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

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Growth Energy

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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.

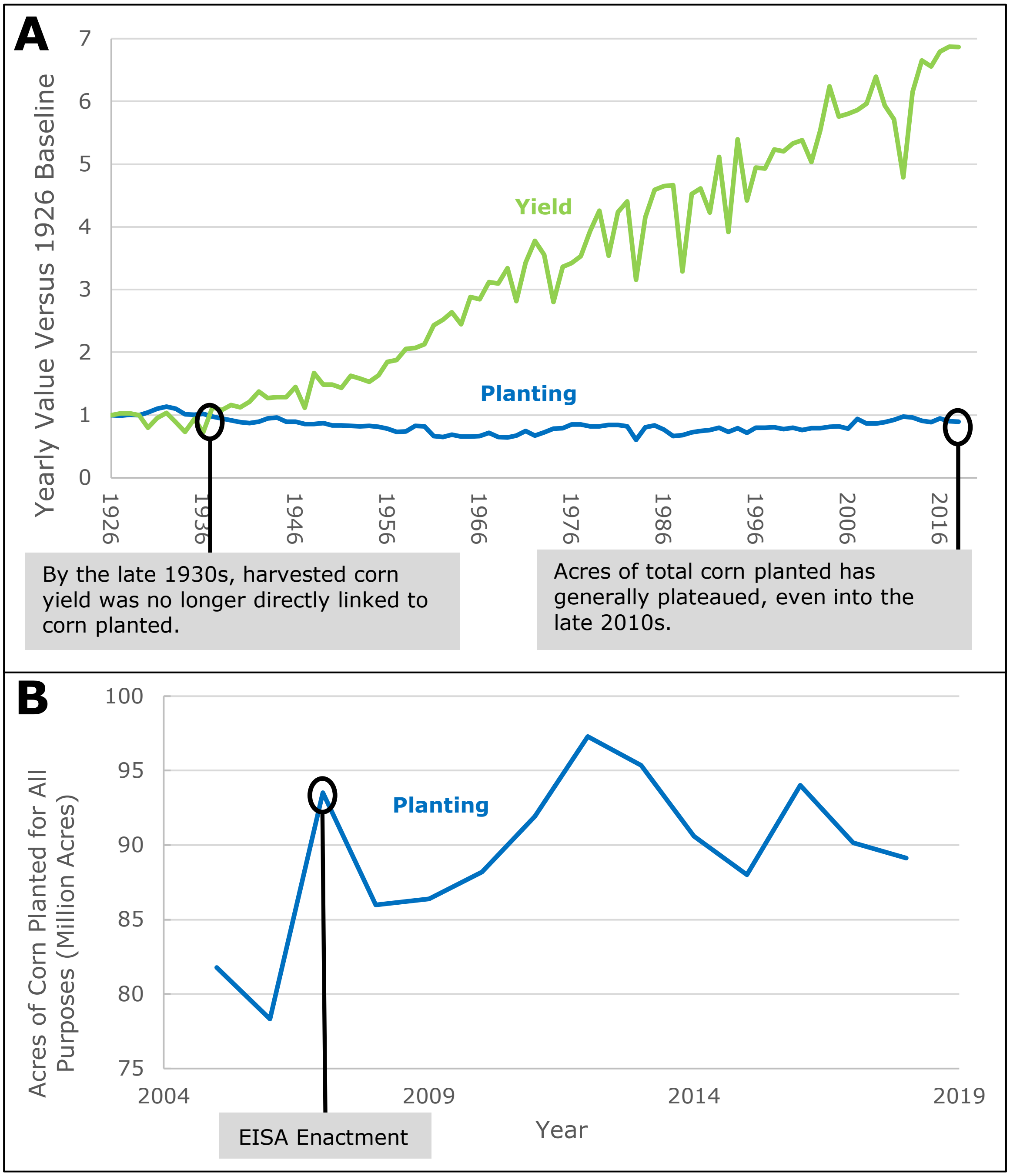
- **The RFS Reset is well-timed to coincide with ongoing improvements in agricultural practices—**Nearly all published investigations Ramboll has reviewed that focus on the potential impact of increased corn growth for biofuel production have focused on past practices with only passing mention of future 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.

