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Abstract

In response to the Renewable Fuel Standard, the U.S. transportation sector now consumes a substantial amount (13.3 billion gallons in 2010) of ethanol. A key motivation for these mandates is to expand the consumption of biofuels in road transportation to both reduce foreign oil dependency and to reduce greenhouse gas (GHG) emissions from the consumption of fossil fuels in transportation. In this paper, we present the impacts of several biofuels expansion scenarios for the U.S. in which scaled increases in the U.S. corn ethanol mandates are modeled to explore the scalability of GHG impacts. The impacts show both expected and surprising results. As expected, the area of land used to grow biofuel crops increases with the size of the policy in the U.S., and some land-use changes occur abroad due to trade in agricultural commodities. Because the land-use changes happen largely in the U.S., there is an increase in U.S. land-use emissions when natural lands are converted to agricultural use in the policy scenarios. Further, the emissions impacts in the U.S. and the rest of the world in these scenarios, including land-use emissions, scale in direct proportion to the size of the U.S. corn ethanol mandates. On the other hand, the land-use emissions that occur in the rest of the world are disproportionately larger per hectare of change due to conversions of more carbon-rich forests to cultivate crops and feed livestock.

1. INTRODUCTION

Increasing awareness of the negative consequences of global climate change caused by anthropogenic greenhouse gas (GHG) emissions (IPCC, 2014) has led to developed countries implementing policies to reduce carbon dioxide (CO_2) emissions in different sectors of the economy. Transportation is an essential aspect of economic activity and development, which is linked to greenhouse gas emissions. These emissions, almost exclusively from oil, contribute to a large share of the total global GHG emissions (13% in 2004); and in the U.S. they are 27.3% of national GHG emissions (Davis *et al.*, 2015). The reduction of these emissions requires replacing oil with a less CO_2-intensive fuel that is ideally as energy-dense, cheap to produce and easy to transport safely. Alternatives to using oil or an oil-substitute have serious limitations: cost of batteries and their limited capacity for electric vehicles; safety[1] and larger fuel storage volumes needed for natural gas vehicles (NGV); etc. (Cazalot *et al.*, 2012). Biofuels offer an alternative that can displace oil without requiring a major overhauling of the fueling networks and, subject to blending restrictions in some cases, can be simply "dropped in." The use of biofuels instead of petroleum-based fuels can reduce GHG emissions, as the tail-pipe carbon emissions from biofuels are compensated by the carbon sequestered during growth of the biofuel crop, but the overall impact depends on the lifecycle emissions of biofuels[2]. At the same time, the cost of producing biofuels remains higher than producing comparable refined oil products (Winchester and Reilly, 2015). Adoption of biofuels thus requires policy instruments to compensate for the production cost difference between fossil fuels and bio-energy. Policies such as the Renewable Fuel Standard[3] (RFS) in the U.S. (EPA, 2010) and the implementation of carbon trading using the U.N. "Reducing Emissions from Deforestation and forest Degradation" (REDD) program in the European Union (Angelsen *et al.*, 2008) are examples of such undertakings. It is important to note that the price of crude has declined significantly and is not likely increase to the earlier high levels of over \$120/barrel (bbl) in the near term, but may remain below that level until 2040 according to the latest IEA projections (IEA, 2015). If this holds true, it will exacerbate the production cost differences between petroleum products and biofuels, increasing the cost of the policies.

Discussions of the benefits of biofuels for climate change should account for their potentially negative interference with food production, livestock feed and other agricultural sectors that need

[1] In the U.S., safety concerns for NGVs impose a higher premium on their production, limiting their penetration despite the lower fuel prices. In some countries, such as Argentina, Iran and Pakistan, the lower price of natural gas relative to oil and limited safety regulations allows for a larger penetration of NGVs.

[2] The well-to-wheel emissions presented in Table 1B in the online supplementary material for Searchinger *et al.* (2008) indicate that if carbon credits for biomass are included and land-use change emissions are excluded, then CO_2 emissions from burning the ethanol are almost equal to the carbon off-set from growing the crop.

[3] The Renewable Fuel Standard (RFS) is a program developed in compliance with the Clean Air Act (amended by the Energy Policy Act, 2005) and the Energy Independence and Security Act (EISA) by the U.S. Environmental Protection Agency (EPA). The goal of this program is to reduce air pollution and GHG emissions through the increased use of renewable fuels. This increase is achieved through requiring petroleum refiners and importers to blend a certain percentage of biofuels into their outputs of gasoline and diesel. The first RFS finalized by 2007 and focused on gasoline and ethanol. The second, known as RFS2, took effect for biodiesel in July of 2010 (http://www2.epa.gov/renewable-fuel-standard-program).

materials from crops. At the same time, the production of biofuels also increases the competition for inputs, especially cropland, that are required by other sectors. To meet the competing demands for food, livestock and ethanol, the area of cropland used to cultivate corn in the U.S. has grown despite increases in yield (USDA, 2015). As more ethanol is produced, this competition for cropland intensifies, leading to increased land rents for crops, which in turn induces land from other uses to be turned into cropland. In the future, to meet demand, some of the current domestically-produced crops may be imported, which would affect global food and agricultural markets. Thus, biofuel adoption in a single region can lead to leakage of impacts into other areas of the world where there is no direct biofuels trade or production.

Perhaps the impact of biofuels that is most relevant to GHG emissions reduction is the change in emissions from terrestrial ecosystems as the adoption of biofuels induces land-use change. The impact of biofuels on carbon emissions from both the economy and land-use change has been the subject of several studies (Searchinger *et al.*, 2008; Wise *et al.*, 2009; Melillo *et al.*, 2009; Hertel *et al.*, 2010; Reilly *et al.*, 2012). These emissions can be both large and of either sign (sequestration is a possibility). Outcomes depend on the details of land transitions and carbon stocks as well as the unique features of the impacted ecosystems.

Estimates of emissions factors for each type of land conversion in different economic regions are used in Wise *et al.* (2009) and Hertel *et al.* (2010) (with 14 and 18 regions, respectively); however, these studies do not have any biogeochemistry feedback between changes in land productivity due to climate conditions and the economic decisions of land use, land rents and land use transitions. Melillo *et al.* (2009) do account for feedbacks of changes in land productivity from climate using a loosely linked economic model and a fine resolution biogeochemistry model. Reilly *et al.* (2012) advance the framework described in Melillo *et al.* (2009) by dynamically linking these models. The framework in Reilly *et al.* (2012) is designed to investigate a global emissions policy that prices carbon to meet emissions targets; in this study, we will use the same framework to examine the scalability of biofuel mandates and their effects in the U.S.

The goal of this paper is to explore any non-linearities in global land-use emissions if corn ethanol mandates in the United States are expanded. We use the MIT Integrated Global Systems Modeling (IGSM) framework, which includes the Economic Projection and Policy Analysis Model (EPPA; Paltsev *et al.*, 2005) and the Terrestrial Ecosystem Model (TEM; McGuire *et al.*, 2001, Felzer *et al.*, 2004). We will analyze the land-use emissions and other economic impacts from a U.S.-only policy out to 2050, both globally and locally. We will examine the response to increased mandates in these policies, focusing on the areas affected by land-use change, associated land-use emissions, oil production and consumption, and the emissions from the rest of the energy sector.

We present our study design and details of the biofuel production scenarios in Section 2. The models used and our methods for this study are presented in Section 3. The economic impacts and land-use emissions for both the U.S. and the rest of the world are explored in Section 4. Conclusions are drawn in Section 5.

2. STUDY DESIGN

The largest share of biofuels produced and consumed in the U.S. comes from corn ethanol. Market penetration of biofuels, especially corn ethanol, is limited by their cost of production and the readiness of vehicles to use them. The higher cost of production means that policy instruments are crucial for biofuel adoption. Consequently, the assessment of biofuels' impact on GHG emissions faces uncertainty from policy mandates.

We consider one reference and three policy scenarios, benchmarked to data from the EIA Annual Energy Review (2012), which shows that the U.S. produced 13.3 billion gallons (bg) of corn ethanol (1.02 Quads[4]) and 0.49 bg ethanol-equivalent (bge) of bio-diesel (0.04 Quads[5]) in 2010. The reference scenario is chosen to be identical to EIA's Annual Energy Outlook 2014 projections. In this scenario, corn ethanol production increases to 15.42 bg and biodiesel production increases to 2.44 bge by 2050. These EIA projections differ from the RFS mandates primarily in the volumes of cellulosic biofuels, a technology still not commercialized, that are assumed to be produced. There is a small volume of cellulosic biofuels introduced in 2015, which reaches 0.43 bge by 2050. The required biofuel volumes in the reference scenario in the U.S. are shown in **Figure 1**.

