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Energy and greenhouse gas emission effects of corn and cellulosic ethanol with technology improvements and land use changes

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ABSTRACT

Use of ethanol as a transportation fuel in the United States has grown from 76 dam^3 in 1980 to over 40.1 hm^3 in 2009 — and virtually all of it has been produced from corn. It has been debated whether using corn ethanol results in any energy and greenhouse gas benefits. This issue has been especially critical in the past several years, when indirect effects, such as indirect land use changes, associated with U.S. corn ethanol production are considered in evaluation. In the past three years, modeling of direct and indirect land use changes related to the production of corn ethanol has advanced significantly. Meanwhile, technology improvements in key stages of the ethanol life cycle (such as corn farming and ethanol production) have been made. With updated simulation results of direct and indirect land use changes and observed technology improvements in the past several years, we conducted a life-cycle analysis of ethanol and show that at present and in the near future, using corn ethanol reduces greenhouse gas emission by more than 20%, relative to those of petroleum gasoline. On the other hand, second-generation ethanol could achieve much higher reductions in greenhouse gas emissions. In a broader sense, sound evaluation of U.S. biofuel policies should account for both unanticipated consequences and technology potentials. We maintain that the usefulness of such evaluations is to provide insight into how to prevent unanticipated consequences and how to promote efficient technologies with policy intervention.

1. Introduction

Since the beginning of the U.S. fuel ethanol program in 1980, production of corn ethanol in the United States has grown from 76 dam^3 in 2000 to 40.1 hm^3 in 2009 [1]. The U.S. Congress in the 2007 Energy Independence and Security Act (EISA) established a corn ethanol production target of 56.8 hm^3 a year by 2015 plus 79.5 hm^3 of advanced biofuel production by 2022 [2]. The production capacity of corn ethanol in the United States has already exceeded 49.2 hm^3 a year, and the construction of new facilities is expected to result in additional production capacity of 5.3 hm^3 of corn ethanol [1]. Meanwhile, investments in research, development, deployment, and commercialization of advanced biofuel technologies have been accelerated in the past several years [3,4]. Parallel to these efforts, regulatory efforts such as the

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California low-carbon fuel standards (LCFS) have been launched [5–7].

The original goals of the U.S. ethanol program were to reduce the nation's dependence on imported oil and to reduce air pollutant emissions from motor vehicles. In 2009, the use of 40.1 hm^3 of ethanol accounted for 5.4% (on an energy basis) of the U.S. gasoline market of 500 hm^3 a year [8]. Although still a small share, ethanol is the single largest fuel alternative to gasoline in the United States [8]. The goals now include the reduction of greenhouse gas (GHG) emissions.

Between the late 1970s and now, it has been debated whether corn ethanol results in a negative or positive energy balance (which is defined as the energy in ethanol minus fossil energy spent during the life cycle of producing it) [9–14]. The GHG effects of corn ethanol and cellulosic ethanol have been examined since the early 1990s [15–21]. Since 2008, studies have been conducted to quantitatively address GHG effects of potential indirect (as well as direct) land use changes (LUCs) that are induced by large-scale corn ethanol production [22–25]. Direct and indirect LUCs have been modeled and included in the EPA's development of EISA's renewable fuel standards (RFS) [26] and CARB's LCFS development [7].

Energy balance and GHG results of using corn ethanol are influenced by two major factors – continuous technology improvements in key stages of its life cycle and expansion of the system boundary of life-cycle analysis (LCA) to address unanticipated consequences of large-scale corn ethanol production. For cellulosic biofuels, no commercial-scale cellulosic biofuel facilities are in place yet. The energy balance and GHG results of using cellulosic biomass for biofuel production are affected by the latest understanding of future plant designs, process engineering modeling, and performance of pilot and demonstration projects. In this paper, we evaluate technology improvements and our recent modeling results of direct and indirect LUCs for corn ethanol. We apply the results of our evaluation of these issues to the LCA of corn and cellulosic ethanol and contrast our results with those published in the past three years.

2. Corn ethanol and cellulosic ethanol production technologies and life-cycle analysis of ethanol and gasoline

Corn ethanol technologies are mature, although they continue to advance. Corn ethanol plants could be dry or wet milling. In a dry milling plant, milled corn is sent into a fermentor, in which starch is fermented into ethanol and the remaining materials become distillers' grains with solubles (DGS) as a commercial animal feed. DGS could be sold as dry or wet (DDGS and WDGS). In a wet milling plant, corn oil and corn gluten feed are separated from milled corn first. The remaining materials are sent to a fermentor, in which starch is fermented into ethanol, and the rest of the materials become corn gluten meal. Both corn gluten feed and meal are animal feed. Recent development of dry milling plants incorporates technologies (such as corn fractionation) to extract corn oil before fermentation. Before the middle 1990s, wet milling plants accounted for the majority of U.S. corn ethanol plants. Since then, the growth of the U.S. corn ethanol production capacity has been with dry milling plants. As a result, dry milling ethanol plants now account for nearly 90% of the total U.S. capacity [26].

In cellulosic ethanol plants, biomass materials are pretreated with chemicals. Cellulose and hemi-cellulose are then fermented into ethanol. The remaining lignin material contains energy, which can be burned on-site to generate the steam and power needed for plant operation. In fact, mass balance simulations of cellulosic ethanol plants indicate that the amount of lignin available can generate an amount of electricity exceeding that needed for plant operation. Thus, cellulosic ethanol plants can be a net electricity exporter, similar to Brazilian sugarcane ethanol plants. The types of cellulosic biomass under consideration in the United States include crop residues, forest residues, and dedicated energy crops (such as switchgrass, miscanthus, and mixed prairie grasses).

Historically, LCA of ethanol began with the manufacture of agriculture chemicals (i.e., fertilizers, pesticides, and herbicides), farming of biomass, and production of ethanol, plus transportation activities of moving materials from one location to another. Since ethanol plants produce multiple products including ethanol, addressing the impact of these co-products in biofuel LCAs is critical [29]. The system boundary of biofuel LCAs was expanded to include the manufacturing of farming equipment [10,30], but the contribution of farm equipment manufacturing to the results of ethanol LCA proved to be small (about 1% for GHG emissions and 2% for fossil energy use; see [30]). Recently, the biofuel LCA system boundary has been expanded again to include indirect LUCs (while direct LUCs have been included in some LCA models), which could be significant, but with great uncertainties [7,23,25,26,31].

