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Biofuels and the Environment:

Third Triennial Report to Congress

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Biofuels and the Environment
Third Triennial Report to Congress

U.S. Environmental Protection Agency
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Acronyms and Abbreviations

ABS	Anti-backsliding study
ACEP	Agricultural Conservation Easement Program
ACP	Acidification potential
ADAGE	Applied Dynamic Analysis of the Global Economy
AEO	Annual Energy Outlook
AEPE	Agricultural Energy Partial Equilibrium
AFDC	Alternative Fuels Data Center
AI	Active ingredient
AKI	Anti-knock index
Al	Aluminum
AMPA	Aminomethylphosphonic acid
ANL	Argonne National Laboratory
ANPRM	Advance notice of proposed rulemaking
APEX	Agricultural policy extender
APHIS	Animal and Plant Health Inspection Service, part of the U.S. Department of Agriculture (USDA)
ARMS	Agricultural Resource Management Survey
ARPA-E	Advanced Research Projects Agency-Energy
ARS	Agricultural Research Service, part of the U.S. Department of Agriculture (USDA)
ATTAINS	Assessment, Total Maximum Daily Load Tracking and Implementation System
B5, B10, B20, B100	Types of biodiesels (blended with 5%, 10%, 20% and 100% of biodiesel relative to diesel)
BAP	Biorefinery, Renewable Chemical, and Biobased Product Manufacturing Program
BAT	Base acres treated
BAU	Business as usual
BBD	Biomass-based diesel
BBS	Breeding Bird Survey
BC	Black carbon
BEA	Bureau of Economic Analysis, part of the U.S. Department of Commerce
BEAD	Biological and Economic Analysis Division, part of the U.S. Environmental Protection Agency (EPA)
BEIOM	Bioeconomy Economic Input Output Model

Bgal	Billion gallons
BIP	Biofuel Infrastructure Partnership
BMP	Best management practice
BOB	Blendstock for oxygenate blending
BRS	Biotechnology Regulatory Services
BSM	Biomass Scenario Model
BT16	2016 Billion Ton Study
BTC	Biodiesel Tax Credit
BTU	British thermal unit
BWF	Blue water footprint
CAA	Clean Air Act
CAAA	Clean Air Act Amendments
CAC	Central America and the Caribbean
Ca	Calcium
$CaCO_3$	Calcium carbonate
CAFE	Corporate average fuel economy
CAG	Crop acres grown
CARB	California Air Resources Board
CARD	Center of Agricultural and Rural Development
$CaSO_4$	Calcium sulfate
CBI	Caribbean Basin Initiative
CCS	Carbon capture and storage
CDL	Cropland data layer
CDPF	Catalyzed diesel particulate filter
CDS	Condensed distillers' solubles
CEAP	Conservation Effects Assessment Project
CEC	California Energy Commission
CFR	Code of Federal Regulations
CGE	Computable general equilibrium
CGF	Corn gluten feed
CGM	Corn gluten meal
CH_4	Methane
CI	Confidence interval
CL	Confidence limit

CLCA	Consequential lifecycle analysis
CMAQ	Community Multiscale Air Quality
CNG	Compressed natural gas
CO	Carbon monoxide
CO_2	Carbon dioxide
CONUS	Contiguous United States
COP23	23rd Conference of the Parties
CPI	Consumer Price Index
CRC	Coordinating Research Council
CRD	Cropland Reporting Districts
CRLF	California red legged frog
CRP	Conservation Reserve Program
CS	Corn-soybean
CSP	Conservation Stewardship Program
D3	D-code for generating RINs with cellulosic biofuel
D4	D-code for generating RINs with biomass-based diesel
D5	D-code for generating RINs with advanced biofuel
D6	D-code for generating RINs with renewable biofuel
D7	D-code for generating RINs with cellulosic biofuel or biomass based diesel
DBP	Disinfection by-products
DCO	Distillers' corn oil
DDG	Distillers' dried grains
DDGS	Distillers' dried grains with solubles
DEER	Diesel Engine-Efficiency and Emissions Research
DEM	Digital Elevation Model
DG	Distillers' grains
DisN	Dissolved nitrogen
DisP	Dissolved phosphorus
DOC	Dissolved organic carbon
DOE	U.S. Department of Energy
DOI	U.S. Department of the Interior
DOM	Dissolved organic matter
DON	Dissolved organic nitrogen
DPPR	Dakota Prairie Pothole Region

DRIA	Draft Regulatory Impact Analysis
DWG	Distillers' wet grains
DWGS	Distillers' wet grains with solubles
E0, E10, E15, E85	Types of gasoline (blended with 0%, 10%, 15% and 85% of ethanol relative to gasoline)
E&P	Extraction and production
EAEP	Enzyme-assisted aqueous extraction process
EAS	Exhaust aftertreatment systems
EC	Elemental carbon
EC50	Median effective aqueous concentrations
EDDMapS	Early Detection and Distribution Mapping System
EEF	Enhanced efficiency fertilizer
EEIO	Environmentally-extended input-output
EIA	U.S. Energy Information Administration
EISA	Energy Independence and Security Act
EJ	Exajoule
EMTS	EPA Moderated Transaction System
EOR	Enhanced oil recovery
EPA	U.S. Environmental Protection Agency
EPAct	Energy Policy Act of 1992
EPIC	Environmental Policy Integrated Climate
EQIP	Environmental Quality Incentives Program
ERD	External review draft
ERS	Economic Research Service, part of the U.S. Department of Agriculture (USDA)
ESA	Endangered Species Act
ET	Evapotranspiration
ETBE	Ethyl-tertiary-butyl-ether
EU	European Union
FAA	Federal Aviation Administration
FAO	Food and Agriculture Organization of the United Nations
FAPRI	Food and Agricultural Policy Research Institute
FAS	Foreign Agricultural Service, part of the U.S. Department of Agriculture (USDA)
FASOM	Forest and Agricultural Sector Model
FCA	Food, Conservation and Energy Act of 2008

