THE GREENHOUSE GAS EMISSIONS CONSEQUENCES OF THE EXPANDED RENEWABLE FUEL STANDARD

A Thesis

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ABSTRACT

This thesis uses a computable general equilibrium model to evaluate the greenhouse gas emissions consequences of the United States' Renewable Fuel Standard (RFS) mandate for corn based ethanol. The RFS can reduce emissions if the potential emissions savings, that result as ethanol displaces gasoline, are greater than the resulting carbon leakage, which occurs because markets and sectors adjust to the increased consumption of ethanol or the decreased US consumption of gasoline. The general equilibrium framework allows for the impact of the expanded RFS on behavior in each of the major US sectors (transportation, fuel production and agriculture) and international markets (crude oil and land) to be measured and mapped to greenhouse gas emissions.

Estimating emissions in the general equilibrium framework differs from previous studies that estimate the carbon leakage from ethanol consumption. In contrast to studies that rely on world agricultural models (Searchinger et al. (2008); Tokgoz et al. (2008)), this framework is able to capture the impact of the Renewable Fuel Standard (RFS) on the domestic market for blended fuel, and the international crude oil market. Compared to lifecycle analysis (Farrell et al. (2006); Hill et al. (2006)), which links sectors using fixed behavioral relationships that are based on the flow of material and energy, the general equilibrium framework is able to incorporate both behavioral adjustments and emissions from sectors that do not necessarily fall in the standard lifecycle boundaries.

We find that the Renewable Fuel Standard mandate directly reduces gasoline consumption by 0.6% and 2.5% relative to baseline levels in 2009 and 2015 respectively. These reductions lead to potential emissions savings of 6.5 $TgCO_2e$ (roughly 0.1% of total US emissions) in 2009 and 26.32 $TgCO_2e$ (0.5% of total US

emissions) in 2015. These savings are very small because we project the baseline consumption of ethanol to increase to levels close to mandated levels had the RFS not been implemented. We also find that these potential savings are totally offset by carbon leakage in other markets. In 2009, approximately 50% of the potential emissions savings is offset by domestic leakage. The main sources of leakage domestically are the expansion of ethanol production, which offsets 21% of potential savings, and the expansion and intensification of the agricultural sector, which offsets 22% of potential savings. A smaller leakage, 6% of potential savings, occurs in the domestic fuel market because the price of blended fuel falls in response to the RFS. In 2015, the domestic leakages are smaller and offset 40% of potential emissions savings. The main sources of leakage are again fuel production and domestic agriculture, which offset 21% and 22% of potential emissions savings. However, we find that the price of blended fuel increases in response to the mandate leading to addition emissions savings of 1% in domestic fuel markets. Internationally, the magnitudes of the leakages are substantially larger. As the RFS depresses the world price of crude oil, the consumption of crude oil increases and results in a leakage that offsets 168% of potential emissions savings in 2009 and 257% of the potential emissions savings in 2015. In addition, the RFS reduces US crop exports and causes a small expansion of world agricultural production on to previously uncultivated land. The carbon emitted as a result of this expansion is at least 1800% greater than the potential emissions savings of the Renewable Fuel Standard for each year between 2009 and 2015.

We also find that our estimation of the emissions consequences of the RFS deviate from the results of standard lifecycle analysis (LCA) because of contrasting behavioral assumptions. Compared to our general equilibrium analysis, standard LCA methods estimate similar reductions in transportation emissions and similar increases in fuel production emissions. Incorporating behavioral adjustments in these two sectors leads to projected emissions savings that are 15% below LCA estimates in 2009 and 1% higher than LCA estimates in 2015. We also find that compared to our general equilibrium analysis, LCA overestimates the increase in corn production due to the RFS, which leads to an underestimate of total emissions savings of close to 25%. However, by not incorporating the expansion of cropland on to CRP, LCA overestimates emissions savings by a similar percentage (approximately 27%). Finally, we find that the assumption in most LCA analyses, that the use of ethanol has no impact on international crude oil or land markets, will cause LCA to overestimate emissions savings relative to the general equilibrium estimates by more than 285% and 3000% respectively.

BIOGRAPHICAL SKETCH

Richard Klotz was born on December 26, 1982. He was raised in Homer, New York and attended Homer Central High School. He attended Hobart and William Smith Colleges in Geneva, New York between 2001 and 2005 and graduated Magna Cum Laude with a major in Economics and minors in Environmental Studies and History. After graduation, he worked as a research assistant to Dr. Thomas Drennen, in Hobart and William Smith's Department of Economics, developing dynamic simulation models that integrated various energy policy decisions with economic and environmental consequences. In 2007, he enrolled in the Department of Applied Economics and Management at Cornell University and completed the requirements for a Master of Science in the field of Environmental and Resource Economics in the fall of 2009.

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Some colleagues deserve mention. First, Joel Landry for the countless hours he has spend writing, debugging and understanding the more than 20,000 lines of code that make up the economic model and for discussing many of the emissions calculations. Also, Nirav Patel for patiently helping me learn about the interactions between agriculture, soils, and greenhouse gas emissions during many cold days in the lobby of Mann Library.

Finally, I would like to thank my family for their continuous support, and Christine for putting up with me.

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LIST OF ABBREVIATIONS

AEO	EIA's Annual Energy Outlook
ARMS	USDA's Agricultural Resource Management Survey
ASD	USDA's Agricultural Statistics Database
CC	Continuous Corn
CCR	USDA's Commodities Cost and Returns dataset
CO_2e	Carbon Dioxide Equivalent
CS	Corn-Soybeans Rotation
CSW	Corn-Soybeans-Wheat Rotation
CT	Conventional Tillage
CTIC	Conservation Tillage Information Center
CRP	Conservation Reserve Program
EIA	US DOE's Energy Information Agency
EISA	2007 Energy Information and Security Act
ERS	USDA's Economic Research Service
FAPRI	Food and Agricultural Policy Research Institute
FAS	USDA's Foreign Agricultural Service
FASOM	Forestry and Agricultural Sectors Optimization Model
FSA	USDA's Farm Service Agency
GREET	Greenhouse Gases, Regulated Emissions in Transportation Model
GTAP	Global Trade Analysis Project
ha	Hectare
HH	Continuous Hay
IEO	EIA's International Energy Outlook
IPCC	Intergovernmental Panel on Climate Change

l	Liter
LCA	Lifecycle Analysis
LPG	Liquid Petroleum Gas
LUC	Land Use Change
MJ	Megajoules
mmt	Million Metric Tons
mt	Metric Tons
MT	Mulch Tillage
NASS	USDA's National Agricultural Statistics Service
NG	Natural Gas
NT	No-till
REAP	USDA's Regional Environmental and Agricultural Programming Model
RFS	Renewable Fuel Standard
RT	Reduced Tillage
SOC	Soil Organic Carbon
SS	Continuous Soybeans
Tg	Teragram (equivalent to 1e+9 kg or 1million metric tons)
US DOE	United States' Department of Energy
US EPA	United States' Environmental Protection Agency
USDA	United States' Department of Agriculture
VEETC	Volumetric Ethanol Excise Tax Credit
VMT	Vehicle Miles (kilometers) Travelled
WW	Continuous Wheat

Chapter 1. Introduction

There is irrefutable evidence that human activities are causing climate change. The current atmospheric concentrations of the main greenhouse gases, carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), are significantly higher than pre-industrial levels, and the IPCC now states that it is 'very likely' that the observed increase in temperature is a direct result of anthropogenic greenhouse gas emissions (IPCC 2007a). While there is a general consensus that global warming is occurring, the potential economic, environmental and social impacts are still being debated. Many expect the effects to be drastic. Stern (2007) estimates that if no action is taken on climate change, the resulting damages and risks will be equivalent to a reduction of 5% of world per capita income. If non-market impacts, a more responsive environmental system and distributional impacts are included, Stern estimates that the damages could rise to as much as 20% of average per capita consumption. The IPCC (2007b) states that global warming will increase the extent of areas suffering draught, increase the risk of species extinction and increase the prevalence of both malnutrition and infectious diseases. In addition, Stern (2007) and the IPCC (2007b) also find that global warming will have a disproportionate impact on the world's poor as agricultural productivity is expected to fall in low latitudes.

The Problem of Carbon Leakage

Policy makers have responded to both the dire outlooks of a warmer climate and public sentiment by implementing policies intended to reduce greenhouse gas emissions. A particular issue with greenhouse gas regulation is the global nature of the problem. Specifically, emissions from any location cause equivalent damage to the global climate, suggesting a global system is necessary to effectively limit emissions (Stavins 1997). Currently, as attempts to construct a binding global agreement have been unsuccessful, most abatement policies are implemented at the national or regional level.[1] These efforts have focused on the two main sources of greenhouse gases, fossil fuel combustion and land use change.[2] Policies that limit the use of fossil fuels generally attempt to either increase the price of fossil fuels relative to a carbon neutral alternative (carbon taxes, cap and trade systems) or support the use of more efficient or renewable technologies (subsidies, fuel economy standards, mandates). Strategies to limit land use change are commonly agreements in which one party 'earns' emissions credits by supporting the mitigation projects of other parties.[3] Essentially, developed countries are able to continue using fossil fuels in exchange for supporting improved forest management and conservation efforts in less developed countries. However, unilateral or joint emissions restrictions can result in increased emissions in unregulated countries, an effect that is referred to as carbon 'leakage'.[4]

The study of carbon leakage from fossil fuel combustion stems from the economic analysis of the proposed emissions trading scheme between industrialized countries of the Kyoto Protocol (Babiker (2005); Felder and Rutherford (1993)) and has been extended to other abatement measures.[5] A common conclusion of these studies was that between 5-20% of the emissions reductions in the constrained countries were offset by increased emissions in the unconstrained countries (IPCC 2001a). This literature suggests two main sources of carbon leakage: the terms-of-trade or producer relocation effect and fuel-market effect. The terms-of-trade leakage

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1 The primary example of a regional agreement is the European Union's Emissions Trading Scheme.
2 In 2004, the largest global source of greenhouse gas emissions, accounting for 57% of total emissions, was the CO_2 released from fossil fuel combustion. The second largest source, 17% of total, was CO_2 emissions from deforestation and other land use change, while N_2O released from agricultural soil management contributed 8% of total anthropogenic emissions (IPCC 2007a).
3 Examples of this type of transaction are the Clean Development and Joint Implementation mechanisms of the Kyoto Protocol.
4 In other contexts, this effect is also known as 'slippage', the 'rebound effect' and 'crowding'.
5 Barker (1999), Barker et al. (2007), Burniaux and Martins (2000), Jacoby and Reiner (1997), Bohringer and Rutherford (2002), Corrado and van der Werf (2008) and Manne and Rutherford (1994).

occurs as the emissions restrictions increase the production costs of emissions-intensive goods in the constrained countries, shifting comparative advantage to unconstrained countries. The result is increased production of these goods, and emissions in unconstrained countries. The fuel-market leakage occurs when emissions restrictions lower the demand for fossil fuels in constrained countries, lowering the world price of these fuels which leads to increased consumption in unconstrained countries.[6]

In the forestry sector, one policy option to reduce carbon emissions, by increasing sequestration, is through afforestation programs, which generally focus on setting aside agricultural land to forestry (Alig et al. (1997); Aukland et al. (2003)).[7] These programs are subject to substantial leakage that can result in 10% to 90% of the total quantity of land set aside returning to cropland in other areas.[8] This leakage occurs because removing land from agricultural production and increasing the land in forestry increases agricultural land rents while simultaneously reducing the forestry land rents, causing a conversion of land to agriculture (Alig et al. 1997). This leakage can occur on many levels. Murray et al. (2004) find that US afforestation programs induce leakage domestically, while Gan and McCarl (2007) and Sohngen et al. (1999) find global leakage resulting from forestry programs. For other agricultural set aside programs, particularly the Conservation Reserve Program (CRP), some have shown that leakage could occur at the farm level (J. Wu 2000).

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[6] Others have applied carbon leakage concepts to interacting state and national regulations. Goulder et al. (2009) studies the potential for carbon leakage due to the interaction between state and national fuel economy standards. Fowlie (2008) finds carbon leakage due to restrictions on California electricity producers.

[7] Increasing the productivity of existing forests through management strategies is the other main option (US EPA 2009c).

[8] Murray et al. (2004), Gan and McCarl (2007), Sohngen et al. (1999) and Chomitz (2002) have all found leakage in forestry programs. Stavins and Jaffe (1990) find leakage in wetland forests resulting from flood control projects.

The evidence of carbon leakage suggests that estimating the total impact of an emissions mitigation strategy on atmospheric greenhouse gas concentrations requires an understanding of all potential leakages. It also suggests that omitting potential leakages from an analysis of a mitigation program could lead to a drastically different assessment of the emissions consequences.

United States' Climate Policy and the Renewable Fuel Standard

Reducing the consumption of gasoline by automobiles is chief among the policy options for lowering the United States' greenhouse gas emissions. As one of the country's main sources of greenhouse gas emissions, 22% of US total fossil fuel related emissions (US EPA 2009a), there is a large potential to reduce emissions through policy measures. It follows that reducing the gasoline consumed by passenger vehicles will increase the United States' energy independence. In 2008, the US imported 66% of the petroleum products it consumed (EIA 2008a) effectively linking the US economy to market disruptions in the rest of the world through changes in the world crude oil price. The gasoline used in passenger vehicles was a major factor for these imports, accounting for 47% of total US crude consumption in 2008 (EIA 2008a).

There are many policy options that could reduce the consumption of gasoline. Three of the most commonly discussed include increased gasoline taxes, increased corporate average fuel economy (CAFE) standards and increased consumption of biofuels. Perhaps due to the political infeasibility of increasing gasoline taxes, the US government has pursued the CAFE and biofuels option.[9]

[9] The most recent modification to the CAFE standards will increase fuel efficiency standards from current levels, 11.7 kilometers per liter, to 14.9 kilometers per liter in 2020 (US Congress 2007).

Much support for biofuels arises because these fuels can be manufactured from domestic feedstock and used as a direct replacement for gasoline derived from crude oil. Ethanol,[10] the primary biofuel in the US, can be mixed at small quantities, typically less than 10%, with conventional gasoline to form a blended fuel that can be used in an unmodified internal combustion engine. Others support biofuels as a method of revitalizing rural communities, through increased demand for agricultural commodities.[11] In addition, the combustion of biofuel has clear environmental advantages over the combustion of gasoline. The use of gasoline increases atmospheric CO_2 because the carbon stored in gasoline, which is released upon combustion, had previously been sequestered within the earth. As biofuels are derived from renewable feedstock, which take in CO_2 from the atmosphere while growing, the combustion of biofuel only releases CO_2 that was recently in the atmosphere.

For these reasons, the US government has actively supported biofuels. Since 1978, the US government has subsidized, through various tax credits, the blending of ethanol with gasoline. These supports are still currently in place, with fuel blenders receiving 0.12 $/liter ethanol blended with gasoline since 2009 (CBO 2009). More recently, the US government has mandated levels of biofuel consumption through the Energy Policy Act of 2005 (EPACT) which mandated the consumption of 15.12 billion liters of ethanol by 2006 and 28.35 billion liters of ethanol by 2012. The mandate effectively had no impact on ethanol consumption, or was non-binding, as in both 2006 and 2007 ethanol consumption was significantly higher than mandated levels (RFA 2009a).

[10] Ethanol is the commonly used name for ethyl alcohol.
[11] See Swenson (2006), Dorr (2006) and Ethanol Across America (2008) for example.

The focus of this study is the United States' current mandate for corn ethanol, which was enacted in the Energy Independence and Security Act of 2007 (EISA).[12] The EISA expanded and extended the EPACT mandate through 2022 under the Renewable Fuel Standard Program (RFS) program. The RFS requires 77.5 billion liters of biofuel to be consumed annually by 2015, with no more than 56.7 billion liters coming from corn based ethanol. The remaining portion of the mandate is to be met by non-corn based 'advanced' biofuels. While the mandate for corn based ethanol remains at 56.7 billion liters after 2015, the mandate for advanced biofuels continues to increase, reaching 79.38 billion liters by 2022.

The Renewable Fuel Standard and Carbon Leakage

To fully capture the emissions consequences of the Renewable Fuel Standard, the potential emissions savings, which is a consequence of the displacement of gasoline by ethanol in the blended fuel supply, and all sources of carbon leakage must be assessed. As the production and use of ethanol impacts the agricultural and energy sectors, both of which are fully integrated into the global economy, the traditional sources of leakage must be considered. In addition the production of ethanol and corn are emissions intense, relative to gasoline production and the production of other crops respectively, leading to sources of leakage that differ from those in the literature. These leakages occur because a product that saves emissions during its use exacerbates emissions during its production.

Domestically, in response to an increased RFS the potential for leakage exists in the transportation, fuel production, and agricultural markets. Internationally, leakage may occur through adjustments in energy consumption and land use.

[12] We focus on the mandate for corn ethanol through 2015 because at this time, our model does not incorporate the allocation of bioenergy crops to cropland that would result from the second phase of the Renewable Fuel Standard.

Domestic Transportation Sector

The RFS constrains the transportation sector by forcing the consumption of a certain quantity of ethanol. The potential leakage is a domestic extension of the fuel-market leakage. As ethanol displaces gasoline in blended fuel, the demand for crude oil used to produce gasoline decreases, lowering the world price of crude oil and gasoline. If the depressed price of gasoline outweighs the increased price of ethanol, the price of blended fuel will fall and blended fuel consumption will increase. The increased consumption of blended fuel will offset some of the emissions saved through the displacement of gasoline with ethanol. This leakage could become a domestic emissions benefit if, in response to a binding mandate, the increased price of ethanol outweighs the depressed price of gasoline. In this situation, the price of blended fuel would increase, the demand for blended fuel would fall and there would be an additional reduction in gasoline consumed.

Domestic Fuel Production

The majority of US ethanol production currently relies on technology that converts the starch from corn, into sugar and then ferments these sugars to produce ethanol.[13] This process uses natural gas and coal to provide the heat needed during the conversion process, and produces a substantial amount of greenhouse gas. The leakage in domestic fuel production does not fall into a traditional leakage category. Instead leakage occurs because the product that is emissions neutral to consume (ethanol) is more emissions intensive to produce than the product it displaces (gasoline). The magnitude of this leakage due to the RFS will be dependent both on the quantity of ethanol added to fuel supply, but also on the total change in blended fuel consumption.

[13] In 2008, 98% of ethanol produced in the United States used corn as a feedstock (US EPA 2009d).

Domestic Agricultural Sector

Leakage in the agricultural sector occurs through the same pathway as the forestry or agricultural set aside programs. The RFS forces corn to be diverted from other sectors to ethanol production which is comparable to taking a portion of cropland out of production. As the supply of corn available to other sectors falls, the price of corn increases, as do the land rents that farmers receive. In turn prices of all other crops increase. Farmers can respond to higher crop prices by adjusting the allocation of land to crops, intensifying agriculture practices or by increasing the quantity of land in production. The extent to which these adjustments occur determines the magnitude of the carbon leakage.

Like the production of ethanol, the production of corn generates more emissions than most other uses of agricultural cropland. Therefore, the RFS is likely to increase the overall emissions from US agriculture as the higher price of corn will lead to increased corn production. In addition, any expansion of cropland could increase the size of the leakage through land use change emissions.

International Energy Markets

The US transportation sector is linked to the world energy markets through the trade of crude oil. As the US accounts for 25% of world crude oil consumption (EIA 2009a), reductions in US demand could depress the world price and lead to increased global consumption and carbon leakage. This leakage is similar to the energy-market leakage described in the climate policy literature, only with the ethanol mandate reducing the demand for fossil fuels as opposed to emissions restrictions or environmental taxes.

International Land Uses

Similar to the global leakage that can result from afforestation programs, the increased consumption of corn by the US ethanol sector and the reduced production of other US crops that induced RFS, will cause US agricultural exports to fall. The result is an increase in world crop prices, and likely an intensification and expansion of global agricultural production. If there is a sizeable expansion global cropland, or a large conversion of native ecosystems, the resulting carbon leakage would be substantial.

<u>Current Study</u>

The goal of this study is to analyze the impact of the Renewable Fuel Standard on greenhouse gas emissions. As such, we assess not only the emissions savings from the displacement of gasoline with ethanol, but also the magnitude of each leakage discussed above. Specifically, we use a multi-market simulation model to examine the impact of the RFS mandate for corn ethanol, between 2008 and 2015, on greenhouse gas emissions from domestic passenger vehicle transportation, fuel production and agricultural production as well as international crude oil consumption and land-use change. The emissions from transportation depend on the ethanol-gasoline mix in blended fuel, changes in fleet composition (shifts in fuel economy) and household demand for vehicle miles travelled (VMT), while the emissions from fuel production are related to the demand for blended fuel and the blended fuel mix.[14] The emissions from agricultural production are related to changes in the land allocated to four major crops (corn, soybeans, hay and wheat), six crop rotations (continuous and multi-crop rotations) and four tillage systems (conventional, reduced, mulch and no-till). Within

[14] Although all values reported in this thesis are in metric units (kilometers), the terminology ‘vehicle miles traveled’ and VMT is standard, so it will be used throughout the text.

the agricultural sector, we also consider emissions related to the conversion of land in the Conservation Reserve Program to agricultural production. Internationally, emissions from crude oil consumption are related to changes in the international price of crude, and emissions from land use change are related to the mandates impact on US crop exports.

Previous Studies

A number of other studies have analyzed the potential for carbon leakage from biofuel consumption. These studies can be grouped based on the leakage that is measured. The most prevalent studies use lifecycle analysis techniques to estimate the carbon leakage in the biofuel and agricultural production sectors. Other studies have used economic models to predict the impacts of an expanded US biofuel sector on US agricultural production and trade, with a subset of these studies estimating carbon leakage by linking these adjustments to greenhouse gas emissions. Another group of studies analyzes the potential leakage through international land use change either by estimating the extent of international cropland expansion due to increases in US biofuel consumption, or by comparing the emissions from converting a unit of native land to agriculture and the emissions savings from the biofuel that could be produced using that unit of land. The last group of studies analyzes the potential leakage through domestic and international energy markets.

Lifecycle Analysis

Most studies attempting to determine the emissions consequences of biofuel rely on lifecycle analysis (LCA) techniques.[15] LCA attempts to estimate all emissions

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[15] There are a number of lifecycle studies that focus specifically on corn ethanol with the same general structure described in the text. Farrell et al.(2006) conducted a meta-analysis of six other corn ethanol LCA studies. Argonne National Laboratory's Greenhouse Gas, Regulated Emissions and Energy in Transportation (GREET) model (2000) is often used in LCA studies and has been consistently updated

that are related to the existence of a single unit of a certain product by tracing and assigning emissions to flows of material and energy used in the production and consumption of that product. As such, LCA is able to compare products that have different emissions intensities for different stages of the lifecycle and estimate possible carbon leakage. This has been important for analyzing corn based ethanol because while the net emissions from ethanol combustion are much less than those from gasoline, ethanol production and corn farming are more emissions intensive than the refining of gasoline and crude oil recovery.[16]

The LCA emissions metric for a unit of corn ethanol would include the emissions from the production of farm inputs (fertilizer, pesticide and energy), the production of corn (energy combustion, fertilizer application and carbon sequestration), the production of ethanol and the combustion of ethanol. The lifecycle methods also assign emissions 'credits' to the animal feeds that are co-produced with ethanol. This emissions credit is calculated by estimating the lifecycle emissions of the traditional animal feeds displaced by these co-products (Farrell et al. (2006), Kim and Dale (2002); Liska et al. (2009); Wang (1999)). If the lifecycle emissions of ethanol are less than the lifecycle emissions of gasoline, then LCA studies conclude that emissions savings of consuming a unit of ethanol instead of a unit of gasoline outweighs carbon leakage.

Farrell et al. (2006) evaluate six lifecycle studies and find that the lifecycle emissions from corn ethanol range from 20% less than to 32% greater than an energy equivalent unit of gasoline. In addition they use the best data from the available studies to construct their own estimate and find that the lifecycle emissions of ethanol

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(Wang, M. Wu, and Huo 2007). Other standard lifecycle analyses include: Pimentel (2003), Patzek (2004), Delucchi (2003), Hill et al. (2006), Kim and Dale (2005) and Liska et al. (2009).

[16] This is a general conclusion of most lifecycle studies of the emissions from corn ethanol.

are 18% lower than the lifecycle emissions from gasoline. This estimate is consistent with Hill et al. (2006) who estimate corn ethanol to have 12% lower lifecycle emissions than gasoline.

Other lifecycle studies have focused on how corn and ethanol production practices may affect the magnitude of carbon leakage and therefore total lifecycle emissions. Wang et al. (2007) study the impact of the fuel efficiency and type of energy used in ethanol plants on lifecycle emissions. They report that given a US average mix of fuels used in ethanol plants, which would be 25% coal fired and the remaining natural gas powered, ethanol's lifecycle emissions are 19% lower than gasoline. If only coal is used, the lifecycle emissions from ethanol are 3% higher than those from gasoline. Liska et al. (2009) find that the lifecycle emissions of corn ethanol can be as much as 59% lower than the lifecycle emissions of gasoline, if the data is based on a recently built, natural gas fired ethanol plant and average corn from Nebraska. Kim and Dale (2005) study how different cropping practices, such as crop rotations and winter cover crops, affect the lifecycle emissions.

Most standard LCA studies find that the emissions savings from ethanol combustion outweigh the carbon leakage in the production of ethanol and corn.[17] Crutzen et al. (2008) however, finds that the N_2O emissions factors used in many LCA studies are 3-5% lower than an emissions factor estimated using a top-down approach. This deviation is large enough to make the lifecycle emissions of ethanol higher than those of gasoline.

[17] There are some notable exceptions. For example, Pimentel has consistently found the lifecycle emissions of corn ethanol to be higher than those of gasoline (Pimentel (2003); Pimentel and Patzek (2005); Patzek and Pimentel (2005)), although the methods and data used in these studies has been criticized (Farrell et al. 2006).

Agricultural Market Impact Studies

Another large group of studies use economic models to predict how the US agricultural sector responds to increased ethanol consumption. Most studies have relied on existing agricultural models that focus on adjustments in the quantity of land harvested, the allocation of crops planted, and adjustments in exports, but not carbon leakage specifically. Peterson (2008) notes that the allocation of crops and the extent of cropland harvested are only two of the potential agricultural adjustments. He proposes a framework for analysis that includes crop choice and pattern (rotation), management intensity (input usage and tillage practices) and the structural diversity of cropland.

Tokgoz et al. (2008) use a set of non-spatial multi-market partial equilibrium models from the Food and Agricultural Policy Research Institute (FAPRI) to analyze the impacts of a 10.0 $/bbl higher crude price in 2016. They find that the higher price of crude oil causes ethanol consumption to increase 55% from 55.9 billion liters to 86.52 billion liters. This results in an increase in corn production of 11.4% at the expense of wheat and soybean production, both of which fall by roughly 6%. Likewise, US exports of corn, soybeans and wheat exports fall by 30%, 21% and 12.5% respectively. Gohin (2008) finds similar responses in the European agricultural sector, increases in the production of crops used for energy and decreased crop exports, to the EU's renewable fuel mandates.

Walsh et al. (2003) use the POLYSYS agricultural simulation model to analyze impact of different prices scenarios for the bioenergy crops between 1999 and 2008 and find that the majority of land used to produce biomass (switchgrass, poplar, willow) crops comes out of cropland and CRP land, while the expansion on to idled and pasture land is small. As would be expected they report a large displacement of hay (including alfalfa), corn, soybeans and wheat.

There are a number of other studies that deserve mention. Feng and Babcock (2008) construct an analytical framework that models the allocation of land to crops as a function of input use, prices, land markets, yields and total cropland area. A significant result of this work is that as higher yields increase profits per unit of land, increased yields could lead to more land use change, rather than less, if crop prices are not dramatically depressed. Westhoff (2007) uses the FAPRI models to analyze impact of the RFS corn ethanol mandate on US agriculture, and agricultural trade. The focus of this study is crop allocation, crop prices, exports as well as impacts on other portions of the agricultural sector. The US EPA (2009b) uses the Forest and Agricultural Sectors Optimization Model (FASOM) to estimate the domestic agricultural emissions that result from the RFS and the FAPRI models to estimate the impacts of the RFS on foreign agriculture, trade and emissions. The ERS (2007) uses the Food and Agricultural Policy Simulator (FAPSIM), to analyze domestic crop production in response to an expansion of the ethanol sector the mimics the RFS. The ERS (2007) also uses Regional Environmental and Agricultural Programming (REAP) model to analyze the impacts of increased ethanol and biodiesel use on the allocation of crops, rotations and tillage practices for different regions of the US.

Carbon Payback Studies

The other set of studies that analyze the land use change leakage focus on 'what if' scenarios that compare the emissions from converting a unit of native land to biofuel production to the lifecycle emissions savings of the biofuel that could be produced on that land over a given period. The carbon leakage induced from land use change includes the direct emissions from the burning and decomposition of plant biomass, soil carbon oxidation and foregone sequestration benefits. In general, these studies find that the carbon leakage from land use change is far greater than the domestic emissions savings of biofuel use (estimated by LCA). Fargione et al. (2008) compare the annual lifecycle emissions savings of various biofuels to the carbon lost as a result of converting land to agriculture and the number of years required to 'pay-back' the emissions from conversion were calculated. If central grassland is converted to corn ethanol production, the payback period is 93 years, while if corn ethanol production occurs on abandoned cropland the payback period is only 48 years.

Gibbs et al. (2008) estimate region specific carbon payback times for biofuels produced in the tropics using a spatial database of crop locations and yields, and updated vegetation and soil biomass data. They test the sensitivity of their estimates to yield improvements, biofuel production technology and more carbon intensive petroleum sources (tar sands and oil shale). They find that replacing tropical forests with biofuel production using current technology (ethanol or soybean based diesel), would lead to payback periods of between 300 and 1500 years. They also find that there could be significant carbon benefits from expanding sugarcane or oil palm production on to already degraded lands. They also find that the carbon savings of biofuel compared to regular gasoline will increase by about 25% over the next 20 years as the lifecycle emissions from gasoline production increase.

Kim et al. (2009) find that the carbon payback times are dependent on agricultural management practices. For grassland and forest converted for ethanol production, the payback period could be as small as 3 and 7 years respectively, if sustainable cropping practices (no-tillage with cover crops for example) are used to produce the corn for ethanol. Righelato and Spracklen (2007) estimate that the carbon benefits of corn ethanol, over a 30 year period, are far less than restoring cropland to native forests.

Land Use Change Studies

A related set of studies focuses on the impacts of increased US ethanol consumption on the world agricultural markets and specifically the expansion of cropland globally. A subset of these studies combine this economic analysis with the methods of the carbon payback studies to compare the emissions incurred through conversion of native ecosystems to cropland to the emissions savings of biofuel.

The most prominent of the studies that include emissions estimates, Searchinger et al. (2008), use the FAPRI models to estimate the worldwide cropland expansion resulting from an expansion of US ethanol production which reached 111.76 billion liters in 2016, an increase of 55.92 billion liters above the baseline scenario. As a result of the expanded ethanol production, US exports of corn, soybeans and wheat decline by 62%, 28% and 31% respectively, and an additional 10.8 million hectares of uncultivated land is brought into production worldwide. This conversion leads to emissions, over 30 years, of 3,801 $TgCO_2e$, or 351 $mtCO_2e$/ha. When factored in to a lifecycle analysis of corn ethanol, these emissions outweigh the lifecycle savings of ethanol over 30 years. In fact, the cumulative lifecycle emissions savings of ethanol would only outweigh the carbon losses from land use change after 167 years.

Ravindranath et al. (2008) find that first generation biofuels are likely to induce carbon leakage that is larger than the potential savings of 30 years of ethanol production. They conclude that biofuel must utilize feedstock, such as waste products, cover crops, or crops grown on marginal lands, so the conversion of native cropland is limited. Dumortier et al (2009) use the FARPI models to test the sensitivity of the payback periods of land use change emissions to assumptions about US deforestation, crop yields and lifecycle emissions savings of ethanol. They find that restricting deforestation from occurring in the US, which account for 36% of new US cropland in the Searchinger et al.(2008) study, the payback period of corn ethanol falls from 180 years to 120 years. Likewise, if international yields attain 1% higher yields by 2022, the payback period is only 31 years.

Studies that analyze the global impact of expanded biofuels consumption but that do not predict emissions are more common. Leemans et al. (1996) use the Integrated Model to Assess the Global Environment (IMAGE) to study the IPCC's Low-Emissions Supply System (LESS) scenario constructed by the IPCC. They find that while large scale biomass use is possible, the competition between food and bio-energy production will increase the probability of deforestation. Keeney and Hertel (2010) use the Global Trade Analysis Project (GTAP) model to analyze a 3.78 billion liter increase in US ethanol production while allowing for an agricultural yield response to crop prices. They find that 31% of the output response to increased ethanol production is due to yield gains above normal trends, which significantly reduces the amount of new cropland brought into production. They suggest that the magnitude of the land use change effects found by Searchinger et al. (2008) could be too large. Tyner and Taheripour (2008) note that most other studies have focused on a single country's mandate. They simultaneously study the EU and US biofuel mandates using the GTAP model and find that the interactions between the two mandates could lead to much larger land use change effects.

Taheripour et al. (2009) demonstrate the importance of incorporating the ethanol co-products into the analysis of agricultural response. They find that an increase in world cereal grain production resulting from US and EU biofuel mandates of 10.8% if biofuel co-products are allowed to displace other agricultural commodities and an increase of 16.4% if co-products are not included. Fabiosa et al. (2009) use the FAPRI models to estimate international land use changes that result from increased ethanol consumption in US, Brazil, EU, China and India and find that trade restrictions, accentuate the domestic agricultural adjustments, while limiting the international land use effects.

Gurgel et al. (2007) use the MIT Emissions Prediction and Policy Analysis (EPPA) model to assess the adoption of cellulosic ethanol technologies over the 21st century. They find that a large scale adoption of advanced biofuel technology would result in a 40% reduction in the world's forest land by 2100, and that pasture land would also be significantly reduced.

Fuel Market Impact Studies

A final group of studies analyze how the increased consumption of biofuel impacts US and international the energy markets. While these studies find the potential for leakage in the energy markets as US fuel prices and the price of crude oil may fall in response to increased biofuel consumption, they do not measure the emissions impacts.

Khanna et al. (2008) suggest a framework for assessing the impacts of a fuel tax and an ethanol tax credit on the consumer's demand for driving in absence of an ethanol mandate. Empirically, using lifecycle emissions parameters, they find that the current fuel tax and ethanol tax credit reduce emissions relative to a scenario without government policy, but that emissions savings could be increased by lowering the tax on gasoline and the ethanol tax credit, and imposing a tax on driving.

De Gorter and Just (2009a) show analytically that an ethanol mandate's impact on the price of blended fuel depends on the relative elasticity of supply of gasoline and ethanol and the ethanol consumption required by the mandate. Empirically, they find that the price of blended fuel decreased in response to expanded ethanol use in 4 of the 6 years of they analyzed between 2001 and 2007. They expect the price of blended fuel to increase in response to the RFS for 2008 and 2015 because in these years ethanol production uses a larger share of US corn produced, causing the supply of ethanol to become less elastic.

Du and Hayes (2009) use pooled time-series data for regions of the US between 1995 and 2008, and find that increased ethanol production has kept wholesale gasoline prices 0.04 $/liter lower than they would have been without ethanol.

Dixon et al. (2007) use the USAGE model to simulate effects of replacing 25% of US crude oil consumption with biomass by 2020. In 2020, they find that the increased use of biomass reduces the world price of crude oil and subsequently the cost of producing US motor fuels.

Finally, the EPA Draft Regulatory Analysis of the Renewable Fuel Standard (2009b) mentions both the potential for the RFS to impact both domestic and international energy markets, but has not fully quantified either effect or the resulting emissions.

Features of Current Model Framework

This study differs from the previous work in a number of ways. First, we estimate greenhouse gas emissions within a general equilibrium framework. This allows us to simultaneously measure the gross emissions savings from the combustion of ethanol and all potential sources of carbon leakage that result from a certain policy. This contrasts with previous studies which have tended to focused on a single source of leakage (agricultural impact studies or land use change impact studies), or have not used an economic framework and therefore not included relevant sectors and behavioral adjustments (lifecycle analysis).

Consistent with the lifecycle analysis literature, we calculate the emissions not only from the use of ethanol, but also from the production of ethanol and all relevant inputs, specifically corn and fertilizer. Unlike the LCA studies, we are able to link the sectors both by the flow of goods, but also by market forces. This framework enables us to determine the impact of biofuel policy on each of the major sectors affected by the mandate, even those not contained in the ethanol 'lifecycle'. This wider scope gives us a better understanding of the impacts of a policy, especially on sectors that compete for the main inputs to ethanol production (land and corn). Specifically, it would be expected that any increase in ethanol production would divert corn from other end uses, such as food production and crop exports as well as spur increased corn production. Likewise, the expanded corn production would come at the expense of other crops and other land uses. We are able to assess the magnitude of the carbon leakage from these adjustments.

Following the group of studies that analyze the agricultural adjustments we measure the impact of expanded ethanol consumption on the allocation of land to crops, the end-use of crops grown and crop exports.[18] However, consistent with the framework of Peterson (2008), we also consider the impacts of increased ethanol consumption on the intensity of agricultural production. We allow the agricultural sector to adjust not only between crops, but also between continuous and multi-crop rotations and tillage practices. Capturing these effects allows for a more accurate assessment of leakage in the agricultural sector as there is considerable variability in the input usage, and emissions consequences, of different agricultural production systems (rotations and management practices).

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[18] Domestic agricultural analyses such as Tokgoz et al. (2008), Feng and Babcock (2008) and Westhoff (2007) focus on the allocation of crops, as do most studies of the international agricultural response to expanded biofuel production. For example: Taheripour et al. (2009), Searchinger et al. (2008) or Keeney and Hertel (2010).

Similar to Searchinger et al. (2008), we estimate the emissions from the expansion of domestic agriculture on to land not previously used for agriculture.[19] However, we allow this expansion to occur only through the re-cultivation of CRP lands and not into forests or other native ecosystems. Likewise, by modeling the international demand for US agricultural products, we are able to infer the magnitude of the potential leakage from international land use change.

Finally, consistent with the framework propose by de Gorter and Just (2009a), we allow the price of crude oil to be endogenous in our model. In addition, we model the fuel blender's decision, which depends on the pre-existing ethanol tax credit, the ethanol mandate and the prices of gasoline and ethanol, and the household's driving decision, which allows the consumer to choose between fuel and non-fuel (fuel economy) expenditures. Combining these features, we are able to capture the impact of expanded ethanol consumption on the price of blended fuel and the world price of crude oil. It follows that we are able to capture the carbon leakage in both the domestic fuel market and the international market for crude oil. Our model's ability to capture the emissions consequences of these effects contrasts with all prior studies. Our study is also different than previous work because each leakage is estimated simultaneously, which allows us to compare the relative magnitude of each leakage and contrast with the gross emissions savings of a policy.