Argonne National Laboratory developed the GREET (Greenhouse gases, Regulated Emissions, and Energy use in Transportation) model for LCAs of transportation fuels, including ethanol and other transportation fuels, and advanced vehicle technologies [32] (see http://greet.es.anl.gov/main). The current GREET version (GREET1.8d) includes more than 100 production pathways for transportation fuels, many of which are biofuel pathways. It also includes major vehicle propulsion technologies, such as internal combustion engines, hybrid electric vehicles, battery-powered electric vehicles, and fuel cell vehicles. Since the late 1990s, we have applied GREET to examine ethanol's energy and emissions effects [17,30,33,34]. In addition, CARB applied GREET for its LCFS development [7], the U.S. EPA used GREET partially for its RFS development [26], and Farrell et al. [20] and Searchinger et al. [23] used GREET to examine ethanol's GHG effects.

Ethanol LCA results are usually compared to those of petroleum gasoline in order to assess energy and GHG effects associated with the displacement of gasoline with ethanol. The LCA of petroleum gasoline includes petroleum recovery and petroleum refining plus transportation activities for petroleum and gasoline. In GREET, the life cycle of petroleum gasoline that is produced for the U.S. market includes recovery of conventional and unconventional crude, transportation of crude, crude refining to gasoline (among many other petroleum products), gasoline transportation and distribution to refueling stations, and gasoline combustion in vehicles. Unconventional crude considered in GREET includes Canadian oil sands. In 2009, U.S. petroleum refineries imported 0.13 hm^3 of oil sands products from Canada, of the 2.30 hm^3 of crude oil input to U.S. refineries [27,28]. In GREET simulations for this study, we assumed that Canadian oil sands products accounted for 5.7% of U.S. crude oil supply.

In the past two years, we addressed four critical issues for ethanol LCA simulations with GREET. First, we upgraded Purdue University's Global Trade Analysis Project (GTAP) model to address simulations of direct and indirect LUCs. Second, we monitored and collected data of ethanol plant operation to examine historical trends and the current situation of ethanol plants' yield and energy use. Third, we analyzed displacement effects of DGS in the animal feed market to address energy and emission credits of DGS in corn ethanol LCA [29,35]. Fourth, we collected and analyzed U.S. farming data on both chemicals and energy use to develop historical trends of U.S. farming chemical and energy use intensity [33]. With GREET1.8d and updated data in these areas, we produced new LCA results for corn ethanol and cellulosic ethanol, as documented in this paper.

3. Key parameters of ethanol life-cycle analysis

3.1. *Land use changes from corn ethanol production with an upgraded GTAP model*

Use of corn for ethanol production generates additional demand for corn. This additional demand induces both direct LUCs (dLUCs) and indirect LUCs (iLUCs). By definition, dLUCs are those for growing corn that is used directly for ethanol production, while iLUCs are those for growing corn for uses other than ethanol production and for growing other crops that experience production shortfalls caused by corn ethanol production. However, in practice, the use of these terms has not been consistent among authors. Since corn is a fungible commodity that is traded in markets around the world and is used for food, feed and industrial purposes, it is difficult to estimate and observe dLUCs for ethanol production. On the other hand, iLUCs are caused by changes in market prices from one crop to another and from one region to another at the global scale, they are simulated (together with dLUCs) but are impossible to observe.

In the past three years, economic models were adapted to simulate potential global LUCs (direct and indirect) as a result of U.S. corn ethanol production. In particular, the Food and Agricultural Policy Research Institute (FAPRI) model at Iowa State University was used by Searchinger et al. [23] and by the U.S. EPA [36] (in conjunction with the FASOM model at the Texas A&M University) for its RFS development, and the GTAP model [37] was used by the CARB for its LCFS development [7,25]. Based on economics, these models predict direct and indirect LUCs together but separate the aggregate LUCs into different global regions. In particular, the models generate LUC results for the U.S. (domestic LUCs) and for other countries (foreign LUCs) from U.S. corn ethanol production. While all foreign LUCs are iLUCs, domestic LUCs are both dLUCs and iLUCs. With some supplement techniques (such as satellite data at a finer level), domestic LUCs could be separated into dLUCs and iLUCs. But no systematic attempt has been made for the separation so far. Because of these difficulties, we were unable to separate aggregate LUCs into dLUCs and iLUCs. In the sections below, we present aggregate induced LUC results.

The early versions of these models did not adequately address a few critical issues, such as crop yield increase in response to increased commodity price, future trends of both supply and demand of grains, closer examination of available land types and amount in key countries, detailed simulation of substitution between DGS and conventional animal feeds inside economic models, energy sector demand and supply elasticities in the modern era, and productivity of new lands brought into biofuels production, among other issues. Since January 2008, Purdue University, with the support of the U.S. Department of Energy and Argonne National Laboratory, has made significant modifications to the GTAP model to remedy these problems [31]. Compared with previous studies, an upgraded GTAP model from this effort shows a lower amount of LUCs for the United States to reach 56.8 hm^3 of corn ethanol production in 2015. Fig. 1 shows LUC results from several recent studies in comparison with those from new GTAP results. Estimates by EPA, CARB, and Hertel et al. [7,25,36] reduce LUCs by 60% from that of Searchinger et al.. Our estimates reduce LUCs by an additional 60%. Furthermore, our GTAP modeling results showed that 25% of the aggregate LUCs from U.S. corn ethanol production are domestic LUCs and 75% are foreign LUCs. This means that at least 75% of the aggregate LUCs would be iLUCs.

