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The economic and environmental costs and benefits of the renewable fuel standard

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Abstract

Mandates, like the renewable fuel standard (RFS), for biofuels from corn and cellulosic feedstocks, impact the environment in multiple ways by affecting land use, nitrogen (N)-leakage, and greenhouse gas (GHG) emissions. We analyze the differing trade-offs these different types of biofuels offer among these multi-dimensional environmental effects and convert them to a monetized value of environmental damages (or benefits) that can be compared with the economic costs of extending these mandates over the 2016–2030 period. The discounted values of cumulative net benefits (or costs) are then compared to those with a counterfactual level of biofuels that would have been produced in the absence of the RFS over this period. We find that maintaining the corn ethanol mandate at 56 billion l till 2030 will lead to a discounted cumulative value of an economic cost of $199 billion over the 2016–2030 period compared to the counterfactual scenario; this includes $109 billion of economic costs and $85 billion of net monetized environmental damages. The additional implementation of a cellulosic biofuel mandate for 60 billion l by 2030 will increase this economic cost by $69 billion which will be partly offset by the net discounted monetized value of environmental benefits of $20 billion, resulting in a net cost of $49 billion over the 2016–2030 period. We explore the sensitivity of these net (economic and environmental) costs to alternative values of the social costs of carbon and nitrogen and other technological and market parameters. We find that, unlike corn ethanol, cellulosic biofuels can result in positive net benefits if the monetary benefits of GHG mitigation are valued high and those of N-damages are not very high.

The renewable fuel standard (RFS) was established by the Energy Independence and Security Act of 2007 with the motive of enhancing energy security, reducing greenhouse gas (GHG) emissions, and promoting rural economic development in the United States (U.S.). The RFS mandated blending of 136 billion l of first-generation biofuels (from food crops) and second generation (cellulosic) biofuels with fossil fuels by 2022. While the corn ethanol mandate has been met and its production has grown to 57 billion l, the production of cellulosic biofuels has been negligible so far (Debnath *et al* 2019)[5]. Although the volumetric targets of various types of biofuels beyond 2022 are yet to be determined, a forward-looking analysis of the potential economic and environmental implications of extending the corn ethanol mandate and adding the cellulosic biofuel mandate to be achieved by 2030 can inform policy discussions.

[5] The commercialization of cellulosic biofuel remains negligible due to technological challenges in converting biomass to reactive intermediates, termed recalcitrance, and to biofuel efficiently at commercial scale, leading to relatively high production costs compared to oil prices (Lynd 2017, Debnath *et al* 2019). However, as recent studies indicate, more advanced bioengineering techniques are being developed to increase the conversion efficiency and lower the production cost of cellulosic biofuel (Jung and Altpeter 2016, Kumar *et al* 2018, Baral *et al* 2019, Liu *et al* 2019).

Biofuel production and their feedstocks can generate multiple environmental impacts; they not only affect GHG emissions by displacing fossil fuels but they also affect nitrogen applications and GHG emissions due to the direct and indirect land use change caused by the increased demand for biofuel feedstocks (see review in Donner and Kucharik 2008, Khanna and Crago 2012, Sun *et al* 2020). This can impact leakage of reactive nitrogen (N) inputs to the environment and affect air, water (surface freshwater, groundwater, coastal) and climate systems. These positive and negative environmental effects of biofuels differ with the feedstock used. Different types of biofuels feedstocks also differ in the economic costs and benefits they will impose on food and fuel consumers and producers depending on their cost of production and competition for cropland. Given these multi-directional environmental and economic impacts of biofuel production, a social cost–benefit approach that monetizes the value of these multiple environmental effects will enable a comparison of the value of the environmental impacts with the economic costs of biofuel mandates and an assessment of the net societal benefits[6] of extending biofuel mandates until 2030.

A key objective of this paper is to examine the economic and environmental consequences of extending corn ethanol and cellulosic ethanol mandates over the 2016–2030 period and to analyze the trade-offs among these multi-dimensional effects using the social cost–benefit approach. This framework allows us to determine the net economic benefit maximizing approach to achieving a biofuel mandate. By comparing net economic benefits with and without the mandate, we assess the net economic benefit or cost of the mandate for the U.S. as well as quantify the distributional effects of these biofuel policies on food and fuel consumers and producers. Since these monetary benefits and costs are occurring over time, and money has a time value[7], we discount future net benefits of these mandates to obtain their present value in 2016 and compare it with the present value of net benefits under a No-Policy scenario. We define the No-Policy scenario as the level of biofuels (24.6 billion l of corn ethanol and 3.52 billion l of biodiesel) that would have been produced in the absence of the RFS being established in 2007 ('No-Policy scenario'). To isolate the effects of the corn ethanol mandate from cellulosic ethanol mandate, we compare the effects of two alternative scenarios; a 'Corn Ethanol Mandate scenario', in which the corn ethanol production remains at the cap of 56 billion l under the RFS and a 'Corn + Cellulosic Ethanol Mandate scenario' in which additional cellulosic ethanol production ramps up to 60 billion l of cellulosic ethanol by 2030. In both these scenarios, biodiesel production increases to 6.03 billion l. We then quantify the effects of these mandates on land use, changes in GHG emissions, and N-leakage over the 2016–2030 time-period.

We undertake this analysis by extending the Biofuel and Environmental Policy Analysis Model (BEPAM). BEPAM is a multi-period, open-economy, multi-market partial equilibrium, optimization model that integrates the agricultural and transportation sectors of the U.S. economy and incorporates trade with the rest of the world (ROW). We apply it to determine the maximum discounted value of the net economic benefits to consumers and producers in the agricultural and transportation sectors over the 2016–2030 period in each of the three scenarios. By comparing these net economic benefits across scenarios, we assess the net economic cost (or benefit) of the biofuel mandate. We quantify the GHG emissions and N-leakage across these scenarios and monetize the change in the value of these environmental impacts using the concepts of social cost of carbon (Interagency Working Group 2013, Khanna *et al* 2017) and the social cost of nitrogen (Sobota *et al* 2015, Keeler *et al* 2016) to determine the net social costs or benefits of the mandates. We consider N-leakage effects caused by the conversion of N to N_2O, NO_x, NH_3, and NO_3 and from N loadings to surface water, groundwater and coastal systems and analyze the trade-offs that the biofuel mandate offers between GHG emissions and N emissions. We examine the effects of various technological, parametric and market conditions on the net societal benefits of these biofuel mandates.

1. Previous literature

There is a large literature analyzing the economic and environmental effects of biofuels (see reviews in Debnath *et al* 2019, Khanna *et al* Forthcoming). Previous studies have focused on examining one environmental impact of biofuels at a time and do not consider the multiple environmental effects of biofuels. They also do not combine them with the economic impacts of biofuel mandates to examine the synergies and trade-offs among them. Many studies have examined the direct life-cycle GHG intensity of biofuels and the market-mediated effects induced by biofuels on food and fuel prices in the world market. These market-mediated effects were shown to lead to indirect land use change (ILUC) in the

[6] Agricultural/fuel consumer benefit is measured by the difference between the maximum consumers are willing to pay for various agricultural/fuel commodities (as given by the demand curves for those commodities) and the price of those commodities. The producer benefit is measured by the profit from producing agricultural/fuel commodities.

[7] Time value of money arises because money we have now is worth more than an identical sum in the future provided money can earn interest. As a result, a comparison of the monetary value of a dollar now and a dollar in the future can only be done after discounting the future dollar and obtaining its present discounted value.

agricultural sector and to fuel rebound effects in the transportation sector[8].

Research examining the ILUC-related emissions intensity of biofuels has largely focused on corn ethanol; there has been limited analysis of the ILUC effects of energy crops produced for cellulosic biofuels[9]. Taheripour and Tyner (2013) examine the land use effects of producing specific levels of miscanthus and switchgrass ethanol and assume that the amount of land needed per-unit ethanol is uniform across geographic regions and linearly related to the volume of biofuels. In contrast to these papers that examined the ILUC effect of a given volume of a specific type of biofuel and do not consider the land use interactions when several types of biofuels will be produced simultaneously, our paper examines these ILUC effects for the endogenously determined mix of cellulosic biofuels from various feedstocks and incorporates the heterogeneity in the yields of these crops across the rainfed region in the U.S. It also uses more updated land use change estimates to obtain forward-looking estimates of the ILUC-related GHG emissions using a 2016 land use baseline.

Previous analyses of the fuel rebound effect of biofuels were undertaken when the U.S. was a major importer of oil and gasoline and a large decrease in its consumption level could be expected to lower the world oil price and lead to a positive rebound effect (such that a liter of biofuel would displace less than a liter of gasoline). A number of early studies showed that this rebound effect in the global fuel market due to biofuel production in the U.S. could be large (Degorter and Drabik 2011, Thompson *et al* 2011, Chen and Khanna 2012, Chen *et al* 2014, Bento *et al* 2015, Rajagopal *et al* 2015, Hudiburg *et al* 2016). Empirical evidence, however, suggests otherwise and that the effect of biofuels on oil price has been small (see review in Khanna *et al* Forthcoming).