In the policy scenarios, we increase the mandated corn ethanol volume by 1, 5 and 10 bg relative to the reference scenario described above. These increases are imposed in linear increments beginning in 2010 so that the additional amount of ethanol specified in each policy

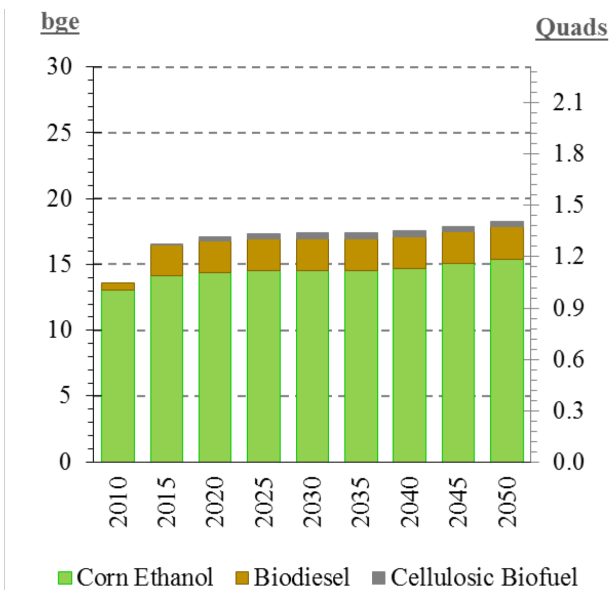

Figure 1. Biofuel volumes in billion gallons of ethanol equivalent, bge, (left axis) and Quads (right axis) produced in the reference case using the projections in EIA AEO (2014).

[4] Quad is the abbreviation used for one quadrillion (10^{15}) British thermal units (BTU). This is related to Joules (J) by BTU = 1,055 J, and 1 Quad = 1.055×10^{18} J = 1.055 EJ (E = Exa = 10^{18}). The energy content of ethanol is assumed to equal its low heating content value = 77,000 BTU/gallon.

[5] The energy content of bio-diesel assumed here is 118,000 BTU/gallon

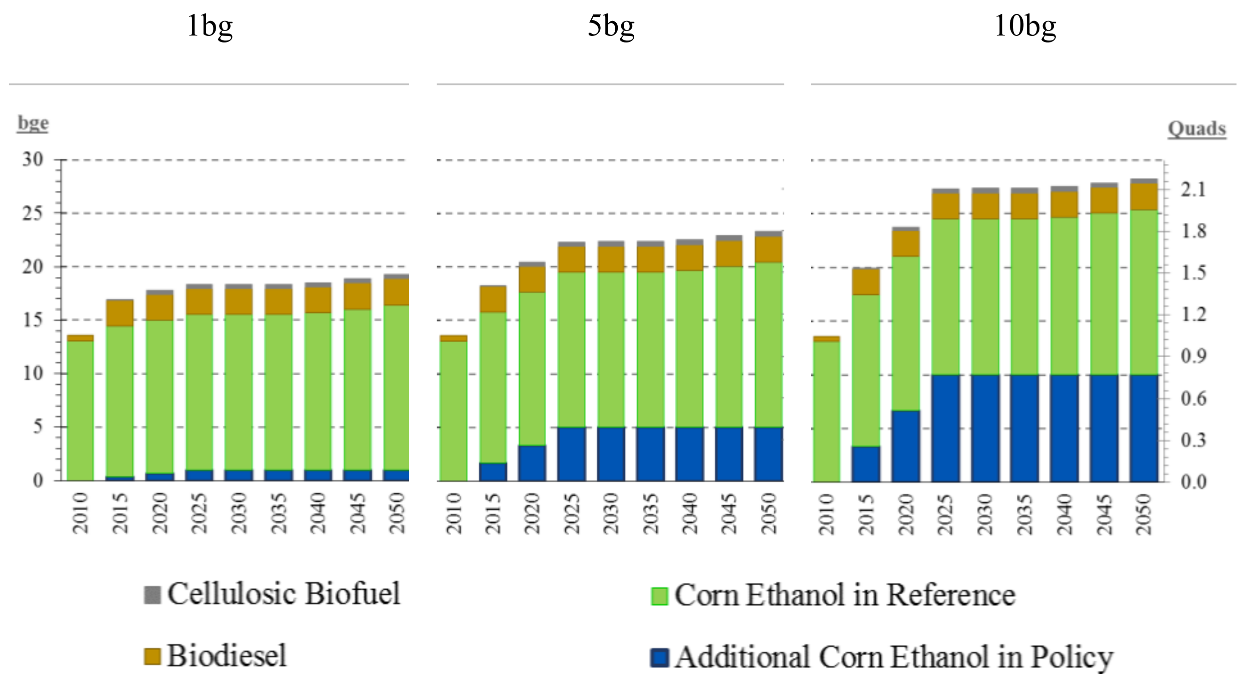

Figure 2. Volumes of corn ethanol produced in the policy scenarios, where the additional ethanol volumes are in blue.

scenario is reached by 2025, after which the additional volume is held constant. We call these scenarios *1bg*, *5bg* and *10bg*, respectively. The policies in *1bg*, *5bg* and *10bg* represent a 7.5%, 37.6% and 75.2% increase relative to 2010 production respectively. As EPPA has a five-year time step, the first policy year in our study is 2015. The biofuel production profiles for the policy scenarios are presented in **Figure 2**, where the increases in corn ethanol production are highlighted in blue.

3. MODELS AND METHODS

We model the impacts of the biofuel policies using components of the IGSM. Land use and land-use changes are computed in the economic module (EPPA), based on demand for agricultural sector activity levels and available land resources. The land-use emissions attributable to the land use and land-use change caused by economic activity are calculated using the biogeochemistry module (TEM).

3.1 Economic Projection and Policy Analysis Model (EPPA)

EPPA is a recursive-dynamic, computable general equilibrium model (CGE) of the global economy (Paltsev *et al.*, 2005). The input-output component of the model draws on the GTAP dataset (Dimaranan, 2006), which is aggregated to 14 non-energy and 25 energy sectors (see **Table 1**), and 16 regions (see **Figure 3**). These sectors are aggregate representations for all economic activity. The base year of the model is 2004, with 2005 and 2010 calibrated to the historic energy and economic data from the International Energy Agency (IEA, 2014) and the International Monetary Fund (IMF, 2015).

Table 1. EPPA sectors and factor inputs.

	Energy Sectors:	
	COAL	Coal
	OIL	Crude Oil
	ROIL	Refined Oil
	BOIL	**Liquid Fuel from Biomass**
Biomass Liquid Fuel	CORNE	Corn Ethanol
Biomass Liquid Fuel	WHEATE	Wheat Ethanol
Biomass Liquid Fuel	SUGARE	Sugarcane Ethanol
Biomass Liquid Fuel	BEETE	Beet Ethanol
Biomass Liquid Fuel	RAPESO	Rapeseed Biodiesel
Biomass Liquid Fuel	PALMO	Palm Biodiesel
Biomass Liquid Fuel	SOYO	Soy Biodiesel
Biomass Liquid Fuel	BIO-OIL	Cellulosic Ethanol
	NGAS	Natural Gas
	SYNG	Synthetic Gas from Coal
Electricity Generation Technologies	ELEC	Conventional Fossil
Electricity Generation Technologies	HYDR	Hydropower
Electricity Generation Technologies	NUCL	Nuclear
Electricity Generation Technologies	ADV-NUCL	Advanced Nuclear
Electricity Generation Technologies	BIOELEC	Biomass
Electricity Generation Technologies	NGCC	Natural Gas Combined Cycle
Electricity Generation Technologies	NGCAP	Natural Gas with CCS
Electricity Generation Technologies	IGCAP	Coal with CCS
Electricity Generation Technologies	SOLAR	Solar
Electricity Generation Technologies	WIND	Wind
Electricity Generation Technologies	WINDGAS	Wind with Natural Gas backup
Electricity Generation Technologies	WINDBIO	Wind with Biomass backup

Non-Energy Sectors:	
CROP	Crops
LIVE	Livestock
FORS	Forestry
FOOD	Food
SERV	Services
EINT	Energy Intensive Products
OTHR	Other Industries Products
TRAN	Industrial Transportation
HTRN	**Household Transportation**
PTRN	Purchased Transportation
ICE	Internal Combustion Vehicles
CNG	Compressed Natural Gas Vehicles
HYB	Conventional Hybrid Vehicles
PHEV	Plug-in Electric Vehicles
EV	Electric Vehicles

Factor Inputs:		
Capital	Gas	Cropland
Labor	Uranium	Grazing Land
Crude Oil	Wind	Managed Forests
Shale Oil	Solar	Natural Forests
Coal	Hydro	Natural Grassland

The regions in EPPA, shown in Figure 3, include eight regions represented by a single country and eight regions represented by an aggregation of countries based on regional similarities in their economies. The single-country regions are the United States (U.S.), Canada (CAN), Mexico (MEX), Japan (JPN), China (CHN), India (IND), Brazil (BRA) and the Russian Federation (RUS). The multi-country regions are Africa (AFR), the Middle East (MES), Rest of Americas (LAM), Australia, New Zealand and Oceania (ANZ), Dynamic Asia (ASI), the EU27 and Norway (EUR), Central Asia and Rest of Europe (ROE), and the Rest of East Asia (REA). Trade among these regions in commodities uses the Armington specification[6].

