Review of GHG Emissions of Corn Ethanol under the EPA RFS2

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DISCLAIMER

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Contents

TABLES

FIGURES

TERMS AND ABBREVIATIONS

aLCA	Attributional Life Cycle Analysis
ANL	Argonne National Laboratory
ARB	California Air Resources Board
CA	California
CA-GREET	The standard GREET model modified for use in CA LCFS
CH_4	Methane
CI	Carbon intensity
cLCA	Consequential Life Cycle Analysis
CO	Carbon monoxide
CO_2	Carbon dioxide
DOE	U.S. Department of Energy
EPA	U.S. Environmental Protection Agency
g CO_2e	Grams of carbon dioxide equivalent
GHG	Greenhouse Gas
GREET	The Greenhouse gas, Regulated Emissions, and Energy use in Transportation model
GWP	Global Warming Potential
HC	Hydrocarbon
IPCC	Intergovernmental Panel on Climate Change
LCA	Life Cycle Analysis or Life Cycle Assessment
LCFS	Low Carbon Fuel Standard
LCI	Life Cycle Inventory
LCFS	Low Carbon Fuel Standard
LHV	Lower Heating Value
N_2O	Nitrous oxide
NG	Natural Gas
RIA	Regulatory Impact Analysis
RFS2	Revised Federal Renewable Fuels Standard
UN	United Nations
UNFCC	United Nations Framework Convention on Climate Change
VOC	Volatile Organic Compound
WTT	Well-To-Tank
WTW	Well-To-Wheel

Executive Summary

Twelve years of experience and improved analysis methods have provided new insight into the life cycle greenhouse gas (GHG) emissions from corn ethanol. This study reviews the key factors that affect the life cycle emissions from corn ethanol production as well as the most recent agricultural data. Some of the key factors affecting corn ethanol have evolved as predicted in EPA's 2010 Regulatory Impact Analysis (2010 RIA), while other factors point towards substantially lower life cycle GHG emissions.

EPA developed a consequential LCA approach that estimated the emissions associated with the incremental ethanol capacity induced by the RFS policy as well as the incremental crop production required to make up for the net effect of corn crops diverted to ethanol production and distiller's grains sold as animal feed. The modeling approach involved a combination of the FASOM model that has been used to develop the U.S. inventory for agricultural emissions, the FAPRI model, which estimates the effect of the use of agricultural products on global agricultural production, and the GREET model, which estimates life cycle GHG emissions from the fuel used in ethanol plants. EPA's analysis aligned the economic modeling of the FASOM and FAPRI modeling and calculated emission impacts that are tied to the model predictions including changes in rice and beef consumption as well as deforestation associated with new crop production.

The 2010 RIA overestimated the GHG impact of corn ethanol due largely to overestimating indirect land use conversion (ILUC) emissions as well as numerous small details associated with the life cycle of corn ethanol. EPA's agro-economic models rely on economic projections to attribute land use change to crop production without considering factors such as changes in farming and cattle production practices. Recent data on deforestation has shown that land ownership is much more important in affecting deforesting than the macro-economic pressure or crop prices. Burning in the Amazon has declined and increased due to policies associated with land ownership. A more accurate representation of the effect of crops on pasture conversion is represented in more recent publications based on the GTAP model and EPA would generate similar results if its ILUC modeling tools included an accurate representation of factors such as flexibility in changing cattle stocking rates. The analysis inputs to GTAP modeling would yield similar results in the FASOM/FAPRI modeling system. If EPA continues to use the FAPRI results for its international LUC analysis, the results could be scaled to reflect the values from GTAP that more accurately represent the interaction between pasture and cropland.

Several other factors affecting corn ethanol have also changed since the publication or the 2010 RIA. Corn ethanol uses about 0.7 kWh to produce one gallon of ethanol and the GHG intensity of electric power has declined substantially with increased natural gas production, a reduction in coal-based power, and growth in renewable power. The RIA also underestimated the adoption of low emission technologies that have resulted in lower emissions from ethanol plants and many small details associated with each step of the ethanol life cycle.

More significantly, EPA underestimated the effect of distiller's grains and corn oil. Much of the corn used for ethanol production has resulted in the displacement of soybean production. The same acre of land that was producing soybeans and converted to corn for ethanol produces the same amount of feed via the distiller's grains from the ethanol plant. Therefore, any change in net feed requirements is subtle at best. The GREET model also underestimates the displacement effect of both soybeans and urea that would otherwise be fed to cattle. Even though soybeans fix nitrogen[1], USDA data shows that they have required more nitrogen fertilizer than projected in the RIA. Also, the emissions associated with urea feed in the GREET model omit the displacement of fossil carbon[2] in urea. Corn ethanol plants have produced significant quantities of corn oil as predicted in the 2010 RIA. However, about half of the corn oil is used as biodiesel which corresponds to about 2.5% of the energy output of an ethanol plant. The GHG emissions associated with corn production and any ILUC should be partially assigned to biodiesel.

These factors should be incorporated in EPA's GHG analysis of corn ethanol in this 2020, 2021, 2022 Renewable Volume Obligation (RVO) rulemaking, including the following considerations:

- ILUC and soil carbon storage should reflect the latest research.
    - ANL soil carbon storage modeling (CCLUB) shows increased soil carbon storage with corn farming that was not taken into account in the 2010 RIA.
    - New analysis based on the GTAP shows the effect of pasture intensification which predicts lower rates of forest conversion to agriculture.
    - CARB revised ILUC for LCFS from 30 g CO_2e/MJ to 19.8 g CO_2e/MJ with the newest GTAP results showing 7.5 g CO_2e/MJ.
- The FASOM and FAPRI modeling system predict effects that are not tied to ethanol use and should be corrected.
    - The latest data and science demonstrate that deforestation rates occur due to many factors and the supply and demand of agricultural products has little effect on this phenomenon.
- Co-product credit value of distillers' grain solubles (DGS) is higher than anticipated due to:
    - Greater emissions from the displacement of soybean meal;
    - Higher nitrogen (N) application rate on soybeans than originally anticipated;
    - Displacement of fossil CO_2 in urea feed.
- A high adoption rate of corn oil extraction has led to the rapid growth in use of corn oil as biodiesel feedstock.
    - The preferred use of corn oil is biodiesel; so, the appropriate co-product treatment for 50% of the corn oil is as an energy product via allocation.

[1] Soybeans and other legumes assimilate nitrogen from the atmosphere into organic compounds through a process known as fixation.

[2] The GHG intensity of urea in the GREET model represents the life cycle emissions per ton of urea. The urea molecule includes carbon that is derived from natural gas. When urea is used as fertilizer or animal feed, the carbon is metabolized to produce CO_2. GREET counts the field emissions for urea when used as fertilizer but omits the emissions when it is used as a co-produce animal feed.

    - Corn oil when used as a biodiesel feed displaces fats such as soy oil and palm oil which have much higher indirect land use change (ILUC) values than corn oil when treated as DGS mass.
- Ethanol plants produce lower GHG emissions than estimated in the 2010 RIA due to:
    - Elimination of coal for dry mill plants with natural gas;
    - Lower carbon intensity for electric power used by ethanol plants;
    - Use of biogas motivated by California low carbon fuel standard (LCFS) program;
    - Ongoing efficiency improvements from many sources;
    - Utilization of CO_2 to displace fossil sources and CO_2 sequestration.
- 2005 Petroleum baseline in the 2010 RIA is underestimated because the baseline fails to adequately account for:
    - Higher rates of methane venting and flaring from oil production;
    - Mix of secondary oil recovery technologies and oil sands.

This study found that corn ethanol has resulted in greater GHG emission reductions compared to those originally predicted in the 2010 RIA. The results for dry mill corn ethanol plants from this Study (Figure S.1) are aligned with the approach in the 2010 RIA. The emissions are based on GREET calculations and adjustments to reflect EPA's categories with projections for energy use in 2022 developed in this study. The emissions include allocation of half of the GHG emissions associated with corn oil to biodiesel. Higher nitrogen application rates for soybean farming, which affect the DGS co-product credit as well as fossil carbon displaced in urea feed are also considered in the analysis. The lower carbon intensity of electric power compared to 2010 projections is reflected in fuel production emissions. The small effect on rice methane and livestock emissions is based on the recent study funded by the U.S. Department of Agriculture by ICF (Rosenfeld, 2018). These results compared with appropriate adjustments to EPA's 2005 baseline translate into about a 48% reduction in GHG emissions as shown in Figure S.1.

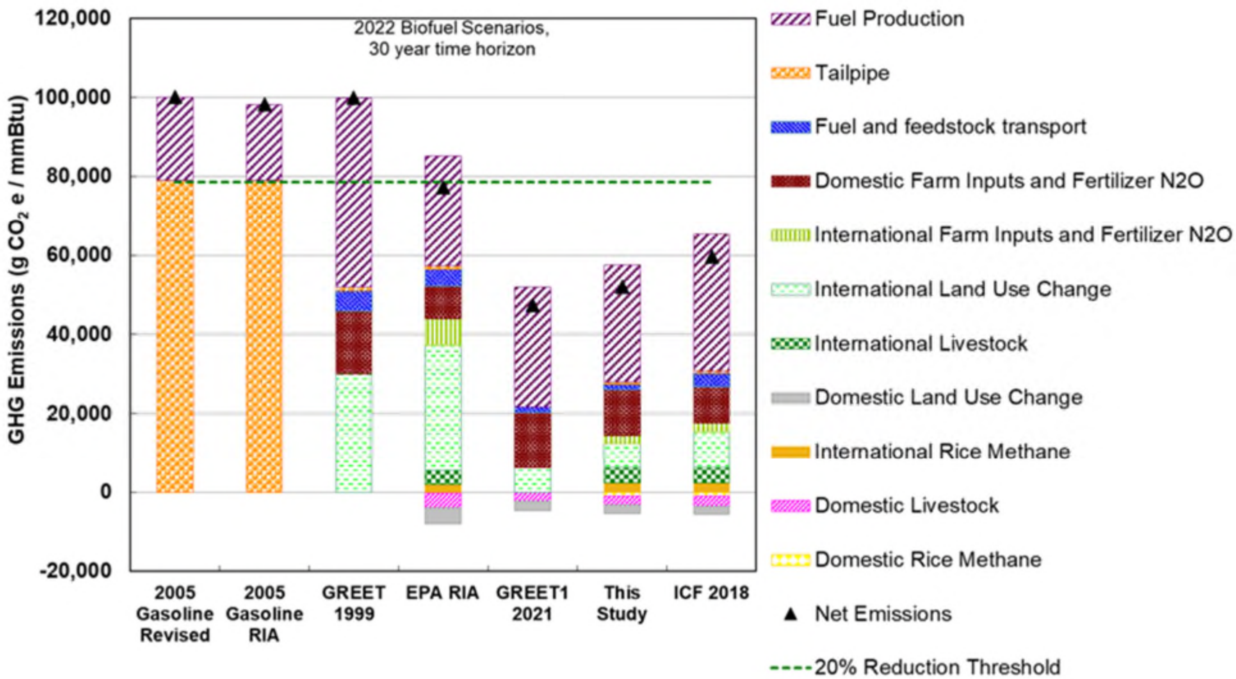

Figure S.1. Life Cycle GHG Emissions from Dry Mill Corn Ethanol and 2005 Petroleum Gasoline.

1. INTRODUCTION

As part of EPA's 2010 Regulatory Impact Analysis (2010 RIA) of the Renewable Fuel Standard (RFS), it conducted a life cycle assessment (LCA) of the biofuels specified in RFS2 by accounting for direct and indirect emissions for the year 2022. The 2010 RIA identified 11 emission sources which capture the full life cycle GHG profile of corn ethanol and compared these emissions with those of gasoline (Figure 1.1). The highest GHG emissions for corn ethanol correspond to international land use change (LUC) followed by Fuel Production. International LUC corresponds to the change in carbon associated with the growth of new crops outside the U.S. EPA estimated that these emissions include the release of soil carbon and avoided carbon storage from forest and pastureland when these lands are converted to cropland. The landcover change is predicted with the FAPRI model and is combined with carbon stock factors developed by Winrock International. Fuel production emissions include the emissions associated with natural gas combustion as well as upstream natural gas and electric power. International farm inputs and N_2O correspond the crop farming activity required to make up for changes in U.S. farm exports. The modeling system estimated the effect of expansion in corn production.

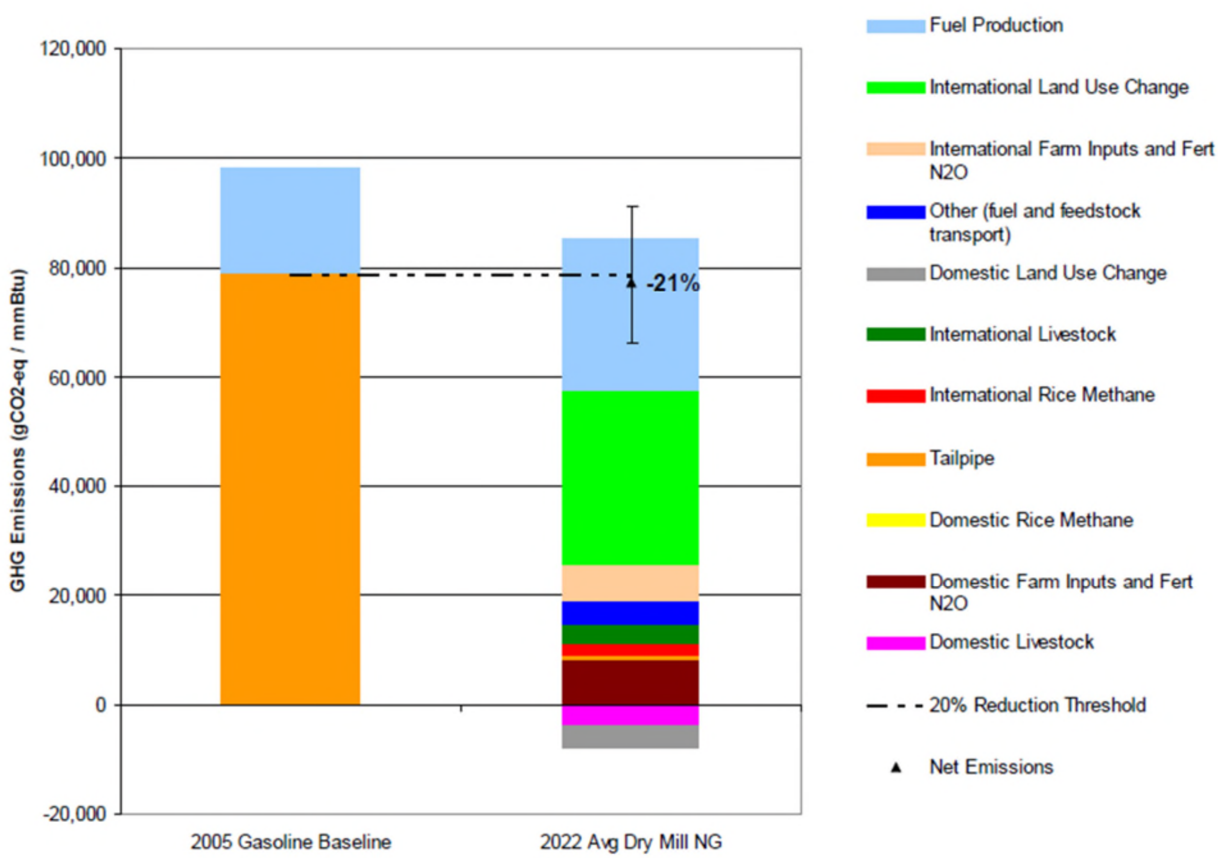

Figure 1.1. EPA's Analysis of Corn Ethanol GHG emissions. (EPA, 2010)

The objective of this study is to evaluate EPA's analysis based on the availability of new data and a better understanding of models and assumptions. This study focuses on emission categories with the highest impacts such as international land use change and compares the results of 2010 RIA with the new findings. Another key effect examined in the study is the impact of co-product credits and different methods of allocation. The study includes the following sections.

- Sections 1.1 to 1.4 provides an introduction to corn ethanol life cycle GHG emissions.
- Section 2 presents domestic and international land use change and their impacts on corn ethanol carbon intensity.
- Section 3 discusses farming inputs and the sensitivity analysis.
- Section 4 presents the impact of different co-products and their allocation factors on corn ethanol carbon intensity.
- Section 5 describes technologies used in ethanol production and their advancements.
- Section 6 analyzes the energy sources used in the fuel production stage.
- Section 7 describes the GHG emissions related to various types of extraction of fossil fuels and their projection.
- Section 8 presents the results of this study and compares them with those of other studies and EPA RIA.
- Finally, Section 9 summarizes this Study's conclusions.

1.1 Life Cycle GHG Analysis

The RFS2 and other biofuel policies around the world require GHG reduction targets relative to the conventional fossil fuels. The GHG reduction is measured through life cycle assessments (LCAs), which account for cradle-to-grave emissions (and/or other environmental impacts), starting with raw material extraction and ending with fuel consumption. LCA is a technique used to model the environmental impacts associated with the production of materials. LCA models assess environmental impacts over a range of categories, including energy consumption, GHG emissions, criteria air pollution, eutrophication, acidification, water use, land use, and others. The analysis includes a full inventory of all the inputs and outputs involved in a product's life cycle. Determining life cycle emissions for all inputs requires an iterative analysis of these components because some components of the life cycle of fuels depend on inputs that are part of the LCA. The net GHG emissions are converted to a CO_2-equivalent basis and then normalized by the energy content of the fuel (e.g. g CO_2e/MMBtu). This carbon intensity (CI), when compared with the CI of petroleum fuels, provides a measure of the net GHG reductions of renewable fuels.

In the case of corn ethanol, with the U.S. the largest producer of corn in the world, the hypothesis is that diverting corn to biofuel feedstock reduces the supply of corn in food and feed markets. This effect is realized through increases in the price of corn and other agricultural commodities globally. In order to address the shift in corn supply, farmers across the globe switch from other crops to corn (direct land use change) or convert grasslands, wetlands, or forests which are carbon sinks to crop production. The conversion of land to cropland results in indirect land use change (ILUC) emissions due to the release of carbon in the soil, above ground biomass, and the foregone sequestration of CO_2. Two different approaches covering the extent of life cycle impacts are referred to as attributional and consequential LCA. Attributional LCA (aLCA) focuses on the direct processes used to produce and consume a product while consequential LCA (cLCA) examines the consequences of possible (future) changes between alternative product systems (Brander et al., 2009). An aLCA identifies the direct energy inputs and emissions associated with corn farming and ethanol production. A cLCA identifies the net change in global emissions due to induced impacts of corn consumption, energy inputs for ethanol plants, and ethanol use. The 2010 RIA is aimed at calculating cLCA emissions based on the displacement effect of corn diverted to ethanol production.

1.2 Land Use Change

The correlation between LUC and an expansion in biofuel is typically estimated with agro-economic models. Economic models that simulate market behavior (particularly those in the agricultural sector) are often linked to predict the location of land cover change and the emissions associated with conversion to crops as illustrated in Figure 1.2

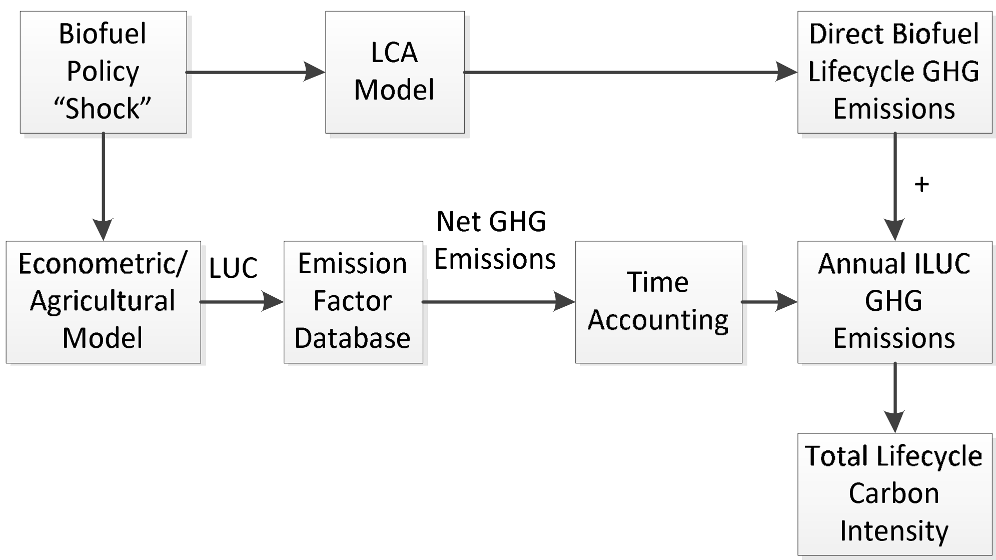

Figure 1.2. Modeling Flow for Determination of Total Biofuel Lifecycle Carbon Intensity, Including Both Direct and Indirect Effects.

1.3 Modeling Approaches

The system boundary defines the scope of activities and emissions associated with a life cycle analysis. The inputs to the system and emission flows are counted in the analysis are defined in a system boundary diagram (SBD). The system boundary identifies how far emissions are tracked and the treatment of co-products.

1.3.1 Approach for Revised GHG Analysis

This study combines new data on corn ethanol production with the methods used by EPA in the RIA to develop a revised estimate of the GHG emissions associated with corn ethanol. Repeating the details of the modeling in the RIA is not practical due to the complexity of the FASOM and FAPRI modeling systems. This study estimated the emission categories within the 2010 RIA methodology based on energy inputs and co-product yields thereby allowing for a comparison with the 2010 RIA results.

The system boundary used in this study is shown in Figure 1.3. Ethanol and corn oil for biodiesel are fuel products. Corn oil is also used as animal feed as modeled in the 2010 RIA but current fuel policies favor the use of corn oil as a biodiesel feedstock. Fermentation CO_2 is another coproduct for many ethanol plants. This study compares data on corn production, ethanol inputs and ethanol plant yields with those in the 2010 RIA and then estimates emissions for each of the RIA categories based on the best available data. The effect of each of the co-products on the net life cycle emissions is examined here.

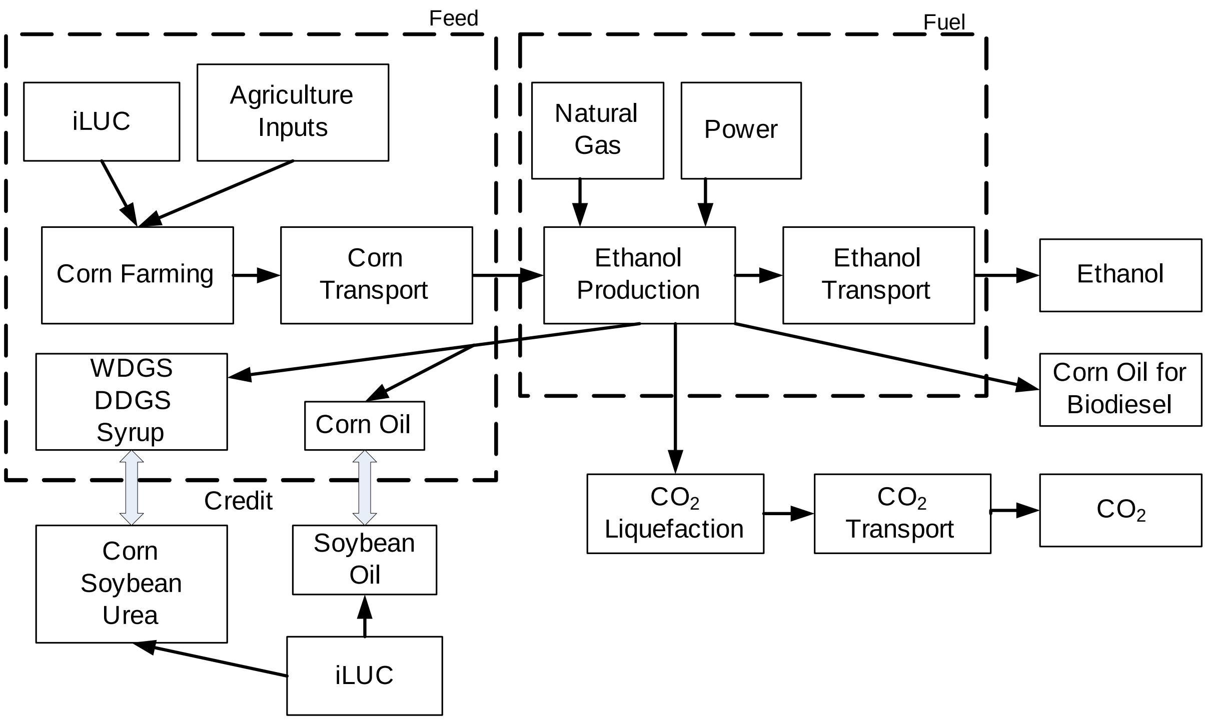

Figure 1.3. System Boundary Diagram for Corn Ethanol Production.

1.4 Global Warming Potential

The global warming potential (GWP) represents GHG emissions based on their radiative forcing and lifetime in the atmosphere on equivalent units of carbon dioxide (CO_2). These factors are estimated by the Intergovernmental Panel on Climate Change (IPCC) and updated in each IPCC Assessment Report (AR). The 2010 RIA used the factors provided by the IPCC's Second

Assessment Report (SAR), however, these factors have been updated since 2010 and the most recent one is the Fifth Assessment Report (AR5) shown in Table 1.1 (IPCC, 2014). This study uses the AR4 factors to calculate the CI of fuels since these values are currently adopted by the EPA for calculations of the national GHG inventory.

Table 1.1. Global Warming Potential (100-year time horizon).

Greenhouse Gas	SAR	AR4	AR5
CO_2	1	1	1
CH_4	21	25	28
N_2O	310	298	265

2. DOMESTIC AND INTERNATIONAL LAND USE CHANGE

Since 2010 when EPA conducted the RIA, new findings and data on actual deforestation across the globe, crop prices, soil organic carbon stocks, corn and ethanol yields have shown that the 2010 RIA overestimated the contribution of LUC towards the CI of corn ethanol. The 2010 RIA's approach, as well as new studies on LUC, are discussed below. EPA's approach to ILUC modeling, improved ILUC estimates, and the estimates used in this study are discussed.

2.1 EPA RIA Approach for Land Use Change

The 2010 RIA takes into account the incremental change of diverting corn crops to biofuel production. The modeling attempts to answer the question: what would change if U.S. ethanol use increased to 15 billion-gallon per year[3] while holding constant the consumption of food. Both the incremental farming inputs as well as the incremental effects of land conversion on crops were estimated through macroeconomic modeling.

2.1.1 EPA Modeling Approach

The system boundary used in 2010 RIA is shown in Figure 2.1. The analysis includes the direct emissions associated with tailpipe emissions, fuel production, fuel and feedstock transport. The carbon in fuel is treated on a carbon neutral basis with zero emissions associated with the short cycle carbon in ethanol and ethanol plant fermentation emissions. The effects of the corn feedstock are analyzed in a cLCA with estimates of the effects of an incremental increase in the use of ethanol and consumption of corn. The modeling takes into account the direct farming emissions in the U.S. and internationally as well as the effect on rice and livestock methane emissions due to shifts in the production of agricultural products. The U.S. emissions are predicted with the FASOM model and the international crop production is predicted with the FAPRI model combined with emission factors for land cover change and agricultural inputs.

The 2010 RIA also includes the indirect farming emissions associated with new crops in addition to LUC. This method is intended to represent the replacement crop inputs as well as land use conversion.

[3] EPA 2010 RIA, Section 1.1.1.1

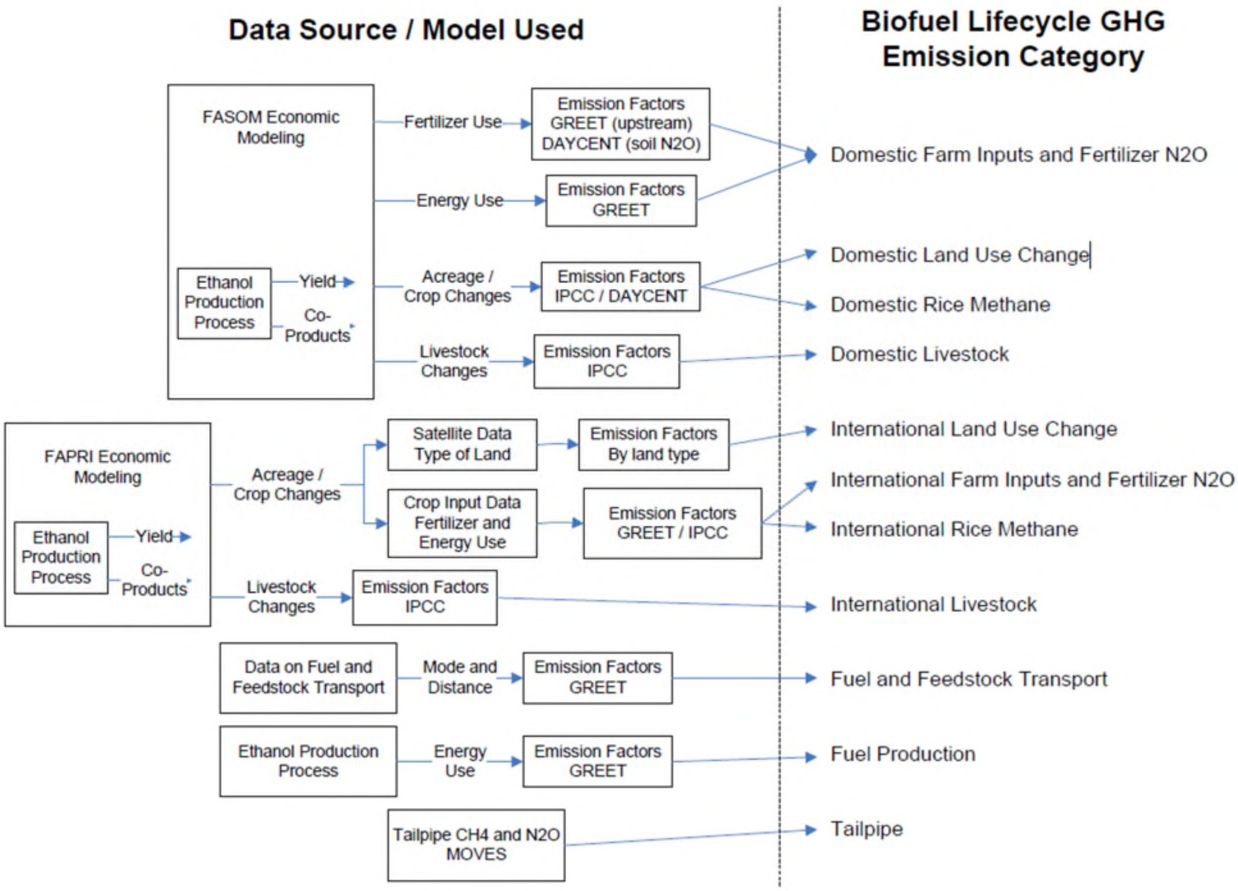

Figure 2.1. System Boundary Used in EPA RIA Study. (EPA, 2010)

For the purposes of discussion in this study direct and indirect land use change are described separately.[4] Direct land use change refers to land already used for a specific purpose (e.g. growing food) and whose future use will achieve the same result. For instance, in response to an increase in production of corn ethanol, lands previously used for food production might be converted to corn for fuel. On the other hand, indirect land use change refers to the land whose ultimate purpose is essentially changed from its previous use (Farm Energy, 2019). For instance, converting forests or grasslands to agricultural land is called indirect land use change. The 2010 RIA aggregated the impacts of direct and indirect land use change in the U.S. and called it "domestic land use change." Also, the RIA assumed international land use change occurred as a result of domestic biofuel production expansion.

EPA used the Forestry and Agricultural Sector Optimization Model (FASOM), developed by Texas A&M University and others, to estimate the changes in crop acres resulting from increased biofuel production. FASOM is a partial equilibrium model of the forest, agriculture,

[4] Some argue that all LUC is indirect since corn used for biofuel production is diverted from the overall U.S. corn supply.

and livestock for the United States. The model tracks U.S. cropland by county and estimates emissions associated with the conversion to cropland (i.e. domestic land use change). Within the model, the linked agricultural and forestry sectors compete for a portion of the land within the U.S. Prices for agricultural and forest sector commodities as well as land are endogenously determined given demand functions and supply processes. The FASOM model maximizes the net present value of the sum of consumers' and producers' surpluses (for each sector) with producers' surplus estimated as the net returns from forest and agricultural sector activities. The GHG calculations are based on available data on inputs from crop budgets coupled with estimates from EPA, the IPCC, and the DAYCENT model developed by Colorado State University. The FASOM model also estimates the energy consumption, as well as fertilizer use, of crop production. The projection of farm inputs by FASOM was used in 2010 RIA to calculate the GHG emissions of corn ethanol in 2022. The model takes into account shifts among agricultural production including changes in livestock population due to changes in corn prices. The population provides the basis for estimating livestock methane emissions.

Since FASOM is only applicable for modeling the land use change within the U.S. (domestic LUC), EPA employed the integrated Food and Agricultural Policy and Research Institute international models, as maintained by the Center for Agricultural and Rural Development (FAPRI-CARD) at Iowa State University (as summarized in CRC, 2014), to estimate the changes in crop acres and livestock production by type and by country globally (international LUC) in the 2010 RIA. While FAPRI-CARD models how much cropland will change, it does not predict what type of lands such as forest or pasture will be converted. Therefore, EPA used Winrock International's data to estimate what land types are converted into cropland in each country (EPA, 2010). EPA also used the GTAP model and confirmed that the GTAP results were consistent with outputs of FASOM and FAPRI models. Since then, the GTAP model has undergone several revisions, but EPA has not compared its findings with the new results from the GTAP model.

FASOM also predicted that cultivation of corn increases the soil carbon storage while conversion of cropland pasture and forestland leads to more GHG emissions. Overall, the FASOM results showed that expanding corn cultivation resulted in carbon storage (negative value for domestic LUC). However, the results from FAPRI showed that production of 15 billion gallons of corn ethanol reduced the corn export from the U.S. which causes other countries to allocate more lands to corn cultivation and subsequently convert more pasture and forestland to corn farms which leads to more GHG emissions. Conversion of Brazilian forests to corn farming had the highest share from total emissions associated with international LUC under the methodology used in the 2010 RIA.

2.1.2 Challenges with 2010 RIA Land Use Change Analysis

While the direct emissions from ethanol production vary among the studies, the table below shows the large variability in estimates which are largely due to LUC. Early studies employed worldwide agricultural models to estimate emissions from land use change (Searchinger et al.,2008; Searchinger, et al., 2015; Fargione et al., 2008) with higher net GHG emissions for corn ethanol compared to gasoline.

More recent studies, (Hertel et al., 2010) found that the emissions associated with land use change were less than one-third of those projected by Searchinger (2008) and even smaller values of land use change effect were reported by Tyner et al. (2010). The inconsistency in indirect land use change predictions is mainly due to the differences in methods and assumptions. Key factors include elasticity factors that affect the selection of land cover change and carbon stocks. Further, some argue the modeled predictions of indirect land use change are not meaningful because there is not a causal relationship between biofuel use and land conversion (Zilberman et al., 2010). In the 2010 RIA, conversion of Brazilian forestland to corn farm had a significant contribution to the international LUC. However, new studies found that agricultural intensification and governmental policies and regulations have had a great impact on GHG emissions reduction as well as decreasing the deforestation in Brazil (Silva et al., 2018; Garrett et al., 2018). Brazil, for example, is seeking to reduce greenhouse gas (GHG) emissions by 37% below 2005 levels by 2025 and 43% by 2030 through its announced Nationally Determined Contribution (NDC). The role of agricultural intensification in response to increasing commodity prices was not fully considered in the 2010 RIA and therefore international LUC was over-estimated (Rosenfeld et al., 2018).

Table 2.1. Addressing Uncertainties in LUC Assessments

LCA parameter	Uncertainty	Recommendation
New LUC studies estimate lower emissions associated with international LUC.	LUC estimates vary greatly with model, structure, assumptions and target year.	While it is true that LUC modeling is greatly based on assumptions and model structure, we believe that after 10 years, with the availability of new data, we can see that most of those assumptions were not realistic. The current rate of deforestations, yield price elasticity, type of land being converted, etc. are not close to what EPA projected in 2010.
Soil C sequestration of corn is higher than what assumed in 2010 RIA.	The SOC data resulted from recent studies are inconclusive due to variation between studies and dependence on experiment duration.	Recent long-term studies on SOC in Midwest such as Poffenbarger et al. (2017) shows that corn farming results in a significant increase in SOC storage. Various practices such as no-tillage and optimum fertilization increases the SOC storage and more farmers are applying these practices now.

2.2 New Findings on Land Use Change

The emissions associated with LUC include the net accumulation of carbon, taking into account both the carbon release from land conversion and the foregone carbon sequestration. Figure 2.2 shows a simplified breakdown of the factors that affect the LUC presented by the CARB and modeled in GTAP. The significant differences between the GTAP modeling and the FASOM/FAPRI modeling include the carbon stock factors for released carbon as well as the regional detail for crop shifting. GTAP, for example, takes into account prior trade history between countries. All agro-economic models solve for prices that result in a supply and demand equilibrium. GTAP is a general equilibrium model that includes all sectors of the economy. FASOM and FAPRI are models including only agriculture and, in the case of FASOM, forestry. Those models are more detailed on individual agricultural commodities. All of the models project changes in land cover and predict changes in carbon stock through different carbon accounting mechanisms and carbon stock data sets. All of the modeling systems need to allocate emissions over time as they are predicting an initial "shock" of biofuel demand that is distributed over a period of biofuel production.

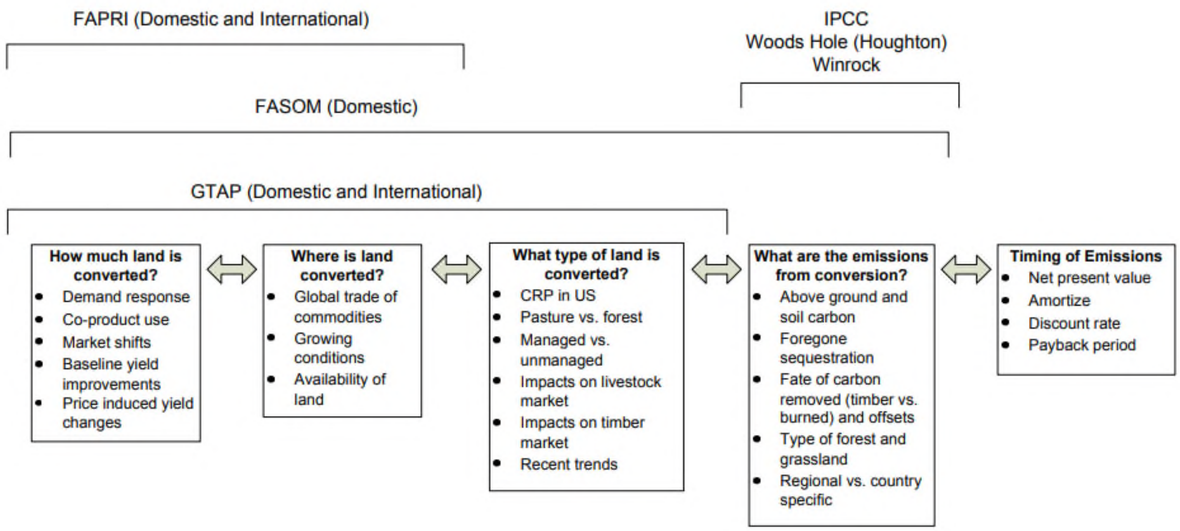

Figure 2.2. Approaches to LUC Modeling.
(CARB, 2018)

While the modeling represents the inputs to the GTAP system, the basic principles are the same for all LUC models. Improving crop yields, production of co-products, and high carbon stocks for converted lands reduce LUC emissions. The recent key findings for corn ethanol affecting LUC with GTAP have been:

- Low conversion of land in the U.S.;
- Increase in soil carbon storage due to corn farming practices;
- Overall decline in deforestation rates globally;
- High substitute value of Distillers' grain solubles (DGS) as feed;
- Increased cattle stock rate with pasture intensification;
- Corn oil producing biodiesel increases overall fuel output.

Since an acre of land producing corn for ethanol produces as much animal feed (i.e. DGS) as an acre of soybeans (soybean meal), the net LUC emissions in recent studies by ANL (Dunn, 2017), which are below 10 g CO_2e/MJ appear reasonable.

2.2.1 CCLUB and GTAP

LUC models also predict changing yields, both to the biofuel crop being examined as well as other crops grown globally. These yield improvements include both projected future improvements due to better farming practices (some of which may have nothing to do with an expansion in biofuels), as well as yield improvements that are due to higher prices sending a signal to the market to incentivize better farming practices, more efficient harvest, and technology improvements. Expanded use of crops for biofuels will also affect feed prices and shift the use of agricultural commodities. The production of DGS from corn affects feed markets. The removal of land from feed production will also result in market shifts due to price mediation. Higher corn prices, for example, could result in a shift from feedlot-fed cattle to other sources of meat that are less feed intensive. The effect of displacement by DGS as well as shifts in crop usage may be the most significant factor. Demand mediation or a reduction in the demand for feed and food also reduces the overall requirement for land. Another key LUC prediction is associated with cattle stocking rates on pasture as well as the selection of forest land, marginal land or grassland. These predictions affect the carbon stock factor for LUC.

2.2.2 Other Corn Ethanol Studies

Two studies conducted by ICF for the U.S. Department of Agriculture (USDA) examined the 2010 RIA. Each study calculated the CI of corn ethanol under different scenarios (Flugge et al., 2017; Rosenfeld et al., 2018). The studies investigated domestic and international land use change based on recent studies and models and concluded that both domestic and international land-use change emissions for corn ethanol are lower than those in the 2010 RIA. Moreover, their estimates of GHG emissions of fuel production stage as well as tailpipe were also lower than those in the RIA.

CARB has revised its estimation of international LUC (CARB, 2015) due mainly to using a newer version of GTAP with an updated database, re-estimating energy sector demand and supply elasticity values, the addition of cropland pasture to the U.S. and Brazil, improved treatment of corn ethanol co-product (DGS), improved treatment of soy meal, soy oil, and soy biodiesel, improved estimation of crop yield across the world, improved estimation of emissions factors, and revision of demand and yield responses to price, among other things. The reduction in estimated forest conversion is an important factor since the GHG emissions associated with conversion of forest is significant.

Argonne National Laboratory (ANL) and California Air Resource Board (CARB) developed GREET and CA-GREET models, respectively, which include the LCA for corn ethanol. CARB's estimates of ILUC have dropped from 30 g CO_2e/MJ to 19.8 g CO_2e/MJ based on refinements in modeling (Tyner, 2010) and the changing CI of ethanol in Table 2.2 reflects both the ILUC and mix of fuel production technologies. CARB's original modeling with GTAP assumed a 1:1 displacement of DGS with corn, but that has since been revised. Subsequent modeling has also taken into account the displacement of other agricultural products.

Table 2.2. Life Cycle Studies Examining Corn Ethanol.

Year	Study	Model/ Database	ILUC CI (gCO_2e/MJ)
2008	Searchinger et al. (2008)	FAPRI-CARD/GREET	100
2009	CARB	CA-GREET.8b/GTAP	30
2010	EPA RIA	GREET/FASOM/FAPRI	28
2018	ANL	CCLUB/GTAP/GREET	3.9 to 7.5
2017	Flugge et al. (2017)	FASOM/ FAPRI	8 to 14
2018	Rosenfeld et al. (2018)	GREET/IPCC/GTAP	7 to 14
2014	CARB[a]	CA-GREET2/GTAP	19.8
2021	Scully (2021)	Review of Models	3.9

[a] Average of approved pathways.

These models however, look backward at prior data crop expansion, yield, and land use data. Ten years of increased biofuel production in the United States allows for a revised assessment of the assumptions and results of the 2010 RIA.

2.2.3 Empirical Data

Showing the effects of LUC is challenging since the effect occurs even absent biofuel production. No experiment can prove the "counterfactual" effect of land use change absent biofuel production. However, significant empirical data suggests that the relationship between crops used for biofuel production and land use change may not be as significant as predicted in the 2010 RIA. Deforestation rates have declined in the past decade and farming practices continue to store carbon in the soil. In fact, the drivers for deforestation are not directly related to crop production (Zilberman, 2017).

The international LUC effect related to the conversion of Brazil's Amazon region was significant in the 2010 RIA, however, this anticipated relationship was not borne out in reality. When comparing the deforestation in Brazil and corn ethanol production in the U.S. from 2004 to 2015, we can see that not only did U.S. corn ethanol production *not* cause an increase in deforestation in Brazil but annual deforestation rates in Brazil's Amazon region actually *decreased* over 75 percent over that decade (Figure 2.3). These trends in forestry loss are decoupled from biofuel use and this lack of correlation is not, but should be, incorporated into EPA's analysis.

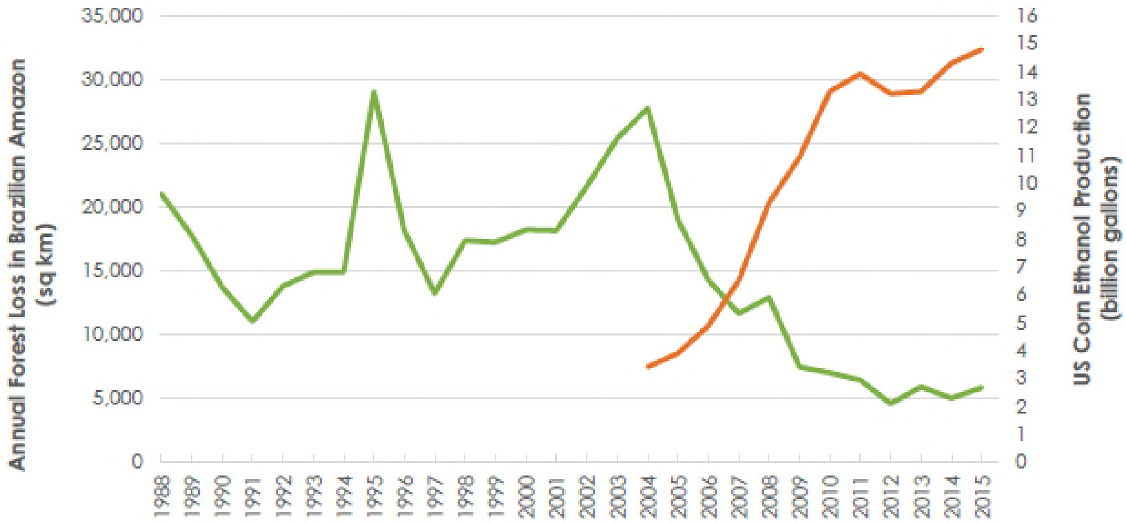

Figure 2.3. Comparison of Brazilian Deforestation and U.S. Corn Ethanol Production. (Rosenfeld et al., 2018)

Moreover, several studies have shown that corn crops produce large amounts of high carbon root and residue and this has a major positive impact on soil carbon stocks (ACE, 2018). Figure 2.5 implies that the organic matter content of the soil has improved over time due to corn farming. Part of domestic LUC is the carbon stock change due to crop cultivation and based on Figure 2.5, the carbon stock due to corn cultivation is improving which leads to more GHG emissions saving and lower impact of domestic LUC. Clay et al. (2012) studied the impact of corn yield on soil carbon sequestration and reported that in many regions, surface soils are carbon sinks when seeded with corn.

The issue of soil carbon storage is illustrated in comments in the literature regarding LUC modeling. The authors of critiques of CCLUB, which represents the newest ILUC analysis from GTAP, (Malins, 2020) argue that the Winrock data for domestic crop conversion is more accurate (which is an option to utilize in GTAP). This is not a defensible position. Much of the debate around LUC estimates as presented in GTAP pertains to the use of emission factors associated with soil carbon release. CCLUB uses the CENTURY emission factors as defaults with Winrock data used by default for international emissions. Figure 2.4 shows the comparison of different emission factors, which support the argument that the higher Winrock emission factors for domestic ILUC would be an appropriate estimate; however, this argument is inconsistent with EPA's GHG accounting as used in the U.S. GHG inventory, which uses FASOM. Shifting to greater corn production from other crops along with the deployment of low carbon farming practices stores carbon, as reflected in FASOM and CCLUB. Accordingly, criticisms of the more recent versions of GTAP are misplaced; the LUC emissions in the U.S. should be negative as shown in the 2010 RIA (which utilizes FASOM) and in CCLUB.

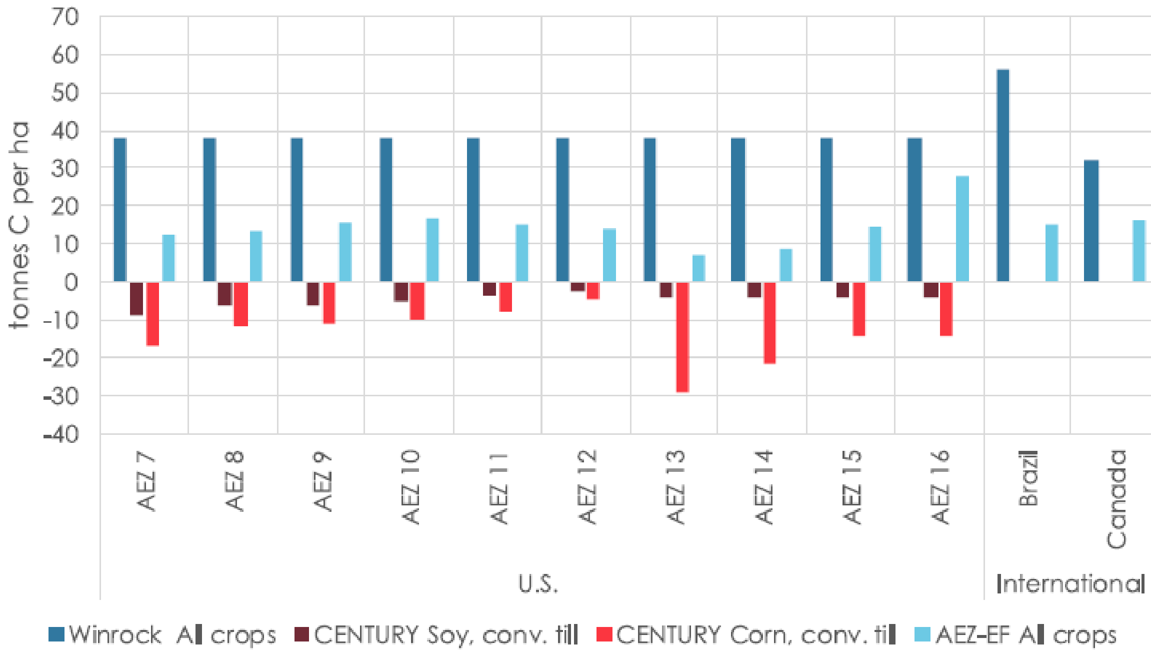

Figure 2.4. Carbon loss following cropland pasture conversion using Winrock, CENTURY and AEZ-EF emission factor models. (Malins, et al., 2020).

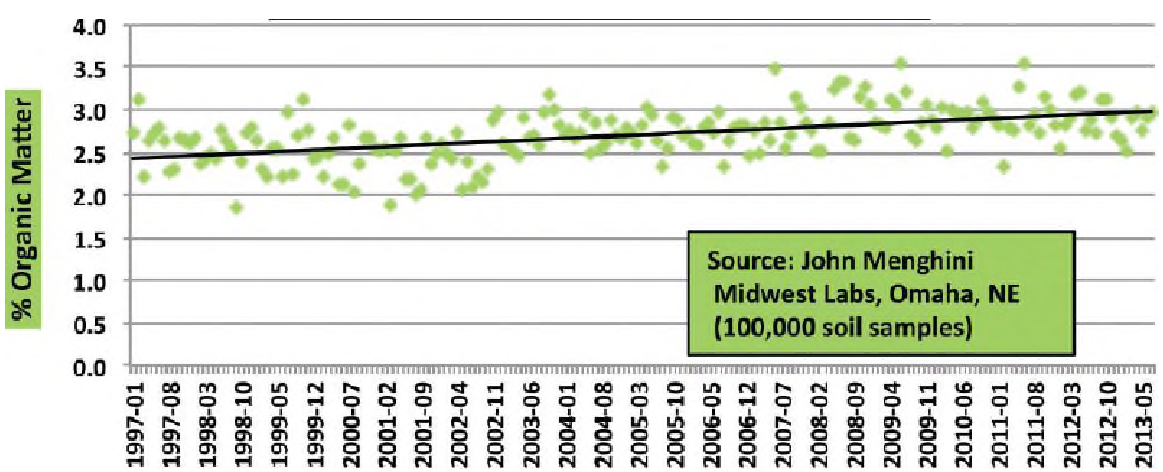

Figure 2.5. South Dakota Top Soil Organic Matter. (ACE, 2018)

2.2.4 Modeling Results

Since 2010, numerous studies have examined the international LUC for corn ethanol and their results showed that the international LUC was significantly lower than the 2010 RIA's estimation (Figure 2.6). These emissions correspond to the land cover change outside the U.S. induced by a change to corn ethanol. Typically, agro-economic models predict a reduction in U.S. crop exports for both corn and soybean as either corn exports are reduced or corn-soy rotation is converted to continuous corn. The models take into account the price effects of agricultural commodities as well as yield improvements and predict the type of land converted to crop production. The initial ILUC estimate from the California Air Resources Board (CARB, 2009), was a total of 30 g CO_2e/MJ of which about half was international LUC (see Figure 2.6). CARB revised its ILUC analysis with a total international component of 15 g CO_2e/MJ. These values are roughly comparable to the EPA international LUC result in Figure 2.5 though the 2010 RIA analysis includes additional categories. A series of peer-reviewed publications have shown that the international LUC is even lower. Publications from Purdue University (Tyner et al., 2010; Taheripour et al., 2017) are based on the GTAP model; which was employed by Argonne National Laboratories and incorporated into GREET (the model used by CARB and other state Low Carbon Fuel Standards, such as Oregon's Clean Fuels Program).

As discussed earlier, several studies based on GTAP evaluated biofuels induced land use changes and GHG emissions. Tyner et al. (2010) estimated the land use change and emissions associated with corn ethanol production using GTAP in support of the LCFS with the newer analysis resulting in lower ILUC emissions. A more recent study (Taheripour et al., 2017) incorporated a newer database (2011 database instead of 2004 database), added an intensification option to the model, and updated the yield price elasticity based on new data from the Food and Agriculture Organization (FAO). As Taheripour et al. (2017) stated, the previous versions of the GTAP model did not account for the intensification of pasture and assumed that a change in the harvested area equals a change in land cover, thus overestimating the emissions associated with ILUC.

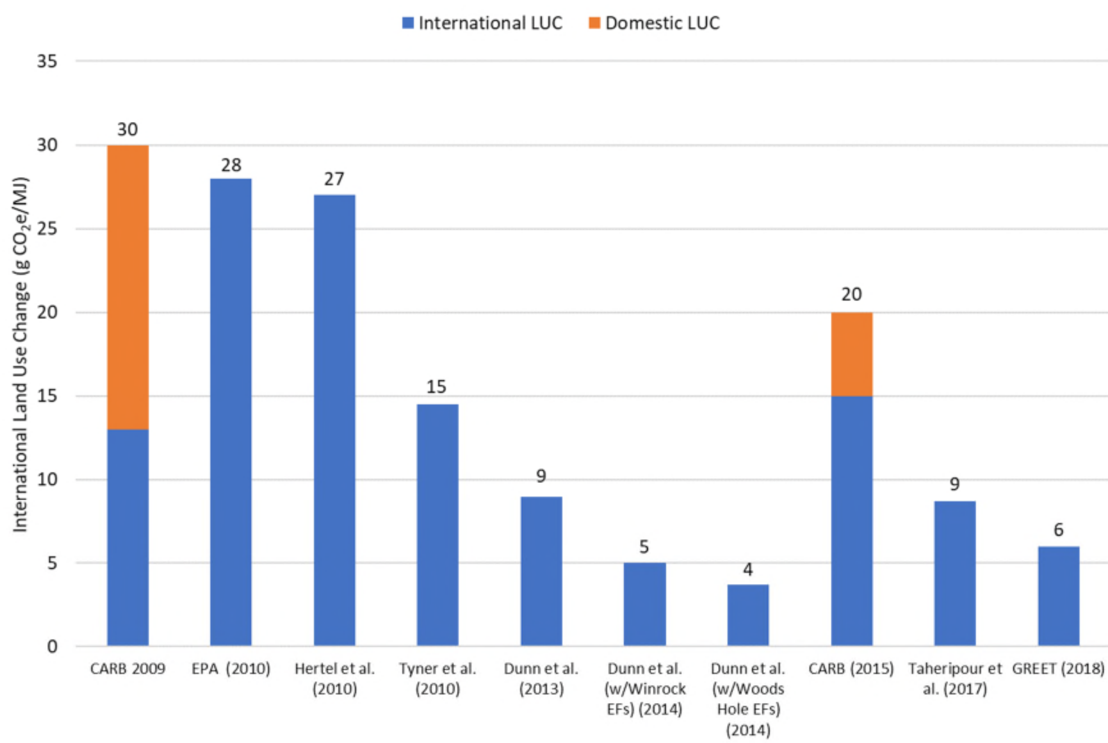

Figure 2.6. International Land Use Change Estimated by Several Studies. (Rosenfeld et al., 2018; ANL, 2018)

2.2.5 Summary of LUC Effects

International LUC for corn ethanol CI was overestimated in the 2010 RIA as shown by recent studies, availability of more recent data, and more realistic assumptions. Any estimation of LUC involves significant uncertainty with the largest uncertainties associated with the yield predictions on new and marginal land as well as the selection of land cover type. Shifts among agricultural commodities further complicates the analysis and adds a level of opacity to the modeling (CRC, 2014). While the results of LUC modeling are intrinsically uncertain, improvements in models such as those documented in recent GTAP studies indicate that EPA's assessment of both international LUC as well as U.S. LUC are overstated. In fact, soil carbon storage effects from corn farming should lead to a negative LUC in the U.S.