With the increase in shale oil and gas production, the U.S. has now transitioned into a smaller importer of oil and a net exporter of petroleum products in the world market (EIA 2017, 2020). Moreover, the literature indicates that the price elasticity of demand for gasoline in the US is low and has been declining (Hughes *et al* 2008, Greene 2012). As a result of both supply and demand side factors, changes in biofuel productions in the U.S. are likely to have a smaller effect on the world oil price. However, blending high-cost biofuels with gasoline in the U.S. can be expected to raise fuel prices in the U.S. and lead to a negative rebound effect, such that a liter of ethanol displaces more than an energy equivalent liter of gasoline, as found to be the case by Rajagopal *et al* (2015). We analyze the extent to which this is the case and its implications on the GHG savings due to biofuels in the transportation sector. The fuel market price impacts affect not only the GHG mitigation achieved by biofuels but also the welfare-economic effects of the RFS. Earlier studies showed that the corn ethanol mandate (Moschini *et al* 2010, Cui *et al* 2011) and the combined corn and cellulosic ethanol mandates (Chen *et al* 2014) increase the economic benefits because they improve the terms of trade of the U.S. (by lowering the price of fuel imports and raising the price of US agricultural exports). However, the changes in the U.S. fuel market discussed above may reduce these beneficial terms-of-trade effects and increase the economic costs of implementing the extended U.S. biofuels mandate in the future.

Previous studies have examined the effects of expanding corn production on N-fertilizer application and its effect on water quality in the Mississippi River Basin (Secchi *et al* 2011, White *et al* 2014). Donner and Kucharik (2008) estimated that meeting the corn ethanol mandate of 57 billion l would result in a 10% increase in nitrogen reaching the Gulf from the Mississippi-Atchafalaya River Basin. More recently, Ferin *et al* (2021) showed that the additional implementation of the cellulosic biofuel mandate would worsen water quality even further by incentivizing the harvesting of crop residues. These studies do not consider damages in and outside the Mississippi-Atchafalaya River Basin due to other forms in which applied N can leak into the environment (as N_2O, NO_x, NH_3, and NO_3) and worsen air quality, contaminate drinking water supply, degrade water quality and negatively contribute to the global climate system (Bennett *et al* 2001, Galloway *et al* 2004, Davidson *et al* 2012, Leach *et al* 2012).

In undertaking this analysis, we extend previous applications of BEPAM to analyze the effects of biofuel policies on the transportation and agricultural sectors (Chen and Khanna 2012, Huang *et al* 2013, Chen *et al* 2014, Hudiburg *et al* 2016, Khanna *et al* 2017) in several ways. First, unlike previous versions of BEPAM that modeled gasoline and diesel markets as independent markets, we have now modeled them as joint products (produced in fixed proportions) from crude oil which is produced domestically and imported from the ROW. We assume the U.S. can affect the price of oil in the world market through its oil import decisions. This is based on the

[8] The ILUC effects in the agricultural sector could occur because biofuels have the potential to raise food crop prices and lead to expansion of cropland in the US and ROW (Searchinger *et al* 2008, Hertel *et al* 2010, Taheripour and Tyner 2013, Witcover *et al* 2013).The rebound effect in the oil market could occur if the large-scale displacement of demand for petroleum products by biofuel in the U.S. reduces demand for oil in the world market; this could lower the world oil price, leading to a 'rebound' in oil consumption in the ROW.

[9] Several studies that have analyzed the potential of corn ethanol (Farrell *et al* 2006, Chen and Khanna 2012, Rajagopal 2013) and cellulosic ethanol (Tilman *et al* 2009, Gelfand *et al* 2013, Hudiburg *et al* 2016, Daioglou *et al* 2017, Khanna *et al* 2017) to lower GHG emissions and the extent to which the ILUC effect due to the large-scale production of biofuels would offset the direct saving due to biofuels displacing fossil fuels (see reviews in Khanna and Crago 2012, Khanna *et al* 2017).

observation that the U.S. is still a fairly large importer in the crude oil market (EIA 2017). However, we now model the U.S. as a small, price-taking exporter of petroleum products to the ROW (EIA 2017) instead of a large importer of gasoline (with the ability to lower world prices through biofuel-induced displacement). We also assume a relatively elastic supply curve for crude oil[10] based on previous studies (Hughes *et al* 2008, Hochman *et al* 2011, Greene 2012, Hochman and Zilberman 2018)[11]. Second, we determine the domestic direct land use change and ILUC, due to the biofuel mandates, endogenously. As a result, the domestic indirect land use related GHG emissions intensity of biofuels is determined endogenously and may vary over time rather than being a constant value assumed in previous applications of BEPAM (such as Khanna *et al* 2017).

2. Methods

BEPAM determines the optimal land use, production and consumption decisions in the agricultural and transportation sectors in the U.S. that maximize the sum of consumers' and producers' net benefits in both the sectors subject to various material balances, land and resource availability constraints, and policy scenarios. It endogenously determines the quantities of row crops, biomass feedstock, biofuels, and fossil fuels and their prices that ensure that market demand and supply are in equilibrium. It incorporates domestic agricultural and fuel markets in the U.S. and trade in agricultural commodities and petroleum products with the ROW.

The model considers spatial heterogeneity in crop and livestock production, where costs of production, yields, and land availability differ across crop reporting districts (CRDs). Crops can be produced using alternative rotation, tillage, and irrigation practices. The model simulates optimal land use allocation for major row crops and energy crops on active cropland as well as land that is considered idle[12] endogenously based on the availability of land, the net returns to crop production, endogenously determined crop prices, historical land mix constraints, policy, and technology constraints. The amount of land under crop production and idle changes annually depending on returns to the land[13]. By comparing the amount of land under crop production in each year after 2016 in the No-Policy scenario, we determine the amount of land that could have been planted but was not planted (and therefore idled) in each year.

The transportation sector in BEPAM consists of downward sloping linear demand functions for vehicle kilometers traveled (VKT) with four different types of vehicles: conventional gasoline, flex-fuel, gasoline-hybrid, and diesel vehicles. We implicitly derive the gasoline and diesel demand from the VKT demand functions for road transportation by incorporating fuel efficiency assumptions for the vehicle fleet[14]. We specify the domestic oil supply and the demand and supply of oil in the ROW. The U.S. is assumed to be an importer in the oil market, but a small, price-taking exporter in the world market for gasoline and diesel. Domestically produced oil and imported oil are converted to gasoline and diesel in a fixed proportion. The model endogenously determines the implicit cost of VKT depending on the marginal cost of oil, costs of conversion of oil to gasoline and diesel, extent and mix of biofuels blended and the operation and maintenance costs of vehicles under the alternative biofuel policies. The transportation sector only considers demand for on-road VKT and does not include the aviation sector. More details of the model can be found in the supplementary information (sections 1 and 2) (available online at stacks.iop.org/ERL/16/034021/mmedia).

To analyze the economic and environmental effects that can be attributed to various biofuels, we simulate the following three alternative scenarios over the 2016–2030 period:

(a) No-Policy: corn ethanol and biodiesel production are assumed to remain at their 2007 levels of annual production of 24.6 billion l and 3.52 billion l each year.
(b) Corn Ethanol Mandate: annual corn ethanol production is assumed to be at the maximum permitted level of 56.78 billion l under the RFS

[10] Domestic price elasticity of oil supply is assumed to be 2, ROW oil price supply and demand elasticities are assumed to be 1 and −4 based on Thompson *et al* (2011). More details are documented in section 1.3 of SI.

[11] Studies that model upstream oil market estimate a smaller effect relative to those that focus only on finished fuel markets (see the review in Hochman and Zilberman 2018). These estimates could be even smaller if the strategic behavior by the cartel of oil producing nations (e.g. OPEC) is considered. For example, Hochman *et al* (2011) argue the cartel would lower oil production in response to increased biofuel production, mitigating the negative impact of biofuel displacement on oil price.

[12] This is defined as cropland that likely earns close to zero net returns from crop production (Jiang *et al* 2020). More details on the how active cropland and idle land available in 2016 were determined can be found in section 2.1 of SI.

[13] Landowners are assumed to use a 10 year rolling horizon to make land use decisions for the next 10 years (for example, investment in establishment of perennial energy crops to meet future targets for cellulosic ethanol) taking land availability, technology and policy as given. From the resulting multi-year equilibrium solution, we take the first-year's solution as the 'realized' solution and used them to update the land that is under crop production and idled and costs of producing biofuels due to learning by doing. More technical details of rolling horizon and multiple-year equilibrium solution can be found in Chen *et al* (2014).