Structure of Thesis

By accounting for each leakage, and linking all economic decisions to government policy, our framework is able assess the impact of the Renewable Fuel Standard on greenhouse gas emissions. The rest of the thesis is organized as follows.

[19] Other domestic agricultural studies such as Tokgoz et al. (2008) and Westhoff (2007) also estimate the expansion of agricultural land, but do not state what land use is being displaced by agriculture or estimate the emissions consequences.

Chapter 2 describes the equilibrium simulation model and the carbon emissions model. Chapter 3 describes the data used and calibration procedure of each model. Chapter 4 presents and interprets the results from simulations of the Renewable Fuel Standard. Chapter 5 presents an analytical and empirical comparison of the emissions estimates from a general equilibrium framework and lifecycle analysis. Chapter 6 offers additional discussion and conclusions.

Chapter 2. Model Description

This chapter outlines the mathematical structure of the economic simulation model in Section I. In Section II, the structure of the emissions model and the parameters that link the two models are discussed.

Section I – Economic Simulation Model

The economic agents modeled are households, producers of agricultural crops, producers of ethanol and producers of food, along with suppliers of regular gasoline and suppliers of blended fuel. The model considers the vehicle miles traveled (VMT) and food consumption decisions of a representative household in accordance with utility maximization. The representative consumer faces a pre-existing fuel tax.

The model considers the allocation of land to four crops, (corn, soybeans, hay and wheat) and the enrollment of land in the Conservation Reserve Program (CRP). Landowners also allocate land to rotations and tillage systems. Corn is used as an input in the production of ethanol and food, while the other crops consumed domestically are used as an input to food production. The model also considers trade in corn, soybeans and wheat, while hay is consumed only domestically.

Blended fuel suppliers equate the marginal cost of producing ethanol with the marginal cost of producing regular gasoline when deciding the quantity of ethanol to demand. For each liter of ethanol blended, the blenders receive a tax credit. In turn, the marginal cost of regular gasoline is linked with the international price of crude oil and the marginal cost of ethanol is linked with the price of corn.

<u>Households</u>

Household Utility

Households obtain utility from consuming VMT, food, and other commodities. Each household has exogenous income, which reflects the returns to the fixed factors in the economy: labor, capital and land. Households' income also includes the returns (positive or negative) of the goods sold (or purchased) internationally, which are crops (corn, soybeans and wheat) and crude oil. The representative agent has the following utility function:

$$U = \left[\alpha_T T^{\left(\frac{\sigma_u - 1}{\sigma_u}\right)} + \alpha_M M^{\left(\frac{\sigma_u - 1}{\sigma_u}\right)}\right]^{\left(\frac{\sigma_u}{\sigma_u - 1}\right)} \tag{2.I.1}$$

where

$$T = \gamma_{UT}\left[\alpha_{UTC} C^{\left(\frac{\sigma_{UT} - 1}{\sigma_{UT}}\right)} + \alpha_{UTX} X^{\left(\frac{\sigma_{UT} - 1}{\sigma_{UT}}\right)}\right]^{\left(\frac{\sigma_{UT}}{\sigma_{UT} - 1}\right)} \tag{2.I.2}$$

and where M, C and X denote the quantities of VMT, a numeraire good and food consumed. T is introduced to allow for flexibility in the elasticity of substitution among these three goods and represents a composite of C and X. The parameters α_M, α_{UTC} and α_{UTX} represent the share of expenditure, relative to total expenditures, for VMT, the numeraire good and food respectively. The parameter σ_u denotes the elasticity of substitution between VMT and the composite good T, while σ_{UT} is the elasticity of substitution between the numeraire good and food.[20]

Households seek to maximize the utility function in (2.I.1) and (2.I.2) by choosing the quantities of M, C, and X subject to a budget constraint. The household budget constraint can be written as:

$$p_c C + p_X X + p_M M = [(1 - t_L) w_L \bar{L} + (1 - t_K) w_K \bar{K} + R_{\bar{A}} + EXP + G] \tag{2.I.3}$$

[20] In the remaining description of the model, each α_i represents the share of total expenditure on input or good i. Likewise each σ is an elasticity of substitution.

where p_c, p_X and p_M denote the price of the numeraire good, food and VMT respectively. The wage rate, normalized to 1, and the interest rate on capital are denoted by w_L and w_K, while $\bar{L}$ and $\bar{K}$ are the endowments of labor and capital. $R_{\bar{A}}$ is the return on land and EXP is the value of net exports. The government transfer, G, includes revenue from the labor tax (t_L), the tax on capital (t_K) and the tax on blended fuel.

Household Production of VMT

The household combines blended fuel with money expenditures on driving to 'produce' VMT. Following Parry and Small (2005), we represent the production of VMT as:

$$M = \gamma_M \left[\alpha_F F^{\left(\frac{\sigma_M - 1}{\sigma_M}\right)} + \alpha_H H^{\left(\frac{\sigma_M - 1}{\sigma_M}\right)} \right]^{\left(\frac{\sigma_M}{\sigma_M - 1}\right)} \qquad (2.I.4)$$

where F is the consumption of blended fuel and H are the monetary expenditures on driving. The household chooses the quantity of blended fuel and the monetary expenditures on driving so as to minimize the costs of VMT, given by:

$$(p_F + t_f)F + H \qquad (2.I.5)$$

where p_f is the pre-tax price of blended fuel and t_f is the tax on blended fuel.

The specification of VMT production in (2.I.4) implies that households do not distinguish regular gasoline from ethanol. To date most of the blends in the market are E10, or 10 percent of ethanol and 90 percent of regular gasoline, thus making it almost identical to regular gasoline. Therefore, we implicitly assume that households do not distinguish between different blends available in the market and treat fuel, F, as a homogeneous commodity. Equation (2.I.4) also allows for a non-proportional relation between blended fuel and VMT. In response to an increase in the price of blended fuel, (2.I.4) allows for reductions in VMT as well as substitution towards more fuel-efficient vehicles (causing an increase in H). The price per kilometer of driving that results from the minimization of (2.I.5) subject to (2.I.4) is given by:

$$p_M = \frac{1}{\gamma_M}\left[\alpha_F^{\sigma_M}\left(p_F + t_f\right)^{(1-\sigma_M)} + (1-\alpha_F)^{\sigma_M}\right]^{\left(\frac{1}{1-\sigma_M}\right)}. \qquad (2.I.6)$$

Production of Crops and Cropland Allocation

The representative agent is endowed with a fixed amount of land. For simplicity we assume that all land is allocated to agricultural uses, and we abstract from agricultural land used for range and pasture, as well as non-agricultural land uses such as rural residential and urban areas. The representative agent seeks to maximize the returns to land by deciding the combination of crop allocation, rotation practice and tillage system. The model considers four major crops indexed by k: corn, soybeans, wheat and hay. We also consider the allocation of land to the Conservation Reserve Program. The rotation practices represented in the model indexed by l are: continuous corn, continuous soybeans, continuous wheat and continuous hay, corn-soybeans and corn-soybeans-wheat. Finally, we consider four tillage systems indexed by t: conventional tillage (CT), reduced tillage (RT), mulch tillage (MT) and no-till (NT).

We model the crop-rotation decisions using two nested constant elasticity of substitution (CES) functions, indexed at the crop level, that are integrated into a non-linear returns maximization objective function. In the first CES nest, land is allocated between single and multi-crop rotation practices, indexed by p. In the second nest, land is allocated between each of the multi-crop rotation practices for a given crop.[21]

[21] Since there is only one single crop rotation per crop, these nests are one-to-one for single crop rotations.

Conditional on the rotation allocation decision, the landowner simultaneously determines the cost minimizing allocation of land to each of the four tillage systems. This level is modeled using fixed proportion production with four inputs: labor, capital energy (calibrated in this model to be natural gas) and fertilizer.

Formally, our representative agent maximizes the returns to land given by:

$$\sum_{k=1}^{4}(\beta_k - \delta_k A_k)A_k q_k + A_{CRP}s_{CRP} - \sum_{l=1}^{6} c_l A_l \tag{2.I.7}$$

taking into account the following constraints:

$$\sum_{k=1}^{4} A_k + A_{CRP} = \bar{A} \tag{2.I.8}$$

$$A_k = \gamma_k \left[\sum_{k\,\in p}^{2} \alpha_{kp} A_{kp}^{\left(\frac{\sigma_k - 1}{\sigma_k}\right)}\right]^{\left(\frac{\sigma_k}{\sigma_k - 1}\right)} \quad \forall k = 1, \dots, 4 \tag{2.I.9}$$

$$A_{kp} = \gamma_{kp} \left[\sum_{l\,\in kp}^{6} \alpha_{kpl}(s_{kl}A_l)^{\left(\frac{\sigma_{kp} - 1}{\sigma_{kp}}\right)}\right]^{\left(\frac{\sigma_{kp}}{\sigma_{kp} - 1}\right)} \quad \forall k = 1, \dots, 4 \; \forall p = 1,2 \tag{2.I.10}$$

$$A_k \cong \sum_{l\,\in k}^{6}(s_{kl}A_l) \;\; \forall k = 1, \dots, 4 \tag{2.I.11}$$

where β_k is the intercept and δ_k is the slope in crop k's yield function, q_k is the price per unit yield of crop k and s_{CRP} is the per unit CPR rental rate. $\bar{A}$ is the total endowment of land, A_k is the amount of land allocated to crop k, and A_{CRP} is the amount of land allocated to CRP. A_{kp} is the amount of land allocated to crop k in a single-crop and multi-crop rotations p, A_l is the amount of land allocated to rotation practice l, c_l is the marginal cost per hectare of land in rotation practice l, and s_{kl} is the share of crop k in rotation practice l.

Equation (2.I.8) requires that the total land allocated to each of the four crops and to CRP land equal the total land available. Constraints (2.I.9) are the crop-single/multi-crop rotation system CES functions that allow for the conversion across single and multi-crop rotation systems for each of the individual crops. Note that multi-crop rotations enter into more than one of these equations, making a closed-form solution intractable. Constraints (2.I.10) are the single/multi-crop and rotation system CES functions that allow for conversion across rotations systems for each of the crop and rotation system aggregate categories. Finally, (2.I.11) are the rotation-share balancing constraints for each crop. In effect, these constraints require that the total land provided by each of rotation for each of the crops reflects the total crop allocation predicted by the model.[22]

The marginal cost per hectare of land in rotation practice l, c_l, is calculated by minimizing the following cost function (total cost denoted by C_l):

$$C_l = \sum_{t=1}^{4} (c_{lt} A_{lt}) \tag{2.I.12}$$

subject to:

$$A_l = \left[\sum_{t=1}^{4} \alpha_{lt} A_{lt}^{\left(\frac{\sigma_l - 1}{\sigma_l}\right)} \right]^{\left(\frac{\sigma_l}{\sigma_l - 1}\right)}. \tag{2.I.13}$$

Finally, we model the input usage by rotation and tillage systems using Leontief technology consisting of four inputs, indexed m: labor, capital, energy and

[22] Without constraints (2.I.11), the hectares predicted for each of the rotations may fail to add up to the hectares in each crop.

fertilizer. We compute c_{lt} as simply the sum of the per hectare expenditure on inputs used in the production. That is:

$$c_{lt} = \sum_{t=1}^{4} a_{ltm} w_m \ \forall l = 1, \dots, 6; t = 1, \dots, 4 \tag{2.I.14}$$

where a_{ltm} is the per hectare quantity of input m used to cultivate land in rotation practice l and tillage system t, and w_m is the price per unit of input m.

Production of Ethanol

Ethanol production is a multi-output production process. In addition to ethanol, four co-products (indexed i) are produced: distillers' grains (DGS), corn gluten meal (CGM), corn gluten feed (CGF), and corn oil. We model this process using a fixed proportion technology, given by:

$$min\left\{\frac{Y_{k1}}{\lambda_{Y_{k1}}}, \frac{L}{\lambda_L}, \frac{K}{\lambda_K}, \frac{E}{\lambda_E}\right\} \tag{2.I.15}$$

where Y_{k1} is the quantity of corn, and L, K, E are the quantities of labor, capital and energy respectively. The λ's denote the quantity of each input needed to produce one unit of ethanol.

The ethanol producer seeks to minimize production costs, after accounting for the income generated by the selling of ethanol co-products (CO_i), subject to (2.I.15). Costs of production are:

$$C_{Fe} = q_{k1} Y_{k1} + w_l L + w_k K + w_E E - \sum_{i=1}^{4} p_{CO_i} CO_i \tag{2.I.16}$$

where w_E is the price of energy and p_{CO_i} is the price of co-product i. The resulting price of ethanol, where λ_{CO_i} is the quantity of co-product i produced per unit of ethanol, is given by:

$$p_{Fe} = \lambda_{Y_{k1}} q_{k1} + \lambda_L w_L + \lambda_K w_K + \lambda_E w_E - \sum_{i=1}^{5} \lambda_{CO_i} p_{CO_i}. \tag{2.I.17}$$

Production of Regular Gasoline

Gasoline producers combine labor, capital and crude oil (R) to produce regular gasoline. Letting w_R denote the price of crude oil, the cost function of regular gasoline can be represented by:

$$C_{Fg} = w_R R + w_L L + w_K K. \tag{2.I.18}$$

Gasoline producers minimize (2.I.18) subject to a nested CES function that represents the production function of regular gasoline. Specifically:

$$Fg = \gamma_{Fg} \left[\alpha_{FgR} R^{\left(\frac{\sigma_{Fg}-1}{\sigma_{Fg}}\right)} + \alpha_{FgLK} LK^{\left(\frac{\sigma_{Fg}-1}{\sigma_{Fg}}\right)} \right]^{\left(\frac{\sigma_{Fg}}{\sigma_{Fg}-1}\right)} \tag{2.I.19}$$

where

$$LK = \gamma_{Fg\_LK} \left[\alpha_{FgLK\_L} L^{\left(\frac{\sigma_{Fg\_LK}-1}{\sigma_{Fg\_LK}}\right)} + \alpha_{Fg\_LK} K^{\left(\frac{\sigma_{Fg\_LK}-1}{\sigma_{Fg\_LK}}\right)} \right]^{\left(\frac{\sigma_{Fg\_LK}}{\sigma_{Fg\_LK}-1}\right)}. \tag{2.I.20}$$

In (2.I.19) and (2.I.20), σ_{Fg} and σ_{Fg_LK} denote the elasticity of substitution between crude oil and a composite of labor and capital, and the elasticity of substitution between labor and capital, respectively. The resulting price of regular gasoline is given by:

$$p_{Fg} = \frac{1}{\gamma_{Fg}} \left[\alpha_{FgR}^{\sigma_{Fg}} w_R^{(1-\sigma_{Fg})} + \alpha_{FgLK}^{\sigma_{Fg}} w_{LK}^{(1-\sigma_{Fg})} \right]^{\left(\frac{1}{1-\sigma_{Fg}}\right)} \tag{2.I.21}$$

where

$$w_{LK} = \frac{1}{\gamma_{Fg\_LK}} \left[\alpha_{FgLK\_L}^{\sigma_{Fg\_LK}} w_L^{(1-\sigma_{Fg\_LK})} + \alpha_{FgLK\_K}^{\sigma_{Fg\_LK}} w_K^{(1-\sigma_{Fg\_LK})} \right]^{\left(\frac{1}{1-\sigma_{Fg\_LK}}\right)}. \quad (2.I.22)$$

Blended Fuel

Fuel blenders seek to mix ethanol and regular gasoline in order to produce blended fuel at a minimum cost. Blenders also face a constraint that mandates a quantity of ethanol to be blended. That is, the ethanol used by the blender must meet or exceed a quantity of ethanol mandated by the federal government. In addition, a tax credit is given to the blender for each liter of ethanol consumed. The blender's cost of production can be represented by:

$$C_F = (p_{Fe} - s_{Fe})Fe + p_{Fg}Fg \quad (2.I.23)$$

where s_{Fe} is the tax credit per liter of ethanol blended. The blender minimizes production costs subject to:

$$F = Fe + Fg \quad (2.I.24)$$

$$Fe \geq \delta_{Fe}\bar{F} \quad (2.I.25)$$

where F is the total blended fuel to be produced. Consistent with the way the Renewable Fuel Standard describes the mandate, (2.I.25) states that the quantity of ethanol to be blended is a function of the share of ethanol δ_{Fe} mandated relative to the total amount of fuel expected to be consumed $\bar{F}$.[23] The resulting, pre-tax, price of blended fuel is:

$$p_F = p_{Fg}(1 - \delta_{Fe}) + (p_{Fe} + s_{Fe})\delta_{Fe}. \tag{2.I.26}$$

Other Sectors

In addition to the key agents described above, the model also considers the production of the numeraire good, the production of food, the production of natural gas and the production of fertilizer. The numeraire good is produced by minimizing costs subject to a CES production function that combines labor and capital. Food producers combine crops, ethanol co-products, labor, capital and energy to produce food by minimizing costs subjected to a nested CES production function. Natural gas is produced with a CES function that combines labor and capital and it is used as an input in the production of ethanol, the production of crops and the production of food. Finally, fertilizer is produced from natural gas and is used as an input to agricultural production.

Rest-of-World Demand and Supply

We also consider the demand for corn, soybeans and wheat from the rest of the world. The inverse rest-of-the world net demand for crop k is given by:

[23] While the RFS mandates a level of ethanol consumption, the mandate is implemented through a minimum blending requirement. The RFS is administered by the EPA, through a credit trading system. Ethanol producers and importers generate Renewable Identification Numbers (RINs) for each unit of biofuel they produce. When the biofuel is sold to a fuel blender the RIN is transferred and is used by the blender to prove compliance with the RFS. If the blender obtains more RINS than the RFS requires, the excess can be sold to other blenders who can then produce blended fuel with a smaller percentage of ethanol than the RFS mandates. To implement the mandate the EPA sets the RFS using EIA projections for gasoline demand (US EPA 2007)

$$q_k = \gamma_K Y_{ROWk}^{\left(\frac{1}{\eta_{ROWk}}\right)} \; \forall k = \text{corn, soybeans, wheat} \tag{2.I.27}$$

where Y_{ROWk} is the quantity demanded, and η_{ROWk} is the rest of the world demand elasticity for crop k.

The inverse rest-of-the-world net supply of crude oil is given by:

$$w_R = \gamma_R R^{\frac{1}{\eta_R}} \tag{2.I.28}$$

where B_R is the quantity of crude oil supplied and η_R is the rest-of-world supply elasticity of crude oil.

Government

Government expenditures are financed by a tax on labor income t_l, a tax on capital t_k and a tax on blended fuel t_f. Government provides a lump-sum transfer to households, G, and a volumetric tax credit to blenders per unit of ethanol blended, s_{Fe}. The government also provides a per hectare rental payment to land that is kept in CRP, s_{CRP}. We assume that, in each period the government budget balances and is given by:

$$t_f F + t_L w_L \overline{L} + t_K w_K \overline{K} = G + s_{CRP} A_{CRP} + s_{Fe} Fe \tag{2.I.29}$$

Solution Method

To solve the model, we must obtain a vector of prices that clears all markets and an aggregate transfer level that equals the government's revenues from the different taxes. To solve the multidimensional system we use Broyden's method, a derivative-based quasi-Newton search algorithm.

Section II – Emissions Model

The emissions model measures the release of the three major greenhouse gases: carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). For most sectors and activities, a simple emissions factor approach was used, with each emissions factor representing the US average greenhouse gas emissions that result from a certain activity. Where possible, emissions factors (denoted ϕ) were derived for the most basic activity, such as the combustion of a given fuel by a given technology, and aggregated to estimate an emissions factor for more complicated processes. By modeling emissions at a relatively disaggregated level, the effects on emissions of economic adjustments in certain production practices, as well as dynamic adjustments can be captured. The final emissions factor for a specific activity, given in greenhouse gas equivalent units of carbon dioxide (CO_2e), is the total quantity of CO_2, CH_4 and N_2O emissions weighted by each gas's potential to contribute to global warming. The IPCC's global warming potential (GWP) concept is the basis for this weighting. While estimates for the relative warming potentials of these gases have changed slightly over time, we use the values from the IPCC's Third Assessment Report (2001b).[24]

The quantities and types of fossil fuel used per unit of a given activity are the basis for that activity's emissions factor. In addition, the recovery of the fossil energy also relies on fossil fuels and these emissions are measured as well. As we do not explicitly model the production of each source of fossil energy, and following lifecycle analysis methods, the emissions from the production of energy inputs to a given practice are included in that practice's emissions factor with gasoline and ethanol used for transportation the two exceptions. For example, the emissions factor

[24] The global warming potential, on a 100-year time horizon of a unit of CH_4 is 23 times greater than a unit of CO_2, while the GWP of a unit N_2O is 296 times greater than a unit of CO_2 (IPCC 2001c).

for the fossil fuels used in agricultural production would include the emissions from the combustion of the fuels plus the emissions from the recovery and production of those fuels. However, the emission factors for transportation would include only the combustion emissions of ethanol and gasoline. Likewise, the emissions from fossil fuel based electricity generation are always attributed to the sector consuming the electricity. For most activities, and depending on the analysis being conducted, the emissions from non-energy inputs are not attributed to the end use sector. As this will change in some parts of the discussion, how the emissions of non-energy inputs are attributed will be explicitly mentioned.

Certain sources of emissions cannot be modeled with emissions factors as current emissions are based not only on current practices, but also previous activities. Specifically, the sequestration of carbon in cropland soils is based on the flux of C to and from soils and is dependent on the overall stock of soil organic carbon (SOC). For this category a more complex method was used to estimate emissions and sequestration.

The measurement of greenhouse gas emissions is conducted by sector for the US economy, including the emissions from transportation (GHG_{TRANS}), ethanol (GHG_{Fe}) and gasoline production (GHG_{Fg}), agricultural production and domestic land use change (GHG_{AG}) and farm input, or 'fertilizer', production (GHG_{J}).[25] Domestically, we measure the baseline levels of emissions and the change in emissions given a policy. The change in international emissions in response to the mandate are also measured and include the change in emissions from non-US crude oil

[25] There are other domestic sectors that would be impacted by the ethanol mandate that are not considered in the current model. Notably, the food production and livestock sectors will respond to changes in crop prices. However, without a more detailed model of food production the effect on emissions will be difficult to capture.

consumption ($dGHG_{R_W}$) and the change in emissions from international land use adjustments ($dGHG_{ILU}$).

The total emissions impact of the US ethanol mandate $dGHG$ can therefore be expressed as:

$$dGHG = dGHG_{TRANS} + dGHG_{Fe} + dGHG_{Fg} + dGHG_{AG} + dGHG_J + dGHG_{R_W} + dGHG_{ILU}. \tag{2.II.1}$$

Transportation

The emissions from transportation GHG_{Trans} are the total emissions from the combustion of gasoline and ethanol, where the emissions factors represent the per unit emissions from the use of gasoline (ϕ_{Fg}^{C}) and ethanol (ϕ_{Fe}^{C}) in passenger vehicles. The total emissions from transportation are given by:

$$GHG_{Trans} = \phi_{Fg}^{C} Fg + \phi_{Fe}^{C} Fe. \tag{2.II.2}$$

Following the lifecycle analysis literature an ethanol combustion credit is calculated. This emissions credit is equal to the amount of CO_2 released when ethanol is combusted and is attributed to the agricultural sector (GHG_{AG}^{Fe}), but is discussed here only because it is a function of ethanol consumed. The emissions factor used to calculate the total size of this credit, $\phi_{Fe}^{CO_2}$, accounts for the CO_2 that was removed from the atmosphere during the growing of corn, stored in ethanol, and then released to the atmosphere when the ethanol is combusted. This credit is calculated as:

$$GHG_{AG}^{Fe} = \phi_{Fe}^{CO_2} Fe. \tag{2.II.3}$$

Ethanol Production

In the economic model, the ethanol sector is modeled as a single production process that represents an average US ethanol plant. As such, the λ's in the ethanol production function (equation (2.I.15)) are the weighted average of four ethanol plant types. Each plant type represents a distinct combination of two primary fuels, indexed h, (natural gas and coal) and two conversion technologies, indexed n, (dry mills and wet mills). For each technology combination an emissions factor ($\phi_{Fe}^{h,n}$), based on efficiency and types of energy used, is calculated per unit of ethanol. Given the shares ($s^{h,n}$) of the four production technologies, the weighted average emissions factor per unit of ethanol produced, (ϕ_{Fe}^{P}), is calculated as:

$$\phi_{Fe}^{P} = s^{h,n}\phi_{Fe}^{h,n} \tag{2.II.4}$$

and the total greenhouse gas emissions from ethanol production are given by:

$$GHG_{Fe} = \phi_{Fe}^{P} Fe. \tag{2.II.5}$$

Gasoline Production

The emissions from gasoline production include the emissions from both gasoline production and the recovery of crude oil. The emissions factors represent the emissions per unit of gasoline for crude oil refining (ϕ_{Fg}^{P}) and the average emissions per unit of gasoline for crude oil recovery (ϕ_{Fg}^{R}). Total emissions from gasoline production are calculated as:

$$GHG_{Fg} = \left(\phi_{Fg}^{P} + \phi_{Fg}^{R}\right)Fg. \tag{2.II.6}$$

Agriculture

The emissions from cropland agricultural production include the combustion of fossil energy in field operations (GHG_{AG}^{E}), the direct and indirect emissions of N_2O

from cropland (GHG_{AG}^{N2O}), the emissions from the liming of soils (GHG_{AG}^{Lime}), the CO_2 flux resulting from soil organic carbon accumulation and oxidation (GHG_{AG}^{SOC}), and the biomass carbon lost from the conversion of CRP to cropland (GHG_{AG}^{CRP}). The emissions from fossil fuel combustion, N_2O and liming are grouped and referred to as 'Direct' agricultural emissions. The basis for this grouping is that, as currently modeled, these emissions are dependent solely on the allocation of cropland to different crops, rotations and tillage practices. The total emissions from agriculture are:

$$GHG_{AG} = GHG_{AG}^{E} + GHG_{AG}^{N2O} + GHG_{AG}^{Lime} + GHG_{AG}^{SOC} + GHG_{AG}^{CRP}. \quad (2.II.7)$$

Direct Agricultural Emissions

As direct agricultural emissions are calculated based on the allocation of agricultural land (A_{lt}), emissions factors are attributed to each unit of land held in a specific cropping practice. These emissions factors are based on the type and quantity of agricultural inputs, specifically nutrients and energy, used in a given cropping and management practice, as well as the crop yield (for the N_2O calculations).

The agricultural sector of the economic model is driven by per hectare expenditures on four broad input categories including labor, capital, fertilizer and energy per unit of land (equation (2.I.14)). These broad categories are an aggregation of many smaller input categories, based on crop, rotation and tillage specific input use, and benchmark prices. To calculate emissions, we rely on the specific inputs as opposed to the aggregated broad categories. This disaggregation of input categories is necessary to accurately assess emissions as a dollar expenditure on the each of the specific inputs has very different emissions consequences. For example, applying a dollar's worth of nitrogen fertilizer to agricultural soil will lead directly to N_2O emissions, while applying a dollars worth of potassium fertilizer will have little effect on the emissions from agricultural soils. This disaggregation is also necessary because within the broad input categories, there is significant variation in the type and quantity of inputs used by a given cropping and management practice. For example, no-till corn production requires significantly less diesel fuel than conventional corn production, but the use of the other energy types is not likely to be substantially different for these two practices.[26]

For the emissions calculations we focus on the disaggregated energy and fertilizer input requirements to agricultural production. More specifically, the rotation and tillage specific energy input variable (a_{ltE}) is disaggregated into specific quantities of diesel, gasoline, natural gas, liquid petroleum gas (LPG) and electricity. The quantities of the specific fossil fuels used per hectare are denoted a_{ltE_e}, where the subscript e is an index that represents the 5 sources of fossil energy. Likewise, the 'fertilizer' input variable (a_{ltJ}) is disaggregated to specific quantities of nitrogen (N), phosphate (P) and potassium (K) fertilizer, pesticide, lime and other farm inputs.[27] The quantities of the specific farm inputs used per hectare are denoted a_{ltJ_j}, where the subscript j is an index that represents the 6 specific farm inputs for which emissions are calculated.

Energy

Cropping practice specific emissions factors are calculated based on the total units each fossil energy (diesel, gasoline, natural gas, LPG and electricity) used, a_{ltE_e},

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[26] The energy use of various cropping practices is described in Nelson et al. (2009), Schenpf (2004) and West and Marland (2002).

[27] In the emissions calculations, the 'other' input category is used to align the system boundary of agriculture with LCA studies, specifically, Farrell et al (2006), Wang et al. (2007) and Hill et al. (2006). This factor varies by crop and represents emissions from the transportation of inputs and crops and energy used in irrigation.

and emissions per unit energy, ϕ_{E_e}, for the combustion and production of each fuel. These factors per unit of land are:

$$\phi^{E}_{AG_{lt}} = \sum_{e=1}^{5} \phi_{E_e} a_{ltE_e} \qquad (2.II.8)$$

and the total emissions from agricultural energy are:

$$GHG^{E}_{AG} = \sum_{l=1}^{6} \sum_{t=1}^{4} A_{lt} \phi^{E}_{AG_{lt}}. \qquad (2.II.9)$$

The emissions factors for the non-electricity energy inputs consist of two parts, the combustion of the fuel and the production and transportation of the fuel. For each of these fuels, the combustion emissions are dependent on the chemical properties of the fuel, (carbon content, heating value) and the technology used for combustion (stationary engine, tractor, industrial boiler etc). The emissions from producing a fuel are dependent on the energy needed for feedstock recovery (coal mining, crude oil drilling), the conversion of feedstock to the final fuel (crude oil refining), as well as the transportation of both feedstock and final product. The emissions factor for electricity also consists of two parts, the combustion of fossil fuels in power plants, and the recovery and transportation of the power plant's feedstock. The combustion emissions for electricity are dependent on the fuels used and efficiency of the US power plant stock. The feedstock emissions depend on the energy required to recover and transport fuels to the power plants.

N_2O Emissions from Agricultural Soils

N_2O emissions from cropland agriculture are produced through the nitrification and denitrification processes, both of which occur naturally in soils. Nitrification is the process of microbial oxidation of ammonium to nitrate, from which N_2O is a byproduct that leaks from the microbes into the soil and eventually the atmosphere. Denitrification is the anaerobic microbial conversion of nitrate to nitrogen gas where N_2O is an intermediate of the chemical reaction. Both processes are dependent on the available mineral nitrogen (N) in soils, and any practice that changes the N content of soils will have an effect on N_2O emissions (IPCC 2006).

We estimate N_2O emissions using the methods outlined in the IPCC Guidelines for National Greenhouse Gas Inventories (2006). The IPCC methods measure N_2O emissions based on the quantity of nitrogen compounds added to soils and the mineralization of nitrogen in soil organic matter from the cultivation of previously native organic soils. We focus on the emissions from N additions to agricultural soils.

While the amount and type of N additions is a major factor in quantifying N_2O emissions, basing emissions solely on this factor is an oversimplification. Other factors such as land cover, soil type, climate, and management practice are known to influence soil N_2O emissions but are not captured by the IPCC (2006) methods.[28] In addition, the IPCC methods assume that all soil N_2O emissions occur in the same year as the N is added to the soil. It could be the case that N inputs, particularly crop residues, in one year cause increased N_2O emissions in subsequent years.[29]

Nitrogen Additions to Soils

We measure emissions from two sources of N additions to agricultural soil, synthetic N fertilizer (denoted N_{SN} and equivalent to a_{ltJ_N}) and N in crop residue

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[28] In reality, N_2O emissions are dependent on many more variables, such as soil characteristics, weather patterns as well as previous crops and cropping practices. See for example: IPCC (2006), Bouwman et al. (2008), Stehfest and Bouwman (2006), Novoa and Tejeda(2006).
[29] US EPA (2009a); Crutzen et al. (2008).

(N_{CR}).[30] Synthetic N additions include all industrially produced N compounds applied to soils, including anhydrous and aqua ammonia, ammonium nitrate and other nitrogen solutions. The crop residue category is a measure of the nitrogen stored in above-ground and below-ground crop biomass that is reincorporated into the soil after harvest.

While the total application of synthetic N is based on agricultural practice data and varies by rotation and tillage practice, the N in crop residue is modeled as a function of crop yields, which vary by crop in our model. The crop specific N additions from crop residue (N_{CR_k}) are calculated as:[31]

$$N_{CR_k} = DM_k \rho_k^{rn} \left(\pi_k^{ag} N_{CR_k}^{ag} (1 - \rho_k^{rm}) + \pi_k^{bg} N_{CR_k}^{bg} \right) \qquad (2.II.10)$$

where DM_k is the average dry matter yield for crop k per hectare of cropland;[32] $\rho_k^{rn}$is the fraction of crop k that is renewed each year (set to 1 for annual crops); $\pi_k^{ag}$is the ratio of above ground residue dry matter to harvested yield for crop k; $N_{CR_k}^{bg}$is the nitrogen content of above ground biomass for crop k (kgN/kg dry matter); ρ_k^{rm} is the fraction of crop k's above-ground residues that are removed after harvest; π_k^{BG} is the ratio of below-ground biomass to the harvested yield of crop k; and $N_{CR_k}^{bg}$is the nitrogen content of below ground biomass for crop k (kgN/kg dry matter).

[30] The IPCC methods consider N inputs from synthetic and organic fertilizer, manure, sewer sludge and crop residues. In the US the N inputs, and N_2O emissions, from organic fertilizer, sewer sludge are small, relative to the N inputs from N fertilizer and crop residue, so are not considered (US EPA 2009a).
[31] Unlike the IPCC (2006) we do not consider crops that are managed with fire. The EPA (2009a) calculates emissions from agricultural residue burning of only 1.1 $TgCO_2e$ in 2007 (0.9 $TgCO_2e$ of CH_4 and 0.5 $TgCO_2e$ N_2O), so the contribution to the total agricultural emissions estimate would be small.
[32] As yields for each crop are reported in terms of fresh weight (not dry), the yield values must be deflated slightly to calculate DM_k. Consistent with the IPCC we assume that the yield to dry matter ratios are 0.87 for corn, 0.91 for soybeans 0.9 for hay (alfalfa and non-legume hay) and 0.89 for wheat.

The biomass to yield ratios, π_k^{bg} and π_k^{ag}, are calculated based on above-ground non-crop dry matter (DM_k^{ag}), and the ratio of below-ground biomass to above ground biomass, π_k^{ag-bg}. These ratios are given as:

$$\pi_k^{ag} = \frac{DM_k^{ag}}{DM_K} \tag{2.II.11}$$

$$\pi_k^{BG} = \frac{\pi_k^{ag-bg}\left(DM_k^{ag} + DM_k\right)}{DM_K} \tag{2.II.12}$$

where DM_k^{ag} is modeled as a linear function of dry matter yield based on crop specific intercept and slope parameters (α_k and b_k):

$$DM_k^{ag} = \alpha_k + DM_k b_k. \tag{2.II.13}$$

Direct N_2O Emissions

Direct N_2O emissions result from nitrification and denitrification of N additions in the soils where the N is added. The direct emissions measure the response of cropland soils to changes in available N. As more mineral N is made available in cropland, the nitrification and denitrification processes produce more N_2O. It follows that less available N will reduce N_2O emissions from cropland.

The per hectare direct N_2O emissions for a given crop, rotation and tillage practice, $\phi_{AG_{lt}}^{N2O-D}$, are calculated by multiplying total N additions from synthetic fertilizer and crop residue by an emissions factor, ϕ_{AG}^{N-D}, which represents the quantity of direct N_2O emissions per unit of N addition. Given the synthetic N additions per hectare for a given rotation and tillage system ($N_{SN_{lt}}$), and the rotation specific per hectare N addition from crop residue ($N_{CR_l} = N_{CR}{}' \cdot s_{kl}$, where $N_{CR_i} = N_{CR_k}, \forall k = 1 \dots 4$), total direct N_2O emissions per hectare of land in agricultural production are:

$$\phi_{AG_{lt}}^{N2O-D} = \left(N_{SN_{lt}} + N_{CR_l}\right)\phi_{AG}^{N-D}. \tag{2.II.14}$$

Indirect N_2O Emissions

Indirect N_2O emissions occur when N is transported from the field in a non-N_2O form and is converted to N_2O elsewhere. Consistent with the IPCC, we consider two indirect pathways of N_2O emissions: volatilization and leaching and runoff.

The volatilization pathway measures the N that is vaporized from agricultural soils as NH_3 and NO_x and subsequently deposited on other soils or water bodies. In the atmosphere the NH_3 is converted to particulate ammonium, while the NO_x is typically hydrolyzed to form nitric acid.[33] These newly formed reactive nitrogen compounds are eventually deposited back onto soils causing N_2O emissions (US EPA 2008). Following the IPCC (2006) we assume that only synthetic N additions are available for volatilization.

The leaching or runoff pathway quantifies the mineral N additions transported from agricultural soils to water bodies as a result of overland runoff or leaching through soil macropores. A fraction of the mineral N that reaches these water bodies is released as N_2O through aquatic denitrification (US EPA 2008).

The N_2O emissions from volatilization are modeled assuming a fraction (denoted π_{AG}^{N-V}) of synthetic N additions are volatilized and that the volatilized N is converted to N_2O according to a fixed emissions factor (ϕ_{AG}^{N-V}). Likewise, the N_2O emissions from leaching and run off are calculated by assuming a fraction (π_{AG}^{N-LR}) of synthetic and crop residue N additions are transported to water bodies and converted

[33] The conversion of NH_3 to particulate ammonium occurs because the NH_3 combines with nitric or sulfuric acid to form ammonium nitrate and ammonium sulfate aerosol, which is then converted to ammonium (US EPA 2008).

to N_2O according to an emissions factor (ϕ_{AG}^{N-LR}).[34] As such, the per hectare indirect N_2O emissions for a given crop, rotation and tillage practice are:

$$\phi_{AG_{lt}}^{N2O-I} = \left(N_{SN_{lt}}\right)\pi_{AG}^{N-V}\phi_{AG}^{N-V} + \left(N_{SN_{lt}} + N_{CR_l}\right)\pi_{AG}^{N-LR}\phi_{AG}^{N-LR}. \qquad (2.II.15)$$

Finally total N_2O emissions from agricultural production are given as:

$$GHG_{AG}^{N2O} = \sum_{l=1}^{6}\sum_{t=1}^{4} A_{lt}\left(\phi_{AG_{lt}}^{N2O-D} + \phi_{AG_{lt}}^{N2O-I}\right). \qquad (2.II.16)$$

Lime

Lime is added to agricultural soils to reduce acidity. The addition of limestone ($CaCO_3$) or dolomite ($CaMg(CO_3)_2$) to acidic soils can result in CO_2 emissions as the acid soils break down the limestone releasing the stored C.[35] Depending on the soil and climate conditions this breakdown can occur very quickly or over a number of years. Following the IPCC (2006) methodology, we model the emissions from agricultural lime as a function of total lime applied to soils using an emissions factor that maps lime applied to CO_2 emissions (ϕ^{Lime}). Given the per hectare limestone applied for a given rotation and tillage practice, ($a_{ltJ_{lime}}$), the per hectare emissions from liming are:

$$\phi_{AG_{lt}}^{Lime} = a_{ltJ_{Lime}}\phi^{Lime} \qquad (2.II.17)$$

and the total agricultural emissions from lime application are:

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[34] In the IPCC (2006) methods, the share of N inputs that are transported away from the field by leaching or run-off is dependent on climate, soil type and irrigation practice. Specifically, it is assumed that run-off and leaching only occurs on land that is irrigated, or in regions that have a necessarily moist climate (total precipitation less evapotranspiration is greater than the soil's water holding capacity. Consistent with the EPA (2004), we assume that the conditions necessary for leaching and run-off occur on all US cropland.