While the study by Searchinger et al. (2008) was the basis of international LUC calculation in the 2010 RIA, Zilberman (2017) has recently evaluated the assumptions made by Searchinger et al. (2008) and concluded that "Searchinger et al. (2008) results may now be seen as fundamentally flawed not just because the ILUC is uncertain and estimates vary considerably, but also because it fails to capture the basic features of agricultural industries and land resources." Dumortier et al. (2011) employed the same model used by Searchinger et al. (2008), but used more realistic assumptions and obtained completely different results (lower emissions). Rosenfeld et al. (2018) used the simulation results of the 2013 GTAP-BIO model available in ANL's CCLUB tool to calculate the impact of international LUC on corn ethanol CI under several scenarios and reported that the emissions associated with international LUC ranged from 1.3 to 16.9 g CO_2e/MJ. These findings that elasticity factors and other contributors to ILUC were overstated by the 2010 RIA were confirmed in a recent paper by Scully, et al. (2021). Finally, studies that compare ILUC modeling place a strong emphasis on Winrock land use conversion factors where a critical assumption is that crop land pasture emission rates are half those of pasture conversion (Malins, 2020). These same studies criticize the overestimation of soil carbon storage from ongoing corn farming practices predicted by CENTURY. However, the studies fail to recognize the merits of FASOM's analysis as used in the U.S. emission inventory that reflects real-world soil carbon storage effects.

Modeling Approach for This Study

This study combines the elements of several approaches to provide an updated assessment of the GHG intensity of corn ethanol. Repeating the steps in the 2010 RIA is a challenging process and EPA acknowledges this issue in the 2021 draft RIA; however, there are reasonable ways to update corn ethanol's CI without undertaking the extensive modeling effort completed in 2010. Here, domestic and international LUC were calculated based on the GREET (2021) model adjusted for the corn oil to biodiesel yield as shown in Table 2.3. The domestic and international ILUC emissions are multiplied by an allocation factor that assigns half of the emissions associated with corn oil production to biodiesel. The GREET model uses CCLUB (Dunn et al., 2017) to estimate the soil organic carbon storage as well as land conversion and associated emissions in response to biofuel expansion. Domestic LUC is based on average tillage practice in the U.S.; however, the more no-tillage practice is used by corn farmers, the more carbon will be stored in the soil and thus the impact of LUC will reduce.

Table 2.3. Change in GHG Emissions Due to Land Use Change (g CO_2e/MMBtu).

Study	Domestic	International
EPA 2010 RIA	-4,033	31,797
Rosenfeld et al. (2018)	-2,038	9,082
GREET1_2020	-2,314	6,300
GREET1_ 2020, allocated to corn oil	-2,199	5,986

The following calculation approach was used in this study. It allows for the assessment of the newest corn farming data, addition of the GTAP analysis for ILUC, and inclusion of the original 2010 RIA emission categories.

Emissions Allocated to Corn Ethanol and Corn Oil by Energy Content
Domestic ILUC: CCLUB
International ILUC: CCLUB
Domestic Rice Methane: ICF 2018
Domestic Farm Inputs: GREET minus international fertilizer
International fertilizer: ICF 2018 (to align with RFS categories, subtracted from domestic farm
International Rice Methane: ICF 2018

Emissions Assigned to Corn Ethanol
Tailpipe: ICF 2018
Fuel Production: GREET

3. CORN FARMING

The consumption of farming inputs such as fertilizers, pesticides, and energy such as diesel and LPG affect the GHG intensity of corn or crops that are grown to make up for corn used for biofuel production. Crop yields yield affect both the land required for crop production and LUC. This section includes new data on corn yield as well as crop inputs. This section also reviews recent data on farming and aligns it with the estimates in the 2010 RIA and the current GREET model.

3.1 Corn Farming

Historical data on corn yield indicates that the yield has increased steadily over time, from 85 bu/ac in 1988 to 172 bu/ac in 2020 as shown in Figure 3.1. The adoption of double-cross hybrid corn, continued improvement in crop genetics, adoption of N fertilizer and pesticides, and agricultural mechanization resulted in a steady increase of corn yield in the U.S. (Nielsen, 2017). Aside from the steady increase of corn yield, the harvested area of corn has increased over time. Due to the continuous improvement of corn yield, the production quantity has an upward trend (USDA NASS, 2018). The 2010 RIA estimated the corn yield for 2022 as 185 bu/ac, based on past 30 years of corn yields from USDA database. EPA's projection of corn yield for 2022 is consistent with the trendline of current data in Figure 3.1.

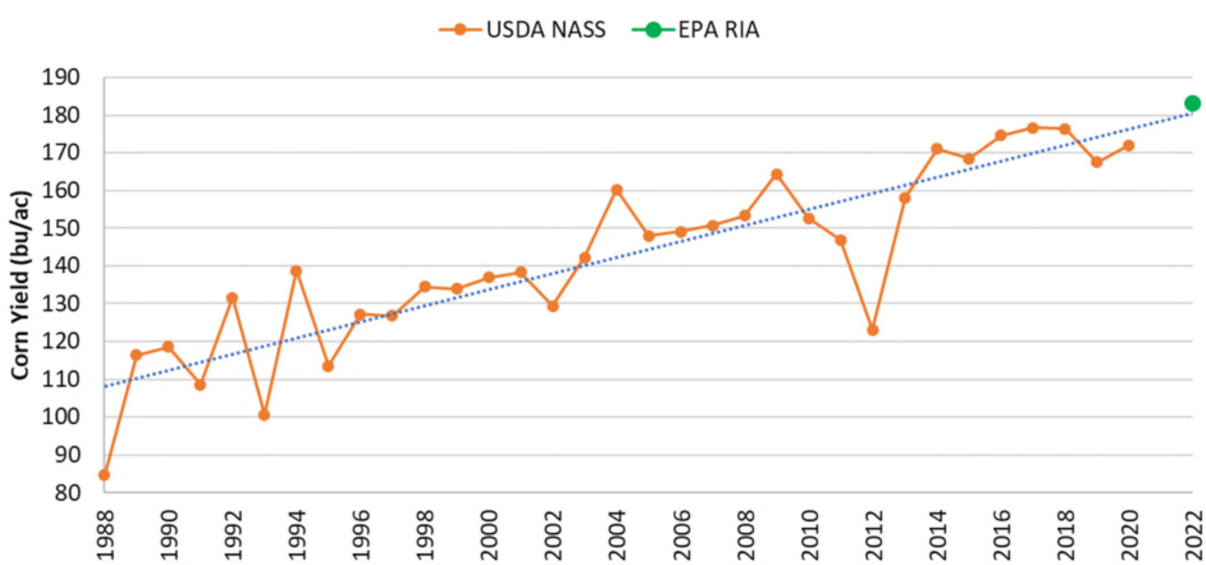

Figure 3.1. Corn Yield Over Time. (USDA NASS, 2020)

Management practices such as tillage, and nitrogen (N) application rate affect the GHG intensity of crops. In order to decrease the environmental footprint and lower production costs, farmers have started using new technologies such as precision agriculture to manage their fertilizer consumption. Reduced tillage has become a common practice across the U.S. farms, reduces soil emissions during the farming stage (Figure 3.2). Nitrogen inhibitors reduce the requirement for nitrogen and also reduce the formation of N_2O. Precision farming and guidance methods also allow for the more efficient application of nitrogen. The combination of all of these methods results in increased yield per acre and reduced nitrogen per bushel.

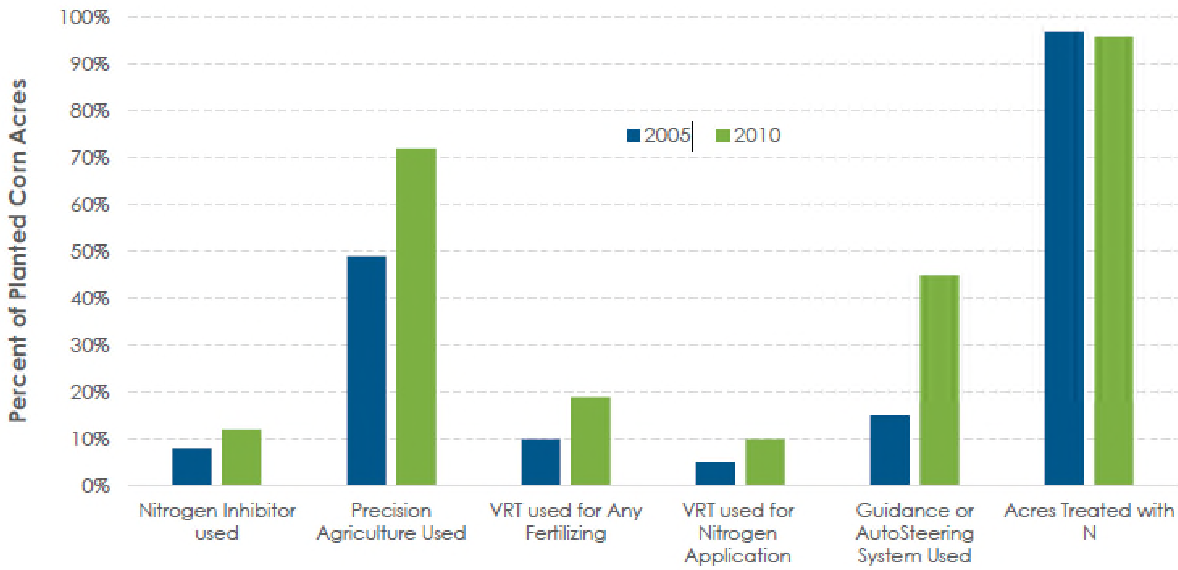

Figure 3.2. Changes in Corn Production Practices from 2005 to 2010. (Rosenfeld et al., 2018)

The leading corn farming states in the U.S. produce most of the ethanol in the country as shown in Figure 3.3. The location of ethanol plants is not surprisingly coincident with corn production. This co-location reduces corn transport distance and growth in corn production is occurring in the states with the highest yield per acre, which is shown in Figure 3.4.

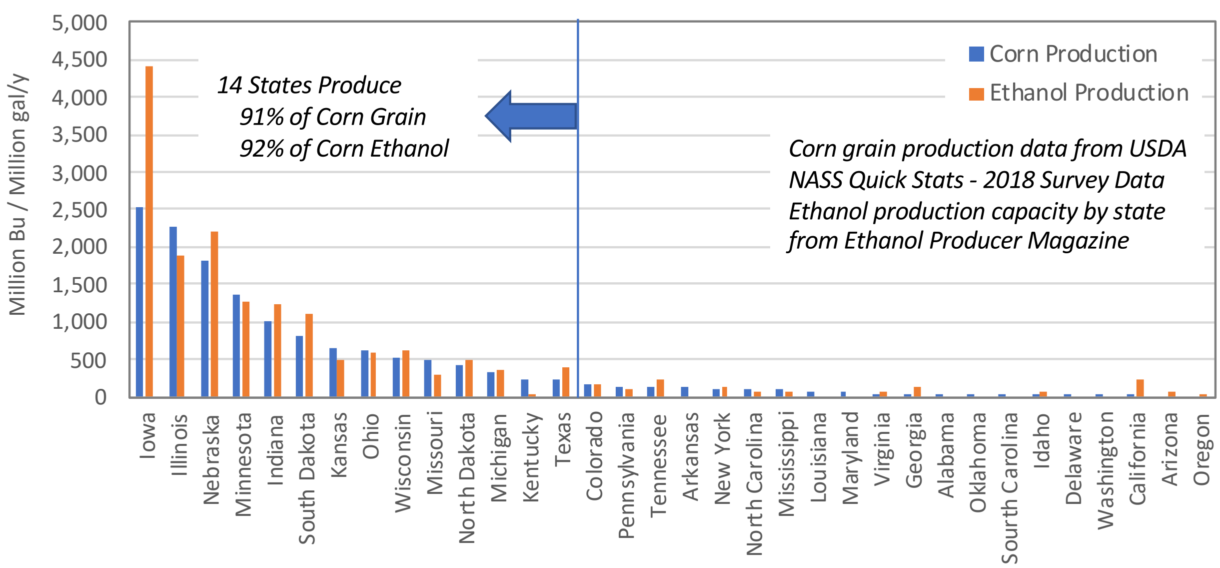

Figure 3.3. Corn Ethanol Production by State. (USDA NASS, 2018)

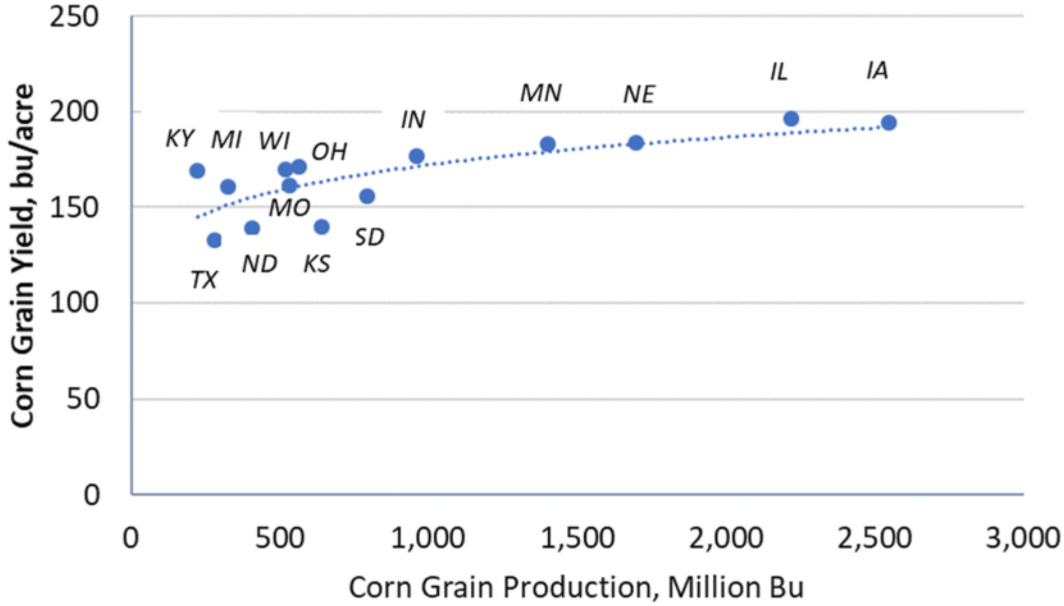

Figure 3.4. Average Corn Grain Yield vs. Production for 14 States.
(2014-2018 Weighted Average) (USDA NASS, 2018)

Iowa, Illinois, and Nebraska are the three states with the highest corn production in the U.S. An analysis of NASS data for applied nitrogen and corn yield shows consistent reduction in the nitrogen application rate per bushel of corn (Figure 3.5). The reduction in nitrogen application rate is consistent with the 2010 RIA estimate discussed below.

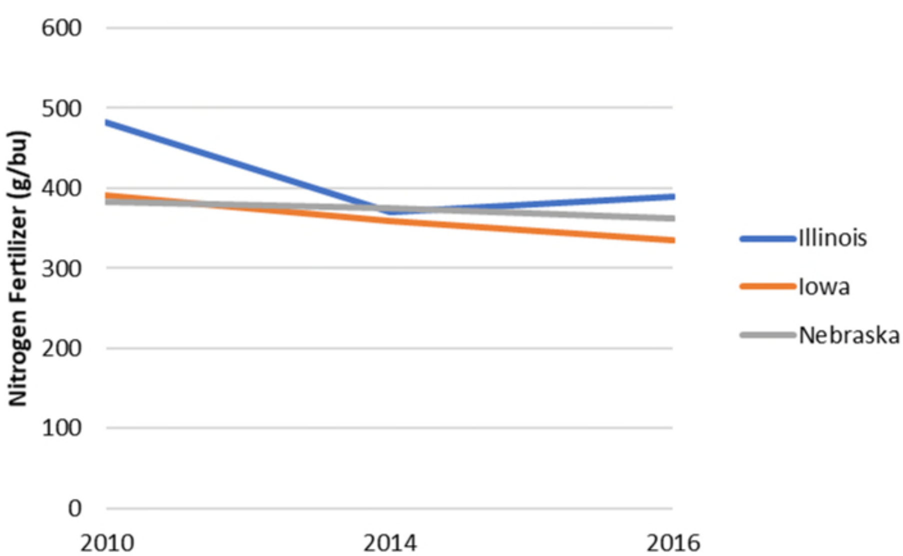

Figure 3.5. Nitrogen Fertilizer Use Rate in the Three Largest Corn Producer States.
(USDA NASS, 2018)

Domestic agricultural use of fertilizers, pesticides, and energy was projected by FASOM in the 2010 RIA. The 2022 projections are compared to several evaluations of NASS data in Table 3.1. The 2010 RIA used the GREET interim emission results to calculate the upstream emissions associated with agricultural inputs. The 2022 projections for farming inputs in the RIA reflect improved yields and advancements in farming techniques, which, in some cases, may not have yet been achieved. Overall, this comprises a small portion of ethanol's CI relative to the LUC portion discussed above.

Table 3.1. Farming Inputs of Corn in the U.S.

Input	Unit	GREET (2021)	Rosenfeld et al. (2018)	USDA NASS (2018)	EPA RIA[d]
Analysis Year		2020	2015	2016	2022
N	g/bu	401.5	373	380	344
P_2O_5	g/bu	150.6	128	165	79
K_2O	g/bu	152.3	130	193	98
Lime	g/bu	1,457	1,150	N/A[c]	260
Herbicide	g/bu	6	6	3	5
Pesticide	g/bu	0.01	0.1	N/A	1
Diesel	Btu/bu	5,200	4,730[b]	6,388	9967
Gasoline	Btu/bu	802	1,413	774	1042
Electricity	Btu/bu	1,326	441	1,089	19
Natural Gas	Btu/bu	479	1,301	1,212	1283
LPG[a]	Btu/bu	1,026	1,723	1,297	-

[a] Liquified Petroleum Gas
[b] The energy usage of corn ethanol was not mentioned in Rosenfeld et al. (2018), however they mentioned that they obtained the data from GREET (2015). To make it comparable, the energy usage data for Rosenfeld et al. (2018) were obtained directly from GREET (2015).
[C] Data was not available.
[d] From EPA RIA, Table 2.4-5. The values are listed per MMBtu of ethanol which appear to incorrectly labeled and not possible. If for example, the N fertilizer of 138.8 lb/MMBtu are taken as lb/acre yield and combined with a corn yield of 183 bu/ac from the RIA the N rate is 344 g/bu.
GREET1_2021 is the study input

3.2 Sensitivity Analysis of Farm Inputs

A sensitivity analysis was conducted to investigate the impact of each input on overall CI of corn and the results are shown in Figure 3.6. Fertilizer application rates, farm yields, transport distances to ethanol plants, and N_2O production rates were examined for 12 corn farming states using the GNOC model,[5] which provides an easy-to-use assessment tool with global applicability. Uncertainty distribution functions were developed based on the standard deviation of historical data and other variability factors to provide inputs for a Crystal Ball™ simulation of the GHG intensity of corn. The analysis shows that nitrogen fertilizer and N_2O

[5] http://gnoc.jrc.ec.europa.eu/

emission are the most sensitive inputs, implying that a reduction in nitrogen fertilizer application rate significantly decreases the GHG intensity of corn and the CI of corn ethanol.

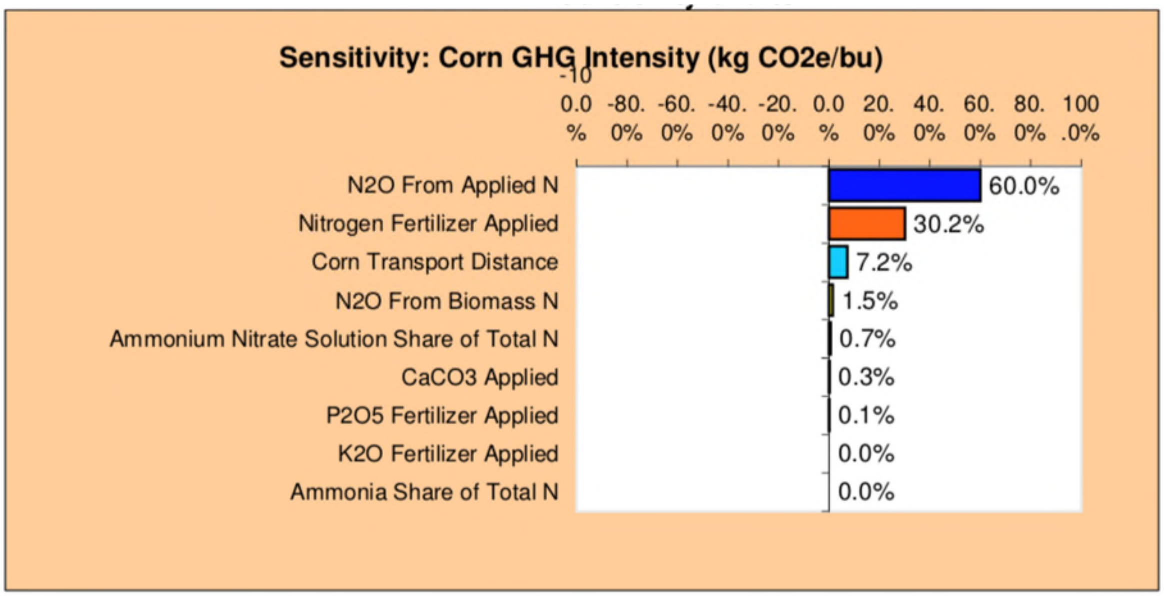

Figure 3.6. Sensitivity Analysis of Farm Inputs.

4. Impact of Co-Products on corn ethanol CI

The corn farming system and ethanol production generate several co-products that were considered in the 2010 RIA. These include DGS, Corn Distillers' Oil (CDO), and stover that is harvested with corn. Stover was considered as a fuel feedstock and not animal feed co-product. The effect of these co-products on GHG emissions is discussed in the following sections. Some ethanol plants also capture fermentation CO_2.

The 2010 RIA also used the study published by Argonne National Laboratory to estimate the DGS replacement rates for corn and soybean meal in animal feed. Production of DGS effectively results in a credit since DGS is a suitable source of animal feed and displaces agricultural crops like corn and soybean meal. However, since FASOM takes the production and use of DGS into account, no further allocation (displacement) was conducted in the 2010 RIA.

4.1 DGS Co-Product

Distiller's grains are the nutrient-rich co-product of the ethanol production process and provide an alternative to corn and soybean meal feed. Wet distiller's grains are sold to local markets due to their high moisture content and low shelf life. But generally, the distiller's grains are dried to increase the shelf life and facilitate transportation over longer distances. The product is referred to as Distiller's Dried Grains with Solubles (DDGS) (Iowa Corn, 2019). In the U.S., ethanol plants have the capacity to produce substantially more than 15 billion gallons of ethanol and 44 million metric tonnes of DDGS (U.S. Grain Council, 2018). This effect is significant since an acre of land producing ethanol for corn produces as much feed as an acre of soybeans. Due to its nutritional value, DDGS is considered a good substitute for soybean and canola meal. A recent study has investigated the effect of DDGS vs. soybean meal and canola cake on feed intake, milk production, and milk quality in dairy cows and concluded that DDGS can substitute for a soybean-canola mixture without affecting feed intake, milk yield, and quality, or sensory quality (Gaillard et al., 2017).

Figure 4.1 shows the prices of DDGS and soybean meal over time with a correlation in price activity. Rises in soybean meal prices are followed by rises with DDGS prices supporting the substitution effect. The replacement value of DGS was less well-understood in 2010 when corn ethanol was a less mature technology. While the overall substitution effects are more complicated, DDGS that displaces soybean meal results in the avoidance of emissions from soybean farming.

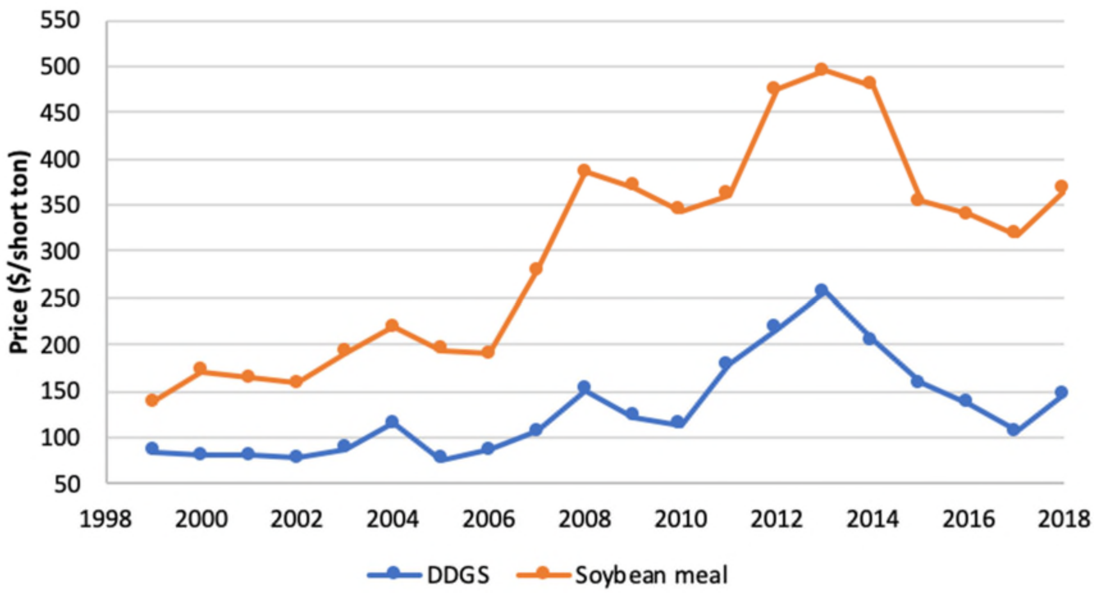

Figure 4.1. Historical Prices of DDGS and Soybean Meal. (USDA ERS, 2018; World Bank, 2018)

Soybeans as legumes fix nitrogen in the soil, which provides nitrogen for soybean crop and the following crop which is typically corn. Thus, the application of nitrogen fertilizer is not required for soybean farming. However, without N fertilizer, the soybean yield is limited to 50 to 60 bu/ac. In order to achieve higher yields, 30 to 60 lb/ac of nitrogen fertilizer is required (Schmidt, 2016). In recent years, more fertilizers, especially nitrogen fertilizer, have been used in soybean farming to increase yields (McGrath et al., 2013) (Schmidt, 2016). The GREET model input for soybean farming (ANL, 2018) is 48 g/bu of nitrogen fertilizer, which is based on a 2008 study (Huo et al., 2008). However, recent USDA data indicates that the consumption of nitrogen fertilizer in soybean is 18 lb/ac which translates to 166 g/bu (USDA NASS, 2018). The application of nitrogen fertilizer on soybean crop is triple the GREET input, which directly affects the emissions related to soybean production.

Since DDGS is a substitute for soybean meal, the avoided emissions are substantially higher than originally anticipated. Correcting the nitrogen fertilizer use for soybeans allows for a better estimate of the displacement value of DDGS with corn ethanol production. The FASOM model estimate for nitrogen usage in soybean farming in 2022 in the 2010 RIA appears to be less than 10 lb/ac (Figure A.**1**) with a projected soybean yield as 50 bu/ac in 2022. These parameters correspond to a nitrogen application rate of 64 g/bu, which is much lower than current nitrogen fertilizer use rate reported by USDA NASS (2018) (166 g/bu). (See Appendix A for a discussion of nitrogen application)

By comparing the nutritional value and moisture content, one lb of DGS is equivalent to 0.781 lb and 0.307 lb of feed corn and soybean meal, respectively. Therefore, one lb of DGS production results in the displacement of 118 g CO_2e plus 96 g CO_2e if replaced for soybean meal and corn (Table 4.1).

Table 4.1. The CI of DGS Using Displacement Method.

Feed Material	Soybean Meal[a]	Corn	Total
CI (g CO_2e/g)			
Production	0.53	0.24	
ILUC	0.32	0.03	
Total	0.85	0.27	
Displacement Ratio	0.307	0.781	
g CO_2e/lb DGS	118.4	95.7	214.1
g CO_2e/MMBtu EtOH	7,694	6,216	13,910

[a]The co-product credit for DGS depends on the crops that it displaces. In order to assess ILUC based on Figure 1.3, the displacement effect of corn to DGS is already taking into account in ILUC modeling in GREET with 5.0 lb DGS, dry basis per gal ethanol. However, the higher ILUC of soybean meal has not been fully taking into account due to the new market introduction of DGS. The displacement effect of urea feed is now shown here.

4.2 Corn Distillers Oil

Another important co-product of the ethanol plant is corn distillers' oil (CDO). Since 2010, corn oil extraction has become a common practice in bioethanol plants due to technological advancements, although it requires additional investment (Batres-Marquez, 2018). In the U.S., almost 85% of dry grind ethanol plants extracted corn oil in 2015, producing about 1.22 million metric tons of CDO (Veljković et al., 2018), and the extraction of CDO has continued to grow (Figure 4.2), which is consistent with the projections in the 2010 RIA. Several studies have shown that CDO has comparable properties to diesel and is used for biodiesel production (Balamurugan et al., 2018; Kumar and Kumar, 2013).

In the U.S., CDO represented the fastest expanding oily feedstock for biodiesel production in 2013 (Grooms, 2014). The California LCFS originally had a very favorable CI for biodiesel produced using CDO as feedstock.[6] This drove increased use of CDO as feedstock. In 2018, about 2,060 million lb of CDO, or 50% or production, was used from biodiesel production based on EIA statistics.

[6] The LCFS CI was 4 g CO_2e/MJ of biodiesel for several years. This value has since been raised to about 22 g CO_2e/MJ, but the low initial value provided an incentive to use CDO as a biodiesel feedstock.

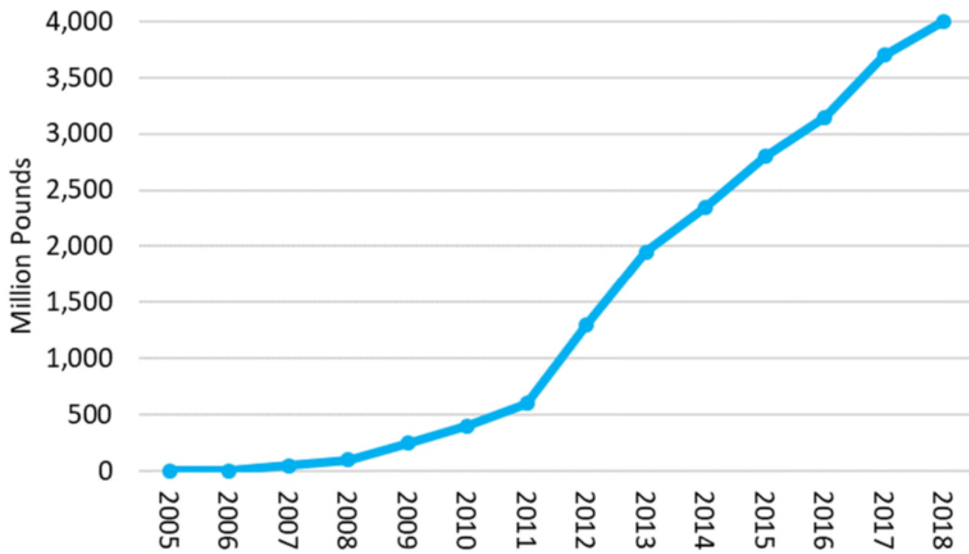

Figure 4.2. Corn Distillers' Oil Production in the U.S. (USDA NASS, 2018; RFA, 2019)

4.2.1 Corn Oil as Coproduct of Ethanol Production in EPA RIA

EPA estimated that by 2022, 70% of dry mill ethanol plants will conduct extraction, 20% will conduct fractionation, and 10% will not extract CDO. These estimates were incorporated into the FASOM and FAPRI/CARD models to account for extracted corn oil as biodiesel feedstock. The 2010 RIA projected that by 2022, 680 Mgal or 4000 million lb of CDO is produced as a by-product of corn ethanol production and used to produce biodiesel. The RIA analyzed the displacement of CDO with other agricultural products such as soy oil in the FASOM model. If CDO were treated as a fuel product, it would receive a greater share of the ethanol plant emissions and the ethanol plant emissions would be reduced. In practice, about half the CDO is used as biodiesel; which means that a corn ethanol biorefinery produces two energy products and the emissions and ILUC should be allocated between ethanol and CDO for biodiesel.

4.2.2 CDO Under Various Allocation Methods

Since CDO is a co-product of ethanol production, emissions from corn farming and ethanol production should be allocated to CDO or treated as a displacement credit. Several allocation methods allow for the treatment of CDO including displacement with soybean oil, and diesel, or energy allocation with ethanol and DGS. Each allocation method results in a different effect on the CI of corn ethanol shown in Table 4.2 as the estimated reduction in ethanol CI due to CDO production. Although the RIA accounted for CDO using the FASOM model, which focuses on the displacement of agricultural products, the energy allocation method is a better choice since corn oil us for biodiesel production has expanded in recent years. The effect of the different allocation approaches is shown in Table 4.2, energy allocation method results in more reduction in CI of corn ethanol than displacing with soybean oil. While displacing CDO with diesel is an extreme case, biodiesel from corn oil is an alternative for diesel fuel, so displacing with diesel is an option. EPA should factor into its analysis the fuel value of CDO. Energy inputs and emissions for ethanol plants as well as ILUC associated with corn usage should be assigned to both ethanol and CDO.

Table 4.2. The Effect of Displacement Method of CDO on CI of Corn Ethanol.

Modeling Approach	**CI (g CO_2e/MJ Ethanol)**
EPA RIA	~-1.14
CDO displacing with soybean oil[a]	-1.20
CDO displacing with diesel	- 4.94
Energy Allocation	-2.12

[a] Based on 166 g/bu of nitrogen fertilizer.

4.3 Replacement Feed

Corn stover (cobs and residue) is an important part of the life cycle of corn, either as fuel or as animal feed, but most LCA models treat them separately from starch ethanol (Welshans, 2014; Mueller, 2015). Corn stover is used as a cellulosic feedstock for ethanol production. Corn stover can also be used as a replacement for corn and hay or corn silage in animal feed. Mueller et al. (2015) conducted a study to investigate the effect of corn stover removal on overall emissions of ethanol. The analysis included a displacement credit for the 30% corn stover used as corn replacement feed (CRF) as well as the DGS produced from the grain corn. The displacement credit for CRF is based on a substitution ratio of 0.5 kg corn and 0.5 kg hay being equivalent to 1.0 kg of CRF on a dry matter basis. Although CRF is a suitable substitute for feed ingredients such as corn and hay, it requires pretreatment which involves consumption of chemicals such as calcium hydroxide. On the other hand, CRF has a feed and LUC credit. The results showed that using corn stover as animal feed has a co-product credit of -6.6 g CO_2e/MJ which potentially reduced the corn ethanol CI. The extent of CRF was not explicitly modeled by EPA in the 2010 RIA, but should be considered by EPA in reassessing the CI of corn ethanol.

5. Biorefinery Technologies

The performance of biorefineries affects life cycle GHG emissions due to the use of feedstock and fuel resources as well as chemical inputs. The key factors affecting GHG emissions for dry mill ethanol plants are shown in Table 5.1. The future energy inputs and yield for ethanol plants were examined in the 2010 RIA. Many of the technologies that affect dry mill ethanol plants were identified. The factors that affect energy inputs and yields, as well as the differences between the performance projected in the RIA and actual performance are examined here.

Table 5.1. Ethanol Plant Performance Parameters.

Performance Trend	Key Drivers	Effect on LCA
Increased Yield	Starch hydrolysis and fermentation efficiency Cellulosic conversion	Higher yield reduces corn upstream emissions and ILUC as well as DGS mass and co-product credit.
Reduced Natural Gas Consumption	Reduced drying energy, plant heat integration, corn oil extraction, advanced separation processes	Natural gas combustion and upstream emissions are proportional to use rate.
Reduced Electric Power Consumption	Ongoing improvements in efficiency and yield and cogeneration reduce power requirement. Corn oil separation requires additional electrical power.	Power generation and upstream emissions are proportional to use rate.
Increased Corn Oil Production	Corn oil in DGS is extracted by centrifuge or with solvents.	Several approaches. Substitution for agricultural products or allocation.
Reduced DGS Mass	Increased ethanol and corn oil yield reduce starch and oil component of DGS without changing protein output.	Affects co-product credit. Protein content is not affected. Only carbohydrate and fat fractions are affected by yield improvements.
Reduced Chemical Consumption	Increased yield and improved monitoring.	Reduced upstream life cycle for chemical production.
CO_2 Capture	Growth in CO_2 capture from ethanol plants which have a pure CO_2 stream. Avoids CO_2 production from other sources.	Several possible approaches, none used in RIA. Credit or allocation for CO_2 storage/ productive use.

The efficiency of corn ethanol biorefineries has improved (see following sections) in the past decade resulting in the use of less corn per gallon of ethanol and lower energy inputs. Corn ethanol plants also produce about 5% of their energy output as corn oil.[7] The primary factors affecting ethanol plant performance are discussed below.

[7] 0.25 lb/gal ethanol × 15,993 Btu/lb (GREET soy and canola LHV) /77,000 Btu/gal denatured ethanol = 5.2%.

5.1 Corn Ethanol Yield

Several technologies have contributed to improvements in the ethanol yield per bushel of corn. Increased ethanol yield results in less corn used per gallon of ethanol which results in lower farming emissions, lower land use, and LUC per gallon of ethanol. Figure 5.1 shows trends in historical yield data as well as projections. Data from the GREET model that was available at the time of the 2010 RIA (Version 1.8c) is compared with industry data. These values are consistent with EPA's projections in the RIA with the trend line from the industry data slightly under the 2022 RIA projection. However, the input to the FASOM and FAPRI modeling system is 5% lower than the yield projected by EPA[8] for dry mill ethanol plants.

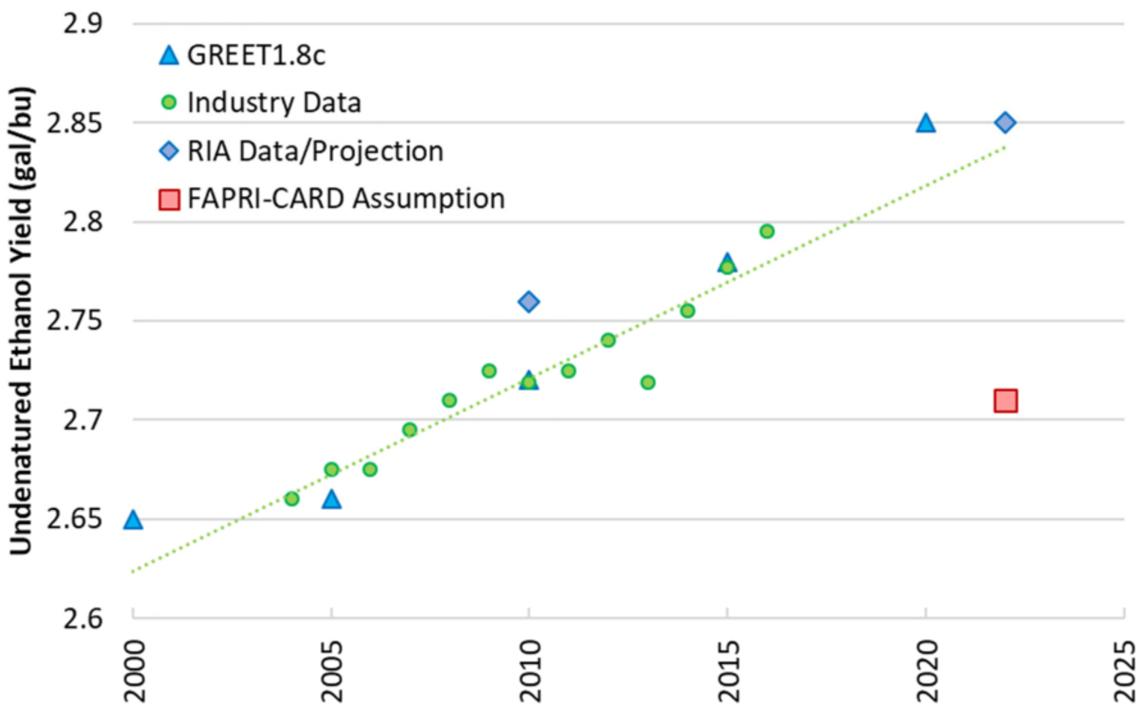

Figure 5.1. Dry Mill Corn Ethanol Yield Data and Projections.

5.1.1 Ethanol Yield in EPA 2010 RIA

EPA assumed ethanol yields of 2.71 gallons per bushel for dry mill plants and 2.5 gallons per bushel for wet mill plants and FASOM and FAPRI-CARD models used these yield assumptions. With the growth of dry mill plants, the aggregate yield should be higher than the values in the 2010 RIA. A higher yield would result in lower fertilizer use and ILUC. A first-order approximation is that corn farming and LUC related emissions should be 10% lower than those predicted by EPA due to actual yield improvements.

[8] 2010 RIA Section 2.4.7.1 EPA states the FASOM assumption

EPA identified yield projections that are consistent with industry data.[9] The discrepancy may be due to the use of the modeling systems for other programs or challenges associated with changing a modeling assumption. In any event, the lower corn ethanol yield overestimates the corn feedstock requirement for ethanol production. An offsetting factor would be that the model predicted higher production of DGS and greater co-product displacement but the net effect would still be an overestimate of corn farming emission and land use effects.

5.1.2 Plant Debottlenecking

The debottlenecking process helps to increase the yield and reduce energy consumption in corn ethanol plants. New technologies and reviews of material and steam flows optimize the utilization of critical processes to boost overall throughput, increase yield from base throughput, or both. Membrane dehydration technology is one such technology which helps in energy reduction, purity flexibility, and debottlenecking distillation capacity and dehydration. These improvements have contributed to the overall improvement in U.S. ethanol plants.

5.1.3 Enzymes and Chemicals

Enzymes are among energy-intensive inputs for corn ethanol production. Companies like Syngenta and DuPont are providing enzymes that are more efficient in terms of increasing the ethanol yield and simultaneously reducing the enzyme consumption. In a new study by Kumar and Singh (2016) that investigates using amylase corn and superior yeast in corn ethanol production, the authors concluded that use of amylase corn and superior yeast in the dry-grind processing industry can reduce the total external enzyme usage by more than 80%. Combining their use with in situ removal of ethanol during fermentation allows efficient high-solid fermentation. Also, their study showed that the ethanol yield in their process is 4.1% higher than the conventional process of corn ethanol production.

5.2 Energy Consumption

Ethanol plants have reduced natural gas and power consumption through numerous factors such as heat integration, combined heat and power technologies, variable frequency drives, advanced grinding technologies, various combinations of front and back end oil separation, and innovative ethanol and dried distillers' grains (DDG) recovery (Mueller, 2016). These technologies directly affect the CI of corn ethanol. These energy-saving technologies were identified in the 2010 RIA and EPA modeled the natural gas and electric power consumption for corn ethanol plants that EPA projected would be built with wet and dry DGS (Figure 5.4).

[9] RIA Section 1.1.1.1

Plant configurations modeled by EPA.

- Baseline plant
- Combined heat and power (CHP)
- CHP with corn oil fractionation
- CHP with corn oil fractionation and membrane separation
- CHP with corn oil fractionation, membrane separation, and raw starch hydrolysis

EPA placed considerable emphasis on modeling CHP. This technology has proven borderline economical with the lower costs of natural gas as well as lower costs of electric power. EPA projected that 70% of dry mill plants would adopt corn oil fractionation and this adoption rate has been exceeded.

Ten years of experience has provided insight on the actual energy use for dry mill ethanol plants. Data from ethanol plant operation has become available from industry surveys as well as pathway registrations under the California LCFS (Cooper, 2008; ACE, 2018; CARB, 2018 list of plants).

The GHG intensity of dry mill ethanol plants that were registered under the LCFS in 2016 is shown in Figure 5.2. These data are based on the CA-GREET2 model and the current CI values for these facilities with the CA-GREET3 model would be lower. However, the broader data set was available for more facilities in 2016. These ethanol plants that register under the LCFS tend to be closer to California and the lower CI ethanol plants are also represented here. The lower CI of advanced corn ethanol is attributed to the use of biomass or biogas from anaerobic digester as sources of energy. The CI values combined with LCFS applications allows for an estimation of the distribution of natural gas usage among these facilities. The range of natural gas usage was distributed equally among six bins and the range of each bin is shown in Figure 5.3. The average natural gas usage is 20,706 Btu/gal, LHV. These energy use rates and trend in reduced energy consumption over time are consistent with a survey of dry mill ethanol plants shown in Figure 5.4. These data are consistent with an industry average natural gas use rate of 22,500 Btu/gal by 2022, which is used in the assessment of GHG emissions in Section 8.

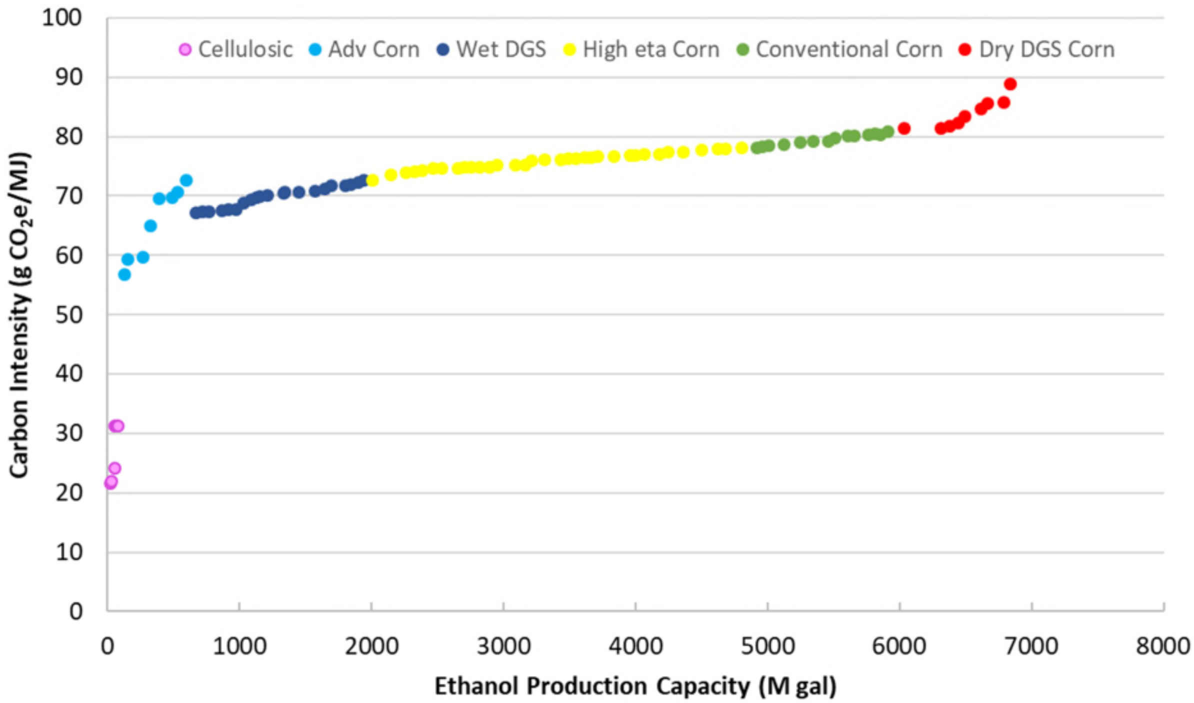

Figure 5.2. CI of Corn Ethanol with Various Technologies Registered under CARB (CA-GREET2 model) (CARB LCFS Pathway List)

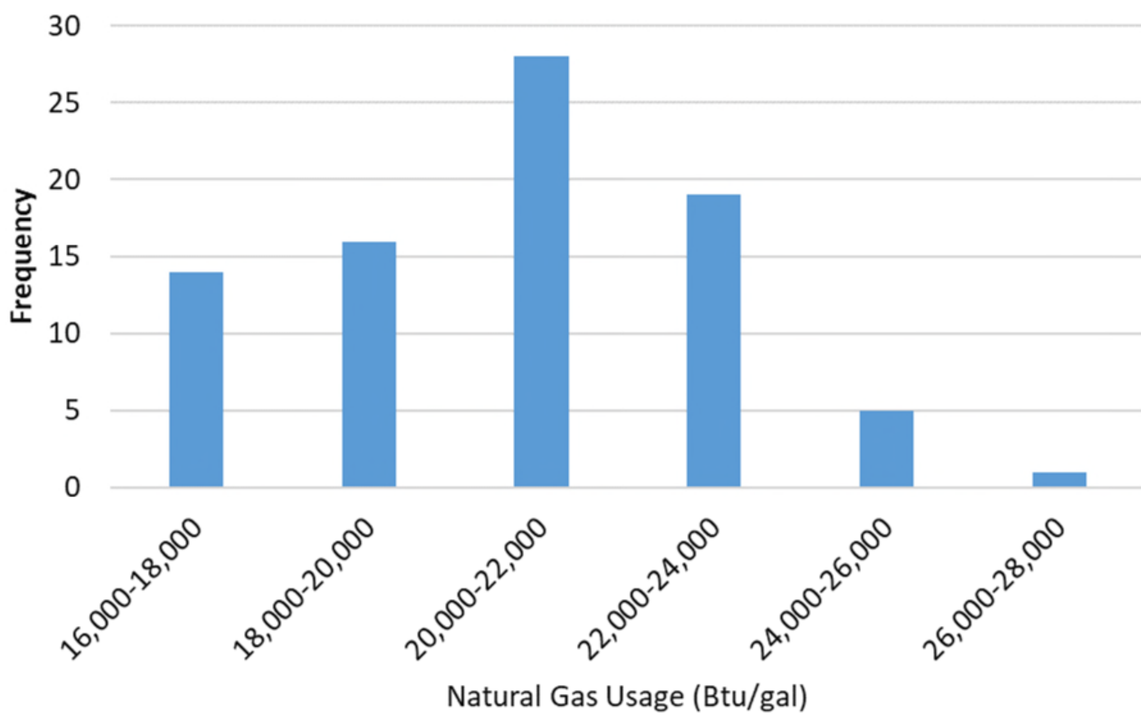

Figure 5.3. Distribution of Natural Gas Usage Among Ethanol Production Facilities.

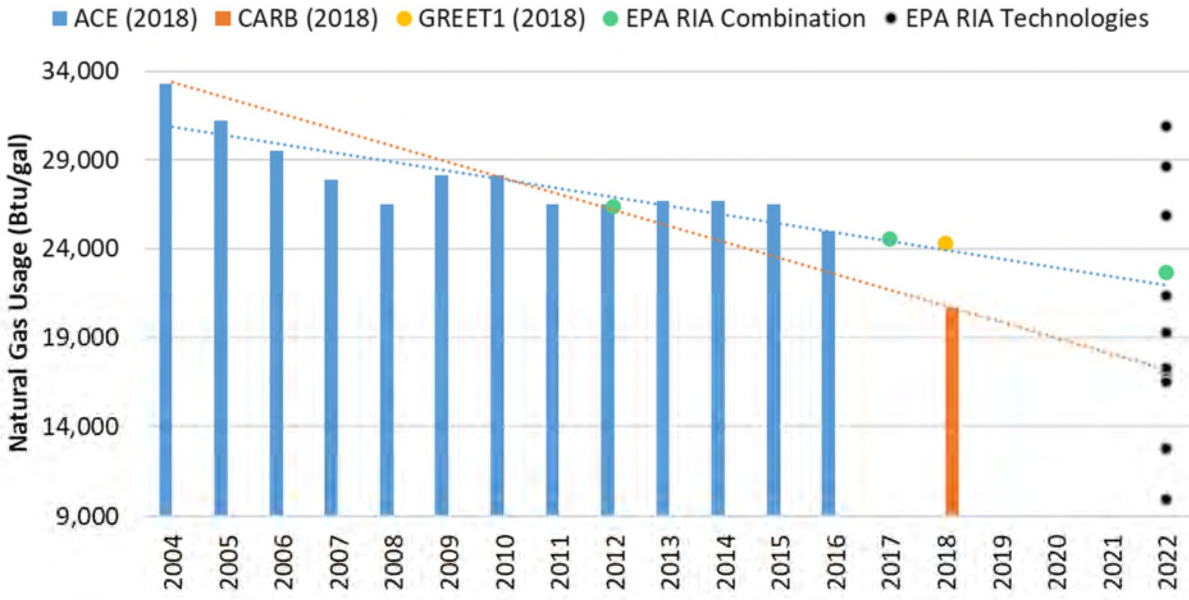

Figure 5.4. Decrease in Natural Gas Usage Since 2004 (EPA RIA combination denotes dry mill plant with only natural gas which produces 63% dry DGD and 37% wet DGS).

While the electricity consumption has not decreased significantly since 2010 (Figure 5.5), it has a decreasing trendline which implies lower electricity is being consumed by ethanol plant due to employing newer technologies. The overall impact of electric power should be examined as described in Section 6.6.

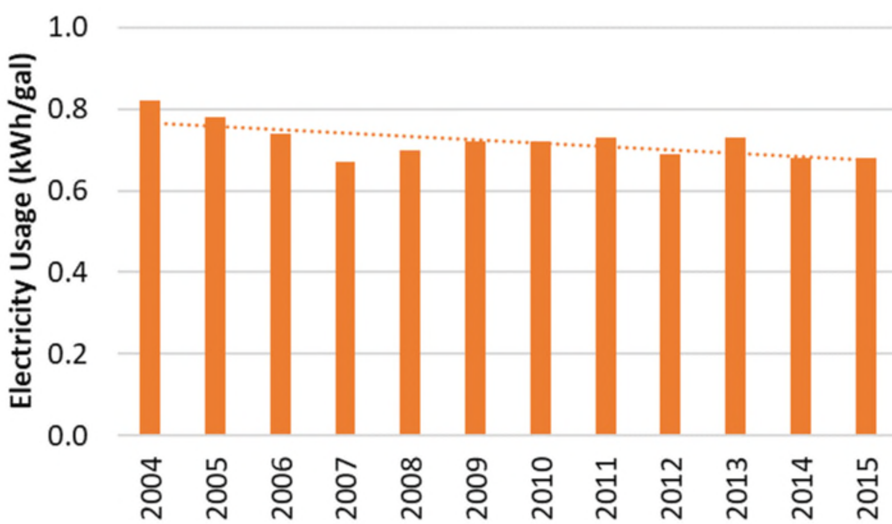

Figure 5.5. Electricity Conusmption in Corn Ethanol. (ACE, 2018)

5.3 CO_2 from Corn Ethanol

Many corn ethanol plants provide CO_2 for beverage and industrial purposes. The CO_2 generated in the fermentation process of corn-ethanol plants has a high market share such that it is the largest single-sector CO_2 source for the U.S. merchant gas markets. As a valuable product for the food industry, not only is the CO_2 not a waste product, but it also generates GHG savings credit which lowers the final CI of corn ethanol (Mueller, 2017). Absent ethanol plants, other sources of CO_2 would need to be utilized for refrigeration, beverages, and other applications (Mueller, 2019). Carbon in the fermentation CO_2 corresponds to half of the carbon in ethanol or about 37,000 g CO_2/MMBtu. After electric power for capture and liquefaction the GHG savings are over 30,000 g CO_2/MMBtu for ethanol plants that capture CO_2. In addition, at least 4 different ethanol plants are deploying carbon capture and EPA did not take into account the benefits of CO_2 capture or utilization in the 2010 RIA. The effect of these technologies is not included in the analysis in Section 8.