[14] The fuel efficiency projections are obtained from Annual Energy Outlook (EIA 2017). We are assuming the demand for VKT with alternative types of vehicles is determined exogenously and are only considering the marginal fuel and operation and maintenance related costs of VKT with different vehicles under alternative biofuel policies.

and annual biodiesel production is assumed to be 6.03 billion l.

(c) Corn + Cellulosic Ethanol Mandate: Cellulosic ethanol production is assumed to grow linearly from 0.87 billion l in 2016 to 60.56 billion l in 2030 while annual corn ethanol and biodiesel production levels are at levels in the Corn Ethanol Mandate scenario.

We implement the mandate in BEPAM by establishing annual blend rates to meet the volumetric goals for ethanol and biodiesel; these blend rates increase beyond the 11.5% level in 2016 to 24.2% by 2030 under the Corn + Cellulosic Biofuel Mandate scenario. To determine the social welfare effects of the two alternative biofuel policy scenarios, we assess the impact of each of these policies on the food and fuel prices and quantities in the agricultural and transportation sectors and, thus, on the discounted value of the sum of consumer and producer net benefits in these sectors relative to the No-Policy scenario over the 2016–2030 period. We assume a social discount rate of 3% (for more discussion of the social discount rate see Boardman *et al* 2017)[15].

2.1. Social costs of GHG emissions estimation

We estimate the direct GHG emissions intensity of the various types of biofuels and the GHG emissions mitigated by displacing fossil fuels. The aboveground GHG emissions related to agricultural activities, including planting, maintenance and harvest are estimated by multiplying various production inputs with corresponding emission factors obtained from the GREET model (see details in the SI in section 3). We endogenously determine the U.S. domestic ILUC related GHG emissions generated by the food crop price effects caused by the production of biofuels to meet different biofuels mandate. Emissions due to the conversion of idle cropland in the U.S. in 2016 to crop production (conventional or energy crops) in response to changes in market demand and crop prices are obtained by multiplying the amount of idle land converted with the emission-factors obtained from Taheripour *et al* (2017). International ILUC emissions due to the conversion of cropland to biofuel feedstocks in the ROW are not accounted for in this study (estimates of those are reviewed in Khanna and Crago 2012, Taheripour *et al* 2017). We also incorporate soil carbon sequestration by both row and energy crop productions; the extent of this varies by crops, land types, and production operations (Hudiburg *et al* 2016, also see more details in section 3 of SI).

[15] There is no consensus in the literature on the appropriate discount rate to use in a social cost benefit analysis. As a result, U.S. EPA (2017) estimated social cost of carbon under various discount rates (e.g. 2.5%, 3% and 5%).

We then estimate the social costs of GHG emissions due to the implementation of the U.S. biofuel policies by multiplying the total GHG emissions with the social cost of carbon, which is a measure of the discounted monetary value of the global damages due to carbon emissions. There is a wide disparity in the range of estimates of the social cost of carbon but a considerable consensus that \$50 per metric ton of CO_2 equivalent (Mg^{-1} CO_2-eq) is a reasonable estimate with the same 3% social discount rate assumed here (EPA 2017, Khanna *et al* 2017). We analyze the sensitivity of our social costs and benefits of the RFS by considering two alternative values, \$50 Mg^{-1} CO_2-eq and \$100 Mg^{-1} CO_2-eq.

2.2. N-damage cost estimation

We apply our modeling approach to assess the spatially varying additions to N induced by the corn ethanol and the cellulosic ethanol mandates over the 2016–2030 period. The extent to which applied N will leak into the environment and the form of that leakage and the amounts that are transported to end-points (drinking water wells, coastal zone, atmosphere) where the environmental damages occur are expected to vary spatially. We estimate the N-leakage to the environment due to agricultural anthropogenic nitrogen fertilizer use in each period by multiplying the simulated total volume of N application with coefficients obtained from Sobota *et al* (2015) that measure the transfer of synthetic N-fertilizer to land, air, and water resources (see more details in section 4 of SI and table S1).

In the absence of N-leakage cost data at a state-specific level for all the states in the U.S., Sobota *et al* (2015) complied the damage costs associated with N applications based on information obtained from existing studies (Compton *et al* 2011, van Grinsven *et al* 2013) (see details in table 1 in Sobota *et al* 2015). We estimate the potential N-damages costs (\$ kg^{-1} N) by the environmental resource affected (air/climate, freshwater, drinking water, and coastal zone)[16]. The total N-damage costs in each CRD are then calculated by multiplying the total amount of N-leaked to the environment with the per-unit damage costs (\$ kg^{-1} N) associated with the corresponding air/climate, freshwater, drinking water, and coastal zone related damages. Assuming these damage costs increase linearly with an additional unit of

[16] Damages due to N_2O arise due to increased ultra-violet light exposure from ozone to humans and to crops and increased emissions of GHGs. Damages to surface water from N loadings include those due to declining property value, loss of recreational use and loss of endangered species. N-damages to surface-water and ground-water to drinking water include those due to increased eutrophication, undesirable odor and taste, nitrate contamination and increased colon cancer risk while those to coastal ecosystems stem from loss of recreational use and decline in fisheries and estuarine/marine habitat.

nitrogen leaked, we sum across these types of nitrogen damage costs to estimate the total N-damages for each CRD and the contiguous U.S. These damage costs range from a low value of \$5.45 kg^{-1} N applied, a medium value of \$10.17 kg^{-1} N applied and a high value of \$16.87 kg^{-1} N applied[17]. We analyze the sensitivity of the overall social welfare to a range of values of the social cost of N estimated by Sobota *et al* (2015). Our analysis here relies on simplifying assumptions that social cost of N-damage is spatially homogenous. However, this estimate, as Sobota *et al* (2015) noted, is compiled from large-scale studies (national or regional); as a result it is reasonable to consider it to be representative of values over the large geographical scale studied here.

3. Results

We validate BEPAM by comparing simulated outcomes in the fuel and agricultural sectors over the 2016–2019 period with observed data. We find the land allocation to the major U.S. crops (corn, soybean, and wheat) deviates by less than 10% from observed data (table S2(a) in SI). Average national food crop prices are generally within 10% of the observed values except for the corn price, which is 11% higher than the actual prices observed in 2016–2018 (panel B in table S2(a) in SI)[18]. Fuel consumption also generally deviate by less than 10% from observed values (table S2(b) in SI). The deviations of the simulated outcomes of the updated BEPAM for 2016–2019 from their observed values over the 2016–2019 period are generally within a similar level of tolerance as in previous studies applying BEPAM (Chen *et al* 2014, Hudiburg *et al* 2016).

3.1. Land use in the No-Policy scenario

The land requirements for conventional crops and biofuel feedstocks and their quantities produced and prices under the various scenarios are presented in table 1. Under the No-Policy scenario, we find that 109 million ha (M ha) land will be used for row crop production in 2030 and there will be 15 M ha of idle cropland (table 1). Of this 11 M ha land is idle land in 2016, while the remaining 4 M ha is cropland that used to be under crop production in 2016 but would become idle by 2030 in the No-Policy scenario (figure S1(a)). The time-varying amount of idle cropland is driven by two assumptions in the model: an exogenous growth in yields of row crops (around 1% yr^{-1} for corn, soybean and wheat) and an outward shift in domestic demand and export demand overtime at exogenously specified rates[19]. Consistent with historical trends in land under crop production in the U.S., we find that the increasing trend in productivity growth will outpace growth in the demand for agricultural production in the U.S.; therefore, the total land used for crop production will decrease by 3.5% by 2030 compared to that in 2016, inducing a net increase of 4 M ha in idle cropland by 2030.

3.2. Effect of the biofuel mandate on agricultural sector

Under the Corn Ethanol Mandate, the demand for corn production will increase by 21%, which will increase land under corn production by 23% (as shown in column 2, table 1); this increase is achieved, in part, by reducing land under soybeans and wheat and in part by increasing land under crop production. The increase in demand for corn will lead to an increase in land under continuous corn and under corn–soybean rotation while reducing land under wheat and continuous soybeans. Overall, land under wheat and soybeans declines by 1.6% and 0.4%, respectively, relative to the No-Policy scenario. Higher corn demand will increase its price by 12%. The price of soybeans increases by 7% as land under soybeans declines marginally while the production of biodiesel increases the demand for soybeans. Despite the reduction in land under wheat, wheat production increases as its production shifts to productive land released from soybeans and its yield per-unit land increases. As a result, the price of wheat decreases slightly by 1.2% relative to the No-Policy scenario (table 1). Overall, the increased demand for corn ethanol will increase land under crop production by 5.9 M ha (5.4%) in 2030 relative to that in the No-Policy scenario, including 1.9 M ha (1.7%) from land that was idle in 2016 and 4 M ha (3.7%) from land that would otherwise have been idled in 2030 under the No-Policy scenario (figure S1(b)).