[35] On weakly acidic soils, the application of lime can actually be a net carbon sink (West and McBride 2005).

$$GHG_{AG}^{Lime} = \sum_{l=1}^{6} \sum_{t=1}^{4} A_{lt} \phi_{AG_{lt}}^{Lime} . \qquad (2.II.18)$$

Soil Organic Carbon

The quantity of organic carbon (SOC) stored in agricultural soils depends on soil type, climate and management factors.[36] This analysis will focus on the soil carbon flux from mineral cropland soils as influenced by management factors.[37] The SOC content of a soil is dependent on the balance between the oxidation and addition of organic carbon (C). This balance influences the CO_2 content of the atmosphere. The accumulation of organic C in soils removes CO_2 from the atmosphere as a portion of the C added to soils is sequestered as SOC instead of being released as CO_2. Likewise, if the organic matter in soils is oxidized, there will be a net flux of organic C from soils, increasing atmospheric CO_2 levels.

As land is held in a certain management practice for longer periods, the level of SOC in the soil reaches an equilibrium state. This means that the soil is no longer accumulating or losing SOC because there is a balance between the amount of organic matter additions and oxidation. Therefore to accurately measure SOC emissions, it is not only necessary to account for the current cropping and management practice and the number of years held in the current practice, but also historic management practices.

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[36] Soils also contain inorganic carbon, but the impact of management practices on these stocks is small and much less well understood (IPCC 2006).

[37] A very small fraction of US agriculture takes place on organic soils (less than 1million hectares compared to the 168 million hectares of mineral soils cultivated (US EPA 2009a)), but draining and cultivating these soils has a very large greenhouse gas emissions impact. The EPA estimates that the cultivation of organic soils leads to SOC losses of 34.8 $TgCO_2e$ (US EPA 2009a). There is no clear link between the cultivation of organic soils and the RFS, so we ignore these emissions.

There are two main components of our estimation of soil carbon, the land allocation model, and the SOC model. The land allocation uses historic data and the results of the economic model to estimate how land moved between different cropping practices. The SOC model estimates how SOC stocks change when a cropping or management practice is used on a particular parcel of land.

Land Allocation Model

The agricultural production sector of the economic model solves for the optimal allocation of cropland, land in CRP, rotations and tillage practices for a given year. As such, the allocation of land in previous years is not considered, and how land moves over time is unknown. To get around this limitation, the land allocation model operates outside the economic model and estimates how land may have moved between cropping practices based on the results of the economic model and historic trends, while tracking specific parcels of land.

The first step in the land allocation is to model historic agricultural land uses from 1980 until the benchmark year of 2003. This allows us to model how certain practices were adopted and how these practices were introduced over time. For example, we are able to estimate how conservation tillage, which began to be adopted in the mid 1980's and the CRP program, which began in 1986, displaced conventional cropping systems. Using the historic cropping patterns, and the results of the economic model, we can construct a time series of the total quantity of land in each land use from 1980 to 2015.

From this time series we estimate how land may have moved between the different management practices each year. We do this in the simplest possible manner. For each practice that expands in area for a given year, we assume that this additional land came proportionately from the practices that decreased in area during that year.[38] As land is shifted amongst the different management practices, all land with the same history is grouped.[39] This procedure generates 'parcels' of land, of various sizes, that have the same land use history. Each year, the number of parcels grows as more land shifts between land uses, increasing the combinations of management practices that could make up a parcel's history. Throughout this process, the history of each parcel is tracked.

SOC Model

The SOC stocks are estimated at the parcel level using methods based on IPCC (2006) recommendations, but adapted for use with our land allocation model. As the history of each parcel is known, it is possible to calculate the current level of SOC, at the parcel level, based on this history, and not just the current land use. This differs from the IPCC (2006) methods which estimate SOC change based on the land use in the inventory year and the land use twenty years prior. Our method also allows us to handle situations where a parcel of land that is still accumulating or losing SOC from a previous management change is converted to another land use.

The change is SOC level is based on the reference soil organic carbon stock ($\overline{SOC}$) and stock change factors (S) which represents how SOC levels will change, compared to a reference SOC level, if soils are managed with a certain activity for twenty years (IPCC 2006). The IPCC methods include three stock change factors, which when multiplied provide the composite stock change factor of a given combination of practices. The first, S_{LU}, measure the influence of land use on SOC, and in our model represents land held as cropland, land held in CRP, and land used for

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[38] This is a simplifying assumption. In reality we would expect farmers are more likely to alter rotations before they adjust tillage systems, due to the capital cost of different tillage equipment.
[39] The 'history' of a given parcel refers to the specific time series of management practices (rotation, tillage, CRP) used and the number of years each practice was used for.

hay production. The second (S_{MG}) accounts for the impacts of tillage practice on SOC, while the third (S_I) accounts for the level of carbon inputs.[40]

The calculations of SOC level occur at the parcel level using a linear function to model SOC accumulation and losses. It is also assumed that any change in SOC resulting from management change will occur in the 20 years following a transition from one practice to another.[41] The SOC stock per hectare, for a given parcel of land, SOC_ρ, is given by:

$$SOC_\rho = SOC_c + \begin{cases} y_c \left(\dfrac{S_\rho \, \overline{SOC} - SOC_c}{20} \right), & y_c \leq 20 \\ S_\rho \, \overline{SOC} - SOC_c, & y_c > 20 \end{cases} \tag{2.II.19}$$

where y_c is the number of years since the last management practice change, S_ρ is the composite stock change factor for the new management practice of parcel ρ and SOC_c is the SOC level of the parcel prior to its conversion.

The annual per hectare change in SOC for parcel ρ, $dSOC_\rho$ is therefore:

$$dSOC_\rho = \begin{cases} \dfrac{S_\rho \overline{SOC} - SOC_c}{20}, & y_c \leq 20 \\ 0, & y_c > 20. \end{cases} \tag{2.II.20}$$

The linear assumption used here is a simplification of the actual SOC response to management or land use changes, as it has been well documented that SOC changes do not follow a linear path (West and Post 2002). For example, after a management change, sequestration rates are expected to be low in the first few years, highest 5 to 10 years after the change, and close to zero 10 to 15 years after the change. Other studies that have come to the same conclusions include Lal et al. (1998) and

[40] Following Ogle et al. (2003) and the US EPA (2004) we assume that all the cropping practices involving corn, soybeans and wheat are classified as 'medium input', such that S_I is effectively removed from our analysis.
[41] This is the IPCC (2006) default assumption which is also used by the US EPA (2004).

Franzlubbers and Arshad (1996). Likewise, in response to a management change, the accumulation of SOC occurs at a much slower rate than SOC losses, especially in shifts from uncultivated land to cropland.[42]

The total soil carbon accumulation or loss in a given year is the sum of total SOC change for all parcels, and is calculated as:

$$GHG_{AG}^{SOC} = \sum_{\rho=1}^{P} A_p dSOC_\rho \qquad (2.II.21)$$

where A_p is the number of hectares in parcel ρ, and P is the total number of parcels used to describe the allocation of cropland in the current year.

CRP Conversion

Converting CRP to cropland impacts emissions in two ways. First, as the land is cleared, there is a significant loss of above and below ground biomass due to burning and decomposition. Second, the soil carbon that accumulated as a result of holding the land in CRP will be released over a number of years (Fargione et al. 2008). As the soil carbon storage of land in CRP is dependent on the type and condition of the land taken out production, SOC storage by CRP land is estimated simultaneously with cropland SOC storage.

To estimate the emissions from lost biomass, we multiply the total quantity of land that was converted from CRP to cropland in a given year (A_{CRP-K}) by a fixed emissions factor (ϕ^{CRP}). Consistent with the EPA (2009b), we assume that all of carbon stored in above and belowground biomass is released as CO_2 in the year CRP was converted to cropland. Therefore ϕ^{CRP} represents the total quantity of carbon

[42] Burke et al. (1995), Ihori et al. (1995), Reeder et al. (1998) and Baer et al. (2000).

stored in the above and below ground biomass of land held in CRP. The total emissions from CRP conversion are:

$$GHG_{AG}^{CRP} = A_{CRP-K}\phi^{CRP}. \tag{2.II.22}$$

<u>Farm Input Production</u>

We measure the emissions from the production and transportation of all farm inputs using an emissions factor approach. The production of fertilizer materials and other farm inputs are emissions intensive because of the fossil fuel energy used to recover feedstock and to produce the fertilizer end products. For example, the production of N fertilizer is typically based on the Haber-Bosch process, which combines hydrogen and nitrogen under high temperature and pressure. Natural gas is used both as a feedstock for this process, as it provides the necessary hydrogen, and to generate the required heat and pressure. Likewise, the production of P fertilizer involves a reaction of phosphoric acid and phosphate rock, both of which are produced using fossil fuels. Phosphoric acid is produced using sulfuric acid and phosphate rock which are both energy intensive. In particular, sulfuric acid is produced using natural gas and crude oil, while there are large fossil energy requirements for the mining and transportation of phosphate rock (Wood and Cowie 2004).

Specifically, this category estimates the emissions from the production, including feedstock recovery, and transportation of the N, P and K fertilizer, pesticide and agricultural lime used to produce corn, soybeans, hay and wheat. Given emissions factors (ϕ_j^J) that map CO_2e emissions to a unit of farm input j produced, and the per hectare use of the specific farm inputs (a_{ltJ_j}), the per hectare emissions from farm input production (ϕ_{lt}^J) are calculated as:

$$\phi_{lt}^{J} = \sum_{j=1}^{6} a_{ltJ_j} \phi_j^J \tag{2.II.23}$$

and the total emissions from farm input production are:

$$GHG_J = \sum_{l=1}^{6} \sum_{t=1}^{4} A_{lt} \phi_{lt}^{J}. \tag{2.II.24}$$

<u>International Land Use Change</u>

By altering international crop prices, reductions in US exports may exacerbate agricultural emissions in the rest of the world as farmers respond to higher crop prices by increasing production.[43] There are two pathways through which production may be increased. The first is through the agricultural intensification, as higher crop prices induce farmers to apply more fertilizer, shift to monocultures or increase irrigation. The second pathway is through the expansion of agriculture onto previously unfarmed land. Without a detailed model of worldwide agriculture we are not able to capture the emissions effects of an intensification of worldwide agriculture. We therefore focus on the land use change pathway.

To calculate the emissions from indirect land use change, we assume that all reductions in US crop exports are replaced by new agricultural production in the rest of the world. This is a simplifying assumption, which provides a rough estimate for the emissions from indirect land use change in the absence of a full international model. The amount of additional cropland (dA_{ROW}) that replaces diverted US exports is estimated using the model's predicted changes in crop exports (dY_{Xk}), and rest-of-

[43] See for example: Searchinger et al. (2008) or Keeney and Hertel (2010).

world yield assumptions (μ_{ROWk}).[44] The total quantity of cropland brought into production given a change in US exports is given as:

$$dA_{ROW} = \sum_{k=1}^{4} \left(\frac{dY_{Xk}}{\mu_{ROWk}}\right). \tag{2.II.25}$$

Carbon losses from land use change include the carbon in above and below ground biomass, changes in soil carbon stocks, non-CO_2 emissions from fire clearing of land and foregone sequestration of regenerating forests (Searchinger et al. (2008) and US EPA (2009b)). Biomass decay and fire clearing causes large CO_2 releases in the first year after conversion, soil carbon losses occur over the next 20 years and lost forest sequestration occurs for approximately 80 years (US EPA 2009b).

To account for the inter-temporal nature of these emissions, we discount the stream of future emissions (ϕ_t^{ILU}), where t is an index of years after the land use change, to calculate the per hectare present value of all future emissions (ϕ_{PV}^{ILU}) from land use change.[45] Given the total number of years for which land use change emissions will occur (T), and a fixed discount rate, r, the present value of these emissions are calculated as:

$$\phi_{PV}^{ILU} = \sum_{t=0}^{T} \frac{\phi_t^{ILU}}{(1+r)^t} \tag{2.II.26}$$

and the total emissions from rest-of-world land use change (dA_{ROW}), in present value are:

$$dGHG_{ILU} = dA_{ROW}\phi_{PV}^{ILU}. \tag{2.II.27}$$

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[44] Hay is not exported, so this index (k), represents corn, soybeans and wheat only.
[45] See for example US EPA (2009c), Stern (2008) or Nordhaus (2007).

World Crude Oil Consumption

Given the per unit emissions from the consumption of crude oil (ϕ^R), the total emissions from rest of the world crude oil consumption (R_W) are given as:

$$GHG_{R_W} = R_W \phi^R. \tag{2.II.28}$$

Chapter 3. Model Calibration

This chapter outlines main data sources and parameter values we use to calibrate the economic and emissions models. The models are calibrated using 2003 as the base year. Section I and Section II describe the calibration procedures for the economic and emissions models respectively. Appendix A provides additional tables.

Section I – Economic Simulation Model

GDP and Value of Fixed Factors

Our estimate for the total GDP in 2003 is $7,667.60 billion, which is derived from the National Income and Product Accounts (NIPA) dataset (BEA 2009). This value represents the sum of compensation to employees plus the consumption of fixed capital. We treat the consumption of fixed capital as the after tax consumption of capital and compute an implied value of the capital endowment of $2,108.52 billion after re-adding the tax component. Subtracting the initial value of land ($22.80 billion) and the implied capital endowment from the above estimate of GDP provides an estimate for the value of the labor endowment in 2003 of $5,490.20 billon.

Elasticities of Demand and Expenditure Shares for VMT and Food

The elasticity of substitution between VMT and non-VMT expenditures (σ_U in equation (2.I.1)) and the elasticity of substitution between food expenditures and the numeraire good, (σ_{UT} in equation (2.I.2)), were jointly determined to imply an own price elasticity of demand for VMT of -0.13, and an own price elasticity of demand for food of -0.08.

Estimates of the elasticity of VMT with respect to the price of VMT vary considerably (from zero to -0.9). Graham and Glaister (2002) conducted a survey and report means for short-run estimates between -0.1 and -0.26 and long run estimates of -0.26 and -0.31. Small and Van Dender (2007) use a pooled cross section of US states for 1966-2001 to estimate a short-run VMT elasticity in the range of -0.022 and -0.085 and long run estimates between -0.11 and -0.34.

Seale et al. (2003) estimate the own price elasticity for a broad consumption group of "food, beverages and tobacco" in the range of -0.075 to -0.098 using data from the International Comparisons Project.

The share of food expenditures to total consumption expenditure (0.14) was computed using the NIPA dataset (BEA 2009). The share of VMT expenditures to total consumption expenditures is 0.07. This is calculated using benchmark quantities of VMT and blended fuel, the fuel price, and the ratio of fuel costs per unit of driving to the total cost per unit of driving. The remaining expenditures represent expenditures on the numeraire good.

Household Production of VMT

Total vehicle miles traveled (VMT) were computed using data from the Federal Highway Administration's (FHWA) Highway Statistics Dataset (Federal Highway Administration, 2003). We consider the annual total fuel consumption for 'passenger cars and other two-axle, four tire vehicles (PC24) and divide by the average fuel economy of this class of vehicles in 2003. This calculation yields a benchmark quantity of 1.6 trillion kilometers.

The initial fuel economy was 8.7 kilometers per liter (FHWA 2003), which is consistent with Parry and Small (2005).

The elasticity of substitution between fuel and non-fuel expenditures on driving, σ_M, in (2.I.4) was selected to imply an own-price elasticity of demand for blended fuel of -0.47. Pre-1990 estimates of the long-run elasticity of fuel demand with respect to the price of fuel vary between -0.5 and -1.10.[46] The value adopted in Parry and Small (2005) is -0.55.

The ratio of fuel costs to total cost per unit of driving is commonly computed by taking the ratio of the long-run elasticity of VMT with respect to the price of fuel and the long-run elasticity of fuel demand with respect to the price of fuel. Following Parry and Small (2005), we adopt a ratio of fuel costs to total costs of 0.4. This value is also consistent with estimates presented in Johansson and Schipper (1997) and US Department of Energy (1996).

Agricultural Production

The agricultural dataset was compiled from a number of sources. The data includes the benchmark allocation of cropland to corn, soybeans, hay, wheat and the CRP, the distribution of rotations and tillage practices, average yields and input use and cost of production data at the tillage level. The key elasticity parameters in the agricultural sector were calibrated to reflect literature estimates. The quantity of land planted to each crop is gathered from the National Agricultural Statistics Service (NASS). The Agricultural Resource Management Survey (ARMS) provides the estimates for input use and land allocation at the rotation and tillage level for the years 1996-2005 (ERS 2008a). The quantity of land allocated to the Conservation Reserve Program was collected from the Farm Service Agency (FSA 2004). The ARMS data is also the basis for our tillage level input use estimates. We supplemented the ARMS data with the USDA's Commodity, Costs and Returns (CCR) dataset (2009), the Regional Environment and Agricultural Programming Model (REAP) dataset (R.

[46] See Graham and Glaister (2002) for a survey, or Dahl and Sterner (1991), Goodwin (1992), Espey (1996) or (1998).

Johansson, Peters, and House 2007) and several academic studies. Benchmark yield data was derived from the Agricultural Statistics Database (ASD) (NASS 2009).

Land Allocation

Crops and CRP

In 2003, we consider 107.8 million hectares of cropland allocated to four crops.[47] Corn was the dominant crop in terms of land area, taking up 31.37 million hectares of cropland, followed by soybeans, hay and wheat, which accounted for 29.33, 25.65 and 21.47 million hectares respectively. The USDA includes many different crops, both legumes and non-legumes, in the definition of hay. From NASS (2003) data, it is given that 37% percent of total hay production was alfalfa, while the remaining is classified as other, which we assume is 50% other legume hay (clover or vetch) and 50% grasses.

The crops we consider account for the vast majority of U.S. agriculture both in terms of area and economic value. In each year since 1980, these four crops have made up at least 80% of total crops harvested and made up 87% of crops harvested in 2003 (NASS 2003). In terms of the economic value of production, the value of corn produced amounted to $24.5 billion, soybean to $18.0 billion, hay to $12.0 billion, and wheat to $7.9 billion according to the USDA (NASS 2003). Out of a total production value for field crops of $82.3 billion, corn, soybeans, hay and wheat represent 76.0%.

There are a number of different CRP programs, each with different contract lengths, payment levels and enrollment qualifications. Two of these programs, the Conservation Reserve Enhancement Program (CREP) and the Farmable Wetland Program (FWP) target specific environmental objectives and offer higher rental rates

[47] The specific NASS data used is land that was harvested.

making this land unlikely to be converted to cropland.[48] We therefore assume that only land in the remaining two major programs, general sign-up and continuous non-CREP, will be available for conversion to cropland. In 2003, land in general sign-up totaled 12.78 million hectares, and the land in non-CREP continuous signup totaled 0.76 million hectares (FSA 2004). Our estimate of total CRP land in 2003 is therefore 13.54 million hectares.

Rotations and Tillage Practices

As discusses in Chapter 2, the model consider six crop rotations: continuous corn (CC), continuous soybeans (CS), continuous hay (HH) and continuous wheat (WW), corn-soybeans (CS) and corn-soybeans-wheat (CSW).[49] Each of the multi-crop rotations is constructed using the previous crop cross-tabulation of the ARMS and were selected to cover the majority of the land in each crop. In 2003, these rotation definitions accounted for 94% of total land producing corn and 93% of total land producing soybeans, but only 71% of land producing wheat.

We follow the USDA definitions (Table 3-11) to construct four tillage practices: conventional (CT), reduced (RT), mulch (MT), no-till (NT).[50] The 2003 share of each tillage practice within a given rotation is derived from the ARMS. To calculate the final values for land planted to each crop, rotation and tillage, the shares of each rotation and tillage practice was calculated using the ARMS data, and these shares were imposed on the NASS crop level data.[51]

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[48] These assumptions do not bias our baseline, as the CREP and FWP contain very little land compared to the general sign-up with only 0.20 and 0.34 million hectares respectively.
[49] In the ARMS, there is not a wheat previous crops planted category. Therefore rotations containing wheat are constructed based on 'small grains' as the previous crop harvested.
[50] The USDA defines 5 tillage practices which are also used in the ARMS. As the land being managed in ridge tillage is a small share of total production, this category is combined with the mulch tillage.
[51] As the ARMS survey does not cover hay, the tillage shares from wheat production are used.

The shares of land managed with certain tillage systems differ significantly by rotation systems (Table 3-12). For example, almost 40% of land in a continuous wheat rotation and continuous soybean rotation is managed with no-till methods. This contrasts with only 6% and 26% of land in continuous corn or corn-soybeans rotation managed with no-till respectively. Residue build up is the main factor that restricts the use of no-till with continuous corn (Ohio State University Extension (2007); Dulker and Myers (2009)). The increased residue results in increased risk of disease, cooler soils, interference with planters, and increased yield drag.

Input Use and Expenditures

As shown in (2.I.14) the agricultural model operates using per hectare expenditures on energy, labor, capital and fertilizer, which vary by rotation and tillage practice (a_{ltm}). We construct estimates for energy use, and therefore expenditures from literature sources. Expenditures on labor and capital are collected at the crop level from USDA cost of production data and variation at the tillage level is imposed using other data sources. Finally, expenditures on fertilizer are estimated using tillage level input use data for fertilizer, pesticide and seed, as well as crop level expenditure data for additional categories from the USDA. The final aggregated expenditure data is reported in Table 3-3.

Energy

The ARMS survey and the USDA Commodities Cost and Returns do not report data that allows for a direct calculation of energy use by rotation and tillage. Therefore, energy use is constructed from literature sources, with five primary energy sources considered. The estimated use of gasoline, natural gas (NG), liquefied petroleum gas (LPG) and electricity varies at the crop level, while estimated diesel use is tillage specific (Schenpf (2004); Gregory, Shea and Bakko (2005); Rathke et al.

(2007); Nelson et al. (2009)). NG and LPG primarily used to run crop dryers, electricity is used to run irrigation equipment and to light, heat and cool barns and houses while gasoline fuels the smaller farm vehicles, cars and pickup trucks, and equipment (Schenpf 2004; USDA 2008). These operations are independent of tillage choice, but highly dependent on the choice of crop. The values for gasoline, NG, LPG and electricity represent estimates from literature that assesses the lifecycle energy use of various crops. The production of corn has much higher energy requirements, for each fuel we consider, than the production of soybeans and wheat (Table 3-1).

Table 3-1. Non-Diesel Energy Requirements by Crop (MJ/hectare)

	Gasoline	Natural Gas	LPG	Electricity	Source
Corn	1277.7	670.1	765.3	819.7	Farrell et al. (2006)
Soybeans	394.5	135.4	87.5	249.7	Hill et al. (2006)
Wheat	302.3	0.0	66.5	133.4	Piringer and Steinberg (2006)

Hay is assumed to have the same energy requirements as wheat

Diesel fuel is used in the farm equipment (tractors, combines, balers etc) as well as large trucks used for the transportation of input factors and final products (USDA 2008). The use of large farm equipment is dependent on the choice of tillage practice as this equipment is used to prepare fields, plant and harvest crops and apply chemicals. For example, conventional tillage entails running a moldboard plow over the field. Under no-till management the moldboard plow is not used, and other operations (such as a disking or cultivation operations) are avoided. This decreases the diesel fuel requirements of a no-till operation significantly compared to a conventional tillage operation.

We estimate diesel use based on literature estimates of the emissions consequences of altering tillage practices (West and Marland 2002). These studies are based on the number, and type, of field operations used in specific tillage systems and the energy requirements of each operation (Table 3-2). For plowing, disking, planting and cultivation operations, the energy requirements are based on whether a crop-tillage combination will undertake a given operation. For instance, a field producing conventional, reduced and mulch tilled corn would be cultivated, but a field producing corn under no-till (or a field producing soybeans, wheat or hay) would not be cultivated, saving 168 MJ/ha of diesel. Similarly, all non-corn crops are assumed to require no cultivation. The application of fertilizers and pesticide will also require diesel fuel. The total diesel used for these operations is based on the estimated energy use of the operation and the percent of land in each rotation and tillage combination that receives a given treatment (ERS 2008a).

The total expenditure on diesel and non-diesel energy by rotation and tillage practice are reported in Table 3-3.

Table 3-2. Diesel Energy Requirements by Tillage Practice

		Conventional		Reduced/Mulch		No-Till	
Operation	MJ/ha	Corn	Other	Corn	Other	Corn	Other
Moldboard Plow	1122	1122	1122	-		-	
Disk	345	690	690	690	690	-	
Planting	254	254	254	254	254	254	254
Cultivation	168	168	-	168	-	-	
Harvest Combine	506	506	506	506	506	506	506
Fertilizer Application	574			based on ARMS values			
Pesticide Application	63			based on ARMS values			
Total		2740	2572	1618	1450	760	760

Values in this table are derived from West and Marland (2002)
Diesel requirements for fertilizer (and pesticide) application are calculated for each rotation-tillage combination using ARMS data for the percent of land receiving any fertilizer (or pesticide).
Other crops are soybeans, wheat and hay
Disking is counted twice to represent two passes over the field

Capital

Agricultural expenditure on capital is based on the USDA's Commodity Cost and Returns data and includes the interest on operating capital and the capital recovery of machinery and equipment. This data is estimated at the national level, so does not include tillage level variation. Capital costs decrease with tillage intensity because equipment purchases can be avoided and because the equipment required are cheaper (a smaller or less powerful tractor). We allow for variation in capital expenditure following the REAP (R. Johansson, Peters, and House 2007) dataset. As the estimated variation in REAP is consistent across regions and crops, for each rotation, we allow the capital costs for mulch tillage to be 5% higher, reduced tillage to be 10% higher and conventional tillage to be 20% higher than the capital costs of no-till. The expenditure on capital is reported for each rotation and tillage practice in Table 3-3.

Labor

The expenditure on labor data is derived from the USDA's Commodity Costs and Returns (CCR) dataset (ERS 2009) and includes wages paid to workers, and the opportunity costs of unpaid workers. While the CCR data varies by crop, consistent with REAP (R. Johansson, Peters, and House 2007) we impose variation at the tillage level following energy consumption. There is likely to be a decrease in labor costs with decreases in tillage intensity, as the number of field operations, and hence diesel fuel consumption, is reduced. With fewer passes over the field, fewer operator hours are required. As the variation in energy costs is driven totally by the amount of diesel fuel required for machinery operations, we allow labor costs follow the same pattern in percentage terms (Table 3-3).[52]

Fertilizer and Other Agricultural Inputs

We construct the expenditure on fertilizer using tillage level data for the application rates of N, P and K fertilizer, seed and pesticides at the tillage level and national prices as well as national level expenditure estimates for some additional input categories.

[52] This is consistent with the REAP dataset. For example, in REAP, a corn-soybean rotation in the Corn Belt under no-till has 46% of the energy expenditure and 45% of the labor expenditure of conventional tillage.

For corn, soybeans and wheat, ARMS data was collected for all available states between the years 1996 and 2005.[53] To generate our input use data at the rotation and tillage level, we aggregated the state data to the national level, while controlling for regional variation. To control for temporal variation we averaged the input use data across the available survey years. In addition, certain relationships are imposed at the rotation level.[54] Fertilization rates for hay were collected from extension reports from universities in major hay producing regions, and represent the central value application levels for N, P and K fertilizer.[55] Variation is then added to the hay data by assuming that the percent difference from the rotation mean of each tillage system is consistent with continuous wheat. The application of pesticide to hay was also assumed to be the same as continuous wheat. The application rates of the specific inputs are reported in Table 3-14.

The total expenditure on each fertilizer was calculated using national average prices (NASS 2003).[56] For expenditures on pesticide and seed, the rotation and tillage variability in input usage, in terms of percentage from the crop mean, estimated from the ARMS data is imposed on the crop level national average expenditure data from the Commodities Cost and Returns dataset (ERS 2009).

Our fertilizer expenditure variable includes the variable costs of production that are not included in the capital, labor or energy categories. The values vary only at

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[53] Not each state or crop is survey each year. Table 3-13 summarizes the ARMS surveys collected.
[54] Specifically, corn in a CS rotation receives 44 kg/ha less nitrogen fertilizer than continuous corn, while corn and wheat planted in a CSW rotation receive 34 kg/ha and 22 kg/ha less nitrogen than a continuous corn and continuous wheat rotation respectively. These adjustments reflect the 'soybeans nitrogen credit' (Vanotti and Bundy 1995; Lory, Russelle, and Peterson 1995; Hesterman et al. 1987; Gentry et al. 2001).
[55] Extension recommendations were collected from North Dakota State, University of Wisconsin, Kansas State, South Dakota State, Pennsylvania State, Auburn, and Louisiana State. The recommendations are based on a 'medium' or 'optimal' soil test. Non-alfalfa legume hay consists of clover and vetch while non-legume hay consists of various grasses (brome, fescue, canary, etc).
[56] We calculate the national average price of fertilizers in 2003from (NASS 2006). These were 0.63 $/kg, 0.31 $/kg and 0.40 $/kg for N, P and K respectively.

the crop level as they were collected from the USDA's Commodities Cost and Returns dataset (ERS 2009). The additional categories included are: soil conditioners, manure, custom operations, repairs, purchased irrigation water, taxes and insurance and general farm overhead.

Crop Yields

The benchmark crop yields for corn, soybeans, hay and wheat are 8.9, 2.6, 5.6 and 3.0 metric tons per hectare respectively. These are US average yields for 2003 from the Agricultural Statistics Database (NASS 2009).

Crop Prices and CRP Rental Rate

The benchmark crop prices are from the USDA's Agricultural Prices Summary (NASS 2006). The price per metric ton of corn, soybeans, hay and wheat are \$95.2, \$269.6, \$94.3 and \$118.6 respectively. The CRP rental rate is 114.48 \$/hectare and is calculated as the weighted average annual rental payment to CRP in 2003 (FSA 2004). The average payment to general sign-up and non-CREP continuous sign-up CRP were 108.16 \$/hectare and 224.16 \$/hectare respectively, with general sign-up making up 94% of total CRP land. These payments include the annual soil rental payments, maintenance allowances and annual incentive payments.

Table 3-3. Agricultural Expenditures by Rotation and Tillage System

	Input Expenditures ($/hectare)					Fertilizer Components ($/hectare)					
	Labor	Capital	Energy	Fertilizer	Total	N	P	K	Seed	Pesticide	Other
Continuous Corn	76.13	144.41	58.96	398.07	677.57	106.73	21.67	18.58	84.54	59.50	107.05
Conventional	84.76	154.79	64.87	392.53	696.95	103.59	20.91	18.53	84.61	57.85	107.05
Reduced	71.04	141.89	55.50	397.92	666.35	108.80	23.45	17.54	83.52	57.55	107.05
Mulch	71.04	135.44	55.50	406.72	668.70	110.95	22.44	19.05	85.30	61.93	107.05
No-Till	60.47	128.99	48.29	393.59	631.34	102.03	19.08	19.27	83.28	62.89	107.05
Continuous Soybean	48.88	111.67	23.60	214.24	398.39	2.59	5.49	8.43	68.35	44.78	84.63
Conventional	64.99	121.95	30.69	211.45	429.08	2.30	4.55	7.71	68.99	43.27	84.63
Reduced	45.94	111.77	22.31	213.48	393.50	2.50	6.94	9.64	66.32	43.44	84.63
Mulch	45.32	106.70	22.04	212.09	386.16	2.50	5.34	7.71	68.72	43.19	84.63
No-Till	33.38	101.61	16.78	218.44	370.22	2.97	5.86	8.82	68.55	47.62	84.63
Corn Soybean	58.54	124.37	38.92	296.43	518.26	43.61	13.32	13.47	76.26	53.94	95.85
Conventional	74.92	138.38	47.79	292.43	553.50	43.54	13.52	13.25	74.92	51.35	95.85
Reduced	58.14	126.84	38.75	297.84	521.55	45.37	14.09	13.49	76.26	52.78	95.85
Mulch	58.17	121.08	38.75	295.12	513.12	43.27	13.07	13.99	76.92	52.02	95.85
No-Till	46.98	115.30	32.57	299.64	494.46	42.40	12.70	13.00	76.43	59.23	95.85
Corn Soybean Wheat	53.38	125.16	33.90	245.03	457.45	41.37	12.26	9.34	56.56	36.87	88.66
Conventional	71.29	139.30	43.47	241.99	496.05	41.07	12.53	9.19	55.77	34.74	88.66
Reduced	53.50	127.68	34.00	245.82	461.03	42.73	12.58	9.56	56.66	35.66	88.66
Mulch	53.55	121.87	34.03	244.34	453.79	41.51	12.13	9.76	57.11	35.16	88.66
No-Till	42.03	116.07	27.80	246.71	432.61	40.70	11.91	9.12	56.79	39.54	88.66
Continuous Hay	49.08	130.13	27.06	153.26	359.52	20.11	15.20	7.69	18.78	17.15	74.31
Conventional	62.39	140.04	33.98	153.83	390.21	21.05	15.72	6.94	18.78	17.05	74.31
Reduced	43.12	128.35	23.97	151.23	346.69	19.55	14.04	8.65	18.78	15.91	74.31
Mulch	43.22	122.52	24.02	151.63	341.36	20.31	14.75	6.99	18.78	16.46	74.31
No-Till	31.38	116.68	17.87	157.68	323.61	18.31	16.56	9.02	18.78	20.68	74.31
Continuous Wheat	50.51	131.14	27.80	174.01	383.46	47.69	11.27	2.64	18.78	19.30	74.31
Conventional	64.42	141.12	35.02	176.21	416.77	49.89	11.66	2.40	18.78	19.18	74.31
Reduced	44.31	129.36	24.59	170.73	368.96	46.33	10.40	2.99	18.78	17.89	74.31
Mulch	44.11	123.48	24.49	173.15	365.20	48.19	10.95	2.42	18.78	18.53	74.31
No-Till	32.37	117.60	18.41	175.18	343.56	43.42	12.28	3.11	18.78	23.28	74.31

The ‘other’ category includes: soil conditioners, manure, custom operations, repairs, irrigation water, taxes and insurance, and other overhead

Elasticities of Crop Supply

We calibrate the parameters δ_k, σ_k and σ_{kp} in equations (2.I.7) to such that the own-price elasticity of corn supply is 0.25 and the cross price elasticities of soybeans, wheat and hay with respect to the price of corn are -0.13, -0.09 and -0.05, respectively. Gardner (2007) reports an elasticity of corn supply of 0.23, which is consistent with our study. De Gorter and Just (2009b) use a value somewhat higher for this elasticity (0.4).

Lin et al. (2000) report several estimates of crop allocation response. They report weighted national averages of own and cross price elasticities for corn covering the years 1991-1995 and calculate an own-price elasticity of corn of 0.29. They also report cross-price elasticity of soybeans with respect to the price of corn of -0.23 and the cross price elasticity of wheat with respect to the price of corn of -0.046. Our wheat elasticity of -0.09 is somewhat higher than the estimate suggested by Lin et al. (2000), which is a result of the aggregation of our dataset, which aims to be nationally representative.[57]

We assume that the cross-price elasticity of CRP with respect to corn is -0.03, which may be an extremely conservative value. Featherstone and Goodwin (1993) use a two-stage least squares Tobit model to estimate conservation investment as a function of various attributes. They find an elasticity of conservation investment with respect to crop efficiency (which is defined as crop production/total variable costs) between -0.95 and -0.13. Since the share of CRP investment relative to the USDA's total investment in conservation is 41.2% (Claassen 2006), these elasticity values become -0.39 and -0.055 respectively.

[57] Lin et al. (2000) use a regional model to generate their elasticity estimates, whereas we consider a model that considers only a single region. Since wheat is produced predominantly in regions that are not large producers of corn and soybeans, their elasticity estimates are smaller than ours.

Tillage Elasticities of Substitution

Following the USDA's REAP model (Johansson, Peters and House, 2007), the elasticities of substitution between the four tillage systems, σ_l, in equation (2.I.13) are set to 10.

Ethanol Production

The baseline quantity of ethanol produced in 2003 is 10.4 billion liters. This value is the total ethanol used by passenger vehicles as reported by the Highway Statistics Dataset (FHWA 2003). We calibrate the λ's in (2.I.15) to represent the national average corn, energy, labor and capital requirements and co-products generated per unit of ethanol produced. These values are reported in Table 3-15 and discussed below.

We consider four ethanol production technologies, which are combinations of conversion technology (wet or dry milling) and fuel source (natural gas or coal). Wet milling and dry milling are inherently different technologies, produce different co-products and have different corn and energy requirements.[58] Our calibration of (2.I.15) reflects a technological make up of the ethanol sector that follows GREET 1.8c (Wang 2009). In the benchmark, we assume that dry mills fired by natural gas and coal account for 57% and 18% of total ethanol production respectively. Wet mills fired by natural gas account for 15% of total production and wet mills fired by coal make up the remaining 10%.

Following GREET 1.8c, we assume that dry mills require 2.53 kilograms corn per liter and that wet mills requires 2.63 kg of corn per liter. Therefore we calibrate

[58] Dry milling differs from wet milling in that the wet milling process separates the corn-starch from the other parts of the corn kernel before the fermentation process. Once the starch has been separated, it is possible to create a number of products, such as ethanol, corn syrup or corn starch. The dry milling process does not separate the corn starch from the rest of the kernel and can produce only ethanol and certain co-products.

$\lambda_{Y_{k1}}$ to be the average conversion efficiency of wet and dry mills of 2.56 kg/l. Likewise, following (Farrell et al. 2006), we assume that to produce a liter of ethanol, dry mills require 13.13 MJ of primary fuel and wet mills require 14.59 MJ of primary fuel. It follows that we calibrate λ_E to reflect the average energy use of 13.49 MJ/liter.

It is also assumed that wet and dry mill ethanol plants have the same capital and labor costs per liter of ethanol. Following (Shapouri and Gallagher 2005) we calibrate λ_K to reflect a capital expenditure of 0.005 $/liter and λ_L to reflect a labor expenditure of 0.07 $/liter. These values are consistent with the capital and labor expenditures reported by McAloon et al. (2000).