6. PROCESS FUELS

6.1 EPA RIA Fuel Production

In 2010, EPA considered several process fuels and different ethanol production practices (dry mill and wet mill) and came up with a combination of use rates for process fuels. EPA used the ASPEN models developed by the USDA to estimate the energy use at dry mill plants. The use rates are for a new dry mill corn ethanol refinery in 2022 that uses natural gas as its process fuel. The plant has a fractionation technology to extract corn oil and will produce a composite DGS coproduct that is 63% dry and 37% wet. Fuel Production emissions for this refinery were estimated as ~28,000 g CO_2e/MMBtu in 2022. The 2010 RIA used the GREET model to estimate the GHG CI of natural gas and electricity. These data have evolved and more recent estimates are included in the analysis in Section 8.

6.2 Phase Out of Coal

The use of coal as a fuel for ethanol plants has declined since 2010. The majority of ethanol plants are using natural gas as process fuel and only a small portion of the energy used in ethanol plants is coming from coal. According to corn ethanol pathways in the 2015 GREET model, on average, only 8 percent of the energy for steam production at U.S. ethanol plants comes from coal (ANL, 2018). EPA's projection of reduction in coal use were consistent with actual experience.

6.3 Natural Gas Production and Methane Emissions

Further refinements of the LCA of natural gas have led to many publications addressing the issue of energy inputs and methane emissions from natural gas production and distribution. GHG emissions associated with natural gas extraction have resulted in an increase in the GHG intensity of natural gas process fuel, which is taken into account in this study. As can be seen from Figure 6.1, the CI used for natural gas in this study was slightly higher than the CI used in the 2010 RIA.

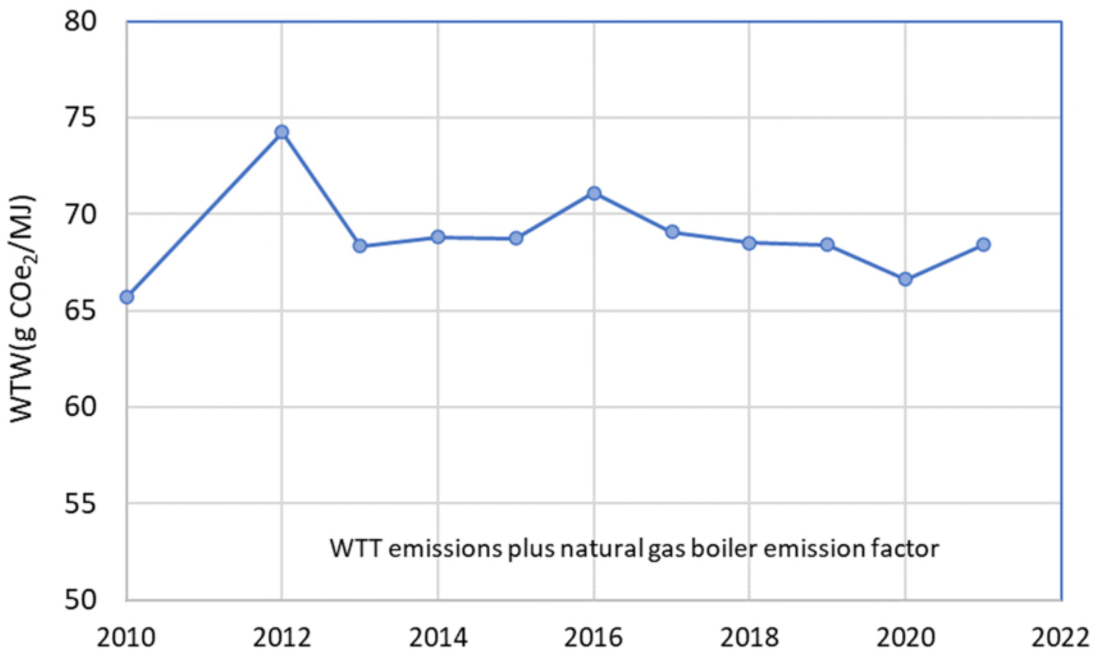

Figure 6.1. Well to Wheel (WTW) Carbon Intensity of Natural Gas plus Boiler Emission Factor in GREET. (ANL, 2018, GREET versions from 1.8b to 2021)

6.4 Biogas and Biomass Process Fuel

Landfill gas and biogas are potential process fuels for biorefineries which help to reduce the CI of biofuel (Table 6.1). The introduction of low GHG process fuel at biorefineries has been motivated by the RFS2 as well as the California LCFS. Below are several strategies employed by biorefineries to reduce the CI. All of these technology improvements lead to low CI ethanol that could be analyzed by EPA in the current rulemaking.

- Landfills collocated with ethanol plants;
- On-site anaerobic digestions of manure with avoided methane emissions;
- Anaerobic digestion of stillage;
- Electricity cogeneration;
- Solid fuel biomass combustion.

Table 6.1. Effect of Biogas on Carbon Intensity of Corn Ethanol.

Process Fuel	Biogas Fraction	CI (g CO_2/MJ), LHV	
		NG/Biogas	Ethanol
Natural Gas	100%	69	50
On-site Landfill	50%	1	40
Dairy Anaerobic Digester	15 to 25%	-250	0

6.5 Electric Power

Corn ethanol plants use electric pumps, hammer mills, and other electrical equipment. The electrical load has steadily declined over time from over 1 kWh per gallon of ethanol to an average of 0.65 kWh per gallon (ACE, 2018) over a 10-year period. Over the same time period, the GHG intensity of the U.S. grid has declined from 750 to 505 g CO_2e/kWh on a life cycle basis. On the other hand, the 2010 RIA projected power use of 1.09 kWh/gal with projects of reduced power consumption. Actual power use had dropped to about 30% less than the projected value.

6.5.1 Grid Carbon Intensity

The carbon intensity of electric power has declined with the expansion of natural gas production and the declining price of natural gas (Figure 6.2). Carbon intensity of electric power based on GREET has declined by 34% from 2010 to 2021 due to reduction in coal use and growth in renewable power generation. The decrease in grid electricity CI directionally reduces the corn ethanol CI since electricity is used in different stages of corn ethanol production, which was not anticipated in the 2010 RIA with an overstatement of about 1000 g CO_2e/MJ ethanol.

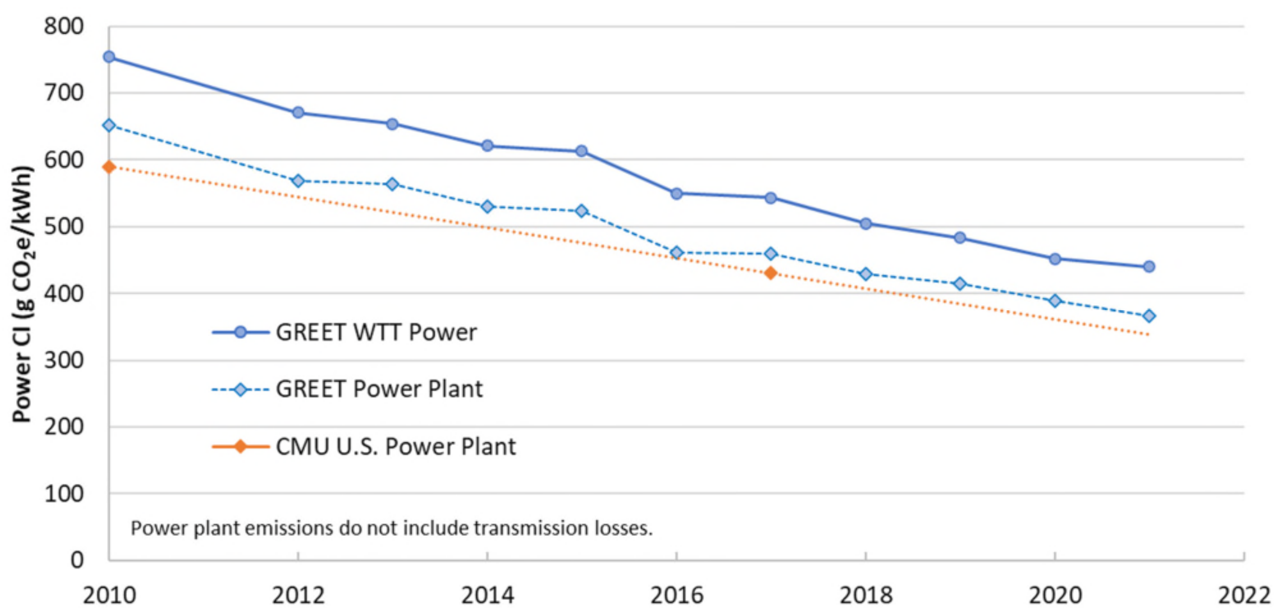

Figure 6.2. Carbon Intensity of Electric Power (U.S. Average).
(Power plant emissions do not include transmission losses, Source GREET)

A study at Carnegie Mellon University (CMU) examined the direct GHG emissions from the power sector in the U.S. and found that between 2001 and 2017 the average annual carbon intensity of electricity production in the U.S. decreased by 30%, from 630 g CO_2e/kWh to 439 gCO_2e/kWh (Schivley et al., 2018; EIA 2021). A similar proportional reduction in emissions occurred for power plants in the corn belt states where most ethanol plants are located (Figure 6.3). Schivley et al. (2018) used the U.S. Energy Information Administration (EIA) database to calculate aggregate GHG emissions and reports only power plant emissions[10]. The power plant emissions are consistent with the power plant component GREET. Based on both EIA and GREET, the CI of electricity is dropping. Note that the more recent EIA data shows a continuous downtrend in the GHG intensity of U.S. electric power.

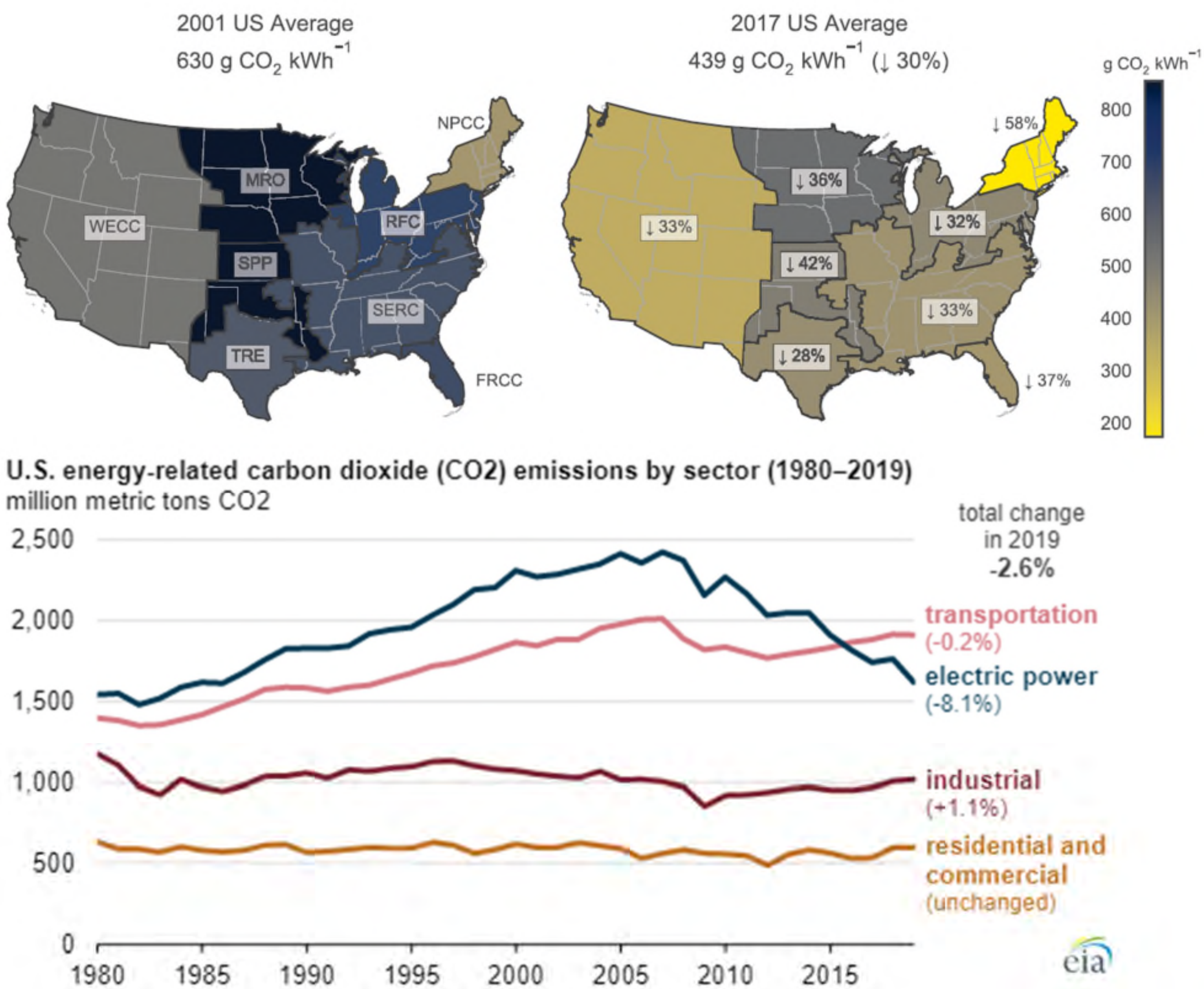

Figure 6.3. Change in Carbon Intensity of Electricity. (Schivley et al., 2018; EIA, 2021)

6.5.2 Renewable Power

Ethanol plants also have the opportunity to obtain lower GHG sources of electric power. Under current fuel policies, such as California's Low Carbon Fuel Standard, ethanol plants must use renewable power that is directly connected to the generation source. However, renewable power had contributed to the overall reduction in GHG emissions from the grid in the U.S.

[10] Power plant emissions at the plant from GREET correspond to the "fuel" phase × (1 – loss factor)

6.6 Summary of Ethanol GHG Analysis Issues

Many factors affect the CI of corn ethanol. A summary of the issues and recommended analysis method is shown in Table 6.2.

Table 6.2. Evaluation Issues related to GHG Analysis.

LCA parameter	Analysis Issue	Recommendation
Ethanol refinery energy efficiency has increased.	The energy efficiency has increased in a few refineries and it does not reflect the average.	Based on our analysis, the current energy usage at the fuel production stage is close to EPA RIA's estimate, however, both electricity and natural gas consumption have a declining trend which should be considered.
Electric power GHG intensity.	The GHG intensity of electric power has dropped faster than projected in the 2010 RIA.	Update electricity mix for electric power generation.
Emissions associated with gasoline is under estimated.	EISA requires that the EPA compare biofuel emissions to a 2005 petroleum baseline.	The 2005 petroleum bassline analysis excluded methane leakage and the thermal cracking of petroleum which has lead to underestimation of emissions associated with gasoline.
Co-product allocation method	EPA RIA used the replacement method which results in lower co-product credit.	Since corn oil is used as biodiesel feedstock (energy source) energy allocation is a better option which results in more reduction in corn ethanol CI.
Fertilizer use rate for soybean	EPA RIA used lower fertilizer use rate for soybean.	According to recent USDA statistics, the N fertilizer use rate in soybean is almost three times more than what EPA used. Higher fertilizer rate for soybean results in more co-product credit for DGS which replaces the soybean meal.

7. PETROLEUM BASELINE EMISSIONS FOR 2005 ARE LARGER THAN PROJECTED.

7.1 EPA 2010 RIA Approach in Estimation of Petroleum Baseline

EPA estimated the lifecycle GHG emissions associated with baseline gasoline transportation fuel using the 2009 analysis performed by the National Energy Technology Laboratory (NETL). The NETL analysis considers the GHG emissions associated with crude oil extraction both in the U.S. refineries and refineries in other countries from which the U.S. imported oil. The emissions from the 2010 RIA for 2005 gasoline fuel are shown in Table 7.1.

Table 7.1. Carbon Intensity of 2005 Gasoline from Well to Wheel (WTW).

	GHG Emissions (g /MMBtu)			
Life Cycle Step	CO_2	CH_4	N_2O	CO_2e
Fuel production	16,816	2,282	103	19,200
Tailpipe	77,278	3	5	78,891

EPA established the baseline RBOB (Reformulated gasoline Blendstock for Oxygen Blending) CI for gasoline at 93.08 g CO_2 e/MJ in the year 2005.[11] EPA has not re-examined the CI of petroleum since the 2010 RIA; however recent studies have shown that EPA underestimated the emissions associated with 2005 gasoline. The key factors analyzed by these studies include:

- Fugitive methane;
- Flaring of associated gas;
- Enhanced production methods including water flooding and thermal oil recovery;
- Mix of oil sands;
- Refinery complexity.

The key findings of recent studies which have more accurate data are discussed below.

7.2 New Findings on Petroleum Baseline

Researchers have studied the life cycle GHG emissions of petroleum fuels for several decades. Many of these studies follow the process for LCA defined by International standards (ISO 14040, 2006). Initial studies examined the national inventory of GHG emissions from crude oil production and refining with calculations of crude oil and fuel transport (Wang, 1999). Even though GHG emissions from oil refineries are reported as part of most national GHG reporting systems, the distribution of emissions among refined products has remained a challenge since multiple refinery units produce a range of products.

[11] California, in 2006, established a baseline CARBOB (California Reformulated gasoline Blendstock for Oxygen Blending) CI of 95.86 g CO_2 e/MJ. However, this value was updated to the 2012 value of 99.18 g CO_2 e/MJ to reflect the steady shift to higher intensity crude oils fed into U.S. refineries.

Aspects of crude oil production including flaring, indirect effects of road building, thermal enhanced oil recovery, and crude production methods were identified as key aspects of the life cycle of petroleum fuels (Unnasch et al., 2009; Keesom et al., 2009). Subsequent studies expanded the modeling methods and detail for crude oil production in regions such as the EU (Keesom et al., 2012; ICCT, 2014; COWI, 2015). More detailed models of crude oil production have also been developed by Jacobs Consultancy (Keesom et al., 2012) and Stanford University (El-Houjeiri et al., 2014). The California Air Resources Board (ARB) also publishes annual estimates of the CI of crude oil (CARB, 2019b). Regional studies of crude oil for the U.S., China, and globally are also part of the scientific literature (Cooney et al., 2016; Masnadi et al., 2018a; Masnadi et al., 2018b; Gordon et al., 2015).

The GHG LCA emissions associated with gasoline have been examined in numerous studies conducted by Jacobs Consultancy, Argonne National Laboratory, MathPro, and the University of Calgary (Keesom et al., 2012; Elgowainy et al., 2014; Kwasniewski et al., 2016; Rosenfeld et al., 2009, Abella and Bergerson, 2012). These studies show that a CI of 97 g/MJ would be more accurate than the 93 g/MJ for the 2005 baseline value estimated in the EPA 2010 RIA due to emissions associated with a range of crude oil production practices including oil sands upgrading, venting and flaring or produced gas, and enhanced oil recovery technologies.

The quality and consistency of the raw crude fed into refineries determines the complexity of processing required such that lower quality crude oil is more difficult to refine into transportation fuels, thus resulting in higher CI. The total energy expended to recover crude oil and the resulting GHG emissions vary depending upon the crude characteristics and the recovery methods used. The carbon intensities per production method were analyzed in a study that examined the CI of fuels under the RFS2 (Boland & Unnasch, 2014). The results for different petroleum fuels are shown in Table 7.2.

Table 7.2. Petroleum Gasoline Carbon Intensity.

Petroleum Source	**Gasoline Carbon Intensity (g CO_2 e/MJ)**		
	Low	**High**	**Average**
Primary	84.50	94.6	89.55
Secondary	93.58	98.18	95.88
TEOR	100.58	120.00	110.29
Stripper Wells	101.95	116.44	109.20
Mining Upgrader	100.42	104.91	102.67
SAGD, Dilbit	105.00	115.36	110.18
Fracking	97.48	111.54	104.51
Oil Shale	113.00	159.00	136.00

Conventional oil includes primary and secondary sources of oil and these are the most well defined and accessible sources of crude and hence the most drawn upon, the carbon intensity for gasoline from these crude oils ranges from approximately 84 to 98 g CO_2 e/MJ. TEOR (Thermally Enhanced Oil Recovery) methods are generally implemented where the crude characteristics (viscosity, API gravity) dictate and also to extend the life of a production well. Heating water to produce the steam or other *in-situ* TEOR techniques require additional energy inputs and can increase emissions by an additional 8 to 9% over conventional production. Compared to conventional oil deposits, oil sands require production techniques that are associated with greater environmental impacts. Shallow deposits are typically accessed using strip-mining techniques, while deeper deposits are generally accessed using in situ techniques whereby steam is injected into the reservoir to heat the bitumen until its viscosity decreases sufficiently to allow it to flow out of the reservoir. On a WTW basis, the GHG emissions from oil sands are generally between 5 to 15% higher than from most conventional oils. Heating water to produce the steam used for in situ techniques and bitumen-sand separation uses large amounts of energy, typically natural gas, and produces correspondingly large amounts of emissions. In addition, bitumen produced from tar sands must go through more extensive refining than conventional oil, producing additional emissions. Upgraded mining techniques have led to advances in emissions reductions by approximately 2% over other oil sands ranges. The emission ranges shown in Figure 7.1 show a range of crude oil types that were in production in 2005 and are higher than the baseline in the 2010 RIA.

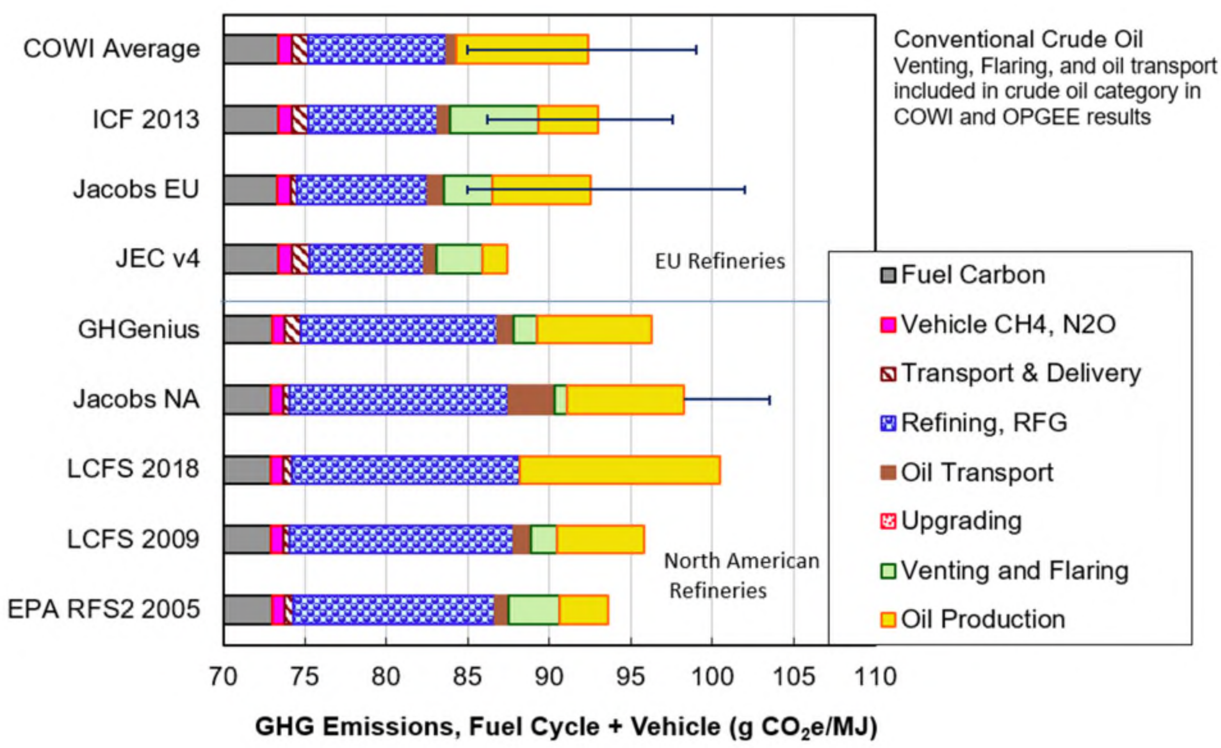

Figure 7.1. CI of Gasoline Estimated by Several Studies. (Unnasch et al., 2018)[12]

[12] The Jacobs EU, JCE v4, GHGenuis, Jacobs NA, LCFS 2018, LCFS 2009, and EPA RFS2 2005 were presented in Keesom et al. (2012), Edwards et al. (2012), S&T (2013), Keesom et al (2012), CARB (2018), CARB (2009), and EPA (2010), respectively.

8. ESTIMATED GHG EMISSIONS FROM CORN ETHANOL

This study evaluated EPA's 2010 LCA of corn ethanol and specifically focused on the emission categories with the highest impacts. Since 2010 when the RIA was conducted, more data have become available, LUC models have been revised several times and more realistic assumptions have been made. Ten years of research provides a better understanding of the impact of biofuel expansion on LUC both in the U.S. and across the globe. Also, the energy consumption in the fuel production stage has been improved continuously since 2010 which should be accounted for in EPA's GHG LCA. Another important factor are the co-product credits where the role of corn oil as biodiesel and the substitute value of soybean meal displacement was not fully reflected in the 2010 RIA. The main factors analyzed in this study are discussed below.

1. International LUC has the highest share from total emissions of corn ethanol in the RIA. Recent studies have estimated much lower values for international LUC compared to EPA RIA. In this study, uses the GREET (2021)/CCLUB, to calculate both domestic and international LUC. GREET uses the GTAP model which has undergone several rounds of revision since 2010 and GTAP's estimate of international LUC due to corn ethanol production is almost five times lower than what EPA RIA estimated. GTAP includes refinements in pasture utilization and projections of yield improvement reflected by elasticities (Taheripour, 2017).
2. Corn ethanol yield affects both domestic and international LUC. EPA projected a yield of 2.71 gal/bu, however, recent data shows that the ethanol yield in dry mill process is 2.88 gal/bu and continues to improve (GREET, 2021).
3. Energy consumption in the fuel production stage has improved due to the application of new technologies. EPA projected the natural gas consumption as the main source of energy for dry mill process with corn oil fractionation as 25,854 Btu/gal. Data from LCFS applications show a trend below 20,000 Btu/gal by 2022. Also, the CI of electricity used as a source of energy in biorefining has a declining trend due to the consumption of cleaner fuels in the production stage.
4. DGS, a byproduct of corn ethanol, is a partial substitute for soybean meal. Nitrogen fertilizer use in soybean farming has increased recently and reached 166 g/bu (USDA NASS, 2018). The RIA assumed a nitrogen fertilizer use rate for soybean of approximately 64 g/bu. Higher nitrogen fertilizer use rates increases the GHG intensity of soybean meal which results in a higher credit for the DGS co-product.
5. Corn oil is a co-product of corn ethanol that has achieved a high adoption rate. The 2010 RIA used the displacement method; however, the evolving use of corn oil is biomass-based diesel production (2021 Draft RIA, Figure 5.2.3-1). Therefore, energy allocation is an appropriate option since the growing use of corn oil is as an energy product. The net effect is a lower CI when both ethanol and biodiesel are treated as energy products.

This study uses the GREET (2021) model to calculate the CI of corn ethanol configured with current ethanol plant and crop data. Since GREET lacks some consequential aspects of corn ethanol LCA such as international rice methane emission and international livestock emissions, the analysis in the ICF study (Rosenfeld et al., 2018) provides the basis for these parameters in order to be consistent with the emissions categories in the 2010 RIA. The allocation treatment of corn oil biodiesel is factored into the analysis also as shown in Table 8.1.

The estimated GHG emissions represent a hybrid between the GREET and consequential LCA approach in the 2010 RIA. The allocation effect of corn oil as a biodiesel feedstock is taken into account with emissions allocated between ethanol and corn oil-based diesel. Note that the substitute value of corn oil is a small fraction of the DGS co-product and an acre of land that produces corn for ethanol makes as much animal feed as an acre of soy beans.

Table 8.1. CI of Corn Ethanol for Dry Mill, Natural Gas Operation with Corn Oil Extraction.

Emission Category	2005 Gasoline Revised	2005 Gasoline RIA	GREET 1999	EPA 2010 RIA	GREET 2021	This Study[a]	ICF
Domestic Livestock				-3,746	**-2,202**	-2,463	**-2,340**
Domestic Farm Inputs and Fertilizer N_2O			16,000	8,281	**11,548**	9,065	**11,023**
International Farm Inputs and Fertilizer N_2O				6,601	**-987**	**-1,013**	-1,013
Domestic Rice Methane				-209	578	578	578
Tailpipe	79,004	79,004	880	880	**2,420**	**2,483**	**2,359**
International Rice Methane				2,089	**3,795**	**3,894**	**3,700**
International Livestock				3,458	**-2,255**	-2,038	**-2,199**
Domestic Land Use Change				-4,033	**1,374**	3,432	**1,374**
Fuel and feedstock transport			5,000	4,265	**2,160**	**2,217**	**2,217**
International Land Use Change			30,000	31,797	**6,139**	9,082	**5,986**
Fuel Production	21,100	19,200	48,000	27,851	29,527	34,518	28,792
Net Emissions	**100,104**	**98,204**	**99,880**	**77,233**	**47,468**	**52,096**	**59,755**

[a] 95.2% allocation factor (fraction of ethanol output/ ethanol plus corn oil) applied to either **GREET** or **ICF** results as indicated in bold.

Natural gas consumption of 24,305 Btu/gal, LHV. International farming inputs are based on the ICF analysis even though the full burden of domestic corn farming is represented with the GREET inputs. Domestic and international rice methane and livestock emissions are based on the ICF values combined with the allocation factor. International and domestics land use change are based on the GREET result combined with the allocation factor. This study does not investigate categories including international farm inputs and fertilizer N2O, domestic and international rice methane emissions and international livestock emissions and relies on the ICF study estimates for these emission categories and are combined with the allocation factor for corn oil. Livestock emissions include two major factors, enteric fermentation, and manure management. It has been shown by several studies that replacing DGS with soybean meal reduces the enteric fermentation. The manure management emissions refer to emissions during collection, storage, transfer, and treatment of manure. While the replacement of DGS reduced the enteric fermentation in domestic livestock, it was not included in estimating the international livestock emissions in RIA analysis. Inclusion of reduction in enteric fermentation for international livestock would decrease the emissions associated with international livestock.

Figure 8.1 shows the estimated CI is 50,417 g CO_2e/MMBtu while 2010 RIA estimated the CI of corn ethanol as 77,233 g CO_2e/MMBtu. The GREET (2021) estimation of corn ethanol CI is the lowest since it does not account for international livestock and rice emissions. The emission estimates from the ICF analysis provide the basis for the analysis presented here. While in GREET (2021) a small percentage (~7%) of energy for fuel production is coming from burning coal, this analysis represents natural gas dry mill facilities, which are the new facilities incentivized by the RFS2 and does not attempt to examine the entire range of ethanol production technologies.

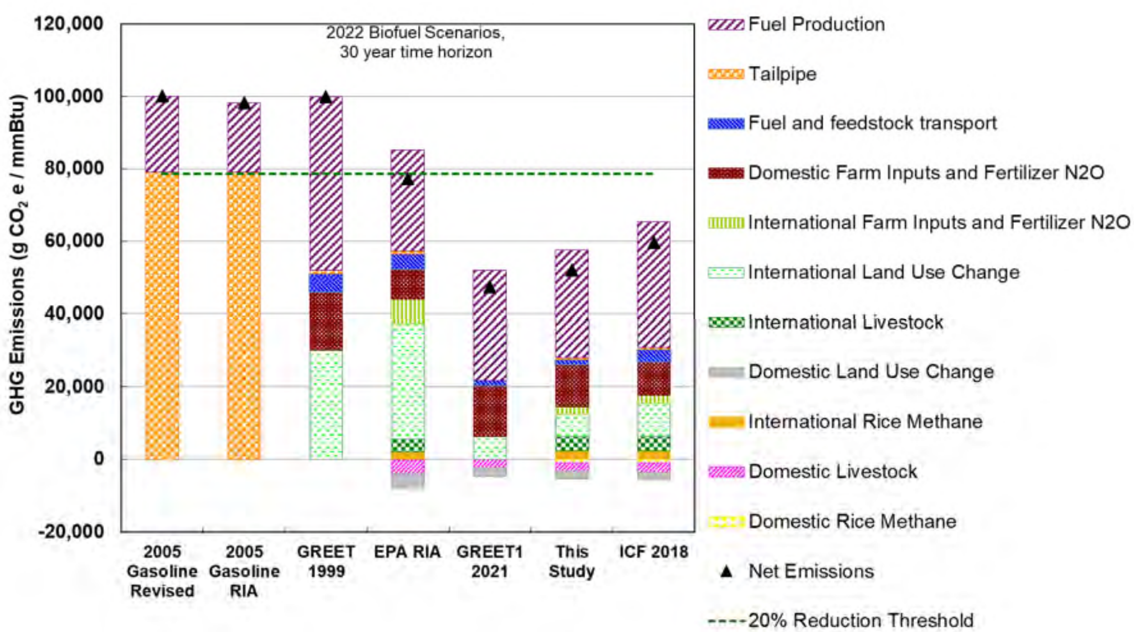

Figure 8.1. CI of Corn Ethanol for Dry Mill, Natural Gas Operation with Corn Oil Extraction.

Under the current situation and in the year 2022, Rosenfeld et al. (2018) calculated the CI of corn ethanol as 59,755 g CO_2e/MMBtu and 54,588 g CO_2e/MMBtu, respectively. Rosenfeld et al. (2018) also defined a scenario in which new technologies and better practices are employed to reduce the emissions in corn and fuel production. They concluded that by employing advanced technologies and introducing new co-products in the fuel production stage, and efficient management practices such as reduced tillage, nutrient management and cover crops in the farming stage the GHG emissions can be reduced to 27,852 g CO_2e/MMBtu. These estimates are consistent with ongoing trends in regenerative agriculture.

9. Conclusions

Life cycle GHG emission from the corn ethanol was analyzed over a range of production technologies and analysis methods. The data in this study show that life cycle GHG emissions for corn ethanol plants can range from 26 to 57 g CO_2e/MJ. Typical dry mill facilities have a CI in the 40 to 55 g CO_2e/MJ range. The CI for the 2005 petroleum baseline is also higher than originally projected; so, most of the ethanol plants in the U.S. produce fuel with a 45 to 55% reduction in GHG emissions. The key factors that result in GHG emissions that are lower than projected in the 2010 RIA include the following:

- Reduced energy consumption;
- Reduced GHG intensity for electric power;
- Shift from coal to natural gas fuel;
- Adoption of corn oil extraction with energy allocation;
- Reduced rates of deforestation;
- Improved rates of DGS use as animal feed;
- Displacement of ILUC and N_2O emissions from soy beans;
    - Higher nitrogen application rates to soybeans than originally modelled;
- Use of corn replacement feed from crop residue;
- Introduction of lower CI process fuels for ethanol plants;
- Higher GHG emissions from 2005 petroleum baseline fuels.

EPA overestimated international land use conversion in the 2010 RIA and has not updated the analysis in the draft 2021 RIA. New ILUC studies that take into account pasture intensification show a lower level of international ILUC and are represented in the CCLUB model from Argonne National Laboratory (ANL). The CCLUB model incorporates the most recent modeling from Purdue University's GTAP program. EPA also analyzed negative direct and indirect land use conversion emissions in the 2010 RIA. These results are confirmed in the CCLUB model from ANL and are consistent with the basic factors affecting the growth of corn ethanol production. Total agricultural land has not increased significantly in the U.S.

In addition, much of the growth in corn ethanol has come from a reduction in soybean production. Corn farming increases soil carbon relative to soy farming with no till practices and due to the fact that corn builds up soil carbon from its root mass. Criticisms of the CCLUB model based on the choice of the CENTURY emission factors associated with crop activity are misplaced as the emission factors based on Winrock and Woods Hole are simple approximations that are unsubstantiated. The CENTURY approach is used in the development of the U.S. emission inventory and is also consistent with regenerative agriculture practices that generate voluntary carbon credits.

In addition, EPA did not sufficiently document advancements in corn ethanol technology. Numerous ethanol plants are starting to use biogas and biomass fuel as well as implementing carbon capture and sequestration.

Finally, EPA understated the 2005 petroleum baseline and has not acknowledged the revised estimates of emissions in the 2021 draft RIA for this rulemaking. The refining of heavy oil as well as flaring emissions from many international sources of crude oil, which occurred in 2005 contribute to higher GHG emissions associated with gasoline than those in the 2010 RIA.

10. Appendix A – Nitrogen Application Rates

Nitrogen application rates affect the GHG intensity of corn production. In addition, ethanol plant DGS provides a replacement for crops with nitrogen application rates that are higher than anticipated in the 2010 RIA.

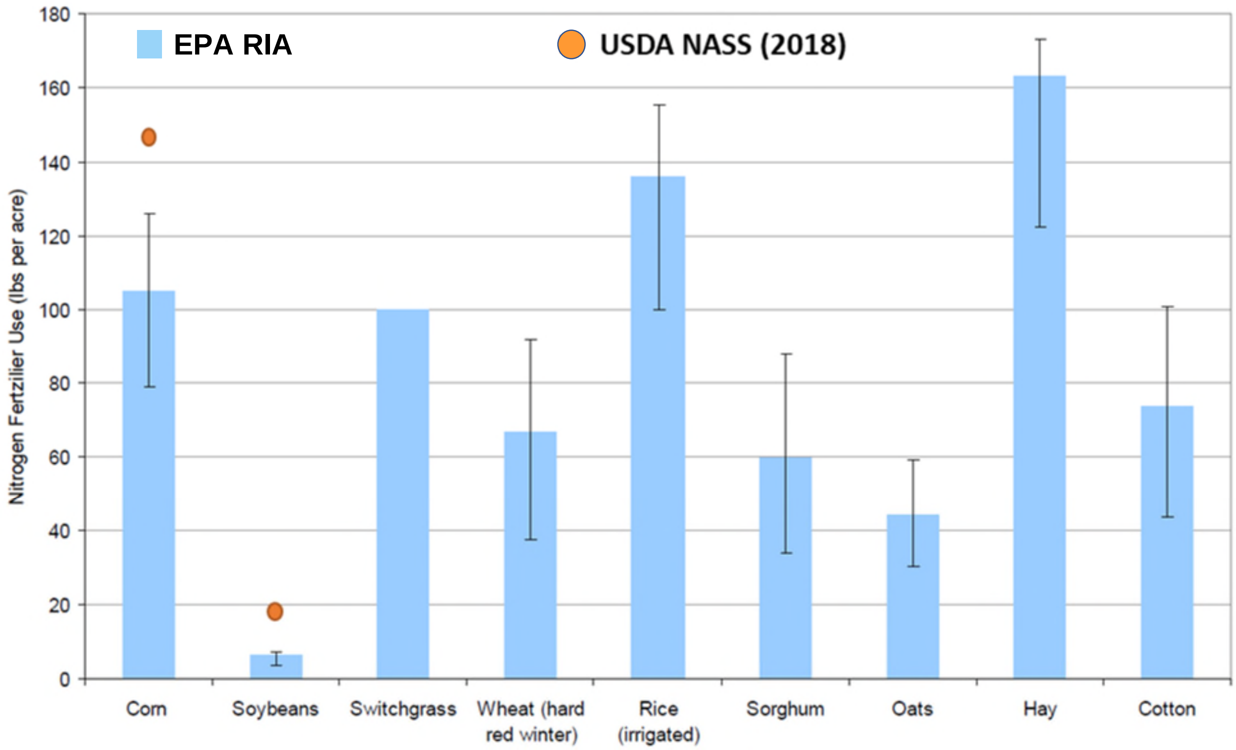

Figure A.1. FASOM Average Nitrogen Fertilizer Use by Crop. (EPA, 2010, not updated in EPA 2021)

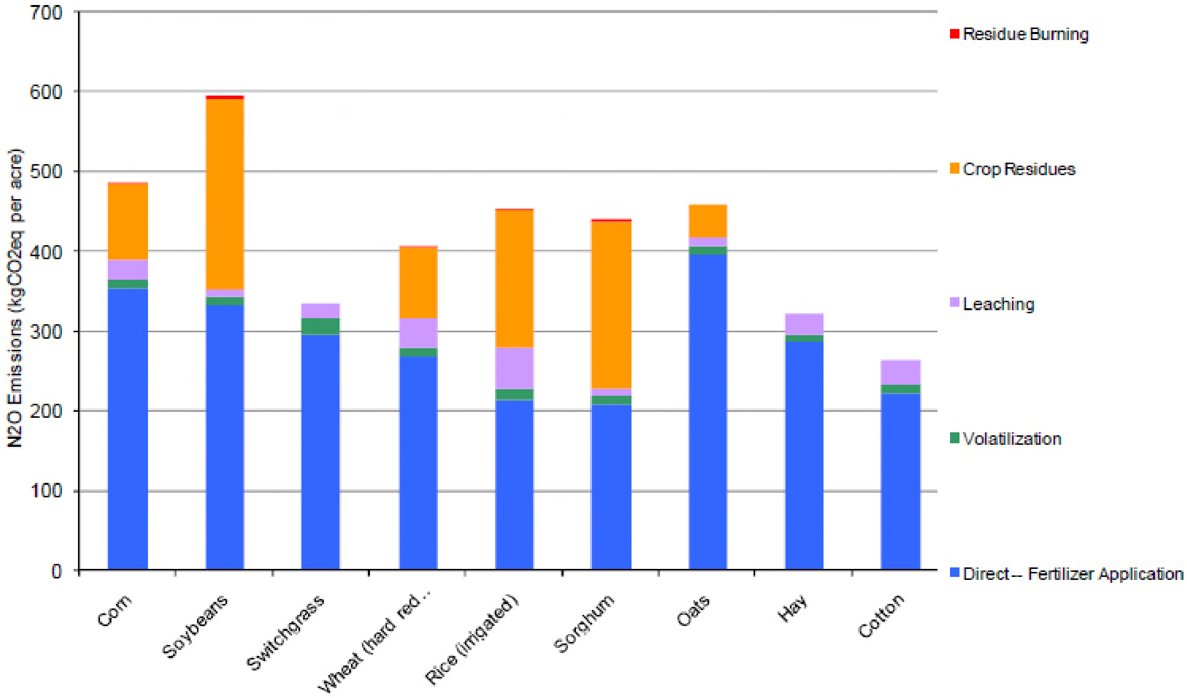

Figure A.2. N_2O emissions per acre from crop production.

Correcting the actual N fertilizer use in soybean farming, i.e., 166 g/bu, results in about a 460 g CO_2e/MMBtu of ethanol reduction in carbon intensity (CI) of corn ethanol with the soybean meal substitution rates in the GREET model.[13]

In summary: EPA attributed a certain amount of N fertilizer to soy production. DGS displaces soybeans that would otherwise be used as animal feed. Soybeans are more energy intense to grow than considered in the 2010 RIA and this displacement credit should be taken into account. The displacement value of DGS may be understated in the 2010 RIA also.

The literature review presented below examines the discrepancy between USDA NASS database and GREET on nitrogen fertilizer use in soybean farming. Soybean, which is an annual legume, requires a high amount of nitrogen (~5 lb of N per each bushel). However, 50 to 60% of the required nitrogen is supplied through the N-fixation process, which is a result of a symbiotic relationship between the plant and soil bacteria (Nafziger, 2014). The nitrogen fixation process consumes about 10% of the soybean's energy in the form of sugars produced by photosynthesis. According to Nafziger (2014), "at high yield levels, the crop might not be able to produce enough sugars to go around, and that either yield will suffer, or N fixation will be reduced." One of the methods to overcome this issue is to add nitrogen fertilizer in the growing season of soybean. Several studies have investigated the impact of nitrogen fertilizer application rate on soybean yield (Mourtzinis et al., 2018; La Menza et al., 2017; Schmidt, 2016. Mourtzinis et al. (2018) conducted one of the most comprehensive studies on soybean yield

[13] (166 – 48) lb/bu ÷60 lb/bu. 0.307 lb SBM displaced per lb DDGS, 3.78 g CO_2e/g N fertilizer, 0.0153 g N_2O/g N.

response to N fertilizer in the U.S. which included 207 environments (experiment × year combinations) for a total of 5991 N-treated soybean yields. While this study reported that the soybean yield increased by an increase in N fertilizer application, in most individual environments, the effect of a greater N-rate on soybean yield was not significant.

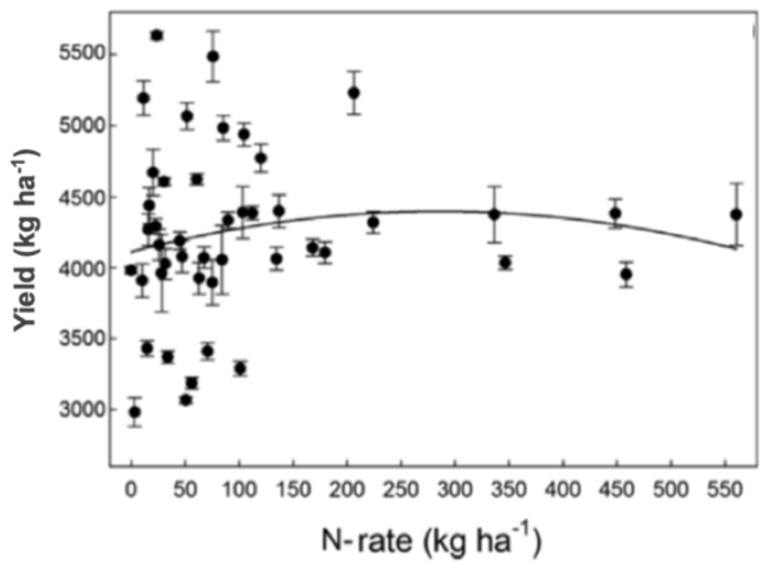

Figure A.3. Effect of Nitrogen Application Rate on Soybean Yield. (Mourtzinis et al., 2018)

While there was a large yield variability among environments within the same N rates, Mourtzinis et al. (2018) generated a second-degree N polynomial function that was significant (p = 0.0297), and it estimated the nitrogen rate of 340 kg ha^{-1} for maximization of soybean yield. This rate translates to 1.8 kg N per bushel of soybean (Figure A.**3**). Similarly, Nafziger (2014) studied the impact of nitrogen fertilizer on soybean yield over several years and concluded that soybean yields response to N fertilizer ranged widely among the trials.

In another study, La Menza et al. (2017) tested the hypothesis that indigenous nitrogen sources (N fixation and soil mineralization) are insufficient to meet crop N requirements for high yields. For this purpose, they developed a protocol to ensure an ample N supply during the entire crop season. They reported that soybean yield under ample N was 11% higher than the zero-N condition. Based on the literature review, we can conclude that adding N fertilizer to soybeans to achieve higher yields is gaining more attention, however, there is no clear trend between N application rate and soybean yields. There are several other factors which can affect the soybean yield such as planting date, N application timing, irrigation, etc. which need further studies. The higher emissions associated with soybean meal have been included in the more recent versions of GREET.

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Growth Energy Comments on EPA's
Proposed Renewable Fuel Standard Program:
Renewable Fuel Standard Annual Rules

Docket # EPA-HQ-OAR-2021-0324

Exhibit 3

Net Gain
ECOLOGICAL SERVICES

ANALYSIS OF **EPA'S PROPOSED RULEMAKING FOR** 2020, 2021, and 2022 RVOs REGARDING LAND USE CHANGE, WETLANDS, ECOSYSTEMS, WILDLIFE HABITAT, WATER RESOURCE AVAILABILITY, and WATER QUALITY

Prepared For: Growth Energy

Date: February 3, 2022

Author: Pieter Booth, Principal[1]

Net Gain Ecological Services

This memorandum provides Net Gain's comments and observations regarding selected technical issues associated with EPA's Proposed Rule for the Renewable Fuel Standard (RFS) Program Rules for 2020, 2021, and 2022 Renewable Volume Obligations (RVOs) (the Proposed Rule; EPA 2021a) and the associated Draft Regulatory Impact Analysis (RIA; EPA 2021b). The Proposed Rule and the RIA rely heavily on EPA's Second Triennial Report (EPA 2018). Therefore, this memo updates and builds upon previous findings and conclusions presented in the following reports[2] attached as exhibits:

- Ramboll. August 18, 2019. The RFS and ethanol production: Lack of proven impacts to land and water. Prepared for Growth Energy. Ramboll, Seattle, WA. (Exhibit 1).
- Ramboll. November 29, 2019. Memorandum: Supplemental analysis regarding allegations of potential impacts of the RFS on species listed under the Endangered Species Act. Prepared for Growth Energy. Ramboll, Seattle WA. (Exhibit 2).[3]

These prior analyses addressed the absence of a demonstrated causal nexus between the RFS and land use change (LUC); adverse impacts to wetlands, ecosystems, and wildlife habitat; and adverse impacts to water resource availability and water quality. Our analyses refuted claims by other investigators that the RFS causes quantifiable adverse impacts to environmental media. We have evaluated more recent scientific literature on this topic and continue to find that there is no evidence the RFS program causes these adverse environmental impacts. Based on this finding, there is no evidence that the Proposed Rule will result in land conversion or cause adverse impacts to wetlands, ecosystems, wildlife habitat, water availability and water quality. We encourage EPA to update its analysis in the RIA to address these findings and correct its potentially misleading discussion of environmental impacts of the program.

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[1] Mr. Booth has over 30 years of experience as an environmental scientist specializing in environmental risk assessment and restoration, natural resource damage assessment, environmental due diligence, and policy analysis. He has published 19 articles and presented his work 43 times at national and international conferences. He has acted as consulting expert on over a dozen environmental damages cases in the U.S. and has been retained as a testifying expert on three international environmental damages cases.

[2] Mr. Booth was lead author on both referenced reports.

[3] Submitted by Growth Energy as "ESA Comments – Attachment B," Docket # EPA-HQ-OAR-2019-0136 Supplemental Notice of Proposed Rulemaking; Renewable Fuel Standards Program: Standards for 2020 and Biomass-Based Diesel Volume for 2021, and Response to the Remand of the 2016 Standards.

EXECUTIVE SUMMARY

The RIA presents only a generalized discussion of the drivers for and nature of potential impacts of biofuel feedstock production and biofuel refining on LUC; wetlands, ecosystems, and wildlife habitat; and water availability and water quality. It fails to recognize the complex causal links between drivers of impacts and the potential impacts described to land and water, which in turn creates the misleading impression that there is a causal relationship between the RFS and impacts to land and water, where no such relationship has been established.

Conversion of Wetlands, Ecosystems, and Wildlife Habitats: The Role of Land Use Change

Crop extensification resulting in LUC is the factor that has garnered the most attention in assessing the potential impacts to land and water from the RFS. The causal chain linking the RFS and LUC consists of a myriad of complex interactions among economic, biophysical, and social factors that are very challenging to model. In its Second Triennial Report and its Endangered Species Act No Effect Finding for the 2020 Final Rule (No Effects Finding) (EPA 2019), EPA has recognized this complexity and the large amount of uncertainty in establishing causation between the RFS and LUC, but EPA fails to adequately consider the implications of this uncertainty and absence of evidence in its conclusions in the RIA. The RIA cites studies such as Lark et al. (2015) and Wright et al. (2017) in its discussion of the RFS and LUC. More recent publications have concluded the findings of Wright et al. (2017) and Lark et al. (2015) were flawed and based on inaccurate data. In addition, EPA does not adequately consider that these studies fail to establish a causal link between the RFS and their reported results. Other recent publications similarly do not find a quantitative causal relationship between the RFS and LUC, with studies confirming our prior findings that the work of Lark et al. (2015) and Wright et al. (2017) is unreliable. The RIA should be updated to acknowledge the shortcoming of such studies and address the more recent literature.

Further, EPA largely ignores several important factors in play that negate or mitigate potential impacts of the RFS on land and water; these include:

- Continued improvement in crop yield satisfies increased demand for corn without the need for extensification and LUC.
- Cropping practices and other practices at the farm level such as conservation tilling, and vegetative buffers minimize impacts to soil, surface water, and groundwater.
- Production of Distillers Dried Grains with Solubles (DDGS) offsets a considerable amount of demand for corn and soy as animal feed.
- Adoption of more efficient irrigation methods and advanced farming technologies minimize use of irrigation water, pesticides, and fertilizers.

EPA should update the RIA to address the complex economic and biophysical links in the causal chain associating the RFS with impacts to land and water. EPA should address each important link in the causal chain, including data gaps and lack of any evidence substantiating one or more of the causal links in the chain. EPA should also address the mitigating factors set forth above.

EPA's consideration of impacts to wetlands, ecosystems, and wildlife habitat in the Second Triennial Report and the RIA consists almost entirely of general descriptions of data on nationwide wetlands losses, conversion of grassland habitat, discussions of waterfowl habitat loss, potential impacts to aquatic habitats, and potential impacts of grassland conversion to agriculture on insect pollinators. In both the Second Triennial Report and in the RIA, EPA acknowledges the uncertainty of efforts to quantify a relationship between the RFS and wetland and grassland habitat losses, yet this is not adequately reflected in its conclusions. For example, the RIA includes the following statements relative to the proposed volumes for 2022:

- There is a possibility that the proposed volumes for 2022 may "inspire" an increase in feedstock production which in turn may affect wetlands (page 91).
- There is a potential to "incent" additional production of biofuels that in turn, may affect grasslands and other ecosystems (page 96).

Given the magnitude of the uncertainties described by EPA and others in establishing causation between the RFS and such impacts, and considering the myriad of economic, biophysical, and social links in the causal chain, the "possibility to inspire" feedstock production or "incent" biofuel production appears to be vanishingly small. This should be given due consideration in the RIA.

Water Quantity and Water Quality

As with the discussion of LUC, the RIA presents lengthy discussions of impacts associated with agriculture in general (including biofuel feedstocks) and water quantity and water quality; however, the RIA fails to adequately describe the complex economic and biophysical links in the causal chain associating these impacts with the RFS. EPA should reevaluate the discussions of generic impacts as well as address the causal chain as set forth above. In addition, to the extent EPA discusses the hypoxic zones in western Lake Erie and the Gulf of Mexico it must explain that there has been no quantitative attribution of these water quality impacts to the RFS and that any such attribution is conjecture. We also recommend that EPA enhance the discussion of technological improvements in agriculture that reduce water and agrichemical use.

THE RFS AND LAND USE CHANGE

The Draft RIA has three critical failings in its discussion of the potential role of the RFS on LUC:

- It relies on flawed studies that underlie the Second Triennial Report without due consideration of more recent work that shows there is no demonstrated causal link between the RFS and LUC, as EPA itself has acknowledged since it prepared the Second Triennial Report.
- It fails to consider the high degree of uncertainty in the causal relationship between the RFS and biofuels prices, which is a fundamental assumption underlying assessments of the RFS, and it fails to acknowledge that such a relationship has not been adequately quantified.
- It fails to address important mitigating factors including the continuing increase in crop yield, adoption of conservation farming practices and modern farming technology, and production of DDGS.

These shortcomings are discussed below.

Reliance on Flawed Research and No Established Causal Relationship

Some investigators have asserted that the RFS has resulted in extensive conversion of non-agricultural land to agriculture due to increased demand for corn for ethanol. Our findings indicate that these claims are not borne out, in part because the studies use unreliable databases, present flawed data analysis, and/or do not attempt to establish a causal link between the RFS, increased ethanol production, and LUC. Indeed, EPA (2019) repeatedly asserts that no causal connection has been established between LUC associated with corn production and the RFS.

As background, in the discussion of potential impacts of the RFS on LUC in its Second Triennial Report, EPA repeatedly cites geospatial analysis conducted by the following researchers who used the Crop Data Layer (CDL)[4] dataset:

- Lark et al. (2015) analyzed LUC nationwide during the period 2008-2012 using CDL calibrated with ground-based data from USDA's Farm Service Agency (FSA), and data from the National Land Cover Database (https://www.mrlc.gov/national-land-cover-database-nlcd-2016). The authors reported that 7.34 million acres of previously uncultivated lands became utilized in crop production and of those 1.94 million acres (785,000 ha.) of converted lands were planted in corn as a first crop.
- Wright et al. (2017) assessed quantitative spatial relationships between the loss of grasslands and the locations of ethanol refineries with the intent of associating this LUC with demand for ethanol. Wright et al. (2017) note that approximately 2 million acres of grassland was converted to row crops within 50 miles of an ethanol refinery between 2008 and 2012.