All economic flows between regions through trade and inputs/outputs of sectors within a region are tracked and accounted for in dollar values. At the same time, in order to project emissions from energy consumption, the use of energy in each sector of the economy is also tracked in energy units, such as Quads. Emissions from energy and other uses are tracked in physical units of metric tonnes (Waugh *et al.*, 2011). The GHG emissions projected in the model are CO_2, methane (CH_4), nitrous oxide (N_2O), sulfur hexafluoride (SF_6), pentafluorocarbons (PFCs) and hexafluorocarbons (HFCs).

[6] Armington specification assumes imperfect substitution between traded commodities from the same sector but from different regions. It allows for cross imports between different regions.

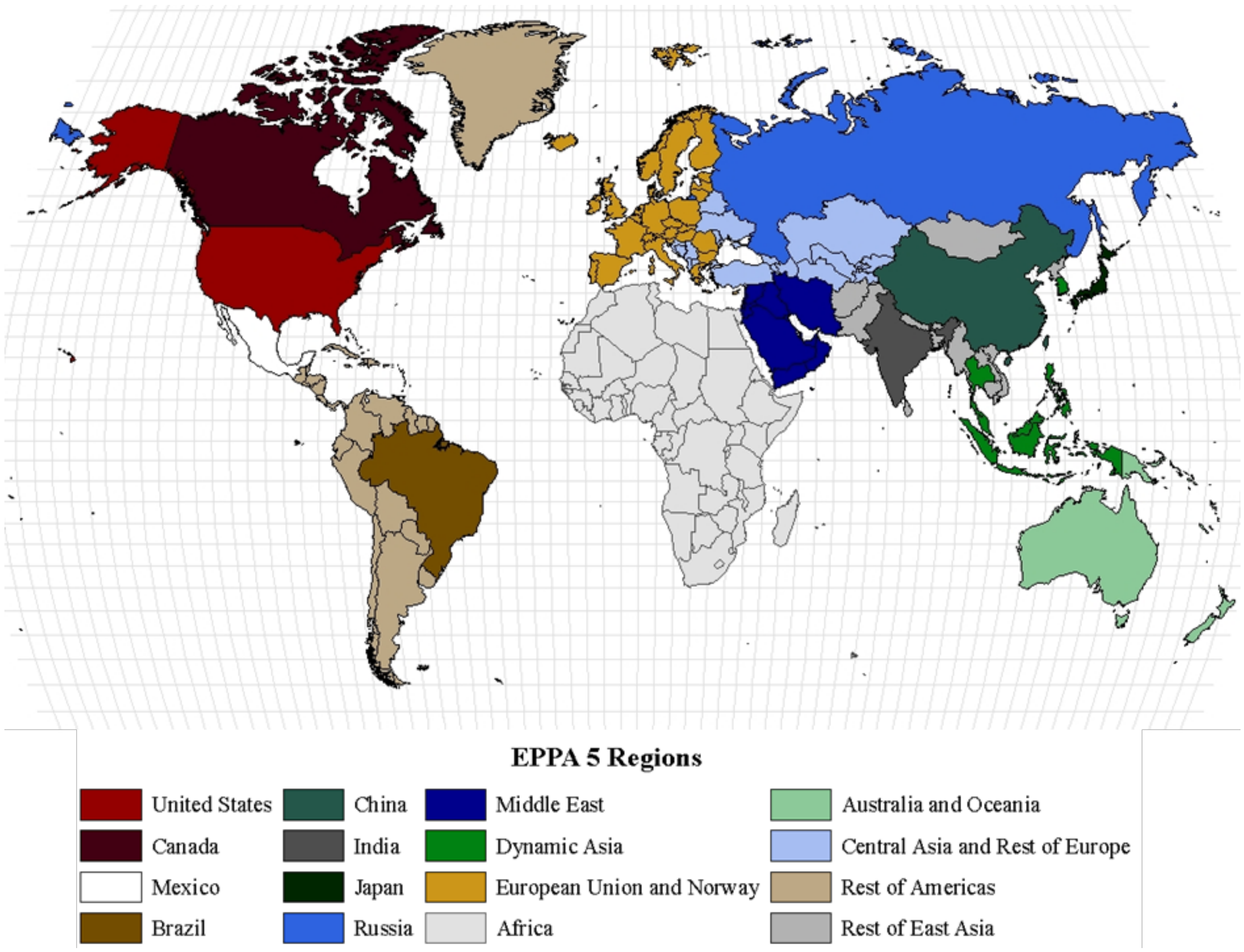

Figure 3. The 16 economic regions in the EPPA model.

Land is also tracked in physical units (hectares), and is required as an input in the three aggregated agricultural sectors: crops (CROPS), which uses cropland; livestock production (LIVE), which uses grazing land/pasture; and forestry (FORS), which uses managed forest land. Cropland is also required by biofuel crops. The model also represents three land categories that are not in use in the economic sectors mentioned above: natural forests (NFORS), natural grasslands (NGRASS), and other (OTHER). Land classified as OTHER is assumed to be unsuitable for agriculture and forestry use and is never converted, but is tracked to ensure conservation of total area in the simulations. The base year for the initial areas of each land-use are from Reilly *et al.* (2012).

Land-use conversions occur based on relative rents and conversion costs. The land available for the agricultural economic sectors can be expanded by converting either natural forests or grasslands available in each region to economic use based on conversation costs, relative land rents, and the willingness to convert natural to managed lands, as outlined by Gurgel *et al.* (2007, 2011). The land rents are deduced from the GTAP land value data (Lee *et al.*, 2009) and the Global Timber Market and Forestry Data Project (Sohngen, 2007).

The CGE structure of EPPA simulates flows between the inputs and outputs of its different sectors based on the relative prices of their inputs and outputs. The sectors are modeled using nested constant elasticity of substitution (CES) functions. The output from each sector in the model uses factor inputs and possibly the output from other sectors. If multiple sectors have a common input, then their competition for that input increases its price. When that input is the output of another sector, then due to its increased price, its production responds to the change in its price. For example, the FOOD sector uses the crops produced by the CROP sectors. If another sector, such as biofuels, increases its uses of CROP output, then the price for crops will increase. The CROP sector uses land as an input, along with livestock (LIVE) and forestry products (FORS). In response to the price increase for CROP output, the rents for cropland increase relative to other uses. This relative rent increase drives conversion of pasture land and managed forest lands to crop land, which increases rents for these land types. At the same time, the output of biofuels needs to be consumed by the transportation sector, so it competes with gasoline (aggregated into the refined oil sector, ROIL). As biofuels take some of the market share from ROIL, its price decreases. This decrease also reduces the demand for inputs by ROIL, including crude oil (OIL). Further, as more expensive biofuels are implemented using mandates, the cost of transportation fuel increases, affecting the demand for transportation in all sectors that use transportation services[7]. Thus the linkages between the sectors in the model allow us to capture the effects of changes in one sector on the whole economy. Consequently, our modeling framework assesses multiple agricultural, energy, and transportation sectors when assessing the impacts of biofuels penetration through policy mandates.

The EPPA model represents first- and second-generation biofuel production technologies, as outlined in Winchester and Reilly (2015). The first-generation technologies produce either ethanol or bio-diesel. Ethanol can be produced from grains (i.e. corn and wheat) or sugary crops (i.e., beets and sugarcane). Biodiesel can be produced from soy and palm oil. The second-generation technology produces ethanol from cellulosic biomass from grassy crops.

The biofuel sectors (Table 1) are based on their technologies and feedstock crops, which are disaggregated from the aggregate CROP sector: corn (CORNE), wheat (WHEATE), sugarcane (SUGARE), beet (BEETE) and grassy crops (BIO-OIL) for ethanol; and rapeseed (RAPESO), soy (SOYO) and palm oil (PALMO) for biodiesel. Thus, while the model tracks corn used specifically for biofuels, corn grown for other uses (i.e. food, feed and fiber) is included in the remaining aggregate CROP sector.

Biofuel crops, in a similar fashion to CROP, are produced by combining crop land and other inputs in nested CES production functions, which have the built-in ability to substitute between inputs of land, energy, materials, resources, labor and capital based on the price of land and other inputs along with the elasticities of substitution among inputs. This simulates price-induced endogenous yield changes in the bioenergy crops. At the same time, the biofuel crops are also subject to the same assumption of 1% per year exogenous yield improvements as the other economic sectors that use land. The amount of biofuel produced from its feedstock crop also responds to prices, making it possible to produce more fuel from the same amount of crop by using more energy and materials.

[7] Simulated changes in demand for transport fuels also include the so-called rebound effect.

Production costs for each biofuel pathway differ by region based on Winchester and Reilly (2015). For example, corn ethanol is the least-cost biofuel in the U.S., but sugarcane ethanol is the cheapest option in Brazil. The biofuels are then treated as equivalent to refined oil products (ROIL) on an energy equivalent basis. However, ROIL cannot be completely substituted away in the U.S. by ethanol in transportation, because a 10% blend-wall is implemented in consumption based on vehicle technology. In addition to producing fuels, some biofuel pathways also produce co-products. For example, dried distiller grains (DDG) are co-produced with corn ethanol, which are treated as a perfect substitute for crops in livestock feed.