The translation of LUCs into GHG emissions requires an understanding of the types of land changes from one practice to another, above- and below-ground biomass and carbon content of given land types, lifetime of a biofuel program

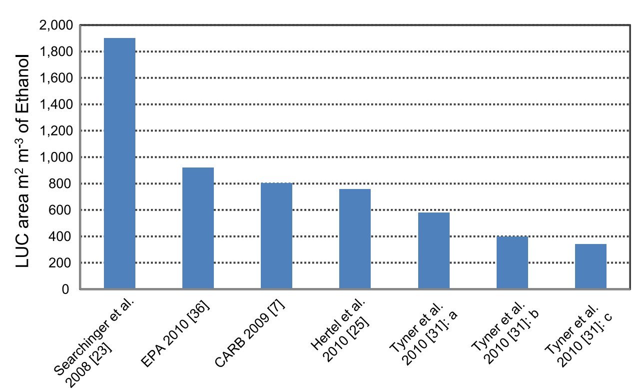

Fig. 1 – Land use change from U.S. corn ethanol production in several studies (m^2 per m^3 of ethanol produced). ***Notes:*** **Searchinger et al. used the FAPRI model [23,36]. EPA used FAPRI and Texas A&M University's FASOM model. CARB and Hertel et al. used an earlier version of GTAP [7,25]. Tyner et al. used an upgraded GTAP version [31]. Of the three cases in Tyner et al., Case a is with the 2001 global baseline; Case b is with a partially updated 2006 global baseline; and Case c is with the updated 2006 baseline and baseline projection of supply and demand of grains through 2015, the target year for 57 hm^3 of corn ethanol in the United States.**

under evaluation, and treatment of emissions occurring at different times [38], all of which are subject to data limitation and uncertainties. We developed a GHG calculator with GTAP LUC results as inputs. The calculator can vary these parameters to estimate GHG emission results from LUCs. The calculator, together with its user manual, is available at Argonne's GREET website (http://greet.es.anl.gov/main). Fig. 2 presents LUC-induced GHG emissions for U.S. corn ethanol production estimated in a few recent studies in comparison with values used in this paper. All estimates in Fig. 2 are with a 30-year lifetime of corn ethanol programs. Estimates by EPA, CARB, and Hertel et al. reduce LUC GHG emissions by 70% from that of Searchinger et al. Our estimates reduce LUC emissions by an additional 50%.

The above-mentioned factors affect the differences among the cited studies. Although advances have been made in the past three years to address LUC modeling in economic models, LUC simulations continue to be subject to great uncertainties. Four of the remaining issues that need further research are as follows. First, more sensitivity tests on prospective growth in crop demand and supply are needed by region and agricultural ecological zone (AEZ) (see [39] for AEZ definition and assessment). The future growth in the demand and supply of agricultural commodities — particularly coarse grains — is a critical determinant of the impacts of biofuel programs. If global income and population growth, and dietary transition lead to greater growth in demand for coarse grains than in supply, the impacts of biofuel mandates would be greater. On the other hand, if new technologies and broader adoption of these technologies lead to greater growth in supply, the impacts of biofuel mandates would be reduced. Second, improved data and information on land use and land cover change could be helpful to improve model parameters and structure. This is particularly important for other regions of the world because less is known about land use and land conversion. Third, as we add cellulosic feedstocks to GTAP for land use analysis, we will need to effectively capture the interactions among the different feedstocks, and between these feedstocks and standard commodity markets. Fourth, the modeling and analysis will need to be dynamic so that we can better capture the dynamics of cellulosic and other second-generation feedstocks.

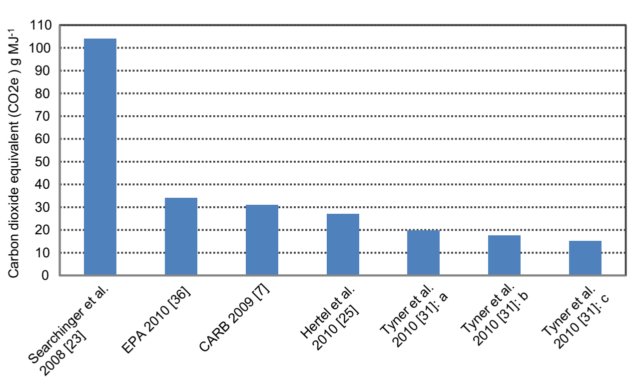

Fig. 2 – Greenhouse gas emissions of land use changes of corn ethanol (g CO_2e per MJ).

3.2. Energy use of corn ethanol production plants

The most significant change in the corn ethanol industry is probably the reduction in energy use in ethanol plants. Fig. 3 shows the energy use by ethanol plants in GJ per m^3 of ethanol presented in studies from the late 1970s to now. Of the 35 studies summarized here, energy use was estimated to be above 39 GJ m^{-3} before the start of the U.S. ethanol program in 1980. After that, and with data from ethanol plant operation, energy use was shown to be reduced to around 19.5 GJ m^{-3}. Furthermore, dry milling plants have experienced a reduction from 19.5 to 7.97 GJ m^{-3} between 1980 and now, while wet milling plants have experienced a reduction from 19.1 to 13.2 GJ m^{-3}. The dramatic reduction in dry milling plant energy use is especially important to industry-wide ethanol energy and GHG results, since at present, dry milling ethanol plants account for nearly 90% of the U.S. total ethanol production capacity.

Energy use is the second largest cost component (after corn feed cost) in ethanol plant operation. Plant owners have economic incentives to reduce their energy expenditure. This was especially true in the middle 2000s when the price of natural gas skyrocketed and U.S. dry milling plants used primarily natural gas to generate steam. During that period, engineering firms and other third parties began to market and introduce energy-efficient technologies and processes into ethanol plants [40,41]. To further reduce fossil energy use in dry milling ethanol plants, some plant owners have experimented with the use of biomass-based fuels, such as wood chips and solubles of the DGS from ethanol plants [33,40,41]. The reduction in dry milling plant energy use has continued in the past several years. In 2007, we assessed an energy use of 10 GJ m^{-3} for dry milling plants in operation in 2005 [33]. The industry survey data for 2008 plant operation [14,21,42] showed the energy use was reduced to 7.97 GJ m^{-3} in 2008 for dry milling plants. On the other hand, an industry survey of wet milling plants in operation in 2006 showed energy use of 13.2 GJ m^{-3} [43].

3.3. Energy and GHG emission credits of co-products of corn ethanol

Dry milling ethanol plants yield 681 kg of DDGS per m^3 of ethanol produced [35]. DDGS has been used as a nutritional substitute of conventional livestock feeds, including corn, soybean meal, and urea. As a result of the substitution between DDGS and conventional animal feeds, energy required for the production of the displaced conventional feeds would be saved and associated emissions avoided. With the 56.8 hm^3 of corn ethanol production in 2015 as established in the U.S. 2007 Energy Independence and Security Act [2], 34.9 Mt of DDGS could be produced. The theoretical maximum U.S. market size for DDGS has been estimated to be 46.8 Mt [44,45]. In the U.S. domestic market, beef cattle and dairy cattle will equally have 40% share of the total, swine 13%, poultry 6%, and the remaining 1% by other animal types [35]. On the other hand, the DDGS export accounts for 19.6% of the U.S. DDGS production.