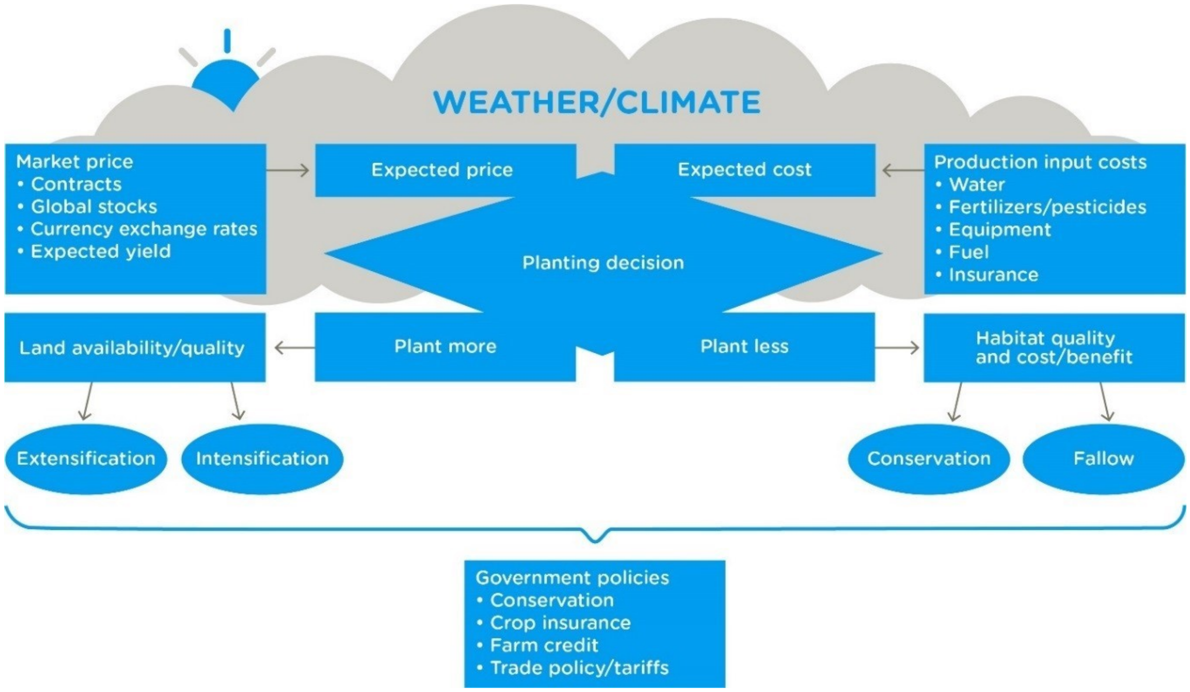


Source: USDA Crop Production Historical Track Records, 2019

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).