Several investigators have shown that reliance on inadequately corrected and verified CDL data leads to an unacceptable level of uncertainty in geospatial analysis and potentially misleading results and conclusions from such analysis. For example, Dunn et al. (2017) examined data for 2006-2014 in 20 counties in the prairie potholes region using the CDL, a modified CDL dataset, data from the National Agricultural Imagery Program, and ground-truthing. Dunn et al. (2017) concluded that analyses

[4] The CDL data set is developed by the U.S. Department of Agriculture National Agricultural Statistics Service (https://www.nass.usda.gov/Research_and_Science/Cropland/SARS1a.php).

relying on CDL data returned the largest amount of LUC by a wide margin. They further concluded that errors associated with CDL-based analyses resulted in estimates where "the number of hectares in the potential error associated with CDL-derived results is generally greater than the number of hectares the CDL-based analysis determined had undergone a transition from grassland, forested land, or wetland to agricultural land." This suggests that errors in classification inherent in the CDL can result in uncertainty bounds that are larger than the estimates themselves (thereby even predicting negative land conversion to agriculture).

Specifically, Dunn et al. (2017) pointed out that the findings reported by Lark et al. (2015) contradict USDA data indicating that cropland area has remained almost constant during the period 2008-2012. Similar critiques have been published of the findings reported by Wright et al. (2017) using the CDL data set. The RIA does not acknowledge a major shortcoming of the study by Wright et al. (2017), namely, the authors' admission that their study "did not consider potential effects of other explanatory variables." The paper also discussed the errors in the data itself, stating that the "conversion of non-cropland to cropland was mapped correctly over 70% of the time," which means that it was mapped incorrectly 30% of the time, a considerable percentage.

Several other recent papers challenge the findings of previous authors who have implicated the RFS in LUC. Of note are papers by Pates and Hendricks (2019), Pristola and Pearson (2019), and Shrestha et al. (2019). Pates and Hendricks (2019) assessed the local impact of ethanol plants on cropland transitions and concluded that ethanol plant expansions *reduce* the probability of cropland conversion by 0.5% on average and that fields near ethanol plants were 10% *less* likely to convert non-agricultural land into cropland than fields farther away. The authors acknowledge that this result contradicts their underlying premise that ethanol plants induce LUC through local effect on corn prices. They speculate (without providing evidence) that this contradiction may result from bias due to concurrent changes in Crop Reserve Program (CRP) policy that disproportionately affected areas near ethanol plants.

Pristola and Pearson (2019) performed a critical review of literature that was relied on by EPA in its Second Triennial Report (and again in the RIA) regarding the RFS and LUC and concluded that major flaws in the work by Wright and Wimberly (2013), Lark et al. (2015), and Wright et al. (2017) render the work by these authors unreliable. Their major findings are as follows:

- All three studies reviewed relied on data from the CDL which has several shortcomings including the inability to differentiate between native prairie, CRP, grass/hay, grass/pasture, and fallow/idle grassland types, especially in earlier years.
- Improvements in the CDL over time make it problematic to compare land cover and land use over relatively long time periods. Thus, results reported in the three studies might be biased due to the CDL's ability to better identify cropland in later years than earlier years. This bias would give the appearance that cropland expanded, as these authors assert.
- All three studies reported cropland expansion over the conterminous United States, but this is contradicted by data from the NASS that show a contraction of cropland from 2008 to 2012, and that by 2017, cropland acres were below 2007 levels.

Both Lark et al. (2015) and Wright et al. (2017) relied on CDL data for Iowa for 2008 and 2012. Data from NASS revealed that during the period 2008-2012 in Iowa there was a net increase of cropland of

38,000 acres as compared to an increase of 263,468 acres as reported by Lark et al. (2015) and 295,100 acres as reported by Wright et al. (2017).

Shrestha et al. (2019) studied the relationship between biofuel demand, food prices, and LUC. One of the authors' objectives was to assess the accuracy of automated land use classification as performed by previous investigators (including Lark et al. 2015) as compared to manual land use classification techniques. For this analysis, the authors selected study areas within three counties near Moscow, Idaho with a total land area of 664 km^2. The areas were selected to represent a range of climates and proportions of land cover types. Their work revealed that 10.90% of non-agricultural land was misclassified as agriculture, whereas only 2.23% of agricultural land was misclassified as non-agricultural. The automated classification showed an 8.53% increase in agricultural land from 2011 to 2015, while the manual classification showed only a 0.31% (±1.92%) increase. The result derived via manual classification was within the margin of error suggesting that there was no significant LUC during the period.

These recent findings further call into question EPA's continued reliance on flawed studies in its discussion of the RFS and LUC, and indeed call into question whether there is any quantifiable causal link between RFS and LUC.[5]

Inadequate Consideration of the Drivers of Biofuels Feedstock Prices

Studies that attempt to link the RFS with impacts to land and water (especially those studies focusing on LUC) include a foundational presumption that there is a causal link between the RFS and biofuel feedstock prices. Econometric models used to quantify the relationship have a high degree of uncertainty, partly because agricultural commodities are traded on international markets and their production is affected by highly uncertain and seasonally variable weather conditions. The RIA acknowledges this uncertainty by stating that "…models that attempt to project prices at specific times in the future, or in reaction to specific demand perturbations, necessarily contain high levels of uncertainty" (page 209). The RIA goes on to discuss the relationship between grain stores and futures prices and how annual volumes of grain stores depend on current year harvests and future year harvest projections. The RIA provides no discussion of the relationship, if any, between grain stores and the RFS. The RIA acknowledges that, in a general sense, grain prices are influenced by "an array of factors from worldwide weather patterns to biofuel policies to international tariffs and trade wars" (page 209). Finally, the RIA presents results from a meta-analysis of the impact of increased biofuel production on corn prices.[6] Based on the results of the single meta-analysis by Condon et al. (2013) conducted almost a decade ago the RIA projects that the proposed ethanol volumes for 2021-2022 will increase the price of corn 3% per billion gallons, or $0.11per bushel.

[5] In addition, several publications released since August 2019 reported on LUC such as conversion of grassland to corn and soy (e.g., Zhang et al. 2021; Lark et al. 2019b; and Arora and Wolter 2018) and cropland expansion and potential wildlife impacts (Lark et al. 2020); however, these studies do not attempt to establish causal linkages between increased demand for ethanol from the RFS and LUC. Rather, these articles and others reveal a trend among researchers toward improving the accuracy of geospatial modeling to discern specific LUC which appears to be a shift from previous efforts to associate LUC with the RFS. In addition, several authors focused on assessing the environmental benefits of improved agricultural practices, conservation, and restoration, and policy actions to reduce grassland losses (e.g., Fargione et al. 2018; Lark 2020; and Runge et al. 2017).

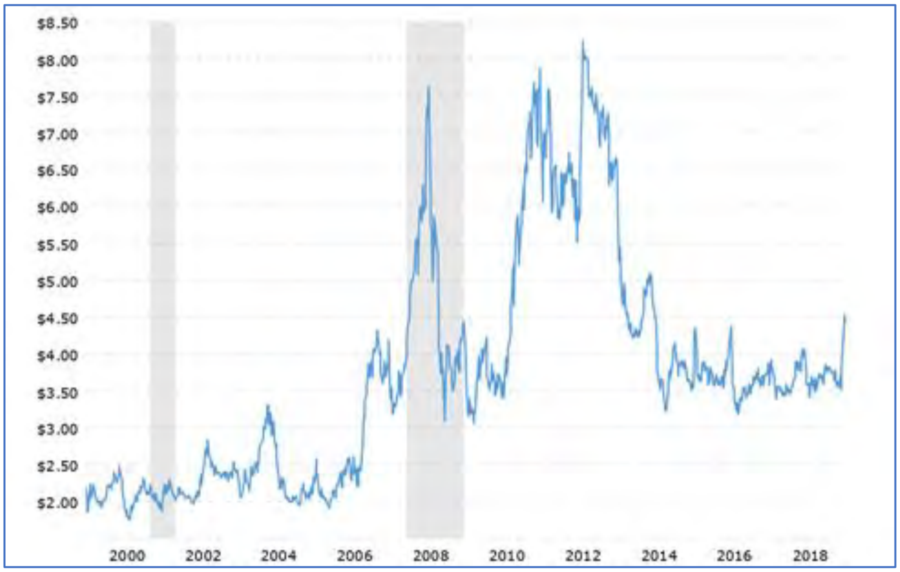
[6] EPA states that Condon et al. reviewed published papers in 2015, when in fact, the working paper was released in 2013 and reviewed papers published between 2008 and 2013.

Despite the acknowledgment of the high degree of uncertainty in econometric models attempting to link the RFS to biofuel feedstock prices, the RIA presents no discussion of such uncertainty and rather proposes the projected price increases estimated by Condon et al. (2013) as fact. For example, the RIA presents no information on whether the meta-analysis conducted by Condon et al. (2013) controlled for how and to what extent corn prices were affected by rapid economic growth in developing countries leading to growing food demand or how corn prices were affected by a dietary transition from cereals toward more animal protein. As a result of these market factors alone, global consumption of agricultural commodities has been growing rapidly. Further, the temporal fluctuation in corn prices is highly influenced by the effect of the price of oil on production inputs such as agrichemicals and fuel for farm equipment, and this relationship is not mentioned by EPA in either the Second Triennial Report or the RIA. Also significant is a study by Shrestha et al. (2019) that analyzed food price inflation and land use classification and concluded that food price inflation since 1973 was lowest during the biofuel boom years of 1991-2016 and was most highly correlated with the price of oil.

Figures 1 and 2 show nominal prices of West Texas Intermediate crude and corn for the latest 20-year period (the shaded areas on the graphs show period of US recessions) and demonstrates that corn prices track very closely to the price of oil.

Figure 1. West Texas Intermediate Crude Price ($/barrel).

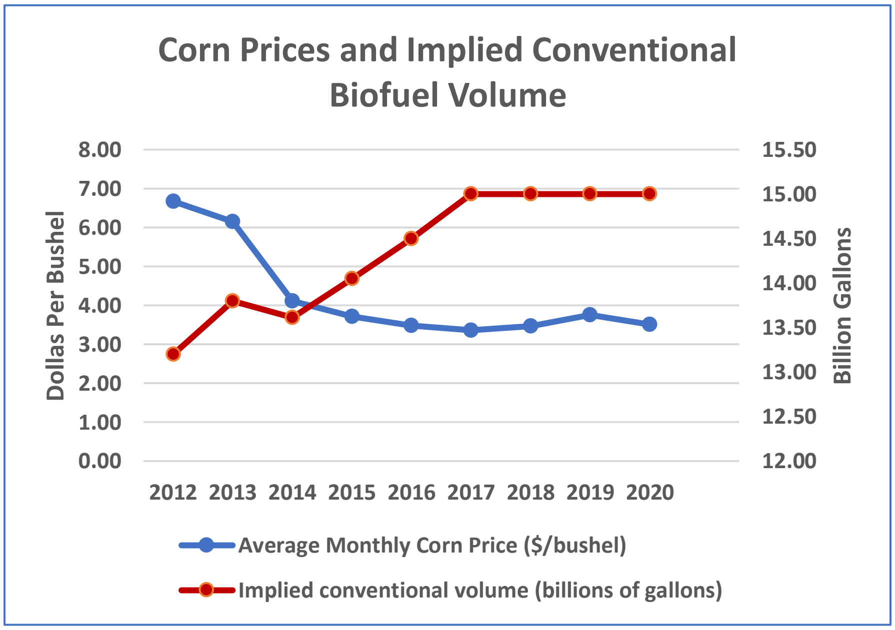
Figure 2. U.S. Corn Price ($/bushel).



SOURCE: Macrotrends n.d.

By contrast to the close relationship between U.S. corn prices and the price of crude oil, Figure 3 shows a plot of U.S. average annual corn prices and the implied ethanol volumes 2012-2020 showing no apparent relationship.

Figure 3. Corn prices versus implied conventional ethanol volumes 2012-2020.



SOURCES: https://www.nass.usda.gov/Charts_and_Maps/graphics/data/pricecn.txt and Congressional Research Service (2020).

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ECOLOGICAL SERVICES

Inadequate Consideration of Mitigating Factors

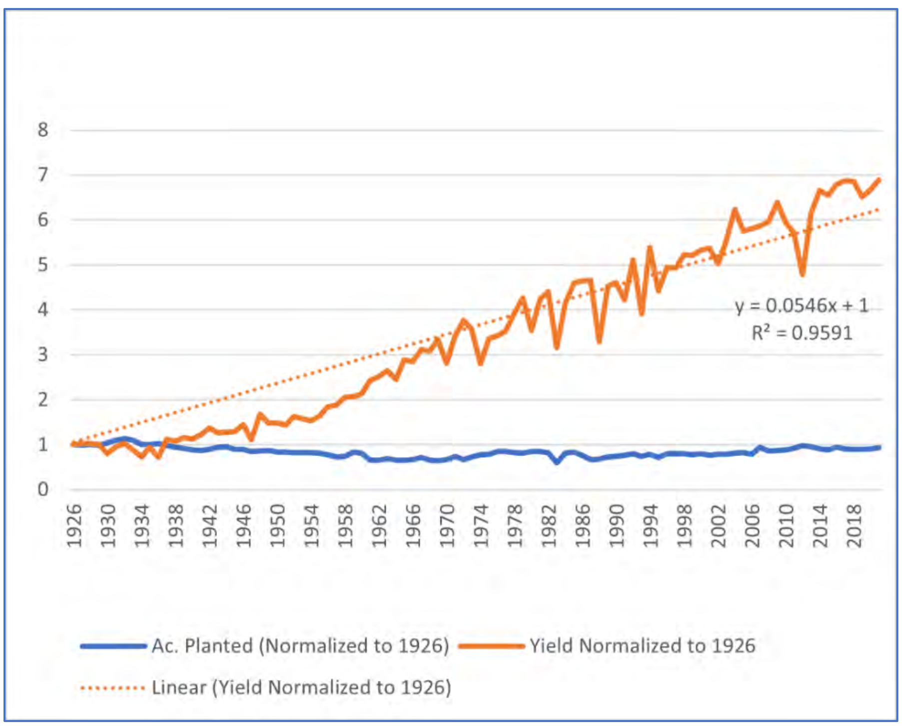
The RIA, like the Second Triennial Report, fails to adequately consider important factors that act to dampen any effect of the RFS on converting non-agricultural land to agriculture for ethanol feedstock, including:

- Increased demand for corn for all uses has mostly, if not fully been met by increased yields.
- Cropping practices such as double cropping increase production with no additional need for land in cultivation.
- Production of dried distillers' grains with solubles (DDGS) by ethanol refineries offset a considerable amount of demand for land to grow corn and soy for animal feed.

Increases in Corn Crop Yield

EPA has not adequately accounted for the fact that increased demand for corn for ethanol and the effect of this increased demand on land conversion, if any, has been offset by increases in yield over time. In fact, the number of acres planted in corn across the United States in the last couple of decades has remained close to or below the total acres planted in the 1930s, despite increases in demand for corn as human food, animal feed, and biofuels over this nearly 90-year period. The increase in demand has largely been met by an approximately 7-fold increase in yield (bushels per acre) (Figure 4). The USDA further anticipates that changes in corn production will result in an increase of approximately 16.1 more bushels per acre by 2028 without a substantial increase in farmed acreage.

Figure 4. Relative Change in Acres of Corn Planted and Yield (1926-2021)



Cropping Practices

Intensification refers to increasing the production of a crop on the same acreage of land and does not directly result in LUC. Extensification refers to increasing production of a crop by planting on land not previously in agriculture. A farmer can intensify production of a crop by switching crop types to a more desired crop or by double planting a single crop (double-cropping) instead of seasonally rotating crops. EPA's Second Triennial Report acknowledges the potential significance of cropping practices in meeting any increased demand for corn due to the RFS. By contrast, the RIA does not mention cropping practices in this context, rather it discusses double cropping only briefly and only as it relates to potential effects on water quality. In its Second Triennial Report, EPA cites a study by Ren et al. (2016) in Iowa (the state with the largest corn production) that examined changes in corn and soybean rotations around 2017 and found that 59% of the area that had been in corn/soy rotation prior to 2007 was in two or more years of continuous corn after 2007. However, EPA fails to acknowledge the most important conclusion related to LUC from this study: "… it is clear that the expansion of corn production after 2007 was realized by altering crop rotation patterns" (Ren et al. 2016).

Further, the RIA mentions the findings reported by Plourde at al. (2013) regarding farmers switching from corn/soy rotation to double cropping of corn as a means of increasing corn production but fails to acknowledge its significance in the context of intensification in lieu of extensification to meet demand. Rather, Plourde et al. (2013) is mentioned in terms of potential effects on nitrogen and phosphorous loads to surface waters.

Use of Distillers Grains and Solubles as a substitute for Corn and Soy in Animal Feed

EPA has not adequately accounted for the fact that the ethanol industry produces large amounts of distiller's grains with solubles (DDGS) and that this biproduct is used as animal feed where it substitutes for traditional grains such as corn and soy. EPA's Second Triennial Report acknowledges the role of DDGS in offsetting overall demand for corn and soy; however, this fact is ignored in the RIA.[7] Production of DDGS and its use as a substitute for corn and soy in animal feed likely has a very important mitigating effect on any potential contribution of the RFS to LUC. This effect can be estimated based on the annual volume of corn grown for ethanol (bushels), the annual figures for yield (bushels per acre) and the following assumptions:

- 17 lbs of DGGS are produced per bushel of corn processed[8]
- 1 lb of DGGS is equivalent to 1.22 lbs of corn/soy (Hoffman and Baker 2021)
- One bushel of corn/soy weighs on average 58 lbs

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[7] The Second Triennial Report states that approximately 12% of the total corn production from 2014-2016 was returned to the feed market in the form of DDGS. The Report also acknowledges a study by Mumm et al. (2014) that concludes that 40% of corn grown in 2011 was estimated to be utilized in ethanol production; however, when the offsetting effect of DDGS is accounted for, the acreage devoted to corn for ethanol goes down to 25%. The report also does not acknowledge that these same authors estimate that the percentage of land devoted to corn for ethanol will drop further to 13% by 2026 due to technological advances increasing crop yield as well as increasing the efficiency of the ethanol distillation process. It is curious that the RIA mentions this study as well, but only in the context of water quality.

[8] Explaining Fluctuations in DDG Prices - Center for Commercial Agriculture (purdue.edu) (accessed 1/5/2022).

Applying these assumptions to the annual amount of corn grown for ethanol and annual yield during the period 2008 to 2020, production of DDGS is estimated to have offset corn/soy equivalent acreage ranging from 8.7 million acres in 2008 to 13.5 million acres in 2012 with an annual average of 11.2 million acres over the period[9].

[9] Figures on corn displaced by DDGS 2010-2020 from World of Corn 2021. By comparison, the U.S. Corn Growers Association estimates 6.0-8.6 million acres per year over the period 2010-2020 with an average of 6.9 million acres per year but it is not clear how its estimate was calculated.

The RFS AND CONVERSION OF WETLANDS, ECOSYSTEMS, AND WILDLIFE HABITAT

The introduction to this section in the RIA simply repeats information presented and conclusions made in the Second Triennial Report regarding land use change. Without providing any supporting information, the introductory paragraph to this section wrongly concludes that "Evidence from observations of land use change suggests that some of this increase in acreage and crop use is a consequence of increased biofuel production."

In the discussion of wetlands, the RIA presents information in reports from several federal agencies that describe the status and trends of U.S. wetlands. These reports and sources of data merely record changes in wetland types nationwide and do not provide any analysis of the cause of the changes that would be useful in the context of the RIA. The RIA goes on to discuss "several regional studies" of changes in wetland area but highlights only one study: Wright et al. (2017). The RIA concludes that this study demonstrated a causal connection between the proximity of an ethanol refinery and loss of wetlands. The RIA also relies on Wright et al. (2017) in its discussion of losses of shrubland and forest ecosystems. As described above, the study by Wright et al. (2017) has been shown to be unreliable.

The RIA also discusses loss of land in the CRP and references a single study by Morefield et al. (2016) who conclude that CRP land lost between 2010 and 2013 largely went to conversion to row crops for corn and soy. The authors of the study do not try to attribute the loss of CRP to increased demand for biofuels, rather they acknowledge that important drivers of extensification at the expense of grassland and wetlands include a combination of "commodity prices, reduced land retirement options, and diminishing interest in land retirement programs…"

In its discussion of wildlife impacts, the RIA mentions loss of wetlands and impacts to ducks, and loss of grasslands and impacts to grassland birds and insects. The RIA acknowledges that the effects of the RFS on wildlife have not been studied, yet presents results from a study of grassland bird diversity and cropland that implicates LUC in reduced species diversity, and studies of impacts to pollinators, including a discussion of the potential role of exposure to agrichemicals.[10] The RIA does not infer a causal relationship between the RFS or crops grown for biofuel feedstock as a driver for effects to wildlife and concludes that "[a]t present it is not possible to confidently estimate the fraction of wildlife habitat loss or of corn or soy production that is attributable to biofuel production or use. Thus, we cannot confidently estimate the impacts to date on wildlife from biofuels generally nor from the annual volume requirements, specifically" (pages 98-99).

The discussion of potential impacts to wetlands ecosystems, and wildlife habitat presented in the RIA (as well as in the Second Triennial Report) is unbalanced and creates a false impression that the generic impacts described are attributable to the RFS. EPA should reevaluate this discussion to present a balanced perspective that accurately presents the current state of knowledge regarding the lack of a quantitative relationship between biofuel feedstock grown specifically to meet the RFS requirements and potential impacts.

An expanded review of literature on this topic since our work on Exhibits 1 and 2 concludes that no publications establish a quantitative or qualitative causal link between impacts from biofuel feedstock

[10] This includes a discussion of neonicotinoids which concludes that the risk of these chemicals to pollinators is poorly understood, and EPA's preliminary determination is that the risk is low.

production and impacts to wetlands, ecosystems, or wildlife habitat.[11] Wiens et al. (2011) suggests a linkage between biofuels and potential biodiversity impacts by implicating demand for ethanol in the loss of CRP lands but provides no analysis of causation. Wimberly et al. (2018) implicate corn and soy extensification in increases in grassland habitat fragmentation in eastern South Dakota and western Minnesota. These authors state that the LUC was driven by higher corn prices driven by increasing demand for ethanol, and they cite Lark et al. (2015), Wright et al. (2017), and Wright and Wimberly (2013)—all studies that have been largely discredited, as described above. Hoekman and Broch (2018) describes benefits and "dis-benefits" of LUC ostensibly driven by higher ethanol prices but provide no quantitative or qualitative causal links to the RFS or corn grown for ethanol.

In sum, the RIA should reflect that the latest scientific literature does not establish any causal relationship between the RFS and impacts to wetlands, ecosystems, or wildlife habitat.

[11] For many publications on this topic, that is not the goal. For example, Pleasants (2017) quantifies milkweed stem abundance in soy and corn fields in the U.S. Midwest and develops estimates of the restoration required to increase monarch butterfly populations but does not mention the RFS or biofuels. A paper by Landis et al. (2018) models biodiversity and ecosystem services under different biofuels cropping systems, but makes no mention of the RFS or corn grown for ethanol.

THE RFS AND WATER RESOURCE AVAILABILITY AND WATER QUALITY

Both the Second Triennial Report and the RIA present detailed discussions of water use to grow corn as well as potential adverse effects to water quality from agriculture in general due to soil loss and transport of pesticides and fertilizers to surface water bodies. However, EPA provides no causal nexus between these potential impacts and agriculture to produce biofuels in general, or more to the point, production of biofuels for the RFS.

Further, existing studies suggest that environmentally protective goals for biofuel production are achievable as best management practices and technological advances in farming continue to be adopted by the farming community.

Water Resource Availability

Water Used for Growing Corn

As is true for LUC, the relationship between corn production and water resource availability and water quality varies geographically and temporally, and studies have failed to demonstrate a quantitative causal link between corn grown for ethanol or the RFS and impacts to water availability. The RIA, EPA's Second Triennial Report, and studies cited therein provide information regarding the geographical distribution of irrigated agriculture in general and biofuel crops relative to known stressed aquifers, but there is no evidence or analysis provided regarding the relative impact of corn grown for ethanol production in response to RFS mandates.

In its discussion of life cycle water use for biofuel feedstock production, the RIA relies on information and analysis presented in the Second Triennial Report, including specific reference to Lark et al. (2015) and Wright et al. (2017) regarding these authors' now refuted conclusions about cropland expansion associated with biofuels and the RFS. The RIA does not present any assessment of the actual water intensity of corn grown for ethanol, rather it presents some general statistics regarding use of water to grow corn and soy. For example, the RIA makes the following statements:

- 90% of corn is grown in areas where corn is non-irrigated.
- If 20% of corn production was used to produce 12 billion gallons per year of ethanol, this would represent only 4.4% of all irrigation withdrawals (citing a study by Dominguez-Faus et al. 2013).
- Nebraska is one of the states with the largest water withdrawals for irrigation and recent increases in irrigation withdrawals have been largely driven by the need to irrigate corn for ethanol (citing a report by the National Academy of Sciences; NAS 2011).

These statements are of little value without providing specific context relative to corn grown for ethanol, and this context is highly geographically variable. In particular, the statement by NAS (2011) regarding irrigation withdrawals in Nebraska and corn grown for ethanol is unsubstantiated by the authors of the report and should not have been cited in the RIA. Moreover, it is outdated by over a decade. However, the RIA acknowledges that "…there have been no comprehensive studies of the changes in irrigated acres, rates of irrigation, or changes in surface and groundwater supplies attributed specifically to the increased production of corn grain-based ethanol and soybean-based biodiesel" (page 123).

A review of literature not previously presented by EPA did not reveal any studies attempting to quantify water use specifically for growing corn that was destined to produce ethanol required to meet RFS requirements. Xie et al. (2019a), Xie and Lark (2021), and Xie et al. (2021) present findings related to mapping of irrigated land in the U.S., but they present no nexus between water use and the RFS or water use and corn grown for ethanol. Xie et al. (2019b) present a method for mapping annual irrigation distribution over the period 2000-2017 and conclude that irrigation over the period 2009-2017 was greater than over the period 2000-2008 and that the greatest increase was in Nebraska and was associated with corn and soy. However, the paper contains no mention of the RFS, biofuels, or ethanol and therefore does not attempt to link the increase in annual irrigation to renewable fuels policy and its conclusion regarding association of increased irrigation with corn and soy is unfounded.

The impacts of irrigation withdrawals to grow corn for ethanol are dependent on the existing condition of available water resources, the use of irrigation water to produce corn relative to other crops, and the proportion of irrigated corn grown that is used in the production of ethanol. Although the RIA acknowledges these complexities, it fails to explicitly relate them to the RFS or to corn grown for ethanol. The RIA postulates three approaches for estimating the change in water demand that may result from increased ethanol volumes: life cycle water requirements for ethanol as compared to gasoline, projected LUC and crop management, and changes in crop prices and associated economic value of irrigation. The RIA acknowledges EPA's inability to perform such analyses yet concludes that there is likely to be some increased irrigation pressure on water resources due to the proposed ethanol volumes. The RIA fails to acknowledge that "increased irrigation pressure," to the extent such a thing may occur, does not necessarily translate to increased overall water use or strain on existing water resources. Moreover, the RIA asserts that the changes in irrigation may result from the proposed volumes impact on crop prices without first establishing that any such impact on crop prices has occurred historically or is likely to occur as the result of the proposed volumes, which are similar to the volumes in 2019.

Water Use for Ethanol Production

Ethanol refineries have made great strides in reducing water consumption. In a 2007 Renewable Fuels Association survey of 22 ethanol production facilities (representing 37% of the 2006 volume produced), dry mills used an average of 3.45 gallons of water per gallon of ethanol produced and wet mills used an average of 3.92 gallons of water per gallon of ethanol produced. Muller (2008) reported declines in water requirements at ethanol dry mills from 5.8 gallons of water per gallon of ethanol (gal/gal) in 1998 to 2.7 gal/gal in 2012. Wu and Chiu (2011) noted that water consumption in existing dry mill plants had, on a production-weighted average basis, dropped 48% in less than 10 years, a reduction that is similar to that reported by Muller (2008). These previously reported improvements in efficiency are confirmed by the latest scientific literature. For example, Wu (2019) reports a 54% decrease in water intensity for the ethanol industry over the period 1998-2017 and a 12% decrease over the period 2011-2017 illustrating the gains in water use efficiency at ethanol plants. Improvements in water use efficiency at ethanol refineries are largely ignored by EPA.

Advancements in Farming Practices Reduce Agriculture's Impacts on Water Resource Availability

What is clear but not adequately recognized by EPA in the RIA or Second Triennial Report is that advancements in farming practices and technology have reduced the negative impact of farming on water resource availability. Over the past decade, there has been increased use of precision agriculture methods as well as standard best practices which retain soil moisture. This trend is expected to continue and is expected to reduce the need for irrigation. As an indication of the trend in irrigation reduction, the University of Nebraska (2018) reports that in Nebraska (as a bell-weather of other dry western states), the percentage of all corn acreage that is irrigated has declined from a high of 72% in 1981 to 56% in 2017.

Farms are increasingly moving away from traditional, less-efficient irrigation systems and adopting water saving irrigation systems. In 2018, 67% of cropland acres irrigated used pressurized systems including sprinklers and low-flow micro systems (Hrozencik and Aillery 2021). The number of farms using inefficient gravity irrigation systems decreased from 62% in 1984 to 34% in 2013, converting mostly to pressure-sprinkler irrigation which is more efficient than gravity irrigation. Water savings associated with advanced irrigation systems relative to typical gravity systems are summarized below[12]:

- Subsurface Drip: 25-35%
- Rainwater Harvesting: 50%
- Precision Agriculture: 13%
- Conservation Structures: 18%

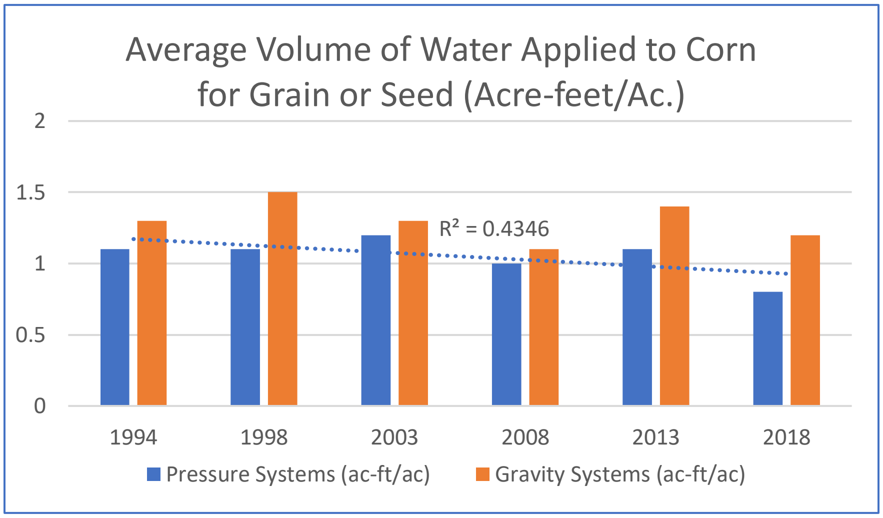
In terms of the adoption of precision agriculture, almost 10% of farms use soil-moisture or plant-moisture sensing devices or commercial irrigation scheduling services. Sensor technology can optimize irrigation scheduling and hence increase water use efficiency. It is also anticipated that additional large industrial farms (which make up a large volume of total production) will employ water use simulation models that are based on corn growth patterns and weather conditions. Adoption of these technologies will continue to grow in the U.S., and particularly in the west, where 72% of water irrigation takes place and farmers have recent experience with low water supply following the 2012-2016 drought. In addition to changes in irrigation technologies, agricultural practices regarding the timing of irrigation have helped reduce the amount of water applied to corn. For example, Xue et al. (2017) have shown that corn crops can forego initial irrigation without significant adverse effects to yield.

Genetic engineering and selection for improved drought tolerant corn cultivars has resulted in corn strains that can tolerate a 25% reduction in water application without affecting yield. The use of drought-tolerant corn, which was commercially introduced in 2011, increased to over 22% of the total U.S. planted corn acreage by 2016 (Mcfadden et al. 2019). More importantly, this percent of use was greatest in the driest corn-producing states of Nebraska (42%) and Kansas (39%). Other states that

[12] See for example Ailen 2013, Barton and Clark 2014, Biazin et al. 2012, Center for Urban Education about Sustainable Agriculture (CUESA) 2014, Gowing et al. 1999, Shangguan et al. 2002, National Research Council 2008, Netafim n.d., and Qin et al. 2015.

are not as drought prone (e.g., Minnesota, Wisconsin, and Michigan) saw drought-tolerant corn planting ranging between 14% and 20% of total acreage in 2016. Figure 5 illustrates the downward trend in the volume of water applied to corn over the period 1994-2018.

Figure 5. Average volume of water applied to corn over the period 1994-2018.



SOURCE of Data:
ttps://www.nass.usda.gov/Publications/AgCensus/2017/Online_Resources/Farm_and_Ranch_Irrigation_Survey/fris_2_0036_0036.pdf Accessed 1/5/2022

Water Quality

The RIA discussion of potential water quality impacts, like the Second Triennial Report, fails to establish a causal link between corn grown for the RFS and water quality impacts. The RIA discusses several studies addressing impacts to soil and surface water from corn and soy, including erosion, soil carbon depletion, and nutrient runoff. These impacts are inextricably linked to LUC as well as crop intensification, but the results of these studies are relevant only to the extent the RFS-LUC or RFS-intensification link can be demonstrated and quantified. The RIA at page 101 presents a simplistic example calculation of the increased nitrogen applied to farm fields nationwide due to corn extensification, but this calculation is based on the work of Lark et al. (2015) which has been shown to be unreliable. This example calculation should be removed from the text of the RIA because it is erroneous and misleading. Similarly, the RIA discusses data from USDA NASS for percentages of planted corn acres that received treatment using herbicides, insecticides, and fungicides, but such information is irrelevant to an analysis of the water quality impacts of the RFS unless it can be quantitatively tied to the program. Finally, the RIA cites work by Garcia et al. (2017) who estimate that corn production between 2002 and 2022 would result in nitrate groundwater contamination > 5 mg/L in areas with sandy or loamy soils. Although Garcia et al. (2017) mention the RFS as a potential driver for increased corn production, the study does not derive a quantitative causal relationship between their conclusions and the RFS, nor is that a stated goal of the research. It is inappropriate for EPA to present this work without adequate context.

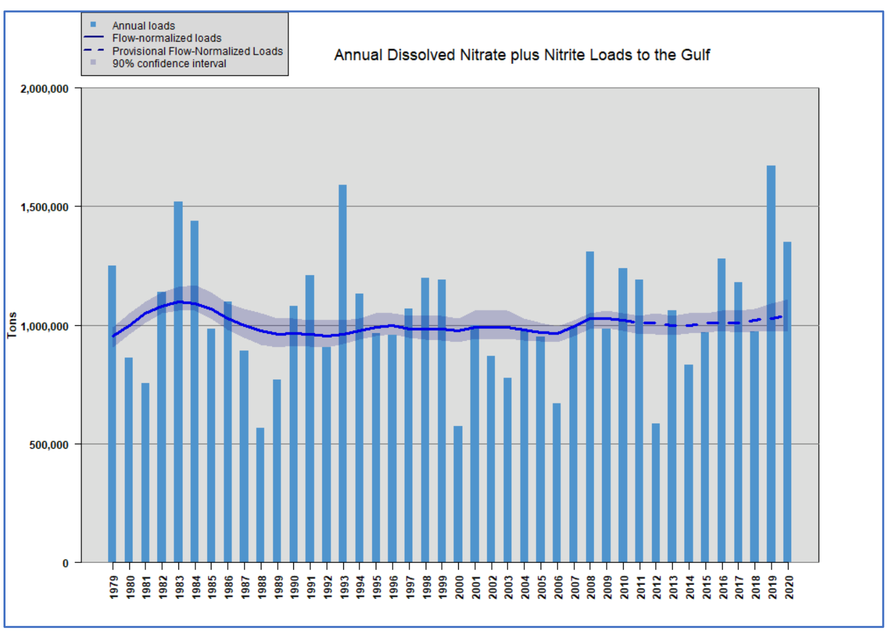
In the absence of evidence of a quantitative causal relationship, the discussion in the RIA of the relationship between agriculture (and corn growing in particular), and proximal water quality is unbalanced and creates the unfounded impression of a direct causal relationship.

Similarly, the RIA presents a discussion of the potential downstream effects of corn and soy cultivation. In terms of aquatic life, the RIA presents a discussion of the biological condition of the nation's rivers and streams and the causative factors contributing to poor conditions. As with its Second Triennial Report, the RIA mentions large scale hypoxia in western Lake Erie and the Gulf of Mexico and the nexus to nutrient enrichment. The RIA also discusses the potential for downstream impacts from herbicides and pesticides applied to corn and soy. Throughout these discussions, the RIA provides no nexus to the RFS.

Like the Second Triennial Report, the RIA fails to acknowledge that there is no established causal connection between corn grown for ethanol and the formation, persistence, or severity of hypoxic events in western Lake Erie or the Gulf of Mexico. The RIA ignores a rich literature describing the complexity of these phenomena as well as characterization and modeling of nutrient loading to these systems from various sources, but such studies fail to establish a causal relationship between the formation and severity of the GoM dead zone and the RFS. For example, econometric modeling by Secchi et al. (2011) predicts an increase in corn acreage due to corn intensification spurred by increasing corn prices and extends that prediction to estimate increased nutrient loading to the Upper Mississippi River basin; however, these authors did not attempt to assert a causal connection between increased corn prices (the variable that drives their analysis) and the RFS.

As illustrated in Figure 6, the RIA fails to acknowledge that nitrogen loading to the Gulf of Mexico has remained fairly stable over the past 40 years.

Figure 6. Annual Nitrate and Nitrite Loading to the Gulf of Mexico 1980-2020.



Source: USGS n.d.

There may be no dispute that excess nutrient loading from the key watersheds that discharge into western Lake Erie and the northern Gulf of Mexico contribute to eutrophication and hypoxia; however, these watersheds contain a complex mix of urban and rural uses that present important sources of nutrients as well as toxic contaminants. In any case, the direct causal link to the RFS or corn grown for ethanol production (compared to all other uses and compared to all other agricultural activities) is not substantiated by the Second Triennial Report or the literature cited therein and should be qualified as such to the extent discussed in the RIA. Regional hypoxic conditions in western Lake Erie and the Gulf of Mexico were increasing in frequency and severity, long before ethanol production increased, and this fact should also be acknowledged by EPA.

The RIA also fails to acknowledge the importance of regional weather on the occurrence and severity of large-scale hypoxia events. The National Oceanic and Atmospheric Administration (NOAA) states that a major factor contributing to the large Gulf of Mexico "dead zone" in 2019 was the abnormally high amount of spring rainfall that resulted in flows in the Mississippi and Atchafalaya Rivers that were 67% above the average flows over the previous 38 years[13]. Data collected by the United States Geological Survey (USGS) indicate that because of these high flows, nitrate loads were about 18% above the long-term average, and phosphorus loads were approximately 49% above the long-term average (USGS 2019).

Notwithstanding the misleading discussions presented in the RIA, the RIA correctly acknowledges 1) that the important determinants of impacts to water and soil quality are not directly determined by the RFS; 2) there are many effective management practices that can act to counterbalance any negative impacts from corn for ethanol; and 3) the magnitude of potential impacts due to the RFS cannot be estimated at this time. The RIA should be edited to present a more balanced discussion of potential water quality impacts from biofuel feedstock agriculture within the specific context of crops grown for biofuels to meet the goals of the RFS program.

[13] Courtney and Courtney (no date) reported that predictions made by NOAA and Louisiana University's Marine Consortium of the areal extent of the GoM dead zone were 31% higher than the actual measured hypoxic areas from 2006 to 2014. The authors of this paper hypothesize that GoM waters are becoming less susceptible to low dissolved oxygen over time.

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EXHIBIT 1

RAMBOLL. AUGUST 18, 2019. THE RFS AND ETHANOL PRODUCTION: LACK OF PROVEN IMPACTS TO LAND AND WATER. PREPARED FOR GROWTH ENERGY. RAMBOLL, SEATTLE, WA.

THE RFS AND ETHANOL PRODUCTION: LACK OF PROVEN IMPACTS TO LAND AND WATER

Prepared at the Request of
Growth Energy

Prepared by
Ramboll

Date
August 18, 2019

CONTENTS

TABLES

FIGURES

ACRONYMS AND ABBREVIATIONS

API	American Petroleum Institute
CDL	cropland data layer
CGF	corn gluten feed
CRP	Conservation Reserve Program
DDGS	distiller's dried grains with solubles
EISA	Energy Independence and Security Act
EPA	U.S. Environmental Protection Agency
FSA	Farm Service Agency
ha	hectare
ITRC	Interstate Technology & Regulatory Council
LUC	land use change
NOx	nitrogen oxide
NASS	National Agricultural Statistics Service
NLCD	National Land Cover Database
NOAA	National Oceanic and Atmospheric Administration
NRCS	National Resources Conservation Service
NWI	National Wetlands Inventory
RFM	Reduced form model
RFS	Renewable Fuel Standard
SOA	secondary organic aerosols
SOx	Sulphur oxide
TPH	total petroleum hydrocarbons
UOG	unconventional oil and gas
USDA	U.S. Department of Agriculture
USGS	U.S. Geological Survey
USEIA	U.S. Energy Information Administration
VOC	volatile organic compound

1. EXECUTIVE SUMMARY

This report was prepared by Ramboll for Growth Energy in anticipation of the United States Environmental Protection Agency (EPA) issuing proposed rulemaking on the Renewable Fuel Standard (RFS), commonly referred to as the "RFS Reset." One of the factors that EPA must consider in resetting renewable fuel volumes in the program is potential environmental impacts.

The key conclusion of this report is that there are no proven adverse impacts to land and water associated with increased corn ethanol production under the RFS. Accordingly, EPA could decide to reset renewable volumes in a manner that would incentivize greater production and consumption of conventional corn ethanol in US transportation fuel without discernible adverse environmental impacts to land and water, to the extent any exist. The major factors supporting this conclusion are that continued improvements in agricultural practices and technology indicate that increased demand for corn grown for ethanol in the United States can be met without the need for additional acres of corn planted, while at the same time, reducing potential impacts to water quality or water supplies.

Our review focused on analyses concerning water quantity and quality; as well as ecosystems, wetlands, and wildlife. Analyses concerning ecosystems, wetlands, and wildlife were presented primarily as part of the body of literature addressing land use change (LUC) and conversion of land from non-agricultural to agricultural uses in the United States. We focused particular attention on EPA's recent environmental review of the RFS, *Biofuels and the Environment: Second Triennial Report to Congress* (EPA 2018a), and studies relied upon by the agency therein. Ramboll also reviewed other key publications pre- and post-dating EPA (2018a). A full list of refences cited in this report is presented in Section 8.

We also reviewed a recent paper by Hill et al. (2019) investigating the air quality-related health impacts of growing corn. Finally, we provide a brief overview of certain environmental impacts of oil and gas exploration and production and gasoline refining, in response to EPA's (2018a) acknowledgement that its assessment is not fully comprehensive because it does not consider a comparative assessment of the impacts of biofuels relative to petroleum-derived fuels.

The principal findings of our review by topic include, but are not limited to:

- Land use change—Some investigators have asserted that the RFS has resulted in extensive conversion of non-agricultural land to agriculture due to increased demand for corn for ethanol. Our findings indicate that these claims are not borne out, in part because the studies do not establish a causal link between the RFS, increased ethanol production, and LUC. Indeed, in a follow-up analysis to its Triennial Report EPA (2018b) reached the same conclusion—that no causal connection has been established between LUC associated with corn production and the RFS.
  - The number of acres planted in corn has remained effectively constant despite significant increases in production. Acres planted in corn across the United States has remained close to or below the total acres planted in the early 1930s, despite increases in demand for corn as human food, animal feed, and biofuels over this nearly 90-year period. The increase in demand has largely been met by an approximately 7-fold increase in yield (bushels per acre).
  - Most studies asserting a connection between the RFS and LUC fail to adequately account for the myriad factors that drive farmers' choices to

plant a given crop or to convert non-agricultural land to cropland. The price of corn is only one of many such factors, and the literature does not support that the RFS is the predominant driver of pricing of this global commodity. Moreover, assertions that the RFS drives LUC, fail to adequately recognize the increased efficiency in corn production per acre as well as the diminished demand for corn crop acreage due to co-products of the ethanol refining process, such as distiller's dried grains with solubles (DDGS). Assessments of LUC and the RFS generally fail to recognize external factors that might be driving expansion of farmland, such as the loss of farmland near urban areas.

- Water use and water quality—EPA (2018a) and other authors raise concerns that increased corn grown for ethanol may be overstressing water sources and resulting in regional water quality impacts. Our findings indicate that these concerns are not borne out primarily due to research that fails to establish a causal relationship between corn grown for ethanol and impacts to water use and water quality. We further find that EPA (2018a) does not adequately acknowledge the role of advances in agricultural practices in mitigating potential water use and water quality impacts.
  - A quantitative or causal relationship between the RFS and concerns over water use has not been established. From a geographical standpoint, much of the corn that is used for ethanol production is grown on non-irrigated land where impacts to water availability are minimal, and while noted, this is not quantitatively considered by EPA (2018a). In addition, the increased adoption of modern farming practices and precision agriculture (Vuran et al. 2018) is reducing the potential impact of agriculture in general, including increased corn production, on water availability. EPA (2018a), in fact, noted that the increased use of these best management practices should substantially limit impacts to water resources. While some investigators have claimed that growth in corn production has resulted in greater stress to water resources, those studies that focus on negative impacts fail to acknowledge, or do not appear to emphasize, that the current focus on best management practices and resource protection is being widely adopted by the corn growing community and incentives to adopt such practices continue. The technical publications we have reviewed do not establish that the RFS drives corn planting decisions and potential associated water impacts.
  - A quantitative or causal link between corn production associated with the RFS and adverse water quality impacts has not been established. While observed environmental impacts, such as excessive algae blooms in western Lake Erie and low oxygen levels in the Gulf of Mexico have been documented, we found that the literature on this issue fails to quantitatively link these regional water quality problems to increases in corn production for ethanol. Indeed, nutrient loading to the Gulf of Mexico, as measured by nitrates and nitrites, has remained relatively constant since at least 1980 despite increases in corn production. In addition, very few investigators have looked closely at agriculture trends over the past decade that show the implementation of modern farming practices are helping to reduce potential watershed impacts; modern farming practices include improved products such as slow-release fertilizers, and improved practices such as precision agriculture and better water and stormwater management. This trend is expected to continue well into the future and provide additional benefits to other agricultural products in addition to corn. Finally, expected future gains in corn yield (bushels produced per acre per year) in combination with steady or even declining fertilizer and pesticide

use (in pounds per acre per year), will naturally result in a decrease in the potential for water quality impacts.

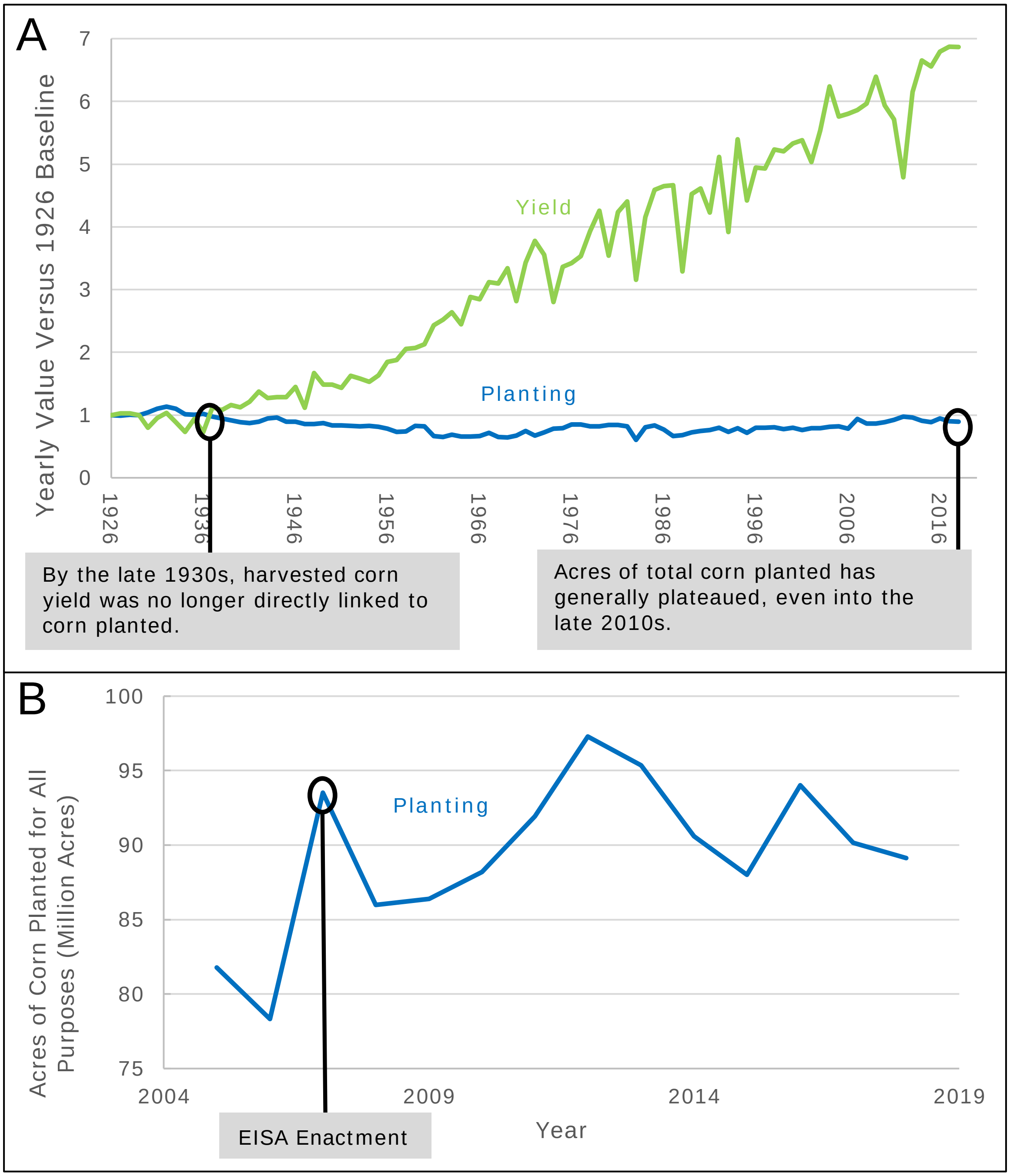
- The RFS Reset is well-timed to coincide with ongoing improvements in agricultural practices—Nearly all published investigations Ramboll has reviewed that focus on the potential impact of increased corn growth for biofuel production have focused on past practices with only passing mention of future expectations. EPA (2018a) acknowledges the benefits of the increased use of best management practices on the environment. Modern agricultural practices are economically beneficial to corn producers when they result in reduced input costs associated with water and agricultural chemicals. The timing for increasing corn production and reduced potential environmental impacts due to precision agriculture coincides with increased biofuel demand, and the coincidence of these trends will benefit both producers and the environment into the future.

1.1 Total Acres Planted in Corn Has Remained at or Below Levels in the Early 1930s While Total Production Increased 7-Fold

The United States Department of Agriculture (USDA) has maintained annual statistics on domestic crop production for decades. Corn production in the United States annually exceeds 10 billion bushels, with approximately 50% of corn currently grown for ethanol production and 50% for grain use. Accordingly, corn is documented to be the most widely produced feed grain in the United States (U.S.), accounting for more than 95 percent of total production and use followed by sorghum, barley, and oats (USDA 2019). Most of the corn crop for feed grain is used for livestock feed. Other food and industrial products include cereal, alcohol, sweeteners, and byproduct feeds.

While the approximate share of U.S. corn (in bushels) dedicated to production of ethanol has increased from 4% in 1986, to 38% in 2015 (USDA-ERS 2019b), and to approximately 50% in 2018, the total corn planting (in acres) has remained relatively stable since the 1930s (Figure 1). On a shorter time-scale, acres of corn planted each year does vary, but when examining data between 2007 and 2018, there is no long-term upward trend. In fact, acres of corn decreased 8.07% in 2008, the year after the enactment of the Energy Independence and Security Act (*EISA*), then rebounded through 2012, then decreased again such that in 2018, acres of corn were 4.7 % lower than in 2007. These data, from the USDA Crop Production Historical Track Record (updated in USDA, 2019) demonstrates the increased efficiency, planting and production of the corn crop without a need to secure appreciable additional acreage for production. Efforts in better crop management, improved fertilizer use, and precision agriculture are all likely contributors to improved yields. The USDA further anticipates changes in corn production to result in an increase of approximately 16.1 more bushels per acre by 2028 without a substantial increase in farmed acres (and with a corresponding reduction in the use of water resources and fertilizer).

Figure 1: A) Annual Yield in Bushels of Corn Per Acre and Annual Acres Planted in Corn Versus 1926. B) Annual Acres of Corn Planted 2004-2018.

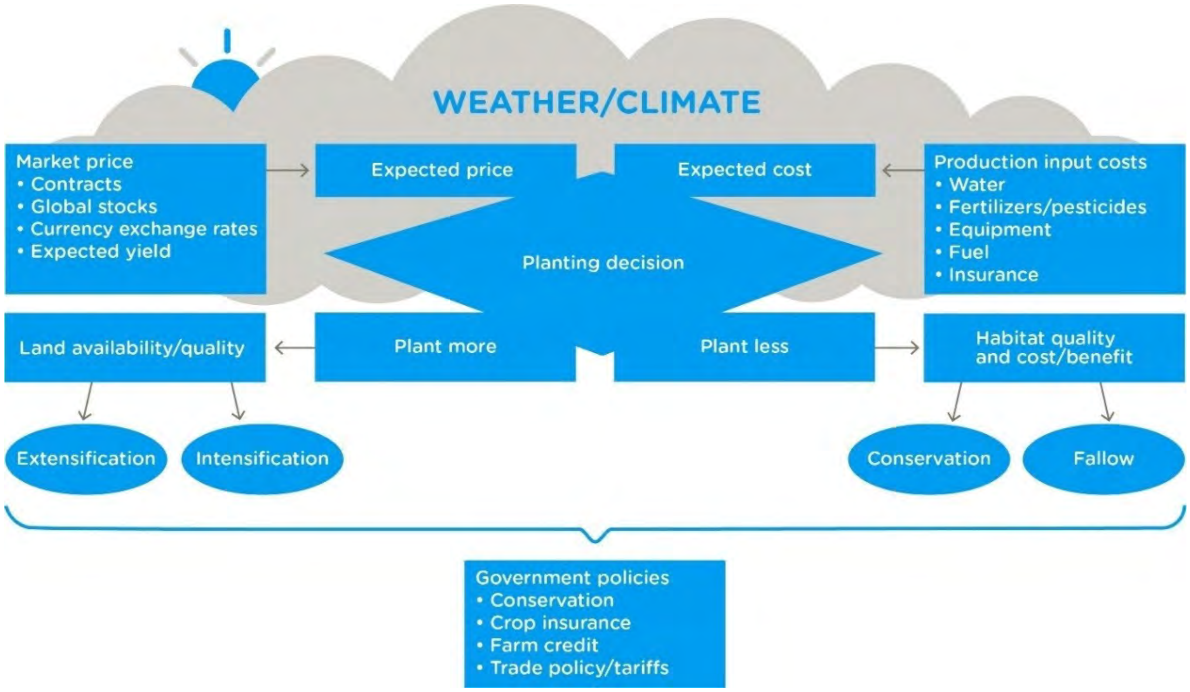


Source: USDA Crop Production Historical Track Records, 2019

1.2 Studies Have Failed to Establish a Quantitative Link Between the RFS and Land Use Change

The decision by farmers and landholders on whether to plant a bioenergy crop such as corn reflects complex relationships between biophysical, economic, and social factors (Figure 2).

Figure 2: Illustration of the Complexity of Biophysical, Economic, and Social Factors Affecting Planting Decisions.



One factor that is of paramount importance is weather and climate. Regional weather patterns largely dictate crop patterns across the country, but this is also influenced by the availability (and price) of water for irrigation in areas with relatively low annual precipitation or highly variable precipitation. The probability of severe weather such as drought and flood as well as severe storm events in any given year, may also influence planting decisions. Government policy is another overarching factor affecting planting decisions, and these include potential monetary incentives associated with the U.S. Department of Agriculture Conservation Reserve Program (USDA CRP), other conservation programs such as local conservation easements, the availability of crop insurance, and market incentives that might affect commodity prices. Other factors include market price and the price of production inputs, which can be strongly influenced by the price of oil, exchange rates and trade policies. Local market prices are influenced by a wide range of factors including status of commodity stores, distance to markets, and competition from regional and even global markets. Input prices are also highly variable due to market prices, and volume requirements for some inputs such as irrigation water are weather and climate dependent. Finally, all of the above factors, plus the availability and quality of land and ecosystem characteristics and ecological value play into decisions regarding land use—whether to plant new acreage (extensification) or plant more of a given crop on existing acreage (intensification).