Co-Products

As previously discussed, a number of products are jointly produced with ethanol and the type produced varies by ethanol conversion technology. Following GREET 1.8c, we assume that dry mills produce only distillers' grains with solubles (DGS), while wet mills produce corn gluten meal (CGM), corn gluten feed (CGF) and corn oil. The quantity of each co-product produced per liter of ethanol is derived from GREET 1.8c and reported in Table 3-15.

The co-products of ethanol production are consumed in the food sector. Consistent with the FASOM and FAPRI models and the US EPA (2009b), we assume a kilogram of DGS displaces 0.95 kilograms of corn and 0.05 kilograms of soybeans. Following GREET 1.8c, a kilogram of CGF displaces 1.53 kilograms of corn and a kilogram of CGM displaces 1.0 kilograms of corn. We allow corn oil to displace corn based on its economic value in 2003, such that $1 of corn oil displaces $1 of corn. We use this method because corn oil is utilized for much more than just an animal feed, and therefore the typical displacement ratio methods used are not reflected in the prices of the two products (Shapouri and Gallagher 2005).

Crude Oil Imports

A central value from reviewed literature of 0.5 is used for η_R in equation (2.I.28).

Krichene (2002) reports a mean long-run rest-of-world supply elasticity of 0.044 and short-run estimates of 0.25, 1.10, and 0.10 for the periods 1918-1999, 1918-1973, and 1973-1999 using an error correction model with co-integration. He also reports short-run estimates using 2 stage least squares for these periods of -0.08, -0.08, and -0.07, respectively. Krichene (2006) reports estimates of long-run supply for the period 1970-2005 of 0.007 to 0.08, again using an error correction model with co-integration. He also provides short-run estimates over this period between 0.005 and -0.03 using 2 staged least squares.

The OECD Economic Outlook uses a supply elasticity of 0.04 (Brook et al. 2004). Huntington (1991) found a range of short-run elasticity estimates from 0.0 to 0.137, with an average or 0.052. They also found long-run elasticities of oil supply ranged from 0.162 to 0.662, with an average of 0.39. Porter (1992) calculates an implied long-run supply elasticity of 0.29. Greene and Tishchishyna (2002) use short-run estimates between 0.028 and 0.049. Bohringer and Rutherford (2002) use values of 0.5 to 2.0. Alhajji and Huettner (2000) have computed a non-US world supply elasticity of 0.21.

Rest-of-World Agricultural Sector

US Export Demand Elasticities

The crop export demand elasticities (η_{ROW_k}), in equations (2.I.27), are set to -0.65, -0.60 and -0.55 for corn, soybeans and wheat respectively, which represent a central value of reported values in the literature. Gardiner and Dixit (1987) report estimates for corn export demand elasticities between –0.47 and –0.16, with a mean value of –0.27. Haniotis et al. (1988) report a value of –1.73, while de Gorter and Just

(2009a) use a value of –0.10. For soybeans, Gardiner and Dixit (1988) report estimates between –2.00 and –0.14, with a mean value of –0.96, while Haniotis et al. (1988) report a value of –0.60. Gardiner and Dixit report estimates wheat export demand elasticities between –3.13 and –0.15, with a mean value of –0.60. Haniotis et al. report a value of 0.74.

Export Shares

The shares of US crops exported are calculated from the USDA's Foreign Agricultural Service (FAS 2009) and historical data reported in USDA's Long Term Projections (2009). In the benchmark, 19%, 14%, and 49% of the corn, soybeans and wheat produced in the US is exported.

International Crude Oil Market

Benchmark world crude oil consumption over time is derived from the EIA's International Energy Statistics dataset (EIA 2009b). In 2003, non-US petroleum consumption was 59,627 million barrels per day (mbd) which is the difference between world consumption (79,660 mbd) and US consumption (20,033 mbd).

For the rest-of-world own price elasticity of demand for crude oil, we use a central value of literature estimates of -0.3. It is generally found that energy demand is relatively price inelastic, and this relationship is also found in own-price demand elasticity for crude oil. Cooper (2003) estimates the short and long run elasticities of demand for crude oil for 23 countries using data from 1971 to 2000. He finds short run demand elasticities between 0.001 and -0.109 and long run elasticities of 0.005 and -0.453. Fattouh (2007) reviews several studies and finds short and long run elasticities range between 0 and -0.64. He concludes that the price of crude has little effect on crude demand in the short run, but the elasticity becomes larger in the long run as there is a higher possibility of substitution and conservation. Krichene (2005) estimates a system of equations model of the world crude oil and natural gas markets. Using an error correction estimation method for the years 1974 to 2004 he estimates a long run price elasticity of -0.12, while a co-integration method produces estimates for the long run price elasticity of -0.26.

<u>Government</u>

In 2003, the average per-liter combined federal and average state fuel tax was 0.10 $/liter as reported by the Federal Highway Administration (FHWA 2003). The Volumetric Ethanol Excise Tax Credit (VEETC) was 0.14 $/liter ethanol (Wright et al. 2006). Therefore, the benchmark fuel tax revenue is $49.08 billion and the government dispersed $1.43 billion to ethanol producers. Given an average CRP rental rate of $114.48 per hectare (FSA 2004), in 2003 we compute total government expenditure on the CRP land of $1.55 billion.

Our estimate of total government expenditures in 2003 of $2.828.90 billion includes the sum of current tax receipts plus contributions for government social insurance from the BEA's NIPA dataset (BEA 2009).

Section II – Emissions Model

<u>Transportation</u>

The calibration of the transportation sector of the emissions model, equation (2.II.2), is based on data Argonne National Laboratory's Greenhouse Gases, Regulated Emissions and Energy Use in Transportation (GREET) model (Wang 2009) and US EPA's Inventory of US Greenhouse Gas Emissions and Sinks (US EPA 2009a).

Gasoline

We assume the combustion of gasoline (ϕ_{Fg}^{C}) causes the emission of 2.47 $kgCO_2e/l$. The majority (2.35 $kgCO_2e$) of this emission is from the conversion of the C stored in gasoline to CO_2.[59] Gasoline does not have a specific chemical formula because as it consists of a number of different compounds. In addition, the components of gasoline vary by grade, region and season. For simplification we assume that all gasoline consumed in the US is conventional gasoline, with a density of 0.75 kg/l and carbon content by weight of 86.3% (Wang 2009).

The combustion of gasoline also releases non-trivial amounts CH_4, and N_2O. These emissions are dependent on the characteristics of gasoline, the air-to-fuel mix, the combustion temperature and the control technologies used in passenger vehicles.[60] Although it is likely that there will be improvements in N_2O mitigation technologies, we assume that N_2O and CH_4 control technologies remain at current levels. As a result of the changing technology, the estimates of N_2O emissions per liter gasoline combusted can vary. For example, GREET 1.8c (2009) estimates CH_4 emissions of 0.003 $kgCO_2e/l$ and N_2O emissions of 0.02 $kgCO_2e/l$ in 2003. The emissions factors implied by the EPA in 2003 are 0.004 $kgCO_2e/l$ for CH_4 and 0.06 $kgCO_2e/l$ for N_2O. We use the emissions coefficients implied by the EPA inventory because this analysis is more sophisticated than the methods used by GREET, and should therefore provide a more accurate assessment of emissions.

[59] The standard assumption is that all C stored in gasoline is released as CO_2 upon combustion.
[60] N_2O emissions from passenger vehicles increased dramatically from 1990 to 1998 because technologies used to control CO, NO_x, VOC and CH_4 emissions exacerbated N_2O emissions. Since 1998, N_2O emissions from passenger vehicles have been dropping because the control technologies have been altered to reduce N_2O as well (US EPA 2009a).

Ethanol

The combustion of ethanol (ϕ_{Fe}^{C}) results in the release of 1.62 $kgCO_2e/l$. The majority of this emission (1.51 $kgCO_2e/l$) results from the conversion of the carbon stored in ethanol to CO_2. The CO_2 emissions are calculated based on the density (0.79 kg/l), and carbon content (52.2%) of ethanol.[61] The CH_4 and N_2O emissions that result from the combustion of ethanol are modeled in GREET 1.8c as a share of the respective emissions for a liter of gasoline and we follow this method. In 2003, it is assumed that the CH_4 emissions from a vehicle burning ethanol are 92% of those of a vehicle burning gasoline, while the N_2O emissions of the two fuels are equal. This calculation leads to CH_4 and N_2O emissions from the combustion of ethanol of 0.003 $kgCO_2e/l$ and 0.06 $kgCO_2e/l$ respectively.

Following Searchinger et al. (2008) and the GREET 1.8c we calculate the ethanol combustion credit ($\phi_{Fe}^{CO_2}$) to be the quantity of CO_2 released during the combustion of ethanol. As discussed above, this value is 1.51 $kgCO_2e/l$.

<u>Fuel Production</u>

The fuel production emissions model, equations (2.II.4) and (2.II.6), is calibrated using the GREET 1.8c model. Many prominent studies have based emissions estimates on GREET including Farrell et al. (2006), Searchinger et al. (2008) and US EPA (2009b).

Gasoline

The emissions for the production of conventional gasoline (ϕ_{Fg}^{P}) are assumed to be 0.41 $kgCO_2e$/liter, which is based on the default assumptions in GREET 1.8c. It is assumed that the refining of gasoline is 87.7% efficient in terms of energy inputs

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[61] Both of these factors are based on the chemical formula of ethanol, C_2H_6O.

and outputs, and that the energy requirements of a refinery are met using still gas (50%), natural gas (30%), coal (13%), electricity (4%) and residual oil (3%).[62,63]

We estimate the emissions factor for crude oil recovery (ϕ_{Fg}^{R}) to 0.22 $kgCO_2e/l$ gasoline using GREET 1.8c. The GREET 1.8c default assumptions are used for the shares of crude oil that come from traditional and alternative sources. We assumed that 92.4% of crude oil used for US gasoline production is from traditional crude oil sources, while 9.4% is from oil sands (with 57.2% of oil sands produced using surface mining and 42.8% from in situ production).[64]

Ethanol

We calculate the emissions per liter ethanol produced using GREET 1.8c, using default assumptions. For wet mills, it is assumed that the primary fuel (natural gas or coal) meets the entire energy requirement of the plant, while dry mills are assumed to use electricity to meet 9% of their energy requirements.[65] We also assume that 50% of the natural gas used in an ethanol plant is combusted in large industrial boilers and 50% is combusted in small industrial boilers while all coal is combusted in industrial boilers.

The technology shares of ethanol production, $s^{h,n}$, and the per liter emissions for each ethanol plant technology ($\phi_{Fe}^{h,n}$) are reported in Table 3-4. The majority (75%) of ethanol production occurs in dry mills, and natural gas is the dominant fuel

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[62] Many of the processes modeled in GREET are based on these energy efficiency assumptions. The energy efficiency parameter is the ratio of energy outputs to energy inputs for a given process. In the case of oil refining, an 87.7% energy efficiency parameter implies that for each energy unit of gasoline produced, 14% more energy is required as input (1/.877 - 1).

[63] These energy share assumptions are identical to those of the US EPA (2009b).

[64] The share of alternative sources of crude oil in the US crude supply are expected to rise over time (Wang et al. (2009); Farrell and Brandt (2006)), which would lead to an increase in lifecycle emissions from crude oil recovery. We assume that the shares of conventional and alternative sources of crude remain fixed.

[65] The energy requirements of ethanol production are discussed in the calibration of the economic model.

source (72%). There is a significant difference in per liter emissions depending on the fuel source used, with coal fired plants emitting close to 40% more than natural gas fired plants. The weighted average emissions factor per liter of ethanol produced (ϕ_{Fe}^{P}) is 1.17 kgCO_2e/l.

Farrell et al.(2006) report emissions from ethanol production of 1.35 kgCO_2e/liter for a 2001 vintage ethanol plant. This differs from our estimate as they assume that 60% of ethanol plants are fuel by coal, compared our assumption of 28%. Liska et al. (2009) find that newer conversion technologies can reduce emissions from ethanol production by 30% relative to our estimate.

Table 3-4. Ethanol Production Emissions by Technology

	2003 Share	kgCO_2e/l
Dry Mill - NG	57%	1.04
Dry Mill - Coal	18%	1.53
Wet Mill - NG	15%	0.96
Wet Mill - Coal	10%	1.57
Average		1.17

Agriculture

Direct Emissions

As shown in (2.II.7), the direct agricultural emissions are based on the quantity of inputs used in agricultural production and input specific emissions factors. A detailed description of the emissions factors is given in the following sections, and the final crop, rotation and tillage specific emissions factors are reported in Table 3-9 and Table 3-10.

Energy Use

The emissions per unit energy (ϕ_{E_e}) used in agricultural production are reported in Table 3-5. As discussed above, these factors include both emissions from the combustion of the fossil fuel plus the emissions from the production and transportation of the fuel. The combustion emissions are based on the chemical properties of the fuel and the assumed combustion technologies of GREET 1.8c. The production emissions were also estimated using GREET 1.8c using default assumptions for the efficiency of the recovery processes and the shares of fossil fuels used in energy production. For each fuel, the emissions from combustion far outweigh the emissions from recovery and production.

The calculation of the electricity emissions factor requires more discussion.[66] The emissions factor for electricity is based on GREET 1.8c assumptions for the mix of US electricity production technologies (1.8% oil, 17.3% natural gas, 50.9% coal, 0.9% biomass, 20.1% nuclear and 9.0% other renewable sources including hydro, wind and geothermal). The value reported by GREET 1.8c (211.8 gCO_2e/MJ) is weighted by the average efficiency of the electricity production capacity and adjusted for an 8% transmission loss.[67] This calculation follows the methods of Farrell et al. (2006), and represents the emissions per MJ of primary energy used to produce a MJ of electricity.

Table 3-5. Agricultural Emissions Factors (gCO_2e/MJ)

	Combustion	Production	Total
Diesel	73.7	15.1	88.8
Gasoline	72.0	18.3	90.3
Natural Gas	63.0	9.3	72.3
LPG	65.9	11.3	77.2
Electricity	54.0	3.4	57.4

[66] The combustion emissions from electricity production occur at the power plant and are listed under the ‘combustion’ category. The ‘production’ emissions for electricity are the emissions from the recovery of fuels used in electricity production.
[67] It is assumed that oil power plants have an overall efficiency of 34.5%, natural gas plants have an overall efficiency of 39.5%, coal plants have an efficiency of 34.1% and biomass fired power plants have an overall efficiency of 32.1%. When accounting for transmission loss, the overall efficiency of electricity production is 27% (GREET 1.8c)

Cropland N_2O Emissions

There is no agreed upon parameterization for estimating the N_2O emissions resulting from agricultural production.[68] To generate an estimate that is consistent with other reports and generally accepted, we used the IPCC (2006) default parameters. The specific parameters are discussed below, and the final emissions factors are presented in Table 3-9 and Table 3-10.

Crop Residue

The parameters used to calculate the nitrogen content of crop residues (equation (2.II.10)) are given in Table 3-6. In addition to these parameters, consistent with the IPCC (2006) we assume that all crops renewed each year (ρ_k^{rn} in (2.II.10) is set to 1 for all k) and that all above ground residues are incorporated into the soils after harvest (ρ_k^{rm} in (2.II.10) is set to 0 for all k). For hay and wheat the weighted average of more specific IPCC parameters is used. In the case of hay, the parameters are a weighted average of IPCC data for alfalfa and non-legume hay based on the make-up of US hay. The IPCC (2006) values for alfalfa were assigned to both alfalfa and other legume hay, while the IPCC non-legume hay value was assigned to the remaining hay production. Likewise, our parameters for wheat are a weighted average of the IPCC (2006) winter and spring wheat categories.[69]

Table 3-6. IPCC Default Crop Residue Nitrogen Content Parameters

	α_k	b_k	DM_k	$N_{CR_k}^{ag}$	$N_{CR_k}^{bg}$
			mt/ha	kg N/kg DM	kg N/kg DM
Corn	0.61	1.03	7.71	0.006	0.007
Soybeans	1.35	0.93	2.08	0.008	0.008
Hay	0.00	0.26	5.03	0.023	0.017
Wheat	0.49	1.52	2.65	0.006	0.009

[68] See for example Crutzen et al. (2008), Smeets et al. (2009) or (2007), or Menichettie and Otto (2008).
[69] Based on NASS data (2003) we assume that 73% of wheat planted is winter wheat. We classify the remaining 27% as spring wheat.

Direct Cropland N_2O Emissions

The emissions factor for direct N_2O emissions for cropland, ϕ_{AG}^{N-D}, is based on IPCC worldwide default value of 1%. This factor represents the percent of mineral N inputs (synthetic N or N in crop residue) by weight that will be released to the atmosphere as N in N_2O. Converting from N in N_2O to CO_2e provides the specific value for ϕ_{AG}^{N-D} of 4.65 kgCO_2e/kg N.[70] The previous version of the IPCC guidelines (1997) suggests a higher direct emissions factor for N inputs of 1.25%, based on the work of Bouwman (1996). The 2006 IPCC methods cite more recent studies as justification for lowering this value. Bouwman et al. (2002) collect 846 estimates of N_2O emissions from agricultural fields and find that N application rates have an impact on N_2O emissions, as does soil texture, crop type and fertilizer type. They found that the global mean fertilizer induced emissions for N_2O to be 0.9% of the N applied. Novoa and Tejeda (2006) conduct a review of studies on the N_2O-N emissions from crop residues and find that 1.055% of N in crop residues is released as N_2O.

Indirect Cropland N_2O Emissions

Following the IPCC (2006) default methods, we assume that 10% of N fertilizer applied to fields is volatilized (π_{AG}^{N-V} in equation (2.II.15) is set to 0.1) and that of the volatilized N, 1% is converted to N_2O after it has been redeposited. The specific value used for ϕ_{AG}^{N-V} is 4.65 kgCO_2e/kg volatilized N. These values are consistent with the 1996 IPCC defaults, but the IPCC (2006) suggests that there is more uncertainty in these values than previously thought. Studies from Brumme et al. (1999) and Denier van der Gon and Bleeker (2005) showed that N_2O emissions from

[70] To convert from N in N_2O to CO_2e, we first multiply by 44/28 to calculate the total weight in terms of N_2O and then multiply by 296 to determine the global warming potential of this N_2O.

atmospheric deposition in certain ecosystems are significantly higher than previously thought. Other studies, specifically Corre et al. (1999), find that some ecosystems have very low emissions factors for atmospherically deposited N.

To estimate the indirect emissions from leaching and runoff, we follow the IPCC (2006) and Reay et al. (2009) by assuming that 30% of total mineral N inputs leave the field as a result of leaching and run off (π_{AG}^{N-LR} in (2.II.15) is set to 0.30) and that 0.75% of this mineral N is converted to N in N_2O (ϕ_{AG}^{N-LR} is set to 3.49 kgCO_2e/kg N).

Discussion of IPCC Method

The accuracy of the IPCC methods is very much debatable. In the lifecycle literature, the IPCC default emissions factor of 1% N in N fertilizer converting to N in N_2O is the most common method for capturing N_2O emissions (Menichettie and Otto 2008). One advantage of this method is that is well known to the international community and is a common reference point. Some studies in the lifecycle literature have adopted the full IPCC (2006) methods including Wang et al. (2007) and Liska et al. (2009). The Forest and Agricultural Sectors Optimization Model (FASOM), which is the basis for the EPA's emissions analysis of the RFS (US EPA 2009b), uses the 1996 IPCC methods for calculating agricultural N_2O.

One problem with the IPCC method is that it does not account for the impact of management, soil type or climate on N_2O emissions. Edwards et al. (2006), uses spatial data for soil, climate and crops along with European Union national data on production practices to estimate N_2O emissions. They find that soil type, climate and ground cover impact N_2O emissions more than N fertilizer inputs. This result would suggest that the IPCC methods are inadequate. More advanced methods can be used and are recommended by the IPCC. For example, US EPA Inventories after 2005 have used the DAYCENT ecosystem model (Parton et al. 2005) to estimate N_2O emissions from cropland.

It has also been argued that the IPCC method underestimates N_2O emissions. Parkin and Kaspar (2006) found N_2O emissions 3 times higher than the IPCC default for corn-soybean rotations under different tillage systems in Central Iowa. They argue that IPCC method underestimates N_2O emissions specifically because the quantity of N inputs explains only a portion of N_2O emissions, which is consistent with the findings of Bouwman et al. (2002). As such, they suggest that an improvement to the IPCC method would be to establish specific emissions factors for different soil types, climates, fertilizer types and application rates, and residue management.

Crutzen et al. (2008) suggest, based on a top-down approach, that the N_2O conversion factor should be between 3-5% as opposed to the default 1%. This factor is calculated by dividing worldwide N_2O emissions from agriculture by the anthropogenic input of newly fixed N. However, if all categories suggested by the IPCC are used (particularly livestock production and grazing), they find total N_2O emissions calculated using the IPCC methods to be consistent with their analysis.

Smeets et al. (2007) argue that the emissions factor suggested by Crutzen et al. (2008) implicitly includes emissions that result from livestock production, and is therefore an over estimate of cropland N_2O emissions. They conduct a similar analysis to that of Crutzen et al. (2008), but omit the N inputs and N_2O emissions from livestock, and propose a 2.7% emissions factor for N_2O emissions from cropland. Smeets et al. (2007) also acknowledge that neither analysis accounts for heterogeneity in crop types, management systems or climate.

Lime

The IPCC (2006) default emissions factors are 0.12 kgC/kg limestone and 0.13 kgC/kg dolomite, which assume that all C in both materials is converted to CO_2. West and McBride (2005) suggest that the default emissions factors may be too high for the US Corn Belt as a portion of the limestone is leached from the field, preventing the carbon from being released. We adopt a liming emissions factor coefficient of 0.12 which represents a weighted average of the two emissions factors based on limestone and dolomite use in US agriculture.[71] Converting to CO_2e, we set ϕ^{Lime} to 0.44 kgCO_2e/kg lime applied. The cropping practice specific per hectare emissions from the application of lime to agricultural soils ($\phi_{AG_{lt}}^{Lime}$) are given in Table 3-9 and Table 3-10.

Soil Organic Carbon

The calculation of soil organic carbon stocks relies on two sets of assumptions. The first set is used to represent the adoption of cropping and management systems between 1980 and 2003. The second set of assumptions is used to estimate the overall size of, and management practices impact on, the US cropland soil carbon stock. For this set of assumptions we follow a number of studies that have applied the IPCC inventory methods to the United States including: Ogle et al. (2003), EPA (2004), Sperow et al. (2003) and Eve et al. (2002).[72]

[71] According to the US EPA (2005), in 2003, 90% of the lime applied to agricultural soils was limestone and the remaining 10% was dolomite.
[72] Note that in some cases these sources overlap. For example, Ogle et al. (2003) is a journal article that explains the results and methods of the EPA's Inventory (2004). Also note that older versions of the EPA Inventory are referenced here because these inventories used methods that are more closely related to our methods. In more recent versions of the Inventory, the EPA used the Century ecosystem model to estimate changes in SOC.

Land Allocation Model

To model historic cropping practices, we combined historic data for the area of land producing specific crops and land held in CRP, with simple assumptions for the adoption of rotation and management practices.

The quantity of land producing each crop was based on data from the NASS (2003). This data was smoothed to isolate the historic trends from annual variability. The cumulative land enrolled in the CRP was collected from the Farm Service Agency (2009).

Our assumptions involved how the use of crop rotations and tillage practices evolved between 1980 and 2003. First, we assume that within each crop, the share of land held in each rotation has remained at the same level from 1980 to 2003. As discussed below, this assumption will have no impact on our estimation of SOC as we assume the impact of multi-crop rotations on SOC is no different than impact of monocultures.

To model the adoption of no-till, reduced tillage and mulch tillage we choose a year after which the share of a given tillage practice increased linearly such that the tillage share matched 2003 data. After the adoption paths of no-till, reduced tillage and mulch tillage were estimated, the remaining land in each crop/rotation category is assumed to be managed with conventional tillage.

The choice of the first year of adoption is based on data from the Conservation Tillage Information Center (CTIC) as reported by the EPA (2004). The CTIC data suggests that in 1982, there was no land managed with no-till. We assume that the adoption of no-till management practices begins in 1985 and allow the share of no-till to increase linearly by 1.2% per year so that by 2003, no-tillage management is used on 22% of cropland. As the CTIC does not report separate values for reduced and mulch tillage, we assume that both practices began to be adopted in 1980. The share of total cropland managed with reduced tillage increases linearly by 1% per year, while the share of mulch tillage to total cropland also increases by 1% per year. These growth rates were set such that by 2003, reduced tillage and mulch tillage are both practiced on 24% of total land respectively. With these adoption rates, conventional tillage drops from being practiced on all cropland in 1980 to being practiced on 30% of cropland in 2003.

The main weakness of this strategy is the assumption of linear adoption. The CTIC data shows that the adoption of no-till and reduced tillage increased steadily from the mid 1980's to the early 1990's and then leveled off in the 1990's. In fact, the share of US cropland in conventional tillage actually increased from 1992 to 1997, while the share in reduced tillage fell.[73] It should also be noted that land in no-till is often overestimated in survey data because farmers use different tillage practices for different crops in same rotation. An example that is often cited is a corn-soybean rotation of conventionally tilled corn and no-till soybeans (Ogle et al. 2003).

Soil Organic Carbon Model

Our calculations of soil organic carbon are based on the average level of organic carbon stored in the soil of cropland and land held in CRP, and estimates for how land uses and management practices impact soil carbon levels.

Reference Soil Carbon Level

As our model operates at a national level, we are unable to stratify cropland into different soil and climate regions, as suggest by the IPCC (2006). This is a significant limitation, as climate and soil type are two important determinants of SOC levels. For example, a high activity clay soil in a moist climate contains roughly 30%

[73] Some attribute this trend to extensive flooding in the Midwest in 1993 (Uri 1998).

more soil carbon than the same soil in a dry climate (Ogle et al. 2003). We assume that all cropland soils are homogenous and if conventionally managed for 20 years would have an organic carbon stock ($\overline{SOC}$) of 56.5 mtC/ha.[74] This factor represents the weighted average SOC stock of 36 combinations of soil classifications and climate regions, as defined by the IPCC (2006), which make up US cropland soils.

Our calculation of the reference SOC stock is based on two sources. Ogle et al. (2003) report the organic carbon, stored in the top 30 cm of the soil profile, per hectare of conventionally tilled cropland for the 6 dominant US soil classifications and 6 climate regions.[75] The SOC levels and the classifications are based on data from the National Resource Conservation Service (NRCS) soil survey. Eve et al (2002) estimate and report the share of US cropland in each soil type, climate region classification. Their estimate is based on National Resources Inventory data and the Parameter-elevation Regressions on Independent Slopes Model (PRISM) climate mapping system (Daly, Neilson, and Phillips 1994). They calculate the share of total cropland classified to each soil type and climate region for the 10 USDA Farm Production Regions.

To generate our national reference SOC stock, we first generate an average SOC stock for each of the Farm Production Regions by assuming that the soil types and climate regions are evenly distributed across each region. We then calculate the weighted average SOC stock of the 10 regions based on NASS (2003) data for total agricultural land in production.[76]

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[74] Land that is conventionally managed is defined by the IPCC (2006) as being cultivated for agricultural purposes, with intensive tillage and average levels of carbon inputs.
[75] The climate regions are based on IPCC definitions (2006). The soil type classifications also follow IPCC (2006) definitions, which are consistent with the USDA soil taxonomy.
[76] High activity mineral soils (67.7%) are the dominant soils for US cropland, followed by wetland soils (17.8%) and low activity mineral soils (9.3%). The majority of U.S. cropland lies in cold temperate moist (39.7%) and warm temperate moist regions (37.9%). Cold temperate dry and warm temperate

Stock Change Factors

We assign SOC stock change factors (S) to our land uses and management practices (crops, rotations and tillage systems) according to the broad land use classifications of the IPCC (2006) and EPA (2004). Each of the IPCC's land use categories is assigned a stock change factor based on Ogle et al. (2003) and EPA (2004). For land that falls into more than one classification, the stock change factors are multiplied. Due to the limited number of rotations in our current agricultural model, our stock change factors vary only by tillage practice, land producing hay and land in CRP.[77] The SOC stock change factors and our classification of land to IPCC (2006) categories are reported in Table 3-7.

Table 3-7. US Specific SOC Stock Change Factors and Classification

IPCC Classification	Model Category	Value
Land Use (S_{LU})		
Cultivated	Corn, Soybeans, Wheat	1.0
Uncultivated	Hay	1.3
Set-asides	CRP	1.2
Inputs (S_I)		
High	N/A	1.07
Medium	Cont. Corn, Cont. Soybeans, Cont. Wheat, Corn-Soybeans, Corn-Soybeans-Wheat	1.0
Low	N/A	0.94
Management (S_{MG})		
Conventional	Conventional Tillage	1.0
Reduced	Reduced Tillage, Mulch Tillage	1.02
No-till	No-Tillage	1.13

We rely on stock change factors that were estimated by Ogle et al. (2003) and used in the EPA Inventories through 2006. Ogle et al. (2003) synthesized a number of paired trials in the US or similar regions in Canada that report SOC levels to least 30

dry regions contain a much smaller percentage of cropland at 5.9% and 7.5% respectively while only 1.1% of U.S. cropland is located in sub-tropical regions.

[77] In our current model set up, there is little variability in the stock change factors for crop land. This is because row crops, small grains, and any rotation combining the two fall into the medium input category. There would be more variability in stock change factors if irrigation and rotations combining row crops or small grains with hay or fallow were considered (Ogle et al. (2003); EPA (2004)).

cm in depth for 20 years after a management change. These studies are used to estimate the change in soil carbon that result from a shift from conventional cultivation, to an alternative management practice. As such, the stock change factor for all conventionally cultivated land with medium inputs is 1.0.

We assume that that converting from conventionally managed cropland to native grassland or forests substantially increases SOC storage, and find U.S specific stock change factor of 1.3. Following Ogle et al. (2003) and the EPA ,we assign this value to land producing hay, but note that this could be an overestimation of the impact of hay production on SOC.[78]

Converting land in CRP to conventional agriculture will lead to a reduction in SOC of 20% over 20 years, or a loss of 0.56 mgC/ha per year.[79] This is consistent with the analysis of Fargione et al. (2008) who estimate a loss of SOC of 0.69 mgC/ha for 15 years after converting CRP to conventional cropland.

In terms of tillage practices, a shift from conventional tillage to no-till increases SOC levels in the top 30 cm of soil by 13% in the 20 years after a management change. Shifts between conventional tillage and reduced and mulch tillage have much smaller effect, with SOC levels increasing by only 2%. These factors are consistent with the findings of West and Post (2002).

Currently we assume that these stock change factors are also valid for shifts from alternative management practices to the reference state, and for shifts between non-reference practices. This is an oversimplification, as it is commonly found that

[78] In estimating this factor, Ogle et al. included studies that measured the SOC losses after land was converted to cropland ('plow-out' studies). Therefore the uncultivated land stock change factor is biased upwards because it also represents SOC losses from the conversion of native land to cropland which occurs quicker than SOC accumulations (see citations below).

[79] Given a reference soil carbon stock of 56.63 mgC/ha, converting from conventionally managed cropland (stock change factor of 1) to CRP (stock change factor of 1.2), and assuming the accumulation occurs over 20 years, yields ((1.2)(56.63) – 56.63)/20 = 0.56 mgC/ha per year.

soil carbon gains resulting from a management change occur more slowly than soil carbon losses that would occur if the management change was reversed.[80]

Impact of Tillage on Soil Carbon

There is some debate over effects of tillage on SOC stocks. A number of studies have argued that reducing tillage intensity leads to increased levels of SOC, effectively sequestering carbon from the atmosphere (West and Post (2002), Lal et al. (1998), Lal (2004), Paustian et al. (1998) and Kern and Johnson (1993) among others),. These studies were convincing enough for the IPCC to consider no-till agriculture a sink for CO_2 (as discussed above) and for the Chicago Climate Exchange (CCX) to offer payments to farmers who converted to no-till production under the assumption that no-till agriculture sequestered roughly 0.3 mtC/ha per year (CCX 2008).

A main criticism of these studies is that most only considered soil samples to 30 cm in depth. Others argue that 30 cm is not deep enough to measure the impacts of tillage on soil carbon because the root systems, a main source of organic carbon to soils, of most plants extend further than 30 cm into the soil (J. M. Baker et al. 2007). Baker et al. claim that the depth of the soil sample is especially important in measuring the impact of no-till on SOC because no-till methods inhibit the growth of plant roots.[81] As the root system is concentrated at a shallower depth in no-till, the amount of SOC stored in the first 30 cm is increased, but the amount at deeper levels is reduced. It follows that in conventional tillage, where the root system extends deeper into the soil, SOC is distributed across the entire soil profile.

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[80] Burke et al. (1995), Ihori et al. (1995) Reeder et al. (1998) and Baer et al. (2000).

[81] The inhibited growth is a result of increased surface residue (which keeps the soils cooler) and increased compaction (which makes it hard for the root system to grow).

A number of studies have found results consistent with this hypothesis. VandenBygaart et al. (2003) conduct a meta-analysis of 100 studies of conservation tillage studies in Canada and find that for studies that measured to below 30 cm there is no statistical difference between the SOC levels of no-till and conventional tillage. Other studies, which sampled to a greater than 30 cm depth, have arrived at similar conclusions.[82] The conclusion that can be made from this debate is that although conservation tillage has many advantages, including reduced fuel usage, soil erosion and production costs, its ability to sequester carbon may be overstated.

Biomass Losses from Converted CRP Land

The biomass on CRP land can take a number of different forms, including native or introduced grasses, hardwood or softwood trees, wildlife habitat, wetland, riparian buffers, etc. However most land held in CRP is grassland. In 2007, at least 77% of continuous signup CPR was classified as native or introduced grasses (Barbarika 2008). This is also a reasonable assumption because the goal of this calculation is to measure the emissions consequences of converting land in CRP to cropland and it is CRP held in grassland that will likely be converted to cropland. This assertion is based on the cost of converting land containing trees or other woody biomass to cropland being higher than the cost of converting grassland. If it is believed that CRP lands held in forest or other woody biomass would be converted to cropland then the biomass carbon losses would be much higher.[83]

Our estimates of biomass stored on CRP lands follow Fargione et al. (2008) and assume that the carbon content of above-ground and below-ground biomass are

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[82] Carter (2005), Dolan et al. (2006), Machado et al. (2003) and Blanco-Canqui and Lal (2008).

[83] If CRP land that was held in woody biomass was converted to cropland then the emissions from conversion would be closer to the value used for international land use change emissions discussed below.

1.6 mtC/ha and 6.7 mtC/ha respectively. These estimates are based on NASS (2009) estimates for non-alfalfa hay, and an assumed value (4.1) for the root:shoot ratio, which represents temperate grassland. As the conversion of CRP to cropland is assumed to release all carbon stored in the above- and below-ground biomass in the year of conversion, ϕ^{CRP} is set to 30.43 $mtCO_2e$ per hectare of CRP converted to cropland.

Farm Input Production

The emissions factors for farm input production (ϕ_j^J) are derived from GREET 1.8c using default assumptions are reported in Table 3-8.[84] The production practices that make up the emissions factors, and the major assumptions used to derive each are discussed below.

Table 3-8. Farm Input Production Emissions Factors

	$kgCO_2e$/kg product
N Fertilizer	2.99
P Fertilizer	1.04
K Fertilizer	0.69
Lime	0.63
Pesticide	21.87

Nitrogen Fertilizer

Deriving the nitrogen production emissions factor from GREET 1.8c yields a parameter of 2.99 $kgCO_2e$ per kilogram nutrient N.[85] This factor is based on an average US nitrogen fertilizer mix of 70.7% ammonia, 21.1% urea and 8.2% ammonium nitrate (ERS 2008b). The production of each N fertilizer material has different emissions intensity. In 2003, the emissions factors calculated by GREET for

[84] As the make-up of pesticide varies by crop (see discussion below), the value reported in the table is weighted average based on US corn, soybeans hay and wheat production.
[85] Our emissions factor is about 1 $kgCO_2e$ less than the factor reported by Farrell et al. (2006) who use GREET 1.6.

a kilogram of ammonia, urea, and ammonium nitrate are 2.62, 1.61 and 9.74 $kgCO_2e$ per kilogram N respectively. These emissions factors include the emissions from producing the feedstock to fertilizer production (primarily natural gas) as well as the emissions from the processing and transportation of the fertilizer itself. While there is considerable variability in the emissions generated in the production of each fertilizer, we assume that the shares of N fertilizer used in US agriculture are fixed over time.[86]

Phosphorus Fertilizer

Our emissions factor for the production of phosphorus fertilizer is 1.04 $kgCO_2e$ per kg nutrient P which is derived from GREET 1.8c. This factor includes the production, processing and transportation of sulfuric acid, phosphoric rock and phosphoric acid. Farrell et al (2006) use a value of 1.6 $kgCO_2e/kg$.

Potassium Fertilizer

Our emissions factor for the production of potassium fertilizer is 0.69 $kgCO_2e/kg$ nutrient K. This factor includes only the emissions from production and transportation of potassium oxide (K_2O). This is consistent with the emissions factor used by Farrell et al. (2006).

Lime

The production emissions of lime include mining, production and transportation. The factor derived from GREET 1.8c is 0.63 $kgCO_2e/kg$ lime. This factor differs by a factor of 10 from the value used by Farrell et al. (2006). We use the

[86] Since the late 1970's the ammonia has made up approximately 70% of total nitrogen applied in the US. Over this same time, ammonium nitrate has been steadily displaced by urea. In 1980, ammonium nitrate made up 18% of total N applied while urea made up only 11%. In 2007, urea made up 20% of total N applied, while ammonium nitrate made up only 9%. This shift away from ammonium nitrate has slowed since the early 1990's, as its share only dropped by 4% since 1990, although the trend is still downward (ERS 2008b). If this trend continues, our emissions factor will be an over estimate as ammonium nitrate has a significantly higher production emissions factor than either ammonia or urea.

GREET 1.8c factor because this value allowed us to more closely match our estimate total emissions from lime production to estimates based on the energy use data reported by the EIA's 2002 Manufacturing Energy Consumption Survey (EIA 2007).

Pesticide

Our emissions factor for the production of pesticide is the weighted average emissions from the production of 4 herbicides and a general insecticide. The crop specific share of herbicide and insecticide in pesticide is derived from the USDA ARMS survey (ERS 2008a). For each crop, herbicide's share is at least 90%.[87]

The insecticide emissions factor derived from GREET 1.8c is 24.90 $kgCO_2e/kg$ insecticide. This factor calculates the emissions from producing and transporting a unit of insecticide. It is assumed that the production emissions intensity of a unit of insecticide is the same for all crops.

GREET estimates the emissions for the production of four major herbicides: Atrazine, Metolachlor, Acetochlor and Cyanazine. As the emissions for each herbicide are substantially different, we assign a different mix of herbicides for each crop. We use the GREET 1.8c assumptions for the mix of corn and soybeans herbicide, and assume herbicide applied to hay and wheat consists of equal parts of the specific herbicide products.[88] Despite the variability in the emissions factors for herbicide production, the weighted emissions factors for each crop are very similar, ranging between 21.7 $kgCO_2e/kg$ (wheat) and 22.3 $kgCO_2e/kg$ (soybeans).