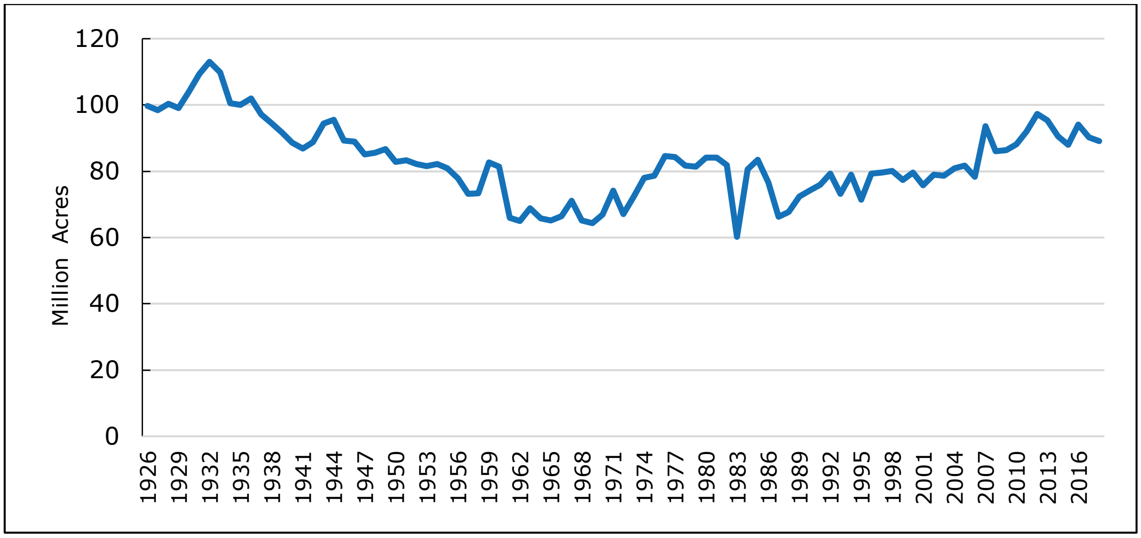
American Petroleum Institute (API) reported approximately 10.8 million gallons of oil were spilled into U.S. Navigable Waters from 1997-2006 with the amount spilled per year varying from 466,000 (2005) to 2.7 million (2004). This figure clearly does not include the Exxon Valdez spill in Alaska in 1989 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).

