tax changes the pattern of global competitiveness. The dramatic expansion of forest lands in ROW squeezes the amount of land available for crops and grazing. Thus, ROW shows a deterioration in its trade balance for all other land-using sectors. Of course, ROW must somehow pay for these increased imports and they do so largely with increased exports of forest products, as well as manufactures and services.

**Table 9**
Changes in regional trade balances due to a $100/TCE global carbon tax in agricultural sectors and forestry.

| Sector | Net exports (Million US$/year) | | |
|---|---|---|---|
| | USA | CHN | ROW |
| Rice | 638 | 17 | −663 |
| Other grains | 2312 | 142 | −2376 |
| Other crops | 2166 | 999 | −2885 |
| Ruminants | 3689 | −541 | −3006 |
| Non-ruminants | 1601 | −654 | −881 |
| Other foods | 1552 | −562 | −490 |
| Forest products | −3982 | 32 | 4431 |
| Fertilizer & energy intensive manufacturing | −1470 | 4849 | −1611 |
| Other manufacturing and services | −4702 | −2931 | 4326 |
| Total | 1804 | 1350 | −3154 |

The USA, on the other hand, benefits from its lower emissions intensities in rice and livestock production, strongly expanding net exports of these products. In the case of other grains production, USA has a high emissions intensity. However, this does not stop the USA from expanding net exports of crops to ROW, in the wake of the reduction in crop area in that region. The only land-using sector where China strongly increases net exports is other crops, where it has a lower emissions intensity than USA and substantial export potential due to low wages in labor intensive sectors (e.g., fruits and vegetables).

### 3.5. Unilateral vs. global abatement costs

An important aspect of climate policy relates to how well countries coordinate their actions. Carbon price differences across regions could distort markets. It is therefore useful to assess how the general equilibrium supply of abatement is affected by regional carbon policies. Abatement analysis is frequently conducted on a country-by-country basis, implicitly assuming that other countries do not have carbon policies (e.g., Murray et al., 2005, for the USA). To explore this issue, we construct a simple example, beginning with the global carbon tax policy described above. The USA general equilibrium abatement supply for forestry (intensification and extensification) and agriculture with a global carbon tax of $100/TCE is 210 MMTCE, with 27 MMTCE from the agricultural sector and 183 MMTCE through forest sequestration. Now contrast this with the case where abatement is implemented in the USA alone. In this case, at $100/TCE, USA abatement is 217 MMTCE—about 4 percent more abatement, with forest sequestration falling slightly to 180 MMTCE and agricultural abatement increasing significantly to 38 MMTCE. These results illustrate important global market effects that affect the cost and net environmental effectiveness of mitigation. Under the US only tax, agricultural abatement is about 40% higher than under the global tax, while forest sequestration is about 2% lower. The domestic only carbon tax increases the cost of USA agricultural products relative to overseas production. As a result, non-USA production increases, as do overseas GHG emissions. On the other hand, when the tax is applied globally, USA agriculture is able to exploit its comparative production advantage; thus USA-based GHG abatement in agriculture becomes more expensive as the opportunity cost of mitigation increases. In short, carbon policies abroad will affect domestic abatement costs, production, net exports, and the het environmental benefits of domestic abatement. Studies that only examine national carbon policies, and do not consider the relative effects of regional carbon policies, could significantly mis-estimate the extent of abatement in agriculture and forestry.[13]

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[13] While many other features of the two studies differ, it is also instructive to compare our results directly to those of Murray et al. (2005), who find that approximately 8 MMTCE of CH4 and $NO_2$ emissions can be abated in the agricultural sector annually, from 2010 to 2019, and about 100 MMTCE per year can be sequestered in the forestry sector for $55/TCE. In our study, at $55/TCE, 10 MMTCE can be abated in the agricultural sector under the global coordinated tax, and 20 MMTCE can be abated each year under the USA only tax. For forest sequestration, around 110 MMTCE can be sequestered under the global coordinated tax and 120 MMTCE under the USA only tax.

## 4. Conclusions

We have developed a global computable general equilibrium model with explicit unique regional land inputs to crop, livestock, and timber production, detailed non-$CO_2$ GHG emissions, and forest carbon stocks. The GTAP-AEZ-GHG model is then augmented with cost and GHG response information from two partial equilibrium approaches to abatement of land-based greenhouse gas emissions and enhanced forest carbon sequestration. For agricultural mitigation of GHGs, we calibrate our model based on mitigation possibilities derived from detailed engineering and agronomic studies developed by the U.S. Environmental Protection Agency (USEPA, 2006). In the case of forest carbon sequestration, we draw on estimates of optimal sequestration responses to global forest carbon subsidies, derived from the model used in Sohngen and Mendelsohn (2007).

Using this framework, we estimate general equilibrium abatement supply for non-$CO_2$ mitigation in agriculture and for forest carbon sequestration. We find that abatement in agriculture and forestry could be as large as 3.0 billion TCE per year over the next 20 years at $100/TCE. Biophysical and economic characteristics, however, are shown to have important influences on the comparative abatement advantage of GHG mitigation across sectors within a given country, and between the same sectors across different countries.

In our results, forest carbon sequestration is found to have the lowest marginal costs for global GHG emissions reduction in the land-using sectors, accounting for around 90% of total abatement at $100/TCE. When compared to the rest of the world, emissions abatement in China comes disproportionately from agriculture, and, within USA agriculture, disproportionately from reductions in fertilizer-related emissions. In the world as a whole, agriculture-related mitigation comes predominantly from reduced methane emissions from ruminant livestock, which is followed in relative importance by reductions in fertilizer use and then methane emissions from paddy rice.