We estimate the GHG emissions due to this indirect land use change of 1.9 M ha that can be attributed to corn ethanol and biodiesel mandates by converting the total changes in idle land to a per-liter estimate using the approach in Chen and Khanna (2018)[20]. We find that the idle land in 2016 converted

[17] The medium value estimated here is similar to the social cost of \$10.79 kg^{-1} N from fertilizer application in Minnesota by Keeler *et al* (2016). The medium values of N-damage costs related to nitrate estimated here is also close to the social value of \$8.7 kg^{-1} N for nitrate reduction in the Corn Belt region by Ribaudo *et al* (2005). All values are in 2016 dollars.

[18] We apply the validated model to simulate outcomes for 2030 and compare outcomes with the No-Policy scenario. In undertaking this comparison, we are keeping all parameters and model assumptions exactly the same in the No-Policy and alternative policy scenarios. We, therefore, expect that the model validation error is the same across the scenarios and washes out when we take a difference across scenarios to evaluate the impact of a policy scenario.

[19] The increase in the national average crop yields resulted in a slight decrease in commodity prices for crops over the study period and lower expected returns for cropland, inducing cropland with relatively low productivity to exit from crop production and become idle. This is offset to some extent by the outward shift in demand which increases the expected returns to cropland.

[20] We estimate the static impact of a biofuel production shock by calculating the change in the idle land in 2016 per unit of the increase in corn ethanol and biodiesel production in 2030.

Table 1. Effects of alternative biofuels mandate on the agricultural sector in 2030.

Scenario	No-Policy (1)	Corn Ethanol Mandate (2)	Corn + Cellulosic Ethanol Mandate (3)	No-Policy (4)	Corn Ethanol Mandate (5)	Corn + Cellulosic Ethanol Mandate (6)
	Land allocation (million ha)			Crop price ($\$\ Mg^{-1}$)		
Row crop (a)						
Corn	27.5	33.7 (22.6%)	33.4 (−0.9%)	126.0	141.0 (11.9%)	143.6 (1.9%)
• Continuous corn	*13.2*	*17.2 (30.4%)*	*17.5 (1.7%)*	—	—	—
• Corn in rotation	*14.3*	*16.5 (15.4%)*	*15.9* (−3.6%)	—	—	—
Soybeans	33.8	33.3 (−1.6%)	32.7 (−1.8%)	261.48	280.0 (7.1%)	295.7 (5.6%)
Wheat	18.4	18.3 (−0.4%)	18.2 (−0.8%)	170.40	168.4 (−1.2%)	174.0 (3.3%)
Other crops	29.7	30 (1.0%)	29.2 (−2.8%)	—	—	—
Energy crop (b)	—	—	4.7	—	—	—
					Cropland rent ($\$\ ha^{-1}$)	
Land under crop production (a + b)	109.4	115.3 (5.3%)	118.2 (2.5%)	335.0	354.8 (6.0%)	366.4 (3.2%)
Idle land	15.1	9.2 (−39%)	6.3 (−31%)	—	—	—
	N-fertilizer applied (million Mg)					
	8.69	9.77 (12.4%)	10.63 (8.8%)	—	—	—

Note: table 1 shows the U.S. land allocation and projected prices of row crops and energy crops by 2030 under the No-Policy, Corn Ethanol Mandate and Corn + Cellulosic Ethanol Mandate scenarios. The percentage changes in the parentheses of column (2) and (5) refer to the changes in the Corn Ethanol Mandate scenario relative to that of the No-Policy scenario. The percentage changes in the parentheses of column (3) and (6) refer to the changes in the Corn + Cellulosic Ethanol Mandate scenario relative to that under the Corn Ethanol Mandate scenario.

to crop production due to corn ethanol and biodiesel over the 2016–2030 period amounts to a combined 49 ha million^{-1} l of corn ethanol and biodiesel in volumetric terms. This is slightly higher than the estimate obtained by Chen and Khanna (2018) which ranged from 12 to 48 ha million^{-1} l of corn ethanol. However, their estimate is only for corn ethanol while ours are for both corn and biodiesel.

The implementation of Corn + Cellulosic Ethanol Mandate will lead to a modest increase in food prices and land rents of cropland since much of the production of energy crops will occur on cropland that would have been idled under the No-Policy scenario. Relative to the Corn Ethanol Mandate, corn and soybean prices and land rent of cropland will increase by 2%, 6%, and 3% in 2030, respectively. The intensive and extensive margin effects on land use under Corn + Cellulosic Ethanol Mandate differ from those under Corn Ethanol Mandate. We find the additional 60 billion l of cellulosic ethanol mandate will lead to an additional expansion of cropland by 3 M ha in 2030 relative to cropland under the Corn Ethanol Mandate, such that total cropland will expand by 9 M ha relative to the No-Policy scenario. The land under row crops will decrease slightly by 1.8 M ha in 2030 (column 3 of table 1). Total land under corn production remains similar to that under the Corn Ethanol Mandate; but there is a shift in corn produced under a corn–soybean rotation to that produced under a continuous corn rotation due to incentives for harvesting corn stover. The shift to continuous corn production lowers the average yield of corn and increases N application to compensate for the absence of nitrogen fixation by soybeans and to compensate for nutrient removal with stover collection (see more details in section 3.4.2.). We find 4.7 M ha land will be converted to energy crops in 2030 under the Corn + Cellulosic Ethanol scenario. We compare the land allocation in each year after 2016 under No-Policy scenarios and Corn + Cellulosic Ethanol scenario at the CRD level. We find that, for the 4.7 M ha land converted to energy crops in 2030 under the Corn + Cellulosic Ethanol scenario, 30% (1.4 M ha) will be from land that was idle in 2016 in both No-Policy and Corn + Cellulosic Ethanol scenario and another 30% (1.4 M ha) will be from land that becomes idle over the 2016–2030 period under the No-Policy scenario due to the growth of productivity, as we noted above. Only the remaining 40% (1.8 M ha) of land for energy crop production will require diversion of land that would have been under food crop production under the No-Policy scenario in 2030 (figure S1(c)). We find that the indirect increase in land use (due to conversion of idle land in 2016) with cellulosic ethanol production amounts to 23 ha million^{-1} l of cellulosic ethanol. This is half of the ILUC effect under the Corn Ethanol Mandate scenario, indicating that the cellulosic mandate can be expected to have a much smaller ILUC-related GHG intensity compared to the Corn Ethanol Mandate.

Table 2. Effects of alternative biofuels mandate on the fuel sector in 2030.

Scenario	No-Policy (1)	Corn Ethanol Mandate (2)	Corn + Cellulosic Ethanol Mandate (3)	No-Policy (4)	Corn Ethanol Mandate (5)	Corn + Cellulosic Ethanol Mandate (6)
		Fuel consumption (billion l)			Consumer fuel prices ($ l^{-1})	
Gasoline	398.8	373.2	324.6	0.64	0.66	0.69
Diesel	202.9	199.5	187.0	0.57	0.59	0.62
Corn ethanol	21.3	56.9	56.8	0.70	0.72	0.72
Cellulosic ethanol			60.6			0.87
		(Billion gasoline-equivalent liters)			($ per gasoline-equivalent liter)	
Total fuel consumption	413.0	411.1 (−0.5%)	402.9 (−2.0%)	0.64	0.66 (2.5%)	0.69 (5.0%)
		(billion km)			($ km^{-1})	
Total VKT	664.9	661.9 (−0.9%)	648.6 (−1.3%)	0.057	0.058 (2.5%)	0.062 (5.1%)

Note: table 2 shows the U.S. fuel consumption and prices 2030 under the No-Policy, Corn Ethanol Mandate and Corn + Cellulosic Ethanol Mandate scenarios. The percentage changes in the parentheses of column (3) and (6) refer to the changes in the Corn + Cellulosic Ethanol Mandate scenario relative to that of the Corn Ethanol Mandate scenario.

3.3. Effect of alternative biofuels mandate on the transportation sector

The implementation of the Corn Ethanol blending Mandate (and biodiesel mandate) will increase biofuel consumption by implicitly subsidizing the biofuel and implicitly taxing the fossil fuel, which create wedges between their consumer and producer prices (Chen *et al* 2014, Holland *et al* 2015). We find the Corn Ethanol Mandate will raise the price of blended fuel ($ per gasoline-equivalent liter) and the price of VKT ($ km^{-1}) by 2.5%, which will decrease blended fuel consumption and VKT by 0.5% and 0.9%, respectively relative to No-Policy scenario (table 2). The reduction in gasoline consumption will be 11% larger than the increase in biofuel consumption in an energy-equivalent base, indicating a liter of biofuel will displace more than an energy equivalent liter of gasoline in the U.S. (negative domestic rebound effect).