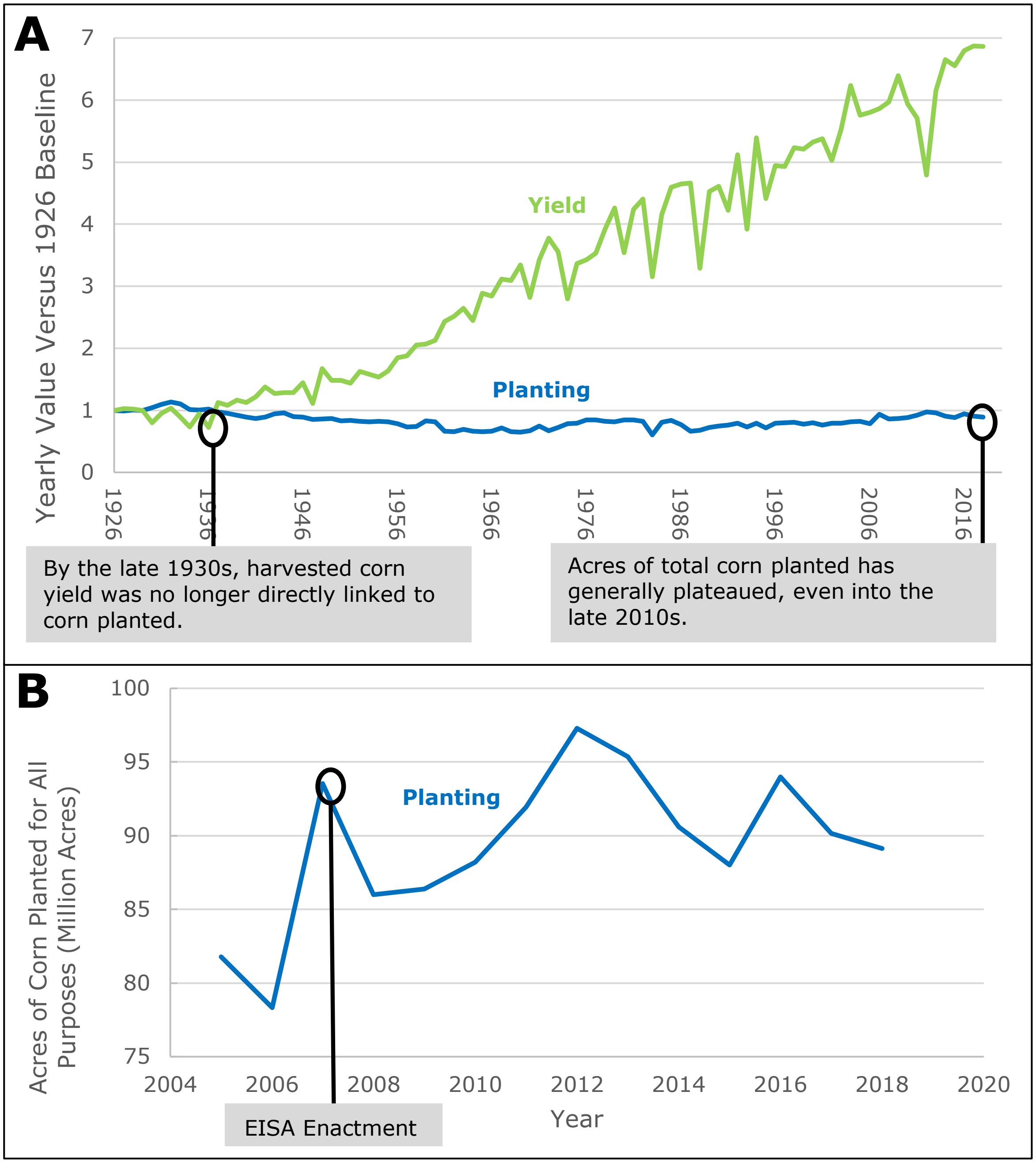


(Source: USDA 2019)

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).

[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).

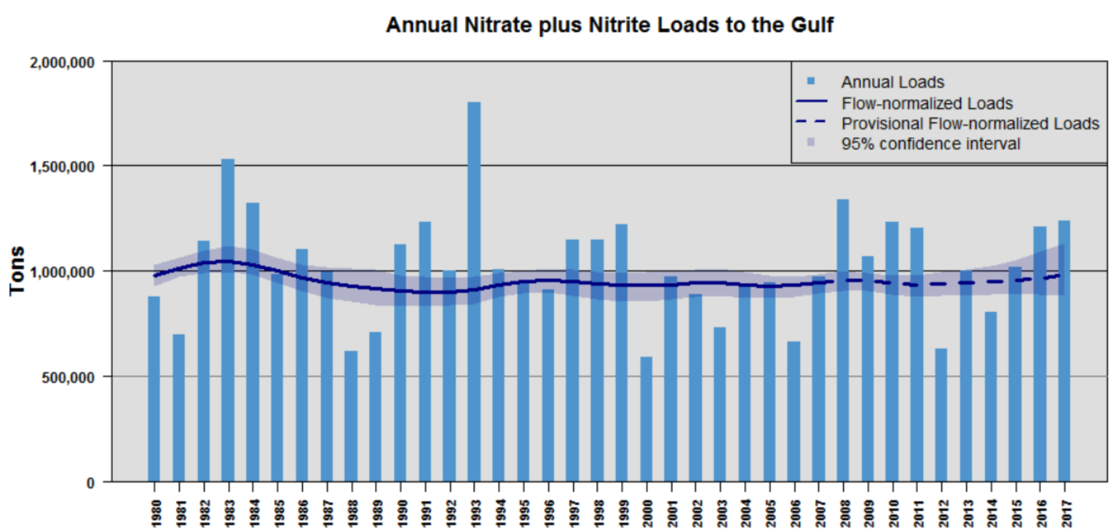
There may be no dispute that excess nutrient loading from the key watersheds that discharge into western Lake Erie and the northern Gulf of Mexico contribute to eutrophication and hypoxia; however, the watersheds are composed of a complex mix of urban and rural uses and wastewater discharges. Agricultural 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.)