[87] We find that corn pesticide is 93% herbicide, soybean pesticide is 99% herbicide and wheat pesticide is made up of 93% herbicide (ERS 2008a). The pesticide used on hay is assumed to have a similar make up to the pesticide used on wheat.

[88] This is the same assumption made in GREET 1.8c for herbaceous biomass. GREET assumes that herbicides applied to corn consists of 31.2% Atrazine, 28.1% Metolachlor, 23.6% Acetochlor and 17.1% Cyanzine. Herbicides applied to soybeans are assumed to consist of 36.2% Atrazine and 63.8% Metaolachlor.

Other

In addition to the production of fertilizer and other farm inputs, we attribute the emissions from the transportation of crops to the farm input category. The transportation emissions from corn is derived from Farrell et al. (2006) and includes a 39.1 kgCO_2e/ha emission for the transportation of corn. The transportation emissions from soybeans (42.3 kgCO_2e/ha) are estimated from GREET 1.8c. In the absence of lifecycle emissions studies for hay and wheat farming, we assume that the other emissions for wheat and hay are identical to those in soybeans.

<u>Direct Agricultural and Farm Input Emissions Factors</u>

The variability in the agricultural emissions factors is primarily driven by the type of crop being produced. Table 3-9 displays the aggregation of the rotation and tillage specific emissions factors to the crop level based on the 2003 allocation of cropland.[89] Corn production proves to be the most emissions intensive use of cropland. The direct emissions of corn production (2848.4 kgCO_2e/ha) are more than 2 times higher than each of the other crops because corn production requires large inputs of energy, synthetic N fertilizer and lime. The difference in direct emissions across the non-corn crops is driven primarily by the nitrogen fertilizer use, with wheat production resulting in more direct emissions (1095.4 kgCO_2e/ha) than either hay (921.7 kgCO_2e/ha) or soybean (484.3 kgCO_2e/ha) production. As the emissions from input production are based on crop specific input use, the large quantities of nitrogen fertilizer and lime required for corn production result in input production emissions for corn (428.7 kgCO_2e/ha) that are roughly three times higher than the other three crops.

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[89] The US average emissions factor is a weighted average, based on the 2003 allocation of cropland, of the emissions factors for the four crops.

Wheat production has the highest input production emissions of the other three crops at 286.8 $kgCO_2e/ha$ while soybean production has the lowest at 112.9 $kgCO_2e/ha$.

Table 3-9. Agricultural Emissions Factors by Crop ($kgCO_2e/ha$)

	Direct				Input Production					
	Total	Energy	N_2O	Lime	Total	N	P	K	Lime	Other
US Average	990.2	307.1	616.0	67.2	407.7	186.0	34.8	22.6	95.5	68.7
Corn	2848.4	476.2	1265.4	199.0	907.7	428.7	54.7	43.2	283.0	98.1
Soybeans	484.3	224.4	146.9	0.0	112.9	12.3	14.0	17.5	0.0	69.1
Hay	921.7	249.6	415.5	22.3	234.3	96.1	39.2	17.5	31.7	49.9
Wheat	1095.4	241.5	547.2	19.9	286.8	176.0	29.2	5.7	28.3	47.7

Combining the direct and input production emissions categories provides a lifecycle emissions factor for agricultural production. Our estimate for total lifecycle emissions from corn production are within 5% of commonly cited studies ((Farrell et al. 2006) and (Wang 1999)). Our emissions lifecycle emissions factor for soybeans is within 2% of a similar study (Wang 1999). Unlike the LCA studies, which generally consider only average emissions by crop, our estimated emissions factors will change with adjustments in rotations and tillage practices (Table 3-10).

With the exception of soybean production the emissions factors for continuous cropping practices are higher than those of multi-crop rotations. For example, continuous corn production results in the emission of 2137.1 $kgCO_2e/ha$ while a corn-soybeans results in the emission of 1140.6 $kgCO_2e/ha$. This is a result of two effects caused by the multi-crop rotations including soybeans. First, the input requirements and emissions (Table 3-9) for soybeans are lower relative to the three other crops, which lowers the overall emissions of multi-crop rotations. Second, the nitrogen fertilizer requirements for corn and wheat planted after soybeans are lower because of the soybean nitrogen credit, which results in lower N_2O emissions from synthetic N fertilizer.[90]

[90] Our model may overestimate the N_2O emissions savings from crop rotations as we are not able to estimate the temporal dynamics of the N cycle.

Table 3-10. Agricultural Emissions Factors by Rotation and Tillage ($kgCO_2e/ha$)

	Direct				Input Production					
	Total	Energy	N_2O	Liming	Total	N	P	K	Lime	Other
Continuous Corn	2137.1	505.5	1432.6	199.0	984.3	509.8	56.0	42.2	283.0	93.4
Conventional	2168.5	567.7	1401.9	199.0	965.6	494.9	53.9	42.0	283.0	91.8
Reduced	2121.0	468.7	1453.3	199.0	994.6	519.8	60.5	39.8	283.0	91.6
Mulch	2142.1	468.8	1474.3	199.0	1009.6	530.0	57.8	43.2	283.0	95.6
No-till	1978.1	392.6	1386.5	199.0	959.7	487.4	49.2	43.7	283.0	96.4
Continuous Soybean	383.8	236.9	146.9	0.0	117.2	12.3	14.2	19.2	0.0	71.4
Conventional	462.5	318.4	144.1	0.0	110.6	11.0	11.7	17.5	0.0	70.4
Reduced	368.1	222.0	146.1	0.0	122.2	12.0	17.9	21.9	0.0	70.5
Mulch	365.0	218.9	146.1	0.0	113.6	12.0	13.7	17.5	0.0	70.4
No-till	309.2	158.5	150.8	0.0	122.5	14.2	15.1	20.0	0.0	73.2
Corn Soybean	1140.6	348.0	691.2	101.4	506.5	212.0	34.7	30.8	144.2	84.8
Conventional	1237.6	445.6	690.5	101.4	504.3	211.7	35.2	30.3	144.2	82.9
Reduced	1156.0	345.9	708.7	101.4	516.4	220.5	36.8	30.9	144.2	84.1
Mulch	1135.4	346.1	687.9	101.4	503.7	210.4	34.1	32.0	144.2	83.0
No-till	1058.8	278.0	679.3	101.4	502.2	206.3	33.1	29.7	144.2	88.9
Corn Soybean Wheat	842.5	269.0	536.6	36.9	313.4	165.1	29.1	11.4	52.5	55.3
Conventional	913.6	346.7	530.0	36.9	310.0	161.9	29.9	10.9	52.5	54.9
Reduced	826.9	246.9	543.0	36.9	316.2	168.2	28.3	12.0	52.5	55.2
Mulch	826.4	247.3	542.2	36.9	316.1	167.8	29.0	11.7	52.5	55.0
No-till	756.5	184.7	534.9	36.9	313.3	164.2	28.8	11.3	52.5	56.4
Continuous Hay	687.3	249.6	415.5	22.3	234.5	96.1	39.2	17.6	31.7	49.9
Conventional	765.4	318.5	424.6	22.3	238.3	100.5	40.5	15.7	31.7	49.8
Reduced	650.9	218.8	409.8	22.3	230.2	93.4	36.2	19.6	31.7	49.3
Mulch	659.0	219.2	417.5	22.3	232.3	97.1	38.0	15.9	31.7	49.6
No-till	578.0	158.0	397.7	22.3	233.8	87.5	42.7	20.5	31.7	51.4
Continuous Wheat	879.9	219.4	640.6	19.9	336.0	221.3	29.7	6.4	28.3	50.3
Conventional	1015.0	319.5	675.7	19.9	352.0	238.3	30.1	5.4	28.3	49.8
Reduced	878.8	218.3	640.6	19.9	332.6	221.3	26.9	6.8	28.3	49.3
Mulch	896.0	217.3	658.8	19.9	341.7	230.2	28.2	5.5	28.3	49.6
No-till	790.1	158.4	611.9	19.9	325.8	207.4	31.7	7.1	28.3	51.4

The variability in the emissions factors for different tillage practices, within the same rotation, is primarily driven by energy requirements. The conservation tillage practices, particularly no-till, require fewer passes across the field than conventional tillage practices, lowering the emissions from diesel fuel combustion. With the exception of continuous corn and corn-soybeans rotations, the emissions from energy combustion for no-till are approximately 50% less than the energy emissions of conventional tillage.[91] While the decreasing emissions from energy use with decreasing tillage intensity trend is consistent with Nelson et al. (2009), our effect is slightly more pronounced.[92]

International Land Use Change

We set the present value emissions from converting a hectare of native land to cropland in the rest of the world (ϕ_{PV}^{ILU}) to 243.9 mtCO_2e/ha land converted to cropland. This factor represents the average stream of emissions from converting a unit of native land to cropland over 80 years, discounted at 2% per year.

We assume that clearing a hectare of native land for cropland will cause emissions of 281.3 metric tons CO_2 over 80 years. This is based on the EPA's Draft Regulatory Impact Analysis of the Renewable Fuel Standard (2009b) and represents the average emissions per hectare of converting native lands to cropland in the rest of the world. International land use change is estimated using the FAPRI models, based on a scenario where the corn ethanol mandate increases ethanol consumption by 9.83 billion liters in 2022. Carbon losses from land use change are based on IPCC (2006)

[91] Continuous corn in conventional tillage emits approximately 30% more CO_2e from fossil fuel combustion than no-till continuous corn. Likewise, in a corn-soybeans rotation, no till offers a 40% reduction in emissions from fossil fuel combustion (Table 3-10).

[92] Nelson et al. (2009) report on-site carbon emissions from fossil fuel combustion for no-till relative to conventional tillage to be 25% lower for corn, soybeans and wheat production and 33% lower for hay production.

default assumptions and include the carbon in above- and below-ground biomass, changes in soil carbon stocks, non-CO_2 emissions from clearing of land by fire, and foregone sequestration of regenerating forests. These emissions are incurred over time. The EPA (2009b) assume that biomass decay and fire clearing cause large emissions in the first year after conversion, soil carbon losses occur over the next 20 years and lost forest sequestration occurs for 80 years. Consistent with Searchinger et al. (2008) we assume that 60% of the overall emissions are from plant biomass, while 20% of the total is due to lost soil carbon and 20% are due to forgone sequestration.

Searchinger et al. (2008) calculate the average carbon loss for land converted to agriculture in the rest of the world to be 351.4 mgCO_2e/ha. This estimate is based on the results of the FAPRI worldwide agricultural model, and estimates of carbon stored in the soils and vegetation from the Woods Hole Research Center.[93] They assume that 25% of the carbon in the top meter of soil and all the carbon stored in the vegetation is released. They also account for 30 years of foregone sequestration as a result of converting regenerating forests.

We use the value from the EPA (2009b) because the FAPRI simulations used to estimate land use change in the EPA analysis represent the projected expansion in US ethanol consumption due to the RFS, while the simulations used by Searchinger et al. (2008) represent a substantially larger expansion in US ethanol consumption.

We have chosen a discount rate of 2%, which is a central value of the discount rates suggested by Stern (2008), who uses 1.3%, and Nordhaus (2007) who suggests 3%.

[93] Searchinger et al. (2008) generate this estimate using a scenario where the US increases corn ethanol consumption to 111.76 billion liters by 2016. This is close to double the RFS mandate for conventional ethanol in this year.

In the benchmark, international crop yields, μ_{ROWk}, in equation (2.II.25) are 3.27, 2.03 and 2.6 metric tons per hectare for corn, soybeans and wheat respectively (FAS 2009).

International Oil Consumption

We let the emissions from crude oil consumed in the rest of the world (ϕ^R) be 369.0 kgCO_2e/barrel. To estimate this factor, the EIA's (2008b) reported total non-US CO_2 emissions from petroleum consumption are divided by total non-US petroleum use.[94]

Estimating the emissions of the use of a barrel of crude oil is not straight forward for a number of reasons. First, crude oil is very rarely combusted directly. Most often it is refined into other liquid fuels, such as gasoline and diesel, which are then combusted. Second, the quality and properties of crude oil are very heterogeneous. Third, not all crude is converted into products that are combusted. For example, oil refineries produce a number of fuels (gasoline, diesel, kerosene among others) as well as other products (asphalt, lubricating oil) in a joint production process. Some of these products, such as asphalt, are never combusted, so the carbon in these products is not released as CO_2.

To address these issues, our emissions factor incorporates the EIA's assumptions for worldwide end uses of petroleum and the chemical properties of crude oil. As such, our emissions factor implicitly assumes that roughly 90% of the carbon in crude oil is not emitted.[95] We also assume that this emissions factor does not

[94] Petroleum is any of a number of liquid hydrocarbon mixtures including crude oil, lease condensate, unfinished oils, refined petroleum products obtained from crude oil and natural gas plant liquids. In the production of petroleum products, the EIA reports that in 2003 crude oil made up about 90% of total petroleum produced (EIA 2008b).

[95] An average barrel of crude in the US (assuming a density of 848 grams per liter and carbon content of 85.3% (Wang 2009)) would release 421 kgCO_2/bbl if totally combusted.

change over time, or in response to changes in crude oil prices. This emissions factor is imperfect, as petroleum is made up of a number of different products in addition to crude oil, each presumably having a different emissions factor. To simplify, we assumed that changes in the price of crude oil will not impact the ratio of total non-crude petroleum consumed to total crude oil consumed.

APPENDIX A

Table 3-11. Tillage System Definitions

Tillage Category	Residue	Other
Conventional	Less than 15%	Typically involves plowing or other intensive tillage. Some combination of cultivation and herbicides is used for weed control.
Reduced	15-30%	Weed control is accomplished with some combination of herbicides and cultivation
Mulch	Greater than 30%	The soil is disturbed prior to planting although less intensive tillage tools are used. Weed control is accomplished with herbicides and cultivation. This category also includes land that the USDA classifies as ridge tillage.
No-till	Greater than 30%	The soil is left undisturbed from harvest to planting except for nutrient injection. Planting occurs is a narrow seedbed and weed control is accomplished primarily with herbicides. Some cultivation may be used for emergency weed control

Table 3-12. Benchmark Tillage Shares by Rotation (million hectares)

	Total	Conventional	Reduced	Mulch	No-till
Continuous Corn	6.88	43%	17%	32%	8%
Continuous Soybean	4.53	38%	17%	11%	34%
Corn Soybean	43.21	20%	24%	30%	26%
Corn Soybean Wheat	21.81	35%	25%	19%	21%
Continuous Hay	25.65	39%	26%	20%	14%
Continuous Wheat	5.75	24%	23%	14%	38%

Table 3-13. ARMS Surveys Collected by State and Year

	Crops		
States	**Corn**	**Soybeans**	**Wheat**
Corn Belt			
Illinois	1996-2005	1996-2002	1997-2004
Indiana	1996-2005	1996-2002	N/A
Iowa	1996-2005	1996-2002	N/A
Missouri	1996-2005	1996-2002	1997-2004
Ohio	1996-2005	1996-2002	1997-2004
Lake States			
Michigan	1996-2005	1997-2002	2004
Minnesota	1996-2005	1996-2002	1996-2004
Wisconsin	1996-2005	1996; 1997; 2000-2002	N/A
Northern Plains			
Kansas	1996; 1998-2005	1997-2002	1996-2004
Nebraska	1996-2005	1996-2002	1996-2004
North Dakota	N/A	N/A	1996-2004
South Dakota	1996-2005	1997-2002	1996-2004
Southern Plains			
Oklahoma	N/A	N/A	1996-2004
Texas	1996; 1998-2005	N/A	1996-2004
Pacific			
Oregon	N/A	N/A	1996-2004
Washington	N/A	N/A	1996-2004
Mountain			
Colorado	1998-2005	N/A	1996-2004
Idaho	N/A	N/A	1996-2004
Montana	N/A	N/A	1996-2004
Delta			
Arkansas	N/A	1996-2002	N/A
Louisiana	N/A	1996-2002	N/A
Mississippi	N/A	1996-2002	N/A
Southeast			
Georgia	2001-2005	N/A	N/A
Appalachian			
Kentucky	1996; 1998-2005	1997-2002	N/A
North Carolina	1996; 1998-2005	1997-2002	N/A
Tennessee	N/A	1996-2002	N/A
Virginia	N/A	2002	N/A
Northeast			
Maryland	N/A	2002	N/A
New York	2000-2005	N/A	N/A
Pennsylvania	1996; 1998; 2000-2005	1997; 1999	N/A

States not surveyed include: **Pacific**: California; **Mountain**: Arizona, Utah, Wyoming, Nevada, New Mexico; **Southeast**: Alabama, Florida, South Carolina; **Appalachian**: West Virginia; **Northeast**: Maine, Vermont, New Hampshire, Massachusetts, Connecticut, Rhode Island, New Jersey, Delaware.

Corn surveys are available for 1996, 1997, 1998, 1999, 2000, 2001, 2005

Soybean surveys are available for 1996, 1997, 1998, 1999, 2000, 2002

Wheat surveys are available for 1996, 1997, 1998, 2000, 2004

Table 3-14. Agricultural Input Requirements

	Energy (MJ/hectare)			Fertilizers (kg/hectare)				
	Diesel	Other Energy	Total	Nitrogen	Phosphoru	Potassium	Pesticide	Lime
Continuous Corn	2656.11	3532.80	6188.91	170.50	53.76	61.09	2.50	448.47
Conventional	3357.16	3532.80	6889.96	165.51	51.81	60.91	2.43	448.47
Reduced	2241.80	3532.80	5774.60	173.84	58.15	57.66	2.42	448.47
Mulch	2242.61	3532.80	5775.41	177.26	55.60	62.64	2.60	448.47
No-Till	1383.26	3532.80	4916.06	163.01	47.32	63.33	2.64	448.47
Continuous Soybean	1920.96	867.11	2788.07	4.13	13.59	27.69	1.31	0.00
Conventional	2839.98	867.11	3707.10	3.67	11.26	25.36	1.26	0.00
Reduced	1753.25	867.11	2620.37	4.00	17.20	31.71	1.27	0.00
Mulch	1717.94	867.11	2585.05	4.00	13.20	25.36	1.26	0.00
No-Till	1036.73	867.11	1903.84	4.75	14.49	28.98	1.39	0.00
Corn Soybean	2004.90	2225.30	4230.20	70.92	33.37	44.65	2.02	228.50
Conventional	3106.09	2225.30	5331.39	70.81	33.87	43.93	1.93	228.50
Reduced	1981.98	2225.30	4207.28	73.75	35.35	44.76	1.98	228.50
Mulch	1983.81	2225.30	4209.11	70.38	32.76	46.38	1.93	228.50
No-Till	1216.77	2225.30	3442.06	68.99	31.85	43.08	2.20	228.50
Corn Soybean Wheat	2237.21	906.68	3143.89	55.20	27.99	16.53	0.61	83.22
Conventional	3112.85	906.68	4019.52	54.13	28.74	15.83	0.59	83.22
Reduced	1988.73	906.68	2895.41	56.24	27.22	17.36	0.60	83.22
Mulch	1992.56	906.68	2899.24	56.11	27.91	16.97	0.60	83.22
No-Till	1286.44	906.68	2193.12	54.93	27.73	16.33	0.66	83.22
Continuous Hay	2361.49	502.43	2863.92	32.14	37.69	25.31	0.35	50.26
Conventional	3138.30	502.43	3640.73	33.62	38.99	22.80	0.34	50.26
Reduced	2014.35	502.43	2516.77	31.22	34.80	28.46	0.32	50.26
Mulch	2019.13	502.43	2521.56	32.47	36.55	23.03	0.33	50.26
No-Till	1329.58	502.43	1832.01	29.25	41.04	29.69	0.42	50.26
Continuous Wheat	2021.14	502.43	2523.57	74.02	28.54	9.24	0.37	44.83
Conventional	3149.39	502.43	3651.81	79.71	28.93	7.86	0.34	44.83
Reduced	2008.99	502.43	2511.42	74.03	25.82	9.81	0.32	44.83
Mulch	1997.50	502.43	2499.93	76.98	27.12	7.94	0.33	44.83
No-Till	1333.21	502.43	1835.64	69.36	30.46	10.24	0.42	44.83

Table 3-15. Benchmark Ethanol Production Data

	Dry Mill	Wet Mill	Average	Source
Share of total production	0.75	0.25	-	GREET 1.8c
Share natural gas	0.76	0.40	0.67	GREET 1.8c
Corn (kg/liter)	2.53	2.63	2.56	GREET 1.8c
Energy (MJ/liter)	13.13	14.59	13.49	Farrell et al. (2006)
Labor ($/liter)	0.07	0.07	0.07	Shapouri and Gallagher (2005)
Capital ($/liter)	0.005	0.005	0.005	Shapouri and Gallagher (2005)
DGS (kg/liter)	0.69	-	0.52	GREET 1.8c
CGM (kg/liter)	-	0.12	0.03	GREET 1.8c
CGF (kg/liter)	-	0.53	0.13	GREET 1.8c
Corn Oil (kg/liter)	-	0.10	0.02	GREET 1.8c

Chapter 4. Results

This chapter will discuss the behavioral and emissions impacts of the Renewable Fuel Standard from 2008 to 2015. The analysis compares a pass of economic outcomes with the RFS in place to a baseline without the RFS, but including the prevailing state and federal gasoline taxes, the Volumetric Ethanol Excise Tax Credit and all dynamic assumptions.[96] For the RFS to lower greenhouse gas emissions, the emissions savings from the reduced consumption of gasoline must outweigh any carbon leakage.

To estimate the gross emissions savings, we analyze the impact of the RFS on ethanol consumption. In particular, we determine for which years the RFS mandate is higher than the baseline consumption of ethanol, and the quantity of ethanol that is forced into the fuel supply by the mandate. Comparing the emissions from the combustion of this additional ethanol, less the ethanol combustion credit, to the combustion emissions of an equal quantity of gasoline, provides the potential emissions savings of the RFS. In the baseline, the underlying price of crude oil leads to an expansion of the ethanol sector to levels close to the RFS mandate, which suggests that the potential emissions savings from the RFS corn ethanol mandate, relative to total US emissions, is limited.

The potential emissions savings are then compared to the various sources of carbon leakage. First, we analyze the potential for leakage in the domestic transportation sector. Although there is leakage, as the price of blended fuel falls as ethanol is added to the fuel supply in some years, the resulting emissions are small compared to the potential emissions savings. Next, we examine the impacts of

[96] As previously discussed, we assume that the aggregate state and federal fuel tax remains at 0.10 $/liter (FHWA 2003) for the entire pass. The volumetric tax credit for ethanol is reduced from 0.13 $/liter to 0.12 $/liter in 2008, reflecting the 2008 Farm Bill (US Congress 2008) and remains at this level for the remaining years.

mandated ethanol consumption on the fuel production sector. We find a substantial carbon leakage as the relatively emissions intensive production of ethanol replaces the production of gasoline.

The response of the domestic agricultural sector to the RFS mandate is then examined. As the mandate forces the ethanol production sector to demand more corn, the price of corn increases and the agricultural sector responds by increasing the production of corn at the expense of other crops. In addition to the shift to corn production, the agricultural sector also expands on to land allocated to the Conservation Reserve Program, and intensifies rotations and tillage practices.[97] Each of these effects leads to carbon leakage. The increased production of corn leads to larger emissions from agricultural energy use and nitrogen fertilizer application, while the conversion of CRP to cropland leads to releases large quantities of carbon stored in plant biomass. Agricultural intensification also leads to increased energy use and N_2O emissions as monocultures replace crop-rotations, and farmers revert to conventional management practices. An additional consequence of the reallocation of cropland is that the potential for agricultural soils to sequester carbon will be reduced as less land produces hay, uses conservation tillage practices or is held in CRP. The changes in the agricultural sector are then related to the increased production of fertilizers, pesticides and other farm inputs and the related emissions.

The simulations illustrates that the emissions consequences of converting CRP land to cropland and increasing the production of corn are substantially larger than the emissions from agricultural intensification. In addition, we find that a portion of the

[97] 'Intensification' is used throughout this text to describe both shifts from less intensive tillage practices to more intensive tillage practices (from no-till to conventional tillage for example) and shifts from multi-crop rotations to monocultures. This differs from the agronomic definition which is a description of tillage only.

increased agricultural emissions is offset because the expansion of corn displaces other crops that were generating emissions in the baseline.

Finally, we investigate the potential for carbon leakage in the international crude oil market and the international land market. The leakage in the international crude oil market occurs when the RFS decreases US demand for gasoline and in turn depresses the world price of crude oil, encouraging the consumption of crude oil in the rest of the world. The leakage in international land markets is an outcome of a reduction in US exports as the RFS diverts corn from exports to ethanol production and increases the relative prices of US crops. We find that the international leakages of the RFS far outweigh the domestic leakages because the international markets are much larger than domestic markets and are larger sources of greenhouse gas emissions.

This chapter is organized as follows. The set of dynamic assumptions underlying the simulations is discussed in Section I. In Section II the baseline simulation is validated for 2008 and contrasted with other projections. Section III will explore the behavioral adjustments in response to the RFS mandate that could lead to carbon leakage and compare the emissions savings from increased ethanol consumption to the total carbon leakage.

Section I – Underlying Dynamic Assumptions

The simulation model solves first for 2003, and then generates a time path of economic outcomes from 2008 to 2015 at one-year intervals. This timeframe encompasses the years for which the Renewable Fuel Standard mandate for corn ethanol is gradually increasing from 34 billion liters to 56.7 billion liters (Table 4-2). We allow for trend changes in the world price of crude oil, the world consumption of crude oil, the domestic price of natural gas, the fuel economy of passenger vehicles, the energy requirements and conversion efficiency of ethanol plants, domestic and international crop yields, CRP rental rates, demand for US crop exports and household income.

World Crude Oil Price Paths

The world price of crude oil follows projections from various years of the US EIA's Annual Energy Outlook (AEO). In our central case we use the price path for imported crude oil from the 2008 AEO Reference Scenario, which falls 4.1% per year between 2008 and 2015 and reaches 48.80 $/barrel in 2015 (EIA 2008c). As part of sensitivity analysis, we consider both a low and high crude oil price path. The low price path follows the AEO 2007 Reference Scenario in which the price of crude falls from 53.35 $/barrel in 2008 to 41.59 $/barrel in 2015. The high price path follows the AEO 2009 Reference Scenario and increases from 79.03 $/barrel in 2008 to 83.26 $/barrel in 2015.[98]

Energy Price

The price of natural gas evolves so that the relationship between the price of imported crude oil and the commercial price of natural gas matches the projections of the AEO. For example, in our central crude oil price case, the relationship between crude oil and natural gas is fixed following the projections of the AEO 2008 Reference Scenario (EIA 2008), while in our low crude oil case the relationship is fixed according to the AEO 2007 Reference Scenario. Fixing the natural gas-crude oil price relationship follows a number of studies that have shown that the two prices are strongly coupled (Bachmeier and Griffin (2006); Villar and Joutz (2006); Hartley, Medlock and Rosthal (2008)) and that as a general rule the energy equivalent prices of

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[98] The EIA offered two reference scenarios in the 2009. We use the reference case that includes the provisions of the American Reinvestment and Recovery Act.

natural gas and crude oil are equal (Brown and Yucel (2008); Bachmeier and Griffin (2006)).

World Crude Oil Consumption

We allow non-US crude oil consumption to increase by 1.1% annually between 2008 and 2015 following the EIA's 2009 International Energy Outlook (IEO) (2009a). This results in an estimate of world crude consumption of 70.4 million barrels per day in 2015.[99]

Passenger Vehicle Fuel Economy

The fuel economy of passenger vehicles increases at 0.22% per year, based on projections of the National Research Council (NRC) on the impact of CAFE standards (2002). This growth rate accounts for both fleet composition across weight classes and new and used vehicle stocks. The NRC provides 10 year projections of fuel economy by vehicle, weight class and model year for a number of scenarios. We calibrate baseline fuel economy to the NRC's "Path 1" assumptions, which reflects improvements in fuel economy of 11% for compact cars and 20% for light trucks. Our assumptions for the composition of the vehicle fleet are from Bento et al. (2009), who report vehicle composition by new and used vehicle stocks by weight class. The final growth rate in fuel economy is the weighted average change across both vehicle class and new and used vehicle stocks. This differs from the assumptions underlying the EIA's AEO 2009, which imply annual average improvement in fuel economy of 0.79% per year between 2008 and 2015.

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[99] The increase in non-US crude oil consumption reflects the EIA's projections that the developing economies in non-OECD Asia and the Middle East will return to trend economic growth in the years following 2009. The expected growth of the industrial and transportation sectors in China and India is the source of much of this increase (EIA 2009a).

Ethanol Production Efficiency

Following GREET 1.8c, the overall energy and conversion efficiency of ethanol production increases. These trends reflect an overall improvement in ethanol plant technology and a shift from less efficient wet mills to more efficient dry mills. We assume that in 2008, 85% of ethanol was produced in dry mills with the remaining produced in wet mills. By 2015, the share of dry mills increases to 88%. The overall energy efficiency of ethanol production improves slightly from 2008 to 2015, from 50.0 MJ/liter to 49.5 MJ/liter. The conversion efficiency of ethanol production increases 2.5 kg corn per liter ethanol in 2008 to 2.43 kg corn per liter in 2015.[100] In these trends, we assume that there will not be any large scale adoption of energy conserving practices such as the production of only wet co-products, using co-products or biomass as fuel, or combined heat and power (Wang, M. Wu, and Huo 2007).

Domestic Crop Yields

US crop yields improve following the USDA's 2009 Long Term Projections (2009). Specifically, the annual growth rates of crop yields are 1.36% for corn, 0.53% for soybeans, 0.08% for hay and 0.66% for wheat.

International Crop Yields

International (non-US) yields increase according to FAPRI (2009) estimates. Between 2008 and 2015, we allow corn, soybeans and wheat yields to increase 0.7%, 0.8% and 0.6% per year. The FAPRI model is also the basis for the international

[100] These improvements in ethanol conversion technology are consistent with, but not as large as those in FAPRI (2009), which has an overall improvement from 2.46 kg corn per liter in 2008 to 2.38 kg corn liter in 2015. At least part of this difference is a result of their estimate that 88.5% and 90% of ethanol production occurs in dry mills in 2008 and 2015 respectively.

agricultural yield projections of the EPA Impact Analysis of the RFS (US EPA 2009b).

CRP Rental Rate

Following historic trends (FSA 2009), Conservation Reserve Rental payments are allowed to increase at 2% annually.

Household Income

Following historic trends reported by the Bureau of Economic Analysis (BEA 2009), household income is allowed to grow at 1% per year.

Crop Exports

The rest-of-world demand for US crop exports is allowed to increase at 1% per year to reflect an increase in global household income.

Section II – Baseline Simulation

The baseline simulation creates a path of economic outcomes that act as counterfactual in our analysis, or a prediction of what would have occurred in absence of the Renewable Fuel Standard mandate. This simulation uses the set of dynamic assumptions described above, and includes the prevailing volumetric tax credit for blending ethanol and the fuel tax.

<u>Baseline Behavioral Trends</u>

Ethanol Consumption

In 2008, our baseline simulation predicts that the RFS mandate does not bind, as baseline ethanol consumption (35.3 billion liters) is above the mandated level of ethanol consumption of 34.0 billion liters (Table 4-2). This level of ethanol consumption is consistent with FAPRI (2009), the EIA (2008a) and the CBO (2009) which have reported ethanol consumption to be above the mandated level in 2008.[101]

Consistent with FAPRI (2009) and the 2009 Annual Energy Outlook (2009c) we project baseline ethanol consumption to increase over time. These sources project that ethanol consumption will be above the RFS mandate for each year until 2015, suggesting that the RFS mandate will never bind. Our model projects that baseline ethanol consumption will be slightly below the RFS mandated level in 2009 and remain below the mandated level until 2015. In 2015, we predict ethanol consumption of 45.5 billion liters compared to the RFS mandated level of 56.7 billion liters. This is consistent with Westhoff (2007) who predicts 47.6 billion liters of ethanol consumption 2015 without the RFS mandate and the EPA who estimate that ethanol consumption will reach 47.6 billion liters by 2022 (US EPA 2009b).

Gasoline Consumption and Vehicle Miles Travelled

In the baseline, our model predicts that passenger vehicles consume 440.1 billion liters of gasoline. This is consistent with, but below, the EIA (2008a) estimate of 469.5 billion liters of gasoline consumed in 2008.[102]

In the baseline simulation (following the central crude oil price path), the decreasing price of crude oil after 2009 causes total vehicle miles traveled to increase and fuel economy to decrease. As the price of miles is linked to the price of crude oil through the price of gasoline, VMT increases 13.5% between 2009 and 2015, from 4.1 to 4.6 trillion kilometers. Likewise, fuel economy falls slightly (0.5%) as consumers

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[101] The RFA (2009a) reports 2008 ethanol consumption at 34 billion liters which suggests the mandate may not have been binding.

[102] The main source for gasoline consumption by vehicle type, the Federal Highway Administration, has not yet reported numbers for 2008. Therefore our only source of comparison is the EIA, which does not disaggregate gasoline use by vehicle type. We therefore assume that 95% of gasoline used in the transportation sector is consumed in passenger vehicles (US EPA 2009b) to compute our estimate.

substitute away from non-fuel expenditures on driving.[103] These two trends lead to an overall increase in gasoline and domestic crude oil consumption of 13.3% and 12.7% respectively. Our estimate of the increase in total VMT consumed between 2009 and 2015 compares with the EIA (2009c) projection of a 12% increase over this period. However, between 2009 and 2015, the EIA (2009c) estimates that fuel economy will increase by 8.42%. The increased fleet fuel economy outweighs the increased driving so that total fuel consumption decreases by 2.5%. This differs from our baseline trends as we find that total fuel consumption increases by 11.5%, which is a consequence of our assumption of lower growth trends in passenger vehicle fuel economy.

Allocation of Cropland

In 2008, our model under predicts the land in corn, soybean and wheat production by 5.9%, 3.0% and 8.4% respectively, but overestimates the land in hay by 8.5% (NASS 2009). Likewise our model predicts the amount of land in CRP to be 5.1% above reported levels from the Farm Service Agency (FSA 2009).[104]

Consistent with the USDA Long Term Projections (2009), our model predicts that land used for corn production will increase slightly between 2009 and 2015. We predict that corn production will increase by 1.3% over this time, while the USDA projects an increase of 2.5%. Likewise, our model projects that land used for wheat production will fall by 3.0% between 2009 and 2015, which is consistent with USDA projections.[105] Unlike the USDA, we project that land dedicated to soybeans will

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[103] As discussed in the introduction, the CAFE standards will actually mandate an increase in overall vehicle fleet efficiency. However, we do not explicitly model these standards.

[104] The National Agricultural Statistics Service reports that 34.2 million hectares of corn, 30.2 million hectares of soybeans, 24.3 million hectares of hay and 22.5 million hectares of wheat were harvested (NASS 2009). Likewise 12.9 million hectares were held in the CRP (FSA 2009).

[105] The USDA Long Term Projections (2009) report a reduction of land harvested for wheat of 1.6% between 2009 and 2015.

remain relatively constant between 2009 and 2015, falling by only 0.47%. The USDA Long Term Projections estimate that the total land used to produce soybeans will fall by 4.0% over this same period.

In contrast to FAPRI (2009) estimates for hay production, our model predicts that land in hay production will increase between 2009 and 2015.[106] This is a result of the price of hay rising by 22% between 2009 and 2015 in our model and falling by 17% in the FARPI projections over the same period. Finally, the model predicts total enrollment in CRP to decrease gradually over time (0.6% per year) which differs from the assumptions made by the USDA (2009) that CRP enrollment will remain constant.[107]

Crop Exports

Our model estimates 2008 crop exports of 38.1 million metric tons (mmt) of corn, 20.6 mmt soybeans and 22.8 mmt wheat. These estimates are below USDA reported values by 16.8%, 31.4% and 17.6% for corn, soybeans and wheat respectively.[108] Between 2009 and 2015 we estimate that the exports of corn, soybeans and wheat fall by 6.8%, 4.3% and 6.6% respectively. This contrasts with USDA projections, which show corn and wheat exports to increase by 7.5% and soybean exports to remain constant over this time.

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[106] The USDA does not report projections for land used to produce hay, but FAPRI (2009) suggests that overall hay production will fall 2% by 2015. Our model estimates that hay production will increase by 1%.

[107] The 2008 Farm Bill (US Congress 2008) lowered the total maximum enrollment in CRP from 15.86 million hectares to 12.94 million hectares starting in October of 2009. We do not address the effect of this change in maximum enrollment in the model.

[108] The USDA reports crop exports in 2008 of 45.7 mmt corn and 25.2 mmt wheat (FAS 2009). The total quantity of soybean exports are not reported by the FAS, but the USDA Long Term Projections (2009) report a value of 32.5 mmt which represents a historic value from USDA sources.

Baseline Emissions Trends

Table 4-1. Baseline Emissions by Sector ($TgCO_2e$)

	2008	2009	2010	2011	2012	2013	2014	2015
Domestic Total	1406.7	1439.9	1465.4	1489.2	1515.7	1543.9	1572.6	1602.3
Transportation	1056.5	1080.0	1098.7	1117.0	1136.8	1157.5	1178.7	1200.4
Gasoline	999.4	1020.6	1032.5	1046.6	1064.2	1083.8	1104.5	1126.7
Ethanol	57.1	59.4	66.2	70.4	72.5	73.7	74.2	73.7
Fuel Production	295.6	302.5	310.1	316.7	322.6	328.5	334.1	339.5
Gasoline	255.7	261.1	264.2	267.8	272.3	277.3	282.6	288.3
Ethanol	39.9	41.4	46.0	48.9	50.4	51.2	51.5	51.2
Farm Input Production	44.6	44.4	44.6	44.7	44.8	44.8	44.7	44.7
Agriculture	63.4	68.4	73.8	76.5	79.2	82.1	84.3	86.6
Direct	109.5	109.6	110.3	110.8	111.1	111.4	111.7	111.8
SOC	-46.2	-41.2	-36.5	-34.3	-31.9	-29.4	-27.4	-25.2
Combustion Credit	-53.4	-55.5	-61.8	-65.8	-67.7	-68.9	-69.3	-68.9
ROW Crude	8821.9	8902.7	8983.5	9083.2	9182.8	9282.5	9382.2	9481.8

Gasoline Combustion

In 2008 we predict that 999.4 $TgCO_2e$ will be emitted from the combustion of gasoline in passenger vehicles (Table 4-1), compared to the EPA (2009a) estimate of 1141.2 $TgCO_2e$ in 2007. Our estimate is lower than, but not inconsistent with, the EPA estimate because there was a large increase (9.45 billion liters (RFA 2009a)) in the amount of ethanol consumed between 2007 and 2008, which displaces gasoline in the fuel supply. The expanded ethanol consumption would likely be reflected in future versions of the EPA Inventory. In addition, our estimate of gasoline consumption is low because our model under predicts total blended fuel consumption by 8.7% in 2008.