Our finding on the rebound effect in the ROW is also significantly different from previous studies that model U.S. as a major importer of gasoline. For instance, earlier studies such as Degorter and Drabik (2011) show that the blend mandate could lead to a positive rebound effect of 50%–65% (a liter of biofuel displaces less than an energy equivalent liter of gasoline) in the world gasoline consumption by using a model that was calibrated to reflect the situation in U.S. and ROW in 2009. Studies using an earlier version of BEPAM, such as Chen *et al* (2014), show that the RFS leads to rebound effects of 9% and 54% on domestic and ROW gasoline consumption in 2030, respectively. However, the optimal solution from our model shows that U.S. would continue importing crude oil in the same quantity as before and will choose to export the excess gasoline displaced by ethanol to the world market rather than reducing the domestic production and import of oil. This is in part because the mandates for ethanol are much larger than for biodiesel and they reduce demand for gasoline by much more than they reduce demand for diesel. Given the assumption that gasoline and diesel are produced in fixed proportions from oil, it is optimal to continue to produce and import oil to meet the demand for petroleum diesel and to export the surplus gasoline. We therefore find that ethanol production does not affect the world oil market price, and hence there is no rebound effect on the ROW.

The implicit tax imposed by the 60 billion l cellulosic mandate will lead to an additional increase in the consumer prices of gasoline and diesel by over 5%, which will lead to a further reduction in gasoline and diesel consumption by 49 billion l (13%) and 13 billion l (1%), respectively, relative to the Corn Ethanol Mandate. The total VKT ($ km^{-1}) increases by 5.1%, which will decrease the total blended fuel consumption and total VKT by 2.0% and 1.3%, respectively, relative to Corn Ethanol Mandate (table 2). We find the domestic rebound effect will be more negative under the cellulosic mandate due to the higher cost of cellulosic biofuels, and the reduction in gasoline and diesel will be 50% larger than the energy-equivalent increase in biofuel.

3.4. Effect of the alternative biofuel mandates on GHG emissions and N-damage

3.4.1. GHG emissions

The cumulative carbon emissions are estimated to be 31.2 billion Mg CO_2-eq with 29.7 billion Mg CO_2-eq (94%) from gasoline and diesel related emissions and 1.6 billion Mg CO_2-eq (5%) from agricultural production, respectively, over 2016–2030 in the No-Policy scenario (table S3). This will decline by 2.5% with the Corn Ethanol Mandate (including those due to domestic ILUC) relative to the No-Policy scenario. We find that the largest decline in emissions

will be in the transportation sector because of the biofuel-induced reduction in gasoline consumption (-1.3 billion Mg CO_2-eq) related emissions. Compared to the No-Policy scenario, the life-cycle GHG emission from the agricultural sector will increase by 0.28 billion Mg CO_2-eq, including 0.2 billion Mg CO_2-eq (71%) from the emissions caused by the expansion of corn production, and 0.1 billion Mg CO_2-eq (29%) from the domestic ILUC emissions induced by the Corn Ethanol mandate and increase in biodiesel production. The total domestic ILUC emissions from the additional corn ethanol and soybean biodiesel production on a per megajoule (MJ) basis amount to 9 g CO_2-eq MJ^{-1}. This estimate is within the range of ILUC emissions estimate for corn ethanol of 8.7 g CO_2-eq MJ^{-1} and that for soybean biodiesel of 18 g CO_2-eq MJ^{-1} (including both domestic and international ILUC effects) in Taheripour *et al* (2017)[21].

The implementation of Corn + Cellulosic Ethanol Mandate will lead to an additional abatement of 1.25 billion Mg CO_2-eq (4.1%) over 2016–2030 relative to the Corn Ethanol scenario. Of the 4.1% reduction in the cumulative GHG emissions, 3.1% will occur in the transportation sector and the remaining 1% in the agricultural sector mainly because of the higher soil carbon sequestration due to the increase in energy crop production. We find a modest change in cumulative ILUC-related emissions of 0.01 billion Mg CO_2-eq in the case of Corn + Cellulosic Mandate scenarios over 2016–2030; this implies domestic ILUC emissions per MJ of 1.5 g CO_2-eq MJ^{-1}, which is only one-fifth of that under the Corn Ethanol Mandate. This is consistent with the findings in the literature that the cellulosic ethanol mandate will only be partly met by energy crops and there will be less diversion of cropland to biofuels (for energy crop production) compared to the diversion of cropland to food crops for biofuel under the Corn Ethanol Mandate scenario.

3.4.2. N-damage

Table S4 shows the N-fertilizer applied in 2030 under No-Policy, Corn Ethanol Mandate, and Corn + Cellulosic Ethanol Mandate scenarios. We find that compared to the No-Policy scenario, the total N-fertilizer applied to corn by 2030 under the Corn Ethanol Mandate scenario will increase by 12.4% while the N-fertilizer applied to other crops will decrease slightly by 2.7% due to the expansion of corn production. The implementation of an additional cellulosic mandate will lead to an additional 0.86 million Mg (8.8% increase) N-fertilizer applied, relative to the Corn Ethanol Mandate. Of the 8.8% increase in N-fertilizer applied, 5.4% can be attributed to the additional N-fertilizer applied to compensate for the soil nutrients removed with the crop residues, 1.4% is attributed to the additional N-fertilizer applied on land that shifts to continuous corn from corn–soybean rotation, while the remaining 2% is applied for producing energy crops.

3.5. Economic and environmental effects of the biofuel mandates

We find that the Corn Ethanol Mandate will lead to a significant decline in the economic net benefits over 2016–2030 (column 2 in table 3). In the transportation sector, the biofuel mandate will increase gasoline and diesel prices, lowering fuel consumers' benefits by \$159 billion (−2%). While the agricultural producers' benefits rise by \$105 billion (12.5%) due to an increase in the food prices and planted acreage, these policies reduce the agricultural consumers' benefits by \$49 billion ((−)2.3%), relative to the No-Policy scenario. Overall, imposing the Corn Ethanol Mandate will lead to a net economic loss of \$109 billion (1% of the total economic net benefits under the No-Policy scenario). This is in contrast to previous studies that show the biofuel mandates increase the net economic benefits (Chen *et al* 2014, Hudiburg *et al* 2016). A major reason for this difference is the structural change in the U.S. fuel market due to which biofuel production in the US does not lower oil imports and the price of oil but it increases gasoline exports instead. As a result, the improvement in terms of trade for the US (defined by the ratio of the price of exports to imports) is not as large as in previous studies. Similar to Chen *et al* (2014) and Hudiburg *et al* (2016), we also find biofuel mandates will lead to net economic benefits in the agriculture sector due to the mandate-induced increase in commodity prices. However, we find a substantial economic cost in the fuel sector since there is no longer a biofuel-induced reduction in the price of oil imports, as discussed above in section 3.3.

The implementation of Corn + Cellulosic Biofuel Mandate will lead to a further reduction in economic benefits compared to the Corn Ethanol Mandate only (column 3 of table 3). Although the economic benefits in the agricultural sector will increase by \$23 billion (0.8%), the benefits in the transportation sector will decrease by \$96 billion (1.3%) over 2016–2030 relative to Corn Ethanol Mandate scenario. We estimate the additional economic loss induced by the Corn + Cellulosic Mandate will be \$69 billion, which is around 0.6% of the total economic benefits under the Corn Ethanol Mandate.

[21] Our model examines land use change only in the U.S. Therefore, this calculation only includes the domestic ILUC-related GHG emissions. If we use estimates from the literature, such as, 8.7 g CO_2-eq MJ^{-1} for corn ethanol and 20 g CO_2-eq MJ^{-1} for soybean biodiesel (Taheripour *et al* 2017), the ILUC-related emissions in the Corn Ethanol Mandate scenario relative to the No-Policy scenario would increase by 0.005 billion Mg CO_2, which would be a negligible reduction in the GHG emissions savings relative to the No-Policy scenario reported here.

Table 3. Effects of alternative biofuels mandate on social welfare over 2016–2030.

Scenario	No-Policy (1) Quantity ($ billion)	Corn Ethanol Mandate (2) Change relative to No-Policy scenario: Absolute change ($ billion)	% change	Corn + Cellulosic Biofuel Mandate (3) Change relative to Corn Corn Ethanol Mandate scenario: Absolute change ($ billion)	% change
Agricultural sector	2992	56	1.9%	23	0.8%
Agricultural consumers	2155	−49	−2.3%	−5	−0.2%
Agricultural producers	837	105	12.5%	28	3.0%
Transportation fuel sector	7305	−161	−2.2%	−96	−1.3%
Crude oil producer	65	−1	−1.7%	0	0.0%
Gasoline consumers	4256	−108	−2.5%	−64	−1.6%
Diesel consumers	2984	−51	−1.7%	−32	−1.1%
Government revenue	854	−4	−0.5%	4	0.5%
Economic benefits (a)	11 151	−109	−1.0%	−69	−0.6%
Social cost of GHG emissions	−1561	39	−2.5%	63	−4.1%
Social cost of nitrogen damages	−1032	−129	12.5%	−42	3.6%
Values of environmental effects (b)	−2593	−90	3.5%	20	−0.8%
Total social welfare (a + b)	**8558**	**−199**	**−2.3%**	**−49**	**−0.6%**

Note: all values are discounted cumulative numbers over 2016–2030 (discounted rate 3%). The environmental costs of alternative biofuel policies by discounting the monetary value of cumulative GHG emissions, and the monetary value of cumulative N-damage costs 2016–2030, as noted in sections 2.1 and 2.2.