In the United States, both DDGS and WDGS are produced and fed to animals to displace corn, soybean meal, and urea in their diets with different inclusion levels [46–49]. In corn

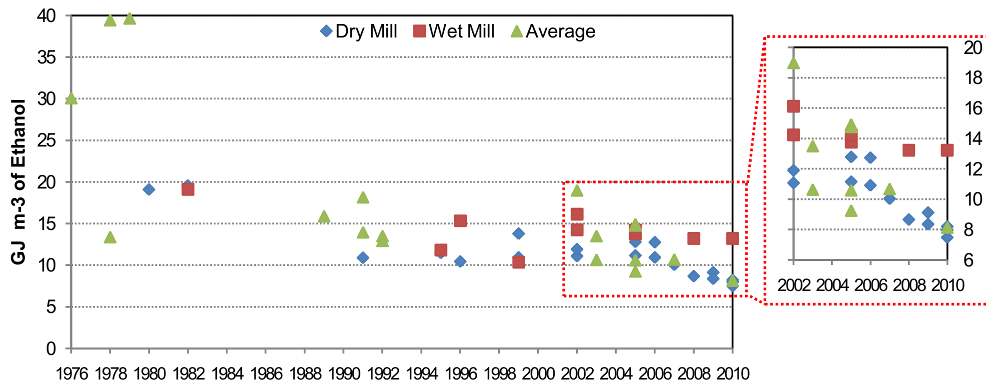

Fig. 3 – Historical trend of corn ethanol plant energy use (GJ per m^3 of ethanol).

ethanol LCAs, methods of dealing with co-product DGS can affect the results significantly [29]. Of the methods available, the systems boundary expansion approach (also called the displacement method in transportation fuels LCAs) is recommended by the International Standard Organization (ISO). We used this method in our study.

For the displacement method, several studies were completed to examine DGS displacement effects in corn ethanol LCAs [21,50,51]. We recently [35] updated our earlier displacement ratios with detailed levels (the feedlot level, the U.S. market level, and the composite U.S. and export market level). Table 1 summarizes displacement ratios we estimated at these three levels of analysis [35]. On average, 1 g of DGS can displace about 0.8 g of corn, 0.3 g of soybean meal, and 0.02 g of urea. That is, 1 g of DGS displaces a total of 1.12 g of corn, soybean meal, and urea combined. The projected 2015 U.S. DGS production of 34.9 Mt could potentially displace 27.9 Mt of corn (about 20% of the corn required for ethanol production in 2015), 10.5 Mt of soybean meal (about 11% of 2009 U.S. production), and 0.7 Mt of urea, if all of the produced DDGS and WDGS is used in the feedlot industry in the United States.

3.4. *Chemicals and energy use intensities of corn farming*

The U.S. Department of Agriculture (USDA) published annual crop yields and fertilizer use until 2005. The USDA historical data from 1960s to 2005 showed that corn yield continued to increase (corn yield per hectare is directly related to LUC effects per unit of ethanol produced, but not directly related to farming emissions per unit of ethanol produced). Meanwhile, nitrogen fertilizer application rates per hectare have been stabilized since the late 1970s, phosphorus fertilizer applications have declined since the late 1970s, and potash fertilizer applications have declined since the mid-1980s [33]. These trends are translated into significant decreases in the intensity of fertilizer use in the amount of fertilizer per unit of corn harvested, which is directly related to corn ethanol LCA results.

Fig. 4 shows historical trends of the relative intensity of U.S. fertilizer use for corn farming. The U.S. corn farms have reduced the intensity of fertilizer use between 1970 and 2005 by 35% for nitrogen fertilizer, 60% for phosphorous fertilizer, and 50% for potash fertilizer.

Table 1 – Displacement ratios between distillers' grains with solubles and conventional animal feeds.

	DDGS and WDGS US market share	DDGS Displacement ratio ($kg\ kg^{-1}$ DDGS)				WDGS Displacement ratio ($kg\ kg^{-1}$ WDGS, dry matter based)			
Feedlot level		Market share (%)	Corn	Soybean meal	Urea	Market share (%)	Corn	Soybean meal	Urea
Beef Cattle	40.6%	35.7%	1.203	0.000	0.068	50.0%	1.276	0.000	0.037
Dairy Cattle	40.6%	35.7%	0.445	0.545	0.000	50.0%	0.445	0.545	0.000
Swine	12.8%	19.5%	0.577	0.419	0.000				
Poultry	6.0%	9.2%	0.552	0.483	0.000				
Average			0.751	0.320	0.024		0.861	0.273	0.019

	DDGS and WDGS Displacement Ratio ($kg\ kg^{-1}$ DDGS & WDGS)		
	Corn	Soybean meal	Urea
U.S. Consumption level[a]	0.778	0.304	0.022
U.S. and Export Market Combined[b]	0.781	0.307	0.023

a In the U.S. market, DDGS and WDGS account for 65.7% and 34.3%, respectively.
b In 2009, export of U.S. DGS accounted for 19.6% of total U.S. DGS production. We assumed that the U.S. DGS export is DDGS. Furthermore, we assumed that the foreign market uses the U.S. DDGS in the same way as the U.S. market uses DDGS.

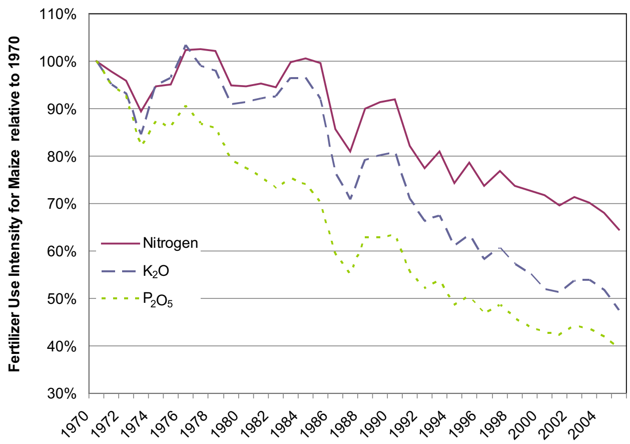

Fig. 4 – Historical trend of relative Intensity of fertilizer use in U.S. corn farming.