Ethanol Combustion Emissions and Emission Credit

The baseline simulation predicts that in 2008, the emissions from ethanol combustion are 57.1 $TgCO_2e$. This compares to the 2007 value reported by the EPA of 37.2 $TgCO_2e$. The deviation is primarily caused by the 9.5 billion liter increase in ethanol consumed between 2007 and 2008. An additional source of deviation is that unlike the EPA (2009a), we attribute the non-CO_2 greenhouse gases resulting from the combustion of ethanol directly to ethanol combustion as opposed to the larger transportation sector.

The EPA (2009a) implicitly assumes that the ethanol combustion credit is 37.2 $TgCO_2e$, in 2007, by not including the emissions from ethanol in their overall emissions estimates. This is comparable to the ethanol combustion credit that our model estimates given the different levels of ethanol consumption between 2007 and 2008.

Agricultural Production

The model estimates the net emissions from cropland agriculture to be 63.4 $TgCO_2e$ in 2008 (Table 4-1). It should be emphasized that our agricultural sector refers only to the production of four crops, while the EPA Greenhouse Gas Inventories include livestock production, pasture land and other crop production in the agricultural category. In the baseline scenario agricultural emissions increase by 26.5% between 2009 and 2015, as there is an overall increase in the production of corn, an emissions intensive crop, and a decrease in the sequestration potential of agricultural soils.

Agricultural Fossil Fuel Use

We estimate the emissions from fossil fuels in the agricultural sector to be 33.1 $TgCO_2e$ in 2008, 25.5 $TgCO_2$ of which resulted from on-farm combustion while 7.6 $TgCO_2$ resulted from the recovery and production of the fuels. The emissions from gasoline and diesel combustion resulted in emissions of 4.8 $TgCO_2e$ and 17.1 $TgCO_2e$ respectively, while natural gas (1.6 $TgCO_2e$) and LPG (2.0 $TgCO_2e$) emissions were smaller.

The EPA (2009a) reports agricultural emissions of 9.4 $TgCO_2e$ from the combustion of gasoline and 39.0 $TgCO_2e$ from the combustion of diesel. Our estimates are reasonable because cropland agriculture accounted for only 57% of total agricultural expenditures on energy in 2002 (Schenpf 2004).

N_2O Emissions

In 2008, the model predicts N_2O emissions to be 69.0 $TgCO_2e$ compared to the US EPA (2009a) estimate of 92.7 $TgCO_2e$ from major crop production in 2007. Analyzed another way, if all N_2O emissions are attributed to the application of N fertilizer, the IPCC methods used in our model would suggest that 1.8% of the mineral N in N fertilizer applied to soils would be released as N in N_2O, while the EPA estimate, which is based on DAYCENT simulations, would suggest an emissions factor 2.6%. Our estimate is low for two of reasons. First our model does not consider sorghum and cotton which accounts for 7.9% of the nitrogen fertilizer application to the major crops in the EPA inventory.[109] Second, the EPA uses the DAYCENT ecosystem model to predict N_2O emissions and is able to account for regional variation in weather patterns and soil types, as well as specific crop and management characteristics (fertilization method and timing as well as tillage) that are not captured in the IPCC methods (US EPA 2009a). Cropland N_2O emissions increase by 2.7% between 2009 and 2015 as a result of shifts to the production of corn, which requires large amounts of N fertilizer (Table 3-14), and away from the production of wheat and soybeans.

Liming

We estimate that the liming of agricultural soils resulted in the release of 7.4 $TgCO_2e$ in 2008, which is higher than the EPA estimate for 2007 of 4.1 $TgCO_2e$. The

[109] The EPA (2009a) also includes cotton and sorghum in the major crops category. Based on national average application rates from the ARMS and production estimates from NASS, these two crops accounted for approximately 7.9% of N fertilizer applied to major crops as reported by the EPA (603 Gg N of 7,587 Gg synthetic N applied in 2007).

reason for this deviation is that the EPA uses a different emissions factor for the application of lime than the IPCC factor we adopt. The EPA (2009a) use emissions factors for the application of limestone and dolomite from West and McBride (2005). These emissions factors (0.059 kgC/kg limestone and 0.064 kgC/kg dolomite) are 50% lower than the 0.12 kgC/kg limestone and 0.13 kgC/kg dolomite the IPCC recommends which leads to the deviation between our model and the EPA estimate.

Soil Organic Carbon Sequestration

Our model predicts that agricultural soils accumulated 12.6 TgC in 2008, leading to a net sequestration of atmospheric CO_2 of 46.1 $TgCO_2e$. The EPA estimate for 2007 of 42.3 $TgCO_2e$ (with a 95% confidence interval between 15.0 and 69.7 $TgCO_2e$) is lower than our estimate as the EPA estimate incorporates 20% more cropland. The main reason for this deviation is that our estimate of SOC accumulation for land held in CRP of 9.6 $TgCO_2e$. The EPA (2009a) estimates CRP soils to sequester 2.0 $TgCO_2e$ in 2007, however their calculation is based only enrollment changes in CRP since 2003 with the assumption that this new enrollment accumulates 0.5 mgC/ha per year. We allow land that was converted to CRP prior to 2003 to also accumulate carbon because CRP contracts range from 10 to 15 years, and because literature estimates suggest that soil carbon will be accumulated or lost for roughly 20 years after a management change.[110]

Consistent with historic trends in the EPA Greenhouse Gas Inventory (US EPA 2009a), the amount of carbon sequestered annually in agricultural soils decreases over time as management practices are maintained and soil carbon levels reach equilibrium levels for a large portion of cropland. The annual sequestration of CO_2 attributed to crop and CRP land falls by 45.3% by 2015 (Table 4-1). In percentage

[110] This literature is discussed in the SOC sections of Chapters 2 and 3.

terms, sequestration from CRP lands falls the most (67.8%) as a significant portion of this land first was set aside in mid to late 1980s, and will have stopped accumulating CO_2 by the mid to late 2000s.[111] Likewise reduced and conservation tillage systems were first adopted in the early 1980s so the sequestration benefits of lower intensity tillage are fully realized for much of this land between 2008 and 2015.

Rest-of-World Crude Oil Consumption

Our model predicts that the non-US consumption of crude oil will result in the emissions of 8,822 tgCO_2e in 2008. This is consistent with EIA (2009b) estimates for 2006 (8,638 TgCO_2e) and 2010 (8,982 TgCO_2e).[112] Following the increasing world consumption of petroleum, our mode predicts these emissions will steadily increase and reach 9,481.8 TgCO_2e by 2015. This is comparable to the EIA's IEO (2009a) which projects emissions 9480 TgCO_2e from non-US crude oil consumption in 2015.

Section III – Emissions Impacts of Increased RFS Mandate

As previously discussed, the overall emissions consequences of the Renewable Fuel Standard mandate can be decomposed into potential emissions savings and carbon leakage. Only if the potential emissions savings of the RFS mandate are larger than the sum of all leakages, will the mandate reduce overall greenhouse gas emissions. This section will describe the behavioral adjustments that drive each source of carbon leakage, and quantity the magnitude of each leakage relative to the potential emissions savings.

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[111]This rate is likely an under estimate of actual soil carbon sequestration on CRP land because our model operates on the cumulative hectares in CRP and does not account for land that might have entered or exited the program as a result of contract expiration and renewal.

[112] At the time of writing the EIA had not yet released its 2008 estimates for the total world emissions from petroleum combustion.

Potential Emissions Savings and Leakage in Domestic Fuel Markets

The stated environmental goal of the Renewable Fuel Standard is to reduce greenhouse gas emissions from domestic transportation by displacing gasoline with ethanol. As such, emissions savings of the RFS are dependent first, on the gap between the baseline and mandated levels of ethanol consumption and second, on the quantity of gasoline displaced by the additional ethanol. The increased quantity of ethanol consumption determines the potential emissions savings from the RFS in the absence of any market adjustments, while adjustments in the fuel market are the first source of carbon leakage we will discuss.

As shown in Table 4-2, the RFS impacts the consumption of ethanol only after 2009, when ethanol consumption increases by 2.8 billion liters. Between 2009 and 2015, the gap between the mandated quantity of ethanol and the baseline becomes substantially larger, with an additional 11.1 billion liters of ethanol consumed in 2015 as a result of the RFS.

Table 4-2. Impact of RFS on Ethanol Consumption (billion liters)

	2008	2009	2010	2011	2012	2013	2014	2015
Baseline Consumption	35.3	36.7	40.9	43.5	44.8	45.5	45.8	45.5
EISA Mandate	34.0	39.7	45.4	47.6	49.9	52.2	54.4	56.7
Actual Consumption	35.3	39.4	45.1	47.4	49.8	52.1	54.4	56.7
Additional Ethanol Due to Mandate	0.0	2.8	4.3	4.0	5.0	6.5	8.6	11.1
Does RFS Mandate Bind?	No	Yes	Yes	Yes	Yes	Yes	Yes	Yes

Potential Emissions Savings from Binding RFS Mandate

The potential emissions savings from the binding RFS mandate are the emissions that would occur if ethanol displaced gasoline in the fuel supply with no other market adjustments. As such, to estimate the maximum potential emissions savings of the binding RFS, each liter of mandated ethanol is assumed to displace a liter of gasoline. The main source of emissions savings from replacing gasoline with ethanol is a result of the 'ethanol combustion credit.' which accounts for the carbon released in the combustion of ethanol that was captured from the atmosphere during the growing of corn. The remaining emissions benefits result because the combustion emissions of ethanol are lower than the combustion emissions of gasoline. We refer to this as the 'substitution effect.'[113] Both sources of emissions savings behave linearly with the increased consumption of ethanol due to mandate (Table 4-5). In 2009, the RFS mandate could have resulted in emissions savings of 6.5 $TgCO_2e$, 64% of which could be attributed to the ethanol combustion credit. Following the increased gap between the baseline quantity of ethanol consumed and the RFS mandate, potential emissions savings increase to 26.3 $TgCO_2e$ by 2015.

Leakage in Domestic Fuel Markets

The emissions from transportation are a function of the fuel mix (the share of ethanol relative to gasoline) and the overall consumption of blended fuel. The extent to which the mandate alters these two factors determines the emissions consequences. Emissions related to the fuel mix are captured in the potential emissions savings of the RFS discussed above. The overall consumption of blended fuel is dependent on the RFS's impact on the price of blended fuel. Carbon leakage occurs if the price of blended fuel falls in response to a binding mandate. In contrast, if the price of blended fuel increases, there could be additional emissions benefits.

[113] There are two ways to view the emissions from ethanol combustion as the carbon that is released during the combustion of ethanol was captured from the atmosphere during the growing of corn. The first option is to ignore the emissions from the combustion of ethanol in the transportation sector. The second option is to measure the emissions from ethanol in the transportation sector, but give an emission 'credit' equal to the amount of CO_2 released through ethanol combustion to the agricultural sector. We use the second option, and therefore account for the emissions from ethanol combustion in this discussion. However, the size of the ethanol combustion credit is discussed here for clarity.

Behavioral Adjustments

The RFS has an ambiguous impact on the price of blended fuel (de Gorter and Just 2009a) which is the weighted average prices of ethanol and gasoline (equation (2.I.26)). The impact of the mandate is uncertain because the prices of ethanol and gasoline respond in opposite directions if the RFS binds. The mandate increases the demand for ethanol and its factors of production, corn in particular, which results in higher factor prices. This causes a subsequent increase in the price of ethanol, as the price of ethanol is the share weighted average price of its factors (equation (2.I.17)). Table 4-3 shows that in 2009 the RFS caused the price of ethanol to increase 1.9% above baseline levels. The magnitude of the ethanol price increase is dependent on the gap between baseline and mandated levels of ethanol consumption. In 2015, we project that the mandate will cause a 10.7% increase in the price of ethanol.

The price of gasoline is depressed by a binding mandate because the increased consumption of ethanol decreases the demand for gasoline. As a result, US demand for crude oil falls, lowering its world price, and subsequently reducing the marginal cost of gasoline production. These effects are illustrated in Table 4-3. In 2009, the RFS mandate decreases US demand for gasoline such that the world price of crude oil falls by 0.4%, which results in a reduction in the gasoline price of 0.3%. This effect is amplified as more ethanol is forced into the fuel supply by the mandate. In 2015, the crude oil price falls by 2.4%, and the gasoline price falls by 1.7%.

We find that the depressed price of gasoline offsets the increased price of ethanol and results in a lowering of the price of blended fuel between 2009 and 2013. In 2009, the blended fuel price falls by 0.07%, while in 2013 the price of blend fuel falls by 0.01%. This effect is consistent with the findings of de Gorter and Just (2009a) who present the necessary condition for a blended fuel price increase is for the price weighted elasticity of gasoline supply to be less than the price weighted share of ethanol, less the volumetric tax credit. This condition suggests that the price of blended fuel is more likely to increase as gasoline supply becomes more elastic relative to tethanol supply. Our results support this finding as between 2009 and 2013 the price weighted elasticity of ethanol ranges between 0.68 and 0.73, while the price weighted elasticity of gasoline ranges between 0.86 and 0.73.

After 2014, we find that the elasticity of ethanol supply becomes inelastic relative to the price of gasoline such that the necessary conditions for a blended fuel price increase occur. The less elastic ethanol supply is a result of ethanol production consuming a larger share of total corn production. In both 2014 and 2015, we find that the price of blended fuel increases by 0.01% and 0.03% respectively.[114] The direction of these results are consistent with de Gorter and Just (2009a), who find that the price of blended fuel will increase as a result of the mandate in 2015.

Table 4-3. Impact of RFS on the Prices of Fuel, VMT and Crude Oil

	2008	2009	2010	2011	2012	2013	2014	2015
Ethanol ($/liter)	0.67	0.63	0.61	0.60	0.59	0.57	0.56	0.55
% Change	0.00%	1.85%	3.28%	3.26%	4.28%	5.87%	7.96%	10.70%
Gasoline ($/liter)	0.53	0.51	0.49	0.48	0.47	0.45	0.44	0.43
% Change	0.00%	-0.31%	-0.51%	-0.50%	-0.65%	-0.90%	-1.24%	-1.70%
Blended Fuel ($/liter)	0.63	0.61	0.59	0.58	0.57	0.55	0.54	0.53
% Change	0.00%	-0.07%	-0.05%	-0.03%	-0.02%	-0.01%	0.01%	0.03%
Driving ($/km)	0.14	0.14	0.14	0.13	0.13	0.13	0.13	0.13
% Change	0.00%	-0.03%	-0.02%	-0.01%	-0.01%	0.00%	0.00%	0.01%
Crude Oil ($/barrel)	65.45	61.79	59.14	56.94	54.77	52.60	50.50	48.51
% Change	0.0%	-0.4%	-0.7%	-0.7%	-0.9%	-1.2%	-1.7%	-2.4%

In response to the lower price of blended fuel between 2009 and 2013, the price of VMT seen by the consumer falls, resulting in increased VMT and blended fuel consumption. In 2014 and 2015, the opposite occurs as the increased price of blended fuel reduces the consumption of VMT and blended fuel. The change in fuel consumption can be decomposed into two ‘fuel price’ effects: the ‘VMT’ effect and

[114] The price weighted elasticity of ethanol supply in our model is 0.74 and 0.75 in 2014 and 2015, while the price weighted elasticity of gasoline supply is 0.71 in both years.

the 'fuel economy' effect. The VMT effect is the result of the change in the price of VMT that leads consumers to alter their consumption of VMT and blended fuel. The fuel economy effect is a caused by consumers shifting away from (or towards) investments in fuel efficiency and towards (or away from) fuel expenditures in the production of VMT as the price of blended fuel changes.

The impacts of the change in the price of blended fuel on VMT, fuel economy and blended fuel consumption are illustrated in Table 4-4. In 2009, the consumption of VMT increased by 0.03%, while the overall fuel economy of the passenger vehicle fleet fell by 0.01%. Combined, these two effects cause an increase in blended fuel consumption of 0.04%. In 2013, the increased consumption of blended fuel due to the RFS is smaller (0.01%) as VMT increases by 0.01% and fuel efficiency decreases only slightly (0.001%). Between 2009 and 2013, the adjustments in the price of blended fuel effectively reduce the quantity of gasoline offset by the expanded RFS mandate.

In 2015, the increased price of blended fuel leads to a reduction in the consumption of VMT of 0.02% and an increase in fleet fuel economy of 0.003%. Overall, these effects lead to a decrease in blended fuel consumption of 0.02%. As such, the adjustments in the price of blended fuel actually result in more gasoline being displaced by the RFS than additional ethanol forced into the fuel supply.

Table 4-4. Impact of RFS on Consumption of Fuel and VMT

	2008	2009	2010	2011	2012	2013	2014	2015
Blended Fuel (billion liters)	440.11	450.14	459.12	467.45	475.89	484.54	493.24	501.92
% Change	0.00%	0.04%	0.04%	0.03%	0.02%	0.01%	-0.01%	-0.02%
Ethanol (billion liters)	35.27	36.69	40.87	43.46	44.77	45.51	45.80	45.52
% Change	0.00%	7.50%	10.42%	9.17%	11.11%	14.38%	18.69%	24.48%
Gasoline(billion liters)	404.84	413.44	418.25	423.99	431.12	439.02	447.44	456.40
% Change	0.00%	-0.63%	-0.97%	-0.90%	-1.14%	-1.48%	-1.92%	-2.47%
Driving (trillion kilometers)	4.06	4.14	4.22	4.30	4.37	4.45	4.53	4.60
% Change	0.00%	0.03%	0.03%	0.03%	0.01%	0.01%	0.00%	-0.02%
Fuel Economy (km/l)	9.22	9.20	9.19	9.19	9.19	9.18	9.18	9.17
% Change	0.00%	-0.01%	-0.01%	0.00%	0.00%	0.00%	0.00%	0.00%
Crude Oil (billion barrels)	2.54	2.59	2.62	2.66	2.71	2.76	2.81	2.87
% Change	0.00%	-0.62%	-0.96%	-0.89%	-1.10%	-1.42%	-1.83%	-2.33%

The ratio of gasoline displaced to additional ethanol consumption due to the mandate is reported in Table 4-22. This analysis shows that between 2009 and 2013, for each additional liter of mandated ethanol consumption, gasoline consumption decreases less than one liter. In 2009 each liter of mandated ethanol displaces 0.94 liters of gasoline. In 2013, the mandate is able to displace 0.99 liters. After 2013 however, each liter of ethanol mandated by the RFS displaces more than one liter of gasoline. In each year, the majority of the fuel price effect (more than 80%) is the result of increased demand for VMT, while disinvestments in fuel economy play a smaller role.

Carbon Leakage

The carbon leakage in the domestic fuel markets is a direct result of the increased consumption of blended fuel, and therefore occurs only from 2009 to 2013. The leakage in the fuel market offsets a small portion of the potential emissions savings in the transportation sector (Table 4-5). Specifically in 2009, the potential emissions savings (labeled 'Substitution Effect') in the fuel market were 2.3 $TgCO_2e$, but the increased consumption of blended fuel (the sum of the 'Fuel Economy' and 'VMT' effects) offsets 0.4 $TgCO_2e$ of these savings. The majority of this leakage (0.3 $TgCO_2e$) results from the VMT effect, while the fuel economy effect has a smaller impact (0.08 $TgCO_2e$). In 2013, the emissions savings from ethanol substitution of ethanol for gasoline of 5.6 $TgCO_2e$ are only slightly offset because of increased demand for VMT (0.07 $TgCO_2e$) and reduced fuel economy (0.01 $TgCO_2e$).

In 2015, the fuel price effects lead to additional emissions benefits. While the potential emissions savings of the mandate are 9.5 $tgCO_2e$, the increased price of blended fuel that results from the RFS leads to additional savings from the decreased consumption of VMT (0.3 $tgCO_2e$) and the increased fuel economy of the vehicle fleet

(0.04 tgCO2e). Overall, the emissions savings in the transportation sector in 2015 are 3% higher as a result of the fuel price effects.

Table 4-5. Impact of RFS on Emissions from Transportation Sector (TgCO_2e)

	2008	2009	2010	2011	2012	2013	2014	2015
Baseline Transportation	1056.49	1080.04	1098.67	1117.03	1136.75	1157.47	1178.71	1200.38
Total Change	0.00	-1.93	-3.16	-3.00	-4.03	-5.47	-7.34	-9.75
Substitution Effect	0.00	-2.34	-3.62	-3.38	-4.22	-5.56	-7.27	-9.46
Fuel Price Effects	0.00	0.41	0.46	0.39	0.19	0.08	-0.07	-0.29
Fuel Economy	0.00	0.08	0.06	0.04	0.03	0.01	-0.01	-0.04
VMT Effect	0.00	0.32	0.40	0.35	0.16	0.07	-0.06	-0.25
Ethanol Combustion Credit	-53.37	-55.52	-61.84	-65.76	-67.74	-68.86	-69.30	-68.88
Change Due to Mandate	0.00	-4.17	-6.45	-6.03	-7.53	-9.90	-12.95	-16.86
Potential Emissions Savings	0.00	-6.50	-10.06	-9.41	-11.75	-15.46	-20.22	-26.32

Leakage in the Fuel Production Sector

The effects of the increased RFS mandate on the emissions from fuel production depend on the total quantity of ethanol and gasoline consumed as blended fuel (Table 4-4). As such, there are two sources of carbon leakage that result from the ethanol mandate. First, as the emissions of ethanol production are 55% higher than the production emissions of gasoline (Chapter 3 Section II), a mandated increase in the share of ethanol in blended fuel results in higher fuel production emissions (labeled ‘substitution effect’ in Table 4-6). In 2009 this effect results in increased emissions of 1.4 TgCO_2e. The magnitude of this leakage is a function of the gap between baseline ethanol consumption and the mandate. Therefore, in 2015, the displacement of gasoline production with ethanol production leads to an emission of 5.5 TgCO_2e.

Secondly, as the mandate lowers the price of blended fuel and causes the overall consumption of blended fuel to increase, there is a subsequent expansion of the fuel production sector in each year from 2009 to 2013.[115] As shown in Table 4-4, the

[115] The VMT effect and the fuel economy effect are combined into a single category, referred to as the ‘Fuel Price Effects’.

fuel price effects on the consumption of blended fuel are small relative to the change in ethanol and gasoline consumption. It follows that the fuel price effects will be a small component of the carbon leakage from the fuel production sector (Table 4-6). We find that the depressed price of blended fuel leads to additional emissions of 0.1 $TgCO_2e$ in 2009 and 0.05 $TgCO_2e$ in 2013.

As in the transportation sector, there are additional emissions benefits in the fuel production sector in 2014 and 2015. As the total quantity of blended fuel produced drops due to the RFS in these two years, there are additional emissions savings of 0.02 $TgCO_2e$ in 2014 and 0.07 $TgCO_2e$ in 2015.

Table 4-6. Impact of RFS on Fuel Production Emissions ($TgCO_2e$)

	2008	2009	2010	2011	2012	2013	2014	2015
Baseline Total	295.63	302.54	310.13	316.67	322.65	328.47	334.11	339.46
Gasoline	166.14	169.67	171.65	174.00	176.93	180.17	183.62	187.30
Crude Recovery	89.56	91.46	92.52	93.79	95.37	97.12	98.98	100.96
Ethanol	39.93	41.41	45.97	48.88	50.35	51.18	51.51	51.20
Total Change	0.00	1.47	2.22	2.06	2.50	3.25	4.20	5.42
Substitution	0.00	1.37	2.10	1.96	2.45	3.23	4.22	5.49
Fuel Price Effects	0.00	0.10	0.12	0.10	0.05	0.02	-0.02	-0.07

Leakage in Domestic Agricultural Production

The effect of the RFS on agricultural emissions result from three behavioral adjustments: changes in the allocation of cropland to corn, soybeans, hay and wheat, the expansion of cropland at the expense of land held in CRP, and the intensification of rotations and tillage practices. With respect to our model, the combustion of fossil fuels, N_2O emissions, and CO_2 emissions from liming (referred to as 'direct' emissions) vary considerably by crop (Table 3-9), and marginally by rotations and tillage practices (Table 3-10). This variability also results in changes in emissions from farm input production. The 'other' agricultural emissions occur as a result of shifts between cropping practices and land uses. Converting land to certain uses, such as hay production, CRP or conservation tillage practices, increases the ability of the soils to accumulate organic carbon. Alternatively, if land that was held in one of these practices is converted to conventional agriculture, the accumulated carbon is released to the atmosphere as CO_2. A similar effect occurs on cropland that is set aside or placed in CRP as these lands accumulate carbon in the above ground plant matter and root systems.[116] It follows that when CRP is converted back to cropland, the carbon stored in the plant biomass is lost to the atmosphere through burning (to clear the land) or decomposition.

Effects of RFS on Crops and CRP

The expansion of the ethanol production sector due to the Renewable Fuel Standard significantly increases the demand for corn, driving the price of corn up. The magnitude of the corn price increase is largely dependent on the amount of ethanol added to the fuel supply by the mandate. In 2009, we project that ethanol consumption will increase by 2.8 billion liters and that the price of corn will increase by 4.7% relative to the baseline (Table 4-23). In 2015 the corn price increases by 21.5% as a result of an 11.1 billion liter increase in ethanol consumption. In response to these prices, the agricultural sector allocates more land to the production of corn at the expense of other crops. This reduces the supply of the other crops and leads to an increase in each of their prices (Table 4-23). For example, in 2015 the price of corn, soybeans and wheat increased by 9.0%, 14.5% and 13.6% due to the mandate. As a result of these price increases, the net returns from crop production increase relative to CRP rental payments and farmers have an incentive to bring CRP land into production.[117]

[116] See for example Fargione et al. (2008) or Righelato and Spracklen (2007).
[117] We abstract from other agricultural adjustment, such as an expansion of cropland into pasture or grazing land, or the displacement of other crops such as cotton or sorghum (US EPA 2009b).

When the RFS binds, the total land planted to corn increases relative to the baseline. In 2009 corn production expanded by 0.6% (0.2 million hectares) relative to baseline levels. As a result, the land used in the production of hay and wheat fell relative to the baseline by 0.3% and 0.6% respectively, and the land set aside to the CRP fell by 0.2%. In the same year, the RFS caused land devoted to soybean production to increase by 0.03 ha (0.1%). In 2015, the RFS's impact on the allocation of land to crops was much larger, with corn production increasing by 2.1%. This increase came mostly at the expense of wheat (2.2% reduction), CRP (0.9% reduction) and hay (0.8% reduction), while the land producing soybeans increases slightly (0.3%) as a result of the RFS.

Table 4-7. Impact of RFS on Allocation of Crops (million hectares)

	2008	2009	2010	2011	2012	2013	2014	2015
Corn Baseline	32.23	31.94	32.25	32.39	32.44	32.45	32.42	32.36
% Change	0.0%	0.6%	0.9%	0.8%	1.0%	1.3%	1.6%	2.1%
Soybeans Baseline	29.30	29.25	29.27	29.25	29.23	29.20	29.16	29.12
% Change	0.0%	0.1%	0.1%	0.1%	0.2%	0.2%	0.3%	0.3%
Hay Baseline	25.98	26.12	26.04	26.06	26.12	26.19	26.28	26.38
% Change	0.0%	-0.3%	-0.4%	-0.3%	-0.4%	-0.5%	-0.6%	-0.8%
Wheat Baseline	20.64	20.81	20.55	20.40	20.31	20.24	20.20	20.19
% Change	0.0%	-0.6%	-0.9%	-0.9%	-1.1%	-1.4%	-1.7%	-2.2%
CRP Baseline	13.52	13.56	13.51	13.49	13.48	13.47	13.47	13.48
% Change	0.0%	-0.2%	-0.3%	-0.3%	-0.4%	-0.5%	-0.7%	-0.9%

Our results for the increased production of corn due to the mandate is similar to, but lower than, the estimates of the US EPA (2009b) and Westhoff (2007). Specifically, the EPA (2009b) analysis uses the FASOM model to analyze the effects of the Renewable Fuel Standard in 2022, when an additional 9.8 billion liters of ethanol are consumed.[118] The EPA estimates that the mandate will result in an increase of corn production of 1.2 million hectares at the expense of soybeans (0.6

[118] This study only reports values for 2022. While this is not an ideal comparison, the quantity of ethanol added to the fuel supply is similar to our projections in 2015, so we expect our results to be comparable to this study.

million ha) and hay (0.3 million ha), and a slight increase in wheat production. The overall increase in agricultural land is 0.13 million hectares which is very close to our estimate of CRP lands converted to cropland in 2015. Westhoff (2007) finds that the RFS will result in an addition 9.14 billion liters of ethanol consumption in 2015, leading to an increase in corn production of 0.9 million ha, and a reduction in the land allocated to soybeans (0.5 million), hay (0.01 million ha), wheat (0.5 million) and CRP (0.14 million ha).

The major difference between our results and these studies is that we find that soybean production may increase with a binding RFS. This effect is a result of our modeling of crop rotations using CES functional forms. More discussion of this issue is provided below.

Effects of the RFS on Rotations and Tillage Practices

Another effect of the increased demand for corn and elevated crop prices are adjustments in crop rotations and tillage practices, which we refer to as agricultural 'intensification'. The impacts of the ethanol mandate on crop rotations are presented in Table 4-8. The increased price of corn caused farmers to increase the production of corn either by utilizing continuous corn rotations, or by shifting from other crops to rotations that include corn. We find that the area planted to continuous corn and corn-soybeans rotations increase by 2.0% and 0.3% as result of the RFS in 2009. These rotations displaced mainly the corn-soybeans-wheat and continuous wheat rotations, both of which fall by 0.6%. In addition, the quantity of land in continuous hay and continuous soybeans fall by 0.3% and 0.2% respectively. In 2015, the rotations displaced by the 6.7% increase in continuous corn and 0.9% increases in corn-soybeans are similar. Specifically, we estimate that land producing corn-soybeans-wheat will fall by 2.2%, continuous wheat will fall by 2.2%, continuous soybeans will fall by 0.6% and continuous hay falls by 0.8%.

Table 4-8. Impact of RFS on Rotations (million hectares)

	2008	2009	2010	2011	2012	2013	2014	2015
Continuous Corn	7.63	7.42	7.67	7.80	7.86	7.89	7.89	7.86
% Change	0.0%	2.0%	2.9%	2.6%	3.2%	4.1%	5.2%	6.7%
Continuous Soybeans	4.45	4.46	4.43	4.42	4.41	4.40	4.39	4.39
% Change	0.0%	-0.2%	-0.3%	-0.2%	-0.3%	-0.4%	-0.5%	-0.6%
Continuous Hay	25.98	26.12	26.04	26.06	26.12	26.19	26.28	26.38
% Change	0.0%	-0.3%	-0.4%	-0.3%	-0.4%	-0.5%	-0.6%	-0.8%
Continuous Wheat	5.53	5.57	5.50	5.46	5.44	5.42	5.41	5.41
% Change	0.0%	-0.6%	-0.9%	-0.9%	-1.1%	-1.4%	-1.7%	-2.2%
Corn-Soybeans	43.61	43.43	43.59	43.64	43.64	43.62	43.57	43.50
% Change	0.0%	0.3%	0.4%	0.4%	0.4%	0.6%	0.7%	0.9%
Corn-Soybeans-Wheat	20.96	21.13	20.87	20.71	20.62	20.56	20.52	20.50
% Change	0.0%	-0.6%	-0.9%	-0.9%	-1.1%	-1.4%	-1.7%	-2.2%

The overall increase in soybean production (Table 4-7) occurs because the reductions in continuous soybeans and corn-soybeans-wheat rotations are offset by the increases in the corn-soybean rotation. This effect is largely driven by our use of the CES functional forms which allocate land to each rotation practice largely on the basis of initial shares.[119] As the corn-soybean rotation is the predominant rotation used to produce corn, accounting for 70% of total corn production in 2003, an increase in corn production will lead to an increase in corn-soybeans production, which could lead to a net increase in the production of soybeans.

Comparable models have report similar effects. The ERS (2007) find that the RFS would increase corn-soybean rotations and overall soybeans in the Corn Belt by 2016. This study uses Regional Environmental and Agricultural Programming model (REAP) to estimate the impacts of 75.6 billion liters of ethanol consumption in 2016. While the structure of this model is very similar to our agricultural model in that both model a choice between crops, rotations and tillage systems using nested CES

[119] See equation (2.I.9) for example.

functions. The REAP model differs from our agricultural model because it is disaggregated to the Farm Production Region level.

The adjustment in cropping practices due to the mandate also changes the distribution of tillage practices. The total land managed with a given tillage system, aggregated across crops and rotations, is displayed in Table 4-9. The mandate has a very small effect on tillage practices, even in years when than mandate has large impacts on the allocation of crops and rotations. For example, in 2015, there is a small increase in the use of mulch (0.6%) and continuous tillage (0.1%) and a 0.1% reduction in the use of no-till due to the mandate. This effect is caused by the reduction of continuous soybeans and continuous wheat, both of which have large shares of no-till and little mulch tillage, and the expansion of continuous corn and corn soybeans, which have large shares of mulch tillage and small shares of no-till (Table 3-12).[120]

Table 4-9. Impact of RFS on Tillage Practices (million hectares)

	2008	2009	2010	2011	2012	2013	2014	2015
Conventional Tillage	30.41	30.35	30.78	31.02	31.18	31.36	31.51	31.59
% Change	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.1%	0.1%
Reduced Tillage	26.00	26.01	26.00	25.99	25.98	25.98	25.98	25.98
% Change	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	-0.1%
Mulch Tillage	26.53	26.49	26.45	26.42	26.39	26.35	26.31	26.28
% Change	0.0%	0.2%	0.2%	0.2%	0.3%	0.3%	0.4%	0.6%
No-Tillage	25.20	25.26	24.88	24.67	24.54	24.38	24.26	24.18
% Change	0.0%	-0.1%	-0.1%	-0.1%	-0.1%	-0.1%	-0.1%	-0.1%

The reduction in the use of no-till as a result of an expanded ethanol mandate is consistent with the previously mentioned USDA study (ERS 2007), which finds large increases of conventional and mulch tillage in the Corn Belt as a result of increased ethanol consumption.

[120] In the benchmark, only 8% of continuous corn was managed with no-till, while 42% and 32% were managed with conventional and mulch tillage respectively. In contrast, 38% and 34% of continuous wheat and continuous soybeans were managed with no-till (Table 3-12).

Carbon Leakage from Direct Agricultural Emissions

The expanded RFS causes total direct agricultural emissions to increase by 1.0 $TgCO_2e$ in 2015. The increased production and intensification of corn production are the main cause of this increase, leading to an additional 1.4 $TgCO_2e$ and 0.09 $TgCO_2e$ respectively. In addition, the increased production of soybeans leads to additional emissions of 0.4 $TgCO_2e$. The emissions from the production of each other crop fell as a result of the RFS. The largest reduction in emissions came from wheat production (0.4 $TgCO_2e$), while the emissions from hay production fell by 0.15 $TgCO_2e$. Each of these reductions are the result of the redistribution of land to corn production, as there were no emissions resulting from the intensification of non-corn crops. In earlier years, the changes in direct agricultural emissions were not as pronounced, as the gap between baseline ethanol consumption and the mandate was small. In 2009 for example, agricultural emissions increased only by 0.27 $TgCO_2e$, which was the result of a 0.4 $TgCO_2e$ emissions increase from the expansion and intensification of corn production, a small increase in emissions from soybean production (0.01 $TgCO_2e$) and a total reduction in emissions from hay and wheat production of 0.15 $TgCO_2e$.

Table 4-10. Impact of RFS on Direct Agricultural Emissions ($TgCO_2e$)

	2008	2009	2010	2011	2012	2013	2014	2015
Total Baseline	109.53	109.59	110.31	110.80	111.14	111.42	111.66	111.83
Change	0.00	0.27	0.41	0.38	0.46	0.60	0.77	1.00
Change in Crops	0.00	0.25	0.37	0.34	0.42	0.55	0.70	0.91
Intensification	0.00	0.03	0.04	0.03	0.04	0.05	0.07	0.09
Corn Baseline	63.70	63.38	64.22	64.71	65.00	65.21	65.34	65.38
Change	0.00	0.41	0.62	0.56	0.69	0.89	1.14	1.47
Expansion	0.00	0.39	0.58	0.53	0.65	0.83	1.07	1.38
Intensification	0.00	0.03	0.04	0.03	0.04	0.05	0.07	0.09
Soybeans Baseline	11.38	11.70	11.76	11.82	11.86	11.89	11.93	11.96
Change	0.00	0.01	0.02	0.01	0.02	0.02	0.03	0.04
Expansion	0.00	0.01	0.02	0.01	0.02	0.02	0.03	0.04
Intensification	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Hay Baseline	17.75	17.84	17.83	17.86	17.92	17.98	18.06	18.14
Change	0.00	-0.05	-0.07	-0.06	-0.07	-0.09	-0.12	-0.15
Expansion	0.00	-0.05	-0.07	-0.06	-0.07	-0.09	-0.12	-0.15
Intensification	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Wheat Baseline	16.70	16.67	16.50	16.41	16.36	16.34	16.34	16.35
Change	0.00	-0.10	-0.15	-0.14	-0.17	-0.22	-0.28	-0.36
Expansion	0.00	-0.10	-0.15	-0.14	-0.17	-0.22	-0.29	-0.36
Intensification	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

These results contrast significantly with the EPA's analysis of the RFS (US EPA 2009b) reports an emissions savings in domestic agricultural emissions of 2.3 $TgCO_2e$ in 2022. A large portion of this savings is due to the reduction in livestock production, but the EPA also reports reductions in emissions from the agricultural use of fossil fuels (0.2 $TgCO_2e$) as well as soil N_2O emissions from fertilizer application (0.93 $TgCO_2e$). For the energy combustion category we estimate an increase in emissions of roughly 0.2 $TgCO_2e$ in 2015, primarily due to the large diesel energy requirements of corn production (Table 3-9). The EPA's estimate of emissions savings comes primarily from a large reduction in gasoline consumption, which is a result of large regional variability in the use of gasoline to produce corn.

The difference between the EPA (2009a) analysis and our work in terms of N_2O emissions are a result of FASOM using the IPCC 1996 Guidelines (IPCC 1997) to estimate N_2O emissions, compared to our use of the 2006 IPCC methods. In the 1996 methods, a large quantity of emissions are attributed to N-fixing crops which increases the per hectare N_2O emissions from soybeans by well over 200%. These emissions have been removed from updated IPCC (2006) methods because studies have shown N_2O emissions of soybean production can be well modeled based on N fertilizer and crop residue inputs.[121] As the EPA expects the increased corn production to displace mostly soybean production, total N_2O emissions would decrease if the updated IPCC methods were used, because the EPA estimates an increase in N_2O emissions from N fertilizer application of 0.4 $TgCO_2e$ which is offset by a reduction in the emissions from N-fixing crops (soybeans) of 1.2 $TgCO_2e$.