FD	Final demand
FDA	U.S. Food and Drug Administration
Fe	Iron
FFV	Flex-fuel vehicles
FIA	Forest Inventory and Analysis
FIFRA	Federal Insecticide, Fungicide, and Rodenticide Act
FOB	Freight on board
FOD	First order draft
FOGs	Fats, oils, and greases
FQAPA	Food Quality Protection Act
FR	Federal Register
FRIS	Farm and Ranch Irrigation Survey
FSA	Farm Service Agency, part of the U.S. Department of Agriculture (USDA)
GAIN	Global Agricultural Information Network
GCAM	Global Change Analysis Model
GCAU	Grain consuming animal units
GCRP	Grassland subprogram of the Conservation Reserve Program
GDI	Gasoline direct injection
GDP	Gross domestic product
GE	General equilibrium (Chapter 4 and 6)
GE	Genetically engineered (Chapter 3)
GHG	Greenhouse gas
GIS	Geographic information system
GLOBIOM	Global Partial Equilibrium Model
GRCAU	Grain and roughage consuming animal units
GREET	Greenhouse Gases, Regulated Emissions, and Energy Use in Technologies
GRSW	Grassed waterways
GTAP	Global Trade Analysis Project
GTW	Grease trap waste
GWP	Global warming potential
H_2O	Water
HAA	Haloacetic acids
HAB	Harmful algal blooms
HACCP	Hazard analysis and critical control points

HBIIP	Higher Blends Infrastructure Incentive Program
HERO	Health and Environmental Research Online database
HP	Horsepower
HPA	High Plains Aquifer
HPCAU	High-protein consuming animal units
HS	Harmonized System
HSMU	Homogenous spatial mapping units
HT	Herbicide-tolerant
HTP	Human toxicity potential
HUC	Hydrologic Unit Code
IAM	Integrated assessment model
ICCS	Industrial Carbon Capture and Storage
ICTSD	International Centre for Trade and Sustainable Development
ILUC	Indirect land use change
IMF	International Monetary Fund
IPCC	Intergovernmental Panel on Climate Change
IPM	Integrated pest management
IPPC	International Plant Protection Convention
IR	Insect resistant
JCA	Jobs Creation Act
K	Potassium
K_2O	Potassium oxide
KOH	Potassium hydroxide
LAMPS	Land-use and Agricultural Management Practice web-Service
LANID	Landsat-based Irrigation Dataset
LC	Land cover
LCA	Life cycle assessment
LCC	Land capability class
LCFS	Low Carbon Fuel Standard
LCI	Lifecycle inventory
LCIA	Life cycle impact assessment
LCLM	Land-cover-land-management
LEMA	Local enhancement management area
LFG	Landfill gas

LM	Land management
LMOP	Landfill Methane Outreach Program
LMRB	Lower Mississippi River Basin
LNG	Liquified natural gas
LTAP	Long term agricultural projections
LUC	Land use change
LULUC	Land use and land use change
Mac	Million acres
MCE	Model Comparison Exercise
MCL	Maximum contaminant level
Mg	Magnesium
MGY	Million gallons per year
MJ	Megajoules
MLU	Major land use
MMI	Multi-metric index
Mn	Manganese
MORB	Missouri River Basin
MOVES	MOtor Vehicle Emission Simulator
MSAT	Mobile source air toxics
MSQA	Midwest Stream Quality Assessment
MSW	Municipal solid waste
MT	Metric tonnes
MTBE	Methyl tert-butyl ether
MU	University of Missouri
MY	Market year
N	Nitrogen
N_2O	Nitrous oxide
Na	Sodium
NAAQS	National Ambient Air Quality Standards
NAICS	North American Industry Classification System
NAIP	National Agricultural Imagery Program
NaOH	Sodium hydroxide
NARS	National Aquatic Resource Surveys
NASA	U.S. National Aeronautics and Space Administration

NASEM	National Academies of Science, Engineering, and Medicine
NASS	National Agricultural Statistics Service, part of the U.S. Department of Agriculture (USDA)
NAWCA	North American Wetlands Conservation Act
NAWMP	North American Waterfowl Management Plan
NAWQA	National water-quality assessment
NCCA	National Coastal Condition Assessment
NCDC	National Climatic Data Center
NCEI	National Centers for Environmental Information
NEI	National Emissions Inventory
NH_3	Ammonia
NLA	National Lakes Assessment
NLCD	National Land Cover Dataset
NMFS	National Marine Fisheries Service
NMHC	Non-methane hydrocarbon
NMOG	Non-methane organic gases
NOAA	U.S. National Oceanic and Atmospheric Administration
NO_X, NO, NO_2, NO_3	Nitrogen oxides
NPL	Northern Plains
NRCS	National Resources Conservation Service, part of the U.S. Department of Agriculture (USDA)
NREL	National Renewable Energy Laboratory
NRI	Natural Resources Inventory
NRSA	National Rivers and Streams Assessment
NSE	Nash-Sutcliffe efficiency
NUE	Nitrogen use efficiency
NWALT	National Wall-to-Wall Anthropogenic Land Use Trends
NWS	National Weather Service
O_2	Oxygen
O_3	Ozone
OCSPP	Office of Chemical Safety and Pollution Prevention, part of the U.S. Environmental Protection Agency (EPA)
ODP	Ozone depletion potential
OPEC	Organization of the Petroleum Exporting Countries