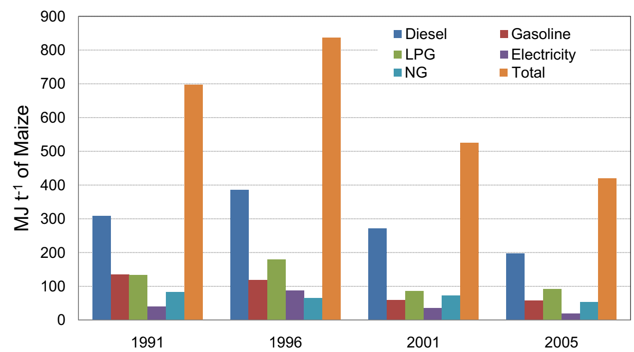

Fig. 5 – Historical trend of energy use in U.S. corn farming. Note: The unusual high farming energy use in 1996 is caused by the wet weather during harvest in that year in the U.S. Midwest.

Lime is also applied to Midwest soils to stabilize soil acidity conditions. The USDA did not have consistent statistics on agricultural lime applications. On the basis of information from Shapouri et al. [14] and our communication with USDA [52], we determined lime use in corn farming, as presented in Table 2. The production of nitrogen fertilizer is energy-intensive and the application of nitrogen fertilizer is GHG-intensive because of the N_2O emissions from nitrification and denitrification of nitrogen fertilizer in cornfields. A reduction in nitrogen fertilizer use for corn farming can thus result in a significant reduction in energy use and GHG emissions of corn ethanol. On the other hand, energy use of lime manufacture (including transportation to cornfields) is low. However, the conversion of limestone to burned lime in cornfields results in CO_2 emissions, which can account for a few percentage points of total corn ethanol GHG emissions and are included in GREET simulations.

Energy use in terms of diesel fuel, natural gas, propane, and electricity in farms is indirectly collected by the USDA in its farm cost surveys for each five years. Shapouri et al. summarized the historical trend of energy use in corn farming [11–14]. Fig. 5 summaries historical trends of energy use in corn farming estimated by Shapouri et al. over the past 15 years. Except for 1996, energy use in corn farming has shown moderate reductions. The increase in 1996 in the energy use in corn farming is due to abnormal weather that year in the Midwest.

3.5. *Feedstocks and key issues of cellulosic ethanol*

Cellulosic biomass supply for ethanol production could come from crop residues (such as corn stover), forest residues, and dedicated energy crops (such as switchgrass, miscanthus, and mixed prairie grasses). In our analysis, we include cellulosic ethanol from corn stover, forest residues, and switchgrass. Corn stover is now primarily left in cornfields after harvest for soil protection and nutrient supplement in the next season. Studies have evaluated sustainable collection rates of corn stover [53]. In our LCA analysis, we treat stover as a residue by considering energy and emissions only for stover collection and transportation as well as supplement fertilizer applications for making up the nutrient loss from stover removal. If stover-to-ethanol production becomes a major ethanol production source to change the decision on farming and converting land for different crops, both stover and corn grain should be treated as commercial products. For forest residue-based ethanol production, we consider collection and transportation (and stumping) of forest residues. For switchgrass-based ethanol, we consider fertilizer use and energy use for switchgrass growth and energy use for its collection and transportation.

LUCs for these cellulosic biomass feedstocks have not been simulated in as much detail as for corn grain. A simulation of corn stover [54] concluded that the LUC impacts of stover collection could be minimal. This is because the revenue from selling stover for farmers will not be high enough to change land uses for farming. For the forest residue pathway, only forest residue collection or forest thinning is assumed, thus no

Table 2 – Intensity of Fertilizer Use in U.S. Corn Farming and Energy Use and GHG Emissions of Fertilizer Production and Use.

	Nitrogen	Phosphate	Potash	Lime
Fertilizer Use Intensity: g of nutrient per kg of corn	17.2	6.1	7.1	43.5
Energy Use for Fertilizer Production: MJ kg^{-1} of nutrient	48.2	13.8	8.6	7.9
GHG Emissions as CO_2e for Fertilizer Production: g kg^{-1} of nutrient	2996	1014	665	605
GHG Emissions as CO_2e from Fertilizer in Field: g kg^{-1} of nutrient	6,536[a]	0	0	440[b]

a This is CO_{2e} emissions of N_2O from nitrification and denitrification of nitrogen fertilizer in cornfields.
b This is CO_2 emissions of converting calcium carbonate (limestone) to calcium oxide (burnt lime) in cornfields.

LUCs would occur. Growing dedicated energy crops (such as switchgrass) could cause LUCs. Analytic efforts are now under way to expand economic models so that this can be modeled. Direct LUCs from dedicated energy crop farming could indeed result in increased carbon in dedicated energy crop farms, since these crops have deep roots and may help increase below-ground biomass stock [55–59]. In our current LCA, we assume no LUC-induced GHG emissions for cellulosic ethanol.

Cellulosic ethanol plants produce both ethanol and electricity. The net electricity export from the plants depends on their energy balance. The National Renewable Energy Laboratory (NREL) has a long history of techno-economic modeling of cellulosic ethanol production. NREL's recent modeling shows a net export of electricity of 0.61 MWh per m^3 of cellulosic ethanol produced from corn stover and switchgrass and zero MWh from forest residues [60]. The type of conventional electricity to be displaced by cellulosic ethanol plant electricity can significantly affect energy and GHG credits of the exported electricity and can vary by region. If cellulosic ethanol plants are to be built in the U.S. Midwest, cellulosic ethanol-based electricity could displace a large share of coal-based electricity, since the production of electricity in the U.S. Midwest relies heavily on coal. However, the future marginal electricity supply is highly uncertain in the United States because electricity supply and demand balance and regulations on power plant emissions can impact the retirement of existing plants and addition of new plants in the future. We assume that cellulosic ethanol electricity will displace electricity with the U.S. average generation mix.