The analysis of the direct agricultural emissions consequences of the RFS offers some interesting insight. First, the intensification of agricultural practices is a much smaller source of leakage than adjustments in crops produced. In each year, the aggregate changes in rotations and tillage practices accounted for less than 10% of total change in direct agricultural emissions. This is caused by the limited variability in input use between rotations and tillage practices relative to the input variability between crops (Table 3-9 and Table 3-10). Second, the increased emissions from the expansion and intensification of corn production are partially offset (roughly 35% in each year the mandate binds) by reductions in emissions from other crops.

Carbon Leakage from Farm Input Production

As the distribution of crops, rotations, and tillage practices adjusts, the quantity of fertilizers demanded also change (Table 4-24). In 2009, the total emissions from input production increase by 0.3% (0.14 $TgCO_2e$) as a result of the RFS. The majority of this increase (0.7 $TgCO_2e$ and 0.5 $TgCO_2e$ respectively) are caused by the increased production of N fertilizer and agricultural lime (Table 4-24) required for additional corn production. In 2015, total emissions from farm input production increase by

[121] The IPCC (2006) cite Rochette and Jenzen (2005).

1.1% (0.5 $TgCO_2e$).[122] As in 2009, the total change in emissions is mostly the result of an increased production of N fertilizer (0.23 $TgCO_2e$) and agricultural lime (0.17 $TgCO_2$). The carbon leakage that results from increased farm input production is roughly 50% less than the leakage from direct agricultural emissions.

Table 4-11. Impact of RFS on Farm Input Production Emissions ($TgCO_2e$)

	2008	2009	2010	2011	2012	2013	2014	2015
Total	44.62	44.42	44.63	44.73	44.76	44.77	44.75	44.70
% Change	0.0%	0.3%	0.5%	0.4%	0.5%	0.7%	0.9%	1.1%
N Fertilizer	20.35	20.26	20.36	20.40	20.42	20.42	20.41	20.39
% Change	0.0%	0.3%	0.5%	0.4%	0.5%	0.7%	0.9%	1.1%
P Fertilizer	3.79	3.79	3.79	3.80	3.80	3.80	3.80	3.80
% Change	0.0%	0.1%	0.2%	0.2%	0.2%	0.3%	0.4%	0.5%
K Fertilizer	2.48	2.47	2.48	2.49	2.49	2.49	2.49	2.49
% Change	0.0%	0.3%	0.4%	0.4%	0.5%	0.6%	0.8%	1.0%
Lime	10.53	10.46	10.53	10.57	10.58	10.59	10.58	10.56
% Change	0.0%	0.5%	0.7%	0.6%	0.8%	1.0%	1.3%	1.6%
Other	7.47	7.45	7.47	7.47	7.47	7.47	7.47	7.46
% Change	0.0%	0.1%	0.2%	0.2%	0.2%	0.3%	0.4%	0.5%

Carbon Leakage from Other Agricultural Emissions

In addition to increasing direct emissions from agricultural production, the RFS causes a reduction in the carbon stored in agricultural soils and the biomass on CRP land. We find that while the reduction in SOC sequestration is relatively small, the emissions from the lost biomass carbon on CRP land are significantly greater than the increased direct agricultural emissions.

Soil Carbon

Even in years were there mandate causes large increases in ethanol consumption, there are relatively small impacts on land planted to hay and CRP (Table 4-7) and the use of reduced tillage practices (Table 4-9). As a result, the effects on soil carbon sequestration are small. In 2015, when the impacts of the mandate on land

[122] This is comparable to the increase in farm chemical production reported by the EPA (2009b). They estimate that the RFS will increase chemical production emissions by 0.3 $TgCO_2e$ in 2022.

allocation were the greatest, the total loss of carbon from agricultural soils was 0.6 $TgCO_2e$. The bulk of these emissions (0.35 $TgCO_2e$) were the result of increased tillage intensity and the reduction in hay production (0.18 $TgCO_2e$). The conversion of land held in CRP to cropland contributed much fewer emissions (0.04 $TgCO_2e$). In years were the impact of the mandate is smaller, the impact of the RFS on soil carbon stocks is minimal. For example, in 2009 the RFS caused only an additional 0.18 $TgCO_2e$ of carbon to be lost by agricultural soils.

Table 4-12. Impact of RFS on SOC Emissions ($TgCO_2e$)

	2008	2009	2010	2011	2012	2013	2014	2015
Total Baseline	-46.15	-41.15	-36.54	-34.32	-31.89	-29.35	-27.35	-25.23
Hay Baseline	-4.07	-4.28	-4.36	-4.44	-4.52	-4.61	-4.71	-4.83
Tillage Baseline	-31.24	-29.38	-27.04	-25.06	-23.21	-21.26	-19.16	-16.91
CRP Baseline	-10.85	-7.50	-5.14	-4.81	-4.15	-3.48	-3.48	-3.49
Total Difference	0.00	0.18	0.25	0.23	0.28	0.36	0.46	0.58
Hay Difference	0.00	0.10	0.09	0.09	0.10	0.11	0.13	0.18
Tillage Difference	0.00	0.04	0.12	0.11	0.15	0.21	0.29	0.35
CRP Difference	0.00	0.03	0.03	0.03	0.03	0.04	0.04	0.04

Biomass Losses from CRP Conversion

The conversion of land allocated to the Conservation Reserve Program to cropland results in substantial carbon emissions through the release of carbon stored in above-ground and below-ground biomass (Table 4-13). Although, the amount of land that comes out of CRP as a result of the mandate is small, the amount of biomass carbon released through CRP conversion is greater than both the direct emissions from agriculture and the total release of carbon from SOC. For example, in 2009 the mandate resulted in emissions from lost biomass of 0.7 $TgCO_2e$, emissions from SOC of 0.2 $TgCO_2e$ and an increase in direct emissions of 0.3 $TgCO_2e$. Likewise, in 2015, the emissions from lost CRP biomass (3.6 $TgCO_2e$) are substantially larger than the increased direct agricultural emissions (1.0 $TgCO_2e$) and the emissions from changes in SOC (0.6 $TgCO_2e$).

Table 4-13. Emissions from Biomass Lost Due to Conversion of CRP

	2008	2009	2010	2011	2012	2013	2014	2015
Losses in CRP (million ha)	0.00	0.02	0.04	0.04	0.05	0.07	0.09	0.12
Lost Biomass Emissions (TgCO2e)	0.00	0.71	1.20	1.17	1.51	2.04	2.72	3.60

Carbon Leakage in International Crude Oil Market

In years when the ethanol mandate binds, the decreased demand for gasoline from the transportation sector lowers US demand for crude oil and causes the world oil price to decrease (Table 4-3). In response to the lower price, the international consumption of crude oil increases, resulting in increased greenhouse gas emissions. The increases in overall crude oil consumption are very small relative to the total quantity of crude oil consumed; only increasing by 0.12% in 2009 and 0.7% in 2015. The resulting changes in emissions are also very small in comparison with total world emissions from crude consumption, but this leakage is extremely large relative to the potential savings of the ethanol mandate and the domestic sources of leakage. In 2009, the increased consumption of crude oil results in an increased emission of 10.9 TgCO_2e, while in 2015 emissions increased by 67.8TgCO_2e.

Table 4-14. Impact of RFS on World Crude Oil Consumption Emissions

	2008	2009	2010	2011	2012	2013	2014	2015
Consumption (mbd)	65.50	66.10	66.70	67.44	68.18	68.92	69.66	70.40
% Change	0.00%	0.12%	0.20%	0.20%	0.27%	0.37%	0.52%	0.71%
Emissions (tgCO2e)	8,821.8	8,902.7	8,983.5	9,083.2	9,182.8	9,282.5	9,382.2	9,481.8
% Change	0.00	10.91	18.29	18.30	24.49	34.57	48.54	67.75

Carbon Leakage in International Land Markets

Impact of RFS on Crop Exports

Crop exports decrease when the RFS binds, because the domestic prices of all crops increases (Table 4-23), lowering the competitiveness of US exports. In addition, corn that would have been exported in the baseline is diverted for use in the ethanol sector, or to replace corn that would have been used in food production. Exports of soybeans and wheat also decrease because there is an overall reduction in the US production of non-corn crops as land has been diverted to corn production.

The reductions in crop exports are greatest when there is a large gap between the baseline quantity of ethanol consumed and the mandated quantity of ethanol (Table 4-15). Corn exports drop the most as a result of the mandate in 2009, falling by 3.0%. Moreover, wheat exports drop by 1.7% and soybean exports fall by 1.1%. In later years, the reductions in crop exports caused by the RFS are much larger as more land is diverted to corn production (Table 4-7) and as the increases in domestic crop prices are larger (Table 4-23). In 2015, the percentage reductions in crop exports from the benchmark are 11.9% for corn, 5.1% for soybeans and 6.8% for wheat.

Table 4-15. Impact of RFS on Crop Exports (million metric tons)

	2008	2009	2010	2011	2012	2013	2014	2015
Corn Baseline	38.05	38.67	37.19	36.35	35.97	35.81	35.83	36.01
% Change	0.0%	-3.0%	-4.7%	-4.4%	-5.5%	-7.2%	-9.3%	-11.9%
Soybeans Baseline	20.56	22.29	21.90	21.68	21.52	21.40	21.37	21.33
% Change	0.0%	-1.1%	-1.9%	-1.8%	-2.2%	-3.0%	-3.9%	-5.1%
Wheat Baseline	22.77	22.30	21.68	21.30	21.08	20.93	20.85	20.83
% Change	0.0%	-1.7%	-2.6%	-2.5%	-3.1%	-4.0%	-5.3%	-6.8%

In terms of weight, the reductions in corn, soybeans and wheat exports in 2015 are 4.3, 1.1 and 1.4 million metric tons respectively. These results are consistent with a number of other studies. However our reductions in wheat exports are higher in comparison. The EPA (2009b) estimate reductions in corn exports in 2022 of 5.19 mmt using the FASOM model and 8.49 mmt using the FAPRI model. Likewise, both models estimate a reduction in soybeans (0.94 mmt using FASOM and 0.71 mmt using FAPRI) and the FAPRI model estimates a reduction in wheat exports of 0.71 mmt. As expected the FAPRI analysis of the RFS (Westhoff 2007) finds very similar reductions in crop exports compared to the EPA analysis. They estimate that corn exports will fall by 6.6 mmt and soybean exports will fall by 0.87 mmt in 2015 as a result of an increase in ethanol production of 9.15 billion liters.

The crop export reductions reported by Searchinger et al. (2008) are significantly different than our analysis because they are measuring the effects of a much larger increase in ethanol consumption. They find that corn exports will be reduced by 38.5 mmt, soybean exports reduced by 6.4 mmt and wheat exports reduced by 8.1 mmt.

Emissions Consequences of Reduced US Crop Exports

We model international land use change such that any decrease in US crop exports, leads to an expansion of cropland worldwide (equation (2.II.27)). The effects of the reduced crop exports from the RFS on international land use change and the resulting emissions are reported in Table 4-16. In 2009, the mandate causes an additional 0.5 million hectares of land to be brought into production, leading to emissions of 130.1 $TgCO_2e$. In later years when the mandate's impacts on crop exports are more severe, the emissions from land use change are significantly greater. For example, the RFS causes 1.9 million hectares of land to be converted to cropland in 2015, resulting in emissions of 472.7 $TgCO_2e$.

Table 4-16. Rest of the World Land Use Change and Emissions

	2008	2009	2010	2011	2012	2013	2014	2015
Land Converted (million ha)	0.00	0.53	0.82	0.75	0.92	1.18	1.52	1.94
LUC Emissions (tgCO2e)	0.00	130.08	198.85	182.91	223.82	288.88	371.26	472.71

Our estimates of land use change emissions are consistent with, but lower than Searchinger et al. (2008) and US EPA (2009b). If we divide our estimate of land use change emissions in 2015 by the increase in ethanol consumption due to the mandate, we calculate an emission of 42.4 $kgCO_2e$/liter of additional ethanol. Searchinger et al. (2008) predict total undiscounted land use change emissions resulting from a 56 billion liter increase in corn ethanol production of 3,796.8 $TgCO_2e$ or 67.8 $kgCO_2e/l$. Likewise, the US EPA (2009b) calculates undiscounted land use change to be 51.1 $kgCO_2e/l$ ethanol added by the mandate in 2022. As their analysis estimates that 9.83

billion liters of ethanol will be added by the mandate, the total international land use change emissions estimated by the EPA are 502.3 $TgCO_2e$.

Total Emissions Impacts of the RFS

Our analysis of the emissions consequences of the expanded Renewable Fuel Standard corn ethanol mandate can be summarized in four points. First, we find that even if the full potential emissions savings of the RFS are achieved, the emissions savings will be insignificant compared to total US emissions. Between 2009 and 2015, the maximum potential emissions savings of the RFS was 26.3 $TgCO_2e$, which occurred in 2015 (Table 4-17). This compares to total net US emissions of 6,087.5 $TgCO_2e$ in 2007 (US EPA 2009a). Assuming that US emissions remain at the same level until 2015, the RFS corn ethanol mandate would directly only reduce emissions by 0.4%.[123]

Second, domestic carbon leakages offset close to 50% of the potential emissions savings of the RFS (Table 4-17). The main sources of domestic leakage are the agricultural and fuel production sectors. The increased production of ethanol offsets at least 21% of potential emissions savings in each year the RFS mandate binds as the production of ethanol is more emissions intensive than the production gasoline. The emissions resulting from adjustments in the agricultural sector offset at least 20% of potential emissions savings between 2009 and 2015 (Table 4-17). As shown in Table 4-10 and Table 4-13 the main components of the agricultural leakage are the increased direct emissions that result from an expansion in corn production, and the emissions consequences of converting land in CRP to cropland.

In the years 2009 through 2013, there is also a minor domestic leakage in domestic blended fuel market that occurs because the mandate depresses the price of

[123] Historic trends suggest that US emissions will continue to rise (US EPA 2009a).

blended fuel. In 2009, this leakage offsets 6% of potential emissions savings and offsets only 1% of potential emissions savings in 2013. In 2014 and 2015, the fuel price effects lead to additional emissions benefits, as total blended fuel consumption falls. In 2015, this price effect results in additional emissions savings of 1% relative to the potential emissions savings of the mandate.

Third, consistent with Searchinger et al. (2008) the carbon leakage from international land use change is several orders of magnitude larger than the potential savings from ethanol use. We find that even if US crop exports are only slightly depressed by the RFS, the resulting land use change emissions are large enough to offset potential savings by over 1800% (Table 4-17).

Finally, we find that the Renewable Fuel Standard's impact on the international crude oil market will totally offset all potential savings independent of the gap between baseline ethanol consumption and the mandate. Between 2009 and 2015, the carbon leakage from crude oil markets are at least 200% larger the potential emissions savings (Table 4-17). The real-world implications of this leakage are clear. While many are debating the legitimacy of indirect land use change (Renewable Fuels Association (2009b); Kim et al. (2009)), the carbon leakage in the crude oil market also challenges the environmental benefits of a biofuel mandate.

Table 4-17. Carbon Leakage as Percent of Potential Emissions Savings

	2008	2009	2010	2011	2012	2013	2014	2015
Potential Savings (TgCO2e)	0.00	-6.50	-10.06	-9.41	-11.75	-15.46	-20.22	-26.32
Domestic Fuel Markets	0%	-6%	-5%	-4%	-2%	-1%	0%	1%
Fuel Production	0%	-23%	-22%	-22%	-21%	-21%	-21%	-21%
Domestic Agriculture	0%	-20%	-21%	-21%	-21%	-21%	-21%	-22%
ROW Crude Oil Markets	0%	-168%	-182%	-194%	-208%	-224%	-240%	-257%
ROW Land Markets	0%	-2000%	-1976%	-1943%	-1904%	-1869%	-1836%	-1796%
Total Leakage	0%	-2216%	-2205%	-2185%	-2157%	-2135%	-2118%	-2094%

Sensitivity Analysis

Here we consider the stability of our central results to different parameter assumptions that affect total greenhouse gas emissions. First, we investigate the sensitivity of the leakage in domestic agriculture to different N_2O emissions factors following the assessment of Crutzen et al. (2008) and Smeets et al. (2009). Second, following Keeney and Hertel (2010) we analyze the impact of international crop yield trends on the international land market leakage. Third, we assess how the assumed rest-of-world demand elasticity for crude oil affects the crude oil market leakage. Finally, we assess the impact of world crude oil price projections on the overall emissions results of our model.

Agricultural N_2O Emissions

As previously discussed, there is considerable uncertainty in the N_2O released from cropland. Our central case uses the IPCC methods for estimating N_2O emissions, which is similar to assuming that 1.8% of the nutrient N applied to cropland is released as nitrogen in N_2O. With this assumption, the total leakage from the domestic agricultural sector ranges between 20% and 22% of the potential savings of the RFS (Table 4-18) for 2009 to 2015. However, the contribution of increased N_2O emissions to the total leakage is small, accounting for an overall leakage of only 3%. This suggests that modifications to the N_2O emissions factor will have a minor effect on both the magnitude of the domestic agricultural leakage, and also the overall emissions impact of the RFS mandate.

Along with the IPCC default values, we estimate N_2O emissions using an emissions factor of 3% following Smeets et al. (2009), and 5% which is the highest emissions factor estimate of Crutzen et al. (2008). In our central case, we find that increased emissions from cropland N_2O emissions result in a leakage of 3% of potential emissions savings. Using the 3% emissions factor, we find that this leakage increases to between 4% and 5% of potential emissions savings. The assumption of a 5% emissions factor has a similarly minor impact. This assumption increases the total cropland N_2O leakage to 6% of potential emissions savings.

Table 4-18. Magnitude of N_2O Leakage under Alternative Emissions Factors

	2008	2009	2010	2011	2012	2013	2014	2015
Potential Savings (TgCO2e)	0.00	-6.50	-10.06	-9.41	-11.75	-15.46	-20.22	-26.32
Non-N2O Agricultural Leakage	0%	-17%	-18%	-18%	-18%	-19%	-19%	-19%
Agricultural N2O Leakage								
Central	0%	-3%	-3%	-3%	-3%	-3%	-3%	-3%
3% N2O Emissions Factor	0%	-5%	-5%	-4%	-4%	-4%	-4%	-4%
5% N2O Emissions Factor	0%	-6%	-6%	-6%	-6%	-6%	-6%	-5%

International Crop Yields

Along with our central case, in which yields reached 4.07, 2.46 and 3.14 metric tons per hectare for corn, soybeans and wheat respectively, we consider four international crop yield projections. We construct two pessimistic scenarios. The 'no yield change' scenario holds yields constant at 2008 levels or 3.85, 2.25 and 3.05 metric tons per hectare for corn, soybeans and wheat respectively. The '10% below' scenario represents the worst case where yields are 10% below projected levels in 2015. In this case corn, soybeans and wheat yields decline from 2008 levels to 3.69, 2.24 and 2.84 metric tons per hectare by 2015. We also consider two optimistic scenarios, where crop yields are 10% and 25% higher levels than projected levels in 2015. In the 10% higher yields scenario, corn yields reach 4.5 metric tons per hectare by 2015, while soybeans and wheat yields reach 2.69 and 3.45 metric tons per hectare respectively. In the 25% higher yields scenario, yields reach 5.03, 3.02 and 3.9 metric tons per hectare for corn, soybeans and wheat respectively.

As was expected, the pessimistic crop yield scenarios result in a larger leakage, although the assumption of zero yield improvement resulted in a leakage that was only 6% larger than the central case in 2015 (Table 4-19). Likewise the 10% lower yield scenario, results in only a 10% larger leakage. While the optimistic scenarios reduce the leakage from international land use change, the magnitudes of the leakage under these assumptions are still substantially greater than the potential savings of the mandate. In 2015, the 10% higher yields scenario produced a leakage that was 16.4 time larger that potential emissions savings, while the 25% higher yield scenario produced a leakage that was 14.5 times greater than potential emissions savings.

Table 4-19. International Land Use Leakage under Alternative Yield Scenarios

	2008	2009	2010	2011	2012	2013	2014	2015
Potential Savings (TgCO2e)	0.00	-6.50	-10.06	-9.41	-11.75	-15.46	-20.22	-26.32
Central	0%	-2000%	-1976%	-1943%	-1904%	-1869%	-1836%	-1796%
No Yield Improvement	0%	-1999%	-1993%	-1980%	-1961%	-1942%	-1924%	-1902%
10% Below 2015 Estimates	0%	-1999%	-2007%	-2008%	-2003%	-1997%	-1992%	-1983%
10% Above 2015 Levels	0%	-1999%	-1942%	-1881%	-1817%	-1756%	-1699%	-1641%
25% Above 2015 Levels	0%	-1999%	-1896%	-1795%	-1699%	-1610%	-1530%	-1453%

Rest-of-World Crude Oil Demand Elasticity

The long-run estimates for the elasticity for non-US demand for crude oil range between -0.1 and -0.5 (Chapter 3, Section I). We use these bounds to check the sensitivity of the crude oil market leakage to the assumed elasticity. In our central case, the carbon leakage in the crude oil market was roughly 2 times larger than the potential savings of the RFS in each year for which the mandate binds (Table 4-20). With the low elasticity assumption, the leakage in the world crude oil market is less than the potential emissions savings of the RFS. For example, in 2009, 56% of the potential emissions savings are offset by increased crude consumption in the low elasticity case, compared to a leakage of 168% in our central elasticity case. In the high crude oil demand elasticity case, the leakage in the crude oil market is substantially larger than the potential savings (279% in 2009). These relationships are similar in 2015 with crude oil leakage under the low and high crude demand elasticity scenarios offsetting 86% and 429% of potential emissions savings respectively, compared to a leakage of 257% in the central case.

Table 4-20. Crude Oil Leakage under Alternative Elasticity Assumptions

	2008	2009	2010	2011	2012	2013	2014	2015
Potential Savings (TgCO2e)	0.00	-6.50	-10.06	-9.41	-11.75	-15.46	-20.22	-26.32
Central Elasticity (-0.3)	0%	-168%	-182%	-194%	-208%	-224%	-240%	-257%
Low Elasticity (-0.1)	0%	-56%	-61%	-65%	-69%	-75%	-80%	-86%
High Elasticity (-0.5)	0%	-279%	-303%	-324%	-347%	-373%	-400%	-429%

Crude Oil Price Path

We model ethanol to be a perfect substitute for gasoline in the production of blended fuel, so the quantity of ethanol consumed is primarily dependent on the relative prices of corn (the main input in ethanol production) and crude oil (the main input in gasoline production). As the potential emissions savings and carbon leakage from the RFS mandate are determined by the quantity of gasoline that is displaced by ethanol, it follows that the crude oil price projection will have a large impact on the magnitude of emissions. The sensitivity of our central results to the crude oil price is tested using the high and low crude oil price paths discussed in Section I of this chapter.

In the high crude oil price path, baseline ethanol consumption increases from 53.7 billion liters in 2009 to 74.9 billion liters in 2015. As such, the RFS mandate, which increases from 39.7 to 56.7 billion liters over the same time period, does not bind in any year. It follows that the potential emissions savings of the RFS mandate are zero, and there is no carbon leakage.

In the low crude oil price path, baseline ethanol consumption increases from 21.3 in 2008 to 41.0 in 2015. It should be noted that using the low crude oil price path our model predicts a value for ethanol consumption in 2008 which is 30% below reported levels (EIA 2008a). This causes our model to estimate the RFS mandate to be binding in 2008. The additional ethanol forced into the fuel supply increases from 14.0 billion liters in 2009 (compared to 2.8 billion liters in the central price path) to 15.7 billion liters in 2015 (compared to 11.1 billion liters in the central price path).

As a larger quantity of ethanol is being added to the fuel supply in the low crude oil price scenario, the potential emissions savings are also significantly larger. In 2009, potential emissions savings are roughly five times higher in the low crude price case relative to the central crude price case (33.05 $TgCO_2e$ compared to 6.5 $TgCO_2e$). In 2015, the potential emissions savings under the low price scenario are 40% higher than the potential savings in the central case (Table 4-21).

While the potential emissions savings of the RFS mandate are larger in the low crude price scenario, each source of carbon leakage increases in proportion. As such, the percent of the potential emissions savings offset by the various sources of carbon leakage in the low price case are very similar to the comparable values in the central price case. For example, in 2009, the leakage in the domestic agricultural sector is 17% and 20% of potential emissions savings in the low crude price and central crude price scenarios respectively. Likewise the leakage in the international crude oil market is similar in 2015 at 291% of total savings in the low price case and 257% of total savings in the central crude price case. In total, the percent of potential emissions savings offset by leakage is nearly identical for the central and low crude oil price cases, with leakages roughly 22 times larger than potential emissions savings in 2009 and 21 times larger than potential savings in 2015.

Table 4-21. Carbon Leakage in Low Crude Price Scenario

	2008	2009	2010	2011	2012	2013	2014	2015
Potential Savings -$TgCO_2e$	-29.56	-33.05	-38.17	-37.32	-39.00	-38.45	-38.77	-37.01
Domestic Fuel Markets	-6%	-11%	-13%	-16%	-11%	-9%	-7%	-3%
Fuel Production	-23%	-24%	-24%	-25%	-24%	-23%	-23%	-22%
Domestic Agriculture	-17%	-18%	-19%	-19%	-19%	-20%	-20%	-21%
ROW Crude Oil Market	-193%	-208%	-226%	-245%	-263%	-276%	-285%	-291%
ROW Land Markets	-1976%	-2011%	-1981%	-1950%	-1912%	-1878%	-1844%	-1806%
Total Leakage	-2215%	-2272%	-2263%	-2256%	-2228%	-2205%	-2178%	-2143%

The sensitivity analysis shows that our central results are stable with respect to the choice of N_2O emissions factors and trends in international crop yields, but highly sensitive to the crude oil price path and the elasticity of non-US crude oil demand. In particular, changing the N_2O emissions factors from the IPCC recommendations to the 5% emissions factor suggested by Crutzen et al. (2008) only increases the leakage in the agricultural sector from 22% to 24% of total emissions savings in 2015. Likewise, the most optimistic international crop yield scenario only reduces international land use change emissions in 2015 by 19% relative to the central case.

With respect to the price of crude oil, we find that in the higher price path, the RFS will have no impact on greenhouse gas emissions as the mandate for corn ethanol will not bind. Given the low crude oil price path, the potential emissions savings much are larger than the central crude path. However, relative to the potential emissions savings, the magnitudes of the carbon leakages are very similar between the central and low crude oil price scenarios. Finally, the non-US elasticity of crude oil demand has a large impact on the leakage in the international crude oil markets. We found that the reported values for this elasticity range from 0.1 to 0.5, and the resulting leakage in 2015 using these bounds falls in the range of 86% to 429% of potential emissions savings.

APPENDIX B

Table 4-22. Gasoline Displacement Ratio

	2008	2009	2010	2011	2012	2013	2014	2015
Gasoline Displaced:Additional Ethanol	0.00	0.94	0.96	0.96	0.98	0.99	1.00	1.01
Driving Effect	0.00	0.05	0.04	0.04	0.01	0.00	0.00	-0.01
Fuel Economy Effect	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.00

Table 4-23. Baseline Crop Prices and Changes Due to Mandate ($/Metric Ton)

	2008	2009	2010	2011	2012	2013	2014	2015
Corn Baseline	138.69	135.26	143.63	148.76	151.24	152.28	152.10	150.97
% Change	0.0%	4.7%	7.7%	7.2%	9.1%	12.2%	16.2%	21.5%
Soybeans Baseline	338.39	295.70	304.63	309.59	313.67	316.60	317.20	318.35
% Change	0.0%	1.9%	3.2%	3.0%	3.9%	5.2%	6.9%	9.0%
Hay Baseline	130.94	127.19	135.98	142.23	146.87	150.68	153.47	155.73
% Change	0.0%	3.3%	5.4%	5.0%	6.3%	8.4%	11.1%	14.5%
Wheat Baseline	153.49	159.33	167.81	173.19	176.50	178.81	180.03	180.49
% Change	0.0%	3.1%	5.0%	4.7%	5.9%	7.8%	10.3%	13.6%

Table 4-24. Changes in Fertilizer Use due to Mandate (1000 metric tons)

	2008	2009	2010	2011	2012	2013	2014	2015
N Fertilizer	6806.48	6774.48	6808.41	6824.09	6829.10	6830.12	6826.83	6819.49
% Change	0.0%	0.3%	0.5%	0.4%	0.5%	0.7%	0.9%	1.1%
P Fertilizer	3647.40	3641.26	3648.08	3651.97	3653.90	3654.89	3655.09	3654.64
% Change	0.0%	0.1%	0.2%	0.2%	0.2%	0.3%	0.4%	0.5%
K Fertilizer	3595.03	3581.08	3595.38	3602.41	3604.96	3605.59	3604.43	3601.69
% Change	0.0%	0.3%	0.4%	0.4%	0.5%	0.6%	0.8%	1.0%
Lime	129.57	128.91	129.53	129.80	129.85	129.83	129.71	129.52
% Change	0.0%	0.3%	0.5%	0.5%	0.6%	0.7%	1.0%	1.2%

Table 4-25. Sources of Corn Used to Meet Expanded RFS Mandate

	2008	2009	2010	2011	2012	2013	2014	2015
Corn Required (Million Hectares)	0.00	0.70	1.06	0.97	1.20	1.55	1.99	2.55
Agricultural Adjustments	0%	28%	27%	27%	27%	27%	27%	27%
Diversion from End-Uses	0%	72%	73%	73%	73%	73%	73%	73%
Food	0%	29%	30%	31%	31%	32%	32%	33%
Co-Products	0%	26%	26%	26%	25%	25%	25%	25%
Exports	0%	17%	17%	16%	16%	16%	16%	16%

Chapter 5. Comparisons with Lifecycle Analysis

Many studies that analyze the emissions consequences of biofuel have relied on lifecycle analysis methods.[124] This method attempts to quantify and allocate all the emissions that result from the existence of one unit of fuel, specifically ethanol and gasoline. A lifecycle emissions factor for a given fuel represents all emissions from the combustion and production of the fuel, the transportation of the fuel, and the production and transportation of all inputs to the fuel production processes. This section compares the assumptions and emissions estimates of lifecycle analysis with the per unit change in emissions estimated within a general equilibrium (GE) framework.

Section I – Analytical Models

To compare the lifecycle analysis and general equilibrium frameworks, stylized models of the change in greenhouse gas emissions resulting from a unit increase in ethanol consumption are developed for each method. We find that both methods rely on the same set of lifecycle emissions factors, but contain very different assumptions for how behavior adjusts to increases in ethanol consumption.

Lifecycle Emissions Model

The lifecycle emissions metric for a biofuel attempts to quantify the total emissions that result from the production and use of one unit of ethanol and all required inputs. It follows that each stage of the lifecycle has a specific emissions coefficient which quantifies both the direct emissions, plus the emissions from the production of inputs used in that sector, per unit of biofuel.[125] The lifecycle emissions of gasoline are calculated in the same manner and the difference between the two

[124] See discussion of lifecycle studies in Chapter 1.
[125] We denote these emissions factors ϕ, following the model description in Chapter 2.

lifecycle estimates is considered to be the economy-wide emissions savings of replacing a unit of gasoline with a unit of biofuel.

The lifecycle of corn ethanol is generally decomposed into two stages, the production of ethanol and co-products and the production of corn feedstock.[126] The lifecycle emissions from ethanol production ($\bar{\phi}_{Fe}^{P}$), include all direct emissions from energy inputs (coal, natural gas, electricity) used at the ethanol plant, plus the emissions from producing these energy inputs.[127] To estimate the emissions from the corn used to produce a unit of ethanol, the lifecycle emissions of producing corn on a unit of land ($\bar{\phi}_{AG}^{k1}$) are attributed to a unit of ethanol based on the volumetric fuel yield (λ_{k1}), which quantifies the units of land planted to corn required to produce a unit of ethanol. The factor, $\bar{\phi}_{AG}^{k1}$, includes the direct emissions from energy use, N_2O and liming, in addition to the emissions from the production of farm inputs.[128] The final component of the corn ethanol LCA is a measurement of the emissions offset by the co-products of ethanol production. While these co-products can substitute for a variety of products, this relationship is simplified here. It is assumed that for each unit of ethanol production, co-products are generated that replace a fixed quantity of land growing corn, λ_{CO}. Mathematically, the lifecycle emissions for one unit of ethanol ($\bar{\phi}_{FE}$) would be written:

$$\bar{\phi}_{Fe} = \bar{\phi}_{Fe}^{P} + (\lambda_{k1} - \lambda_{CO})\bar{\phi}_{AG}^{k1}. \tag{5.I.1}$$

[126] This follows Farrell et al. (2006) and Hill et al. (2006). To simplify the notation, we do not consider the emissions from the combustion of ethanol and the combustion credit in the analytical model.
[127] Note that the lifecycle emissions with a 'bar' accent character, $\bar{\phi}$ are used to described emissions factors that are unaffected by policies or price changes. In the standard LCA, all coefficients are assumed to be fixed in this sense. Here we assume that there is no efficiency adjustment or fuel switching in the production of ethanol as a result of the mandate.
[128] This differs from the analysis in Chapter 4 where the emissions from farm input production were attributed to a distinct sector.

Similarly, the lifecycle emissions for of regular gasoline would include the lifecycle emissions of gasoline production and crude oil recovery ($\bar{\phi}_{Fg}^{P}$),[129] and the emissions from combusting gasoline in passenger vehicles ($\bar{\phi}_{Fg}^{C}$). The total lifecycle emissions for gasoline ($\bar{\phi}_{Fg}$) would be expressed as:

$$\bar{\phi}_{Fg} = \bar{\phi}_{Fg}^{C} + \bar{\phi}_{Fg}^{P}. \tag{5.I.2}$$

Following the standard lifecycle technique, the total change in emissions (ψ_{LCA}) from replacing one unit of gasoline with one unit of ethanol would simply be the difference between the total lifecycle emissions of ethanol and the total lifecycle emissions of gasoline. Describing the change in emissions mathematically and grouping terms by sector would result in expression:

$$\psi_{LCA} = -\bar{\phi}_{FG}^{C} + \left(\bar{\phi}_{Fe}^{P} - \bar{\phi}_{Fg}^{P}\right) + (\lambda_{k1} - \lambda_{CO})\bar{\phi}_{AG}^{k1}. \tag{5.I.3}$$

<u>General Equilibrium Emissions Model</u>

The goal of the general equilibrium analysis is to construct a metric that is comparable to the LCA emissions statistic (ψ_{LCA}) described above, but that incorporates all changes in emissions from our general equilibrium analysis. To construct this metric (ψ_{GE}), the total change in emissions resulting from expanded ethanol consumption is averaged across the additional units of ethanol. Analytically, the expression for general equilibrium change in emissions from ethanol can be derived from the equation for total greenhouse gas emissions.

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[129] The parameter used here, $\bar{\phi}_{Fg}^{P}$, is equivalent to $(\phi_{Fg}^{P} + \phi_{Fg}^{R})$ in our emissions model (Chapter 2).

Grouping the terms of equation (2.II.1) such that GHG_{k1} are the direct lifecycle emissions from land producing corn (A_k) and GHG_{LU} are the total lifecycle emissions of all other land uses ($\boldsymbol{A_{LU}}$), yields:[130]

$$GHG = GHG_{Trans} + GHG_{Fe} + GHG_{Fg} + GHG_{k1} + GHG_{LU} + GHG_{R_W} \tag{5.I.4}$$

which can be expanded in terms of lifecycle emissions factors:[131]

$$GHG = Fg\bar{\phi}^{C}_{Fg} + Fe\bar{\phi}^{P}_{Fe} + Fg\bar{\phi}^{P}_{Fg} + A_{k1}\phi^{k1}_{AG} + \boldsymbol{A_{LU}\phi^{LU}} + R_W\bar{\phi}^{R}. \tag{5.I.5}$$

Differentiating this function with respect to the quantity of ethanol (Fe), to represent an increase in ethanol consumption, and recognizing that Fg, A_{k1}, A_{LU} and R_W are all functions of Fe, produces:

$$dGHG/dFe = \left(\frac{dFg}{dFe}\right)\bar{\phi}^{C}_{Fg} + (dFe)\bar{\phi}^{P}_{Fe} + \left(\frac{dFg}{dFe}\right)\bar{\phi}^{P}_{Fg} + \left(\frac{dA_{k1}}{dFe}\right)\phi^{k1}_{AG} + \left(\frac{d\boldsymbol{A_{LU}}}{dFe}\right)\boldsymbol{\phi^{LU}} + \left(\frac{dR_W}{dFe}\right)\bar{\phi}^{R} \tag{5.I.6}$$

Finally, letting θ represent the change in the use of a given input or final product x, divided by the change in ethanol consumption:

$$\theta_x = \frac{\left(\frac{dx}{dFe}\right)}{dFe}$$

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130 The other land uses category includes the behavioral adjustments and emissions of domestic non-corn cropland, the conversion of CRP to cropland, changes in soil organic carbon stocks and international land use change.

131 The emissions from ethanol combustion are again assumed to be zero to simplify the expression. The land use variables are bolded to represent a vector of land uses and emissions factors.

and normalizing the total change in emissions to a unit of ethanol, provides the expression for the general equilibrium emissions estimate:[132]

$$\psi_{GE} = \theta_{Fg}\bar{\phi}^{C}_{Fg} + \left(\bar{\phi}^{P}_{Fe} - \theta_{Fg}\bar{\phi}^{P}_{Fg}\right) + \left(\theta_{k1}\phi^{k1}_{AG} - \boldsymbol{\theta}_{LU}\boldsymbol{\phi}^{LU}\right) + \theta_{R_W}\bar{\phi}^{R}. \quad (5.I.7)$$

The final expression represents the total change in emissions in terms of lifecycle emissions coefficients, and behavioral adjustments resulting from expanded ethanol consumption. It should be noted that the other land use emissions can be decomposed to the behavioral adjustments and emissions factors of non-corn domestic agricultural production ($\boldsymbol{\theta}_{K\neq 1}$ and $\boldsymbol{\phi}^{\mathrm{k}\neq 1}_{AG}$), the conversion of CRP land to cropland and changes in soil carbon sequestration ($\boldsymbol{\theta}_{DLU}$ and $\boldsymbol{\phi}^{DLU}$) and international land use change ($\boldsymbol{\theta}_{ILU}$ and $\boldsymbol{\phi}^{\mathrm{ILU}}_{PV}$) such that:

$$\boldsymbol{\theta}_{LU}\boldsymbol{\phi}^{LU} = \boldsymbol{\theta}_{K\neq 1}\boldsymbol{\phi}^{k\neq 1}_{AG} + \boldsymbol{\theta}_{DLU}\boldsymbol{\phi}^{DLU} + \boldsymbol{\theta}_{ILU}\boldsymbol{\phi}^{ILU}_{PV}. \quad (5.I.8)$$

For comparison purposes, the sectoral emissions equations for the LCA and GE methods are presented in Table 5-1.