3.2 Terrestrial Ecosystem Model (TEM)

We use the Terrestrial Ecosystem Model (TEM), a biogeochemistry model that computes monthly carbon and nitrogen dynamics of different land ecosystems (McGuire *et al.*, 2001, Felzer *et al.*, 2004), to calculate carbon emissions from land use and land-use change in our scenarios. In our work, TEM estimates are resolved to a ½° longitude × ½° latitude grid using most model inputs at this resolution. In the next subsection, we will discuss how TEM estimates for 67,420 grid cells are aggregated for input into EPPA's 16 regions and how EPPA's regional estimates are disaggregated to the grid cell level for use in TEM.

The atmospheric chemistry inputs for TEM include mean annual global atmospheric carbon dioxide concentrations and gridded monthly atmospheric ozone concentrations. The climate inputs are gridded monthly air temperature, precipitation, and solar radiation. The land inputs include gridded soil texture, elevation, and annual land cover, which may contain a mosaic of land covers within a grid cell (Schlosser *et al.*, 2007, Reilly *et al.*, 2012). The annual land-cover time-series data may be prescribed by spatially explicit data sets or generated from land-cover models. Vegetation-specific parameters are used for 35 land-cover types (Schlosser *et al.*, 2007). A schematic representation of TEM is provided in **Figure 4**.

We estimate land-use emissions by computing the net carbon exchange (NCE) between terrestrial ecosystems and the atmosphere from: 1) net ecosystem productivity (NEP), which is calculated as the difference of the uptake of atmospheric carbon by plant photosynthesis (known as gross primary productivity or GPP), and the carbon released due to both plant autotrophic respiration (R_A) and soil heterotrophic respiration (R_H) associated with the decomposition of litter and soil organic matter by microbes and fungi; 2) the carbon emissions that occur when natural land is converted to agricultural use (E_C); and 3) the carbon emissions caused by the decomposition of agricultural and forestry products (E_P). We use TEM to compute all these carbon fluxes and estimate $NCE = NEP - E_C - E_P$. NEP may be positive or negative based on how local environmental conditions influence the metabolism of the land ecosystems in the grid cell. E_C and E_P will always have positive values or a value of zero. Negative values of NCE represent positive land carbon emissions (i.e. a carbon source) and positive values of NCE represent negative land carbon emissions (i.e. a carbon sink). In Section 4, we describe our results in terms of land carbon emissions.

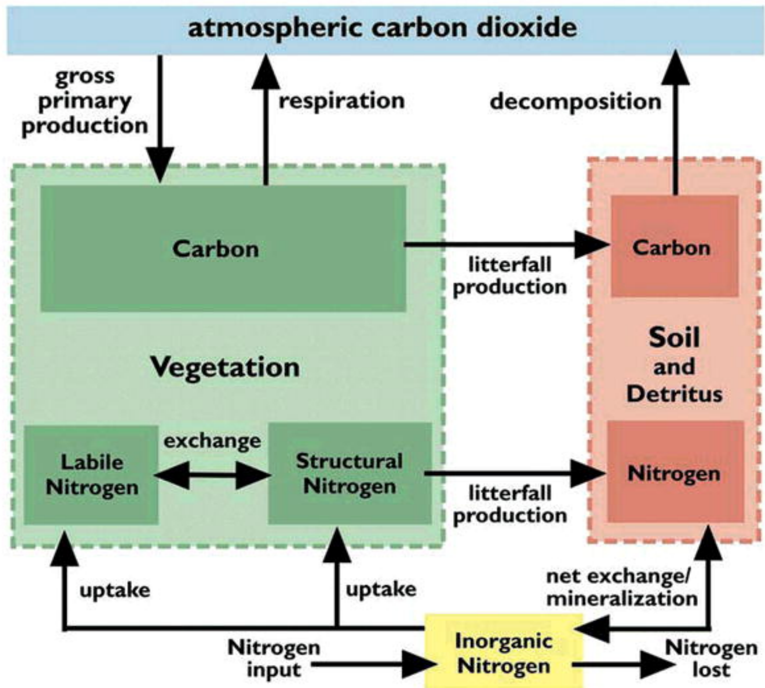

Figure 4. Terrestrial ecosystem model.

The model tracks how disturbance influences the carbon stored in vegetation and soil by keeping a record of disturbance cohorts and the time since the cohort was last disturbed (Reilly *et al.*, 2012). For example, if part of a cohort covered by temperate forest is clearcut, a new cohort is created that keeps track of the area that was cut and the carbon stocks of soil organic matter within that area, but resets vegetation carbon stocks to zero. The age of the new cohort is initially set to zero, but is updated each year afterwards from the last disturbance. Although the vegetation and soil carbon stocks of a cohort generally increase as a cohort ages, future carbon stocks do not depend explicitly on stand age, but rather depend on the local environmental conditions found in the grid cell including land-use activity and changes, climate conditions (air temperature, precipitation, solar radiation) and atmospheric chemistry (atmospheric carbon dioxide concentrations and the AOT40 ozone damage index; see Felzer *et al.*, 2004, 2005).

When forests are cleared/harvested, the carbon in vegetation biomass is released to different pools. Of the vegetation biomass carbon released, 40% is assumed to be released to the atmosphere within a year of the harvest as a result of burning slash or fuel wood as E_C; 30% is assumed to be added to the soil organic carbon pool (e.g. remaining slash, roots) where environmental conditions determine the rate of carbon emissions from decomposition; 20% (for temperate/boreal forests) or 27% (for tropical forests) is added to a 10-year wood products pool where 10% of the cohort is returned to the atmosphere each year from decomposition; and for temperate/boreal forests only, 7% is added to a 100-year wood products pool where 1% is returned each year from product decomposition. The emissions from the decomposition of the woody products contribute to the flux E_P, which also includes the release of carbon back to the atmosphere from the grid cell by the consumption and decomposition of agricultural products within a year after harvesting crops. All releases of carbon are assumed to occur in the same grid cell where harvest occurs; thus, no horizontal carbon flows are assumed to occur in the TEM simulations. The simulated release of carbon to the atmosphere from the 1-year agricultural product pool and the 10-year and 100-year wood product pools would implicitly include the release of carbon to the atmosphere from any later processing, consumption or decomposition of these products in landfills or elsewhere. Further description of how TEM handles carbon fluxes from forest and agricultural harvests is given in McGuire *et al.* (2001) and Felzer *et al.* (2004). After timber harvest, forests are allowed to regrow with the regrowth rates of trees and the decomposition rates of detritus determined by local environmental conditions in each ½° × ½° grid cell.

3.3 The Dynamically Linked Model (EPPA-TEM)

We use a dynamically linked system to connect EPPA and TEM. In this linkage, the land-use need to produce agricultural output in EPPA is adjusted to account for estimated changes in land productivity, known as net primary productivity (NPP), by TEM. NPP represents the creation of new plant biomass and is calculated as the net carbon uptake by plants—the difference of carbon taken up from the atmosphere by photosynthesis and the carbon released back to the atmosphere by plant autotrophic respiration: $NPP = GPP - R_A$. Note here that this land productivity is also related to land carbon source/sink activity as $NEP = NPP - R_H$. The gridded NPP is aggregated to the regional level used by EPPA for each land-use type. As NPP increases for a certain land-use type under more favorable environmental conditions, EPPA requires less land to

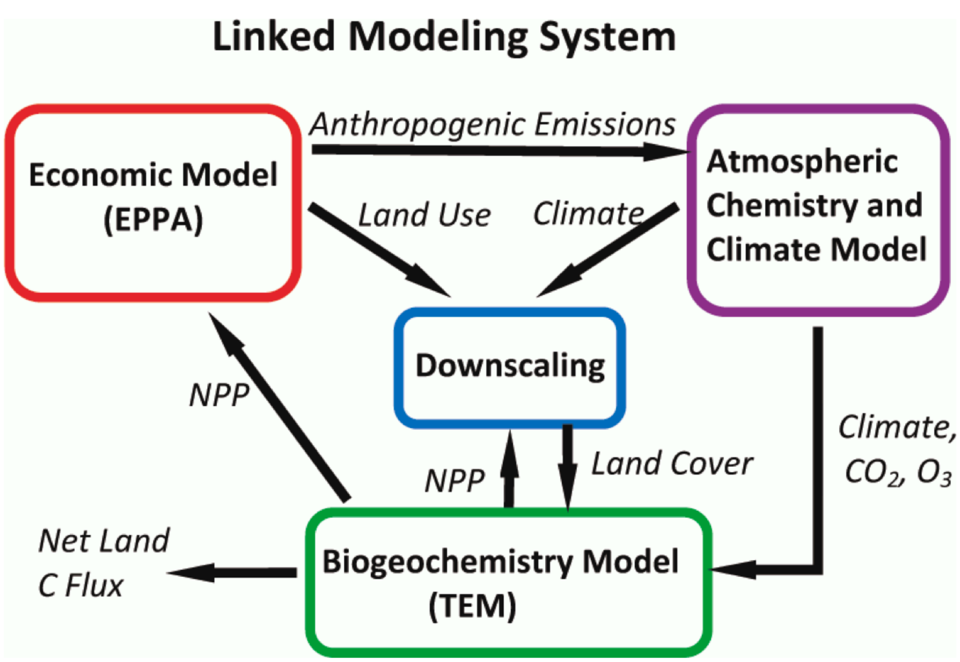

Figure 5. EPPA-TEM Schematic (Reprinted with permission from Reilly *et al.*, 2012).

produce the same output. This reduces both the land-use emissions and the land rents for that land use.