The influence of the RFS on LUC is poorly understood and likely weak. To the extent it suggests otherwise, EPA (2018a) inadequately assesses the range of market and nonmarket factors influencing land use change and does not consider key studies that suggest that the RFS likely had a small, and perhaps negligible effect on LUC, especially changes in land use from non-agriculture to biofuels feedstock (corn and soy). In particular, EPA (2018a) does not adequately consider the role of farm policy such as crop insurance, land characteristics, input and output prices, and technology on growing decisions by farmers.

Studies relied upon by EPA (2018a) to quantify LUC around the time of enactment of the RFS are based on unreliable data and likely overestimate LUC. In particular, EPA (2018a) cites work by several authors who report findings of considerable LUC, including LUC in ecologically sensitive areas such as the Prairie Pothole Region, but do not sufficiently acknowledge or discuss findings by more recent research that indicates that many of the earlier studies were flawed or substantially overstated the extent of LUC in and around the enactment of the RFS. EPA (2018a) also does not sufficiently acknowledge that the studies it relied on do not establish a causal relationship between the RFS and LUC. In addition, EPA (2018a) makes no attempt to quantify, or even describe in any detail, the potential ecological impacts of the alleged LUC, so the actual environmental harm, if any, associated with the RFS remains nebulous. Notwithstanding these shortcomings of the report, EPA clarifies in a subsequent discussion of environmental impacts of the RFS that it *does not* view the literature it identified in the Triennial Report as supportive of a causal link between LUC and the RFS; rather, there is a myriad of “complex regulatory and market factors that are relevant to such a relationship” (EPA 2018b).

A recent effort by Lark et al. (2019) to develop a quantitative link between the RFS and LUC may be the most exhaustive effort to date, but their reliance on an uncertain “business as usual” baseline and on estimating price increases attributable to the RFS are major weakness of the work. Most important, the entire analysis presented by Lark et al. (2019) rests on estimating price increases attributable to RFS, yet the authors fail to adequately acknowledge the role of important factors such as the dietary transition from cereals toward more animal protein in developing countries resulting in rapid growth in the consumption of agricultural commodities. Other important factors affecting corn prices over the period include higher oil prices and the link between the U.S. dollar exchange rate and commodity prices. In addition, the data sets and models used in their analysis are not made explicit, and some data are not in the public domain, precluding a thorough independent review of their work.

EPA (2018a) also failed to adequately account for the role of cropping practices and production of DDGS at ethanol refineries as important LUC offsetting factors. Several studies indicate that a substantial portion of increase corn production following the introduction of the RFS was met via farmers’ cropping practices, including switching from other row crops to corn or double cropping corn instead of rotating between corn and soy (or other crops). These studies are not given adequate consideration by EPA (2018a). Although EPA (2018a) acknowledges that production of DDGS may offset some demand for corn as livestock feed, key studies estimate this offsetting effect is considerable. In addition, EPA (2018a) does not discuss whether and to what extent this offset for demand for corn is a market driver that provides downward pressure for LUC to corn.

1.3 Changes in Agricultural Practices Broadly Reduce the Likelihood of Environmental Impacts to Water Resource Availability and Quality

Advancements in technology and water management techniques have continued to increase the efficiency in water resource management by stabilizing, and potentially reducing, the overall volume of water necessary for corn growth. Agriculture accounts for an estimated 80 percent of national consumptive water use in the US according to the USDA’s Economic Research Service (2018) and reaffirmed by the National Academy of Science (2019). According to the 2012 statistics from the USDA, irrigated corn acreage represented about 25% of all irrigated acreage in western states, and about 24% of all irrigated acreage in the eastern states (USDA-ERS 2018a). Additionally, the USDA has shown that irrigation for all crops, including corn, has decreased even as the farming acreage has essentially been stable over the past 35 years. The USDA attributes this trend to improvements in physical irrigation systems and water management. The USDA also notes that significant capital investments in on-farm irrigation is continuing, particularly in the western states, where most of the irrigated farm-land is concentrated. As an indication of a positive trend in irrigation reduction, the University of Nebraska, Lincoln reports that in Nebraska (as a bell-weather of other dry western states), the percentage of all corn acreage that is irrigated has declined from a high of 72% in 1981 to 56% in 2017 (University of Nebraska 2018).

Increasing crop yield per area of farmed land is taking place on both irrigated and unirrigated corn crops, suggesting that changes in yield are not attributed to irrigation alone. In certain areas, more corn is now being grown on the same number of acres, which has resulted in increases in irrigation. However, watersheds where most intensification has occurred are mostly in Western states which account for less ethanol feedstock than the less- or non-irrigated Midwest and Eastern States.

Trends and expectations in the biofuel refining process also show increasing water use efficiency and lower water demand over time (upwards of 50% reductions in recent years). This trend is anticipated to continue as ethanol refining technology advances.

Advances in sustainable farm management, including substantial improvements in nutrient formulation and use, and technological improvements in pesticide and fertilizer application, will continue to reduce the potential for impacts to water quality in regional watersheds near corn growing areas regardless of the cause of historical water quality impacts. Additionally, the EPA acknowledges that corn production for ethanol has not been reliably linked to large scale degradation of water quality. The hypothesized causal relationship between the hypoxic zones in the northern Gulf of Mexico and eutrophication in Western Lake Erie with corn grown specifically for ethanol production is weak and lacks supporting data. It is recognized that urban and agricultural runoff in the subject watersheds have likely contributed to the conditions; but EPA (2018a) notes that attributing these water quality issues to ethanol production is speculative and not based on specific data.

1.4 Recent Estimates of Health Damages from Corn Production are Unreliable and Misleading

Although the primary focus of this report is on studies assessing the implications of the RFS program and corn ethanol production for land and water, a recent report that attempts to link corn production to adverse public health impacts from air emissions merits a brief response. A recent publication in Nature Sustainability (Hill et al. 2019) purports to estimate US annual health damages caused by particulate air quality degradation from all direct farm and indirect supply chain activities and sectors associated with maize (corn) production. Although the authors do not reference the RFS, they do mention corn grown for ethanol, and the publication has been referenced by third parties in a manner suggesting that corn grown for ethanol may be associated with adverse health outcomes. Ramboll's review indicates that the conclusions presented by Hill et al. (2019) are unsubstantiated and likely overestimate adverse health impacts, where it is not clear any health impacts exist.

The direct and indirect activities explored by Hill et al. (2019) include air emissions from farms and upstream processes that produce the chemical and energy inputs used in corn crop production: fuel, electricity, agrichemical production, transportation, and distribution. The paper focuses on particulate matter smaller than 2.5 microns in diameter ($PM_{2.5}$), which is a concern for human health because particles of this size can penetrate deep into the lungs and enter the bloodstream, and potentially result in both acute and chronic effects to the respiratory and cardiovascular systems. Ramboll reviewed the underlying models and assumptions employed in the Hill et al. (2019) analysis and we present the following findings:

- The model relied upon by the authors uses annual-average data for emissions, meteorology, and chemical/removal rates to estimate annual-average $PM_{2.5}$ impacts. Use of annual averages is inappropriate for representing processes that operate over shorter time scales ranging from minutes to several months (e.g., atmospheric dispersion and chemical formation of $PM_{2.5}$) and results in a high level of uncertainty. The authors acknowledge that this weakness in their approach results in spatial errors in annual average $PM_{2.5}$ calculations. These spatial errors can significantly impact the resulting exposure and mortality estimates. The authors, however, do not present sensitivity analyses to assess the impact of the model assumptions, nor do they include any plausible range of uncertainty or variability with their modeled $PM_{2.5}$ concentration or mortality estimates.
- The 2005 modeling year upon which modeling is based is not representative of more recent chemical conditions of the atmosphere in the U.S., which may lead to an overestimate of the $PM_{2.5}$ contributions from corn production by more than a factor of 2, and this overestimate results in overestimates of health and economic damages.
- Several major sources of uncertainty in the modeling are not acknowledged or accounted for by the authors, including the following key uncertainties:
  - Ammonia emission estimates, which are the largest driver of mortality in the Hill et al. (2019) modeling analysis, are also the most uncertain aspects in any $PM_{2.5}$ air quality modeling, because: (1) emissions are largely from agricultural sources that vary both spatially and temporally due to weather and farming practices; (2) many different methods are used to estimate ammonia emissions, and each can yield very different emission rates and exhibit a high degree of error; (3) annual average ammonia emission inventories used in the modeling fail to account for important seasonal variations and related complex interactions with sulfate and nitrate chemistry; and (4) ignoring diurnal and intra-daily ammonia emission variations have been shown in the literature to overestimate ambient ammonia concentrations by as much as a factor of 2.
  - The health impact assessment is based on a single epidemiological study that found associations between $PM_{2.5}$ concentrations and mortality, but a clear causal link has not been established in the scientific community. In fact, the components of $PM_{2.5}$ that may be associated with adverse health effects are yet unknown, but evidence suggests that carbonaceous particles are more toxic than inorganic particles such as those derived from ammonia and nitrate or sulfate.

Based on our review of literature documenting the development and testing of the simplistic model employed by Hill et al. (2019), we conclude that the model is not able to faithfully reproduce $PM_{2.5}$ impacts estimated by more complex state-of-the-science air quality models. In fact, its performance is at its worst for the very $PM_{2.5}$ component (ammonium) that the Hill et al. (2019) model indicates is the largest contributor to PM mortality from corn production. This renders the modeling especially unreliable for this key PM component. Overall, the uncertainties enumerated above result in unreliable estimates of $PM_{2.5}$ exposure, mortality and related costs associated with corn production, each associated with a large range of variability.

1.5 Environmental Impacts Associated with Ethanol Production Cannot be Viewed in a Vacuum, Without Consideration of Such Impacts Associated with Gasoline Production

EPA (2018a) acknowledges its Triennial Report fails to address environmental impacts associated with gasoline production, but it is important not to view environmental impacts of ethanol in a vacuum given the biased view this presents.

Land use for oil and gas production is extensive. In 2011, the direct footprint of oil and gas production was approximately 1,430,000 acres (Trainor et al. 2016). By 2040, Trainor et al. (2016) estimate the direct footprint of oil and gas production will be approximately 15,890,000 acres.

Habitat fragmentation from oil and gas production is also high and is known to decrease biodiversity (Butt et al. 2013). For example, the fragmentation caused by the dense placement of over 55 pads per square mile in Texas is known to cause a reduction in habitat quality for lizards in the short term (Hibbitts et al. 2013), while in the long term, habitat restoration after the removal of oil and gas infrastructure does not eliminate adverse effects to biodiversity (Butt et al. 2013).

Figure 3: Illustration of Habitat Fragmentation in Jonah Field, Wyoming from Oil and Gas Production.

SOURCE: EcoFlight (USDA 2012)

Oil and gas products, production fluids, and refinery effluent have negative impacts on soil and water quality and flora and fauna when released in the environment (EPA 1999, Wake 2005, Pichtel 2016). The toxicity of crude oil and its individual components has been well studied and these products are known to have negative impacts on wildlife depending on the exposure and dose received (Interstate Technology & Regulatory Council [ITRC] 2018).

Production water, fracking fluids, and refinery effluent, though less well-studied, have also been found to have adverse effects on plants and wildlife, resulting in decreased populations and biodiversity (Wake 2005, Pichtel 2016).

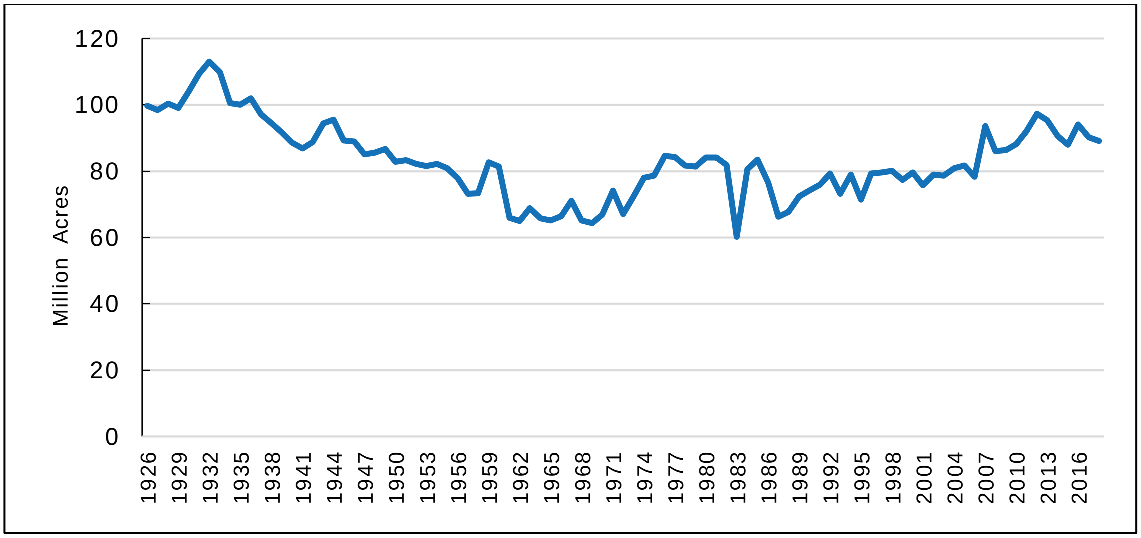
American Petroleum Institute (API) reported approximately 10.8 million gallons of oil were spilled into U.S. Navigable Waters from 1997-2006 with the amount spilled per year varying from 466,000 (2005) to 2.7 million (2004). This figure clearly does not include the Exxon Valdez spill in Alaska in 1989 or the Deepwater Horizon spill in 2010. National data suggest that spills from unconventional oil and gas may amount to one million gallons each year (Patterson et al. 2017). These data are exclusive of major offshore releases and incidents.

The findings and conclusions summarized above and set forth in the remainder of this report are subject to the limitations stated in Section 7.

2. ACRES PLANTED IN CORN HAVE REMAINED AT OR BELOW LEVELS IN THE EARLY 1930s WHILE TOTAL PRODUCTION INCREASED 7-FOLD

The total acres of corn planted in the U.S. has remained relatively stable and in fact has decreased slightly since the 1930s as shown in Figure 4, while the approximate share of U.S. corn (in bushels) dedicated to production of ethanol has increased from 4% in 1986 to 38% in 2015 and currently to approximately 50% in 2018 (USDA-ERS 2019b).

Figure 4: Total U.S. Planted Acres of Corn Per Year (million acres).

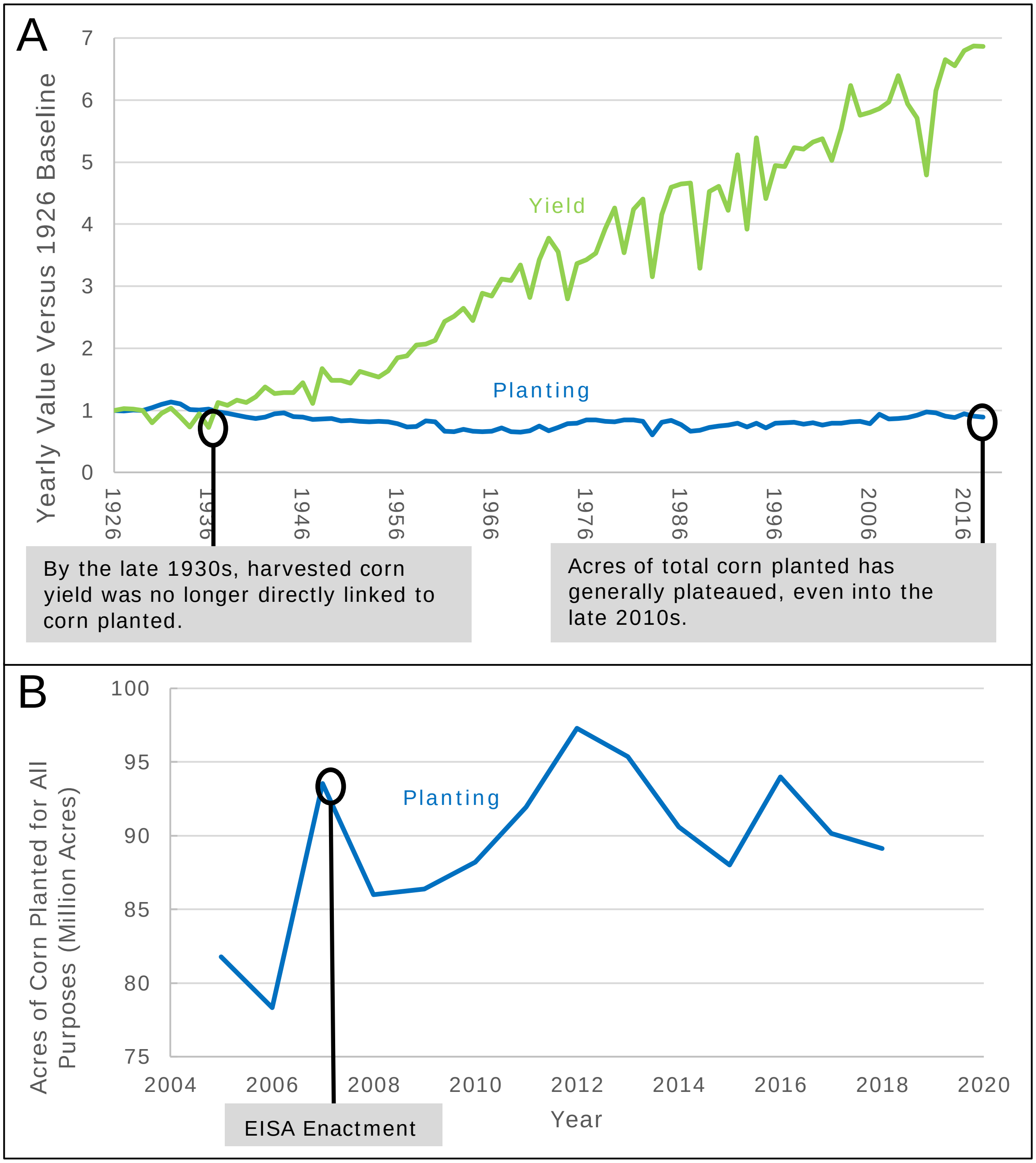


(Source: USDA 2019)

Even as the total corn acreage has been relatively stable or has slightly decreased since the early 1930s, the yield in bushels per acre during this same approximate period has increased dramatically as illustrated by Figure 5: A) Annual Yield in Bushels of Corn Per Acre and Annual Acres Planted in Corn Versus 1926. B) Annual Acres of Corn Planted 2004-2018.

These statistics reported by the U.S. Department of Agriculture (USDA) are a positive sign of the ability of farming practices to become more efficient and optimized to generate more yield without adding additional acreage. Also noticeable is that the stability of farming acreage and continued increase in yield extends into the last decade, following the enactment of the EISA. In 2018, 4.7% fewer acres of corn were planted for all purposes in the U.S. as compared with 2007, even though the approximate percentage of corn for ethanol versus other uses has increased. There was regional variation in changes in corn planting; for example, comparing data from 2017 with 2007, approximately two million fewer acres of corn were planted for all purposes in Illinois, with approximately 860,000 additional acres in North Dakota. Regional changes are driven by a wide range of competing macroeconomic conditions, mostly unrelated to ethanol production, including the relative value of crops like spring wheat and cotton, or changes in corn outputs from other countries. Indeed, the EPA confirmed that, for a variety of reasons, even the proposed 2019 RFS renewable volume obligation standards would not be expected to result in an increase in farming acreage (EPA 2018b).

Figure 5: A) Annual Yield in Bushels of Corn Per Acre and Annual Acres Planted in Corn Versus 1926. B) Annual Acres of Corn Planted 2004-2018.



(Source: USDA Crop Production Historical Track Records, 2019)

According to 2018 USDA projections, annual U.S. corn production is anticipated to surpass 15 billion bushels by 2025, while the USDA projects a 2.1-million acre decline in planted corn acres for 2018/19 (Capehart et al. 2018, USDA-ERS 2018b). Schnepf and Yacobucci (2013) cite the following projections by USDA and industry for future increases in corn yield: USDA predicts yields will reach about 240 bushels per acre by 2050 (overall increase of 55% over the 37-year period), whereas the outlook from biotechnology seed company Monsanto is an increase of 300 bushels per acre by 2030, (overall increase of 93% over the 17-year period).

The continued trend of decreases in farmable acres and increases in yield will likely continue to some stable equilibrium that will be controlled by economic and general land resource conditions. There appears to be little or no discussion in reports and documents, such as EPA (2018a), Lark et al (2019) and others, of the significance of these trends.

3. STUDIES HAVE FAILED TO ESTABLISH A QUANTITATIVE RELATIONSHIP BETWEEN THE RFS AND LUC

3.1 Overview of LUC and Environmental Impacts

In this section we first present a discussion of the lack of evidence for a quantitative causal link between increased demand for ethanol from the RFS and LUC. Second, we present a summary of some of the largest sources of uncertainty in studies that EPA (2018a) relies on to assert that the RFS may have resulted in considerable LUC. Third, we discuss the information presented by EPA (2018a) on the topics of cropping practices as well as the role of distiller's dried grains with solubles (DDGS) in offsetting LUC potentially associated with the RFS.

The literature attempting to relate LUC to ethanol production generally acknowledges shortcomings in some of the major data sets, and authors such as Lark et al. (2015) and Dunn et al. (2017) attempt to address these shortcomings by using advanced geospatial analysis techniques and data corrections (Lark et al. 2015, Dunn et al. 2017). Importantly, studies relied upon by EPA (2018a) to quantify LUC around the time of enactment of the RFS are based on unreliable data and likely overestimate LUC.

Assertions made by EPA (EPA 2010, 2018a) to link LUC (including land taken out of the CRP as well as non-agricultural land converted to agriculture) to increased demand for ethanol due to the RFS cannot be substantiated by the underlying literature for a variety of reasons, including, but not limited to the following:

- There are a myriad of complex, interrelated market and non-market factors affecting farmers' decisions on land use and a thorough assessment of the causative factors was not undertaken in the literature cited by EPA (2018a).
- Many studies do not differentiate among crop type (e.g., corn and soy) when looking at LUC and thus it is not possible to establish a causal linkage between LUC and demand for ethanol versus demand for biodiesel from those studies.
- Most studies of LUC are regional or state-specific and there is substantial inconsistency between studies regarding the geographical area of focus. This inconsistency precludes arriving at broad regional or national conclusions. For example, several studies focus on LUC in the Prairie Pothole Region due to this region's environmental fragility; whereas other studies assessed the "western corn belt", "lake states", or the entire continental United States.
- Many studies focus on specific land use types prior to conversion to agriculture (e.g., grassland, wetlands, or land in the CRP) and thus are not inter-comparable.
- Increased demand for all uses of corn may be met via either expansion of agricultural land onto previously uncultivated land (extensification) and by increased production from existing land (intensification). Intensification does not result in LUC and EPA (2018a) does not adequately represent the role of intensification in mitigating the propensity for extensification and LUC.
- Use of corn in ethanol refining produces substantial amount of DDGS and the use of DDGS as a substitute for corn as livestock feed reduces the demand for corn as livestock feed. This issue is not adequately accounted for in the assessment by EPA (2018a) of the potential role of RFS in LUC.

- The literature assessing LUC relative to the RFS generally fails to consider the considerable loss of agricultural land in urban areas and the role this loss may have in extensification elsewhere.

EPA (2018a) reviewed a wealth of information documenting LUC to biofuel crops and potential environmental impacts, but the report presents no coherent arguments or convincing lines of evidence of: (1) a quantitative relationship between ethanol production spurred by increase demand from the RFS and the documented LUC, or (2) quantitative impacts to ecosystems, wetlands, or wildlife. EPA (2010 and 2018a) reference numerous studies of LUC around the time of the enactment of the EISA. Many of these studies combine data over the period pre- and post-2007, making it difficult or impossible to confidently associate observed LUC to the time the RFS came into effect. Many authors also simply infer that there is a relationship between LUC and the RFS without any meaningful exploration of the market drivers for such change. In fact, EPA (2018b) asserts that historically the annual RFS requirements have not driven increased ethanol production and consumption. EPA asserts that this is due to the fact that consumption of ethanol has remained fairly steady since 2013 (when the 10% ethanol/gasoline blend became the predominant fuel), yet corn starch ethanol production has continued to rise well beyond the volumes required by the RFS standard, driven by favorable export markets. Ethanol exports more than doubled over the 2013-2017 period from about 0.62 billion gallons to 1.72 billion gallons (US EIA 2018).

Irrespective of market drivers, EPA (2018a) acknowledges that attributing the causes of land use change to any one factor, including the RFS, is difficult and speculative. Interestingly, EPA (2018a) acknowledges many of these shortcomings, especially in their concluding statement that "*we cannot quantify with precision the amount of land with increased intensity of cultivation nor confidently estimate the portion of crop land expansion associated with the market for biofuels*".[1] EPA (2018a) acknowledges that contributing factors to LUC include market dynamics such as crop prices and input prices (e.g., fuel, transportation costs, costs of equipment, etc.) and nonmarket costs such as those resulting from adverse weather and pests. EPA (2018a) further acknowledges that these and other factors influence land use change and that these factors may be "*coincident with the passage of EISA and therefore correlated in an empirical analysis*".[2] A fundamental problem with many of the studies cited by EPA (2018a) is that they focus on establishing correlations, or simply temporal associations between observed LUC and the RFS, and do not establish causation. EPA (2018b) succinctly summarizes the issue of relating LUC to the RFS as follows: "*…there is no scientific consensus about how to accurately and consistently attribute land use change in the context of biofuels*".[3]

3.2 The Impetus for LUC is Influenced by Complex Factors; and the Influence of the RFS is Poorly Understood and Likely Weak

EPA (2018a) identifies LUC as one of the primary drivers of potential environmental impacts from increased biofuels production, and they devote an entire section to the topic. However, EPA (2018a) also acknowledges the weakness and lack of certainty in many reports that attempt to establish a quantitative link between the RFS and LUC. For example, EPA (2018a) points out that the U.S. Department of Agriculture National Agricultural Statistics Service (USDA NASS) data indicate increases in corn crops but in the absence of comprehensive land classification "*it is impossible to know whether these increases came from existing*

1 EPA (2018a) at page xi

2 EPA (2018a) at page 22

3 EPA (2018b) at page 16

agricultural lands or new lands that were not recently in cultivation".[4] EPA (2018a) additionally notes weaknesses in empirical approaches in general, including difficulty in comparing observations and differences in how measured attributes are defined. Consequently, EPA (2018a) acknowledges that it is difficult to attribute the causes of land use changes, including where such changes are coincident with the passage of the EISA.

Several authors have examined LUC from the standpoint of decisions made at the individual farm level. Wang et al. (2017) conducted surveys of 3,000 randomly selected farmers in 37 counties in South Dakota and 20 counties in North Dakota to gain an understanding of the relative importance of different factors affecting land use decisions, and how that relative importance changes with operator and farm characteristics. The results of their survey indicated that the importance of crop output and input prices, innovations in cropping equipment, and weather patterns all increase closer to the economic margin. The authors also found that highly sloped areas are more sensitive to crop prices and crop insurance policies than less sloped land and that as farm size increases, farmers are more sensitive to policy issues and technological innovations (Wang et al. 2017).

Claassen et al. (2011) assessed the effect of farm policy on LUC and found that crop insurance, disaster assistance, and marketing loans contributed to a 2.9 percent increase in cropland acreage between 1998 and 2007 in the northern plains (Claassen et al. 2011). Miao et al. (2015) found that crop insurance reduced the effective cost of land conversion by stabilizing crop revenues (Miao et al. 2016).

Efroymson et al. (2016) use classical causal analysis to elucidate shortcomings of existing studies of the relationship between biofuels policy and LUC. The authors point out that such studies are often based on assumptions that the production of feedstock for biofuels results in the increase in demand for food crops, which in turn, results in an increase in crop prices and expansion of the total area devoted to agriculture; and that this cascading process results in the loss of areas of natural vegetation, including grasslands. EPA (2018a) acknowledges the general premise by Efroymson et al. (2016), describes the methods the authors used, but does not describe the authors' principal conclusion that for LUC, single lines of evidence considered individually are insufficient to demonstrate probable cause. Many of the studies cited by EPA (2018a) in describing a putative relationship between the RFS and LUC indeed focus on single lines of evidence such as the temporal association between LUC and the enactment of the RFS, correlations between LUC and farm proximity to ethanol plants, or LUC and increased production of corn.

Fausti (2015) explored the causal linkages among genetically modified corn, ethanol production, and corn production, hypothesizing that genetically modified corn allowed for the expansion of corn acreage, increased corn production incentivized increased ethanol production, and the RFS allowed this economic feedback mechanism to intensity (Fausti 2015). The author examined pre-RFS data (1996-2000) as well as post-RFS data (2009-2013) and found that the policy-induced [RFS] increase in ethanol production after 2006 had a statistically significant and positive effect on change in corn acres planted. However, although this relationship was statistically significant, Fausti (2015) found that the "policy-induced" change was responsible for only 0.69% to 0.88% percent of the change in corn acres planted.

One line of evidence for a link between RFS and LUC that has been explored by several authors is the relationship between increased acres in corn or LUC and proximity to ethanol

[4] EPA (2018a) at page 21

plants. EPA (2018a) asserts that "*The finding of higher rates of conversion closer to the biorefineries is important and suggests a causal link*".[5] In support of this assertion, EPA (2018a) cites "Motamed and Williams (2016)".[6] EPA (2018a) also states that "*for instance* [Motamed et al. 2016]*, estimated that for every 1% increase in an area's ethanol refining capacity, its corn acreage and total agricultural acreage increased by 1.5% and 1.7%, respectively".*[7] *However, EPA (2018a) ignores the authors' own caveats about interpretation of this finding.* In particular, the authors implicate the observed spatial linkages to food and animal feed, as well as ethanol production, conceding that "*[t]hese outcomes may reflect the efficient response of different producers to new economic incentives, but any externalities associated with these evolving arrangements remain unknown*".[8] In other words, no causal link to the RFS was established.

Wright et al. (2017) is cited several times by EPA (2018a) to provide evidence of the association between land use change (loss of grasslands) and refinery location. In particular, Wright et al. (2017) note that approximately 2 million acres of grassland was converted to row crops within 50 miles of a refinery between 2008 and 2012. However, EPA (2018a) again does not acknowledge a major shortcoming of the study, namely, the authors' admission that their study "*did not consider potential effects of other explanatory variables".*[9] The paper also discussed the errors in the data itself, stating that the "*conversion of non-cropland to cropland was mapped correctly over 70% of the time* " which means that it was mapped incorrectly 30% of the time, a considerable percentage.[10]

Li et al. (2018) examine the determinants of change in corn acreage and aggregate crop acreage as a function of the establishment of ethanol plants and changes in crop prices in the United States between 2003 and 2014. In this nationwide study, the authors report that corn acreage is fairly inelastic with respect to both changes in nearby ethanol refining capacity as well as changes in crop prices (Li et al. 2018). Unlike previous studies of the relationship between LUC and ethanol refinery location that have regional focus, Li et al. (2018) base their findings on the analysis of data for 2,535 counties in the contiguous United States. Li et al. (2018) found that a 1% increase in ethanol capacity in a county was associated with approximately 0.03% to 0.1% increase in corn acreage in that county and a 1% increase in corn price was associated with an approximately 0.18% to 0.29% increase in corn acreage in a county. The authors conclude that previous studies may have overestimated the effect of the proximity of ethanol refineries on planting of corn. The authors did find that the expansion in corn ethanol alone, all else being equal, resulted in a 2.9-million-acre increase in acres planted in corn in 2012 relative to 2008. Critically, however, they noted that most of the increase came from conversion of other crops to corn rather than LUC to corn from a non-agricultural land use. Li et al. (2018) also refute previous studies that purported to show considerable and irreversible LUC to corn, and they recognize that the overall effect of corn ethanol production on total crop acreage was negligible (Stein 2018).

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[5] EPA (2018a) at page 35

[6] This study is mis-cited by EPA and should have been Motamed et al (2016). See Section 8 *References* of this report for full citation.

[7] EPA (2018a) Box 3 at page 53

[8] Motamed et al. (2016) at page 741

[9] Wright et al. (2017) at page 9

[10] Wright et al. (2017) at page 3

A review of the above studies indicates that a causal relationship between the RFS and LUC has not been definitively established, and to the extent there is a causal linkage, the relationship is likely weak. These studies as well as EPA (2018a) do not consider in a quantitative way, the potential role of agricultural land loss on extensification. Although EPA (2018a) present some information on agricultural land loss, these studies are not discussed in any detail nor is the potential relationship to extensification.[11] American Farmland Trust estimates that between 1992 and 2012, almost 31 million acres of agricultural land were lost to development—an average rate of loss of 1.55 million acres/year (Sorensen et al. 2018). By comparison, Li et al. (2018) in their nationwide study noted an increase of 2.9 million acres in 2012 as compared to 2008 (an average increase of 725,000 acres per year). It is clear that farmland loss is considerable and very likely affects extensification.

3.3 Studies Relied Upon by EPA (2018a) to Quantify LUC Around the Time of Enactment of the RFS Are Based on Unreliable Data and Likely Overestimate LUC

One of the most pervasive issues in many studies of LUC around the time of the enactment of the RFS is reliance on data sets that have proven to be inaccurate. Some of the key publications that present estimates of LUC post-2007 and were relied on by EPA (2018a) include the following:

- Wright and Wimberley (2013) reported that between 2006 and 2011, based on an analysis of USDA's National Agricultural Statistics Service's Cropland Data Layer (CDL), there was a 1.0-5.4% annual increase in the rate of change of WCB grasslands to corn and soy with total LUC of 530,000 ha (Wright and Wimberly 2013).
- Johnston (2013) assessed wetland to row-crop transition rates in the Dakotas by geographical information system analysis of the intersection of CDL with US Fish & Wildlife's National Wetlands Inventory (NWI) and the U.S. Geological Survey's National Land Cover Database (NLCD) and reported an annualized loss rate of 0.28% (5,203 ha./yr. over a 25-32 year period for NWI data) to 0.35% (6,223 ha./yr. over a 10 year period for NLCD data) (Johnston 2013).
- Lark et al. (2015) analyzed LUC nationwide during the period 2008-2012 using CDL, calibrated with ground-based data from USDA's Farm Service Agency (FSA), and further refined using data from the NLCD. They reported that 7.34 million acres (2.97 million ha.) of previously-uncultivated lands became utilized in crop production while during the same period 4.36 million acres (1.76 million ha.) of existing cropland were abandoned with most of this being land enrolled in the CRP. They also reported that 1.94 million acres (785,000 ha.) of converted lands were planted in corn as a "first crop."
- Morefield et al. (2016) studied LUC using the USDA's CDL over the 12-state Midwest Region and report that between 2010 and 2013, 530,000 ha. (1.3 million ac.) of land formerly in the CRP were converted to row crops with the "vast majority" of these lands converted to soy and corn (Morefield et al. 2016). Of this 530,000 ha., 360,000 ha. (890,000 ac.) were grassland, 76,000 ha. (188,000 ac.) were wildlife habitat, and 53,000 ha. (131,000 ac.) were wetland. They further report that areas in the Dakotas, Nebraska and southern Iowa were hotspots for LUC.
- Mladenoff et al. (2016) assessed LUC in the Lakes States (MN, WI, and MI) and determined that during the period 2008-2013, 836,000 ha. (2,066,000 ac.) of non-agricultural open lands were converted to agricultural use, with conversion to corn

[11] EPA (2018a) Figure 14 at page 33

accounting for 480,000 ha (1,186,000 ac.) (Mladenoff et al. 2016). The authors used USDA's CDL data but combined shrubland and grass/pasture classifications into a single "open land" classification and combined wetland/forest into a single class.

- Wright et al (2017) assessed grassland losses as a function of proximity to ethanol refineries over the period 2008-2012 using USDA's CDL and found that almost 4.2 million acres (1.7 million ha.) of arable non-cropland was converted to crops within 100 miles of refinery locations, including 3.6 million ac. (1.46 million ha.) of converted grassland. Their analysis was based on applying a bias correction factor as per Lark et al. (2015) and making other adjustments.

A major shortcoming of these studies is that the primary data set relied on (CDL) is poor at differentiating between non-crop land classifications. Some authors acknowledged and attempted to correct for this problem to varying degrees. These shortcomings limit the confidence of conclusions regarding the form of the conversion, and even whether actual land use conversion has occurred in some areas.

An illustration of the effect of CDL data uncertainties on many studies relied upon by EPA (2018a) is a paper by Dunn et al. (2017). These authors examined data for 2006-2014 in 20 counties in the PPR using the CDL, a modified CDL dataset, data from the National Agricultural Imagery Program, and in-person ground-truthing, and conclude that analyses relying on CDL returned the largest amount of LUC by a wide margin. They further conclude that errors associated with CDL-based analyses are a major limitation of conclusions drawn from such analyses. In fact, the authors conclude that "*the amount of hectares in the potential error associated with CDL-derived results is generally greater than the number of hectares the CDL-based analysis determined had undergone a transition from grassland, forested land, or wetland to agricultural land".*[12] This suggests that errors in classification inherent in the CDL can result in uncertainty bounds that are of a larger magnitude than the estimates of LUC.

As an example, Dunn et al. (2017) point out that the findings reported by Lark et al. (2015) contradict USDA data indicating that cropland area has remained almost constant during the period 2008-2012. Dunn et al. (2017) is of particular interest because the study focused on the PPR, which has received the greatest attention due to documented ecosystem impacts from habitat loss and wildlife impacts to sensitive species, including population declines of prairie-dependent birds. It is interesting to note that EPA (2018a) acknowledges the specific conclusions reported by Dunn et al. (2017) by stating that adjustments to data made by Dunn et al. (2017) "*led to much lower estimates of land use than either unadjusted CDL and the NAIP for almost all counties examined* [*in the PPR*]".[13] Despite this explicit acknowledgment, EPA goes on to state that "*Nevertheless, these earlier studies* [referring to the studies critiqued by Dunn at al. (2017)] *qualitatively agree with patterns reported in more recent national studies".*[14] EPA's use of the term "qualitatively agree with patterns" in the context of studies that are attempting to quantify LUC after 2007 has little meaning and is misleading to the extent it suggests agreement between studies where little to no such agreement exists.

Table 1 presents a summary of selected results on the analysis conducted by Dunn et al. (2017).

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[12] Dunn et al. (2017) at pages 8 and 9

[13] EPA (2018a) at page 35

[14] EPA (2018a) at page 35

Table 1: Summary of Selected Results as Reported by Dunn et al (2017).

	Forest to Cropland (1000 ha.)			Wetland to Cropland (1000 ha.)		
	Dunn et al. (2017)		Lark et al. (2015)	Dunn et al. (2017)		Lark et al. (2015)
State	NAIP (2013)	CDL	modified-CDL	NAIP (2013)	CDL	modified-CDL
MN[a]	1.7	249	5.6	0	38	10
ND	0.83	222	0.44	0.01	25	7.4
SD	1.2	94	0.47	0	47	5.1
TOTAL	3.73	565	6.51	0.01	110	22.5

[a]Includes forest and grassland that was converted to cropland.

CDL data has 30 m resolution and is tested for inaccuracy each year. The accuracy of the CDL data varies yearly and regionally, which is why authors like Lark et al. (2015) make modifications to the data in an attempt to make it more accurate. Dunn et al. (2017) tested the accuracy of the modifications used by Lark et al. (2015) using NAIP data (see Table 1). NAIP data are images that are have 1 to 2-meter resolution and allow side-by-side viewing across years with high levels of accuracy. Dunn et al. (2017) found that even with the corrections that Lark et al. (2015) made to the CDL data, the modifications produced "*less land flagged as undergoing LUC but the result may not be any more accurate than a result produced without any modification".*[15] These results suggest that for the areas assessed, estimates using only uncorrected CDL data may overestimate actual LUC by a factor of 150 for forests and a factor of 11,000 for wetlands.

Further, EPA (2018a) mischaracterizes the accuracy of the CDL data[16], as the Agency states that CDL accuracies are generally > 90% for corn and soy and cites a study by Reitsma et al. (2016) in support of that assertion; however the accuracies found in the article were actually much lower than 90% for croplands (Reitsma et al. 2016). Reitsma et al (2016) used high resolution imagery to distinguish between cropland, grassland, non-agricultural, habitat, and water body land uses based on data from 2006 and 2012 in South Dakota. They found that cropland accuracy ranged from 89.2% to 42.6% depending on whether there was more cropland than grassland or the reverse. The authors chose data from South Dakota because the state represents a climate transition such that row crops predominate in the eastern portion of the state and grasslands predominate in the western portion of the state; the change in the dominant vegetation allowed them to examine how the surrounding habitat affected accuracy (Reitsma et al. 2016). The authors state that CDL errors that are inherent to the data sets introduce uncertainty into land-use change calculations. EPA's (2018a) failure to recognize the difference in CDL accuracy is especially important since many authors have documented that most of the observed LUC since 2007 has occurred at the margins of cropland/grassland transition areas. While EPA (2018a) falls short of addressing those specific data set concerns, EPA (2018b) recognizes that although satellite imagery can provide information on the types of crops grown on a given parcel of land in a given year, there is no nationwide system for tracking how crops from a particular parcel of land are used, whether for domestically or internationally consumed biofuels or feed or other uses. Thus, as EPA determined, its Triennial Report "did not purport to establish any causal link between the RFS . . . and increased crop cultivation."

[15] Dunn et al. (2017) at page 10

[16] EPA (2018a) at page 32

3.4 Recently Released Research Purporting to Establish a Quantitative Link Between the RFS and LUC is Poorly Documented and Flawed

A recent presentation of research results by Lark et al. (Lark et al. 2019) appears to be an ambitious effort to establish quantitative causal linkages between enactment of the RFS as a policy to a variety of environmental outcomes using a series of interlinked models. However, their approach rests on the assumption that the price of corn is heavily influenced by increased demand for ethanol due to the RFS, yet the authors ignore other important factors that could be equally or more important. Nor can they differentiate between price drivers associated with global vs. domestic ethanol demand.

The modeling effort begins with estimates of increased demand for corn for ethanol and effects of the increased demand on the price of corn. The authors then model the effect of this increased demand on crop intensification and extensification and abandonment. The authors then apply a “suite” of models, including what they describe as “causal economic models” to evaluate the resultant land use changes as well as the following environmental outcomes: NO_2 emissions, carbon emissions, and consumptive water use.

With respect to the effect of RFS implementation in 2007 on LUC, the authors conclude that during the period 2008-2016, the RFS resulted in an annual average increase of 6.9 million acres of corn planted on existing cropland. In addition, the authors conclude that during the same period, the RFS resulted in an annual average increase of 2.8 million acres of corn planted on new cropland (i.e., cropland converted from other land cover types), or 43% of the total increase in new cropland observed over the period. The authors attribute these changes to a 30% increase in price of corn attributable to ethanol demand created by the RFS.

The authors attempted to construct the counterfactual case; that is, simulate what the world would have looked like without the RFS (called the “Business as Usual” scenario) and then compare it to existing conditions in order to obtain and isolate the effects of the RFS. However, when a counterfactual is posed that is too far from the real-world data, conclusions drawn from even well-specified statistical analyses become based on speculation and indefensible model assumptions, rather than empirical evidence. Unfortunately, standard statistical approaches assume the veracity of the model rather than revealing the degree of model-dependence, so this problem can be hard to detect. It is well understood that the greater the distance from the counterfactual to the closest reasonably sized portion of available data, the more the counterfactual depends upon model assumptions and inferences. The seemingly large effects of the RFS reported by the authors are simply their comparison between reality and a manufactured counterfactual situation which may or may not reflect a realistic alternative state.

The authors’ entire analysis rests on estimating price increases attributable to RFS, and that is the primary weakness evident in the work. The pricing model drives the rest of the analysis. By not examining other model specifications, the inherent assumption regarding the association of prices to the RFS remains speculative. In fact, corn prices over the period of analysis were affected by a variety of other factors. For example, rapid economic growth in developing countries led to growing food demand and a dietary transition from cereals toward more animal protein. As a result, global consumption of agricultural commodities has been growing rapidly. Further, most of the increase in corn prices has been driven by higher oil prices. Figures 6 and 7 show nominal prices of West Texas Intermediate crude ($/bbl) and corn ($/bu) for the latest 20-year period. The shaded areas reflect US recessions.

Figure 6: West Texas Intermediate Crude Prices ($/barrel).

(Source: Macrotrends. n.d.)

Figure 7: US Corn Prices ($/bushel).

(Source: Macrotrends. n.d.)

Regarding the ability to "measure" land use change, Lark et al. (2019) explicitly recognize many problems with spatial data interpretation and state that land use change was mapped at the field level using the updated recommended practices by Lark et al (Lark et al. 2015). However, the specific data sets used are not disclosed, and there is no description of how the "recommended practices" were applied. The authors also do not provide an assessment of whether and how the "recommended practices" improved estimates of LUC; rather they simply present the results of their analysis. In addition to not presenting a full description of the methods used, the authors rely on at least some data sets that are not publicly available, therefore limiting the ability of a third party to replicate their work. For example, the authors state that their analysis relies on a database built using field boundary data from the 2008 USDA Common Land Unit (CLU) among other data sources. The CLU database is compiled by the USDA FSA and is not in the public domain.[17]

3.5 EPA (2018a) Failed to Adequately Account for the Role of Cropping Practices and Production of Distillers Dried Grains with Solubles (DDGS) at Ethanol Refineries as Important LUC Offsetting Factors

Numerous authors cited by EPA (2018a) who have researched LUC or increasing corn production, and the relationship of these two phenomena to ethanol production have acknowledged that much of the observed change (either LUC to agriculture or increasing corn) may be attributable to cropping practices rather than conversion of non-agricultural land to corn production. The primary cropping practices that may contribute to increased production of corn, without implicating conversion of noncropland to row crops, are switching fields to corn from other crops and double cropping of corn. The use of DDGS also reduces the need for additional acreage of corn, which is often overlooked in analysis of LUC. Similarly, EPA (2018a) fails to discuss the role of DDGS in potentially offsetting market forces that may contribute to LUC occurring to meet demand for corn for ethanol.

3.5.1 Cropping Practices Have a Major Role in Meeting Increased Demand for Corn

EPA (2018a) acknowledges the potential significance of cropping practices by citing, among other studies, a study by Ren et al. (2016) in eastern Iowa that examined changes in corn and soybean rotations around 2017 and found that the most common rotation over the period 2002-2007 was corn/soy, but this rotation was not evident in 2007 and 2012 (with 59% of the area that had been in rotation prior to 2007 was in two or more years of continuous corn after 2007). The most important conclusion reached by Ren at al. (2016) is ignored by EPA (2018a): "*From our analysis, it is clear that the expansion of corn production after 2007 was realized by altering crop rotation patterns* " (Ren et al. 2016).[18] Although this study pertains to eastern Iowa it is of particular importance since Iowa is the largest producer of corn in the US (17.4% in 2018; USDA-NASS, 2019).[19]

EPA also refers to a study by Plourde et al. (2013) when discussing intensification, but EPA does not underscore the primary conclusion of these authors (Plourde et al. 2013). In assessing data for two distinct time periods (2003–2006 and 2007–2010) in a nine state "Central United States" area (states of AR, IL, IN, IA, MS, MO, NE, ND, and WS) these authors found that the total area impacted by corn production only increased slightly between the two periods, while there was a much greater increase in the intensity of continuous corn rotation patterns. Similarly, in discussion about corn acres increasing mostly on farms that were previously soy over the period 2006-2008, EPA cites Beckman (2013) "*...that increases in corn acreage from 2001-2012 resulted in a net decrease in barley, oats, and sorghum* " (Beckman et al. 2013).[20]

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[17] In fact, the FSA website states the following:
CLU is not in the public domain. Section 1619 of the Food, Conservation, and Energy Act of 2008 (Farm Bill), only allows the sharing of this data to individuals or organizations (governmental or non-governmental) certified by FSA as working in cooperation with the Secretary of Agriculture. Users of the data must be providing assistance to USDA programs, and must require access to CLU data to complete that work (USDA 2012).

[18] Ren et al. (2016) at page 157

[19] Calculated from p. 11 in USDA-NASS 2019

[20] EPA (2018a) at page 40

Although EPA (2018a) acknowledges that changes in cropping practices "could be significant," they do not provide a quantitative or even qualitative assessment of how significant cropping might be in meeting increased demand for corn for ethanol. Inadequate accounting of the role of cropping practices in discussion of ethanol and LUC contributes to the misperception that the increase in corn production to fulfill demand for corn for ethanol necessarily results in adverse LUC.

3.5.2 Production of DDGS Has Offset a Substantial Amount of Demand for Corn as Livestock Feed But this was Not Adequately Acknowledged by EPA (2018a)

EPA (2018a) states that approximately 12% of the total corn production from 2014-2016 was returned to the feed market in the form of DDGS which is produced during the distillation of corn for ethanol. EPA (2018a) also acknowledges a study by Mumm et al. (2014)(Mumm et al. 2014) who conclude that although 40% of corn grown in 2011 was estimated to be utilized in ethanol production, when the offsetting effect of DDGS is accounted for, this acreage is reduced to 25%.[21] Although EPA (2018a) cites some of the findings reported by Mumm et al. (2014), they fail to acknowledge some very important conclusions of these authors regarding potential future projections. Mumm et al. (2014) evaluate four scenarios considering the impact of technological advances on corn grain production, two scenarios focused on improved efficiencies in ethanol processing, and one scenario reflected greater use of DDGS. For each scenario, Mumm et al. (2014) estimate the land area attributed to corn ethanol. Assuming reasonable increases in corn grain yield with anticipated new yield technologies coming into play between 2011 and 2026, the authors estimate that the percentage of land devoted to corn for ethanol will be reduced from the 25% estimated for 2011 to 13% in 2026.

Irwin and Good (2013) reported that DDGS account for much of the decline in feeding of whole corn to livestock since 2007-2008. According to the National Corn Growers Association, between 1,013 and 1,222 million bushels of corn were displaced by DDGS and Corn Gluten Feed (CGF; produced by wet milling at ethanol refineries) between 2009 and 2016 (National Corn Growers Association 2019). For illustration purposes, if we assume an average yield of corn per acre per year of 125 bushels (USDA-NIFA n.d.), then over the period 2009 to 2016_DDG/CGF may have displaced ~8.1 – 9.8 million acres of corn production per year that otherwise would have gone for livestock feed. This offsetting factor is more than the 6.9 million acres (yearly average) of corn planted on existing cropland and the 2.8 million acres (yearly average) of new cropland alleged by Lark et al. (2019) to be attributable to the RFS for the period 2008-2015.

[21] Mumm et al. (2014) Box 3 at page 53

4. CHANGES IN AGRICULTURAL PRACTICES REDUCE THE LIKELIHOOD OF ENVIRONMENTAL IMPACTS TO WATER RESOURCE AVAILABILITY AND QUALITY

The relationship between corn production and water resource availability and water quality varies geographically and temporally. What is clear but not quantitatively recognized by EPA (2018a), is that advancements in farming practices and technology have reduced the negative impact of farming on the environment. Recent technological advances have resulted in considerable improvements in water use in agriculture in general, and for corn growing, as well as reducing the use of agrochemicals such as fertilizers and pesticides. These improvements have the effect of reducing the likelihood of adverse impacts to water resource availability and quality.

There is no dispute that all agricultural production is strongly tied to the availability and quality of fresh water. Farming practice is based on local and regional climatic and soil conditions which determine whether crops are grown using irrigation from surface water or groundwater sources or are non-irrigated and rely solely on precipitation. Approximately one-quarter of US cropland is irrigated (NAS 2019). The total US irrigation withdrawals for all crops in 2010 averaged approximately 115 billion gallons per day (NAS 2019). The availability of sustainable water sources, more so than any other issue, poses the greatest threat to crop productivity into the future. Corn is a water intensive crop; however, most corn grown in the US is non-irrigated, and this is recognized by EPA (2018a). Over the past decade, there has been increased use of modern and precision agriculture methods (for both water use and agrochemical application) which retain soil moisture and reduce tilling. This trend is expected to continue into the future, with increasing efficiency and effectiveness of resource use, which will result in reducing water and fertilizer needs.

4.1 The Triennial Report's Discussion of Water Use and Water Quality

Key conclusions in EPA (2018a) relevant to the RFS reset discussion include:

- The environmental impacts of increased biofuel production on water resource use and water quality were likely negative in the past but limited in impact.
- A potential exists for both positive and negative impacts in the future with respect to water resource use and availability, and impact to water quality both locally and regionally.
- Environmental goals for biofuels production could be achieved with minimal environmental impacts (including water and fertilizer/pesticide use) if best practices were used and if technologies advanced to facilitate the use of second-generation biofuels feedstocks.

These messages are consistent with our findings that the environmentally protective goals for biofuel production are highly achievable as best management practices and technological advances in farming continue to be adopted by the farming community. While challenges for fully distributing and implementing these approaches will remain in certain areas (e.g., NAS 2019), the economic drivers for implementing best practices such as increased productivity and savings derived from resource conservation, will undoubtedly continue to steer the farming community toward greater implementation of modern approaches.

The most important statements presented by EPA (2018a) are the forward-looking considerations that biofuel production can (and will) achieve environmental goals by using modern practices. EPA (2018a), however, paints a picture of negative impacts from biofuels feedstock production without using specific and conclusive data to support the claims. For example EPA (2018a):

- Asserts that increased intensity of corn production on existing cultivated land and expansion of crop land negatively impacts water quality but presents no direct evidence of a causal link.
- Does not rely on direct analysis to assess the magnitude of potential water quality impacts but instead makes general statements with no quantitative analysis that connects the water quality impact to specific areas, land, or conditions.
- Recognizes that quantitative assessments are necessary to evaluate whether increases in water demands can be directly attributed to feedstock production. However, EPA (2018a) does not provide the studies or backup to support this evaluation, rather merely speculates that negative impacts must exist.

EPA (2018a) suggests that growing corn for ethanol feedstock is a major contributor to eutrophication and hypoxic conditions in the northern Gulf of Mexico and eutrophication in western Lake Erie. EPA (2018a) attributes these conditions to substantial nutrient loading from agricultural runoff. However, the impact, if any, from corn grown for ethanol production on water quality and availability is not substantiated with data. For example, the attribution by EPA (2018a) that biofuel feedstock production is a contributing factor to these conditions appears to rely on models such as those presented by Michalak et al. (2013) that state corn production "could" be a contributing factor and LaBeau, et al. (2014) that speculate biofuel production "could" contribute to increased nutrient loading to surface water (Michalak et al. 2013, LaBeau et al. 2014).

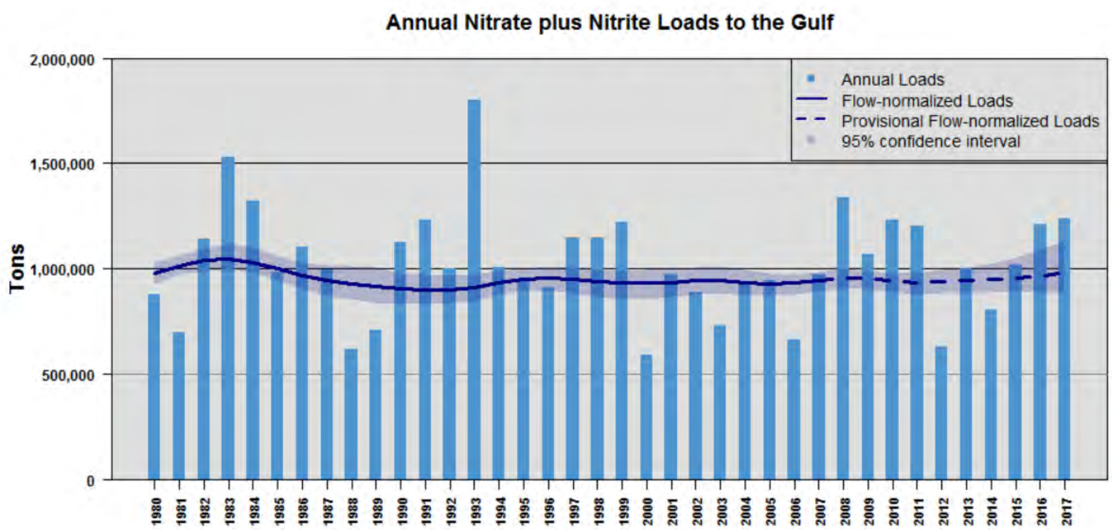
There may be no dispute that excess nutrient loading from the key watersheds that discharge into western Lake Erie and the northern Gulf of Mexico contribute to eutrophication and hypoxia; however, the watersheds are composed of a complex mix of urban and rural uses and wastewater discharges. Agricultural runoff should be considered an important component; however, the direct causal link to corn grown for ethanol production (compared to all other uses and compared to all other agricultural activities) is not substantiated. Indeed, no studies reviewed by Ramboll convincingly link increases in biofuel production to regional hypoxic conditions in surface water bodies. Such conditions have been increasing in frequency and severity since the 1950s, long before ethanol production increased.