Table 5-1. Comparison of LCA and General Equilibrium Emissions Models

Sector	LCA	General Equilibrium Analysis
Fuel Combustion	$\bar{\phi}^{C}_{Fg}$	$\theta_{Fg}\bar{\phi}^{C}_{Fg}$
Fuel Production	$\bar{\phi}^{P}_{Fe} - \bar{\phi}^{P}_{Fg}$	$\bar{\phi}^{P}_{Fe} - \theta_{Fg}\bar{\phi}^{P}_{Fg}$
Agriculture	$(\lambda_{k1} - \lambda_{CO})\bar{\phi}^{k1}_{AG}$	$\theta_{k1}\phi^{k1}_{AG} - \boldsymbol{\theta}_{LU}\boldsymbol{\phi}^{LU}$
Crude oil	N/A	$\theta_{R_W}\bar{\phi}^{R}$

[132] For comparison with the LCA expression, the expected signs of each θ for an increase in ethanol consumption have been added here. That θ_{Fg} is negative and $\theta_{k1}$is positive is straight forward, but the sign of $\boldsymbol{\theta}_{LU}$ is ambiguous as land producing other crops will fall but the amount of cropland worldwide will increase.

As illustrated in Table 5-1 there are two main differences between the LCA and GE estimates. First, the GE method incorporates behavioral adjustments, θ into the estimates of emissions. As shown in Chapter 4, the adjustments in gasoline and crude oil consumption, and land uses in response to increased ethanol consumption are dependent not only on amount of ethanol added to the economy but other economic factors, such as the relative prices of crude oil, gasoline, corn and ethanol, and economic trends, such as domestic and international crop yields and improvements in passenger vehicle fuel economy. This suggests that the GE method may provide a different estimate for the per unit change in emissions for different mandated ethanol requirements, and different underlying economic conditions. As the LCA estimate is a function of only fixed lifecycle emissions factors, the predicted change in emissions will not be altered by the magnitude of biofuel policy or subsequent economic adjustments.

The other major difference is that the general equilibrium method incorporates the impact of increased ethanol use on sectors that are not part of the corn lifecycle. This plays a critical role in the international energy markets as well as the domestic and international land markets. In LCA, there is no estimate of the change in demand for crude oil that results from the ethanol use. This means that the emissions changes from international energy consumption are not captured by the LCA (Table 5-1). In terms of the land uses, only the emissions corn production is consider in LCA.[133] In the GE estimate, the change in emissions from all other uses of land, domestic and

[133] It should be noted that some more recent studies (Searchinger et al. 2008) have incorporated other land uses into the LCA model. These studies have focused on the impacts of US biofuel policy on the conversion of international native lands to biofuel production and not on adjustments in domestic land uses and cropland. These studies essentially add the adjustment in international land uses ($\boldsymbol{\theta}_{ILU}$) and estimated emissions (ϕ_{PV}^{ILU}) to the standard lifecycle calculations.

international, are incorporated, in addition to the change in emissions from corn production.

Section II – Comparison of Lifecycle and General Equilibrium Methods

Here, the emissions savings of lifecycle analysis are compared to the emissions savings estimated using the general equilibrium framework when applied to the expanded Renewable Fuel Standard. First, we use LCA estimates to generate a reference point for our general equilibrium analysis. Next, we estimate the behavioral parameters (θ) using the general equilibrium framework. The behavioral parameters are then incorporated into the standard LCA to determine how the behavioral adjustments implied by LCA impact estimated emissions savings. Finally, the results of the general equilibrium analysis are extended to cellulosic biofuel.

Estimated Lifecycle Emissions Savings

Following the equations 3 and (5.I.2), the LCA emissions of corn ethanol and regular gasoline have been calculated using the data from the economic model's 2008 solution (Table 5-2).[134] In total, the lifecycle emissions for gasoline are 3.10 kgCO-$_2$e/liter, while the lifecycle emissions from ethanol production are 1.82 kgCO_2e/liter. Therefore, the change in greenhouse gas emissions predicted by the LCA method (ψ_{LCA}) would be1.28 kgCO_2e/liter.

Table 5-2. Standard LCA Savings Relative to Gasoline for 2008 (kgCO_2e/liter)

	Gasoline	Ethanol	Savings
Total	3.10	1.82	1.28
Combustion	2.47	0.11	2.36
Production	0.63	1.13	-0.50
Agriculture	-	0.77	-0.77
Co-products	-	-0.18	0.18

[134] Note that the LCA emissions savings of gasoline versus ethanol will grow over time as improvements in corn yields and ethanol conversion technology outweigh the intensification in corn farming (Table 5-6). Also, note that the ethanol CO_2 credit is accounted for in the ethanol combustion category, so the emissions reported are only non-CO_2 emissions from ethanol combustion.

While our estimated emissions from ethanol production, corn production, and the complete lifecycle of gasoline are consistent with the LCA literature, our estimate of the co-product credit is slightly low compared to other studies. Our emissions from ethanol production (1.13 kgCO_2e/liter) are lower than the estimate of Farrell et al. (2006) who report ethanol production emissions of 1.35 kgCO_2e/liter.[135] However, other studies report ethanol production emissions that are well below our estimate (Liska et al. (2009); Wang et al. (2007)). Likewise, our estimate of emissions from corn production of 0.77 kgCO_2e/liter are within the literature estimates, which range from 0.93 kgCO_2e/liter (Wang, M. Wu, and Huo 2007) to 0.61 kgCO_2e/l (Liska et al. 2009).

Our co-product credit (-0.18 kgCO_2e/l) is lower than the estimates of Farrell et al. (2006), Wang et al. (2007) and Liska et al. (2009). This difference is driven by the quantity of animal feeds that are assumed to be displaced by ethanol co-products.[136]

A final deviation from the LCA literature is that we report results in terms of volume, by comparing a liter of ethanol to a liter of gasoline, while common practice is to report results in energy equivalent units.[137] Ethanol is much less energy dense than gasoline and therefore a larger volume of ethanol is required to match the energy stored in gasoline. In this sense comparing a liter of ethanol to a liter of gasoline is inadequate. However, in the economic model, the consumer chooses to purchase blended fuel without knowledge of the energy content of that fuel. Therefore the more natural comparison for our analysis is in terms of volume. For example, the main

[135] This difference is a result of the assumed share of fossil fuels used in ethanol production. Specifically, we assume that 33% of ethanol production occurs in coal fired ethanol plants while Farrell et al. (2006) assume that 60% of ethanol plants use coal.
[136] For example, Farrell et al. (2006) assume that ethanol co-products displace 0.93 kg of corn, 0.18 kg of soybean meal 0.10 kg soy oil, 0.03 kg urea per liter ethanol produced. We assume that co-products only displace 0.72 kg corn and 0.025 kg soybeans per liter ethanol produced.
[137] See for example Farrell et al. (2006) and Hill et al. (2007a).

result of Farrell et al. (2006) is that the use of ethanol provides 18% emissions savings over an energy equivalent unit of gasoline. In terms of volume, the savings reported is 50% or a savings of 1.64 kgCO_2e per liter of ethanol. Other studies have found similar results in terms of emissions savings per liter of ethanol including Wang et al. (2007), Liska et al. (2009) and Hill et al. (2006).

Incorporating Behavioral Adjustments

Standard lifecycle analysis does not have a mechanism to capture behavioral adjustments. Instead, behavioral adjustments are implied by the manner in which the lifecycle emissions are calculated. Further, by comparing the LCA expression to the GE expression (Table 5-1), the behavioral adjustments implicitly assumed in LCA can be derived. As the LCA and GE estimates rely on identical emissions factors, there is a certain set of behavioral adjustments for which the two methods would estimate the same change in emissions. How well the behavioral adjustments implied by LCA match the behavioral adjustments predicted in the GE method will determine how the final emissions predictions compare. The discussion that follows will compare the behavioral adjustments predicted by the GE analysis to the behavioral adjustments implied by LCA and will discuss how each behavioral assumption impacts the emissions savings estimated by LCA.

The normalized behavioral adjustments (the θ's) as estimated by the general equilibrium analysis for the expanded RFS are reported in Table 5-3. We find that the behavioral results of the general equilibrium analysis differ substantially from the assumptions of LCA in terms of domestic and international land use and world crude oil consumption, but are comparable to the LCA assumptions for domestic fuel markets.

The effects of incorporating behavioral adjustments on the emissions savings estimated using LCA are reported in Table 5-4. We find that LCA's restrictive behavioral assumptions can cause both underestimates (domestic corn and non-corn production and domestic fuel markets) as well as overestimates (domestic fuel markets, domestic and international land use and crude oil consumption) of emissions savings. If only domestic behavioral adjustments are incorporated into LCA, offsetting effects would cause estimated emissions savings to be very close to those of a standard LCA. When international adjustments are considered, the domestic savings from ethanol use are completely offset by large emissions increases in the rest of the world.

Table 5-3. Estimated θ's from General Equilibrium Analysis

	LCA	2008	2009	2010	2011	2012	2013	2014	2015
θ_{Fg} - l gasoline/ l ethanol	1.00	0.00	0.94	0.96	0.96	0.98	0.99	1.00	1.01
θ_{k1} - ha corn/1000 l ethanol	0.19	0.00	0.07	0.07	0.07	0.06	0.06	0.06	0.06
θ_{k2} - ha soybeans/1000 l ethanol	0.00	0.00	0.01	0.01	0.01	0.01	0.01	0.01	0.01
θ_{k3} - ha hay/1000 l etanol	0.00	0.00	-0.03	-0.02	-0.02	-0.02	-0.02	-0.02	-0.02
θ_{k4} - ha wheat/1000 l ethanol	0.00	0.00	-0.05	-0.05	-0.04	-0.04	-0.04	-0.04	-0.04
θ_{CRP} - ha CRP/1000 l ethanol	0.00	0.00	-0.01	-0.01	-0.01	-0.01	-0.01	-0.01	-0.01
θ_{ILU} - ha/1000 l ethanol	0.00	0.00	0.19	0.19	0.19	0.18	0.18	0.18	0.17
θ_{RW} - barrel/l ethanol	0.00	0.00	0.01	0.01	0.01	0.01	0.01	0.02	0.02

Table 5-4. Impact of Behavioral Adjustments on LCA Emissions (kgCO_2e/liter)

	2008	2009	2010	2011	2012	2013	2014	2015
LCA	1.31	1.32	1.33	1.34	1.35	1.35	1.36	1.36
Domestic Fuel Markets	0%	-14%	-10%	-9%	-4%	-1%	1%	2%
Domestic Corn	0%	24%	24%	24%	24%	24%	23%	23%
Other Domestic Cropland	0%	5%	5%	5%	5%	4%	4%	4%
All Domestic Land	0%	-24%	-26%	-26%	-27%	-27%	-27%	-27%
International Land Use	0%	-3566%	-3498%	-3424%	-3340%	-3263%	-3191%	-3108%
ROW Crude	0%	-285%	-312%	-334%	-362%	-389%	-418%	-448%

The top row 'LCA' represents the emissions savings that are calculated when using the standard LCA assumptions: $\theta_{Fg} = 1$, $\theta_{k1} = 0.19$, $\boldsymbol{\theta}_{LU} = 0$ and $\theta_{Rw} = 0$; Each subsequent row reports the percent change in lifecycle emissions that occurs by allowing a given behavioral parameter, in parenthesis, to match that of the general equilibrium analysis.

Domestic Gasoline Consumption

In the transportation and fuel production sectors, the difference in the LCA and GE emissions estimates is determined by the adjustment in domestic gasoline consumption in response to increased ethanol consumption (θ_{Fg}). The assumption made in the LCA methods is that a unit of ethanol displaces a unit of gasoline, such that θ_{Fg} is equal to 1 (Table 5-3). As discussed in prior sections, the GE estimate for the amount of gasoline displaced by ethanol is dependent on how the mandate affects the price of blended fuel suggesting that θ_{Fg} could be greater than or less than 1.

In the years 2009 through 2013, the general equilibrium analysis finds that for each liter increase in ethanol consumption there is a less than one liter decrease in gasoline consumption (Table 5-3). In 2009, each liter of mandated ethanol displaces only 0.94 liters of gasoline, while in 2013 each liter of mandated ethanol displaces 0.99 liters of gasoline. In 2015, each liter of mandated ethanol displaces slightly more than one liter of gasoline (1.01 liters).

These results are very close to the LCA assumption of a 1 to 1 displacement. This suggests that for the fuel combustion and fuel production sectors the estimated change in emissions of the LCA and GE methods will be similar. However, the LCA methods over estimate emissions savings compared to the GE method from 2009 to 2013 and under estimate emissions savings in 2015. In 2009 the assumption that a liter of ethanol displaces a liter of gasoline causes LCA to over estimate emissions by 14% relative to the GE method if all other LCA behavioral assumptions remain unchanged (Table 5-4). As the reduction in the price of blended fuel due to the RFS is smaller in 2013, the 1 to 1 displacement ratio assumption of LCA leads only to a 1% over estimate of emissions savings relative to the GE methods. In 2015, this assumption leads to a 2% underestimate of emissions savings compared to the GE methods.

Corn Production

In the general equilibrium analysis, the increased land devoted to corn production (θ_{k1}) is limited by the diversion of corn from other end uses, as well the increased use of co-products in the food sector. If θ_{k1} is close to 0, most of the corn used for ethanol is being diverted from other end uses, while if the value is higher, more corn is being produced on land that would have otherwise not been producing corn.

LCA assumes that the additional land devoted to corn would be equal to the land required to produce a unit of ethanol less the land that is no longer used because of ethanol co-products. As such, the LCA and GE emissions estimates for corn production would be equivalent only if θ_{k1} is equal difference between λ_{k1} and λ_{CO}. Explicitly, in 2009, every 1000 liters of ethanol produced requires 2.5 metric tons of corn, but co-products are produced that are equivalent to 0.68 metric tons of corn in food production. Given corn yields of 9.86 metric tons per hectare, the LCA assumption is that for each 1000 liters of mandated ethanol, corn would need to be grown on an additional 0.18 hectares (Table 5-3).[138]

The behavioral assumption made by LCA is substantially different than the projected behavior of the GE analysis. Between 2009 and 2015, for every 1000 liters of additional ethanol consumption, our model estimates an increase of less than 0.07 hectares of corn production, because the majority of the corn used by the expanded ethanol sector is diverted from the food sector and crop exports (Table 4-25). This immediately suggests that LCA will overestimate the change in emissions from corn production, and estimate overall lifecycle emissions savings that are less than the savings estimated by the GE methods if all other LCA assumptions are maintained.[139]

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[138] λ_{k1} = 0.25 ha corn per 1000 liters ethanol and λ_{CO} = 0. 07 ha corn per 1000 liters ethanol.

[139] For this analysis the emissions consequences of taking corn away from other sectors, specifically food production, are not considered. However, if the production of livestock was reduced, there could

In fact, between 2009 and 2015, the LCA assumption for increased corn production causes LCA to under estimate the emissions savings by 23-24% relative to the GE analysis (Table 5-4), if all other LCA assumptions were maintained.

<u>Other Land Uses</u>

A standard LCA does not incorporate the impacts of ethanol on other land uses, which reflects an assumption that an increase in ethanol consumption will not change how non-corn producing land is used. In the analytic model, the implied assumption of standard LCAs is that $\boldsymbol{\theta}_{LU}$ equals zero for all non-corn domestic and international land. As there are a variety of different land uses, we will discuss each land use following the decomposition in equation (5.I.8).

Other Domestic Cropland

The LCA assumption that the domestic production of other crops is not impacted by increased ethanol consumption ($\boldsymbol{\theta}_{k\neq 1}$ = 0) contrasts with our general equilibrium analysis of the RFS (Table 5-3). Instead, between 2009 and 2015, for each 1000 liters of mandated ethanol, the amount of land producing hay drops by between 0.02 and 0.03 hectares and the amount of land producing wheat drops by between 0.05 and 0.04 hectares. In addition, there is an increase of 0.01 hectares of land producing soybeans for every 1000 liters of mandated ethanol. As there are reductions in the production of two relatively emissions intensive crops, wheat and hay, the LCA estimate of emissions savings under predicts the emissions savings of a liter of ethanol by between 4% and 5% (Table 5-4) by not incorporating adjustments in non-corn agricultural production.

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be substantial emissions benefits (US EPA 2009b). The emissions consequences of diverting corn from US export are captured in the emissions from international land use change.

As corn and other crop production are competing for a relatively fixed quantity land, there is an inverse relationship between changes in the land used for corn production ($\boldsymbol{\theta}_k$) and changes in the land used to produce other crops ($\boldsymbol{\theta}_{k\neq 1}$). If there is a large increase in the amount of corn produced per liter of ethanol produced, then there would also be a large decrease in the land used to produce other crops. As shown above, this relationship serves to mitigate some of the emissions increases from expanded corn production.

Other Domestic Land-Use Change

The general equilibrium analysis also predicts that in response to expanded ethanol consumption, there will be a decrease in the amount of land held in CPR ($\theta_{CRP} < 0$). For years when the RFS mandate binds, each additional 1000 liters of ethanol consumed leads to a reduction of land in CRP of 0.01 hectares. While this is a small adjustment, each hectare held in CRP has significant sequestration benefits that are lost with conversion. As a result, incorporating changes in enrollment in the CRP into the lifecycle analysis reduces the estimated total emissions savings by 24% and 27% relative to a standard LCA in 2009 and 2015 respectively (Table 5-4).

Relative to LCA, the reduction in emissions savings from the conversion of CRP to cropland are as large as the increased emissions savings estimated for adjustments in corn and non-corn agriculture. As such, if all domestic behavioral adjustments, in both agricultural and fuel markets, are incorporated into a lifecycle analysis, the resulting emissions savings are 10% lower than a standard LCA in 2009 and 2% higher in 2015.

International Land Use Change

One of the reasons that LCA overestimates emissions from domestic corn production, relative to the GE analysis, is because a significant portion of the corn used to produce ethanol is diverted from crop exports. While this reduces the need for increased corn production, the reduced corn exports, combined with the higher prices and reduced exports of soybeans and wheat, will cause and expansion of cropland worldwide ($\boldsymbol{\theta}_{ILU}$ > 0). The general equilibrium analysis estimates that 0.19 and 0.17 hectares of uncultivated land are converted to cropland per 1000 liters of additional ethanol in 2009 and 2015 respectively (Table 5-3).[140] The increase in cropland worldwide is greater than the overall increase in US cropland because international yields are lower than US yields for all crops except wheat.[141]

The emissions from land use change are substantial and result in the lifecycle analysis under predicting emissions by 3566% in 2009 and 3108% in 2015. The land use change emissions are substantially larger than any other source of emissions, and as a result, if increased ethanol consumption induces even minor land use change, there is little chance that mandated corn based ethanol will reduce greenhouse gas emissions globally.

International Crude Oil Consumption

All previous lifecycle studies have not considered how increased ethanol consumption could increase the consumption of crude oil in the rest of the world (θ_{R_W} > 0). As crude oil consumption is a substantial source of emissions worldwide, a small percentage change in consumption could lead to extremely large emissions consequences. Using the general equilibrium framework, we estimate that for each liter of mandated ethanol, non-US consumption of crude oil would increase by by 0.01

[140] As mentioned, more recent lifecycle emissions studies have focused on the impact of biofuel policy on international land uses. The most prominent study, Searchinger et al. (2008), estimates for 2016 an increase in worldwide cropland of 0.19 hectares per 1000 additional liters of ethanol consumed.
[141] Both FAS (2009) and FAPRI (2009) show international corn and soybean yields to be consistently lower than US yields. In our central case, international wheat yields are lower than US yields in the benchmark, but grow at a faster rate, such that by 2015 US and international yields are both 3.2 mt/ha.

barrels in 2009 and 0.02 barrels in 2015 (Table 5-3). Incorporating the impact of ethanol use on world crude oil consumption into LCA leads to predicted emissions savings that are between 285% and 448% less than the savings estimated by standard LCA (Table 5-4).

The relationship between the change in gasoline consumption (θ_{Fg}) and the change in world crude oil consumption (θ_{R_W}) is troublesome for climate policy. As the two effects act in opposite directions, a desirable domestic policy (θ_{Fg} larger than 1) would lead to larger international emissions, as the domestic demand for crude oil would be reduced farther and the world price of oil more severely depressed. This relationship is evident in Table 5-3. Moving from 2009 to 2015, the RFS more effectively reduces the consumption of gasoline in the US, such that by 2015 more than one liter of gasoline is displaced for every additional liter of mandated ethanol. As a result, the increase in world crude oil consumption due to the RFS becomes larger, increasing from 0.01 additional barrels of crude per liter ethanol in 2009 to 0.02 additional barrels of crude per liter ethanol in 2015.

The emissions consequences of this effect are severe as the carbon emissions potential of the world crude oil market is substantially larger than the emissions potential of the US passenger vehicle transportation sector. While the emissions savings per mandated liter of ethanol in the US domestic fuel markets increased between 2009 and 2015 as a result of the fuel price effects, these savings were very small compared to the exacerbated emissions in world crude markets (Table 5-4).

Extension to Cellulosic Ethanol

While the emissions consequences of corn ethanol are highly debated, most conclude that cellulosic ethanol, produced on marginal lands or with biomass waste, will provide substantially greater emissions savings (Farrell et al. (2006); Hill et al.

(2006); Searchinger et al. (2008); Fargione et al. (2008)). However, these studies do not consider the impact increased biofuel use on the domestic transportation market or international crude market. Even for an unrealistic best-case cellulosic ethanol scenario, the emissions from adjustments in world crude consumption are likely to dominate any emissions savings in most years.

Suppose that a biofuel feedstock can be grown totally on marginal land, such that it will have no impact on domestic or international land markets ($\boldsymbol{\theta}_{k \neq 1}$ and $\boldsymbol{\theta}_{ILU}$ are 0), and that a biofuel can be produced from this feedstock for roughly the same costs as a unit of corn ethanol. Finally, suppose that the RFS mandate from 2008 to 2015 required the consumption of this cellulosic biofuel as opposed to corn ethanol. As with corn ethanol, the consumption of this biofuel would save 2.36 kgCO_2e/liter if it displaced gasoline on a volumetric basis (Table 5-5). However, due to adjustments in domestic fuel price, between 0.15 and 0.01 kgCO_2e/liter of this emission savings is offset by increased fuel consumption between 2009 and 2013, and there are additional emissions savings of 0.01 and 0.03 kgCO_2e in 2014 and 2015. Likewise, the reduced demand for US gasoline would lower the world price of crude oil and result in emissions of 3.96 kgCO_2e/l in 2009 to 6.08 kgCO_2e/l in 2015. In order for this cellulosic feedstock to provide emissions savings, the growing of feedstock and production of fuel would have to sequester or offset a total of 1.75 kgCO_2e/l in 2009 and 3.69 kgCO_2e/l in 2015.

As cellulosic ethanol technologies are still emerging, the emissions savings estimates vary significantly. Farrell et al. (2006) report lifecycle emissions savings from cellulosic ethanol produced using switchgrass, while coproducing electricity, of 2.8 kgCO_2e/l relative to gasoline. Adler et al. (2007) find lifecycle emissions savings of 1.9 kgCO_2e/l and 1.36 kgCO_2e/l for ethanol produced from switchgrass and reed canarygrass respectively, and 1.9 kgCO_2e/l for hybrid poplar. Sheehan et al. (2003)

find that ethanol produced using corn stover generates lifecycle emissions savings of 1.85 $kgCO_2e/l$. While these emissions savings are substantial from a lifecycle perspective, each estimate fails to outweigh the impacts of increased biofuel use on fuel markets in any year beyond 2013 (Table 5-5).

An optimistic study (Tilman, Hill, and Lehman 2006) found that when soil carbon sequestration is factored into the lifecycle emissions of producing ethanol from low-input high diversity grasses, emissions savings are 6.56 $kgCO_2e/l$ in the first 10 years of production and fall to 5.14 $kgCO_2e$/liter in subsequent years. These projected savings suggest that certain types of cellulosic ethanol could in fact outweigh the emissions consequences of the increased biofuel consumption on the world price of oil.

Table 5-5. Emissions Savings Required for Cellulosic Ethanol ($kgCO_2e$/liter)

	2008	2009	2010	2011	2012	2013	2014	2015
Combustion Savings	0.00	-2.36	-2.36	-2.36	-2.36	-2.36	-2.36	-2.36
Domestic Fuel Market	0.00	0.15	0.11	0.10	0.04	0.01	-0.01	-0.03
International Crude Oil	0.00	3.96	4.29	4.59	4.92	5.28	5.67	6.08
Production and Agricultural Savings Required	0.00	1.75	2.04	2.33	2.60	2.93	3.30	3.69

Conclusions

The change in emissions from the consumption of an additional unit of ethanol calculated using a general equilibrium framework are considerably higher than the change in emissions calculated using lifecycle analysis. The difference in the emissions savings calculated by the two methods is driven completely by how behavioral adjustments are modeled, as the emissions factors used in the two methods are identical (Table 5-1). In LCA, behavioral adjustments are based on the quantity of each input used in the production process. This leads to a very restrictive set of implied behavioral assumptions that do not adjust with changes in policy or other underlying economic conditions. Compared to the general equilibrium analysis, LCA assumes much larger increases in corn production and larger reductions in gasoline consumption. Until recently, the lifecycle methods have assumed that increased ethanol use has no impact on non-corn land uses, while no lifecycle analysis has assessed the impacts on the international energy markets.

Domestically, the estimated change in emissions of the LCA and general equilibrium methods are similar, with the general equilibrium method predicting 9% lower emissions savings in 2009 and 2% higher emissions savings in 2015. However, the reason for this is that the general equilibrium method incorporates emissions domestic changes in land use, notably the conversion of CRP to cropland, that are assumed to be zero in LCA. These emissions offset other emissions categories that are either over predicted (corn production) or not included (non-corn cropland) in LCA.

The bigger deviation between the two methods stems from LCA not including the impacts of increased ethanol consumption on international markets. When the emissions from land use change and crude oil consumption are included in the analysis, the domestic emissions savings predicted by LCA of roughly 1.32-1.36 kgCO_2e/liter become emissions increases more than 3500% larger. We also find that the increased emissions from world crude oil consumption are large enough to offset the lifecycle emissions savings estimates of many forms of cellulosic ethanol.

APPENDIX C

Table 5-6. LCA Emissions Savings, 2008 to 2015 (kgCO_2e/liter)

	2008	2009	2010	2011	2012	2013	2014	2015
Combustion	1.62	1.62	1.62	1.62	1.62	1.62	1.62	1.62
Production	1.13	1.13	1.12	1.12	1.12	1.12	1.12	1.12
Co-Product Credit	-0.20	-0.19	-0.19	-0.18	-0.18	-0.17	-0.17	-0.17
Corn Production	0.75	0.73	0.72	0.71	0.70	0.69	0.68	0.67
Combustion Credit	-1.51	-1.51	-1.51	-1.51	-1.51	-1.51	-1.51	-1.51
Ethanol Total	1.79	1.78	1.77	1.76	1.75	1.75	1.74	1.74
Gasoline Total	3.10	3.10	3.10	3.10	3.10	3.10	3.10	3.10
Ethanol Savings vs. Gasoline	1.31	1.32	1.33	1.34	1.35	1.35	1.36	1.36

Table 5-7. Impact of RFS on Emissions by Sector (kgCO_2e/liter)

	2008	2009	2010	2011	2012	2013	2014	2015
Transportation	0.00	-2.21	-2.25	-2.27	-2.32	-2.35	-2.37	-2.39
Substitution	0.00	-2.36	-2.36	-2.36	-2.36	-2.36	-2.36	-2.36
Fuel Price Effects	0.00	0.15	0.11	0.10	0.04	0.01	-0.01	-0.03
Fuel Production	0.00	0.53	0.52	0.52	0.50	0.50	0.49	0.49
Substitution	0.00	0.50	0.49	0.49	0.49	0.49	0.49	0.49
Fuel Price Effects	0.00	0.04	0.03	0.02	0.01	0.00	0.00	-0.01
Agriculture	0.00	0.47	0.49	0.49	0.50	0.50	0.51	0.51
Corn	0.00	0.22	0.21	0.20	0.20	0.20	0.19	0.19
Other Crops	0.00	-0.07	-0.07	-0.06	-0.06	-0.06	-0.06	-0.06
CRP	0.00	0.32	0.34	0.35	0.36	0.37	0.37	0.37
ROW Crude	0.00	3.96	4.29	4.59	4.92	5.28	5.67	6.08
International Land Use	0.00	47.24	46.67	45.90	44.98	44.14	43.37	42.42

Chapter 6. Conclusions

This thesis has examined the effect of the increased Renewable Fuel Standard (RFS) on greenhouse gas emissions using a model that simultaneously predicts how the US transportation and agricultural sectors will respond to the policy and impact domestic and world markets. Through this framework we are able to estimate both the potential emissions savings of the RFS mandate, and many sources of carbon leakage, that is, the unintended emissions consequences of the policy.

Model Structure and Features

In the transportation sector, we model the RFS mandate's impact on household demand for VMT and fuel economy and therefore blended fuel. Through these adjustments we are able to estimate the changes in emissions that result the RFS's impact on the price of blended fuel, either due to changes in demand for VMT or changes in expenditure on fuel efficiency technologies.

In the agricultural sector, we allow for multiple levels of adjustment that capture both the extensive and intensive responses to the mandate, and estimate the emissions consequences of each response. On the extensive margin we model the landowner's decision to allocate land to cropland or the Conservation Reserve Program and estimate the aboveground, root biomass, and soil carbon lost when CRP is converted to cropland. On the intensive margin, the landowner allocates land between six rotations (which incorporate four crops) as well as four tillage practices. In disaggregating agricultural production to this level, we are able to consider the heterogeneity in energy and fertilizer use, and therefore greenhouse gas emissions, of the various cropping and management practices. This disaggregation also allows us to understand what effects changes in management practice may have on the rate of soil organic carbon sequestration. As the market prices and domestic supplies of crops adjust, the impact on crop exports are related agricultural expansion and land use change emissions in the rest of the world.

To link the transportation and agricultural sectors we model the fuel blender's decision, as well as the production of gasoline from crude oil and the production of ethanol from corn. The fuel blender chooses the minimum cost shares of gasoline and ethanol to include in a unit of blended fuel, based on the prices of the fuel and the RFS mandated share of ethanol. This decision establishes a linkage between price of crude oil and gasoline, the price of corn and ethanol, and the level of the RFS mandate. The linkage between the price of crude oil and the price of ethanol allows us to understand the quantity of ethanol that will be added to the fuel supply as a result of the RFS. Therefore the potential emissions savings in the transportation sector and the increased emissions from the agricultural sector can be estimated. Finally, this linkage also allows us to understand the impact the RFS has on the international price and non-US consumption of crude oil.

Principle Results

Our simulations show that the consumption of ethanol increases over time in absence of the Renewable Fuel Standard, and as a result, the mandate has no effect on ethanol consumption before 2009 and only increases ethanol consumption by 11.1 billion liters in 2015. As little ethanol is added to the fuel supply, little gasoline is displaced and the potential emissions savings are small relative to total US greenhouse emissions (less than 0.5%). We also find that the potential emissions savings are more than offset by the carbon leakage in various domestic and international markets.

Domestically, leakage occurs in both in the transportation, fuel production and agricultural sectors. In the transportation sector leakage occur only when the price of blended fuel falls in response to the mandate, which occurs from 2009 to 2013. This leads to an increase in the consumption of blended fuel which offsets roughly 6% of the potential emissions savings from expanded ethanol consumption in 2009 and 1% of potential emissions savings in 2013. In 2015, the price of blended fuel increases in response to the mandate leading to a 1% increase in emissions savings. The leakage in the fuel production sector occurs because the production of ethanol is emissions intensive relative to gasoline refining. This leakage is much more substantial than the leakage in the transportation sector, as it offsets 23% of the potential emissions savings of the RFS in 2009 and 21% of potential emissions savings in 2015.

The leakage in the domestic agricultural sector occurs as the production of the most emissions intensive crop (corn) expands, land is converted from CRP to cropland, and rotation and management practices intensify. We find this to be a substantial source of leakage, offsetting 20% and 22% of potential emissions savings in 2009 and 2015 respectively. The conversion of land in CRP to cropland is the major component of this leakage because the per hectare emissions from carbon released by CRP soils and biomass upon conversion are far greater than the N_2O and energy use emissions of agricultural production. We find that in response to the RFS, the conversion of CRP to cropland contributes the most (64%) to the agricultural leakage. Of the other components, increased energy and fertilizer use is the largest, accounting for between 13% and 20% of the leakage, depending on the emissions factors used. The majority (roughly 90%) of the increased energy and fertilizer use emissions are the result of shifts from other crops to corn production, while the emissions from the intensification of rotations and management practices are minor.

We also find leakage in the international land and crude oil markets. These leakages are substantial, as the magnitude of each is greater than both the total domestic leakage and the potential emissions savings. The leakage in the international land market occurs because US crop exports fall in response to the mandate, raising international prices for crops. As a result, farmers worldwide are induced to expand production on to previously uncultivated land. Due to the large quantities of biomass and soil carbon stored in native ecosystems, a small increase in cropland worldwide will lead to a carbon leakage that is more than 14 times larger than the potential savings even when the most optimistic yield growth scenario is used.

Finally, we find that the international price of crude oil is depressed by the RFS because US demand for crude oil falls. This results in an increase in the world consumption of crude oil and a carbon leakage more than 1.5 times as large as the potential emissions savings of the RFS in our central case and between 56% and 86% of potential savings in the low crude oil demand elasticity case.

Our results suggest that even if the Renewable Fuel Standard mandate is coupled with other domestic policies that serve to limit leakage, the potential of the RFS to reduce greenhouse gas emissions is low. Domestically, the majority of the carbon leakage occurs through the increased production of ethanol and the expansion of agriculture on to land held in the Conservation Reserve Program. The leakage in the fuel production sector could be reduced through improvements in the efficiency of ethanol production, or if biomass fuels were used to fire the ethanol plants. However, reducing the emissions from ethanol production to be comparable to the emissions from gasoline production would only serve to eliminate half of the domestic leakage. The conversion of CRP land to cropland could be limited by increasing CRP rental payments to levels that would eliminate the incentive for farmers to convert CRP to cropland. However, this type of policy would only serve to reduce US crop exports further and exacerbate leakage through the international land markets.

While any policy that reduces US demand for crude oil will be subject to leakage in the crude oil market, policies that increase the production of biofuel feedstock at the expense of other crops will also result in leakage in domestic and international land markets. Therefore, if the policy goal is to reduce greenhouse gas emissions globally, a more direct policy that reduces gasoline consumption without impacting the agricultural sector, such as a gasoline tax or increased fuel economy standards, would be more desirable. More generally, for effective climate policy, caution must be taken so as not to inadvertently disrupt markets that are large sources of emissions (land markets and crude oil market).

The main tool used to estimate the emissions savings of ethanol consumption, lifecycle analysis, does not consider behavioral responses to increased ethanol consumption and therefore does not capture important sources of carbon leakage. We find that compared to our general equilibrium analysis, standard LCA methods estimate similar reductions in transportation emissions and similar increases in fuel production emissions. In the domestic agricultural sector, the increased emissions estimated by LCA are similar to the emissions increases estimated in the general equilibrium framework, but the sources of emissions are different. LCA assumes that the increased emissions come as a result of increased corn production. Compared to the general equilibrium analysis, we find that LCA overestimates the increase in corn production which results in an underestimate of total emissions savings of close to 30%. However, by not estimating the expansion of cropland on to CRP, LCA overestimates emissions savings, relative to the general equilibrium analysis, by a similar percent.

Finally, we find that the assumption in most LCA analyses, that the use of ethanol has no impact on international crude oil or land markets, will causes LCA to substantially overestimate emissions savings relative to the general equilibrium estimates by more than 285% and 3108% respectively. While other studies, such as Searchinger et al. (2008), attempt to include land use change into LCA, these studies do not model domestic agricultural intensification (rotation and tillage practice adjustments) or behavioral adjustments in the transportation fuel or crude oil markets. It is the ability of our model to capture the behavioral adjustments in these key sectors that drive the substantial deviations with LCA.

Model Limitations

Our model has limitations that warrant mention. First, we acknowledge our simplified treatment of international land use effects and inability to capture agricultural intensification worldwide. As shown in our domestic agricultural sector, modeling the land-owner's decision to expand or intensify production will likely reduce the international carbon leakage as the increased use of fertilizer and pesticides have smaller emissions consequences than the conversion of most native ecosystems to agriculture.

Second, our modeling of the crude oil markets is overly simplified. These markets are extremely complicated and involve the interactions of countries, multi-national corporations and cartels. A more accurate assessment would require the modeling of the major players in these markets.

Third, our model does not track parcels of land over time, so our ability to capture agricultural emissions that are based on historic management practice and soil quality, such as cropland N_2O emissions, which are dependent on residual nitrogen in soils from previous crops and fertilizer applications, and soil carbon sequestration, which is dependent on all historic management practices.

Future Work

There are a number of areas in which the current model could be improved in future work. First, we plan to expand the land-use categories we consider. In terms of non-agricultural land uses, incorporating range, pasture, and forestry into the land owner's decision will provide more realistic adjustments and more accurate emissions consequences in response to the expanded RFS. It would be expected that at higher crop prices, agricultural production may also expand at the expense of land held in forest or pasture, with sizeable emissions consequences, particularly forest is converted. Within agricultural production, additional crops, particularly cotton and sorghum and rotations could be incorporated in addition to other management practices, such as irrigation and manure application.

Along with expanding the number of land uses consider, the agricultural sector could be improved by disaggregating to a finer spatial resolution. This will allow the model to account for the heterogeneous soil characteristics and climate patterns of US cropland and the regional allocation of cropland to particular cropping and management practice. In particular, yields and input use are will vary depending on soil characteristics and climate. This variability suggests that the regional responses to the expanded RFS mandate will be quite different. Likewise, climate patterns will cause the allocation of crops and rotations to be spatially explicit. By incorporating this heterogeneity, we will be able to capture a more realistic adjustment between crops and rotations and be able to use more detailed methods for the greenhouse gas emissions calculations.

Finally, a livestock production sector could be incorporated as an intermediate sector that consumes crops and provides livestock to the food sector. As approximately 40% of all corn produced in the US is used as feed (A. Baker and Lutman 2008), it is likely that this sector will contract in response to higher corn prices caused by the mandate. Incorporating the potential contraction of this sector into our greenhouse gas emissions could yield significant emissions savings as enteric fermentation from livestock is the largest domestic source of anthropogenic methane emissions (US EPA 2009a) and manure management is a large source of emission[142] Including this sector will also allow us to better treat the demand for ethanol co-products, and more accurately attribute emissions savings to these products.

[142] Enteric fermentation from animal livestock contributed 24% of total US methane emissions in 2007, or 139 tgCO_2e, while manure management contributed 44 TgCO_2e of methane (US EPA 2009a).

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