The land-use changes computed in EPPA occur at the regional level. TEM needs this information to be resolved down to the ½° × ½° grid. This is carried out in our downscaling algorithm (Wang, 2008), which uses NPP, air temperature, precipitation and distance to urban areas to allocate these transitions to the 67,420 grid cells. A schematic of this dynamic linkage is shown in **Figure 5**.

4. RESULTS AND DISCUSSION

In this section, we first review the projected land-use emissions for the U.S. and the world. Next we use the land-use projections to understand these emissions profiles, and then wrap up with economic impacts, focusing on food and oil.

4.1 Land-use Emissions for the U.S. and the World

In the reference scenario, we see a variety of patterns of emissions (**Figure 6**) related to land use and land-use change. In Figure 6, the global emissions are disaggregated to the U.S. and the Rest of the World (ROW). We find that the world has positive emissions from land use and land-use change, even though the U.S. is a sink. This sink is quite stable, decreasing only slightly over the projection time frame. The U.S. sink is 847 million tonnes (Mt) of CO_2 per year in 2010, which decreases by 9% to 772 $MtCO_2$/year in 2050. In constrast, total world land-use emissions, which are mostly influenced by land-use emissions in ROW, are 3520 $MtCO_2$/year in 2010, peaking at 6135 $MtCO_2$/year in 2030, and decreasing to 5350 $MtCO_2$/year in 2050.

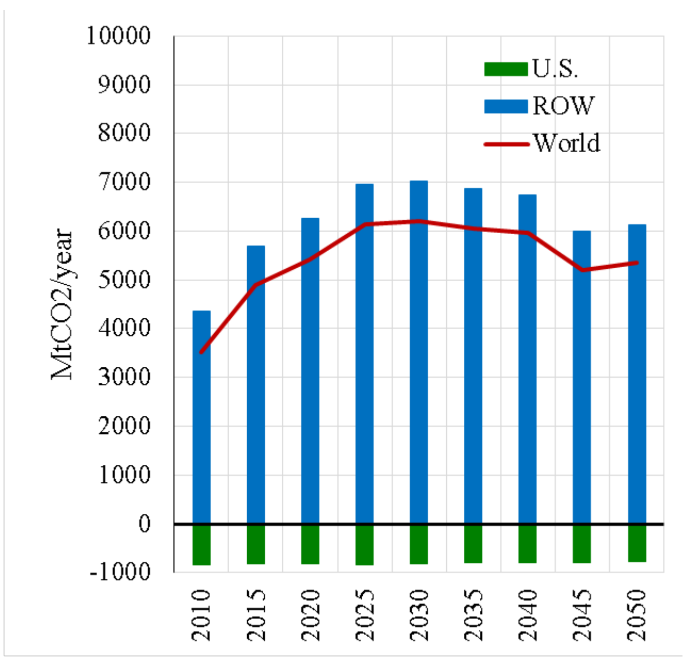

Figure 6. In the reference, global land use contributes a net source to the GHG emissions. The U.S. by itself is a sink due to its stable natural forest lands, while the Rest of the World (ROW) is a source.

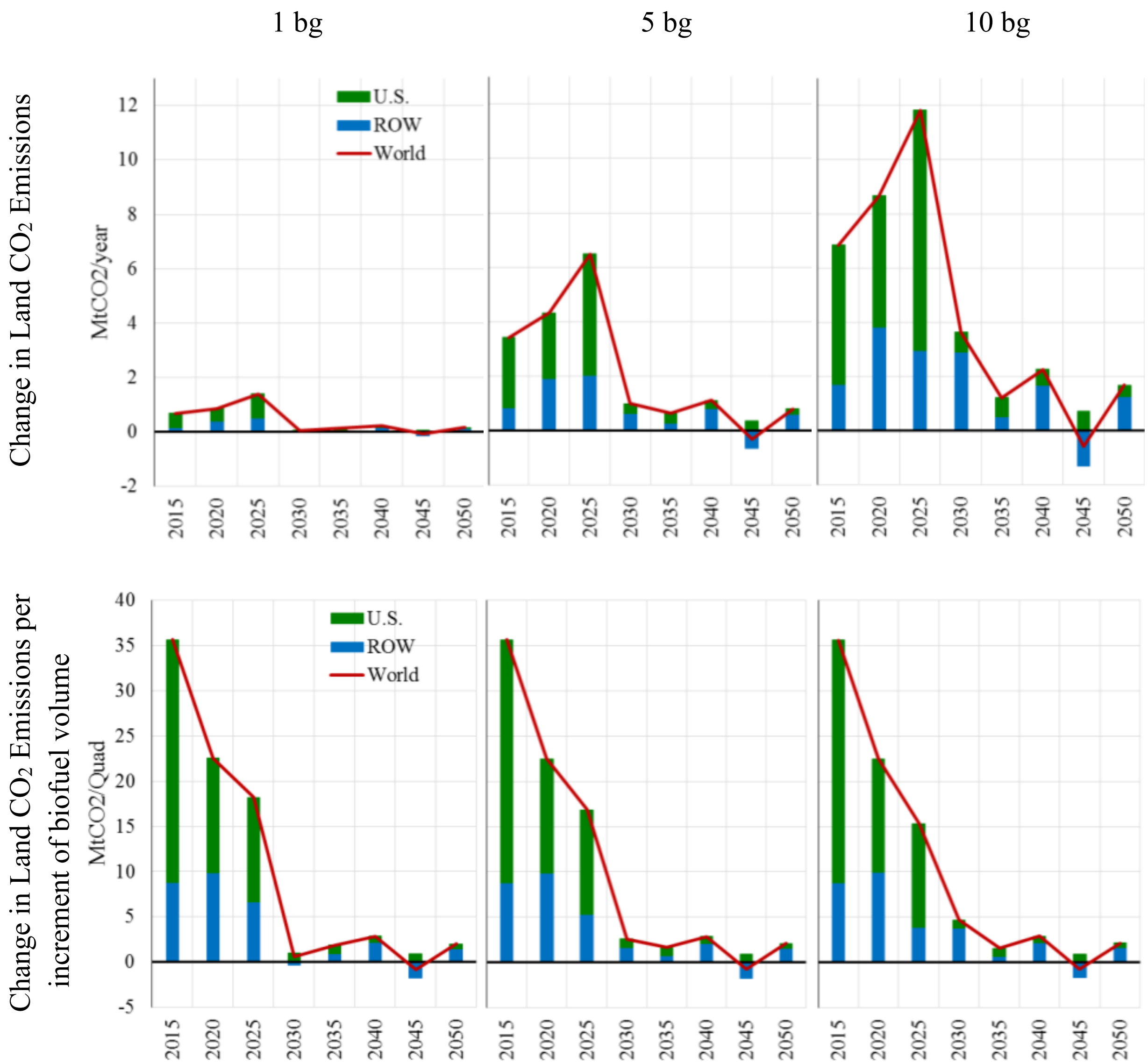

Figure 7. The top row shows the changes in global land-use emissions relative to the reference scenario. The bottom row shows the emissions in the top row divided by the added biofuels in each scenario.

The policy scenarios change the land-use emissions profile both globally and within the U.S. These changes are shown in **Figure 7**. The top row shows changes in these emissions from the reference scenario for the three policy scenarios. Here we see that at each time step, the change is larger for the policies with larger mandates. The U.S. sink decreases with the decrease in the sink peaking in 2025, which is the same year that biofuels reach the maximum size of the mandate shock. After 2025, the changes in emissions are larger outside the U.S. If we normalize the emissions by the amount of added biofuels in each policy scenario, as shown in the bottom row of Figure 7, we see a clear demonstration of the scalability of the impacts at each time step, i.e., the three scenarios produce nearly identical per unit emissions profiles. In 2010, the normalized emissions are 26.8 Mt CO_2/Quad in the U.S. which drop to 11.5 Mt CO_2/Quad in 2025. We will further explain these patterns in the next subsection.

4.2 Land-use Change Effect on Land-use Emissions for the U.S. and the World

The land-use emissions profile for the reference in Figure 6 are the result of the existing (2010) land cover and changes to it over time. To understand these land-use change emissions trajectories, we first look at the reference land-use profile for the U.S. and the ROW shown in **Figure 8**. The negative U.S. land-use emissons are a consequence of it having a large and stable forest mass. We divide forest land into managed and natural parts. The managed forests are the forest areas that are used to produce forestry products, such as timber, paper pulp, etc, and tend to have a lower density of carbon stocks than natural forests due to the removal of carbon from previous disturbances such as timber harvests (Lu *et al.*, 2015). Natural forests are assumed to be undisturbed and can only produce forest products after they are first converted to managed forests with timber harvests or have been converted to agricultural land. Current land management practices and regulations are assumed to protect the area of natural forests and its carbon stocks in the U.S., and such protections are expected to continue in the future. The potential impact of natural disturbances (wildfires, insect infestations, wind and ice damage) on the carbon dynamics of natural forests are problematic (Zhang *et al.* 2012, 2015) and have not been considered in our simulations. EPPA is calibrated to reflect this inertia in the system, which is seen as the natural forests staying untouched in our projection. Instead, the expansion of croplands and pastures in the U.S. come at the expense of natural grasslands and managed forests, which have a lower carbon density than natural forests. In addition, regrowth of trees in managed forests tend to enhance carbon sequestration (Lu *et al.*, 2015). This maintains the U.S. sink in the reference.