In our framework, we explicitly model forest carbon sequestration intensification (increased carbon per hectare) and extensification (increased forest hectares). The results show that intensification has significant mitigation potential in all regions. The potential is relatively larger in the USA and China in this analysis, which is intuitive given the substantial experience with managing timber in those regions. Forest extensification has the largest abatement potential in the ROW region. Over the next 20 years, ROW extensification largely implies a reduction in deforestation. We also find that extensification has a positive feedback effect to the agricultural sector, as more land is maintained in forests rather than converted to agriculture and overall emissions in the agricultural sector decline.

A comparison of carbon tax policies implemented globally, on the one hand, and only in the USA, on the other, shows the importance of this general equilibrium and global analysis. For USA agriculture, abatement potential is diminished by 29 percent when we move from a USA-only carbon tax to a global carbon tax. This is a consequence of the strong export orientation of USA agriculture, which responds to reduced production in the rest of the world (under a global tax) by increasing its own production and hence emissions. These results imply that national level analyses for the USA could under-estimate the costs of emissions abatement and overestimate the net GHG reduction benefits because they do not account for the implications of price changes that occur elsewhere in the world. In general, these results suggest that analyses of large-scale agriculture and forestry mitigation potential need to consider relative changes in regional prices.

A natural extension of this work is to integrate the analysis of non-$CO_2$ emissions and carbon forest sequestration with the more conventional analyses of $CO_2$ emissions from fossil fuel combustion and industrial sources. The latter have been extensively analyzed in global general equilibrium models. The approach outlined in this paper will allow for more structured and rigorous consideration of the trade-offs between these two broad types of mitigation options. Preliminary simulations with our model, augmented with $CO_2$ emissions from fossil fuel combustion in all sectors, suggest that, at $100/TCE, global agriculture and forestry sector mitigation could rival abatement of fossil fuel emissions, with the relative contributions varying significantly across regions. This result, in addition to the complex responses and interactions we've illustrated in this paper, further motivates the need for comprehensive economic assessments of GHG mitigation that explicitly captures the heterogeneous opportunity costs of land management and land-based mitigation. Furthermore, in this context, one could also model and assess the complex relationships between land and energy markets associated with biofuels and bioelectricity, which simultaneously modify the opportunity costs of alternative land-use and energy feedstocks.

## Appendix A. Appendix

In this appendix we explore the restrictions on our aggregate specification of technology in each country stemming from the "true model" in which we have a separate production function for each crop/AEZ pair in the country. Begin with the zero profit condition, as dictated by the maintained hypothesis of perfect competition. In the notation below, lower case letters denote percentage changes, and upper case letters denote levels variables. We know that for zero profits to hold, the percentage change in output price for AEZ $j$, denoted by lower case $p_j$, must equal the cost share-weighted sum of the percentage changes in price paid (denoted by lower case $w_{ij}$) for input $i$ employed in AEZ $j$, the level of which is given by $L_{ij}$:

$$p_j = \sum_i \theta_{ij} w_{ij} \tag{A.1}$$

where $\theta_{ij} = W_{ij}L_{ij}/P_jQ_j$ is the share of total costs expended on input $L_{ij}$ at price $W_{ij}$ in the production of output in AEZ $j$, $Q_j$ (upper case denotes levels variables). In the context of a global model, where there is a single factor market clearing condition for the non-land factors in each country, there must be a unique national market price for non-land inputs (e.g., fertilizer, or labor), so that $w_{ij} = w_{ik}$ for input $i$ used in both AEZs $j$ and $k$.[14] Similarly, if two AEZs produce an identical commodity (e.g., wheat), then product prices will be the same, so their percentage changes will also be equal: $p_j = p_k$. If, in addition, we make the assumption that *non-land* input–output ratios ($L_{ij}/Q_j$: e.g., kilogram fertilizer per bushel of maize) are the same across AEZs, then the non-land cost shares must also be equalized across sectors: $\theta_{ij} = \theta_{ik}$.[15] Therefore, we have the following result, where the $L$ subscript refers to land, and subscripts $j$ and $k$ refer to different AEZs producing the same product:

$$\theta_{Lj} w_{Lj} = p_j - \sum_{i \neq L} \theta_{ij} w_{ij} = p_k - \sum_{i \neq L} \theta_{ik} w_{ik} = \theta_{Lk} w_{Lk} \tag{A.2}$$

From equation (A.2) we see that the cost-share weighted percentage change in land rents across sectors must be equalized. Furthermore, since the cost shares must sum to one, and since the cost shares for non-land inputs across AEZs are equal as a consequence of equal input prices and equal input–output ratios, then so too must the land cost shares be equalized across AEZs: $\theta_{Lj} = \theta_{Lk}$. Importantly, this *does not imply* that the *level* of land rents will be equalized across AEZs. With differing crop yields, *land rents must vary in direct proportion to yield*, so that a low yield (high input–output ratio for land) will be precisely offset by a low level of land rents, thereby resulting in an equalization of land cost shares across AEZs. Since $\theta_{Lj} = \theta_{Lk}$, equation (A.2) gives us the result that: $w_{Lj} = w_{Lk}$. In order to ensure that the return to land in a given crop changes at the same rate, regardless of AEZ, we must assume that the AEZs are (nearly) perfect substitutes in aggregate agriculture and forestry production as asserted in the text.

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[14] Of course, the firms' factor prices could differ due to taxes or subsidies that varied by AEZ subsector, but we do not have data at this level of detail (taxes are only reported at the sector level).

[15] The assumption of equal non-land factor intensities could be questioned. For example, the labor intensity (hours/bushel of corn) might be higher on low productivity land, and pesticide use could vary with rainfall or frost days. However, the only data available to us is that for the entire corn sector at the national level. So we have no real choice other than to make this assumption.

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