EPA (2018a) also fails to acknowledge the importance of regional weather on the occurrence and severity of large-scale hypoxia events. For example, one major variable determining the size of the hypoxic zone (colloquially known as the "dead zone") in the Gulf of Mexico is the rate of flow in the Mississippi River, which may be highly-variable on an annual basis. The National Oceanic and Atmospheric Administration (NOAA) is predicting that the 2019 dead zone in the Gulf of Mexico will cover an area of 7,829 square miles which is close to the record size of 8,776 square miles in 2017 and more than one third larger than the 5-year average size of 5,770 square miles (NOAA 2019). NOAA states that a major factor contributing to the dead zone in 2019 is the abnormally high amount of spring rainfall that has resulted in flows in the Mississippi and Atchafalaya Rivers that are 67% above the average flows over the last 38 years. Data collected by the United States Geological Survey (USGS) indicate that because of these high flows, nitrate loads are about 18% above the long-term average, and phosphorus loads are approximately 49% above the long-term average (USGS 2019).

Finally, EPA (2018a) also fails to recognize that changes in flood-control and navigation improvements in the Mississippi River watershed during the first part of the 20th century dramatically affected the amount of flow from the upper Midwest watersheds that would enter the Gulf of Mexico without environmental buffering from natural tributaries (NOAA 2000). The higher flow rates allowed greater unimpeded flow of water containing nutrients to the Gulf of Mexico than would otherwise have occurred (NOAA 2000).

It is interesting that while EPA (2018a) relies on speculation and qualitative studies to associate corn grown for ethanol to hypoxia in western Lake Erie and the Gulf of Mexico, EPA (2018a) also reports that there has been a reduction in total nitrogen concentrations in surface water bodies in Iowa (the highest corn producing state and an area of corn growth intensification). We note that nutrient loading to the Gulf of Mexico has been relatively stable on average since at least 1980 – an important consideration as corn yield has increased during this time period (USGS n.d.) even as farmed acreage has been stable. This indicates that even during the increased use of corn for ethanol, there has been no net change to nutrient loading to the Gulf of Mexico and thus there is no support for the assertion of a direct relationship between ethanol production on the hypoxia conditions in the Gulf of Mexico. This evidence refutes claims made to the contrary by EPA (2018a).

Figure 8: Annual Nitrate and Nitrite Loading to the Gulf of Mexico 1980-2017.



(Source: USGS n.d.)

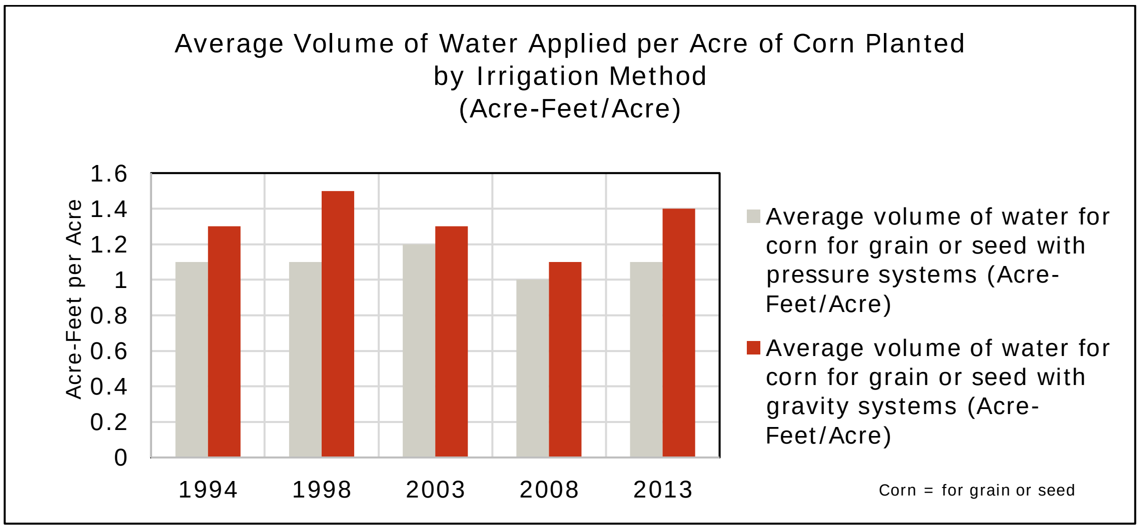
The fact that agricultural practices in general can result in nutrient runoff is acknowledged, although modern efficiencies and conservation methods have improved over time. Modern practices apply technology for increased efficiency and harness continuously improving data analysis to develop and implement best management practices. There is strong evidence that the agricultural community, including biofuel feedstock producers, are adopting modern agricultural practices (Vuran et al. 2018). EPA (2010 and 2018a) acknowledge and strongly advocate for these modern practices and note that negative impacts to environmental resources will be reduced with the use of modern approaches to tilling, fertilizer use, water use, and precision agriculture. If these practices were not being implemented, the expectation is that nutrient loading, and thus hypoxic conditions, should have been increasing along with the increased yield over the past several decades. However, the data from NOAA and the USGS show stability in nutrient loading, which would thus indicate that the net flux of nutrients has not been increasing even while crop yields may have been increasing.

4.2 Agricultural Improvements in Irrigation are Reducing Water Use

The trend of increasing yield per acre farmed extends to both irrigated and unirrigated corn crops, indicating that changes in yield are not likely attributed to irrigation alone. According to the 2012 statistics from the USDA (USDA-ERS 2018a) irrigated corn acreage represented about 25% of all irrigated acreage in western states, and about 24% of all irrigated acreage in the eastern states. Additionally, the USDA has shown that irrigation for all crops, including corn, has decreased even as the farming acreage has essentially been stable over the past 35 years. The USDA attributes this trend to improvements in physical irrigation systems and water management. The USDA also notes that significant capital investments in on-farm irrigation is continuing, particularly in the western states, where most of the irrigated farm-land is concentrated. As an indication of a positive trend in irrigation reduction, the University of Nebraska, Lincoln reports that in Nebraska (as a bell-weather of other dry western states), the percentage of all corn acreage that is irrigated has declined from a high of 72% in 1981 to 56% in 2017 (University of Nebraska 2018).

USDA data indicate that there has been no substantial change in the volume of water applied to corn crops (for grain or seed) since the 1990s (Figure 9) (USDA-NASS 2013). This stability in the average volume of water applied to corn crops, combined with the plateau in area of corn planted, suggests that the quantity of water applied to corn crops has not substantially increased since at least the 1990s, despite intensification.

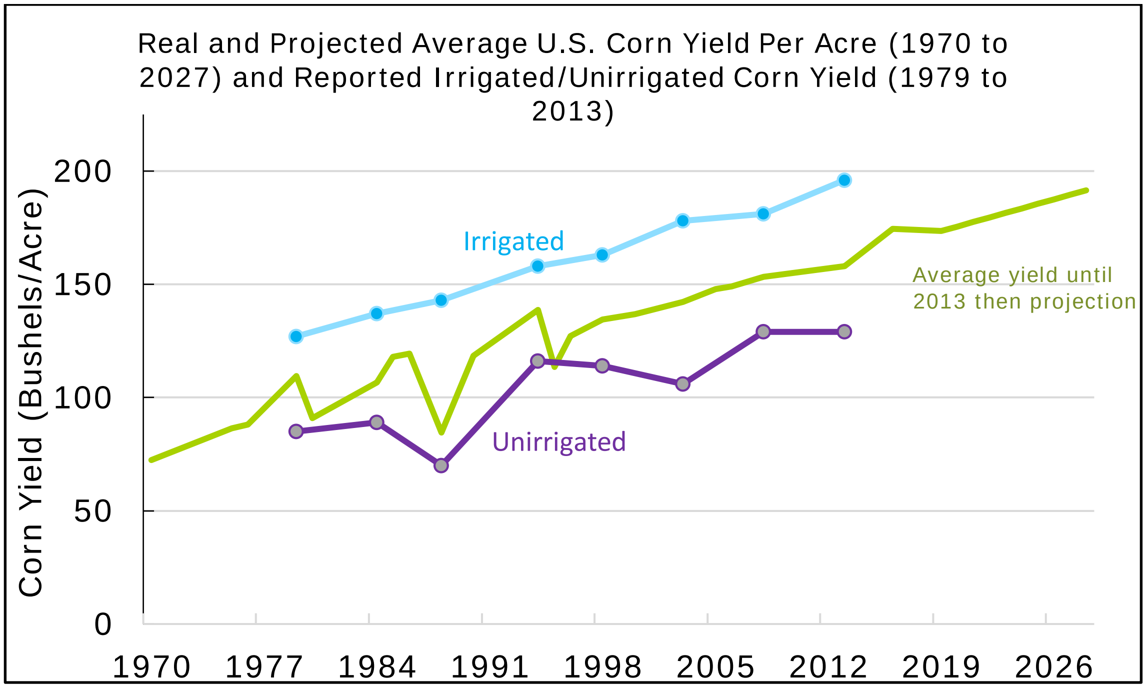
Figure 9: Volume of Water Applied to Irrigated Corn Crops Since 1994, by Irrigation Method.



(Source: USDA-NASS 2013)

Because irrigation provides a stable water resource to the farmed field (assuming the water source that supplied irrigation is also stable), crop yields on irrigated land are generally more regular (e.g., less variable and often more substantial) than for non-irrigated land (Figure 10). Note, however, that from at least 1979 to 2013, increases in yield also have been observed in unirrigated corn crops (USDA Farm and Ranch Irrigation Survey). Specifically, in 1979, irrigated land produced 127 bushels per acre on average, versus 85 for unirrigated land. By 2013, irrigated land produced 196 bushels per acre on average, versus 129 bushels per acre for unirrigated land, representing a 54% and 52% increase, respectively.

Figure 10: While Irrigated and Unirrigated Corn Crops Have Both Experienced General Increases in Yield, Irrigated Crops More Reliably Produce Higher Yields.



(Source: USDA-NASS 2013)

Regions of greatest corn production are moving eastward away from the regions of greatest irrigated water use, providing further evidence that year-to-year changes in corn planting have little to negligible impact on total U.S. water supply. For example, in 2016, the five leading states in annual corn production (Illinois, Nebraska, Iowa, Minnesota, and South Dakota) produced over 60% of the corn grown in the U.S. (USDA-ERS 2016). This statistic is a change from the 2010s when the irrigation of corn crops was even more concentrated in the drier Northern Plains (Colorado, Montana, Nebraska, Wyoming, and North and South Dakota) and dry Southern Plains (Kansas, Oklahoma, Texas) regions. In 2007, the USDA reported that the thirteen leading states in total irrigated acres for all crops of farmland, accounted for nearly 80% of all U.S. irrigated land, but that they were concentrated in arid western states (USDA-ERS 2018a). Of the top five corn-producing states, none made up more than 15% of the total U.S. irrigated acreage. The increased growth in wetter states such as Illinois and Minnesota eases the water supply demand for the total yield of all irrigated corn acres.

USDA anticipates that changes in corn production will result in appreciable yield increases (e.g., 16.1 more bushels per acre by 2028) (USDA-NASS 2017). It is therefore reasonable to expect that technological and methodological changes to farming will continue to result in significant reductions in water use per unit of corn production. Table 2 presents an overview of prevailing opportunities for water savings in irrigated agriculture.

Table 2: Technological and Methodological Improvements to Irrigation of Corn Crops.

Technological Advancement	Approximate water savings factor	Baseline scenario	Demonstrated potential yield increase	Notes
Subsurface drip irrigation	25-35%	vs. center pivot system	15-33%	Costs 40-50% higher than center pivot systems but returns on investment can accrue within 2—5 years. In 2007, only 0.1% of irrigated corn farms used this.
Rain water harvesting and storage	50+%	vs. natural soil runoff	20-52%	Includes 1) harvesting of surface runoff from roads; 2) field micro-catchment to increase fallow efficiency in rain.
Precision agriculture	13%	vs. without government-run weather network	8%	Includes use of global positioning system, geographical information systems, in situ soil testing, remote sensing crop and soil status, real-time weather info. Adoption rate slightly higher in corn belt.
Conservation structures	18%	vs. conventional agriculture	27%	Examples include grass vegetation strips. Adoption is higher in areas of highly erodible land.

(Sources: Netafim n.d., Gowing et al. 1999, Shangguan et al. 2002, National Research Council 2008, Biazin et al. 2012, Allen 2013, Barton and Elizabeth Clark 2014, Center for Urban Education about Sustainable Agriculture (CUESA) 2014, Qin et al. 2015)

Subsidized government programs offer farmers incentives to implement water conservation strategies. For example, because of prolonged drought conditions, California recently installed a network of 145 automated statewide weather stations, so that farmers could manage their water resources more efficiently (CIMIS 2019).

With the focus on drought and long-term reductions in supplied water in some states (such as California), more farms are moving away from "traditional, less-efficient application systems" (USDA-ERS 2018a). For example, the number of farms using inefficient gravity irrigation systems decreased from 62% in 1984 to 34% in 2013, converting mostly to pressure-sprinkler irrigation which is more efficient than gravity irrigation, but which still leaves room for improvement. Currently, almost 10% of farms use soil-moisture or plant-moisture sensing devices or commercial irrigation scheduling services. Sensor technology can optimize irrigation scheduling and hence increase water use efficiency. Though less than 2% of farms use simulation models right now (USDA-ERS 2018a), the anticipation is that additional large industrial farms (which make up a large volume of total yield) also will employ water use simulation models that are based on corn growth patterns and weather conditions. Adoption of these technologies will continue to grow in the U.S., and particularly in the west, where 72% of water irrigation investment takes place and farmers have recent experience with low water supply following the 2012-2016 drought.

Barriers to implementing these measures are lessening but it is recognized that issues relating to the following are still at play: (1) farmer concerns about the impact of new practices on yields; (2) tenant or lease issues that discourage the installation or use of new equipment; (3) institutional issues related to Federal Crop Insurance Program; (4) irrigation water rights laws like "use it or lose it;" and (5) cost of implementation. The Great Plains area had traditionally been risk-averse to implementing subsurface drip irrigation techniques because of the upfront costs and uncertain lifespan of the systems; however, there have been improvements in the technology and irrigators are increasingly aware of the additional incentives for water conservation and protecting water quality (Lamm and Trooien 2003).

Genetic engineering or selection for improved drought tolerant corn cultivars has also contributed to increases in corn crop productivity. Additionally, genetic breeding has shown that yields can be maintained with lower water requirements (nearly 25% reduction), in addition to studies that suggest corn crops can forego the initial irrigation without significant adverse effects to the harvest (Xue, Marek, et al., 2017). Mcfadden et al. (2019) reported that with drought being among the most significant cause of crop yield reduction, the spike in use of irrigation water to reduce such losses can be a major negative impact to water resource availability particularly in the drier western states. Even though many water-intensive crops, including corn, are grown on non-irrigated land, the use of drought-tolerant corn, which was commercially introduced in 2011, had increased to over 22 percent of the total U.S. planted corn acreage by 2016 (Mcfadden et al. 2019). More important, this percent of use was greatest in the driest corn-producing states of Nebraska (42 percent) and Kansas (39 percent). Even the less severe drought-impacted though important corn-growing states of Minnesota, Wisconsin, and Michigan saw drought-tolerant corn planting ranging between 14 and 20 percent of total acreage. There is no guarantee that drought-tolerant crops will be effective against the most severe droughts; however, this use can be seen as similar to the use of crop-insurance to protect farmers against loss while still providing product for use during low-water years. The longer-term advantage is that less irrigation water would be required even under normal water years.

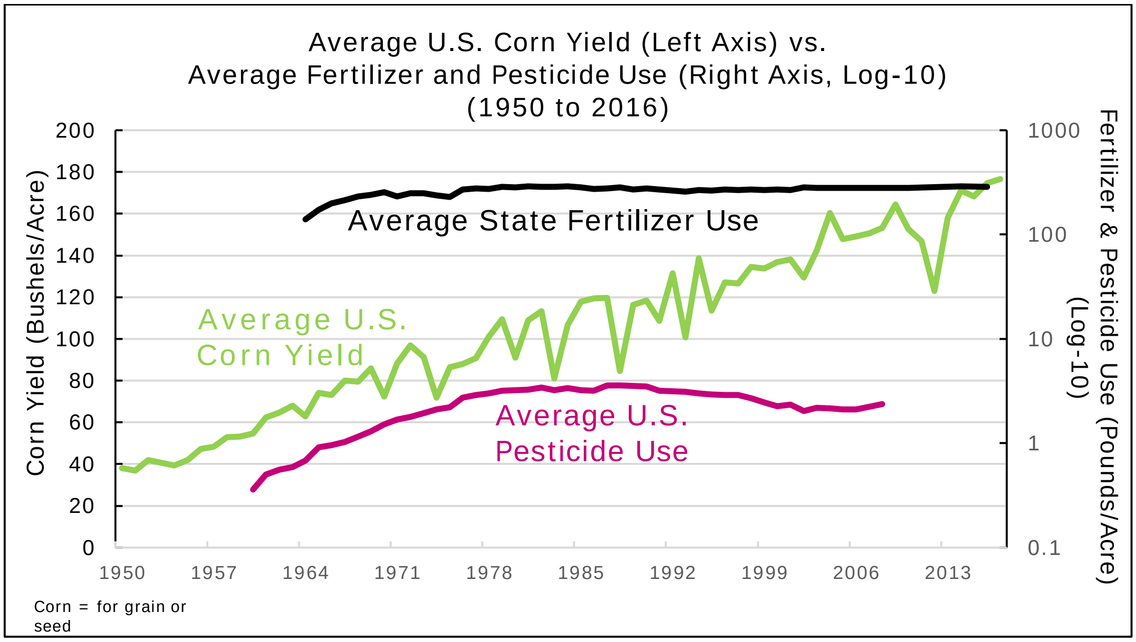
Liu, et al (2018) states that best management practices for reducing agricultural non-point source pollution are widely available even with the challenges related to the large number of agricultural producers and the spatially variable and temporally dynamic nature of the nutrient loading cycles. Greater adoption of the improved practices will rely on: (1) better identification of the higher risk areas; (2) a commitment from local, state and federal authorities to assist the farming community in applying the new approaches by allowing innovations to be implemented without unnecessary regulatory impediments; and (3) better financial incentives. Liu, et al (2018) also note that lack of information and misdirected communications can negatively impact the adoption of new techniques and encourages government, consumers, and farmers to work together to more consistently communicate the advantages of technology adoption.

4.3 Technological Improvements in Agriculture Translate to Reductions in Potential Water Quality Impacts

Government institutions including USDA and academic institutions such as California State University, Fresno have promoted research into the use of precision agriculture to reduce the need for both nutrient and pesticide use (as well as supplied water) because in addition to a reduced environmental impact, the techniques result in cost savings for farmers by improving yield per acre. In addition, the greater use of area-wide databases that provide better information and awareness of water quality conditions helps to identify areas where additional best management practices can be applied. For example, utilization of the USGS water quality mapping reports (e.g., USGS 2017) helps provide data for surface water chemistry trends (i.e., nutrients, pesticides, sediment, carbon, salinity) and aquatic ecology from 1972 to the current editions.

Recent advancement in technology for fertilizers and pesticides have reduced the use of agricultural chemicals while increases in crop yield continue. While use of fertilizer on corn typically accounts for more than 40% of commercial fertilizer used in the U.S. since the 1980s (USDA-National Resources Conservation Service [NRCS] 2006, EPA 2018c), there has been a plateau in the mass of fertilizer applied to corn crops (on average on a state-by-state basis), as well as an overall decrease in the mass of pesticide applied to corn crops (see Figure 11;(USDA-NASS 2013, Fernandez-Cornejo et al. 2014, USDA-ERS 2018c). In 1987, the average mass of pesticide active ingredient application per area of corn planted in the U.S. peaked at approximately 3.58 pounds per acre. In 1984, fertilizer use peaked at approximately 290 pounds per acre. The USDA and EPA report similar trends; for example, U.S. spending on pesticides for all crops peaked in 1998, and consumption of commercial fertilizers peaked in 1981 (Fernandez-Cornejo et al. 2014, EPA 2018c).

Figure 11: Both Pesticide and Fertilizer Use on U.S. Corn Crops Appear to Have Peaked in the 1980s



(Source: USDA [ibid.])

The application of slow released (or controlled) nitrogen fertilizer during peak uptake is one key to improving nutrient efficiency and utilization (Lal, R. (Ed.), Stewart 2018). Under optimum moisture and temperature conditions, use of slow released nitrogen fertilizer can greatly reduce leaching of nutrients. However, further research is necessary to discern the best slow release fertilizer for a given crop species (Rose 2002). Other advanced chemical technologies such as use of bioreactors, can offer additional reductions in pesticide and fertilizer in corn production. Bioreactors such as those that redirect water in farm fields through tiles to underground woodchips where nitrate is removed by microorganisms, can reduce nitrogen in run-off by 15% to 90% (Iowa Corn n.d., Christianson 2016).

Recent surveys and data from the use of the modern and technology-based agricultural management systems have shown reduced resource needs and significant cost savings (NAS

2019; Liu, et al. 2018). The USDA also has shown that a “guidance-based” system for corn production can save thousands of dollars each year with a return of investment of two to three years for this technology (USDA-NRCS 2006). Furthermore, the USDA reports that “…precision agriculture reduces environmental pollution and improves water quality by reducing nutrient runoff [while] other benefits include: improved crop yield; reduced compaction [of fields]; labor savings; and more accurate farming records.” Finally, there are fewer barriers to nearly all farmers in using precision technologies because of grants that are available for purchasing equipment and free public access to the Federal Global Position System that makes it economically possible for producers to use the new precision tools to save energy and reduce costs by improving or implementing the following: (a) yield monitoring, (b) grid soil sampling, (c) precision and variable-rate nutrient application; and (d) soil moisture monitoring. Precision agriculture technologies are quickly adopted by farmers in the United States; the rate of adoption for all precision technologies was 72.47 percent in 2010, as compared to just 17.29% in 1997 (Vuran et al. 2018). USDA found that if guidance-based farming was used on just 10 percent of planted acres in the U.S., fuel use would be cut by 16 million gallons, herbicide use would be reduced by 2 million quarts and pesticide use would lower by 4 million pounds per year(USDA-NRCS 2006). The results would be better environmental conditions and substantial increase in financial savings for the farmer/producer.

4.4 Reduction in Water Usage for Ethanol Processing

Opportunities exist for implementing water reduction programs during biofuel production. Excluding the non-fuel component, the primary processes that require water consumption in ethanol production include heating and cooling. Water losses occur through: (1) evaporation, drift, and blow down from cooling towers; and (2) blow down from boilers. Losses vary with both the ambient temperature of the production plant, and the degree of boiler condensate and blow down water reuse and recycling. Generally, dry mills use less water than wet mills. In a 2007 Renewable Fuels Association survey of 22 ethanol production facilities (representing 37% of the 2006 volume produced), dry mills used an average of 3.45 gallons of water per gallon of ethanol produced and wet mills used an average of 3.92 gallons of water per gallon of ethanol produced. Efforts to use recycled waste water are increasing and will reduce the need for using supplied water during the conversion process.

Keeney and Muller (2006) report that in Minnesota, water use by dry mill ethanol refineries ranged between approximately 3.5 and 6.0 gallons of water per gallon of ethanol in 2005 which followed a 21% reduction in water use by dry mill ethanol refineries from 1998 to 2005 (representing an annual reduction of approximately 3%). More recently, Dr. Steffen Mueller of the University of Illinois (Chicago) Energy Resources Center notes that water consumption by ethanol plants is continuing to decrease and dramatically so. Mueller (2016) documents a reduction of approximately 5.8 to 2.7 gallons of water per gallon of ethanol produced between 1998 and 2012 in dry mills.

Wu and Chiu (2011) noted additional trends that suggest decreases in the water demands of existing and new ethanol plants. Freshwater consumption in existing dry mill plants had, in a production-weighted average, dropped 48% in less than 10 years to water use rates that are 17% lower than typical mill values. Water use can be minimized even further through process optimization, capture of the water vapor from dryers, and boiler condensate recycling to reduce boiler makeup rates.

5. RECENT ESTIMATES OF HEALTH DAMAGES FROM CORN PRODUCTION ARE UNRELIABLE AND MISLEADING

A recent publication in *Nature Sustainability* (Hill et al., 2019) estimates US annual health damages caused by particulate air quality degradation from all direct farm and indirect supply chain activities and sectors associated with corn production. Although the authors do not reference the RFS, they do mention corn grown for ethanol, and the publication has been referenced by third parties in a manner suggesting that corn grown for ethanol may be associated with adverse health outcomes. Ramboll's review indicates that the conclusions presented by Hill et al. (2019) are unsubstantiated and likely overestimate adverse health impacts if any.

These "life-cycle" activities and sectors examined by Hill et al. (2019) include air emissions from farms and upstream processes that produce the chemical and energy inputs used in corn crop production: fuel, electricity, agrichemical production, transportation and distribution. Downstream activities such as corn distribution and food/fuel processing are not considered in the study. The authors develop an annual county-level emissions inventory of air pollutants for all related sectors, then apply a specific "reduced form model" (RFM) that converts those emissions into spatial distributions of annual fine particulate air concentrations (or $PM_{2.5}$) and resulting human exposure, premature mortality, and monetized health damages.

$PM_{2.5}$ comprises microscopic particles smaller than 2.5 microns in diameter, with chemical constituents that include direct (primary) emissions (dust and smoke) along with the several secondary compounds chemically formed in the atmosphere from gas precursor emissions: nitrate from nitrogen oxide (NOx) emissions, ammonium from ammonia emissions, sulfate from sulfur oxide (SOx) emissions, and secondary organic aerosols (SOA) from volatile organic compound (VOC) emissions. $PM_{2.5}$ is a concern for human health because particles of this size can penetrate deep into the lungs and enter the bloodstream, which can potentially result in both acute and chronic effects to the respiratory and cardiovascular systems. Epidemiological studies have found associations between $PM_{2.5}$ exposure and mortality and these associations are used by Hill et al. (2019) to calculate health impacts from corn production. The authors find that impacts to annual-average $PM_{2.5}$ concentrations from corn production are primarily driven by emissions of ammonia from nitrogen fertilizer.

Ramboll reviewed details of the specific RFM used by Hill et al. (2019), called the Intervention Model for Air Pollution (InMAP; Tessum, Hill, et al., 2017) to calculate ambient $PM_{2.5}$ impacts from corn production. InMAP calculates atmospheric dispersion, chemistry and removal (deposition) from direct $PM_{2.5}$ and precursor gas emissions. It then converts resulting annual $PM_{2.5}$ concentrations to human exposure metrics from which premature mortality and associated damages are determined. Hill et al. (2019) provide only an overview of the process to develop emission inventories, which limits our capacity to review. However, given the importance of ammonia emissions to the results reported by Hill et al. (2019), we enumerate well-known uncertainties involved in estimating emissions from agricultural activities. In addition, although Hill et al. (2019) did not provide explicit details on the impact assessment, we provide a summary of the key uncertainties associated with estimating health and associated costs from $PM_{2.5}$ exposures. It is noteworthy that the authors do not provide any uncertainty or sensitivity analyses that can provide important context for the interpretation of the results and conclusions.

Based on our review of Hill et al. (2019) and of Tessum et al. (2017), we draw the following conclusions:

- InMAP uses annual-average data for emissions, meteorology, and chemical/removal rates to estimate annual-average $PM_{2.5}$ impacts. Use of annual averages is inappropriate for representing processes that operate over shorter time scales ranging from minutes to several months (e.g., atmospheric dispersion and chemical formation of $PM_{2.5}$). The authors acknowledge that this weakness in their approach results in spatial errors in annual average $PM_{2.5}$ calculations. These spatial errors can significantly impact the resulting exposure and mortality estimates. The authors, however, do not present sensitivity analyses to assess the impact of the model assumptions, nor do they include any plausible range of uncertainty or variability with their modeled $PM_{2.5}$ concentration or mortality estimates.
- The 2005 modeling year upon which InMAP is based is not representative of more recent chemical conditions of the atmosphere in the U.S. because there have been significant reductions in precursor emissions that directly reduce the capacity to form $PM_{2.5}$. We estimate that this leads to an overestimate of the $PM_{2.5}$ contributions from corn production by more than a factor of 2. Therefore, resulting health and economic damages are likely overestimated.
- Ammonia emission estimates, which are the largest driver of mortality in the Hill et al. (2019) analysis, are the most uncertain aspects in any air quality modeling exercise because: (1) emissions are largely from agricultural sources that vary both spatially and temporally due to weather and farming practices; (2) many different methods are used to estimate ammonia emissions, and each can yield very different rates and exhibit a high degree of error; (3) annual average ammonia emission inventories fail to account for important seasonal variations and related complex interactions with sulfate and nitrate chemistry; (4) ignoring diurnal and intra-daily ammonia emission variations have been shown in the literature to overestimate ambient ammonia concentrations by as much as a factor of 2. These numerous uncertainties and compounding error rates call into question the estimates of emissions that drive the rest of the Hill analysis.

Based on our review, InMAP is not typically able to reproduce $PM_{2.5}$ impacts estimated by more complex state-of-the-science air quality models. In fact, its performance is worst for the very $PM_{2.5}$ component (ammonium) that Hill et al. (2019) model indicates is the highest contributor to PM mortality from corn production. This renders InMAP especially unreliable for this key PM component.

In addition to the number of significant uncertainties in all modeling aspects of the Hill et al. (2019) analysis, including the emissions estimates and the RFM InMAP modeling, there is also a significant amount of uncertainty associated with estimating health impacts from air pollution concentrations and from quantifying the costs of these health impacts.

The health impact assessment is based on a single epidemiological study that found associations between $PM_{2.5}$ concentrations and mortality. While these studies suggest that such an association exists, there remains uncertainty regarding a clear causal link. This uncertainty stems from the limitations of epidemiological studies to establish causality because these studies are based on inadequate exposure estimates and these studies cannot control for many factors that could explain the associations between $PM_{2.5}$ and mortality – which, for example, may not be related to $PM_{2.5.}$ from the source being investigated (e.g., lifestyle factors like smoking). In fact, the components of $PM_{2.5}$ that may be associated with adverse health effects are yet unknown, but evidence suggests that carbonaceous particles are more toxic, than inorganic particles such as those derived from ammonia and nitrate or sulfate.

Overall, the uncertainties enumerated above result in unreliable estimates of $PM_{2.5}$ exposure, mortality and related costs associated with corn production, each associated with a large range of variability.

6. ENVIRONMENTAL IMPACTS ASSOCIATED WITH ETHANOL PRODUCTION CANNOT BE VIEWED IN A VACUUM, WITHOUT CONSIDERATION OF SUCH IMPACTS ASSOCIATED WITH GASOLINE PRODUCTION.

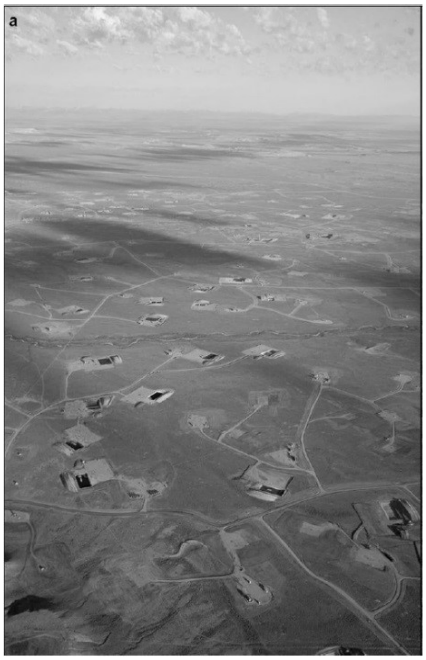
EPA (2018a) acknowledges that it fails to address environmental impacts associated with gasoline production. Spills of petroleum, gasoline, and a wide range of other fluids used in the exploration, production, and refining processes as well as land use change to support those activities all have adverse effect on water quality, ecosystems (including wetlands), and wildlife. Additionally, both conventional and unconventional oil and gas extraction place demands on water supply. Failure to address impacts associated with gasoline production relative to impacts from ethanol production does not present a balanced view of alternative energy sources and casts a negative bias on ethanol production. Parish et al. (2013) recognize the importance of understanding differences in environmental effects of alternative fuel production so that the relative sustainability of alternatives can be adequately assessed in policy-making and regulatory decisions. Parish et al. (2013) assessed negative environmental impacts through the supply chain for ethanol production and gasoline production and found that impacts from ethanol production are more spatially limited, are of shorter duration, and are more easily reversed than those associated with gasoline production. It was beyond the scope of this report to expand upon the work of Parish et al. (2013) or other comparative studies, rather this Section presents a brief description of the wide range of potential impacts associated with petroleum production stemming from land use changes as well water use and impacts to water quality.

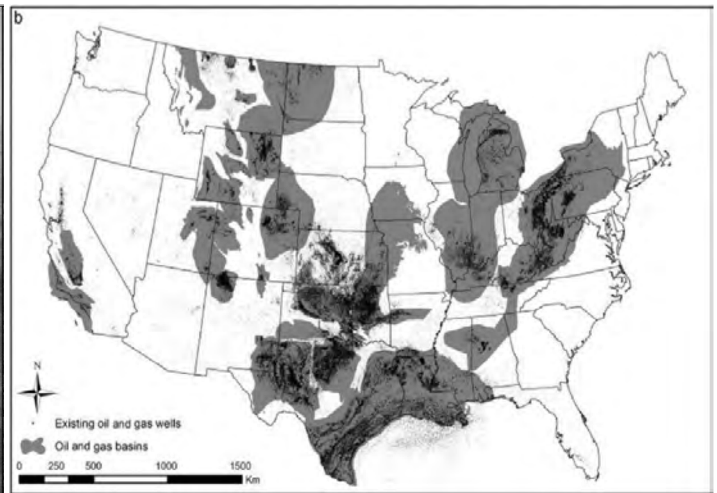
6.1 Impacts of Gasoline Production Associated with Land Use Change

Oil and gas can be extracted using conventional or unconventional (i.e., hydraulic fracturing) methods, with some resultant variability in associated land use change impacts. Both methods require the construction and maintenance of a well pad and placement of pumping machinery. To install any onshore well pad, the land must be cleared and leveled, which requires the construction of access roads in most cases. A water well to provide water to the site and a reserve pit for cuttings and used drilling mud may also be necessary. Once this infrastructure is in place, the oil rig can be assembled on site. Diesel engines and electrical generators provide the power for the rig. Once the oil has been reached, for a conventional well, a pump is installed and much of the rig and other machinery can be removed and some altered areas can be restored. However, the pad area and some access roads and pipelines must remain throughout the life of the well. A typical lease area has many different oil wells and pads that are connected by roads and utilities which fragment the surrounding habitat. In Texas, well pad density may be over 55 pads per square mile (Hibbitts et al. 2013). The typical lifespan of an oil or gas well is 20-30 years, though this varies due to geology and the amount and type of oil present (Encana Natural Gas 2011). Once the well and pad have reached the end of their life, they may be removed, and the area can be restored. However, restoration does not eliminate the environmental damage the well caused; research has shown that local biodiversity loss can have cascading effects on ecosystem productivity and function (Butt et al. 2013).

In the United States, the land use change caused by wells is considerable due to the high numbers of wells in many locations (Figure 12).

Figure 12: Oil and gas field in Wyoming; Areas with Suitable Resources for Future Extraction.





Both panels reproduced from Jones *et al*. 2015. a) Oil and gas field in Wyoming; Photo by David Stubs. b) Current oil and gas wells (dots) and areas with suitable resources for future extraction (shading).

In 2017, there were 990,677 onshore and offshore oil wells in the US, down from 1,038,698 in 2014 (U.S. EIA 2018a). The average size of an onshore unconventional well pad is 3.5 acres (Helmholtz Centre Potsdam GFZ German Research Centre for Geosciences n.d.), while an onshore conventional well pad in Texas is about the same, or roughly 3.4 acres (Young et al. 2018a). When only the direct footprint of onshore domestic wells is considered, the US had over 1,429,999 acres of well pad infrastructure in 2011 (Trainor et al. 2016). Trainor et al. 2016 predicted that by the year 2040, the direct footprint of oil and gas land use could increase to 15,891,100 acres. The actual landscape impacts are almost double the footprint, due to the spacing requirements of wells (Trainor et al. 2016). Thus, the full landscape impact of oil and gas estimated for 2040 is roughly 31,782,200 acres. The large landscape effects of oil and gas have implications for environmental effects.

Conventional and unconventional wells require roads and other impermeable infrastructure that result in highly altered landscapes (Jones et al. 2015, Garman 2018). The land use change to altered landscapes has direct effects on habitats and wildlife (Butt et al. 2013, Garman 2018, Young et al. 2018b). Land use change for well construction increases habitat fragmentation, pollution, noise and visual disturbance, and causes local habitat destruction; all of which can decrease biodiversity (Butt et al. 2013, Garman 2018, Young et al. 2018b). Some of these disturbances, such as fragmentation, are not unique to oil and gas extraction, and research on their effects is explained in other literature (Brittingham et al. 2014). For example, it is well known that fragmentation can split breeding populations and reduce genetic variability within each population, potentially making them less adaptable to other disturbances (Keller and Largiadèr 2003, Langlois et al. 2017).

Wildlife populations have been shown to decrease near areas with oil and gas production due to habitat fragmentation, density of wells, human activity, noise and light pollution, avoidance, and other factors (Jones et al. 2015). For example, habitat fragmentation by well pads reduced the use of preferred habitats of lizards in Texas, which is likely to decrease the populations of habitat specialist species (Hibbitts et al. 2013). Density of well pads has been shown to decrease the population size of several species of songbirds in Wyoming (Gilbert and Chalfoun 2011). Greater sage-grouse (*Centrocercus urophasianus*) in Montana and Wyoming were found to avoid sagebrush habitats that would otherwise be high quality when those areas are near natural gas development (Doherty et al. 2008). Threatened woodland caribou (*Rangifer tarandus caribou*) avoid areas within 1000 m of oil and gas wells and 250 m of roads in northern Alberta, Canada, especially during calving season (Dyer et al. 2001). This avoidance reduces available habitat and can decrease caribou population size (Hervieux et al. 2005). Direct mortality from contact with infrastructure is also a problem; an average of 8.4 birds die in each uncovered reserve pit each year (Trail 2006), thousands more birds die due to gas flare stack emissions (Bjorge 1987), and many more may die due to the gas flare stacks and gas compressors on well sites (Jones et al. 2015).

Development of areas for oil and gas production causes secondary land use conversion as more people move into the production area. If the well is in a remote area the increase in population size can cause other cascading negative effects such as illegal hunting and the increase in introduction of exotic species of plants and animals. Both direct and cascading environmental impacts can be especially harmful in delicate ecosystems, such as the Prairie Pothole Region (Gleason and Tangen 2014).

The United States is composed of many different habitats that energy development affect (McDonald et al. 2009), as shown in Figure 13. When comparing Figure 12 and Figure 13, it is clear that oil and gas resources and well locations fall into many habitat categories, although temperate grassland and temperate forest may be the most highly affected.

Figure 13: Major Habitat Types in the United States.

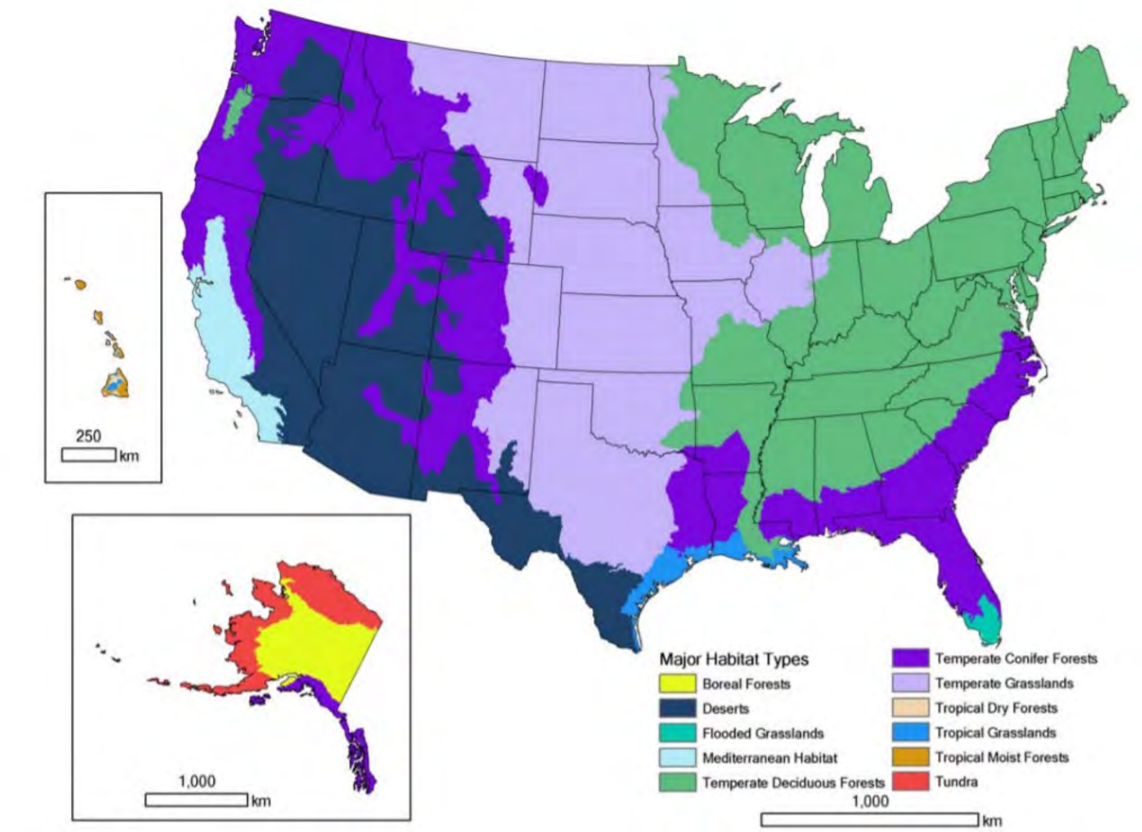


Figure reproduced from McDonald et al. 2009

6.2 Water Quality Impacts Associated with Spills

6.2.1 Unconventional Oil and Gas (UOG)

The most common UOG production method in the U.S. is hydraulic fracturing. A study of UOG wells sites in Colorado, New Mexico, North Dakota, and Pennsylvania estimated 55 spills per 1,000 well-years (where a well-year is a unit denoting the operation of one well for a period of one year; Patterson, Konschnik, et al., 2017). Actual spill rate varied by state, from about 1% (Colorado) to 12% (North Dakota). Median spill size by state varied from 120 gallons (0.5 m^3, Pennsylvania) to 1,302 gallons (4.9 m^3, New Mexico). Total spill volume over ten years (2005 to 2014) was estimated to range from 1,447 m^3 (380 thousand gallons; Pennsylvania) to 33,937 m^3 (9 million gallons; North Dakota). The study found that over 75 percent of UOG production sites spills occur during the first three years of a well's life. It also found that wells with one spill have a higher probability of future spills (Patterson et al. 2017).

Relative to total oilfield spills, the number of spills at UOG production sites is relatively small. EPA (2015) associates only 1% of spills (457 of 36,000 spills across nine states) with hydraulic fracturing. Of the 457 spills assessed by EPA (2015), 300 were reported to reach soil, surface water, or groundwater. The total reported spill volume includes an estimated 540,000 gallons released to soil, 200,000 gallons released to surface water, and 130 gallons reaching groundwater (EPA 2015). Patterson et al. estimate of 6,648 spills associated with all stages of UOG production covering ten years (2005 to 2014). By contrast, the estimate by EPA (2015) focuses only on hydraulic fracturing and covered seven years (2006 to 2012).

6.2.2 Conventional Oil and Gas

The movement of raw petroleum and petroleum products consists of a complex distribution and storage system, which has many chances for accidents, spills, leaks, and losses from volatilization. Consistent national statistics are lacking for many stages in the overall oil distribution and storage system. (ATSDR 1999). Statistics from the American Petroleum Institute (API) based on U.S. Coast Guard data exist for U.S. Navigable waters, but these are limited primarily to coastal areas and large rivers but can include lakes and estuaries.

Data were readily available for the period 1997-2006 from API (API 2009) and are presented for illustration purposes. API reported approximately 10.8 million gallons of oil was spilled into U.S. Navigable Waters from 1997-2006. This includes spills by vessels and facilities (onshore and offshore). The amount spilled per year varied from 466,000 (2005) to 2.7 million (2004). Of the 10.8 million gallons of oil spilled over the period:

- 3.7 million gallons were from onshore facilities;
- Just over 620,000 gallons were from pipelines;
- 226,000 gallons were from offshore facilities;
- 36,000 gallons were from railroads, tank trucks, and passenger cars;
- And most of the remaining spills (5.7 million gallons) were from vessels.

The figures above do not include the Exxon Valdez spill in Alaska in 1989 of 10.8 million gallons (API 1998 as cited in ATSDR 1999) or the Deepwater Horizon spill in 2010 (which post-dated the API study) where EPA reports that 4 million barrels (approximately 168 million gallons) spilled during the 87-day period of the incident (EPA n.d.).

6.3 Toxicity and Other Ecological Impacts of Oil and Associated Products

Total petroleum hydrocarbon (TPH) toxicity to ecological receptors depends on the hydrocarbon composition, exposure pathway, and exposure duration (i.e., acute or chronic). Additionally, TPH in the form of product (e.g., crude oil) can cause physical and chemical toxicity. Acute exposure typically occurs following an accidental release, which causes immediate exposure to high concentrations of petroleum products. Chronic exposures are typically associated with low-level releases over long periods of time, such as from a leaking underground storage tanks and groundwater contamination. Acute exposure following a large oil spill has both physical and chemical impacts and can have immediate ecosystem impacts. In contrast, chronic low-level releases have more subtle impacts typically related to chemical toxicity (Interstate Technology & Regulatory Council [ITRC] 2018).

EPA (1999) describes oil toxicity effects on wildlife according to four categories: physical contact, chemical toxicity, reproductive problems, and destruction of food resources and habitats. These categories of toxicity are described relative to acute and chronic exposures below.

6.3.1 Physical Contact

Terrestrial plants, invertebrates, small animals (mammals, amphibians, reptiles) and birds can become smothered by oil and aquatic organisms can similarly become smothered and lose their ability to uptake oxygen. When fur or feathers of larger mammals or birds contact oil, they get matted down, causing the fur and feathers to lose their insulating properties, placing animals at risk of freezing to death. Additionally, in the case of birds, the complex structure of feathers that allow birds to float or to fly can become damaged, resulting in drowning for aquatic birds (EPA 1999).

6.3.2 Chemical Toxicity

Toxicity to the central nervous system is the major mechanism of toxicity to ecological receptors. Early life-stage aquatic invertebrates and fish can also exhibit phototoxicity (ITRC 2018). These and other toxicological effects are summarized below. Chemical toxicity is typically associated with chronic exposures, however, if petroleum products are present in high enough concentrations, negative health effects, including mortality can occur from acute exposure.

Oil vapors may be inhaled by wildlife, which can cause damage to some species' central nervous system, liver, and lungs. Animals are also at risk from ingesting oil, which can cause red blood cell, intestinal tract, liver, and kidney damage. Skin and eye irritation can also occur from direct contact with oil (EPA 1999). Fish that are exposed to oil may suffer from changes in heart and respiratory rate, enlarged livers, reduced growth, fin erosion, a variety of biochemical and cellular changes, and reproductive and behavioral responses. Chronic exposure to some chemicals found in oil may cause genetic abnormalities or cancer in sensitive species (EPA 1999).

6.3.3 Reproductive Effects

Oil can be transferred from birds' plumage to the eggs they are hatching. Oil can smother eggs by sealing pores in the eggs and preventing gas exchange. Also, the number of breeding animals and the number of nesting habitats can be reduced by a spill.

Scientists have observed developmental effects in bird embryos that were exposed to oil. Long-term reproductive problems have also been shown in some studies in animals that have been exposed to oil (EPA 1999).

6.3.4 Destruction of Food Resources and Habitats

Species that do not directly contact oil can be harmed by a spill. Predators may refuse to eat their prey because oil contamination gives fish and other animals unpleasant tastes and smells, which can lead to starvation. Alternatively, a local population of prey organisms may be destroyed, leaving no food resources for predators. Predators that consume contaminated prey can be exposed to oil through ingestion. This causes bioaccumulation of oil compounds in the food chain. Depending on the environmental conditions, the spilled oil may linger in the environment for long periods of time, adding to the detrimental effects. In freshwater lentic systems, oil that interacts with rocks or sediments can remain in the environment indefinitely, leading to persistent ecological impacts (EPA 1999)

6.4 Additional Water Quality Impacts Associated with Petroleum Production

Production water and fluids used in conventional and unconventional oil and gas production are an additional source of potential contaminants and may have negative impacts on the environment. In the U.S., an estimated 21 billion barrels of produced water is generated each year (Aqwatec n.d.). Production water can be highly saline (up to 15 times saltier than seawater) and can contain elevated levels of chemicals and radioactive elements. This water can kill vegetation and prevent plants from growing in contaminated soil (Miller and Pesaran 1980, Miller et al. 1980, Adams 2011, Pichtel 2016). Hydraulic fracturing fluids contain numerous chemicals to enhance gas and oil extraction. EPA identified 1,173 chemicals associated with hydraulic fracturing activities and chronic oral toxicity values are available for 147 of the chemicals identified (Yost et al. 2016). The potential for toxicity to wildlife and ecosystems depends on the quality of the production water, which varies by production site.

6.5 Additional Water Quality and Supply Impacts Associated with Exploration, Production, and Refining

Water is necessary for both conventional and unconventional oil and gas extraction as well as refining with unconventional oil and gas exploration and production having the higher water demand requirements. This makes oil and gas development a competitor for limited water resources with nearby populations and agriculture, in a time when water rights are often hotly contested (Strzepek and Boehlert 2010). High source water consumption can alter stream flows and affect aquatic ecosystem function, including declines in specific fish species around production sites (Dauwalter 2013, Jones et al. 2015). Additionally, produced water, especially from unconventional oil and gas development, has high total dissolved solids and may be contaminated with other chemicals, making it a pollutant that is expensive and difficult to treat (Gregory et al. 2011, Gleason and Tangen 2014).

There are 135 petroleum refineries in the United States (U.S. Energy Information Administration [USEIA] 2018b, 2019). Over time, the number of petroleum refineries has decreased, but the capacity per refinery has increased (ATSDR, 1999; USEIA 2018b). Gross crude oil inputs to refineries averaged 16.6 million barrels per day in 2017 (USEIA 2018c). An estimated 2.3% of total refinery output is released to the environment through spills or leaks (ATSDR 1999).

Petroleum refinery wastewaters are made up of many different chemicals which include oil and greases, phenols (creosols and xylenols), sulfides, ammonia, suspended solids, cyanides, nitrogen compounds and heavy metals. Refinery effluents tend to have fewer of the lighter hydrocarbons than crude oil but more polycyclic aromatic hydrocarbons, which are generally more toxic and more persistent in the environment (Anderson et al. 1974, Wake 2005). Aquatic ecosystems around refinery discharges are often found to have low biodiversity and a low abundance of fauna. Often the impacted area is limited to a specific distance from the discharge point. This distance varies depending on the site and the effluent. Studies have estimated the impacted range to be 200 m to 1.6 km from the effluent site (Petpiroon & Dicks, 1982; Wharfe 1975 as cited in Wake, 2005). Refinery effluent has also been attributed as the cause of lack in recruitment in some areas, that it may either kill early life stages of aquatic organisms (e.g., settling larvae) or deter them from settling near discharges (Wake 2005).

7. LIMITATIONS

The conclusions, opinions and recommendations presented herein represent Ramboll's professional judgment based upon reasonably available information and are products of and limited by Ramboll's assigned and agreed upon scope of work. In preparing this report, Ramboll relied upon information provided by its client and/or third parties, and also relied upon certain additional publicly available information. Ramboll, however, did not conduct an exhaustive search or review/analysis of all potentially relevant information. The conclusions, opinions and recommendations presented herein, and all other information contained in this report, necessarily are valid only to the extent that the information reviewed by Ramboll was accurate and complete. Ramboll reserves the right to revise this report if/when additional relevant information is brought to its attention. In addition, Ramboll did not consider matters outside of its limited scope of work. Accordingly, the conclusions, opinions, recommendations and other information contained herein may not adequately address the needs of all potential users of this report, and any reliance upon this report by anyone other than Growth Energy, or use of a nature, or for purposes not within Ramboll's scope of work is at the sole risk of the person/entity so relying upon or otherwise using this report. Ramboll makes no representations or warranties (express or implied) regarding this report beyond those made expressly to its client, and Ramboll's liability in relation to this report and its related scope of work is limited under its client contract.

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EXHIBIT 2

RAMBOLL. NOVEMBER 29, 2019. MEMORANDUM: SUPPLEMENTAL ANALYSIS REGARDING ALLEGATIONS OF POTENTIAL IMPACTS OF THE RFS ON SPECIES LISTED UNDER THE ENDANGERED SPECIES ACT. PREPARED FOR GROWTH ENERGY. RAMBOLL, SEATTLE WA.

Growth Energy

ESA Comments - Attachment B

Docket # EPA-HQ-OAR-2019-0136

Supplemental Notice of Proposed Rulemaking; Renewable Fuel Standards Program: Standards for 2020 and Biomass-Based Diesel Volume for 2021, and Response to the Remand of the 2016 Standards

November 29, 2019

ENVIRONMENT
& HEALTH

MEMORANDUM

SUPPLEMENTAL ANALYSIS REGARDING ALLEGATIONS OF POTENTIAL IMPACTS OF THE RFS ON SPECIES LISTED UNDER THE ENDANGERED SPECIES ACT

Prepared for	Growth Energy
Date	11/29/2019

OBJECTIVES AND SCOPE

This memorandum supplements the analysis in our August 2019 report, "*The RFS and Ethanol Production: Lack of Proven Impacts to Land and Water*" ("Ramboll Report"), in which we analyzed potential environmental impacts of the RFS program and concluded that there are no proven adverse impacts to land and water associated with increased corn ethanol production under the RFS. The impetus for this supplemental memorandum is a recent D.C. Circuit opinion on a petition for review of EPA's final rule setting the renewable fuel standards for 2018 (the "2018 RVO Rule"). *Am. Fuel & Petrochemical Mfrs. v. EPA*, No. 17-1258 (D.C. Cir. Sept. 6, 2019). The Court remanded the rule back to the agency to further consider petitioners' claims that EPA failed to comply with the Endangered Species Act (ESA). Specifically, the Court directed that under ESA Section 7, EPA must make an appropriate determination as to whether the 2018 RVO Rule "may affect" a listed species or critical habitat.

We are aware that the ESA Section 7 consultation issue is relevant not only to the remand in the above case, but also to future EPA rulemakings with respect to the Renewable Fuel Standard Program (RFS), including EPA's proposed rule setting the renewable fuel standards for 2020 (the "2020 RVO Rule"). Following on our 2019 Report, we are providing this supplemental analysis to explore further whether there is any evidentiary basis in the record for EPA to conclude that the RFS program "may affect" a listed species or critical habitat. This memorandum focuses on the technical aspects of the record relied upon by the Court that were supplied by petitioners' exhibits, including:

- Declaration of Dr. Tyler Lark (July 27, 2018; referred to herein as the Lark Declaration)
- U.S. Environmental Protection Agency, *Biofuels and the Environment: Second Triennial Report to Congress.* Washington, D.C. (June 29, 2018)
- Declaration of C. Elaine Giessel (July 27, 2018)
- Declaration of Aaron Viles (July 20, 2018)
- Declaration of William A. Fontenot (July 24, 2018)
- Declaration of Katherine M. Slama (July 26, 2018)
- Declaration of Andrew E. Whitehurst (July 26, 2018).

Problem Understanding

The allegations of potential impacts to listed terrestrial species that are presented in the Lark Declaration (and referenced in the Court opinion) center on an assumed relationship between the RFS and habitat loss or degradation due to presumed land conversion to grow biofuel feedstock. The Lark Declaration also references potential impacts to aquatic species due to an assumed relationship between biofuel feedstock grown for ethanol production and water quality degradation due to use of agrichemicals (e.g., fertilizers and pesticides) and the potential for increased erosion.