OPIS	Oil Price Information Service
OPP	Office of Pesticide Programs, part of the U.S. Environmental Protection Agency (EPA)
ORAU	Oak Ridge Associated Universities
ORB	Ohio River Basin
OrgN	Organic nitrogen
OrgP	Organic phosphorus
OTAQ	Office of Transportation and Air Quality, part of the U.S. Environmental Protection Agency (EPA)
P	Phosphorus
P2O5	Phosphorus pentoxide
P&E	Palustrine and estuarine
PADD	Petroleum Administration for Defense District
PAN	Peroxyacetyl nitrate
PBIAS	Percent bias
PCT	Percent crop treated
PE	Partial equilibrium
PEP	PM exposure potential
PFI	Port fuel injection
pH	Expression of the hydrogen ion in water
PHEV	Plug-in hybrid vehicles
PM	Particulate matter
$PM_{2.5}$, PM_{10}	Particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 or 10 µm
PMI	Particulate matter index
PMP	Particle measurement programme
PN	Particle number
POC	Precursor organic compounds
POLYSYS	Policy Analysis System Model
POTW	Publicly owned treatment works
PPQ	Plant protection and quarantine
PPR	Prairie Pothole Region
PRELIM	Petroleum Refinery Life Cycle Inventory Model
PRIA	Pesticide Registration Improvement Extension Act
PWS	Public water systems
QBTU	Quadrillion British thermal units

RB	Riparian buffer
RBSB	Riparian buffer/saturated buffers
RCAU	Rough consuming animal units
RFA	Renewable Fuels Association
RFG	Reformulated gasoline
RFS	Renewable Fuel Standard
RIA	Regulatory Impact Analysis
RIN	Renewable Identification Number
RMP	Risk mitigation plan
RNG	Renewable natural gas
ROA	Real options analysis
RRB	Republican River Basin
RRR	Registration, reporting, and recordkeeping
RSPO	Roundtable on Sustainable Palm Oil
RtC1	First Triennial Report to Congress
RtC2	Second Triennial Report to Congress
RtC3	Third Triennial Report to Congress
RVO	Renewable volume obligations
RVP	Reid vapor pressure
SB	Saturated buffer
SCOPE	Scientific Committee on Problems of the Environment
SCR	Selective catalytic reduction
SDWA	Safe Drinking Water Act
SEDS	State Energy Data System
SFP	Smog formation potential
SHP	Southern High Plains
SO_2	Sulfur dioxide
SOC	Soil organic carbon
SOM	Soil organic matter
SPA	Structural path analysis
SPN	Solid particle number
SRE	Small refinery exemption
SRWC	Short-rotation woody crop
SS	Suspended sediments

STATSGO	State Soil Geographic Database
STEO	Short Term Energy Outlook
STIR	Soil Tillage Intensity Ratings
SWAT	Soil and Water Assessment Tool
T&E	Threatened and endangered
TAME	Tertiary-amyl-methyl-ether
TBA	Tertiary-butyl-alcohol
THC	Total hydrocarbons
THM	Trihalomethanes
TN	Total nitrogen
TOC	Total organic carbon
TP	Total phosphorus
TPL	Temperate Plains
TRACI	Tool for Reduction and Assessment of Chemicals and Other Environmental Impacts
TRF	Total renewable fuel
TSS	Total suspended sediment
TVA	Tennessee Valley Authority
UCO	Used cooking oil
UL	Underwriters Laboratories
ULSD	Ultra-low sulfur diesel
UMRB	Upper Mississippi River Basin
UMW	Upper Midwest
UN	United Nations
UPGM	Unified Plant Growth Model
USDA	U.S. Department of Agriculture
USEEIO	United States Environmentally-Extended Input-Output
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey
USMCA	U.S., Mexico, Canada
UST	Underground storage tank
VEETC	Volumetric Ethanol Excise Tax Credit
VOC	Volatile organic compounds
VRT	Variable rate technology
WATER	Water Analysis Tool for Energy Resources

WF	Water footprint
WRA	Weed risk assessment
WRE	Wetland Reserve Easement
WSA	Wadeable Streams Assessment
WTI	West Texas Intermediate
WTP	Willingness-to-pay
WTW	Well-to-wheel
ZLD	Zero liquid discharge

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Unit Abbreviations and Conversions

Volume

1 gallon (gal) (U.S. gallon)	=	3.8 liters (L)
1 bushel (bu)	=	35 liters (L)
1 barrel (bbl)	=	42 gallons (gal)
1 acre-foot (acre-ft)	=	325,851 gallons (gal)

Area

1 acre (ac)	=	0.4 hectares (ha)
1 hectare (ha)	=	2.5 acres (ac)
1 square kilometer (km^2)	=	247 acres (ac)

Weight

1 pound (lb)	=	0.45 kilograms (kg)
1 ton (U.S. ton)	=	907 kilograms (kg)
1 gram (g)	=	0.035 ounces (oz)
1 kilogram (kg)	=	2.2 pounds (lb)
1 metric ton or tonne (MT)	=	2,200 pounds (lb)
1 teragram (Tg)	=	1,102,000 tons (t)

Length

1 mile (mi)	=	1.6 kilometers (km)
1 inch (in)	=	2.5 centimeters (cm)
1 kilometer (km)	=	0.6 miles (mi)

SI Prefixes

peta = 10^{15}	centi = 10^{-2}
tera = 10^{12}	milli = 10^{-3}
giga = 10^{9}	micro = 10^{-6}
mega = 10^{6}	nano = 10^{-9}
kilo = 10^{3}	
hecto = 10^{2}	

Executive Summary

This is the Third Triennial Report to Congress on Biofuels (RtC3) as required under Section 204 of the Energy Independence and Security Act of 2007 (EISA). The purpose of the report is to examine the effects of the Renewable Fuel Standard (RFS) Program on the environment, including the impacts to date and likely future impacts to the nation's air, land, and water resources. The statute requires a focus on environmental and resource conservation issues, including effects on air quality, soil quality and conservation, water quality and availability, terrestrial ecosystems, aquatic ecosystems, and wetlands, and consideration of invasive or noxious species. This report emphasizes domestic effects, but also examines effects overseas. The RtC3 considers all 17 types of biofuels produced in or imported to the U.S. from 2005-2020 and focuses on the four biofuels that dominated U.S. production and consumption over this period: (1) ethanol from U.S. corn, (2) biodiesel from U.S. soybean, (3) biodiesel from U.S. fats, oils, and greases (FOGs), and (4) imported ethanol from Brazilian sugarcane. Although these four biofuels are the focus of the RtC3, other biofuels (cellulosic biofuels, algae, palm oil, and others) are also discussed where appropriate. While EPA acknowledges the importance of greenhouse gases (GHGs) in assessing the environmental impacts of biofuels and the RFS, consistent with earlier reports, the RtC3 does not assess them here; EPA evaluates GHGs while administering the RFS Program (Sections 201 and 202 of EISA[1]).