The monetized values of GHG emissions and N-damage are substantial across the three scenarios (table 3). We find that there is a trade-off between higher benefits from GHG emissions saving and higher value of damages due to N-leakage. For instance, while the Corn Ethanol Mandate will lead to savings in GHG emissions by 2.5% valued at $39 billion over the 2016–2030 period relative to the No-Policy scenario, it will also increase the N-damage costs by $129 billion (12.5%). Similarly, Corn + Cellulosic Biofuel Mandate will increase the savings in GHG emissions by 4.1% valued at $63 billion but also lead to additional $42 billion in N-damage costs; thereby generating an additional $20 billion benefit relative to Corn Ethanol Mandate scenario (as shown in table 3).

Overall, we estimate that implementing the Corn Ethanol Mandate will lead to a net cost of $199 billion (−2.3%) over 2016–2030 relative to the No-Policy scenario, while the implementation of Corn + Cellulosic Biofuel Mandate will impose an additional economic cost of $49 billion (0.6%) compared to Corn Ethanol Mandate over the same period.

3.6. Sensitivity analysis

The future values of several technical and market parameters in BEPAM are uncertain and can affect the costs and benefits of biofuel mandates. These include the export prices of gasoline and diesel, the demand for US exports of agricultural commodities, the processing cost of cellulosic biofuel, as well as key assumptions such as N-fertilizer replacement rates for crop residues and the values of the social cost of carbon and N-damage. We examine the robustness of our key results to alternative assumed values in BEPAM. First, the economic cost of blending biofuels is expected to be lower if the price of petroleum products is high. We examine the extent to which this is the case by simulating a scenario in which the export prices of gasoline and diesel are 5% higher than those in the benchmark. Second, an increase in the demand for US exports for agricultural commodity could substantially increase due to the potential growing demand for food as incomes in developing countries increase and diets diversify. Increased production of biofuels in this case could lead to a larger increase in the prices of agricultural commodities and improvement in US terms of trade; this would lower the economic costs of biofuels. We examine the sensitivity of our results to assumptions about the annual rate of growth of demand for US exports of agricultural commodities by simulating economic costs with a substantially higher growth rate (3% instead of 2% in the benchmark version of BEPAM) of export demand[22].

Third, the economic costs of the cellulosic biofuel mandate for fuel sector will depend on the processing cost of producing cellulosic biofuel since it will affect blended fuel prices for fuel consumers. We consider a case where improved technology for producing cellulosic biofuels lowers the initial production cost of cellulosic biofuel by 25% from $0.57 l^{-1} to $0.43 l^{-1} and increases the rate of learning-by-doing over the study period by 25% relative to the benchmark scenario. Lastly, we analyze the effect of

[22] In our benchmark scenarios, the rate of growth of export demand for major agricultural commodities in the ROW, including meat, cereals and oil crops is assumed to be 2%, which is consistent with FAO projections (Alexandratos and Bruinsma 2012). More details can be found in table S10 of the SI.

Table 4. Change in social welfare with Corn Ethanol Mandate relative to No-Policy scenario over 2016–2030 period ($ billion).

Scenarios	Low social cost of carbon ($50 Mg^{-1})			High social cost of carbon ($100 Mg^{-1})		
	Low social cost of N-damage (1)	Medium social cost of N-damage (2)	High social cost of N-damage (3)	Low social cost of N-damage (4)	Medium social cost of N-damage (5)	High social cost of N-damage (6)
Benchmark economic assumptions	−139	−199	−284	−99	−159	−244
Export prices of gasoline and diesel increase by 5%	−127	−185	−267	−89	−147	−230
3% annual growth rate of export demand for all food commodities	−117	−172	−250	−77	−132	−210

Note: all values are discounted cumulative numbers over 2016–2030 (discount rate 3%). The environmental costs of alternative biofuel policies by discounting the monetary value of cumulative GHG emissions, and the monetary value of cumulative N-damage costs 2016–2030, as noted in sections 2.1 and 2.2. The low, medium and high social cost of N Damage are $5.45, $10.17, $16.87 Kg^{-1} N applied, respectively. More details can be found in table S1 of SI.

the assumption of replacement N-fertilizer application with corn stover for the social cost of N-damage with cellulosic biofuels by simulating a scenario in which this replacement N application is assumed to be zero.

Additionally, for each alternative scenario, we also consider two alternative values for the social cost of carbon: $50 and $100 and three sets of alternative social costs of N-damage: low values, median values and high values, as shown in tables 4 and 5. For the purpose of comparability, we estimate the change in total social welfare (including economic costs, costs of GHG emissions and cost of N-damages) under the Corn Ethanol Mandate relative to the No-Policy scenario (table 4) and the change in total social welfare under the Corn + Cellulosic Mandate relative to Corn Ethanol Mandate (table 5) under each alternative scenario. We then compare these values to those under benchmark economic assumptions to estimate the potential benefit or loss induced by the changes in these assumptions.

We find that while the social welfare under the Corn Ethanol Mandate relative to the No-Policy scenario does change in the expected direction, the difference between the two remains large and negative, with social costs ranging from (−)$77 billion to (−)$284 billion across alternative assumptions and alternative values of the social cost of carbon and N-damage. The increase in export prices of petroleum products and growth rate of export food demand would increase the benefits of oil producers and agricultural producers, respectively, since they will increase the value of our exports and the terms of trade in favor of the U.S. However, the net change in social welfare is still negative even at a high social cost of carbon ($100 Mg^{-1}) because of the substantial welfare loss for the fuel and agricultural consumers due to the higher commodity prices.

The additional social costs of the Cellulosic Ethanol Mandate relative to the Corn Ethanol Mandate are lower (and benefits are higher) if the cost of production of cellulosic biofuels is lower or if no replacement N is applied with corn stover. More importantly, the Cellulosic Ethanol Mandate leads to smaller additional costs compared to the Corn Ethanol Mandate in all cases and it generally leads to positive net benefits if the social cost of carbon is high ($100 Mg^{-1}). Overall, the change in total social welfare of Corn + Cellulosic Mandate relative to Corn Ethanol Mandate will range between (−)$76 billion and (−)$18 billion with the low social cost of carbon, and between (−)16 billion and $43 billion with the high social cost of carbon (table 5).

We also find that there is a trade-off between the monetized value of the social cost of carbon and N-damage; a high social cost of N-damage could substantially decrease total social welfare even with a high social cost of carbon. For instance, when the social cost of carbon is $100 Mg^{-1} CO_2-eq, total social welfare with the Corn + Cellulosic Mandate relative to Corn Ethanol Mandate will decrease by 34%–228%, if the social cost of N-damage is high instead of at the median level.

4. Discussion

This paper estimates the economic and environmental costs due to the imposition of the Corn Ethanol and Corn + Cellulosic Ethanol mandate on the agricultural and transportation sectors in the U.S. over the 2016–2030 period. We apply a multi-period, multi-market, partial equilibrium,

Table 5. Change in social welfare with Corn + Cellulosic Mandate relative to Corn Ethanol Mandate over 2016–2030 period ($ billion).

	Low social cost of carbon ($50 Mg^{-1})			High social cost of carbon ($100 Mg^{-1})		
Scenarios	Low social cost of N-damage (1)	Medium social cost of N-damage (2)	High social cost of N-damage (3)	Low social cost of N-damage (4)	Medium social cost of N-damage (5)	High social cost of N-damage (6)
Benchmark economic assumptions	−29	−49	−76	34	14	−14
Export prices of gasoline and diesel increase by 5%	−29	−48	−76	32	12	−16
3% annual growth rate of export demand for all food commodities	−28	−47	−73	32	13	−13
Cellulosic biofuels conversion cost decrease by 25% and the learning rate increase by 25%	−18	−28	−43	43	32	18
No additional N is applied for crop residues	−18	−26	−37	41	33	22

Note: all values are discounted cumulative numbers over 2016–2030 (discount rate 3%). The environmental costs of alternative biofuel policies by discounting the monetary value of cumulative GHG emissions, and the monetary value of cumulative N-damage costs 2016–2030, as noted in sections 2.1 and 2.2. The low, medium and high social cost of N Damage are $5.45, $10.17, $16.87 Kg^{-1} N applied, respectively. More details can be found in table S1 of SI.

open-economy model (BEPAM) of the agricultural and transportation sectors to endogenously determine land allocation, food, and fuel, mix of cellulosic biofuels, consumption, and prices to meet the corn ethanol and cellulosic mandates. We also determine the social costs of GHG emissions and N-damages associated with the biofuel policies and add those to the economic costs to determine the total social costs (or benefits) of biofuel policies.