The relationship between the RFS and impacts to land and water, if any, would be effected via a complex causal chain consisting of the following major relationships:

A. RFS and increased demand for ethanol

B. Increased demand for ethanol and upward pressure of this demand on the price of ethanol feedstock (mainly corn)

C. Price of corn and an individual farmer's decision to plant corn

D. Individual farmer's decision to double crop corn, switch from another crop to corn, or farm new land

E. Location, type, and magnitude of adverse impact to terrestrial and aquatic habitat

Each of the above relationships, in turn, encompasses several interrelated variables, each variable is likely to change on an annual basis, and many of the relationships are co-dependent. The Lark Declaration does not consider these relationships in a meaningful way, and instead relies on unsupported assumptions and speculation.

There are several lines of evidence indicating that increased demand, if any, for ethanol resulting from the RFS has not been a discernible driver of land use change. One of the most basic lines of evidence has to do with the historical trend in the number of acres in the U.S. devoted to growing corn. Historical data generated by the U.S. Department of Agriculture (USDA) shows that acres planted in corn nationwide is currently at or below levels reported in 1926 and in the last 2 decades has generally fluctuated between 80 million acres and 100 million acres (Figure 1).

The amount of land in the U.S. devoted to growing corn has remained at or below historical levels despite the following trends:

- Total corn production (bushels per year) has increased about 7-fold over the period of record
- Corn produced for ethanol has increased by a factor of 12.5 since 1986 and now accounts for about 50% of corn grown.

This increase in corn production and corn production devoted to ethanol, without an apparent increase in acres planted, is attributed to a 7-fold increase in corn yield (bushels per acre).

The 7-fold increase in corn production nationwide over the period of record has not been accompanied by a nationwide increase in the acres of corn planted. This lack of association in itself calls into question whether there is a causal link between increased demand for corn grown for ethanol and demand for increased acreage of corn, which in turn calls into question the causal relationship between increased demand for corn for ethanol and land conversion. The remainder of the report delves more deeply into each step in the potential causal chain between the RFS and impacts to species. In particular, causal steps B, C, and D are discussed in Section 2 below, and causal step E is discussed in Section 3 (for terrestrial listed species mentioned in the Lark Declaration) and Section 4 (for aquatic listed species mentioned in the Lark Declaration). Analysis of the effect of the RFS on increased demand for ethanol (causal step A above) is outside the scope of this memorandum.

Summary of Findings

Our technical review of the assertions made in the Lark Declaration lead to the following overall conclusions:

- Assertions that increased corn ethanol production under the RFS has resulted in land use change and conversion of non-agricultural land to production of biofuel feedstock are unsubstantiated for several reasons, including the following:
  - Acres planted in corn across the United States has remained close to or below the total acres planted in the early 1930s despite increases in demand for corn as human food, animal feed, and biofuels over this nearly 90-year period. This fact by itself calls into question the relationship between the RFS and land use change.
  - The causal relationship between the RFS and the price of corn is not supported by the evidence, and therefore, the Lark Declaration's presumption that increased corn prices drive land use change are unsubstantiated.
  - The Lark Declaration does not adequately consider the many disincentives to the farmer of converting non-agricultural land to growing any given crop, and thus assertions in the Lark Declaration that the RFS and price of corn has resulted in land conversion are also unsubstantiated.
- Assertions that RFS-driven land use change has resulted in impacts to particular ESA listed species are without foundation for multiple reasons, including:
  - The Lark Declaration asserts that land use change spurred by the RFS has resulted in impacts to listed terrestrial species of birds, mammals, and insects.
  - The evidence presented in the Lark Declaration to support the alleged impacts are poorly researched and the examples used to support many assertions instead actually *refute* the assertions.
- Assertions that RFS-driven biofuels agriculture adversely impacts water quality are also unsubstantiated for multiple reasons, including:
  - The Lark Declaration asserts that biofuels (corn and soy) agriculture has worsened the Gulf of Mexico dead zone, imperiling Gulf sturgeon, loggerhead turtles, and sperm whales, yet provides no supporting evidence; no studies are cited that specifically quantify the effect of corn or soy crops as threatening these species or their habitats.
  - The Lark Declaration also asserts that biofuel (corn and soy) agriculture is associated with impaired waters pursuant to Section 303(d) of the Clean Water Act but fails to acknowledge cases in which such designations were made well before the RFS came into effect. Our independent assessment of specific examples presented in the Lark Declaration indicates that the allegations of impacts from corn or soy on impaired water bodies is unsubstantiated.

In sum, there are least two important causal chains that must be quantified and linked together to demonstrate a relationship between increased corn ethanol production under the RFS and impacts to ESA-listed species: 1) a causal chain linking the RFS to land use change and water quality impacts; and 2) a causal chain linking impacts to land and water with specific impacts on the survival or reproduction of ESA-listed species. Each of these causal chains is made up of many embedded biophysical and economic relationships that, in turn, are influenced by a myriad of interrelated variables. The Lark Declaration fails to consider these causal relationships in a meaningful way, relying instead on unfounded assumptions and speculation to support its thesis.

1. Examination of the Causal Link Between the RFS and Impacts to Listed Species

1.1 Overview of Causal Analysis

Causal analysis is a method that is used to determine root causes for observed outcomes. It is used in many fields such as medicine, business management, economics, ecology, and has been used to explore the causes of land use change (Efroymson et al. 2016). The point of causal analysis is to look behind outcomes or symptoms to determine the actual cause, instead of assuming the most obvious cause is the root of the issue. For example, if a patient presents to a doctor with knee pain because they hurt themselves gardening, the doctor may simply give pain medication. If the doctor looks deeper using a more holistic causal analysis approach, the doctor may find that the patient is out of shape or that they have arthritis. If the symptom is treated without fully understanding the root cause of the problem, the problem will not be solved in the long term.

Causal analysis begins with creating a causal diagram that includes all causal components of an outcome. In the next section, we use a causal diagram to examine how farmers make decisions about crop species planted and land expansion.

1.2 The RFS/Land Conversion Causal Chain

The relationship between the RFS and the potential for land conversion is addressed in the Ramboll report, primarily in Section 3.2. The decision to alter land from non-agricultural uses to agriculture in general is made at the farm level and is influenced by a myriad of factors including predicted weather conditions, crop output and input prices, innovations in cropping equipment, crop insurance, disaster assistance, and marketing loans. The Ramboll report cites three publications in particular which address the complexity of the causal relationship between increased production of corn ethanol and land use change (Section 3.2 page 16-17). As one example, Efroymson et al. (2016) discusses the use of formal causal analysis to clarify the relationship between biofuels policy and land use change and concludes that studies relying on single lines of evidence alone are insufficient for establishing probable cause. Many such studies are cited by EPA (2018) and indeed, many such studies rely on simple temporal changes occurring around the time of the enactment of the Energy Independence and Security Act (ESIA) or simple spatial associations (e.g., land use change proximity to ethanol plants) in an attempt to link land use change and increasing corn production.

The assertion that the EISA increased the expected market price of corn and directly caused land use change is not supported by a causal analysis. Figure 2 illustrates a simplified causal diagram including the many components that influence planting decisions by farmers. It is clear from this diagram that the expected market price of any given crop is not the only relevant factor in planting decisions. Farmers often have limited freedom to change crop types or expand their farmed areas. For example, planting, cultivating, and harvesting machinery is not interchangeable between all crop types. Farmers may only be able to choose between two or three crops that their current machinery is capable of handling. Additionally, farmers are locked into crop rotation schedules to maintain soil conditions and crop health. Furthermore, all fields cannot be harvested at the same time due to limited machinery, so crops with different harvest times must be planted to ensure high-quality output. If farmers are participating in government subsidy or incentive programs, they may be limited in the types of crops they can plant. Areas in the conservation reserve program cannot be planted until the term of the contract expires, and water use restrictions, or limitations of irrigation machinery, can limit expansion of field size.

For farms, even if the species of crop or the expansion of field size were not restricted as described above, market forces themselves affect planting decisions. Deciding what to plant is a gamble. Farmers must consider many factors, including their own costs, resources, and market price estimates.

Successful farmers must bring in enough profit for both salaries and capital costs; meaning that costs must be well below profits. Besides the obvious costs of fertilizer, water, labor, and machinery, the price of transportation to get products to market must be considered, as well as costs of insurance given the location, climate, and predicted weather. The decision to expand farmed areas could be a poor one if the marginal costs exceed marginal profits. This is especially a concern when expanding farmland into areas around currently farmed fields, which may be less suitable for farming because of steeper inclines or poorer quality soil. Additional costs will also be incurred when expanding into natural areas where drainage of wetlands or removal of trees and other obstructions will be required, which is a disincentive to increase farmed acreage. These dynamics are explored in more detail in the following section.

2. Lack of Evidence of a Causal Link Between RFS and Land Use Change

A recent report by Lark et al. (2019) is a comprehensive attempt to establish quantitative causal linkages between the enactment of the RFS and a variety of environmental outcomes using a series of interlinked models. The fundamental premise of their work is the assumption that the price of corn is heavily influenced by increased demand for ethanol due to the RFS, yet the authors ignore other important factors that have a considerable effect on demand and supply conditions (Lark et al. 2019 is discussed in detail in the Ramboll Report at Section 3.4.) Staab et al. (2017), for example, find that there are many other contributing factors affecting demand for corn, including market speculation, stockpiling policies, trade restrictions, macroeconomic shocks to money supplies, currency exchange rates, and economic growth. As one example, rapid economic growth in developing countries led to growing food demand and a dietary transition from cereals toward more animal protein and the corn products used as cattle feed. As a result, global consumption of agricultural commodities has been growing rapidly. In fact, it appears that most of the increase in corn prices has actually been driven by higher oil prices (Figure 3). The U.S. Energy Information Administration estimated that of the total cost per acre of producing corn in 2013 (approximately $350/ac.), nearly two thirds was spent on fuel, lubricants, electricity and fertilizer[1]; and fertilizer is known to be closely linked to oil prices[2].

Moreover, Lark et al. (2019) and other authors who have attributed land use change to the RFS do not adequately consider the wide range of factors that influence farmers' individual planting decisions. These factors determine the relative prices expected to be faced by farmers. That is, the futures prices of different crops relative to each other help a farmer determine the crop planting mix (what and how much). While relative prices may help a farmer determine the potential crop mix farmed on the land, other supply factors influence the potential costs of production. These include weather, soil quality and temperature conditions; pests and disease (McConnell 2018); moisture (Queck-Matzie 2019);energy and fuel costs; interest rates; storage costs; seed and fertilizer costs; and "preventive" planting (Schnitkey et al. 2019) programs such as COMBO (Crop Insurance), the Cropland Reserve Program (CRP), and others.

Temporal uncertainty is something that farmers face in all their planting decisions. Farmers need to decide today what and how much to plant in the next growing season. Farmers are responsive to crop prices which act as a clearing house to reflect future demand and supply conditions and help alleviate the uncertainty associated with future conditions. This means that a variety of factors described above determine planting decisions, and these factors, coupled with the uncertainty of future prices and costs, weakens the link between the supposed increase in price of corn due to the RFS and planting decisions.

In making crop mix decisions, farmers consider relative futures prices and expected profitability of plantings (futures price vs. cost to produce; (Kleiber 2009, Staab et al. 2017, Hecht 2019, Springborn

[1] https://www.eia.gov/todayinenergy/detail.php?id=18431#
[2] https://agmanager.info/sites/default/files/pdf/2019.4.pdf

2019). Weather is also an important consideration in a farmer's decision on whether to implement "prevented" planting (Reiley 2019, Springborn 2019). Futures prices, profitability, and weather forecasts are factors assessed by a farmer to determine where to plant and how much of each crop to plant (Kleiber 2009, Reiley 2019, Springborn 2019). Farmers examine, among many other factors, the relative price ratios of crops to determine an optimal planting mix, and if a farmer decides to increase production of a certain crop, this can be accomplished by either producing more of the crop on existing land (intensification) or putting new land into production (extensification, which may result in land use change). All else being equal, extensification is the least preferred option as it is the option most likely to involve additional expenditures such as land clearing and other preparation. This option will also be dependent on the expected yield of new fields, which relative to existing fields, is most likely to be sub- or infra-marginal and will require more intensive inputs to achieve desired yields (Schiller 2017). Given these considerations, farmers will typically consider switching crops and increasing yield on existing acreage (Ling and Bextine 2017) before farming new land. Intensification efforts can include precision farming as well as traditional techniques regarding plant spacing, pest management, etc. (Queck-Matzie 2019). (The positive environmental effects of precision farming and other technological advances in agriculture are described at length in the Ramboll report at Section 4).

In summary, studies have shown only a modest effect on corn prices potentially associated with the RFS (Kleiber 2009, Babcock and Fabiosa 2011, Carter et al. 2018, Renewable Fuels Association 2019). In addition, factors affecting farmers' planting decisions include much more than the expected market price of the crop (Kleiber 2009, Staab et al. 2017). Other important factors include the expected yield of the crop (Reiley 2019); and a myriad of production costs including the cost of seed, fertilizers and pesticides, machinery, crop insurance, labor, fuel, and land rental costs (Corn Agronomy 2006, Staab et al. 2017, Hecht 2019). The decision to expand crops onto new land entails additional hurdles and costs beyond costs associated with changing crops or intensifying production on existing acreage. For these reasons and those discussed more extensively in the Ramboll report (at Section 3.4), it is unreasonable to draw a direct causal connection between the RFS and land use change.

3. Lack of Evidence of a Causal Link Between the RFS and Impacts to Terrestrial Species

In the absence of a causal link between the RFS and land use change—and in particular land conversion from grassland, wetland, or forest to corn and soy—there can be no causal link between the RFS and impacts to terrestrial species due to loss or degradation of habitat. In an attempt to establish a causal link between the RFS and impacts to terrestrial listed species, the Lark Declaration presented several examples of quantitative analysis of land conversion from presumably natural land cover to presumably corn and soy. These examples relied on approaches to land conversion analysis presented by Lark et al. (2015). Lark et al. (2015) analyzed land use change nationwide during the period 2008-2012 using the U.S. Department of Agriculture (USDA) Cropland Data Layer (CDL), calibrated with ground-based data from USDA's Farm Service Agency (FSA), and further refined using data from the National Land Cover Database (NLCD). The approach used by Lark et al. (2015) purportedly included methods to "correct" for known errors and uncertainties in the CDL database. However, the approach used by Lark et al. (2015) has been shown to be flawed, resulting in a gross overestimate of land use change.

The Ramboll Report (Section 3.3 pages 19 and 20 and Table 1) discusses work by Dunn et al. (2017) which examined data for 2006-2014 in 20 counties in the prairie potholes region using the CDL, a modified CDL dataset, data from the National Agricultural Imagery Program, and in-person ground-truthing. Dunn et al. (2017) concluded that analyses relying on CDL returned the largest amount of land use change by a wide margin. They further concluded that errors associated with CDL-based analyses are a major limitation of conclusions drawn from such analyses. In fact, Dunn et al. (2017) concluded that “the number of hectares in the potential error associated with CDL-derived results is generally greater than the number of hectares the CDL-based analysis determined had undergone a transition from grassland, forested land, or wetland to agricultural land”. This suggests that errors in classification inherent in the CDL can result in uncertainty bounds that are of a larger magnitude than the estimates themselves (thereby even predicting “negative” land conversion to agriculture within the uncertainty bounds). Specifically, Dunn et al. (2017) pointed out that the findings reported by Lark et al. (2015) contradict USDA data indicating that cropland area has remained almost constant during the period 2008-2012.

The Lark Declaration also cited other authors who purport to establish a quantitative link between the RFS and land use change based on geographic associations (e.g., increased conversion of land to biofuel feedstock in close proximity to ethanol refineries). The Ramboll report specifically identified the following key flaws in studies that attempt to quantify land use change to biofuel feedstocks (Section 3.1 pages 14 and 15):

- Like Lark et al. (2015), many other studies of land use change to agriculture depended on unreliable data sets such as CDL data, lacked ground-truthing, and were regional or state-specific. These problems preclude extrapolation of results nationwide.
- The literature assessing LUC relative to the RFS generally fails to consider the considerable loss of agricultural land due to growth in urban areas and the role this loss may have on the pressure to expand agricultural lands elsewhere.

It is reasonable to presume that the Lark Declaration presented the best examples that could be found to make the case for the habitat of a particular species having been impacted by land conversion to corn or soy spurred specifically by the RFS. In the following sections, we analyze and respond to specific examples presented in the Lark Declaration. In each case we analyzed, we found fatal flaws in the examples presented in the Lark Declaration. These flaws are either associated with a lack of temporal association or a lack of geographical association (or both) and a lack of potential causative mechanism.

3.1 Whooping crane (***Grus americana***)

The Whooping Crane is currently classified as an endangered species. Current places of residence include Florida, Texas, central Canada, and Wisconsin. Migrating flocks reside in either Texas, Florida, central Canada, or Wisconsin (Cornell University 2019) primarily in wetlands or muskeg (swampy woods with lakes). In 1941, the total population had declined to 21 birds. Conservation efforts, including protection of wintering grounds and educating hunters, has helped increase the population. As of 2019, more than 350 whooping cranes reside in North America, including 174 migrating cranes (USFWS n.d.). The population has been increasing over time, with no dip apparent after the RFS in 2008 (Figure 4). In fact, after 2007, the population of whooping cranes appears to have increased even faster than it did between 1990 and 2007 (Figure 4)

A total of three known flocks currently exist throughout North America: two migrating flocks and one non-migrating flock. One migrating flock spends summers in the Wood Buffalo National Park in Canada and winters in Texas at the Aransas National Wildlife Refuge. The other migrating flock nests in Wisconsin for the summer and flies south to Florida in the winter. These flocks have been sighted taking short rests in Kansas at either Cheyenne Bottoms or Quivira National Wildlife Refuge (QNWR) whilst migrating. A non-migratory flock remains in Florida year-round (USFWS n.d.).

The migrating flocks reside in national refuges or national parks that have protection plans in place. For example, the QNWR prohibits hunting when whooping cranes are present to avoid accidental shootings. The U.S. Fish & Wildlife Service reports that refuges only integrate farming for specific wildlife conservation efforts.

Whooping Cranes spend time in marshes, shallow bays and tidal flats, with the occasional venture to nearby farmland. Their diet varies by area but may include fish, mice, insects, berries, seeds, crabs and snakes. The Whooping Crane's wide variety of food preferences opens opportunity to scavenge in several locations, including corn fields (USFWS n.d.).

The Lark Declaration argues that conversion to cropland "adjacent to its critical habitat and wintering grounds" may negatively impact the livelihood of the Whooping Cranes. Lark does not discuss the landscape of the adjacent land at issue nor the distance of these adjacent habitats from the whooping crane's current nesting grounds. The images Lark refers to in support of this claim are in Appendix 7 to the Lark Declaration. The image in Appendix 7 includes boundary locations of the critical habitat (Cheyenne Bottoms and QNWR) briefly visited by the Whooping Cranes (Bloomberg n.d.), as well as the nearest ethanol refinery. Ramboll further investigated these images and found that they did not support Lark's claims.

Multiple areas on the Lark Declaration's maps that show corn or other crops growing in or near Cheyenne Bottoms and QNWR were errors in the USDA Cropland Data Layer (CDL). One particularly egregious error shows corn growing in the southeast corner of Cheyenne Bottoms Pool 2 (Figure 5). It is clear from aerial imagery in Google Earth going back to 1992 that no corn is growing in Pool 2 (Figure 5). Ramboll further confirmed the lack of corn by contacting staff at the reserve on November 18 and 19 of 2019. Reserve staff confirmed that Pool 2 was usually under water, and although they had planted a cover crop for the benefit of wildlife in some dry years, the cover crop had never been corn[3]. Ramboll investigated multiple years of Google Earth aerial imagery for areas near to Cheyenne Bottoms and QNWR and also within QNWR that Lark showed as converted to corn; these images failed to show any new crop cultivation after 2007.

In summary, the data presented in Lark's declaration does not support his assertion that the RFS spurred land use change to biofuels in or near Cheyenne Bottoms and the QNWR. To the contrary, it appears that there is little or no land use change to agriculture near either reserve that supports whooping crane migrations in Kansas, and that such land use change, if any, has not been attributed to the RFS. Further, the population of whooping cranes in the United States has risen and continues to rise since the RFS, suggesting that even if the RFS has resulted in some land conversion in areas potentially used by the whooping crane, this conversion has not resulted in any discernible adverse impact on whooping crane populations. Due to the lack of evidence of land use change, any assertion that the recovery of whooping crane populations would have been more rapid had it not been for the RFS would be purely speculative[4].

3.2 Piping Plover (***Charadrius melodus***)

There are three distinct populations of piping plover in the U.S.: Great Lakes, Northern Great Plains, and Atlantic Coast. Piping plover populations on the Great Lakes are listed as endangered, whereas populations in the Northern Great Plains and Atlantic coast are listed as threatened. Piping Plover population declines have been attributed to human disturbance, habitat loss and predation. Piping

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[3] Phone communication between Ramboll and Cheyenne Bottoms Ranger Station (620-793-3066) on November 17th and 19th, 2019

[4] This point applies as well to other species discussed below, where the data show recovery of the species during the time frame in which the RFS program has been implemented.

plover management strategies are targeted at limiting access to beachgoers and off-road vehicles, pet restrictions, and public education[5].

The Lark Declaration implicates land conversion for crop production (and presumably by extension, land conversion to corn or soy as a result of the RFS) as a potential impact to piping plover populations, citing Cohen (2009)[6] as documenting "disruption of plover habitat" in the Great Lakes endangered population. In fact, Cohen et al. (2009) studied two Atlantic Coast piping plover breeding areas in West Hampton Dunes, a barrier island in New York State. The only mention of land conversion made by the authors was in reference to urban development. In addition, the authors cite predation management (domestic cats and fox) as key to the recovery of the populations at the sites they studied. Thus, this study cited in the Lark Declaration is not relevant to the premise that land conversion spurred by the RFS results in impacts to piping plover, and in fact specifically points to urban development and predation as the primary stressors to these populations.

The U.S. FWS Midwest Region fact sheet describes the following threats to Great Lakes piping plover populations: Coastal beach habitat loss due to commercial, residential, and recreational developments; and effect of water control structures on nesting habitat; vehicle and pedestrian use of beaches; harassment or mortality of birds by dogs and cats; and predation by fox, gulls, and crows[7]. Habitat protection measures include controlling access to nesting areas, nest monitoring and protection, limiting residential and industrial development, and properly managing water flow[8]. Thus, like the Atlantic Coast populations, land use change due to agriculture is not a recognized threat to the Great Lakes populations.

In the Northern Great Plains, piping plover breed on river sandbars, along reservoir shorelines, and in manmade habitat such as commercial sand mines. Similar to the Atlantic Coast and Great Lakes populations, declines of this population are attributed mainly to harassment of birds and nests by people, domesticated animals, and vehicles; shoreline habitat loss due to development projects; human-induced increased predation; and water-level regulation policies that disrupt nesting behavior or destroy nesting habitat (NRC 2005). Appendix 8 to the Lark Declaration provides an example of conversion of some riparian forest habitat adjacent to a farm field along the Missouri River sometime between spring 2012 and late winter 2015. This example, however, does not portray any loss of critical habitat for this species (i.e., critical habitat for the piping plover in the Missouri River is sand bar or sandy shoreline habitat and not forest), and therefore does not support the premise in the Lark Declaration that land conversion spurred by the RFS results in impacts to piping plover.

Thus, based on a review of the specific citations relied upon by the Lark Declaration as well as publications by the U.S. Fish and Wildlife Service (USFWS) regarding endangered and threatened populations of piping plover, we find no evidence that agriculture in general, or land conversion to corn and soy due to the RFS in particular, results in impacts to piping plover. Such claims in the Lark Declaration are unsubstantiated.

It is worth noting that, in addition to discussing land conversion, the Lark Declaration cites a study by Fannin (1993)[9] when suggesting that pesticides or other contaminants from agricultural practices (and by extension, presumably agriculture for biofuels feedstock spurred by the RFS) could jeopardize piping plover egg survival. Fannin and Eamoil (1993) collected 16 piping plover addled (unhatched) eggs in 1989 and 3 piping plover addled eggs in 1990 and analyzed the contents for a wide range of metals and

[5] https://naturalhistory2.si.edu/smsfp/irlspec/Charad_melodu.htm
[6] The Lark Declaration incorrectly cites Cohen et al. (2009)
[7] https://www.fws.gov/midwest/endangered/pipingplover/pipingpl.html
[8] https://www.fws.gov/midwest/endangered/pipingplover/pipingpl.html
[9] We believe that the Lark Declaration incorrectly cites Fannin and Eamoil (1993)

several organochlorine pesticides, including DDT and its breakdown products. DDT and to a lesser extent, other organochlorine pesticides are known to cause eggshell thinning and reproductive failure, principally in raptors and fish-eating birds. DDT was banned from use in the United States in 1972, and chlordane was banned in 1988. It is also widely known that many species made dramatic recoveries in the years following the ban of DDT, most notably the bald eagle. Use of these and other organochlorine pesticides in agriculture were terminated decades prior to the enactment of the EISA and implementation of the RFS. Thus, any suggestion in the Lark Declaration that the use of pesticides on biofuel crops may be resulting in eggshell thinning in piping plover lacks foundation.

3.3 Yellow-Billed Cuckoo (***Coccyzus americanus***)

The Western U.S. Distinct Population Segment of *C. americanus* (western yellow-billed cuckoo) was proposed as threatened on October 3, 2013 (FR 79:192, October 3, 2014; USFWS 2014). Within the last 50 years the species' distribution west of the Rocky Mountains declined substantially mainly due to loss of streamside habitat. USFWS (2014) reports that current impacts from agricultural activities on yellow-billed cuckoo habitat are mainly associated with livestock overgrazing in riparian areas.

Yellow billed cuckoo breed in dense willow and cottonwood stands in river floodplains. The Lark Declaration states that their threatened status is due largely to the "destruction of these habitats from anthropogenic activities, including agriculture," and presumably by extension, land conversion to biofuel feedstock (corn and soy). However, the Lark Declaration fails to acknowledge that with the exception of Glenn County in California, there is no overlap between significant corn or soy growing areas and critical habitat for the species. This is primarily due to the fact that most corn and soy production in the U.S. occurs east of the Rocky Mountains.

Figures 6, 7, and 8 show areas reported to be in corn in the USDA CDL database for 2018 within the boundaries of designated critical habitat for the Western yellow-billed cuckoo in Glenn County, along with available Google Earth aerial images of these areas. The maps in figure 6 show that with only a couple of exceptions there is no overlap between Western yellow-billed cuckoo habitat and counties with corn and soy cultivation. The Google Earth images in figure 7 and 8 clearly show that these areas were in agricultural production as early as 1998, a decade before the RFS could possibly have influenced land conversion, and at approximately 55.5 acres, they account for only 0.036% of the total available critical habitat for the species in California (155,635 acres). Thus, not only is there no overlap between critical habitat for this species and significant corn growing areas, but in the two instances in California where the CDL reports corn to be grown in critical habitat, areas were in agricultural production long before the RFS.

As with the piping plover, the Lark Declaration also suggests that western yellow-billed cuckoo is adversely impacted by eggshell thinning due to pesticides. For the reasons described above, any eggshell thinning observed in this species cannot possibly be associated with the RFS and any such implied association is unsubstantiated.

3.4 Poweshiek Skipperling (***Oarisma poweshiek***)

The Poweshiek skipperling, was once abundant in remnant native prairie habitat in Indiana, Illinois, Iowa, Michigan, Minnesota, North Dakota, South Dakota, Wisconsin, and Manitoba, Canada; but is now thought to be present only in Wisconsin, Michigan, and Manitoba. The USFWS lists several stressors that may be acting to reduce populations of the butterfly, with loss and degradation of habitat being one of the initial stressors for its decline. The USFWS states that other stressors are unknown but might include disease or pesticides.

The Lark Declaration (paragraph 15, page 12) states “Habitat fragmentation poses a key threat to the Poweshiek skipperling, and there are several instances where land has recently been converted to cultivate either corn or soybeans within close proximity to its critical habitat in Minnesota, North Dakota, and South Dakota”. Paragraph 15 refers to Appendix 6, which we presume to be their best example to illustrate land conversion due to RFS. Appendix 6 presents a map showing Poweshiek skipperling critical habitat in Minnesota, the location of an ethanol refinery, and polygons depicting presumed land conversion from native tall grass prairie to corn or soy. Appendix 6 is based on a comparison of data from 2008 to 2016 and methods documented in Lark et al. (2015; see above description of shortcoming of these methods). The second page of Appendix 6 shows two images from Google Earth, one from May 21, 2008 and another from June 23, 2011--presumably showing conversion of two farm fields adjacent to Poweshiek skipperling critical habitat from grassland to cropland. The refinery depicted in Appendix 6 was confirmed by Ramboll to be the Valero refinery in Aurora, North Dakota; approximately 28 miles from the illustrative farm fields.

Several facts indicate that the assertions in the Lark Declaration regarding the Poweshiek skipperling are flawed, and, in fact, land conversion from tall grass prairie to corn or soy due to the RFS could not have had an impact on this listed species:

- The last confirmed sightings of *O. poweshiek* in Minnesota were in 2007, despite extensive annual surveys beginning in 2013[10]. The RFS went into effect in 2008 so could not possibly have had an adverse effect on this species in Minnesota. Similar trends were seen in other states (Environment Canada 2011):
  - In Iowa, the species was in decline by 2003 and was last observed in 2008
  - In North Dakota, the species was thought to be extirpated by 2008; with only 8 individuals seen in a survey in 2001.
  - In South Dakota, The species began to disappear from five South Dakota sites in 2002 and many of these sites were observed to be idle with no range or grass management. At these sites, the decline was attributed loss of floral diversity, increase in grasses and forbs, and an increase in exotic species. The species was last observed at Hartford Beach State Park and the Waubay National Wildlife Refuge in 2002, Pickerel Lake State Recreation Area in 2004, Wike Waterfowl Production Area in 2006, and Scarlet Fawn Prairie in 2008. Several sites where they had been recorded in the past were surveyed in 2010 and no adults were observed.
- The Valero Aurora refinery in North Dakota began operation in 2003 and reached a capacity of 120 million gallons per year (MGY) in 2005[11], three years before the enactment of EISA. Therefore, any increased demand for corn in the vicinity of the refinery would have been met prior to any possible effect of the RFS, and therefore there cannot be a causal relationship between the RFS and land conversion impacting *O. Poweshiek* in Minnesota.
- Dana (1997) conducted a survey for the Dakota skipper (*Hesperia dacotae*) butterfly in several critical habitat areas of Minnesota, including Hole-in-the-Mountain Prairie Lincoln County, Minnesota South of the town of Lake Benton (the same area depicted as critical habitat in the Lark Declaration Appendix 6). The Dakota skipper has very similar habitat requirements as Poweshiek skipperling. Dana (1997) states that the principal threat to this species in this area is probably the use of herbicides for weed and brush control in privately owned pastures as well as overgrazing and mowing

[10] https://www.dnr.state.mn.us/rsg/profile.html?action=elementDetail&selectedElement=IILEP57010
[11] https://www.valero.com/en-us/AboutValero/ethanol-segment/aurora

by County Park staff, and possibly excavation for construction materials. Dana (1997) specifically states that conversion of additional prairie to cropland (in general) is at most, a minor threat[12].

- Appendix 6 of the Lark Declaration shows satellite images from Google Earth for the years 2008 and 2011 presumably to contrast land use in the year the EISA was enacted and several years after the RFS went into effect. However, there is no information provided to substantiate the claim that the highlighted areas were indeed grassland in 2008. In fact, when other satellite images readily available on Google Earth are examined, it is clear that the subject areas were in agriculture as early as 1992. Further, upon viewing the Google Earth images available in subsequent years for the subject areas, there is no evident expansion of cropland since 1992 into what is now designated as critical habitat for the Poweshiek skipperling (Figure 9).

For the reasons described above, the assertion in the Lark Declaration that land conversion spurred by the RFS has adversely impacted critical habitat of the Poweshiek skipperling are unsubstantiated.

3.5 Other Insects

The Lark Declaration also mentions the threatened Dakota skipper (*Hesperia dacotae*), the endangered rusty patched bumble bee (*Bombus affinis)*, the endangered Hine's emerald dragonfly (*Somatochlora hineana*), and the endangered Salt Creek tiger beetle (*Cicindela nevadica lincolniana*) as other insect species that could “potentially be affected by biofuel feedstock production”. In no case, does the Lark Declaration provide any evidence to support that assertion. The Dakota skipper and rusty patched bumble bee are both prairie/grassland species. Although there has been habitat loss and fragmentation to varying degrees across the ranges of these species, there is no evidence presented that habitat loss occurring after 2008 was directly linked to the presumed RFS-induced land conversion.

As to Hine’s emerald dragonfly, USFWS (2013) states the following:

- The greatest current threat to this species is from invasive plants
- There are effective protections against habitat loss (wetland filling and draining)
- Past habitat loss was due to commercial and industrial development.

USFWS (2013) does not mention any impact related to agriculture. Therefore, the Lark Declaration’s assertion of impacts to Hine’s emerald dragonfly due to the RFS is unsubstantiated.

With regards to the Salt Creek tiger beetle, the Lark Declaration also provides no evidence or discussion of the causal relationship between the RFS and impacts to this species. The Salt Creek tiger beetle is currently found at only three sites in Lancaster County, Nebraska occupying 15 acres in saline wetland habitat[13]. The Nebraska Game and Parks Commission states that the biggest threat to the habitat of this species is stream channel modification[13]. The USFWS (2013b; page 33284) cites two publications from 2003 and 2005 in its statement that “in the past 150 years, approximately 90 percent of these wetlands have been degraded or lost due to urbanization, agriculture, and drainage” but does not mention agriculture as a threat to habitat of this species after the implementation of the RFS in 2008. In fact, the USFWS (2013; page 33285) shows a graph presenting results of surveys of adult Salt Creek tiger beetles 1991 to 2012 which indicates a consistent increase in population over the period 2008-2012 with an approximate doubling in numbers over that period (Figure 10). Based on the information presented above, we find that the Lark Declaration’s assertion of impacts to the Salt Creek tiger beetle due to the RFS is also unsubstantiated.

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[12] Note that these observations, including the statement regarding habitat threats, predate the EISA by 11 years.
[13] Nebraska Department of Game and Parks

3.6 Blackfooted Ferret (***Mustela nigripes***)

The black-footed ferret was listed as endangered across its entire range on March 11, 1967.[14] The is no critical habitat designated for this species. Black-footed ferret population status and distribution is closely tied to that of prairie dogs. Prairie dogs make up more than 90% of the black-footed ferret's diet and prairie dog burrows provide shelter and den habitat for the species. Major threats to black-footed ferret populations include conversion of native grasslands to agriculture, prairie dog eradication programs that were once widespread, and disease; and much of the remaining habitat for black-footed ferret is fragmented due to fragmentation of prairie dog towns by agriculture and human development (USFWS 2018).

The Lark Declaration states that "Given the connection between the Renewable Fuel Standard and the conversion of grasslands to agricultural land within the Black-footed ferret's range, further assessment seems warranted", but provides no explanation or evidence to support such a "connection". The Center for Biological Diversity (CBD 2019) reports that the last captive black-footed ferret died in 1980, and at that time, the animals were thought to be extinct in North America. In 1981 the species was re-discovered in a Wyoming prairie dog colony. Between 1991 and 1999, about 1,200 ferrets from that population were released at sites in Wyoming, Montana, South Dakota, Arizona and along the Utah/Colorado border (CBD 2019). It is estimated that about 1,410 black-footed ferrets are currently living in the wild (CBD 2019). Figure 11illustrates the estimated population status of black-footed ferrets in the wild, including a rapid recovery beginning in about 2000 and extending past 2008, the year the EISA was enacted and the RFS was implemented. The continual and unabated recovery of black-footed ferret populations after 2008 also serves to undermine the assertion in the Lark Declaration that the RFS has had adverse impacts on black-footed ferret.

Figure 12 illustrates the locations of black-footed ferret populations (reintroduced) and acres planted in corn and soy in 2018. With few exceptions, there is no overlap between counties with some acreage in corn or soy and locations where black-footed ferret have been introduced. The Lark Declaration presents no evidence of impacts from land conversion spurred by the RFS, and, in fact, evidence suggests that impacts due to loss of habitat (for all reasons) occurred long before any potential influence of the RFS and most of the species recovery has occurred since 2008. Therefore, we conclude that the Lark declarations' assertions of a causal relationship between the RFS and impacts to black-footed ferret lack foundation.

4. Lack of Evidence of a Causal Link between the RFS and Hypoxia in the Gulf of Mexico or the RFS and Water Quality Impacts in Streams Supporting Listed Species

4.1 Lack of Evidence of a Causal Relationship Between the RFS and Hypoxia in the Gulf of Mexico

The alleged link between increased corn production for ethanol and hypoxia in the Gulf of Mexico (and western Lake Erie) is addressed in Section 4.1 of the Ramboll report. While it is not unexpected that nutrient loading (including from agriculture in general) to the Gulf of Mexico via the Mississippi River contributes to the formation of a seasonal hypoxic "dead zone" in the Gulf of Mexico, there is no information demonstrating a link between increased corn ethanol production under the RFS and specific and quantifiable causes of observed hypoxic conditions in the northern Gulf of Mexico. The consensus, based on the vast majority of technical articles we have reviewed is that hypoxia is due to algal

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[14] https://www.fws.gov/mountain-prairie/es/blackFootedFerret.php

production driven by excess nitrogen that enters the northern Gulf of Mexico via the Mississippi River and related watersheds together with certain hydrologic conditions, including vertical stratification and temperature dynamics within the Gulf of Mexico water column. The hypoxia condition is not new and as shown by the U.S. Geological Survey and other institutions, has been an ongoing phenomenon for several decades and well before the RFS was initiated in 2008. The loading of annual nitrate plus nitrite to the Gulf of Mexico has been relatively consistent since comprehensive monitoring began in approximately 1980[15] with the three largest measured annual loading values occurring in 1993, 1983, and 1984, respectively, and thus well before the RFS was envisioned. Bianchi et al. (2010) conclude that understanding the complexity of this highly dynamic system or predicting flux and source areas with high precision is not reliable by simply referring to the numerous mostly general models that are relied on by recent authors (including Lark).

The Lark Declaration (at page 20), for example, refers to the pre-RFS study by Donner and Kucharik (2008) that "predicts" an increase in flux of dissolved inorganic nitrogen (DIN) by the Mississippi and Atchafalaya Rivers of between 10% and 34% using models that rely on hypothetical predictions of land use scenarios and discharge. Although Donner and Kucharik (2008) discuss the model validation approach, the validation results are imperfect indicating considerable overestimates in some cases, and underestimates in others. Furthermore, the model does not appear to provide a precise fit between the simulated results and the observed DIN discharge numbers collected from the few field stations identified in the study. An important complication in using a model like this to make predictions is the differentiation between urban (e.g., septic, industrial and municipal waste water plants, and residential runoff) and agricultural sources. As noted by Alexander, et al (2007), additional complications also are that nutrient sources typically are statistically estimated in the models, and then adjusted based on the model calibration. Model calibration uses "trial and error" processes for simulating numerous parameters that are themselves influenced by hydrologic and biogeochemical processes, nutrient uptake by a wide variety of soil types, climatic (short and long-term) conditions, and (as most relevant currently), improvements in fertilizer application and cropping and drainage patterns. Essentially, by providing examples of failed predictions using models, Bianchi, et al. (2010) make the case to not rely solely on numerical models.

Notably, the influence of weather is a very important condition for the formation of the Gulf of Mexico "dead zone" and is totally independent from loading of DIN from any particular sources. The influence of weather on the formation of the Gulf of Mexico "dead zone" is discussed in the Ramboll report in Section 4.1, page 26, where for example, the U.S. Geological Survey[16] estimated that flooding in the spring of 2019 resulted in an increased loading of nitrate and nitrite of approximately 18% when compared to the long-term average loading to the Gulf of Mexico.

The alleged quantitative relationship between increased corn grown for ethanol and nutrient loading to the Gulf of Mexico is further called into question by data from the U.S. Geological Survey indicating that annual nitrate plus nitrite loading to the Gulf of Mexico has remained relatively constant over the period 1980 to 2017 (Figure 13). This indicates that even during the period of increased use of corn for ethanol, there has been no appreciable net change to nutrient loading to the Gulf of Mexico. For this reason alone, there is no support for the assertion of a direct relationship between ethanol production on the hypoxia conditions in the Gulf of Mexico. In addition, EPA (2018) reports that there has actually been a reduction in total nitrogen concentrations in surface water bodies in Iowa which is the highest

[15] https://nrtwq.usgs.gov/mississippi_loads/#/GULF
[16] https://www.usgs.gov/news/very-large-dead-zone-forecast-gulf-mexico?qt-news_science_products=4#qt-news_science_products

corn producing state. This further refutes the broadly stated allegation that there is a link between expanded corn production (for any reason) and increased nutrient loading to the Gulf of Mexico.

The Lark Declaration mentions the following listed species as potentially impacted by the Gulf of Mexico dead zone: the threatened Gulf sturgeon (*Acipenser oxyrinchus desotoi*), the loggerhead turtle (*Caretta caretta* listed as endangered and threatened depending on location), and the endangered sperm whale (*Physeter macrocephalus*). With regards to Gulf sturgeon, it is instructive to look at the geographical location of critical habitat for the species and the occurrence of the dead zone in the Gulf of Mexico. The dead zone forms west of the Mississippi River Delta over the continental shelf (< 200 m water depth) of Louisiana and sometimes extends to Texas[17]. Figure 14 depicts Gulf sturgeon critical habitat occurring exclusively to the east of the Mississippi River delta and the hypoxic zone in 2019 (the largest recorded) located exclusively to the west of the Mississippi River delta. NOAA's Gulf of Mexico Hypoxia Watch site presents results from dissolved oxygen monitoring for the period 2001 to 2019[18]. These results show that hypoxia rarely extends near critical habitat areas for Gulf sturgeon, and when these conditions exist, they are limited to a relatively small area offshore of Biloxi, Mississippi. Waters to the east and south did not exhibit hypoxic conditions in any year monitored.

Moreover, the migratory behavior of Gulf sturgeon minimizes the probability of encountering hypoxic waters, should they occur in their critical habitat. Oxygen depletion in the Gulf of Mexico increases in late spring, worsens over the summer, then dissipates in the fall; whereas gulf sturgeon move into rivers in the spring and fall and spend the summer months in the riverine habitat, then subadults and adults move into estuarine waters in the fall to feed and then move into marine waters in the winter. Thus, the Lark Declaration provides no evidence of a relationship between Gulf sturgeon critical habitat and potential impacts from hypoxia in the Gulf of Mexico due to nutrient inputs from the Mississippi River basin. In addition, NOAA does not list hypoxia as a threat to the species, rather it lists contaminants, dredging, dams, and climate changes as the threats[19]. For these reasons, the presumption in the Lark Declaration that the RFS has resulted in impacts to Gulf sturgeon is unsubstantiated.

Loggerhead turtles and sperm whales have pan-global ranges and only a limited number of individuals over a limited portion of their life spans would be likely to encounter the Gulf of Mexico dead zone. Because both are air-breathing animals, adverse effects to these species from hypoxia, if any, could only be indirect (e.g., reduced prey abundance).

The loggerhead turtle is the most common sea turtle in the southeastern U.S., and they nest mainly along the Atlantic coast of Florida, South Carolina, Georgia, and North Carolina and along the Florida and Alabama coasts in the Gulf of Mexico[20]. The Lark declaration states that "The increasing frequency of red tides and harmful algae blooms in the Gulf of Mexico as well as the increased duration and extent of the hypoxic dead zone caused by agricultural runoff in the Mississippi River have been reported to both directly and indirectly affect sea turtles" and cites NMFS et al. (2011) for this proposition. Yet, NMFS (2011) makes no mention of hypoxia, and red tide is only mentioned in the context of the west coast of Florida. The Lark Declaration also states that "Loggerheads in the near-shore northern Gulf of Mexico waters may be exposed to hypoxia…", citing Hart et al. (2013) for this proposition. However, Hart et al. (2013) studied nesting sites and movement patterns only along the Alabama and Florida coasts and reported movement patterns southward along the Florida west coast, away from the Gulf of Mexico dead zone. As noted above, since 2001, hypoxia in the Gulf of Mexico did not extend to the west

[17] https://pubs.usgs.gov/fs/2006/3005/fs-2006-3005.pdf
[18] https://www.ncddc.noaa.gov/hypoxia/
[19] https://www.fisheries.noaa.gov/species/gulf-sturgeon
[20] https://www.fisheries.noaa.gov/species/loggerhead-turtle

coast of Florida. Therefore, the assertions in the Lark Declaration that the RFS has resulted in impacts to loggerhead turtles by means of hypoxia in the Gulf of Mexico lack foundation.

Sperm whales inhabit all of the world's oceans, having one of the widest distributions of any marine mammal. NOAA does not list hypoxia as a threat to this species, rather vessel strikes, entanglement, ocean noise, marine debris, climate change, oil spills, and contaminants are listed as threats.[21] Several researchers have investigated the distribution of sperm whales and other cetaceans in the Gulf of Mexico. Davis et al. (2002) report a resident breeding population within 100 km of the Mississippi delta and suggest that the edge of the continental slope south of the Mississippi River delta provides the oceanographic deep-water conditions with locally enhanced primary and secondary productivity. The Gulf of Mexico dead zone does not extend to the continental slope, rather it is oceanographically limited to the continental shelf where water depths are less than 200 m.

In sum, attributing adverse impacts to these species to hypoxia induced by nutrient enrichment in the Mississippi River basin is speculative. Attributing any potential for adverse effects due solely to theoretical increases in nutrient inputs from expanded corn production spurred by the RFS is unsupportable.

4.2 Lack of Evidence of a Causal Relationship Between the RFS and Water Quality Impairment in Streams Supporting Listed Species

Surface water use impairment is determined under Section 303(d) of the Clean Water Act, which predates initiation of the RFS program by several decades. The Lark Declaration at Appendix 5 provides maps of 303(d) impaired water bodies in several geographic regions and asserts a causal relationship between the RFS and the 303(d) listing. Figure 15 compares the 303(d) maps for 2002 (as produced by the State of Illinois) and the 2015 map presented in the Lark Declaration. This figure clearly shows that a major water body near Carbondale has been impaired for more than 17 years—well before the RFS went into effect in 2008. Similar comparisons can be made for areas of North Dakota used for illustration in the Lark Declaration at page 92 where 303(d) impairments were tracked by the State in 2004[22]), and for areas of central Minnesota (Lark Declaration at page 94) where in 2002, the Minnesota Pollution Control Agency (MPCA) provided a list of its 303(d) impaired lakes[23] noting that nutrients were part of the root cause in many of them. The attempt in Lark's Declaration to tie such impairments to the RFS using the 303(d) maps (Appendix 5) is fundamentally flawed, for the reasons described below.

The maps shown in the Lark Declaration (Appendix 5) do not show watershed hydrology that explicitly links areas of crop production, ethanol refining, and impaired water bodies. As Bianchi, et al (2010) observed, general maps and information, such as the geographic placement of crop production in a regional map, is insufficient to establish a causal link between the RFS and water quality due to the complexities of numerous factors, including timing, weather, local farming practices, soil chemistry and physical properties, hydrology, other rural and urban release mechanisms.

Another example of the Lark Declaration's presentation of faulty data—with respect to the alleged link between the RFS and streams with impaired water quality—is a sub-basin in northeastern Kansas depicted on figure 5-6 of the Lark Declaration. We selected this sub-basin for closer examination because it appears to be a worst-case example of the purported causal relationship, based on the relatively large proportion of area identified as land converted to corn or soy and the proximity of a relatively large area to a 303(d) listed water body. Figure 16 presents a reproduction of figure 5-6 from

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21 https://www.fisheries.noaa.gov/species/sperm-whale

22 https://deq.nd.gov/publications/WQ/3_WM/TMDL/1_IntegratedReports/2004_Final_ND_Integrated_Report.pdf

23 https://mn.gov/law-library-stat/archive/urlarchive/a042033-1.pdf

the Lark Declaration, together with the selected sub-basin and presumed land conversion area immediately adjacent to the stream. Figure 17 presents a simple examination of publicly-available Google Earth aerial images of this area over time is instructive. Google Earth aerial images of this area clearly depict it in agricultural use as early as 1991 with no apparent expansion since that time including the period 2008-2018.

Further, we performed a spatial analysis of land allegedly converted to biofuels feedstock cultivation after 2008 (using figure 5-6 from the Lark Declaration) within the watershed depicted in Figure 17 (even though we know that in at least the case illustrated by Figure 17; this allegation is incorrect). Such an analysis indicates that in the watershed area of 55,840 acres, the total area devoted to crops (exclusive of grassland) in 2015 based on LCD NASS data, was approximately 18,940 acres (or 34% of the total watershed area). Of the total acres in crops, approximately 880 acres (or 1.6% of the watershed) was in corn or soy in 2015. Of the allegedly "converted" fields identified in the watershed in figure 5-6 of the Lark Declaration, the closest field to the impaired water body is approximately 390 feet (the field shown in Figure 17) and the average distance of all presumably converted fields to the impaired stream is approximately 4,860 feet. Barring mass wasting of agricultural soils, very poor practices, or spills of fertilizers, loading of nutrients to water bodies from agricultural fields (e.g., in pounds per acre per year) is expected to decrease with distance; even at a distance of 390 feet, an appropriately managed farm field would be expected to have very little transport of nutrients over that distance. Even if one assumed that all of the presumably "converted" areas were indeed converted, the total loading of nutrients from these fields (all else being equal) compared to all other agriculture would be expected to be vanishingly small (e.g., the presumably "converted" soy and corn area is only about 4.6% of the total crop area).

As an additional example, we performed a spatial analysis of the watershed associated with critical habitat for the Arkansas shiner (*Notropis girardi*) as depicted in the Lark Declaration at page 105. For this watershed area of 471,400 acres, the total area devoted to crops (exclusive of grassland) in 2016 based on LCD NASS data, was approximately 175,500 acres (or 37% of the total watershed area). Of the total acres in crops, approximately 590 acres (or 0.13% of the watershed) was in corn or soy in 2016. Of the allegedly "converted" fields identified in the watershed in the figure at page 105 of the Lark Declaration, the closest field to the impaired water body is approximately 2.5 miles and the average distance for all fields to the critical habitat is approximately 10 miles. For this example, even if one assumes that all the area devoted to corn or soy in 2016 was the direct result of the RFS, the proportion of total crop area and the distance between the corn or soy fields is so vanishingly small as to undermine any claims of impact to the Arkansas shiner.

These quality control checks on the evidence presented in the Lark Declaration demonstrate the flawed nature of the assertions presented therein. This analysis, along with the fact that the 303(d) designations predate the RFS, undermines the assertion that there is a causal relationship between the RFS, reduced water quality in Section 303(d) impaired streams, and potential adverse impacts to listed aquatic species.

5. Conclusions

Our conclusions follow from the technical review of the assertions made in the Lark Declaration including an evaluation of the literature cited and an independent check of the geographical information presented in the Declaration. Our conclusions include the following:

- Assertions that increased corn ethanol production under the RFS has resulted in land use change and conversion of non-agricultural land to production of biofuel feedstock are unsubstantiated

- Acres planted in corn across the United States has remained close to or below the total acres planted in the early 1930s despite increases in demand for corn as human food, animal feed, and biofuels over this nearly 90-year period. The increase in demand has largely been met by an approximately 7-fold increase in yield (bushels per acre). The lack of causal relationship between demand for corn and acres planted in corn calls into question the causal relationship between increased demand for corn for ethanol and land conversion, and, in turn, potential impacts of land conversion on endangered species.
- The causal relationship between the RFS and the price of corn is unsupported by the evidence. Recent efforts to quantify the relationship ignore the multiple domestic and international economic factors affecting the price of corn. These factors include the overall increase in global consumption of agricultural commodities in general, due to expanding economies. In addition, most of the increase in the price of corn (as well as other crops like soy and wheat) since 2005 has been attributed to higher oil prices.
- The Lark Declaration (and the literature relied upon therein) does not adequately consider the myriad factors that influence a farmer's decision to convert non-agricultural land to growing any given crop. In addition, the Lack Declaration fails to consider that converting new land is likely the least preferred option a farmer has for increasing production because it most likely involves additional expenditures such as land clearing and other preparation. Nor does it consider that the potential yield that can be expected of new fields, which, relative to existing fields, may be sub- or infra-marginal and may require more intensive inputs to achieve desired yields. For these and other reasons, assertions in the Lark Declaration that the RFS has resulted in land conversion are unsubstantiated.

- Assertions that RFS-driven land use change has resulted in impacts to particular ESA listed species are without foundation —The Lark Declaration asserts that land use change spurred by the RFS has resulted in impacts to listed terrestrial species of birds, mammals, and insects. However, the evidence presented is poorly researched (including citations to irrelevant documents and misinterpretation of data) and the examples used to support many assertions instead actually *refute* the assertions. For example, eggshell thinning in birds is mentioned as a potential impact of biofuels production, yet the chemicals responsible for this adverse effect were banned decades before the RFS took effect. In addition, several examples of supposed land use change were presented using approaches that are shown to be flawed, among other things, by testing the assertions against images from Google Earth. Specifically, we checked several claims of land conversion that are based on methods by Lark et al. (2015) against historical Google Earth Images that clearly show fields had been converted long before the RFS went into effect (e.g., in areas allegedly impacting the whooping crane, Poweshiek skipperling, and yellow-billed cuckoo).
- Assertions that RFS-driven biofuels agriculture adversely impacts water quality are unsubstantiated—The Lark Declaration asserts that biofuels (corn and soy) agriculture has worsened the Gulf of Mexico dead zone, imperiling Gulf sturgeon, loggerhead turtles, and sperm whales, yet provides no supporting evidence. The Lark Declaration fails to cite any studies that associate corn or soy crops (let alone corn or soy crops directly traced to the RFS program) to any impacts to these species or their habitats. In fact, information related to the life histories of all three species indicates that the area within which the dead zone forms each summer does not overlap geographically (or temporally, in the case of the Gulf sturgeon) with critical or important habitat of any of the species. The Lark Declaration also fails to consider that the Gulf of Mexico dead zone had been forming on a regular basis for decades before the RFS went into effect. The Lark Declaration also asserts that biofuel (corn and soy) agriculture is associated with state designation of impaired waters pursuant to Section 303(d) of the Clean Water Act but

fails to acknowledge cases in which such designations were made well before the RFS came into effect. it also presents no assessment of the potential loading of nutrients to impaired water bodies. Our independent assessment of specific examples indicates that an assertion of impacts from corn or soy on impaired water bodies is unsubstantiated.

In sum, there are at least two important causal chains that must be quantified and linked together to demonstrate a relationship between increased corn ethanol production under the RFS and impacts to ESA-listed species: 1) a causal chain linking the RFS to land use change and water quality impacts; and 2) a causal chain linking these impacts to land and water with specific impacts on the survival or reproduction of ESA-listed species. Each of these causal chains is made up of many embedded biophysical and economic relationships that, in turn, are influenced by a myriad of interrelated variables. The Lark Declaration fails to consider these causal relationships in a meaningful way, relying instead on unfounded assumptions and speculation to support its thesis.