In the First and Second Triennial Reports to Congress on Biofuels (RtC1 and RtC2, respectively), the Agency could not separate the effects of the RFS Program from the effects of other factors (e.g., market or other policy effects). Many studies assessed the impacts from biofuels on the environment, but very few separated the effects of the RFS Program from other factors that also affect biofuel production and consumption in the United States. As attribution was identified as a major knowledge gap in previous reports, this report includes a new emphasis on attribution, referred to in this report as an "attribution analysis."

This report examines the many factors that simultaneously influenced the production and use of domestic corn ethanol in the United States to assess attribution. These factors include the need for fuel oxygenates in gasoline during the phaseout of methyl-tert-butyl-ether (MTBE) from 2003–2006, the Volumetric Ethanol Excise Tax Credit (VEETC) from 2004–2010, high oil prices from 2005–2015, and dozens of individual state biofuel programs and MTBE bans over this period. The RFS Program has

[1] Energy Independence and Security Act of 2007, Pub. L. No. 110-140, § 202, 121 Stat. 1492, 1521-28 (2007) (codified as amended at 42. U.S.C. § 7545(o)). Detailed assessment of the GHG balance of corn ethanol and other biofuels are not in scope of this report series. See Chapter 2 (Box 2.2) for an overview and see Federal Registry (FR) FRL–9307–01–OAR (https://www.epa.gov/renewable-fuel-standard-program/workshop-biofuel-greenhouse-gas-modeling) and EPA's 2023 lifecycle analysis Model Comparison Technical Document (https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1017P9B.pdf).

changed as well over this period, from the first version (RFS1) created under the Energy Policy Act of 2005, to a more robust version (RFS2) created under EISA in 2007. Because of these complexities, assessing the effect from the RFS Program as required under EISA, as opposed to the biofuels industry more generally, is challenging. Furthermore, the policy and the market are dynamic, so that the effect of the policy changes over time. Despite these challenges, by assembling multiple lines of evidence from empirical records and simulation modeling from the peer-reviewed literature, this report finds that corn ethanol production and consumption in the United States attributable to the RFS Program varies through time and may include no attributable effect. This finding is expressed throughout the report as a range in billions of gallons per year and includes zero in the range. This report estimates that from 2006 to 2011 the RFS Program—in isolation—accounted for 0–1.0 billion gallons per year of ethanol, mostly by establishing market certainty and encouraging capital investment from the RFS2, and to a lesser extent by stabilizing demand during the Great Recession of 2008–2009. From 2012 to 2018, the RFS Program accounted for an estimated 0–2.1 billion gallons per year, a wider range than from the previous period. This suggests the RFS Program is responsible for 0–9% of cumulative corn ethanol production and consumption in the United States over the historical period assessed (2005–2018).

Many uncertainties are associated with this estimate of the volume of ethanol attributable to the RFS Program. The growth of corn ethanol production in the United States over the years coincided with the MTBE phaseout by 2006, expiration of VEETC at the end of 2010, and lower oil prices after 2015. Disentangling the effect of the RFS Program, as required under EISA Section 204, is difficult given the many cooccurring factors that affect biofuels in the United States. As a mandate, the RFS Program *could have* driven most of the increase in ethanol production and consumption in the United States if it were the only factor affecting ethanol. However, as events played out, non-RFS factors that are known to also influence the market were favorable and appear to explain much of the increase in ethanol production and consumption in the United States. There are many unquantified factors not included in the attribution analysis contained in this report, including the effect of the existence of the RFS Program in influencing state biofuel programs to be enacted and the costs or willingness of refiners to switch back to producing finished gasoline if ethanol were no longer economical, to name a few. Notwithstanding various uncertainties, these ranges are estimated based on currently available information for the historical effect of the RFS Program on corn ethanol production and consumption in the United States.

For biodiesel and renewable diesel, the attributional effect of the RFS Program is estimated to be different. Using similar lines of evidence as for corn ethanol, where available, this report concludes that the RFS Program has driven a significant portion of the use of these biofuels from 2010–2020. However, there is insufficient information available at present to confidently quantify the attributional effect annually of the RFS Program for these years. This is mostly due to a lack of data and peer-reviewed studies that focus on biodiesel that control for key factors important in the biodiesel market such as the Biodiesel Tax Credit (BTC) and state incentive programs. Together, the RFS Program and the BTC are likely responsible for roughly 70–100% of soybean biodiesel and renewable diesel production and consumption in the United States.

Using the estimated range in the volume of corn ethanol attributable to the RFS Program, this report estimates the RFS Program's effect on corn ethanol production and consumption resulted in 0–1.9 million acres of cropland expansion from direct and indirect effects domestically between 2005 and 2016, and 0–3.5 million acres of corn expansion, with many years of no effect. The 1.9 million acres of cropland corresponds with less than 1% of all cropland in 2017, but also represents approximately 19% of the estimated cropland *expansion* between 2008 and 2016. The maximum of 3.5 million acres of corn corresponds with less than 5% of all planted corn in 2017 but represents an almost 35% *increase* in corn acreage between 2008 and 2016. Thus, though these upper range estimates are still small relative to the total acreage of cropland or corn, potential effects from the RFS Program may be locally significant where any land use changes may have occurred. Cropland expansion often leads to increases in soil erosion, pesticide and fertilizer applications, and losses of seminatural habitat. These upper range estimates of the effects on total cropland due to the RFS Program would have had modest negative impacts on many of the environmental effects reviewed in this report, as concluded but not quantified in the RtC1 and RtC2. However, specific areas where environmental effects may have occurred cannot be quantified with confidence due to the vast quantity of potential cropland in the United States and the multitude of factors that contribute to an individual farmer's decision whether to bring additional land into crop production. The ranges analyzed represent an updated estimate based on the currently available science and literature and may be revised as further research is conducted.