4. Life-cycle energy and GHG results of ethanol

4.1. *Energy balance of corn ethanol*

Fig. 6 summarizes the fossil energy balance values of corn ethanol from 26 studies completed between 1978 and now. We add to the figure our estimated energy balance values for dry milling ethanol, wet milling ethanol, and average ethanol (dry and wet milling ethanol averaged with their production shares). The figure shows that most studies completed in the 1980s and before concluded negative energy balance values for corn ethanol. However, since the mid-1990s, most studies showed positive energy balances. The trend in corn ethanol's positive energy balance over time is supported by the trends of improvement in corn productivity and ethanol plant energy efficiency as we presented above, which is especially the case for the period between 2002 and 2010. In summary, our own estimates show a positive energy balance of 9.2 GJ m^{-3} of corn ethanol for dry milling plants, 5.3 GJ m^{-3} for wet milling plants, and 8.8 GJ m^{-3} for weighted average of dry and wet milling plants. For the three cellulosic ethanol types, there is a positive energy value of 23.1 GJ m^{-3} for corn stover ethanol, 22.4 GJ m^{-3} for switchgrass ethanol, and 15.8 GJ m^{-3} for forest residue ethanol. In the cases of corn stover and switchgrass ethanol, the positive energy balance values are larger than the energy content of ethanol due to the large fossil energy displacement credit from exported electricity associated with these two feedstocks.

A positive energy balance by corn ethanol is possible because only fossil energy used to produce ethanol is taken into account in energy balance calculations. The energy for corn plant growth via photosynthesis is solar energy and is not considered. Two indexes have been used to measure ethanol's energy performance. One is energy balance values, as presented in Fig. 6. The other index is energy ratio, which is defined as the energy output in ethanol divided by fossil energy input. Farrell et al. [20] pointed out that the energy ratio index could be misleading. For example, if zero fossil energy were used (which is conceivable for certain cellulosic ethanol types), the energy ratio of ethanol would be infinite. Energy balance, not energy ratio, should be used to measure ethanol's energy performance.

In contrast, any energy products based on fossil energy conversion would result in a negative energy balance. This is because the energy in fossil-based products is part of the energy in the fossil feedstock. Furthermore, additional fossil energy is spent to transform the primary fossil energy sources to final products. For example, petroleum gasoline has a negative energy balance of 0.23 MJ per MJ of gasoline produced. This value translates into a negative energy balance of 4.9 GJ per ethanol-equivalent m^3. Similarly, average electricity generation in the United States (nearly 50% of electricity is from coal) consumes 2.34 units of fossil energy per unit of electricity energy, resulting in a negative energy balance of 1.34. That is, U.S. electricity generation has a negative energy balance of 28.5 GJ per ethanol-equivalent m^3.

Thermodynamic laws dictate energy losses of energy conversion, resulting in a negative energy balance for fossil energy conversion by default. Thus, energy balance values may have little value in the broad evaluation of different energy

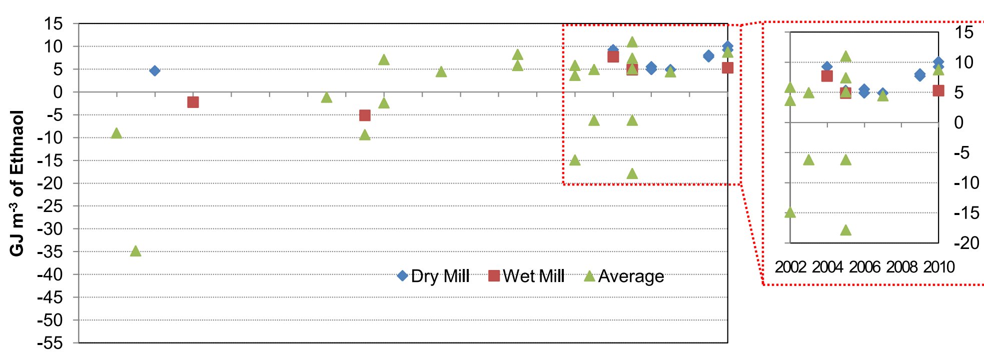

Fig. 6 – Life-Cycle fossil energy balance of corn ethanol (GJ per m^3 ethanol; energy in ethanol minus fossil energy inputs).

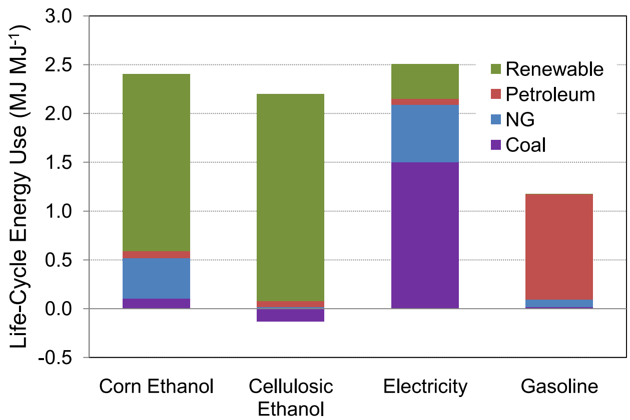

Fig. 7 – Energy use by type for corn ethanol, cellulosic ethanol, U.S. average electricity, and petroleum gasoline (MJ per MJ of fuel produced and used). *Note*: Renewable energy for corn ethanol is the energy in corn kernels, and renewable energy for cellulosic ethanol is that in cellulosic biomass (both of which are from solar energy). Renewable energy for U.S. electricity is that in electricity from hydro power, nuclear power, biomass, solar photovaltaics, and wind power. Renewable energy in gasoline is that embedded in electricity used in the petroleum-to-gasoline cycle.

products. Furthermore, the energy balance calculation combines three fossil energy sources (petroleum, natural gas, and coal), which have very different energy security implications in individual countries. Fig. 7 shows the amount of the three fossil energy sources, as well as renewable energy, to produce and use 1 MJ of energy for corn ethanol, cellulosic ethanol, U.S. average electricity, and petroleum gasoline. If a unit of electricity is used to displace a unit of gasoline (without even considering the difference in efficiency between electric-drive vehicle technologies using electricity and internal combustion technologies using gasoline), the displacement can result in a large reduction in petroleum use, which is an energy security benefit to the United States. Similarly, displacement of gasoline with ethanol results in huge reduction in petroleum use. Energy balance values, as shown in Fig. 6, do not indicate reductions in certain types of fossil energy sources. Also, the three fossil energy sources have very different carbon intensities. Aggregated fossil energy use for a given energy product does not indicate the GHG emissions associated with producing and burning that energy product.