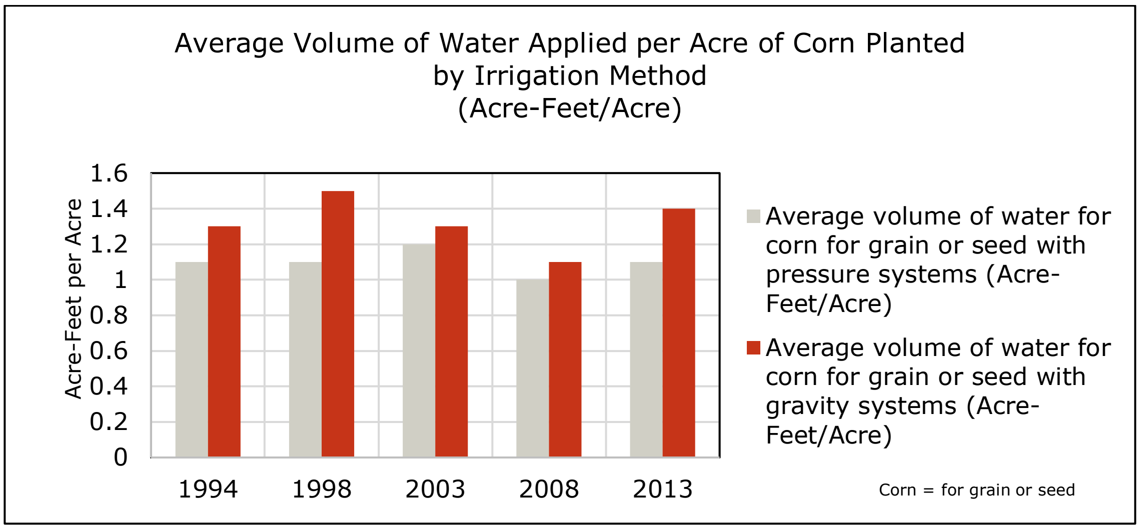
The fact that agricultural practices in general can result in nutrient runoff is acknowledged, although modern efficiencies and conservation methods have improved over time. Modern practices apply technology for increased efficiency and harness continuously improving data analysis to develop and implement 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.

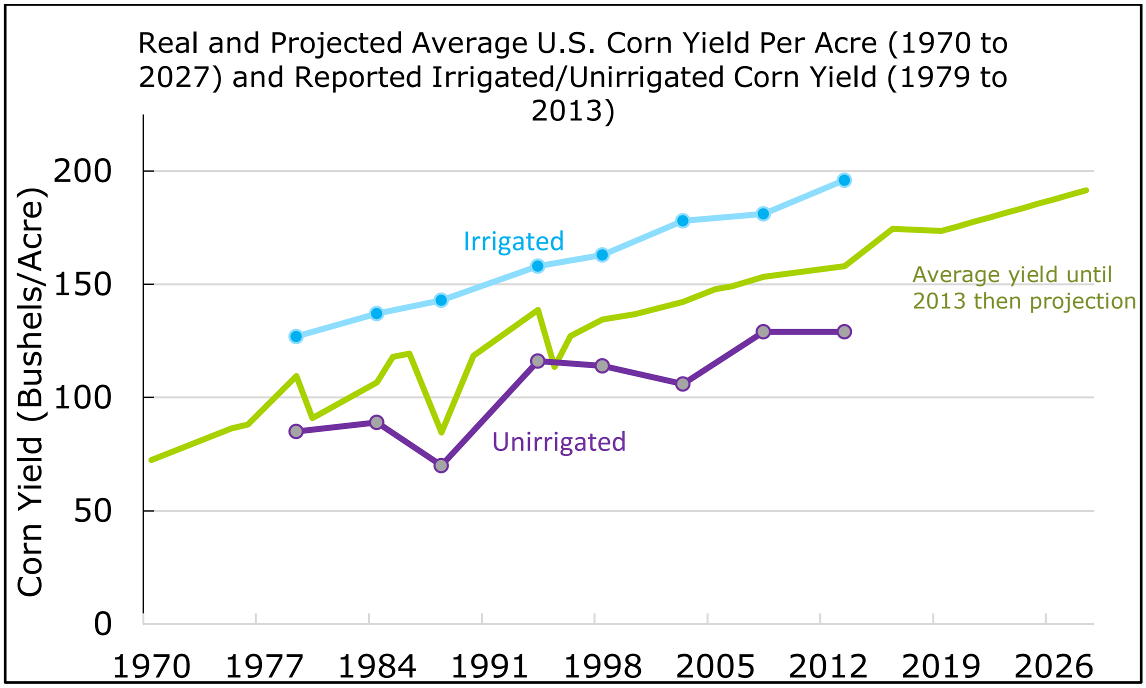
Figure 9: Volume of Water Applied to Irrigated Corn Crops Since 1994, by Irrigation Method.