Despite the finding of potentially modest annual effects of the RFS Program nationally for the environmental impacts assessed, these may have important cumulative impacts on the environment. For example, by 2004–the year before enactment of the Energy Policy Act–over half of the historical wetlands in the lower 48 states had already been lost (>100 million acres lost), with several Midwestern states losing more than 80% of their historical wetlands. Additional losses of up to 275,000 acres of wetlands are estimated to have occurred between 2008 and 2016 from all causes, only a portion of which are attributable to the RFS Program. This acreage is small compared with historical losses but could have cumulative environmental effects or landscape level effects in some areas. Similarly, according to national surveys conducted by the EPA, 67% of the wadeable streams in the United States were already in poor or fair biological condition as of 2004. Thus, even though the RFS Program may not result in *new exceedances* of numerical nutrient thresholds, it does represent additional strain on already strained ecosystems. Moreover, the effects of the RFS Program likely fall disproportionally in certain areas of the

United States, such as in rural areas with greater amounts of grassland habitat lost to corn or soybeans. Some of these areas may contain locally endemic species and other important local environmental resources, which may appear underrepresented in a large national-scale assessment. Thus, even modest national effects do not preclude potentially larger effects at the local level. At this time, however, EPA cannot identify with any specificity and certainty which parcels of land at the local level may have been affected by the RFS Program.

International effects associated with imported biofuels, and market mediated effects on crop and biofuel production in other countries, are even more uncertain than national effects. These effects are highly uncertain due to the large range of estimates among studies, poorly evaluated differences among models, and a lack of adequate representation in these models of biofuel policies in countries other than the United States. However, effects from imported biofuel are likely modest given the relatively small quantity of imports relative to domestic biofuel production since the RFS Program went into effect. It does not necessarily follow that overall international effects of the RFS Program have been small, as research has shown the indirect effects of increased biofuel production on feedstock commodity trade flows could be substantial.

Domestically, some of the agricultural practices that can mitigate environmental impacts are becoming widely adopted (e.g., conservation tillage), while others are not (e.g., cover crops). While some of these adoptions may explain regional improvements in some environmental conditions, they do not yet appear to be large enough to improve many of the environmental effects reviewed in this report. Greater adoption of these conservation practices could help offset potential effects from the RFS Program or broader effects from agriculture.

This report reinforces the broad conclusions from the RtC1 and RtC2 on biofuels in general and further evaluates attribution of those effects to the RFS Program more specifically. Although the overall environmental effects attributable to the RFS Program to date are likely modest but negative, biofuels continue to have the potential for both positive and negative environmental effects, depending on the many factors discussed in this report.

For the future period, EPA included the estimated effects of the RFS Program for 2023–2025 as a part of the Final Set Rule 88 Federal Register 44468 (July 12, 2023). EPA projected an increase of approximately 3.9 billion ethanol equivalent gallons of renewable fuel use in the United States by 2025 due to the RFS Program. This overall increase is estimated to be primarily from compressed/liquified natural gas (CNG/LNG) derived from biogas (+932 million gallons), biodiesel and renewable diesel from soybean oil (+1,484 million gallons) and canola oil (+614 million gallons), and ethanol from corn (+787 million gallons). EPA expects smaller effects from the RFS Program on other biofuels (e.g., +110 million gallons of renewable diesel from FOGs). For the crop-based biofuels with potential effects on cropland, EPA estimated the RFS Program could potentially lead to an increase of as much as 2.65 million acres of cropland by 2025. These estimated increases in the future are on top of historical effects. As with the historical land use changes, EPA cannot at present identify with any specificity and certainty which parcels of land at the local level may be affected by the RFS Program. Several factors contribute to uncertainty in these estimates of the likely future, including ongoing recovery from the global COVID-19 pandemic, uncertainty in the penetration of E15 in the marketplace, competition with other technologies such as electric vehicles, and continued but slow growth of cellulosic ethanol production from agricultural or marginal lands. As policy and market conditions change, so may the factors to consider and the estimate of the likely future effects of the RFS Program. Further details can be found in the associated docket (EPA-HQ-OAR-2021-0427).

Detailed recommendations are discussed in this report and primarily include research recommendations to fill key knowledge gaps to support policy decision making. These include, but are not limited to, research to improve estimates of the attributional effect from the RFS Program on all types of biofuels that include realistic industry and economic detail, methods to link these attributional effects to specific land areas domestically and internationally, improved remote sensing and local data to enable verification of these estimated changes on the land, and more research overall on the environmental effects from newly emerging biofuels. Furthermore, conservation practices exist to offset many of the environmental effects from the cultivation of conventional biofuel feedstocks (e.g., corn, soybean) and agricultural effects more generally; and, while some of these have been widely adopted (e.g., conservation tillage), some have not (e.g., cover crops). A sustained effort to deploy these practices across a wider area, especially in areas of recent cropland expansion may be needed to offset the potential negative effects from the RFS Program specifically and biofuels more generally.

Integrated Synthesis

This is the Third Triennial Report to Congress on Biofuels (RtC3) as required under Section 204 of the Energy Independence and Security Act of 2007 (EISA[1]). The purpose of this report and its predecessor reports (i.e., the First and Second Triennial Reports to Congress on Biofuels, RtC1 and RtC2, respectively) is to assess the "impacts to date and likely future impacts" of the Renewable Fuel Standard (RFS) Program on a range of environmental and resource conservation issues. Section 204 states:

"(a) In General. Not later than 3 years after the enactment of this section and every 3 years thereafter, the Administrator of the Environmental Protection Agency, in consultation with the Secretary of Agriculture and the Secretary of Energy, shall assess and report to Congress on the impacts to date and likely future impacts of the requirements of Section 211(o) of the Clean Air Act on the following:

1. *Environmental issues, including air quality, effects on hypoxia, pesticides, sediment, nutrient and pathogen levels in waters, acreage and function of waters, and soil environmental quality.*
2. *Resource conservation issues, including soil conservation, water availability, and ecosystem health and biodiversity, including impacts on forests, grasslands, and wetlands.*
3. *The growth and use of cultivated invasive or noxious plants and their impacts on the environment and agriculture.*

In advance of preparing the report required by this subsection, the Administrator may seek the views of the National Academy of Sciences or another appropriate independent research institute. The report shall include the annual volume of imported renewable fuels and feedstocks for renewable fuels, and the environmental impacts outside the United States of producing such fuels and feedstocks. The report required by this subsection shall include recommendations for actions to address any adverse impacts found."