4.2. *Life-cycle GHG emissions of ethanol*

Fig. 8 shows life-cycle CO_2e emissions of GHGs. They include carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O), weighted with their global warming potentials (1, 25, 298 for the three, respectively, based on the 100-year values developed by IPCC [61]) of petroleum gasoline (discussed in a section below), six types of corn ethanol, and three types of cellulosic ethanol. Corn ethanol plants using coal do not offer GHG reductions. Average wet milling plants (with 73% of process fuel being natural gas) offer moderate reductions in GHG emissions. The lower GHG benefits associated with wet milling ethanol are the result of higher energy use in wet milling plants and lower emission credits with wet milling co-products (we used the system boundary expansion method to estimate emission credits of ethanol's co-products). On the other hand, dry milling plants (except those using coal) offer significant GHG reductions, especially when wet DGS is produced and a significant amount of energy use can be avoided. On the other hand, the three types of cellulosic ethanol offer large reductions in GHG emissions because the steam required in cellulosic ethanol plants is generated with lignin of biomass and electricity is exported to displace conventional electricity generation.

Fig. 9 presents life-cycle GHG emission sources for corn ethanol, cellulosic ethanol, and gasoline. For both corn and cellulosic ethanol, the CO_2 uptake during biomass growth is the reason for GHG reductions by the two types of ethanol. GHG emission credits of co-products (animal feeds for corn ethanol and electricity export for cellulosic ethanol) offer additional GHG benefits. For both types of ethanol, the largest GHG emission source is ethanol combustion (which is offset by the CO_2 uptake during biomass growth). For corn ethanol, of the other GHG emission sources, fuel use in ethanol plants is the largest source; GHG emissions from nitrogen fertilizers in cornfields (N_2O emissions from nitrogen fertilizer and CO_2 from lime) comes next; and GHG emissions from LUCs is third. For switchgrass-based cellulosic ethanol, GHG emissions from nitrogen fertilizer in farms are the largest source. For petroleum gasoline, gasoline combustion is the main GHG emission

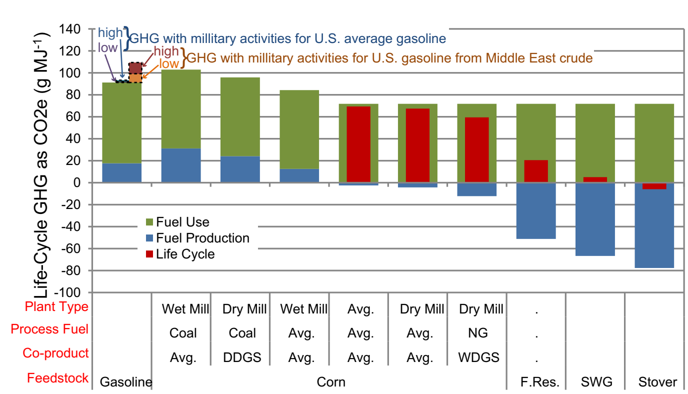

Fig. 8 – Life-cycle greenhouse gas emissions of petroleum gasoline, corn ethanol, and cellulosic ethanol (g CO_2e per MJ).

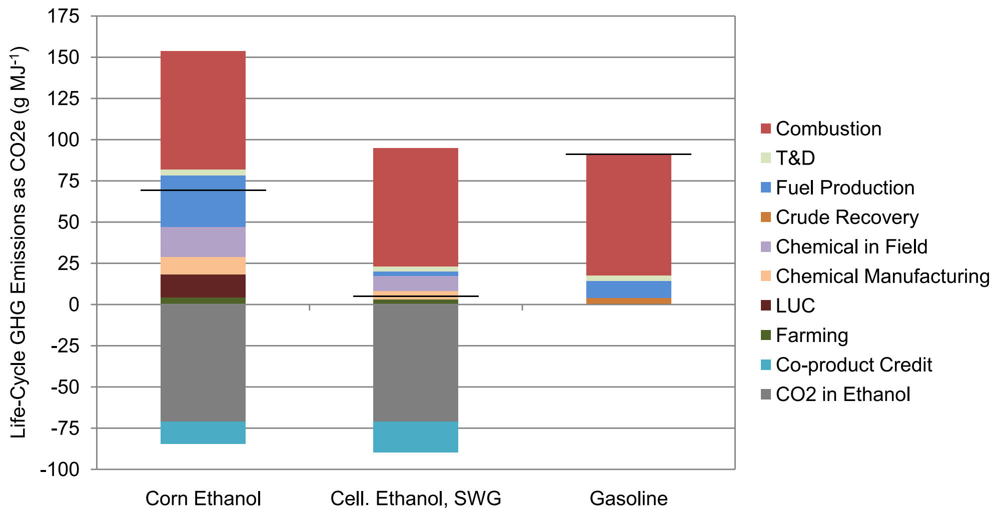

Fig. 9 – Life-cycle greenhouse gas emission sources of corn ethanol, switchgrass-based cellulosic ethanol, and petroleum gasoline (g CO_2e per MJ).

source, while petroleum refining produces noticeable amounts of GHG emissions.

5. Life-cycle GHG emissions of petroleum gasoline

There are direct and indirect effects for the petroleum gasoline cycle. Direct effects (or activities) include petroleum exploration and recovery, petroleum transportation, petroleum refining, gasoline transportation and distribution, and finally gasoline combustion. Energy efficiencies and emissions of these activities in the GREET model are documented in [62]. Petroleum refineries produce several major products (e.g., gasoline, diesel, jet fuel, residual oil, etc.). Allocation of energy use and emissions in refineries among products was studied with several methods in GREET [63]. On the other hand, Canadian oil sands products require oil sands recovery (with either surface mining or in-situ recovery), bitumen upgrading to bring it to a carbon and hydrogen ratio close to that of conventional crude, resulting in a product called synthetic crude oil (SCO) for refining in conventional petroleum refineries [64,65]. GHG emissions of gasoline combustion are dependent on gasoline carbon content, which is known with certainty. GHG emissions of gasoline upstream emissions from wells to station pumps (WTP) are primary from petroleum recovery and refining. Recovery emissions are determined by location and age of oil fields, recovery techniques, flaring and/or venting of associated gas [66]. Petroleum refining emissions are determined by crude quality (such as crude gravity and sulfur content), refined product quality and refinery configuration, among many other factors [63,66,67]. Furthermore, energy use and emissions of oil sands products are affected by mining methods (surface mining or in-situ production), in-situ production techniques (steam assisted gravity drainage or cyclic steam stimulation), and type of fossil energy used for steam and hydrogen production ([64–66,68]). Table 3

Table 3 – Well-to-Pump Greenhouse Gas Emissions of Petroleum Gasoline as CO_2e (g MJ^{-1}).