(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.

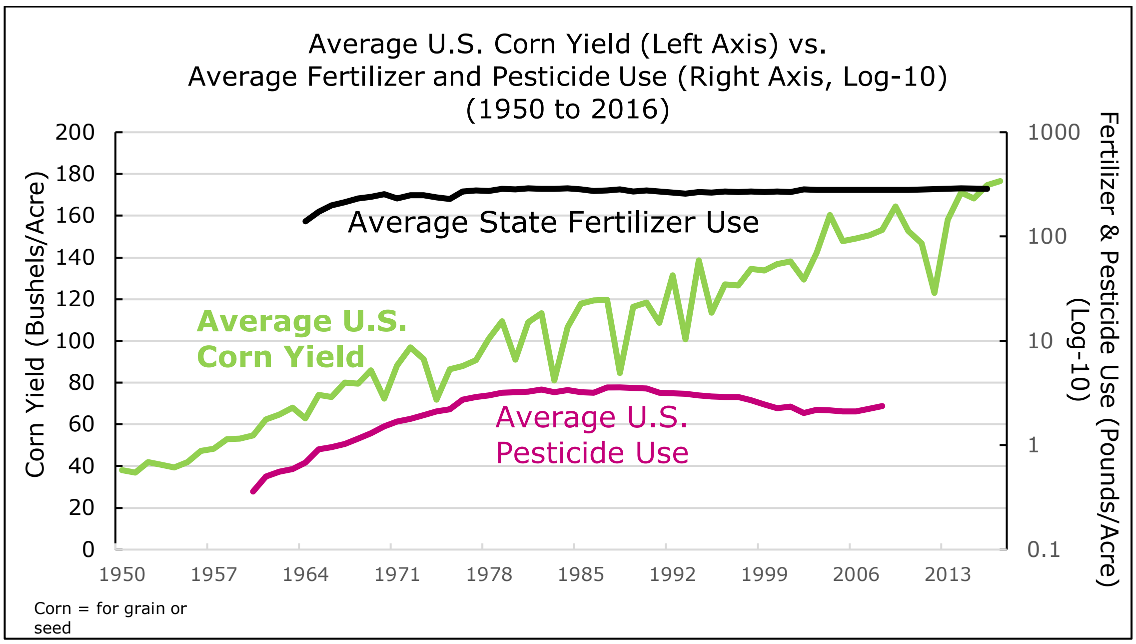
Liu, et al (2018) states that best management practices for reducing agricultural non-point source pollution are widely available even with the challenges related to the large number of agricultural producers and the spatially variable and temporally dynamic nature of the nutrient loading cycles. Greater 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.

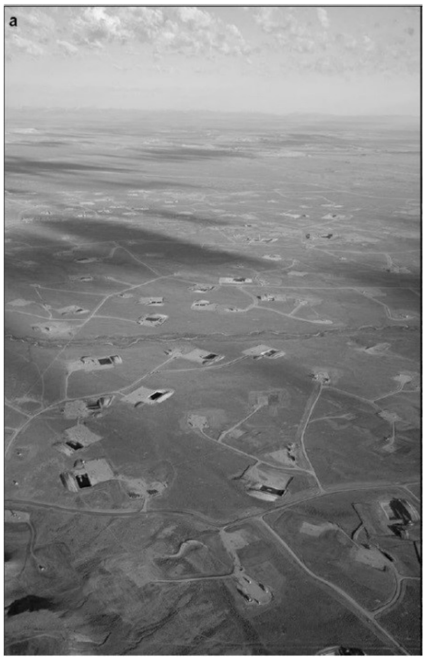
EPA (2018a) acknowledges that it fails to address environmental impacts associated with gasoline production. Spills of petroleum, gasoline, and a wide range of other fluids used in the exploration, production, and refining processes as well as land use change to support those activities all have adverse 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.