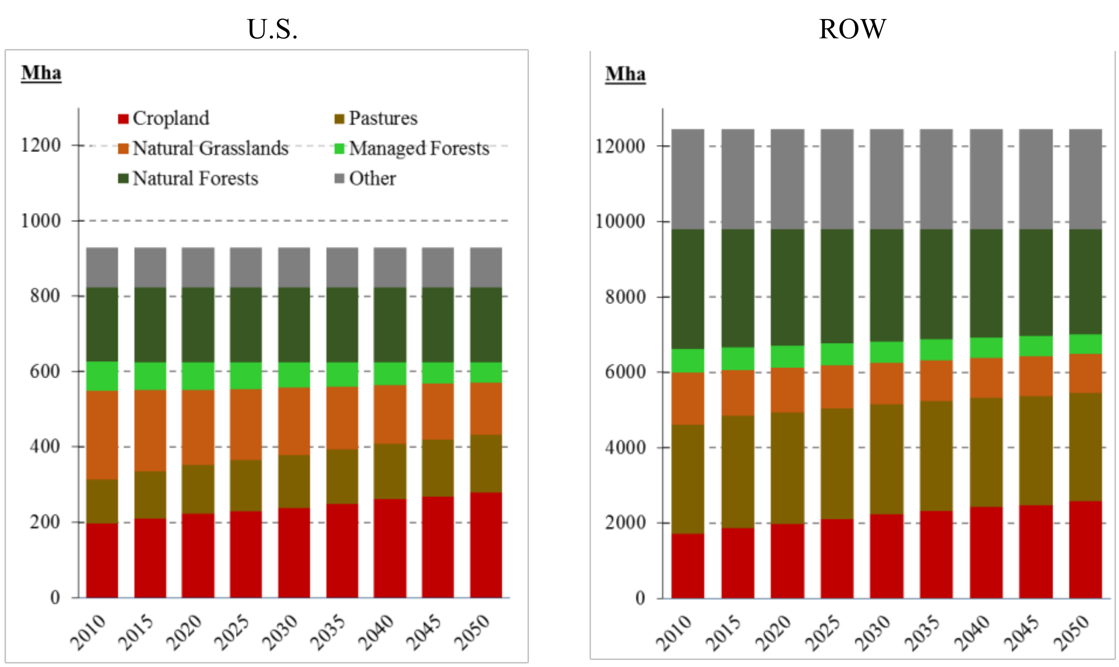

Figure 8. Land use profile for the U.S. and the ROW for agricultural and natural land-uses in the reference. Note the difference in scales.

Cropland in the U.S., including that for biofuel crops, increases from 197 million hectares (Mha) in 2010 to 280 Mha in 2050 (Figure 8). Our results show that this is driven almost entirely by demand for crops in the non-biofuel producing sectors (e.g. food, livestock). The biofuel volumes shown in Figure 1 require 12 Mha of cropland in 2010, reaching a maximum in 2015 of 18 Mha and falling to 11.5 Mha by 2050. The 50% increase between 2010 and 2015 is driven by the initial ramp-up of biodiesel, which causes biofuels to increase from 1.04 Quads to 1.28 Quads during this period. The decrease in area of cropland used to grow biofuels after 2015 is driven by our assumed exogenous 1% per year improvement in crop yield (see Section 3.1), and is more than enough to compensate for the continuing increases in the mandated biofuel volumes. Additionally, the economy becomes more efficient over time through improvement in labor productivity. If we take away these efficiency improvements, the land used for cultivating biofuel crops would be approximately proportional to the profile of the biofuel production.

In concert with cropland, the area of pastures used for livestock in the U.S. increases from 118 Mha in 2010 to 153 Mha in 2050. In Figure 8, we can see that the increases in livestock pastures and cropland come from natural grasslands and managed forests. By 2050, the natural grasslands shrink by 69% and managed forests by 43% relative to 2010.

The increased corn ethanol mandates in the policy scenarios (shown in Figure 2) increase total cropland in the U.S. This can be seen in the first column of **Figure 9**. This increase comes from pastures, managed forests and natural grasslands. The changes are the smallest in the *1bg* scenario (see **Figure 9a**) and increase with the size of the mandate (see **Figure 9d** and **g**). In all policy scenarios, the increase in cropland peaks in 2025, the same year that the policy reaches its maximum. After 2025, the increase in cropland relative to the reference becomes less over time due to yield and efficiency improvements discussed earlier in this section and due to the continuing expansion of croplands in the reference scenario after 2025 (Figure 8). The difference in emissions is greatly reduced after 2025 because croplands are no longer expanding rapidly to meet the biofuel mandates, and the carbon fluxes from converting lands to agriculture (E_C) associated with this expansion of croplands for the additional biofuels are greatly reduced. However, the additional land-use emissions associated with the mandates after 2025 are still being influenced by decomposition of the 10-year and 100-year woody products created during the expansion of croplands for biofuels before 2025.

The land-use changes outside the U.S. in the policy scenarios are completely driven by international trade in food crops and livestock because in our scenario designs we do not have additional biofuels mandates outside the U.S., nor do we allow direct trade in biofuels. As a result of this restriction, the global land-use changes—shown in the third column of Figure 9—are dominated by the land-use changes in the U.S. We find that the size of the global land-use change areal footprint scales in direct proportion to the additional corn ethanol volumes in the policy scenarios. Further, the ratio of cumulative land-use emissions and the cumulative net areal footprints for ROW and U.S. also scale in direct proportion to the size of the U.S. policy mandates. In the U.S., this ratio is 0.9 tCO_2/ha in all our policy scenarios, while in ROW, its value is

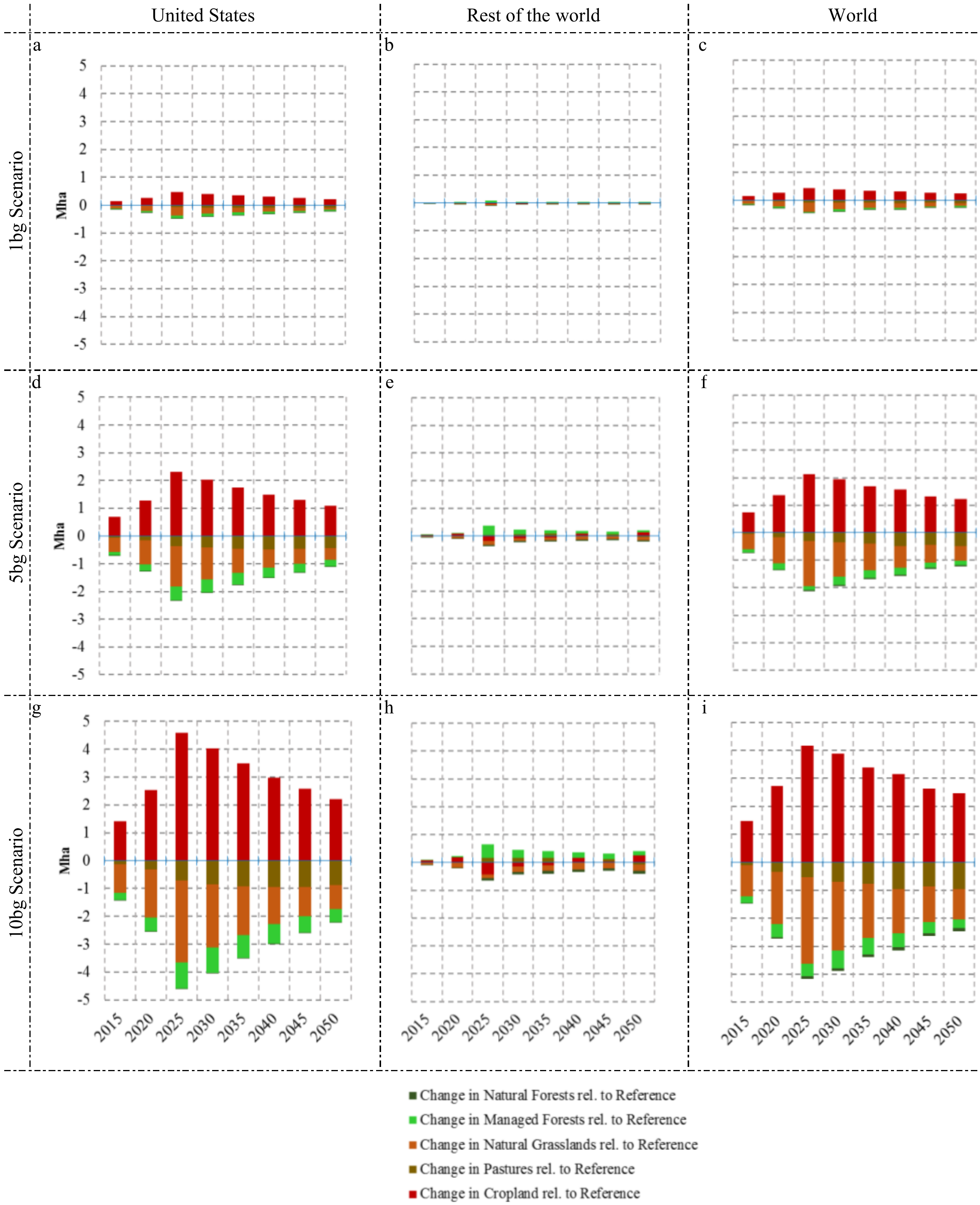

Figure 9. Changes in land use of the 3 policy scenarios relative to the reference scenario for the U.S., the Rest of the World, and the global total. The policies are along the rows and the regions are along the columns.