What follows is the "Report at-a-Glance," which provides a high-level bulleted overview of the entire RtC3. The Integrated Synthesis then describes the background on the scope and content of the RtC3 and compares the overall conclusions from the RtC3 with the RtC2. Subsequently, the Integrated Synthesis presents the specific conclusions from individual chapters on the impacts to date and likely future impacts from the RFS Program.[2] The Integrated Synthesis then closes with discussion of uncertainties and limitations, and future recommendations.

[1] Energy Independence and Security Act of 2007, Pub. L. No. 110-140, 121 Stat. 1492, preamble (2007).
[2] In the RtC3, the term "impacts" is used to generally mean negative effects, while "effects" are more general and may be positive or negative.

Report At-a-Glance

Main Conclusions

- The impacts to date from the RFS Program are separate from, but overlap with, the effects of biofuels as an industry more generally. The estimated impacts to date from the RFS Program varied through time and for different biofuels as conditions in the market and co-occurring policies at the state and federal levels changed. The estimated impacts are expressed throughout the report as a range in billions of gallons per year and may include zero in the range.
- The impacts the RFS Program may have had in the past do not dictate the potential future effects of the Program, which can change as feedstocks, production, and conversion processes change.

Background

- The RtC3 assesses all 17 types of biofuels that were produced in or imported to the United States from 2005 through 2020. Emphasis is placed on the environmental and resource conservation issues specified in Section 204 from the production and use of biofuels that dominated U.S. production and consumption over this interval. These include: (1) domestic corn ethanol, (2) domestic soybean biodiesel, (3) domestic biodiesel from fats, oils, and greases (FOGs), and (4) imported ethanol from Brazilian sugarcane [Chapter 2, sections 2.3 and 2.5, Table 2.1, 2.2]. Although the focus of the RtC3 is on these four biofuels, other biofuels and their effects are discussed where appropriate [Chapters 8–15, sections 8.6, 9.6, etc., and Chapter 16].
- The period of rapid growth in the domestic corn ethanol industry was from 2002 to 2012. The RFS Program has changed over this period. The two versions of the RFS Program are commonly called the "RFS1" (in effect 2006–2008) and "RFS2" (in full effect since 2010). Nearly 40% of the increase in ethanol consumption had already occurred by the first full year of the RFS1 in 2006, and over 90% of the increase in consumption had already occurred by the first full year of the RFS2 in 2010 [Chapter 6, section 6.2].
- After decades of decline in cultivated cropland since at least the 1980s, increases in cultivated cropland by roughly 6–10 million acres have been recorded in multiple federal datasets, using a variety of methodologies, following the 2007 to 2012 period. This increase in cultivated cropland was largely driven by a net 26.5 million-acre increase in corn and soy with small grains and hay in rotation decreasing by 16.5 million acres. More than half of the corn and soybean acreage increase has come from other cultivated cropland (56%), while the rest has come from smaller proportions of pasture (13%), noncultivated cropland (20%), and the Conservation Reserve Program (CRP, 11%). Many of these changes are taking place throughout the Midwest, with

hotspots in northern Missouri, eastern Nebraska, North and South Dakota, Kansas, and parts of Wisconsin [Chapter 5, section 5.3].

- While EPA acknowledges the importance of greenhouse gases (GHGs) in assessing the environmental impacts of biofuels and the RFS, consistent with earlier reports, the RtC3 does not assess them here; EPA evaluates GHGs while administering the RFS Program (Sections 201 and 202 of EISA[3]).

Attribution

- Data allows for quantitative attribution of the potential impacts of the RFS Program on corn ethanol production and consumption. Information from economic models, observed prices for compliance credits (i.e., Renewable Identification Numbers [RINs]), and other sources suggest that from 2006 to 2011 the RFS Program—in isolation—accounted for 0–1 billion gallons per year of the U.S. corn ethanol produced and consumed. The estimated effect in these early years may have been primarily driven by encouraging market growth and capital investment and to a lesser extent by stabilizing demand during the Great Recession of 2008–2009. Other factors together likely played a more significant role in these earlier years (e.g., replacement of methyl tert-butyl ether [MTBE], volumetric excise tax credit [VEETC], and changes in refining operations). In more recent years other factors impacted the corn ethanol marketplace as well, such that the effect of the RFS Program is estimated to be 0–2.1 billion gallons per year [Chapter 6, sections 6.2, 6.3]. Based on these data, it is estimated that 0–9% of the corn ethanol production and consumption in the United States from 2005–2018 is attributable to the RFS Program.
- Uncertainties in the estimated effect of the RFS Program on domestic corn ethanol production and consumption remain, including the effect of the RFS Program in establishing market certainty and infrastructure buildout before the mandates were in full effect, future crude oil prices, the costs or willingness of refiners to switch back to producing finished gasoline without ethanol if blending ethanol were no longer economical, and many others. These factors are difficult to quantify. Thus, notwithstanding the many uncertainties, the ranges above represent the most current estimates based on current information for the effect of the RFS Program on domestic corn ethanol production and consumption in the United States [Chapter 6, sections 6.3.7, 6.4.4, 6.6].