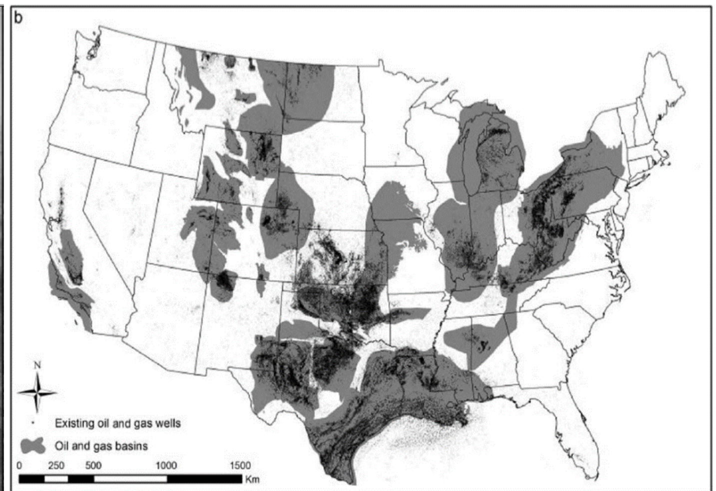
6.1 Impacts of Gasoline Production Associated with Land Use Change

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.

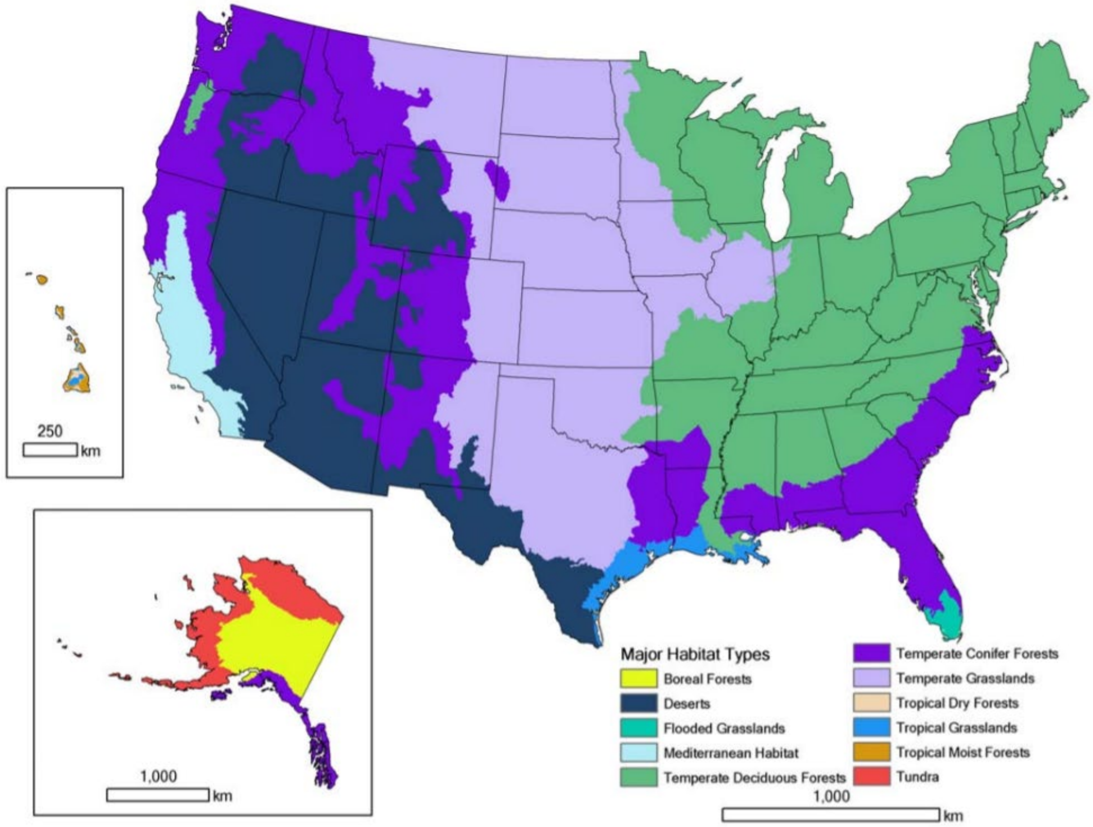


Figure reproduced from McDonald et al. 2009

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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