4.0 tCO_2/ha, 4.3 tCO_2/ha and 4.7 tCO_2/ha in the *1bg*, *5bg* and *10bg* scenarios, respectively. The fact that this metric in each region is almost the same across all our policy scenarios provides another demonstration of the scalability of the impacts of biofuels in our analysis.

The land-use emissions in the rest of the world (ROW) are explained by the land-use changes shown in the second column of Figure 9. The first thing to note is that these land-use changes are much smaller compared to the U.S. than its share in the land-use change emissions presented in Figure 7. Closer examination of these changes shows that ROW converts natural forests to other uses, accounting for the higher emissions. In 2015 and 2020, ROW increases its cropland relative to the reference, presumably to compensate for more crops being diverted to corn ethanol production in the U.S. However, in 2025, the loss of managed forests in the U.S. forces their increase abroad.

The dominant share of land-use changes for ROW occur in AFR, BRA and LAM (regions are defined in Section 3.1). In the reference, AFR increases only its cropland in our projection. This increase occurs at the expense of all the other four uses (pastures, managed forests, natural grasslands and natural forests). In BRA and LAM, both cropland and pastures grow over time by reducing natural grasslands and forests, both managed and natural. However, the response to the policy scenarios in each region varies: AFR and LAM increase their cropland and managed forests while reducing the size of pastures and natural lands; BRA increases its pastures and managed forests at the expense of cropland and natural grasslands and keeps the natural forests unchanged relative to the reference.

Corn for ethanol cannibalizes cropland from other crops in the U.S. in our policy scenarios. This result illustrates the relationships among all sectors of the economy and price-induced yield changes. While total cropland increases relative to the reference, the increase in total cropland is less than the increase in land used for cultivating corn dedicated to ethanol production. For example, in the *10bg* scenario, total cropland increases by 4,592 kilo-hectares (Kha) in 2025 relative to the reference case. This is less than the increase in land used to grow corn for ethanol for the same period (6,692 Kha). The difference is accounted for by the decrease in land used by non-biofuel crops (2,078.4 Kha), soy to produce biodiesel (21.1 Kha), and biomass to produce cellulosic ethanol (0.5 Kha).[8] The U.S. mandates also influence the relative amount of land used for biofuel production in other countries. For example, as production of Brazilian crops for consumption outside of the biofuel sector increases in response to the U.S. reduction due to corn ethanol, the land available for sugarcane crop dedicated to ethanol production in Brazil decreases, reducing the production of sugarcane ethanol. For example, in 2025 in the *10bg* scenarios, the land for sugarcane crop dedicated to ethanol production is reduced by 1,135 Kha to 4,004 Kha relative to the reference value of 5,139 Kha. This reduction is caused by both the loss of total cropland by 0.8 Mha (due to expansion in pastures and managed forests) and the expansion of non-biofuel cropland by 0.3 Mha (to increase its crop production for export to the U.S.). The changes in the other scenarios follow similar patterns and scale with the size of the mandates.

[8] Although land used for crops to produce biodiesel and cellulosic ethanol decreases, the mandates for these fuels continue to be met in the model due to price-induced yield and conversion efficiency improvements.

4.3 Impacts on Energy and its Emissions for the U.S. and the World

The size of biofuel impacts on energy increases with the size of the mandates in our policy scenarios. Mandating more biofuel production reduces U.S. domestic refined oil production and its net imports of both crude oil and refined oil products. While the consumer price for transportation fuel increases, the producer price for refined oil products as a whole decreases. Our results show that the reduction in energy units of refined oil consumption is larger than the biofuels added. For example, in the *10bg* scenario in 2025, 0.77 Quads of biofuels are produced, while the reduction in domestic refined oil consumption is 1.1 Quads, resulting in an overall reduction in consumption of transportation fuels of 0.33 Quads. This reduction in total refined oil uses reflects a demand response due to an increase in the average price of refined oil products.

The domestic production of refined oil does not go down by 1.1 Quads, which is the reduction in refined oil consumption. This is because refined oil producers are able to capitalize on the existing crude oil export ban in the U.S. They increase their net exports of refined products by 0.16 Quads. This is due to the combination of a lower input cost of crude oil price into the refined oil prodution (that makes additional exports of the refined oil products profitable) and inability to export crude oil from the U.S.

In ROW, the consumption of refined oil increases with increased U.S. exports. However, this increase is less than the reduction in U.S. consumption, resulting in overall global reduction. In particular, in 2025 for the *10bg* scenario, refined oil consumption outside the U.S. increases by 0.4 Quads from 183.8 Quads in the reference. Thus the domestic reduction of 1.1 Quads is blunted by the ROW to 0.7 Quads globally. We find the same qualitative results in the other years and for the other policies.

In the reference case, U.S. energy emissions from all fossil fuels are 5.758 Mt CO_2-equivalent[9] (CO_2-eq) in 2010, and decrease to 5,571 Mt CO_2-eq in 2020, after which they rise steadily from 5,690 Mt CO_2-eq in 2025 to 6,613 Mt CO_2-eq in 2050. In the ROW, emissions from fossil fuels increase steadily from 26,310 Mt CO_2-eq in 2010 to 45,310 Mt CO_2-eq in 2050. These changes in emissions from fossil fuels in the policy scenarios relative to the reference reflect changes in the refined oil sector, e.g., for the *10bg* scenario, we find that these emissions are reduced in the U.S. by 98 Mt CO_2-eq in 2025 and 107 Mt CO_2-eq in 2050, while in ROW, these emissions increase by 24 Mt CO_2-eq in 2050, leading to a global reduction of 83 Mt CO_2-eq in 2050.

The land-use emissions presented in Section 4.1 are the carbon emissions calculated using TEM and do not include N_2O emissions from application of fertilizer. In the GHG accounting in EPPA, N_2O emissions related to agricultural uses are tracked and vary with agriculture output. The emissions coefficients are taken from Waugh *et al.* (2011). As corn is heavily fertilized, further refinement of our results based on more detailed representation of N_2O fluxes associated with fertilizer use in TEM might modify our current results.

[9] Non-CO_2 GHGs are converted to CO_2-equivalence by using their 100-year global warming potential (GWP) used in IPCC AR4.