6. References

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7. Figures

Figure 1. Total U.S. Planted Acres of Corn Per Year

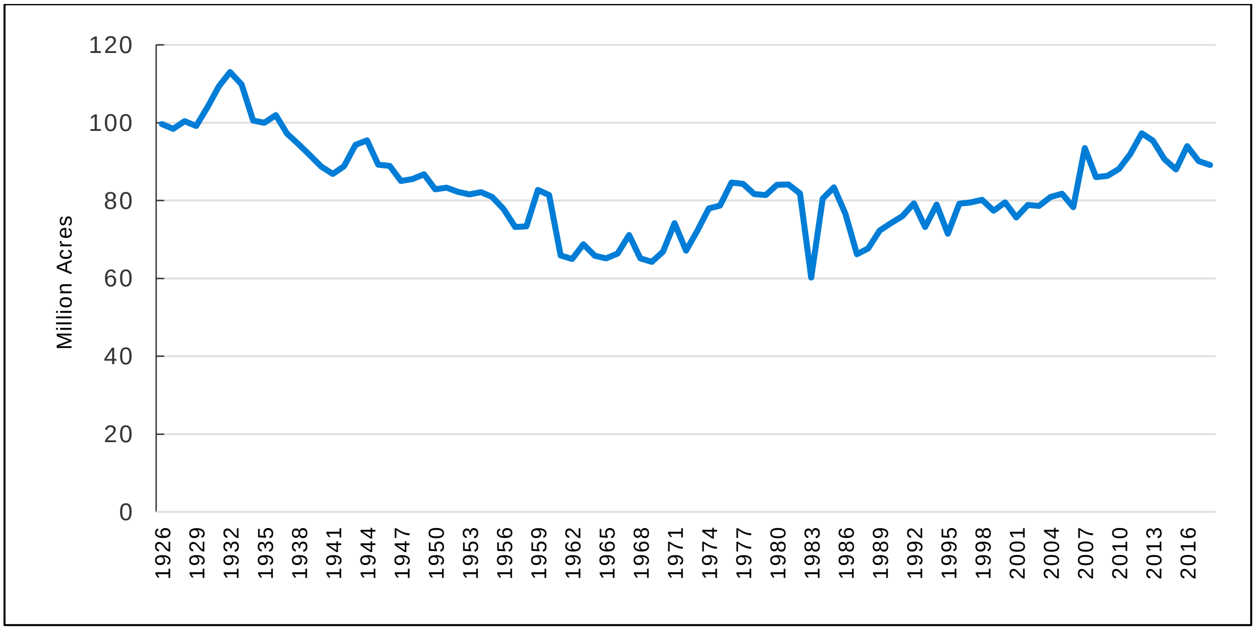


Figure 2. The decision about which crop to plant is made at the farm level, and takes many different components into account

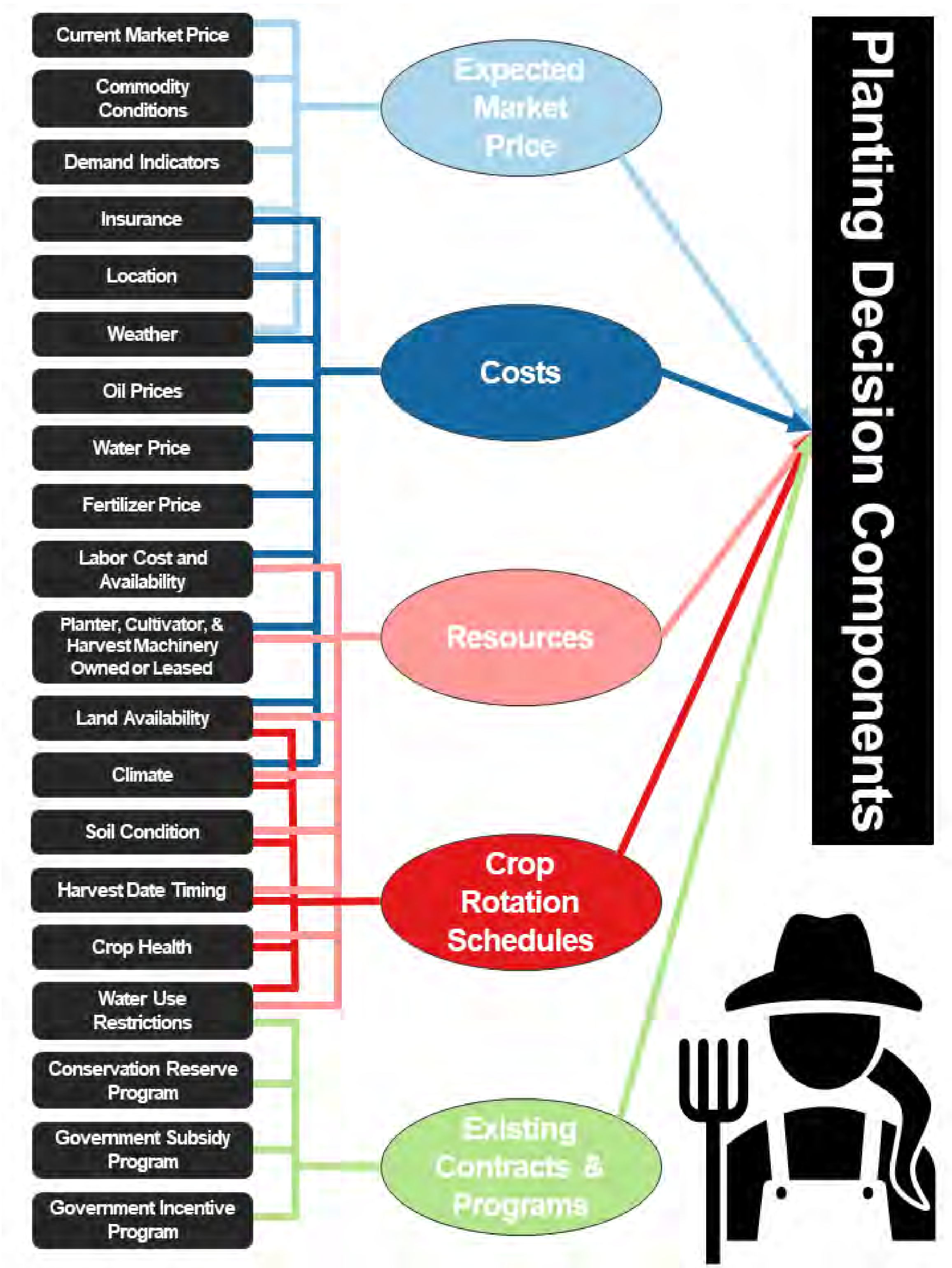


Figure 3. U.S. crude oil prices compared to crop prices, 2005 to 2015. From Staab, et. al. 2017

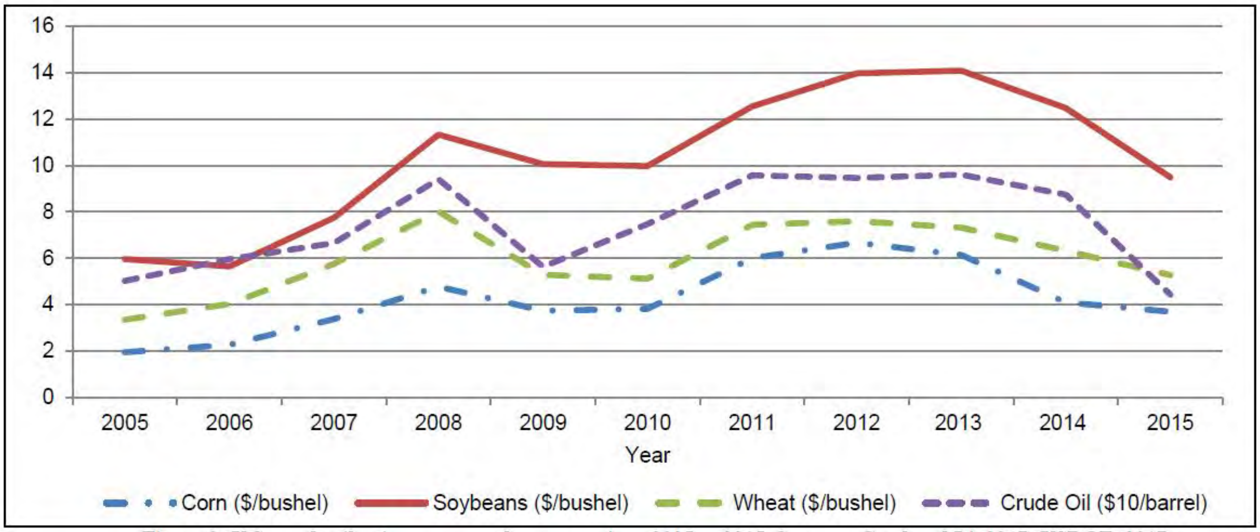
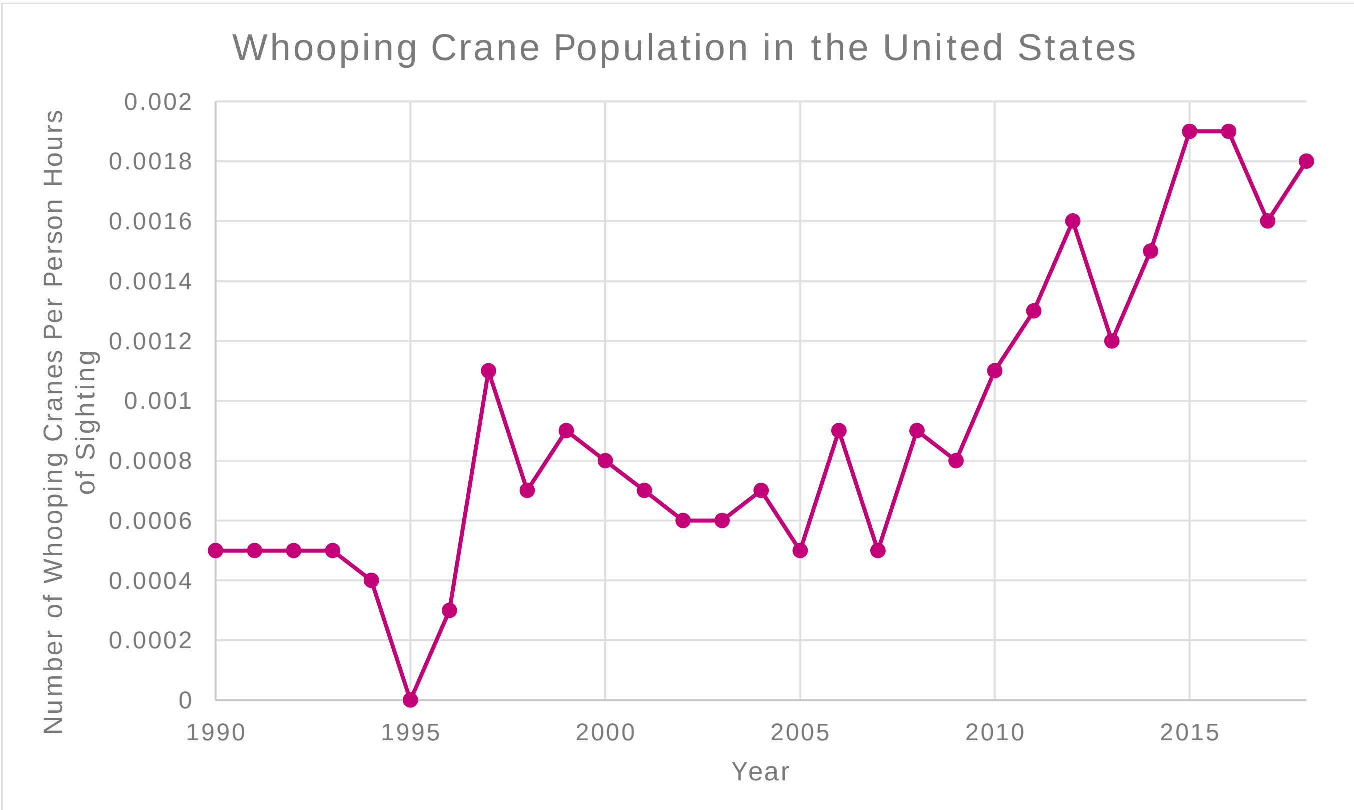


Figure 4. United States whooping crane population 1990 to present. Data are from the Audubon Society's Christmas Bird Count Database[24] and are shown here by the number of cranes per person hour of observation time.



[24] http://netapp.audubon.org/CBCObservation/

Figure 5. Example of error in USDA Cropland Data Layer upon which Lark's argument rests. An area within the Cheyenne Bottoms Reserve was identified as corn by the USDA CDL. Examination of aerial imagery showed no corn, and conversations with staff at the reserve confirmed that corn was never planted there.

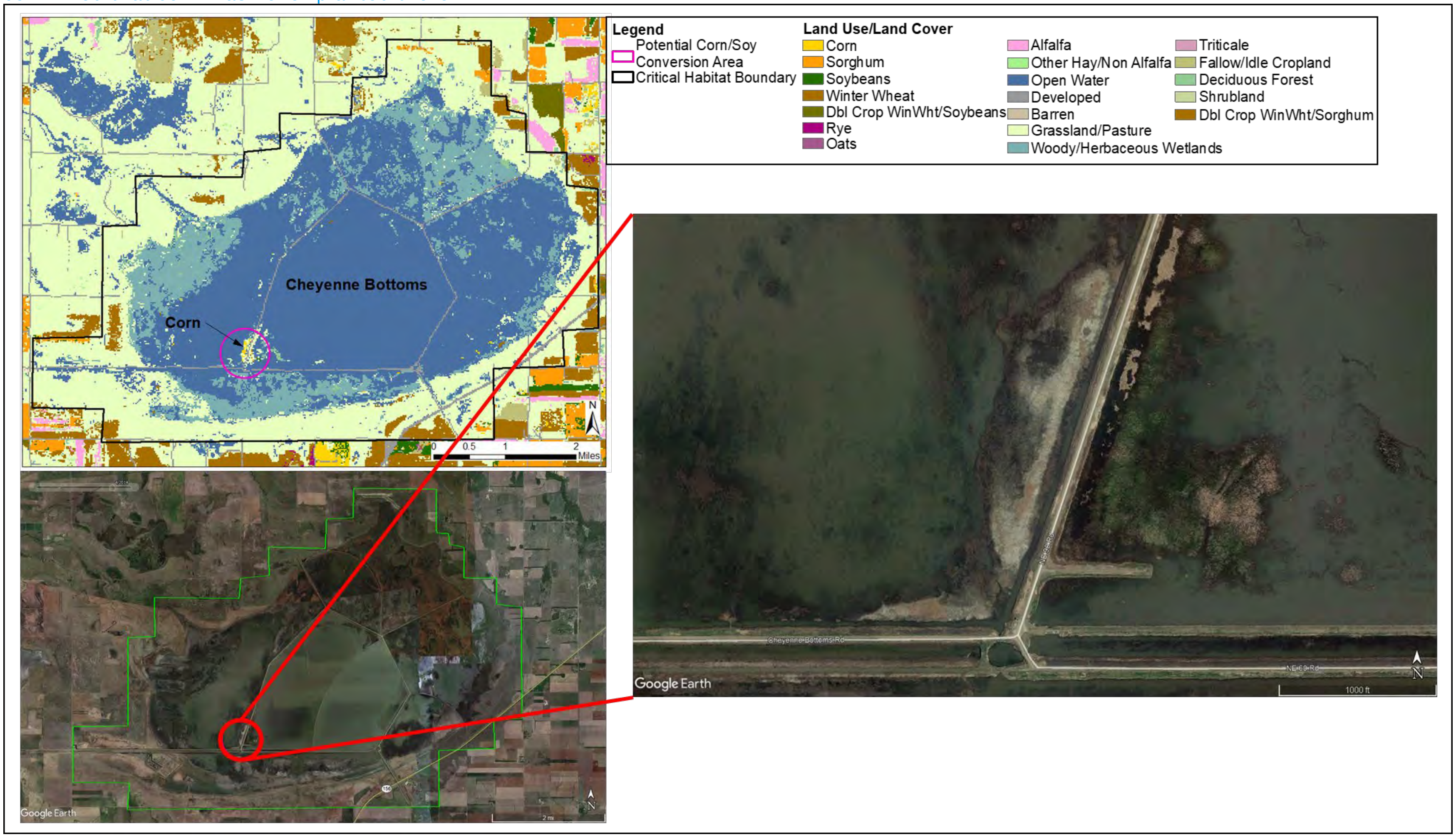


Figure 6. Western yellow-billed cuckoo critical habitat and corn and soy production by county

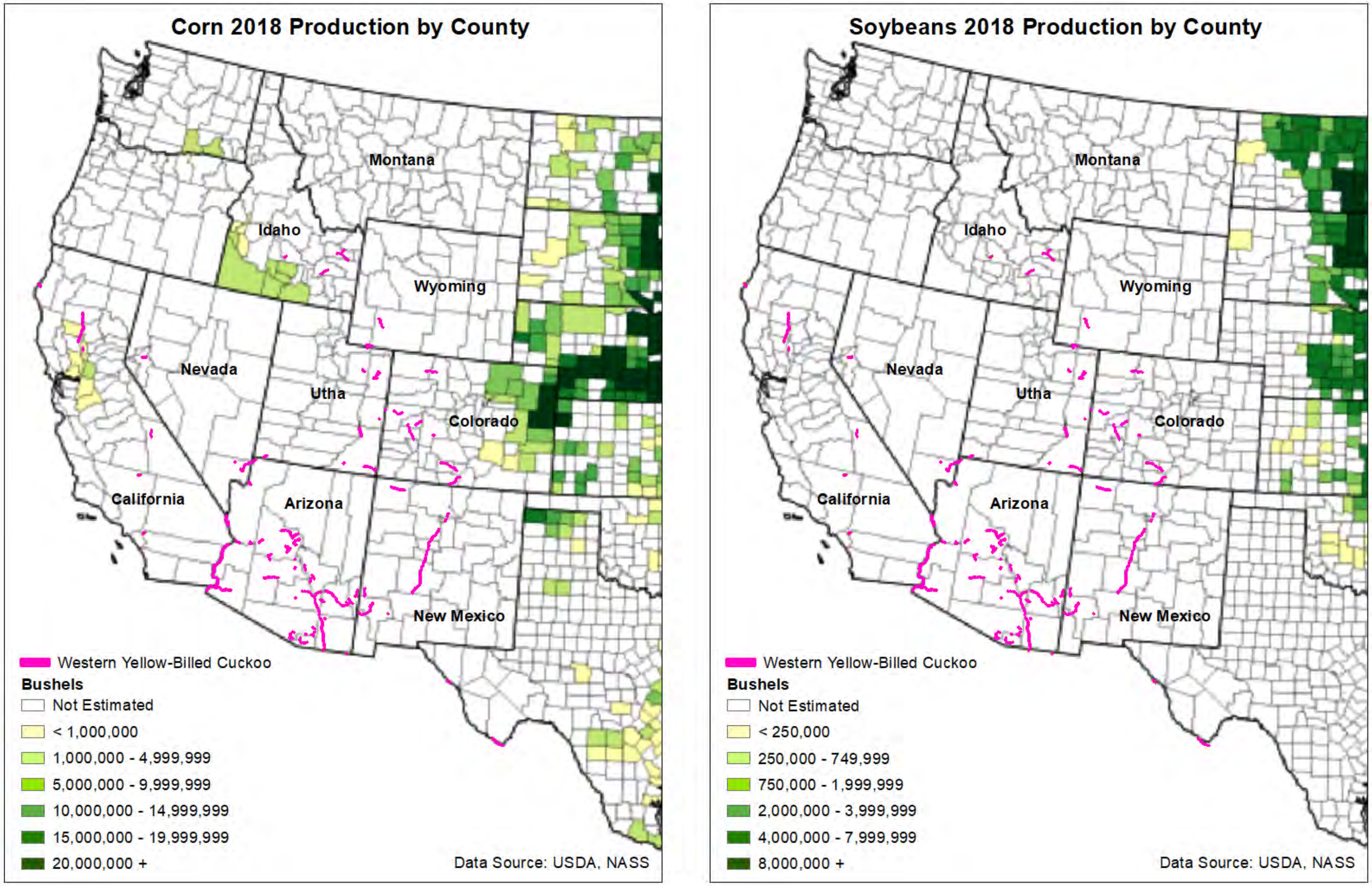


Figure 7. Area near Los Molinos, CA where 2018 CDL show corn within the boundaries of the critical habitat for Western yellow-billed cuckoo and Google Earth images from 1998 2014 document no conversion after 2008.

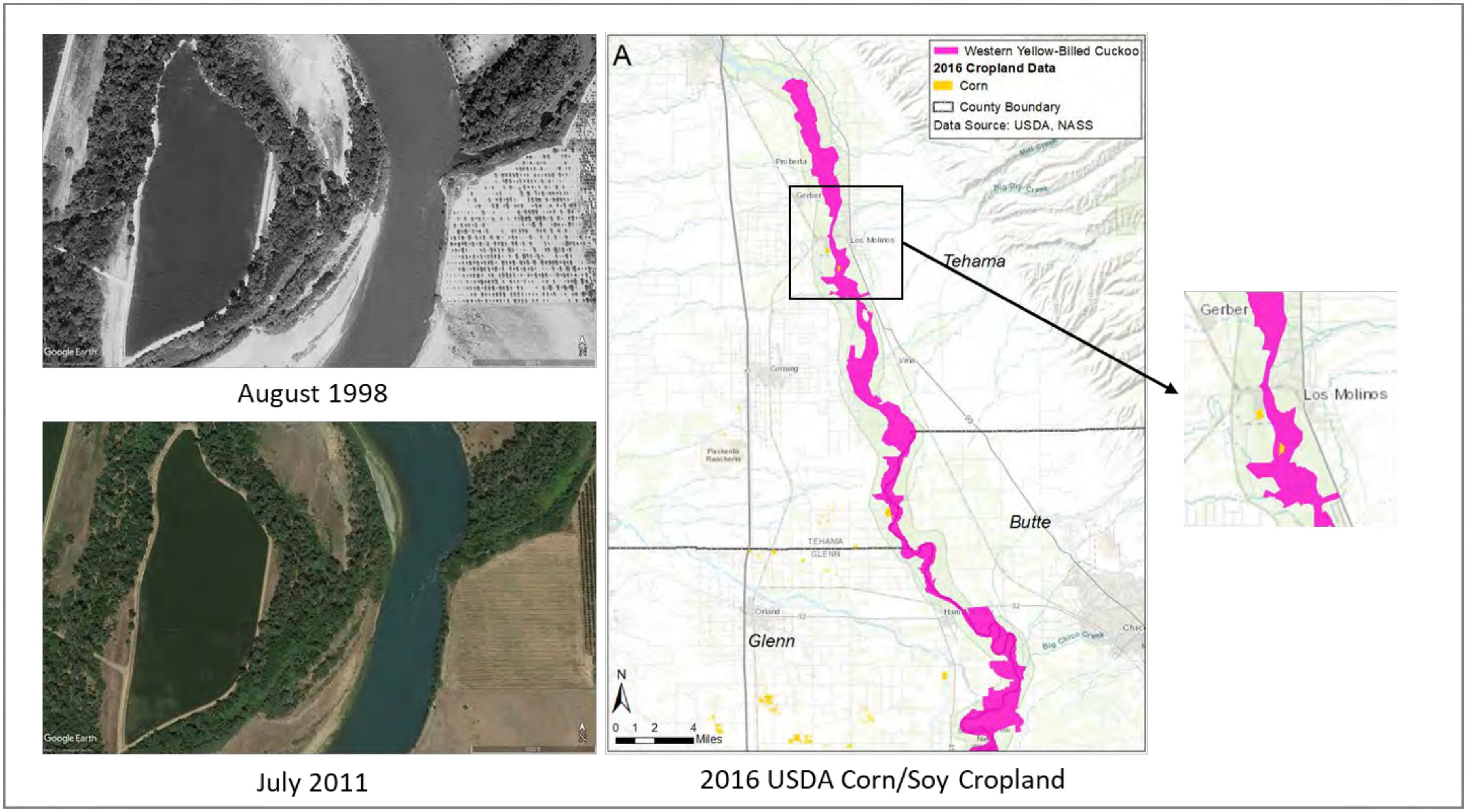


Figure 8. Area near Butte, CA where 2018 CDL show corn within the boundaries of the critical habitat for Western yellow-billed cuckoo and Google Earth images from 1998 2014 document no conversion after 2008.

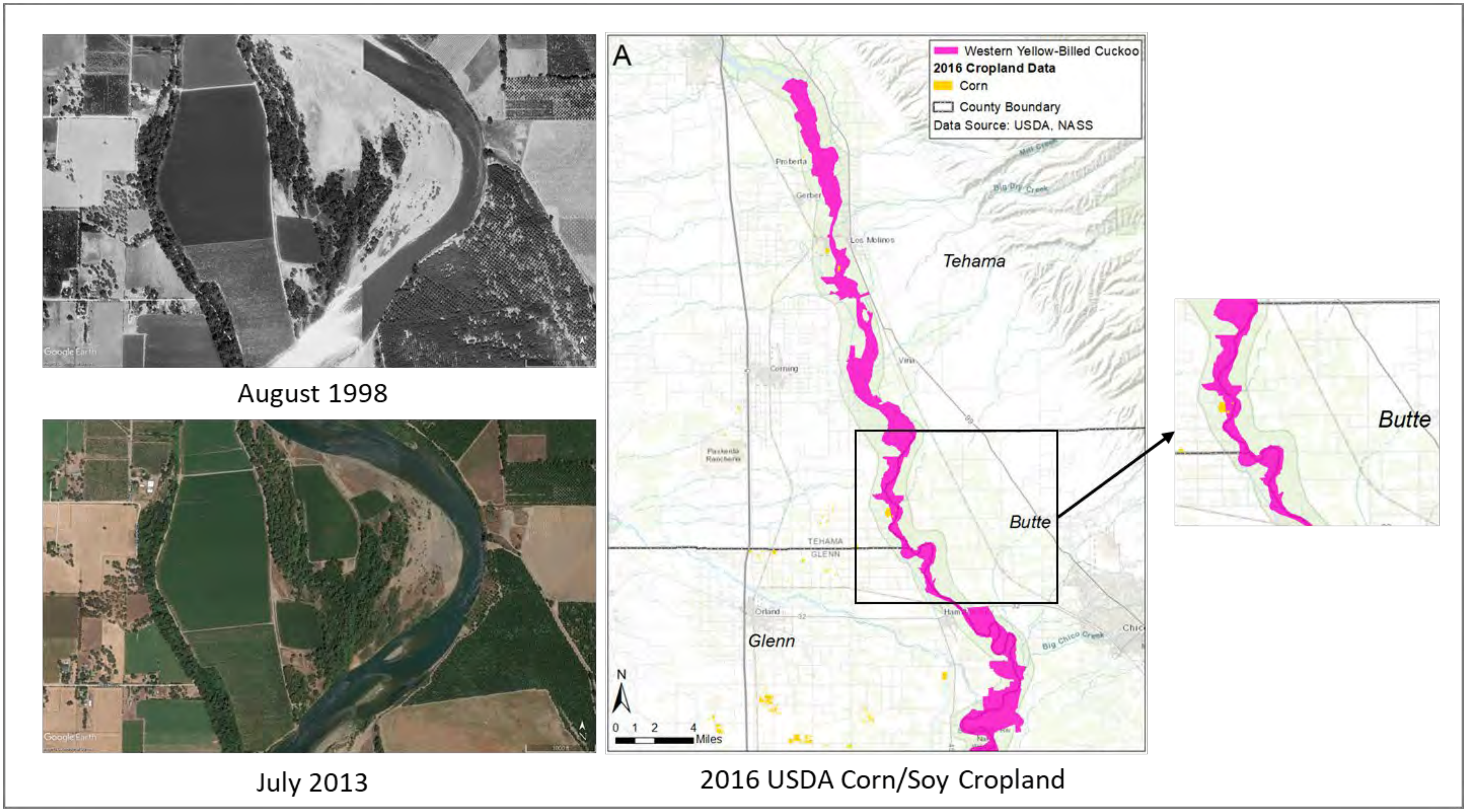


Figure 9. Aerial images from Google Earth demonstrating that the area highlighted in the Lark Declaration Appendix 6 was clearly in agriculture as early as 1991, and there was no evident expansion of the area into what is now designated as critical habitat for Poweshiek skipperling after 2008

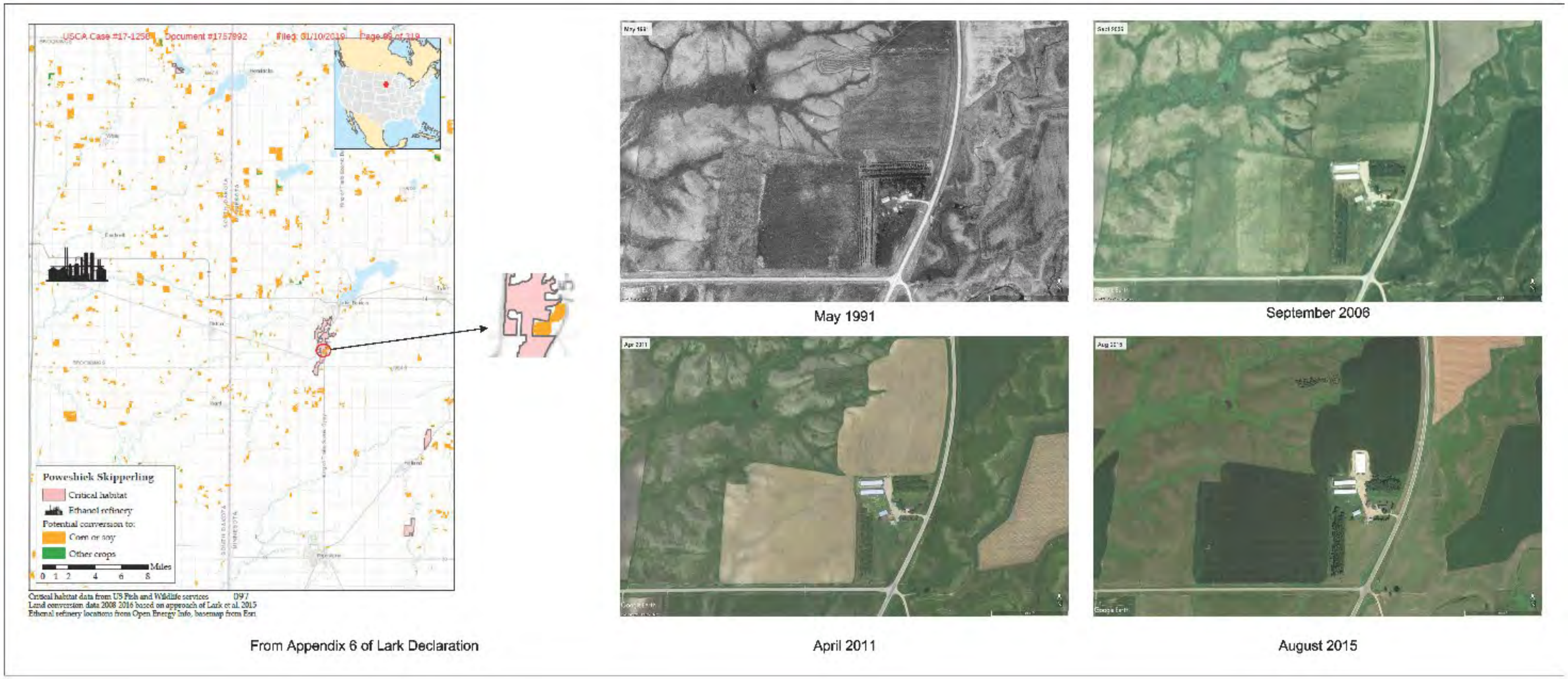


Figure 10. Adult Salt Creek tiger beetles counted during visual surveys 1991-2012 (excerpted from Federal Register 2013)

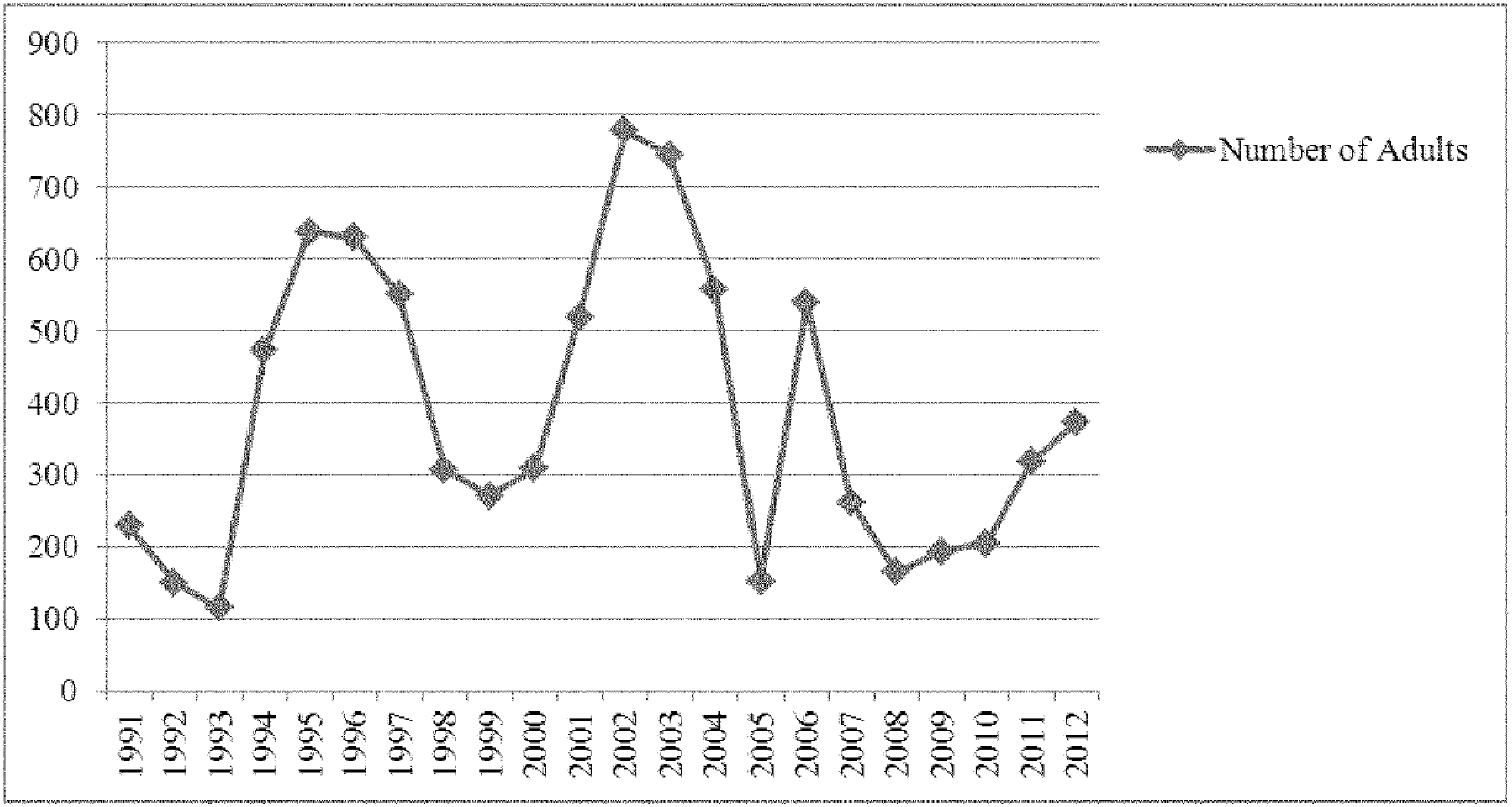
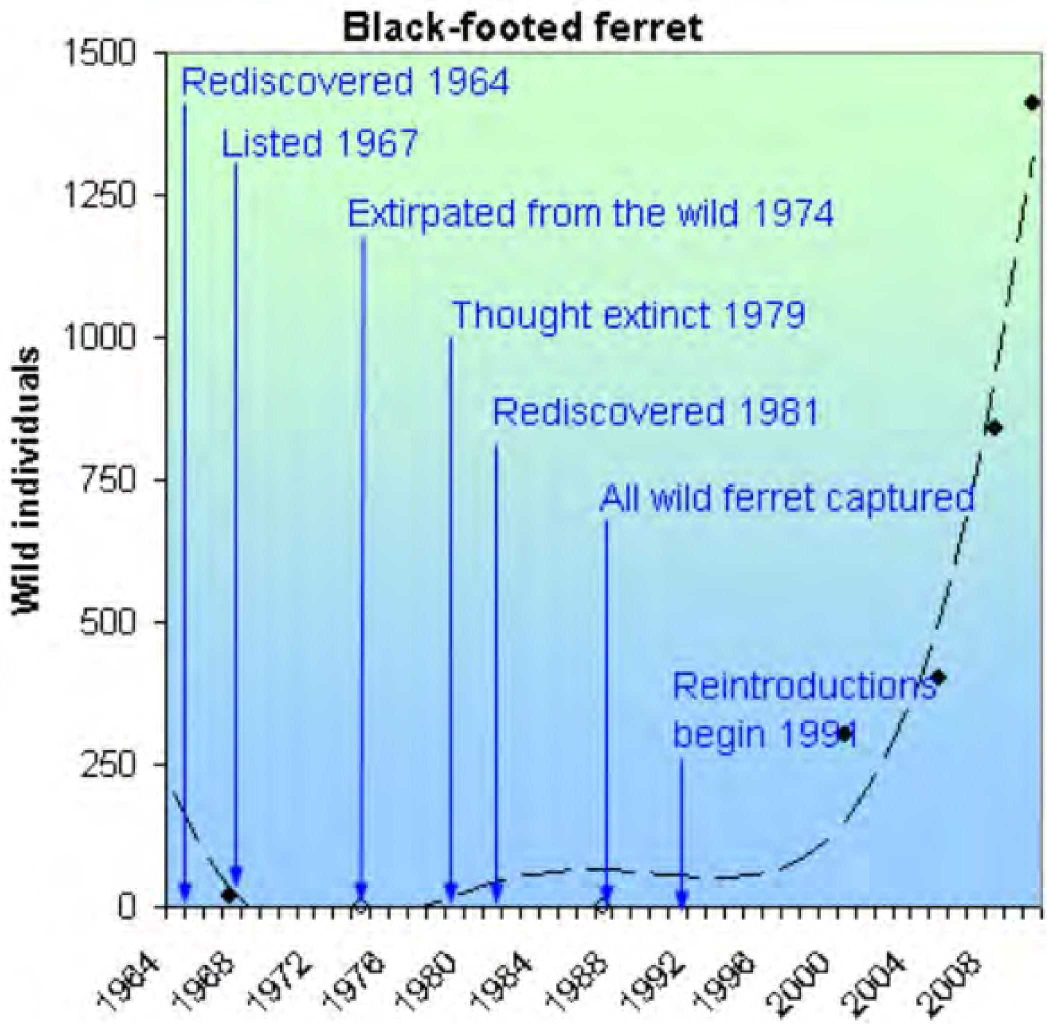


Figure 11. Wild black footed ferret population status 1964 to 2012



SOURCE: https://www.biologicaldiversity.org/species/mammals/black-footed_ferret/

Figure 12. Location of black-footed ferret populations and counties with corn and soy planted 2018

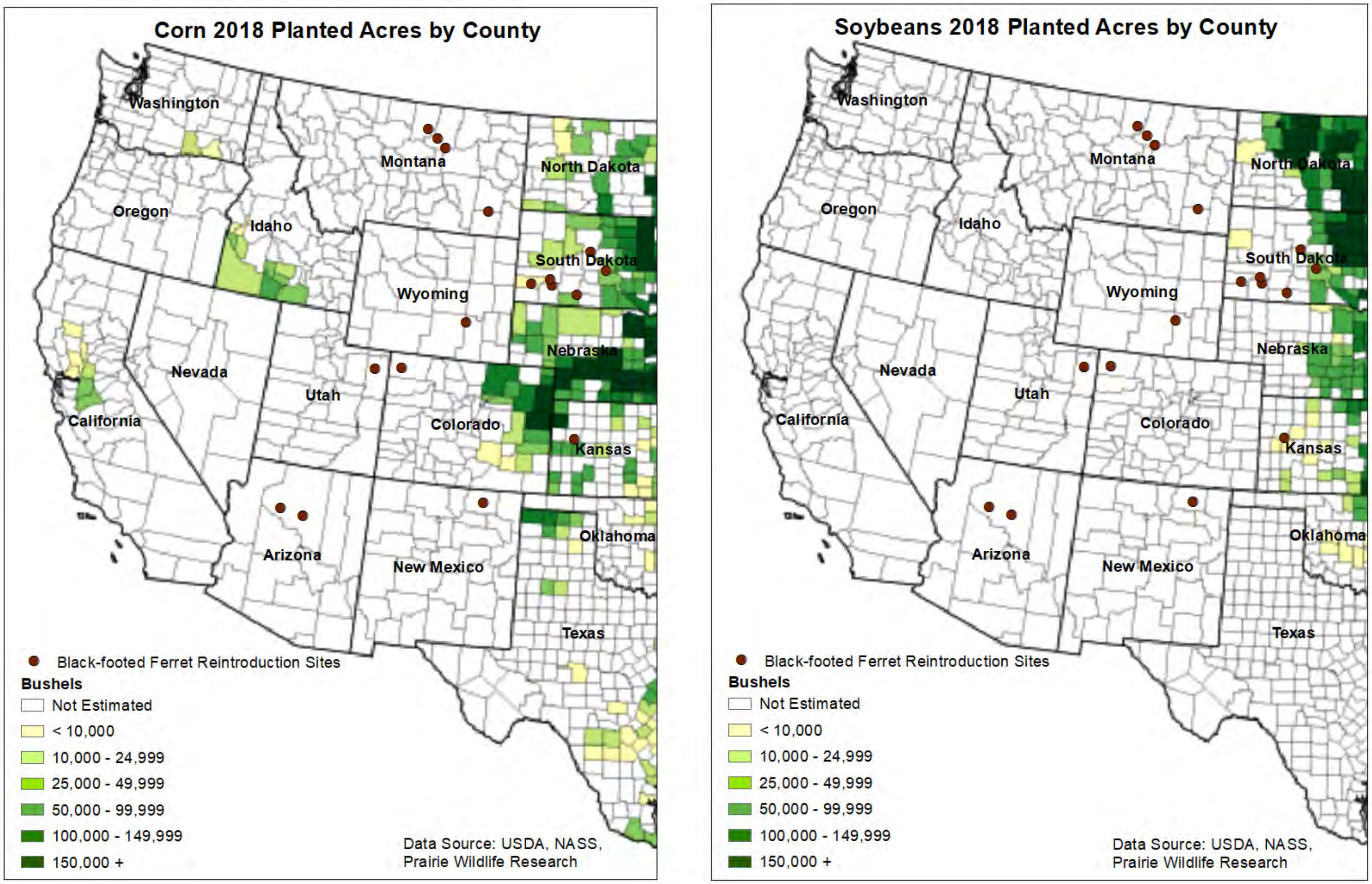
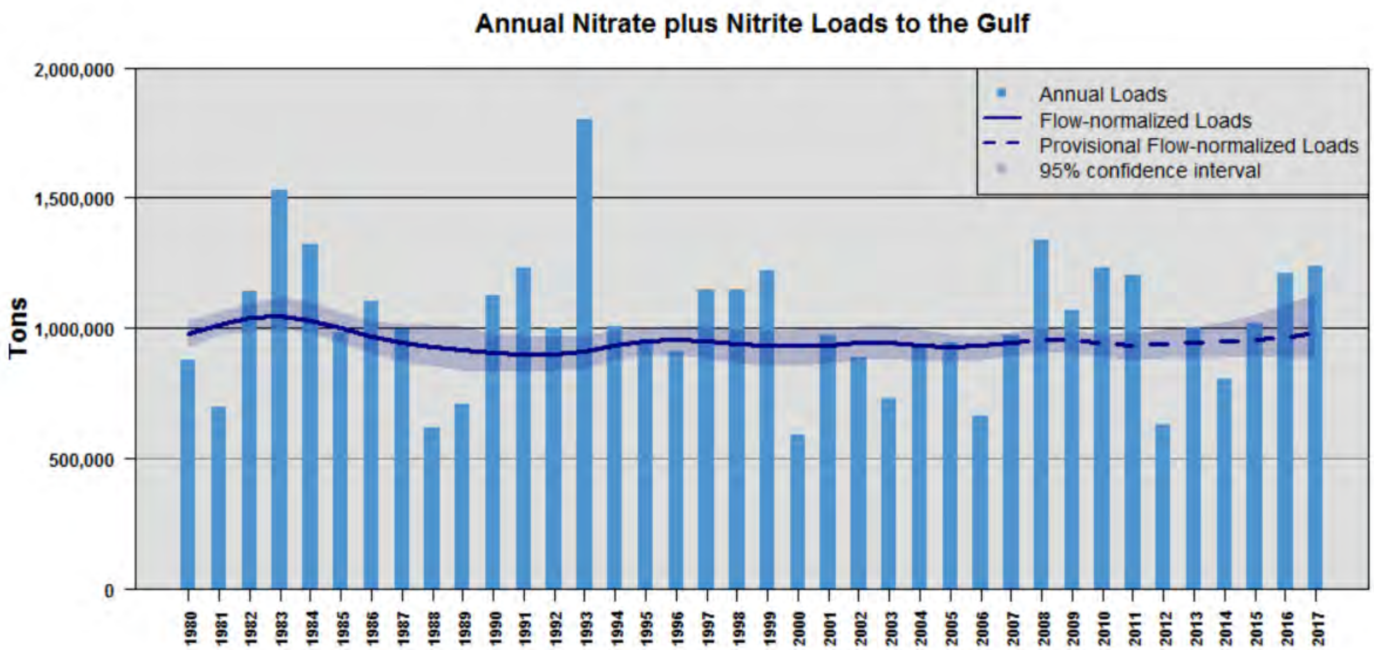
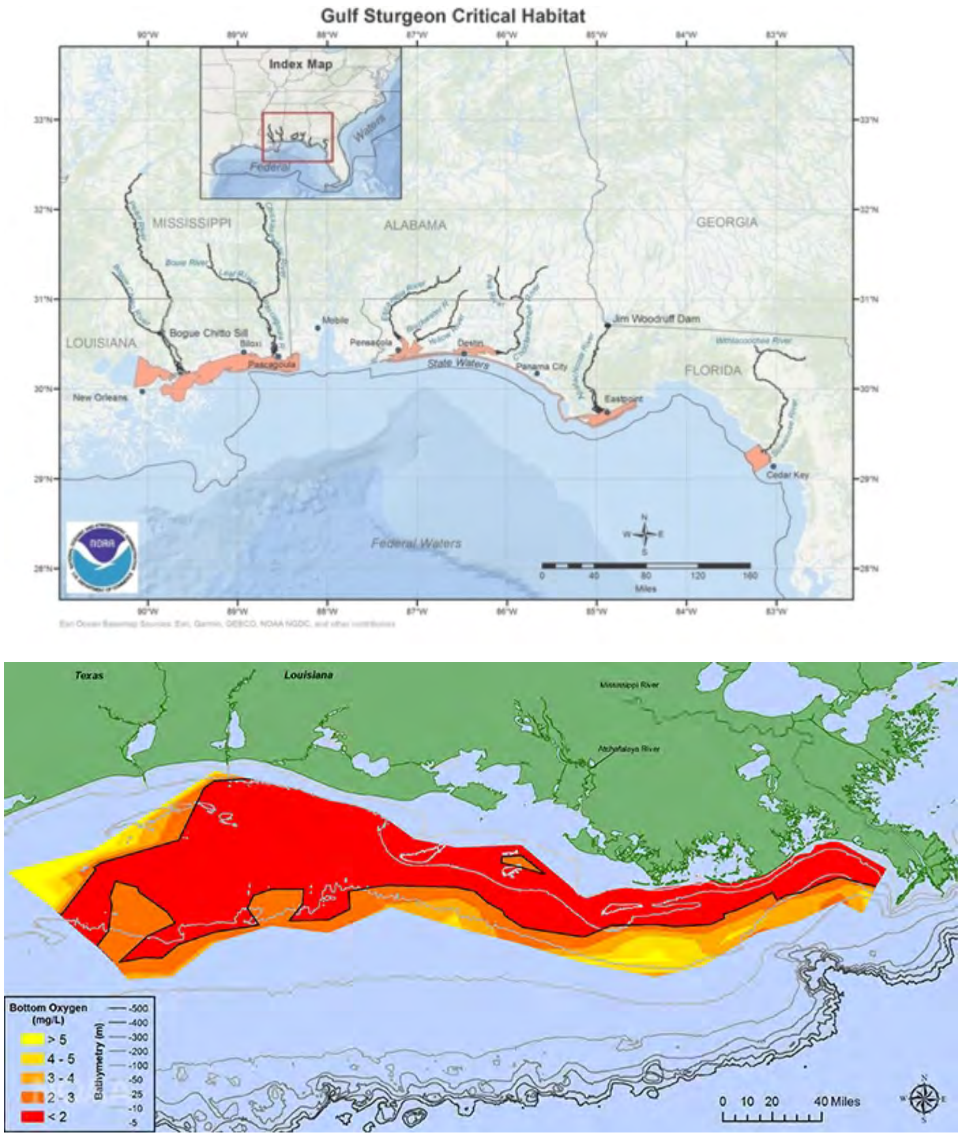


Figure 13. Annual nitrate plus nitrite loading to the Gulf of Mexico 1980 to 2017



Source: USGS n.d.

Figure 14. Gulf sturgeon critical habitat and the Gulf of Mexico dead zone in 2019; the largest dead zone recorded



SOURCE: https://www.noaa.gov/media-release/gulf-of-mexico-dead-zone-is-largest-ever-measured by (Courtesy of N. Rabalais, LSU/LUMCON)

Figure 15. 303(d) maps for 2002 (as produced by the State of Illinois) and the 2015 map presented in the Lark Declaration showing that a major water body near Carbondale has been impaired for more than 17 years—well before the RFS went into effect in 2008.

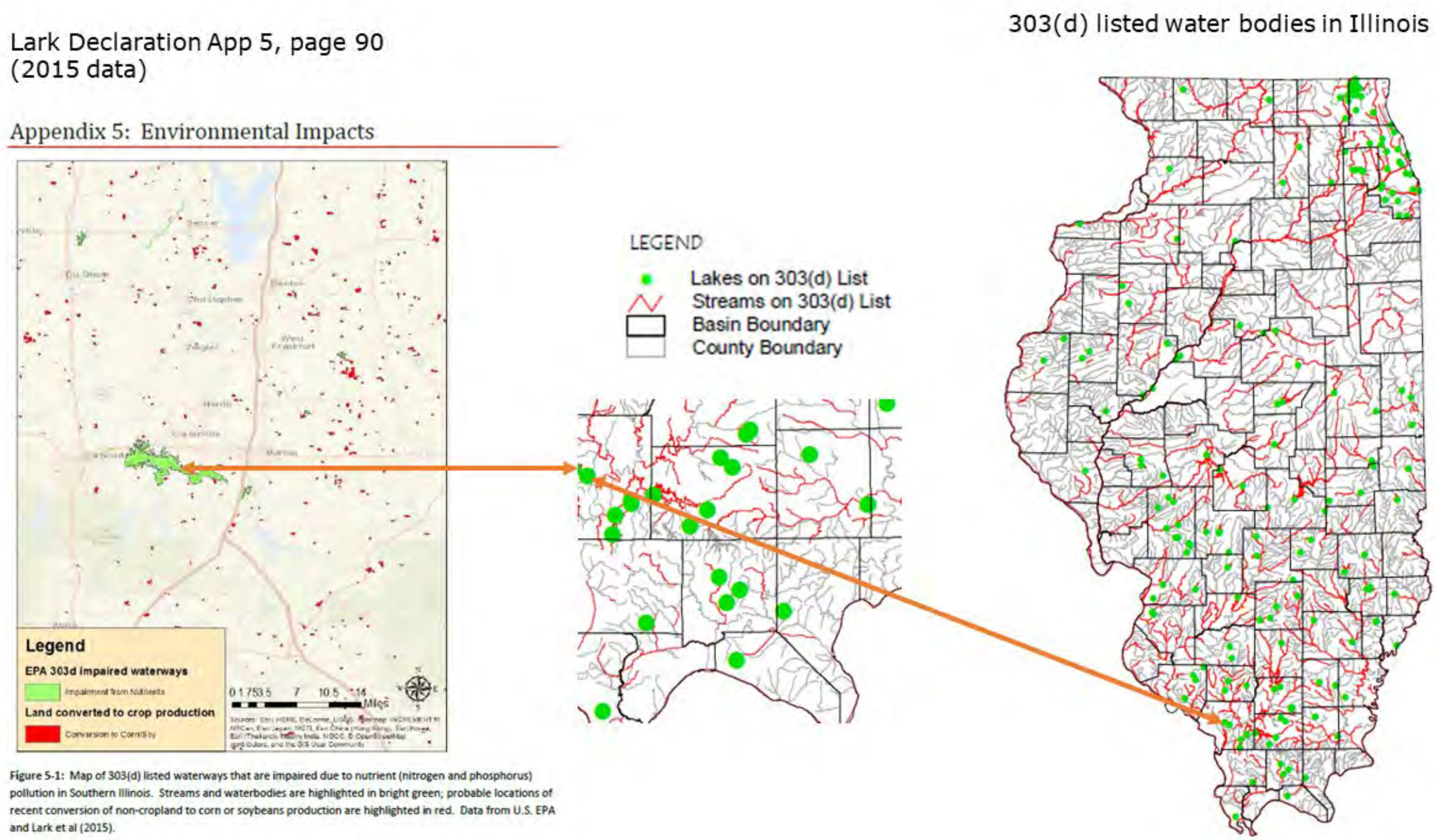


Figure 16. Watershed area selected for spatial analysis of presumed land conversion relative to 303(d) designated streams as identified in Figure 5-6 of the Lark Declaration

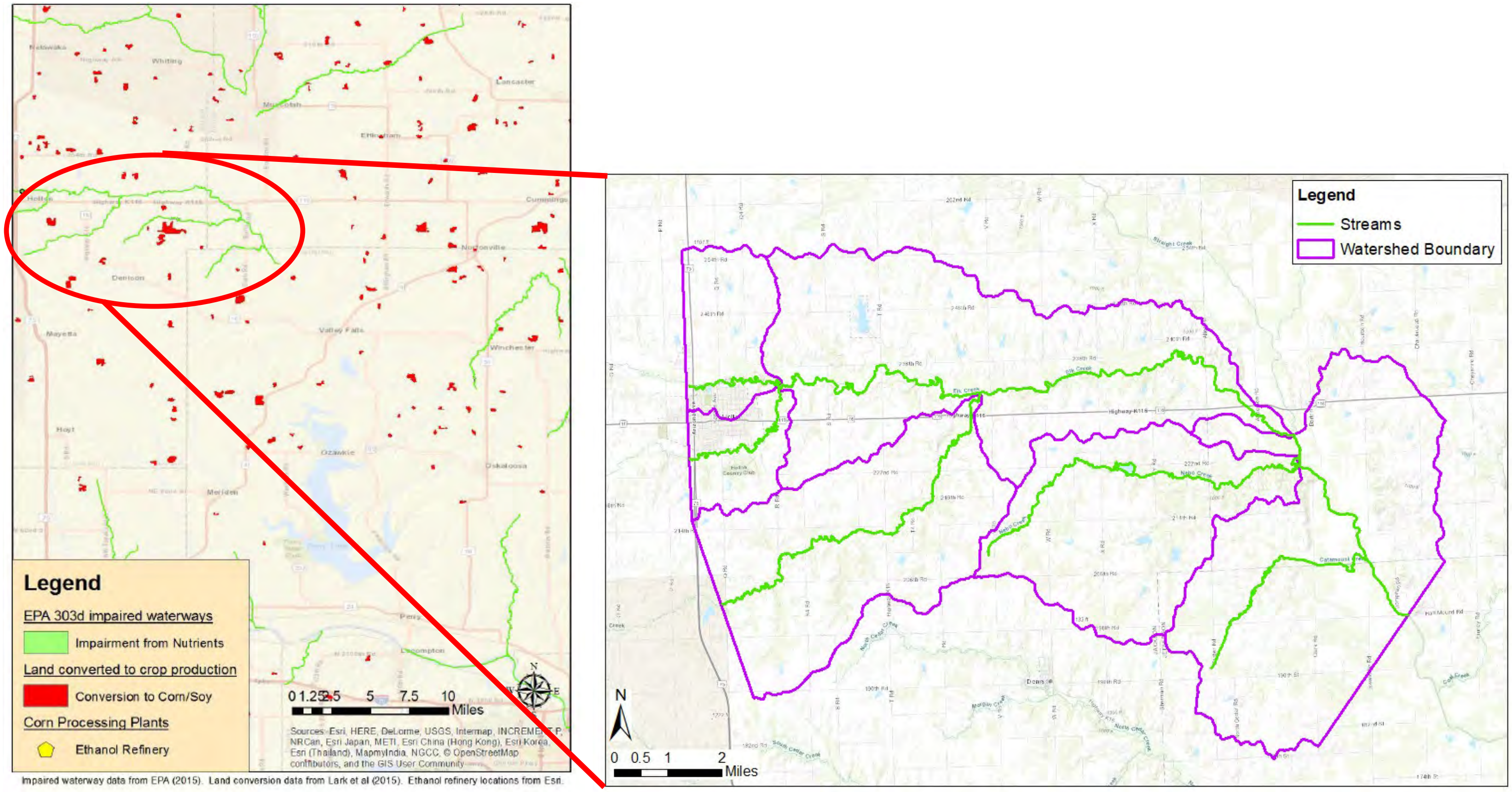


Figure 17. Google Earth Images for the period 1991 through 2018 for fields adjacent to a 303(d) impaired water body identified in Figure 5-6 from the Lark Declaration as having been converted from grassland to corn or soy after 2008