One of the concerns with the biofuels is their consequences for food and fuel prices. Our analysis shows that the mandate increases fuel prices as well as agricultural commodities, mainly corn price, which results in reducing consumers' benefits. We find that the corn ethanol mandate raises corn prices by 12% and soybeans prices by 7% in 2030 compared to the No-Policy scenario; the corresponding additional increase in these prices due to the Cellulosic Ethanol Mandate are much smaller (2% and 6%, respectively in 2030). Blended gasoline prices increase with the mandates are relatively small; by 2% with the Corn Ethanol Mandate and by 5% with the Cellulosic Ethanol Mandate.

While the agricultural producers gain more than $100 billion over 2016–2030, on the whole the Corn Ethanol Mandate creates a welfare loss as the reduction in consumers' benefits due to increased food and fuel prices outweigh the increase in crop producers' profits. We estimate the economic costs of the Corn Ethanol Mandate to be $109 billion over 2016–2030. It can also reduce gasoline consumption by more than 6% and diesel consumption by 2% in 2030 relative to the No-Policy scenario. The benefits due to GHG savings are however far outweighed by the costs due to additional N-leakage, resulting in an overall social cost of the Corn Ethanol Mandate of $199 billion over 2016–2030. These social costs remain large under alternative assumptions in the food and fuel sectors and values of social costs of carbon and nitrogen.

The implementation of the cellulosic mandate will also lead to higher agricultural producers' benefits, although by a smaller amount than Corn Ethanol Mandate (by $28 billion). However, it will lead to larger GHG savings (4.1%), higher GHG benefits ($63 billion) and smaller damages due to N-leakage ($42 billion) compared to the Corn Ethanol Mandate. Overall, the additional costs of the Cellulosic Ethanol Mandate are smaller than those of the Corn Ethanol Mandate. These results are sensitive to assumptions of social cost of carbon and

nitrogen and assumptions about export prices of petroleum products and cellulosic ethanol processing costs. The Cellulosic Ethanol Mandate generally leads to net social benefits when the social cost of carbon is high.

Our analysis has several policy implications. It shows the conditions under which cellulosic biofuel mandate has the potential to provide substantial economic and environmental benefits and would, therefore, warrant policy support. It also shows the benefits of policy support for research and development to lower the cost of conversion of biomass to cellulosic ethanol; this could result in social benefits that range from \$9 to \$34 billion. Similarly, policies that limit the application of replacement N with corn stover removal can reduce the N-damages and lead to positive net social benefits that range from \$8 to \$39 billion.

Our findings should also lead policymakers to question the effectiveness of technology mandates, such as the RFS, for promoting cellulosic biofuels. A key limitation of the RFS is that all cellulosic feedstocks that reduce GHG emission intensity below a 60% threshold relative to conventional fuel are treated as identical. This limits incentives for high yielding and low carbon energy crops, such as miscanthus and switchgrass, that have relatively lower carbon intensity and lower N-leakage but are more costly than corn stover. Performance-based policies, such as the low carbon fuel standard or supplementing the RFS with and carbon/N-leakage taxes, or limits on residue harvest and replacement N application can induce a shift towards more environmentally friendly cellulosic biofuels.

Our analysis relies on a simplifying assumption that the social cost of N-damage is spatially homogenous. However, this estimate of social cost of N-damage, as Sobota *et al* (2015) noted, is compiled from large-scale studies from diverse regions (national or regional); this contributes to its geographical or context similarity which is a core determinant of the validity and reliability or transferring monetary measures of environmental benefits across locations. An alternative approach, as some recent studies suggest, is to use advanced methods, such as meta-analysis and structural benefit transfer to synthesize site-specific information by estimating a benefit function and predicting the site-specific social cost of N-damage (Johnston and Thomassin 2010, Kaul *et al* 2013, Johnston *et al* 2018). However, there is mixed evidence in the literature about whether and the extent to which these advanced and flexible transfer methods would enhance validity and reliability compared to the simpler unit value transfer method used here (Navrud and Ready 2007, Czajkowski *et al* 2017). We leave it to future research to examine and compare the social cost of N-damage using different approaches. Our approach does provide a first approximation of the N-damage costs of continuing the RFS over the 2016–2030 period and enables an assessment of the trade-offs offered by the different types of biofuels that it will induce.

Our analysis does not consider the international ILUC-related emissions since modeling land use in the ROW is outside the scope of BEPAM. To provide an outer range of how large these emissions could be we use estimates from the literature for the total (domestic and international) ILUC effect and examine their implications for the GHG savings with each of the two biofuel mandates. Using the estimates of 8.7 g CO_2-eq MJ^{-1} and 20 g CO_2-eq MJ^{-1} for the total ILUC effects of corn ethanol and soybean biodiesel, respectively, as reported by Taheripour *et al* (2017), the overall ILUC-related emission in Corn Ethanol Mandate scenario relative to the No-Policy scenario would increase by 0.005 billion Mg CO_2 and the total GHG savings in this scenario would be 2.48% instead of 2.5%. We leave it to future research to provide more accurate assessments of the international ILUC-related emissions with the specific biofuel mandates analyzed here. Additionally, this paper does not examine the effects of biofuel mandate on other ecosystem services, such as, wildlife biodiversity. We leave it to future research to provide more generalizable evidence of these effects and quantify their environmental costs.

Data availability statement

Code and data to replicate all simulations results in this study are freely available at the Illinois Data Bank (https://doi.org/10.13012/B2IDB-9851670_V1).

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References

Alexandratos N and Bruinsma J 2012 World agriculture towards 2030/2050: the 2012 revision *ESA Working paper* (https://doi.org/10.22004/ag.econ.288998)

Baral N R, Sundstrom E R, Das L, Gladden J, Eudes A, Mortimer J C, Singer S W, Mukhopadhyay A and Scown C D