[3] Energy Independence and Security Act of 2007, Pub. L. No. 110-140, § 202, 121 Stat. 1492, 1521-28 (2007) (codified as amended at 42. U.S.C. § 7545(o)). Detailed assessment of the GHG balance of corn ethanol and other biofuels are not in scope of this report series. See Chapter 2 (Box 2.2) for an overview and see Federal Registry (FR) FRL–9307–01–OAR (https://www.epa.gov/renewable-fuel-standard-program/workshop-biofuel-greenhouse-gas-modeling) and EPA's 2023 lifecycle analysis Model Comparison Technical Document (https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1017P9B.pdf).

- The RFS Program likely had a larger effect on biodiesel and renewable diesel (and other biofuels) throughout the years of the RFS2. However, there is insufficient information available at present to confidently quantify the attributional effect annually of the RFS Program alone on biodiesel and renewable diesel. This is mostly due to a lack of data and peer-reviewed studies that focus on biodiesel and control for key factors important in the biodiesel market, such as the Biodiesel Tax Credit (BTC) and state incentive programs. Initial estimates suggest that taken together, the RFS Program and the BTC are likely responsible for 70–100% of historical biodiesel and renewable diesel use in the United States [Chapter 7].
- The development of the dry mill ethanol industry in the United States was largely underway and mostly completed by the time that the RFS2 was passed and took effect. These legislative and regulatory actions, or the prospects of them, likely provided policy certainty for investors, including farmer cooperatives. The use of corn and soybean surpluses for transportation—driven by a variety of factors—was in effect a market clearing mechanism that reduced surplus stocks, sustained crop prices above the costs of production, and partially shifted the support of agricultural surpluses from the Farm Bill to the transportation sector.
- This report only quantifies the volumes of corn ethanol attributable to the RFS Program alone and therefore the effects on land and other environmental and resource conservation issues are only quantified for the RFS-effect on corn ethanol, and not for the RFS-effect on soybean biodiesel or other biofuels.
- As the effect of the RFS Program on corn ethanol varies through time and includes zero, so do estimates on cropland expansion from the RFS Program [Chapter 6, section 6.4]. Between zero and 1.9 million acres of new cropland (0–20% of the observed *increase* in cropland, 0–0.5% of all cropland) and between zero and 3.5 million acres of additional corn (0–35% of the observed *increase* in corn, 0-3.7% of all corn), mostly in the Midwest, are estimated to be attributable to the RFS Program. Data limitations prevent the isolation of the exact areas of cropland expansion that were estimated attributable to the RFS Program. For the high end of these ranges, there is a greater estimated increase in corn acreage than overall cropland acreage because some new corn may come from switching of crops on existing cropland (commonly from soy, wheat, or cotton).

Environmental Effects

- Applying the estimated ranges of cropland expansion potentially attributable to the RFS Program suggests that the RFS Program may have been responsible for a range of effects, from no effect to small negative effects on soil quality [Chapter 9, section 9.3.3], water quality [Chapter 10, section 10.3.3], and other environmental effects covered in this report, as concluded but not quantified in the RtC1 and RtC2. More precise descriptions or quantifications of effects on various environmental end points are not possible because data identifying specific areas of RFS-induced land use change are unavailable.

- The RtC3 reiterates the conclusions from the RtC1 and RtC2 that emissions of nitrogen oxides (NO_x), sulfur oxides (SO_x), carbon monoxide (CO), volatile organic compounds (VOCs), ammonia (NH_3), and particulate matter ($PM_{2.5}$) can occur at each stage of biofuel production, distribution, and usage and impact air quality [Chapter 8]. In addition, impacts on ambient concentrations vary depending on the geographic location and local conditions. The EPA's anti-backsliding study, which focused on changes in air quality associated with vehicle and engine emissions (rather than the full lifecycle) using "pre-RFS" fuel and "with-RFS" fuel, found ozone and $PM_{2.5}$ can increase or decrease depending on location, and in general, NO_2 and acetaldehyde increase, while CO and benzene decrease [Chapter 8, section 8.3.2.2].
- Lifecycle assessments of criteria air pollutants and precursors using GREET (Greenhouse Gases, Regulated Emissions, and Energy Use in Technologies) suggest that lifecycle emissions per unit energy from corn ethanol are generally higher than from gasoline for VOCs, SO_x, $PM_{2.5}$, PM_{10}, and NO_x, and that lifecycle emissions per unit energy from soybean biodiesel are generally higher than from diesel for VOCs, SO_x, and NO_x. However, the location of emissions from biofuel production tends to be in more rural areas where there are fewer people. How this translates to health effects on communities is complex, as it depends not only on the number of people, but on their demographics and vulnerability, as well as the dose-response relationship, which is pollutant-specific, among other factors. Trends suggest that the potential lifecycle effects per unit energy from biofuels are decreasing over time as industries mature and practices improve. These lifecycle inventories from GREET estimate emissions rather than estimate actual effects to biological receptors (e.g., humans, ecosystems) and may underestimate effects from fossil fuels due to the omission of factors such as oil spills [Chapters 8, 10, 11; sections 8.5, 10.5, 11.5].
- Although this report estimates that nationally 0 to 1.9 million acres of additional cropland and 0 to 3.5 million acres of additional corn may be attributable to the RFS Program for the historical period assessed, there are insufficient data to determine potential land, water, and species impacts in specific areas below the county scale. If a portion of the observed cropland expansion was due to the RFS Program, it may have had some effect on critical habitat and threatened and endangered species; however, whether that effect would have constituted an adverse effect in the context of the Endangered Species Act (ESA) is unknown [Chapter 12, sections 12.3.2 and 12.3.3; Chapter 13, sections 13.3.2.2 and 13.3.3]. EPA has separately evaluated the potential effects on threatened and endangered species for 2023–2025 in the Set Rule (docket #EPA-HQ-

OAR-2021-0427) and determined that the rule is not likely to adversely affect listed species and their designated critical habitats.[4]

- Overall, even though the estimated environmental impacts from the RFS Program may be small, any impacts may represent additional strain to already strained environments and could be significant locally. Some conservation practices are becoming widely adopted in the United States with positive effects on the environment, while others are not. Many of the potential impacts from the RFS Program specifically and biofuels more generally could be offset with greater adoption of conservation practices [Chapter 3, section 3.2.1].