	GREET[a]	Charpentier et al., 2009 [65] [b]	Bruijn 2010 [68] [c]	Energy-Redefining LLC. 2010 [66]
Weighted average of conventional and unconventional crude	17.7			
Conventional crude: average	16.9	4.5–9.6	22	NA
without gas flaring	NA	NA	NA	8 with SD of 7–10
with significant gas flaring	NA	NA	NA	20 with SD of 15.5–25
Weighted average of oil sands	29.5	NA	37.5	30 with SD of 27–33
Oil sands: surface mining	27.8	9.2–26.5	35.3	NA
Oil sands: in-situ production, average	31.3	16.2–28.7	NA	NA
with steam assisted gravity drainage	NA	NA	38.4	NA
with Cyclic steam stimulation	NA	NA	37.6	NA

NA: not available; SD: standard deviation.
a GREET results are based on LHV. The GHG emissions of petroleum refining are 10.5 g MJ^{-1}.
b The results by Charpentier et al. is a synopsis of 13 studies [65]. The emissions of petroleum refining are not included.
c The results by Brujin is derived using GHGenius and based on HHV. The GHG emissions of petroleum refining are 12.3 g MJ^{-1}, 27.2 g $MJ^{-1}$and 27.2 g MJ^{-1} for conventional crude, oil sands and the average of conventional and unconventional crude, respectively.

summarizes WTP GHG emissions of petroleum gasoline from several sources including the GREET values used in this study.

On the other hand, indirect effects of the petroleum gasoline cycle could include land change from petroleum exploration and recovery and activities of ensuring access to oil in political and military unstable regions. Because oil production per unit of land is significantly larger than biofuel production, the GHG emissions of potential land changes for oil exploration and recovery, when amortized over the amount of oil produced per unit of land over the lifetime of an oil field, could be small. However, military activities in the Middle East by the U.S. to secure U.S. access to the oil in that region could result in large indirect GHG emissions for U.S. petroleum gasoline. Liska and Perrin laid out a method to determine military GHG emissions for gasoline production [69]. They concluded that these GHG emissions could be 8.1–18.2 g per MJ of gasoline produced from Middle East crude. Amortized military GHG emissions for securing foreign oil over the total U.S. gasoline production, they concluded 0.9–2.1 g per MJ of average gasoline produced in the U.S. For comparison with U.S. ethanol, if one maintains that U.S. ethanol at margin displaces Middle East oil, one may use the value of 8.1–18.2 g per MJ. If one is not certain what crude oil is to be displaced by U.S. ethanol, the average value of 0.9–2.1 g could be added to direct GHG emissions of gasoline.

An adequate assessment of military activities related to U.S. gasoline demand may require some kind of economic models to simulate U.S. gasoline demand in a global context. This consequential modeling approach can avoid some difficult decisions such as how much the complete U.S. military activities should be allocated to U.S. oil import and what are the GHG intensities of U.S. military expenditure, among many other issues that Liska and Perrin identified. This is an area where research efforts are needed, similar to the needed research effort for biofuel LUC modeling.

In Fig. 8, the error bars above the gasoline result show the life-cycle GHG emissions including the military GHG emissions for U.S. gasoline production average across gasoline from Middle East crude (the upper bar) or for U.S. average gasoline production (the lower bar).

6. Relative GHG emission changes from gasoline to ethanol

Some studies on impacts of corn ethanol on GHG emissions presented results in terms of the payback period for GHG emissions [22,23]. The payback concept is applicable only to the fuels that could eventually result in GHG reductions over time. The payback concept is similar to the energy balance concept in that both concepts are used for evaluation of ethanol itself rather than a comparison of ethanol against intended products for displacement (i.e., gasoline). If the GHG payback period were to be calculated for gasoline, it would be infinite, since the use of gasoline turns underground carbon into CO_2 to the air and will never result in a reduction of CO_2 to the air.

A reliable way to evaluate ethanol's GHG effects is to compare life-cycle GHG results between ethanol and gasoline, as depicted in Figs. 8 and 9. On the relative change basis, when energy use and GHG emissions from direct activities for the petroleum gasoline cycle are considered, corn ethanol on average can result in 24% reductions in GHG emissions.

If one included military GHG emissions amortized over gasoline from Middle East crude, corn ethanol GHG reductions could be increased from GREET estimated value of 24% (without military GHG emissions) to 30–37% reductions. If one included military GHG emissions amortized over the total U.S. gasoline production, GHG reductions by corn ethanol would be increased to 25–26%.

7. Conclusions

Using updated data on energy use in ethanol plants and updated chemical and energy use in farms, both of which reflect technology improvements, simulated LUC results with improved economic modeling of ethanol, and updated DGS displacement ratios, we estimate that U.S. corn ethanol at present, on average, results in a life-cycle reduction in GHG emissions of 24% relative to the emissions associated with gasoline. Dry milling plants have larger reductions, while wet milling plants have less reductions in GHG emissions. Our results are in contrast to some of the studies published in the past three years for two major reasons: updated data to reflect technology improvements over time and detailed simulations of a few critical issues in modeling the LUCs of corn ethanol. On the other hand, cellulosic ethanol achieves overwhelming GHG reductions.

In the past several years, policy debates of fuel ethanol have been concentrated on potential unanticipated consequences of corn ethanol production in the United States. However, similar levels of efforts should have been made to address technology improvement potentials. In addition, indirect effects for petroleum fuels may need research efforts. There has been a disconnection in modeling and assessment between LUCs and technology potentials. Although LUC simulations have been evolving significantly, critical LUC issues still need to be addressed more accurately [70]. On the other hand, technology potentials for both first- and second-generation biofuels should not be overlooked. Only by focusing on both unanticipated consequences and technology potentials can biofuel policy debates be constructive and sound biofuel policies be formulated. In this sense, the usefulness of such evaluations of biofuel technologies is to provide insight into how to prevent unintended consequences and how to promote efficient technologies with policy intervention.

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