2019 Approaches for more efficient biological conversion of lignocellulosic feedstocks to biofuels and bioproducts *ACS Sustain. Chem. Eng.* **7** 9062–79
Bennett E M, Capenter S R and Caraco N F 2001 Human impact on erodable phosphorus and eutrophication: a global perspective *BioScience* **51** 227
Bento A M, Klotz R and Landry J R 2015 Are there carbon savings from U.S. biofuel policies? The critical importance of accounting for leakage in land and fuel markets *Energy J.* **36** 75–109
Boardman A E, Greenberg D H, Vining A R and Weimer D L 2017 *Cost-benefit Analysis: Concepts and Practice* (Cambridge: Cambridge University Press)
Chen X, Huang H, Khanna M and Önal H 2014 Alternative transportation fuel standards: welfare effects and climate benefits *J. Environ. Econ. Manage.* **67** 241–57
Chen X and Khanna M 2012 Explaining the reductions in US corn ethanol processing costs: testing competing hypotheses *Energy Policy* **44** 153–9
Chen X and Khanna M 2018 Effect of corn ethanol production on Conservation Reserve Program acres in the US *Appl. Energy* **225** 124–34
Compton J E, Harrison J A, Dennis R L, Greaver T L, Hill B H, Jordan S J, Walker H and Campbell H V 2011 Ecosystem services altered by human changes in the nitrogen cycle: a new perspective for US decision making *Ecology Lett.* **14** 804–15
Cui J, Lapan H, Moschini G and Cooper J 2011 Welfare impacts of alternative biofuel and energy policies *Am. J. Agric. Econ.* **93** 1235–56
Czajkowski M, Ahtiainen H, Artell J and Meyerhoff J 2017 Choosing a functional form for an international benefit transfer: evidence from a nine-country valuation experiment *Ecol. Econ.* **134** 104–13
Daioglou V, Doelman J C and Stehfest E 2017 Greenhouse gas emission curves for advanced biofuel supply chains *Nat. Clim. Change* **7** 920–4
Davidson E A *et al* 2012 Excess nitrogen in the US environment: trends, risks, and solutions ESA Issues in Ecology (15) (https://www.esa.org/esa/wp-content/uploads/2013/03/issuesinecology15.pdf)
Debnath D, Khanna M, Rajagopal D and Zilberman D 2019 The future of biofuels in an electrifying global transportation sector: imperative, prospects and challenges *Appl. Econ. Perspect. Policy* **41** 563–82
Degorter H and Drabik D 2011 Components of carbon leakage in the fuel market due to biofuel policies *Biofuels* **2** 119–21
Donner S D and Kucharik C J 2008 Corn-based ethanol production compromises goal of reducing nitrogen export by the Mississippi River *Proc. Natl Acad. Sci.* **105** 4513–8
EIA 2017 U.S. petroleum and other liquids supply, consumption, and inventories (available at: www.eia.gov/outlooks/steo/) (Accessed 3 April 2017)
EIA 2020 U.S. energy information administration *Petroleum and Other Liquids: Supply and Disposition* (www.eia.gov/petroleum/data.php) (Accessed 1 May 2020)
EPA 2017 The Social Cost of Carbon: Estimating the Benefits of Reducing Greenhouse Gas Emissions (Accessed 9 May 2020) (https://19january2017snapshot.epa.gov/climatechange/social-cost-carbon_.html)
Farrell A E, Plevin R J, Turner B T, Jones A D, O'Hare M and Kammen D M 2006 Ethanol can contribute to energy and environmental goals *Science* **311** 506–8
Ferin K M, Chen L, Zhong J, Acquah S, Heaton E A, Khanna M and Vanloocke A 2021 Water quality effects of economically viable land use change in the Mississippi river basin under the Renewable Fuel Standard *Environ. Sci. Technol.* (https://doi.org/10.1021/acs.est.0c04358)
Galloway J N *et al* 2004 Nitrogen cycles: past, present, and future *Biogeochemistry* **70** 153–226
Gelfand I, Sahajpal R, Zhang X, Izaurralde R C, Gross K L and Robertson G P 2013 Sustainable bioenergy production from marginal lands in the US Midwest *Nature* **493** 514–7
Greene D L 2012 Rebound 2007: Analysis of U.S. light-duty vehicle travel statistics *Energy Policy* **41** 14–28
Hertel T W, Golub A A, Jones A D, O'Hare M, Plevin R J and Kammen D M 2010 Effects of US maize ethanol on global land use and greenhouse gas emissions: estimating market-mediated responses *BioScience* **60** 223–31
Hochman G, Rajagopal D and Zilberman D 2011 The effect of biofuels on the international oil market *Appl. Econ. Perspect. Policy* **33** 402–27
Hochman G and Zilberman D 2018 Corn ethanol and U.S. biofuel policy 10 years later: a quantitative assessment *Am. J Agric. Econ.* **100** 570–84
Holland S P, Hughes J E, Knittel C R and Parker N C 2015 Unintended consequences of carbon policies: transportation fuels, land-use, emissions, and innovation *Energy J.* **36** 35–74
Huang H, Khanna M, Önal H and Chen X 2013 Stacking low carbon policies on the renewable fuels standard: economic and greenhouse gas implications *Energy Policy* **56** 5–15
Hudiburg T W, Wang W, Khanna M, Long S P, Dwivedi P, Parton W J, Hartman M and Delucia E H 2016 Impacts of a 32-billion-gallon bioenergy landscape on land and fossil fuel use in the US *Nat. Energy* **1** 1–7
Hughes J, Knittel C R and Sperling D 2008 Evidence of a shift in the short-run price elasticity of gasoline demand *EJ.* **29**
Interagency Working Group 2013 Technical update on the social cost of carbon for regulatory impact analysis-under executive order 12866 (https://www.epa.gov/sites/production/files/2016-12/documents/sc_co2_tsd_august_2016.pdf)
Jiang C, Guan K, Khanna M, Chen L and Peng J 2020 Assessing marginal land availability based on high resolution land use change information in the Contiguous United States *Under Review*
Johnston R J, Rolfe J and Zawojska E 2018 Benefit transfer of environmental and resource values: progress, prospects and challenges *Int. Rev. Environ. Resour. Econ.* **12** 177–266
Johnston R J and Thomassin P J 2010 Willingness to pay for water quality improvements in the United States and Canada: considering possibilities for international meta-analysis and benefit transfer *Agric. Resour. Econ. Rev.* **39** 114–31
Jung J H and Altpeter F 2016 TALEN mediated targeted mutagenesis of the caffeic acid O-methyltransferase in highly polyploid sugarcane improves cell wall composition for production of bioethanol *Plant Mol. Biol.* **92** 131–42
Kaul S, Boyle K J, Kuminoff N V, Parmeter C F and Pope J C 2013 What can we learn from benefit transfer errors? Evidence from 20 years of research on convergent validity *J. Environ. Econ. Manage.* **66** 90–104
Keeler B L, Gourevitch J D, Polasky S, Isbell F, Tessum C W, Hill J D and Marshall J D 2016 The social costs of nitrogen *Sci. Adv.* **2** 1600219
Khanna M and Crago C L 2012 Measuring indirect land use change with biofuels: implications for policy *Annu. Rev. Resour. Econ.* **4** 161–84
Khanna M, Rajagopal D and Zilberman D Forthcoming Lessons learnt from a decade of experience with biofuels: comparing hype with evidence *Rev. Environ. Econ. Policy*
Khanna M, Wang W, Hudiburg T W and Delucia E H 2017 The social inefficiency of regulating indirect land use change due to biofuels *Nat. Commun.* **8** 1–9
Kumar D, Long S P and Singh V 2018 Biorefinery for combined production of jet fuel and ethanol from lipid-producing sugarcane: a techno-economic evaluation *GCB Bioenergy* **10** 92–107
Leach A M, Galloway J N, Bleeker A, Erisman J W, Kohn R and Kitzes J 2012 A nitrogen footprint model to help consumers understand their role in nitrogen losses to the environment *Environ. Dev.* **1** 40–66
Liu C G, Xiao Y, Xia X X, Zhao X Q, Peng L, Srinophakun P and Bai F W 2019 Cellulosic ethanol production: progress, challenges and strategies for solutions *Biotechnol. Adv.* **37** 491–504

Lynd L R 2017 The grand challenge of cellulosic biofuels *Nat. Biotechnol.* **35** 912

Moschini G, Lapan H, Cui J and Cooper J 2010 Assessing the welfare effects of U.S. biofuel policies *AgBioForum* **13** 370–37

Navrud S and Ready R C 2007 *Environmental Value Transfer: Issues and Methods* (Berlin: Springer)

Rajagopal D 2013 The fuel market effects of biofuel policies and implications for regulations based on lifecycle emissions *Environ. Res. Lett.* **8** 24013

Rajagopal D, Plevin R, Hochman G and Zilberman D 2015 Multi-objective regulations on transportation fuels: comparing renewable fuel mandates and emission standards *Energy Economics* **49** 359–69

Ribaudo M O, Heimlich R and Peters M 2005 Nitrogen sources and Gulf hypoxia: potential for environmental credit trading *Ecological Economics* **52** 159–68

Searchinger T, Heimlich R, Houghton R A, Dong F, Elobeid A, Fabiosa J, Tokgoz S, Hayes D and Yu T 2008 Use of US croplands for biofuels increases greenhouse gases through emissions from land-use change *Science* **319** 1238–40

Secchi S, Gassman P W, Jha M, Kurkalova L and Kling C L 2011 Potential water quality changes due to corn expansion in the Upper Mississippi River Basin *Ecol. Appl.* **21** 1068–84

Sobota D J, Compton J E, Mccrackin M L and Singh S 2015 Cost of reactive nitrogen release from human activities to the environment in the United States *Environ. Res. Lett.* **10** 25006

Sun S, Ordonez B V, Webster M D, Liu J, Kucharik C J and Hertel T 2020 Fine-scale analysis of the energy–land–water nexus: nitrate leaching implications of biomass cofiring in the Midwestern United States *Environ. Sci. Technol.* **54** 2122

Taheripour F and Tyner W 2013 Induced land use emissions due to first and second generation biofuels and uncertainty in land use emission factors *Econ. Res. Int.* **12** 1

Taheripour F, Zhao X and Tyner W E 2017 The impact of considering land intensification and updated data on biofuels land use change and emissions estimates *Biotechnol. Biofuels* **10** 191

Thompson W, Whistance J and Meyer S 2011 Effects of US biofuel policies on US and world petroleum product markets with consequences for greenhouse gas emissions *Energy Policy* **39** 5509–18

Tilman D, Socolow R, Foley J A, Hill J, Larson E, Lynd L, Pacala S, Reilly J, Searchinger T, Somerville C and Williams R 2009 Beneficial biofuels–the food, energy, and environment trilemma *Science* **325** 270–1

Van Drecht G, Bouwman A F, Harrison J and Knoop J M 2009 Global nitrogen and phosphate in urban wastewater for the period 1970 to 2050 *Global Biogeochem. Cycles*. **23**

Van Grinsven H J, Holland M, Jacobsen B H, Klimont Z, Sutton M A and Jaap Willems W 2013 Costs and benefits of nitrogen for europe and implications for mitigation *Environ. Sci. Technol.* **47** 3571–9

White M J *et al* 2014 Nutrient delivery from the Mississippi River to the Gulf of Mexico and effects of cropland conservation *J. Soil Water Conserv.* **69** 26–40

Witcover J, Yeh S and Sperling D 2013 Policy options to address global land use change from biofuels *Energy Policy* **56** 63–74