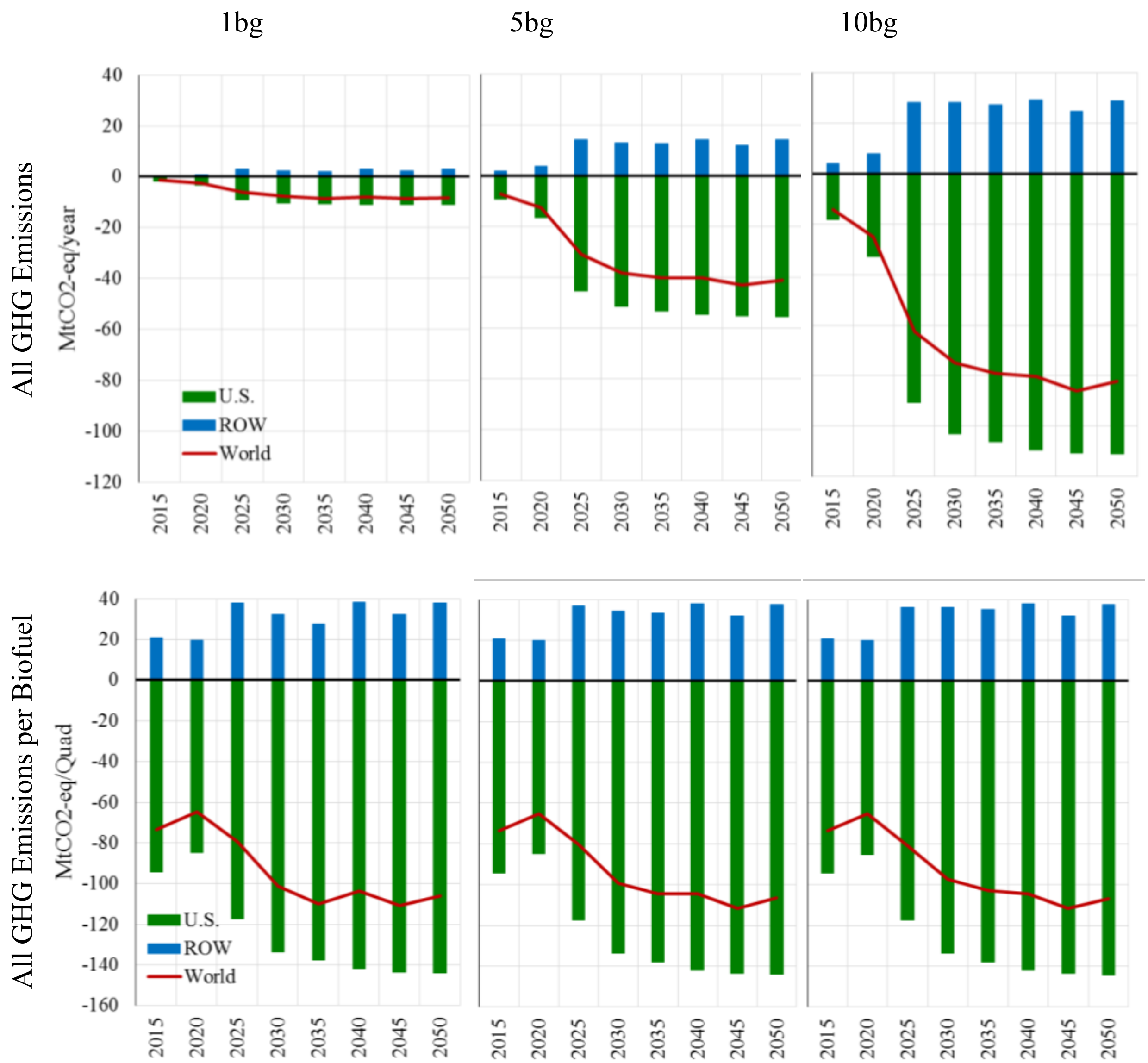

Figure 10. Comparison of the change in GHG emissions (top row) and change in GHG emissions per added volume of biofuels (bottom row) in the 3 scenarios from the reference scenario for the U.S., ROW and the World.

The total GHG emissions reductions in the policy scenarios are presented in the top row of **Figure 10**. These emissions include CO_2 from land use calculated by TEM (Figures 6 and 7), and GHG emissions, including N_2O, from economic activity estimated by EPPA. The total reduction in GHG emissions increase with the size of the shock. The bottom row of Figure 10 demonstrates the scalability of the changes in total GHG emission per increment of biofuels. In the U.S., these emissions range from -85 to -144 Mt CO_2-eq/Quad.

In the above discussion of changes in GHG emissions from fossil fuel consumption and total GHG emissions in Figure 10, we see that even though emissions increase in ROW, the reduction in U.S. emissions is much larger than that increase, leading to emissions reductions for the whole world (U.S. and ROW combined). On the other hand, this also suggests that by considering potential leakages, we get lower global reductions in emissions from the dynamic, linked economic system in the model than we would by considering only the emissions reductions from the introduction of biofuels in the U.S.

4.4 Impacts on Agriculture for the U.S. and the World

Crops, livestock and forestry products are affected by the increased biofuels mandates. Except those used to produce biofuels, all crops are included in an aggregated sector, and we can only report the monetary value of its production, consumption, imports and exports. In our reference, the crop commodities sector in the U.S. (including biofuels) grows from $190 billion in 2010 to $500 billion in 2050. Also in the reference case, U.S. net exports of crops increase from $22 billion in 2010 to $52.75 billion in 2050. The production of crops for biofuels is valued at $3.3 billion in 2010, increasing rapidly in 2015 to $4.5 billion due to the large increase in biodiesel production, and then rising steadily to $4.7 billion by 2050. The production of corn itself for biofuels is valued at $3.1 billion (4.7 billion bushels[10] of corn) in 2010, increasing in 2015 to $3.3 billion (5.2 billion bushels) and then rising steadily to $3.7 billion (5.5 billion bushels) by 2050. The crops used as livestock feed, which we track as intermediate consumption of crops in the livestock sector, is $12.5 billion in 2010 and increases to $40.7 billion in 2050. This increase in feed is reflected in the livestock production, which grows from $129 billion in 2010 to $368 billion in 2050.

In the policy scenarios, the U.S. increases its production and consumption of crops, while reducing the size of its exports. The consumption of crops to produce biofuels increases by 2050 to $5.1 billion, $6.1 billion and $7.25 billion for the *1bg*, *5bg* and *10bg* scenarios, respectively. In physical units, this translates to an increase of 0.35, 1.79 and 3.58 billion bushels in 2050 for *1bg*, *5bg* and *10bg* respectively relative to the reference consumption of 5.51 billion bushels. These increases are in proportion to the increase in biofuels production measured in energy terms (Quads). As these increases are modest, their economic impacts are also modest. Because of the increase in demand for biofuel crops, there is less feed available for livestock production. This aligns with the land-use results where the pastures used for livestock are reduced in the U.S. For the livestock sector, this means a reduction in its size as well. Thus, we find that the land-use results presented in Section 4.2 are mirrored in the agriculture sectors.

5. CONCLUSIONS

Biofuels compete for inputs with agriculture. Due to the inter-linked nature of agricultural markets, the effect of U.S. biofuels policies within a single region (i.e. the U.S.) may have different consequences in different regions of the world. An open question is how biofuel production in one region effects land-use change in other regions when these linkages are accounted for and whether the relationship changes for different levels of biofuel production. We use our global dynamically linked framework of economic and biogeochemistry models to

[10] We assume that 31 lbs of shelled corn produces 1 gasoline equivalent gallon (GEG) (see Winchester and Reilly, 2015), and that 1 GEG = 1.5 ethanol-equivalent gallon. Also, 1 bushel of corn = 56 lbs.

analyze how land use and associated emissions vary with regional policies in the U.S. using different magnitudes of biofuel production. Previous studies have shown that large increases in biofuel production produce non-linearities in the emissions. But to our knowledge, there is no study that systematically increases the volume of biofuels among scenarios in one particular region. Our analysis explores the question of scalability.

The scenarios presented in this paper were designed to test how the relative magnitude of land-use emissions may respond to increased biofuel mandates in the U.S. We find that with increased amounts of biofuels, the land-use changes increase proportionately. In other words, the *5bg* biofuels profile, which is five times the size of the *1bg* scenario's biofuel profile, requires five times more area of land-use change to occur than the *1bg* mandate. Similarly, the *10bg* scenario, which has double the corn ethanol volume than the *5bg* scenario, requires twice as much area of land-use change when compared to the *5bg* scenario. This scalability of the land-use changes between the scenarios is then reflected in the relative sizes of the land-use emissions—i.e., the *10bg* scenario's emissions are twice as large as those of the *5bg* scenario, which in turn are five times those of the *1bg* scenario.

While the scalability of the results holds among the scenarios, the dynamics of land use and associated emissions within each scenario are not linear. The initial ramp-up period for corn ethanol from 2015 to 2025 produces the largest response. Afterwards, as the added volume of ethanol is held fixed, the land needed to grow the corn to produce ethanol decreases. This is the result of improvements in land productivity and more efficient manufacturing of biofuels.

With the implementation of the U.S.-only policy mandates, we also find that the U.S. has the largest share of changes in land use and land-use emissions. The land-use changes affect international trade of agricultural commodities among regions, with the U.S. reducing its exports of crops. Other regions, primarily Africa and South America, increase their croplands. These increases are achieved at the expense of carbon-rich forests. Consequently, the land-use emissions response relative to the areal footprint of land-use change—i.e. the ratio of change in land-use emissions to total land area impacted by the policy—is larger in the rest of the world than in the U.S. alone (relative to the reference scenario). One way we see this is from the ratio of cumulative land-use emissions and the cumulative net areal foot print for the rest of the world and the U.S. In the rest of the world, this value is (4.3 ± 0.4) tCO_2/ha while in the U.S. it is 0.9 tCO_2/ha in all of our policy scenarios. The fact that this metric in each region is almost the same across all our policy scenarios provides another demonstration of the scalability of the impacts of biofuels in our analysis.

The added corn ethanol production time profiles for the U.S. are by design piece-wise smooth: an initial linear ramp up period till 2025, and then the level is held constant out to 2050. These profiles provide us with an insight into both the initial shock and then the reaction of the global economic system as the shock is absorbed. The response to the policies is different in the two periods. We first see that the area of land-use changes and the associated emissions increase during the ramp up. But once the policy levels are reached in 2025 and then maintained, the effects dissipate over time. Population growth over time increases the demand for land for crop cultivation, livestock grazing and forestry products. However, due to efficiency improvements, the amount of inputs required for the same production level decreases. Thus, the response to the policy mandates is non-linear in time and depends on whether the impact of higher corn ethanol volume or the interplay between population growth and efficiency improvements dominates.

The understanding of impacts of bioenergy requires an explicit evaluation of both economic and biophysical implications. It calls for additional research that integrates these two aspects. More detailed representation of these processes would provide additional insights into potential unintended consequences of bioenergy expansion.

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