Likely Future Effects

- The likely future effects from the RFS Program were published in the Final Set Rule for 2023–2025 and projected an increase of 3.9 billion gallons in 2025 due to the RFS Program over the baseline (with no RFS Program) [Chapter 6, Table 6.12]. This increase in 2025 from the RFS Program is primarily from increases in biodiesel and renewable diesel from soybean oil (+1.5 billion gallons), increases in cellulosic biofuel from compressed natural gas (CNG)-liquified natural gas (LNG) biogas (+932 million gallons), and increases in corn ethanol (+787 million gallons). Domestic production and consumption of other biofuels are expected to change little by comparison. These estimated impacts in 2025 from the RFS Program are different from the trends through time from 2022 to 2025. Though highly uncertain, EPA determined in its ESA biological evaluation for the Set Rule that the RFS-attributable volumes could potentially lead to an additional increase of up to 2.65 million acres of cropland by 2025.
- While the projected cropland expansion for 2023–2025 is slightly larger than the estimated historical cropland expansion from the RFS Program, it cannot be said with reasonable certainty that any particular environmental and resource conservation effect will be impacted, due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes. These projected future effects remain uncertain due to many other factors, including ongoing recovery from the global COVID-19 pandemic, uncertainty in the penetration of E15 in the marketplace, uncertain growth of cellulosic ethanol production from

[4] On August 3, 2023, EPA completed its Endangered Species Act informal consultation on the Renewable Fuel Standard (RFS) Program: Standards for 2023-2025 and Other Changes rulemaking (also known as the RFS "Set Rule"). With the Biological Evaluation that EPA submitted to the National Marine Fisheries Service (NMFS) and Fish and Wildlife Service (FWS) on May 19, 2023, EPA determined that the RFS Set Rule is not likely to adversely affect listed species and their designated critical habitats. EPA received letters of concurrence with this determination from NMFS on July 27, 2023, and from FWS on August 3, 2023, thereby concluding the consultation. The Biological Evaluation and Letters of concurrence are available at https://www.epa.gov/renewable-fuel-standard-program/final-renewable-fuels-standards-rule-2023-2024-and-2025.

agricultural or marginal lands, and complex transportation market dynamics, among other factors [Chapter 2, section 2.3.2; Chapter 6, section 6.5].

Background

In August 2005, the Energy Policy Act of 2005 (EPAct)[5] was enacted, which included the creation of the RFS Program to be administered by the EPA. In December 2007, EISA was enacted with the stated goals of "mov[ing] the United States toward greater energy independence and security [and] to increase the production of clean renewable fuels." In accordance with these goals, Section 202 of EISA revised the RFS Program to nearly double the volume of renewable fuel required to be blended into transportation fuel from 5.4 to 9 billion gallons in 2008 and to 36 billion gallons per year by 2022. EISA also included Section 204 which required this report every three years. The two versions of the RFS Program under the EPAct and EISA are commonly called the "RFS1" (in effect 2006–2008) and "RFS2" (in full effect since 2010).[6]

More than a decade after the full implementation of the RFS2, there is sufficient data and scientific literature to assess partially the historical effects of the RFS Program. These data and information were not available for the 2011 RtC1, which was primarily forward looking; and, much of it was not available for the 2018 RtC2. Many important analyses have been published since 2018. The detail and sophistication of the literature has evolved over time, with earlier studies often presuming the RFS Program was the only factor affecting biofuels in the United States and assuming higher levels of biofuel production than later occurred (e.g., cellulosic biofuels). More recent studies include more market and industry detail, with more realistic assumptions of biofuel production levels informed by observations. Thus, more than a decade after implementation of RFS2, there exist data to more fully assess the potential impacts of the RFS Program since its inception.

One of the emphases in the RtC3 is on attribution of effects to the RFS Program as opposed to biofuels in general. Impacts from the RFS Program may overlap partly or entirely with the impacts from biofuels more generally. Many studies have assumed either implicitly or explicitly that U.S. biofuel production was driven solely by the RFS Program, which has limited the ability of previous assessments to attribute effects to the Program. There are many policies—federal and state—and economic and agronomic factors that affect biofuel production, not just the RFS Program. It is not the purpose of the RtC3 to assess the effect of all these other drivers on biofuels, nor to assess the environmental effects of

[5] Energy Policy Act, Pub. L. No. 109-58, 119 Stat. 594 (2005).

[6] 2009 was a transition year between programs, where the total biofuel volume standards were based on the RFS2-level volumes, but there was only a single total renewable fuel standard as with the RFS1. The RFS2 with its four nested renewable fuel standards [Chapter 1, section 1.1] was not fully implemented until 2010.

all of agriculture or even all agricultural feedstocks that may be used for biofuels. However, many of these contexts are discussed for comparison. Rather, the purpose of this report, as stated clearly in EISA, is to assess the impacts to date and likely future impacts of the RFS Program to inform Congress and EPA in the administration of the Program.

The RtC3 evaluated all biofuel-feedstock-region combinations that produced RINs (e.g., biodiesel-soybean-Argentina, ethanol-corn-U.S.) since the inception of the RFS Program (2005) to 2020, and focused on those that dominated the U.S. biofuel marketplace. Thus, while 17 combinations were evaluated for this report (Figure IS.1, Chapter 2, section 2.3), four were identified as potentially having substantive impacts on the environmental effects covered in this report: (1) domestic corn ethanol, (2) domestic soybean biodiesel, (3) domestic fats, oils, and greases (FOGs), and (4) imported ethanol from Brazilian sugarcane. Although the emphasis of the RtC3 is on these four biofuels, other biofuels and effects are also discussed in the chapters where they may be particularly relevant (e.g., cellulosic biofuels in Chapter 9 [section 9.6], palm biodiesel from Southeast Asia in Chapter 16 [section 16.4 and 16.5]).

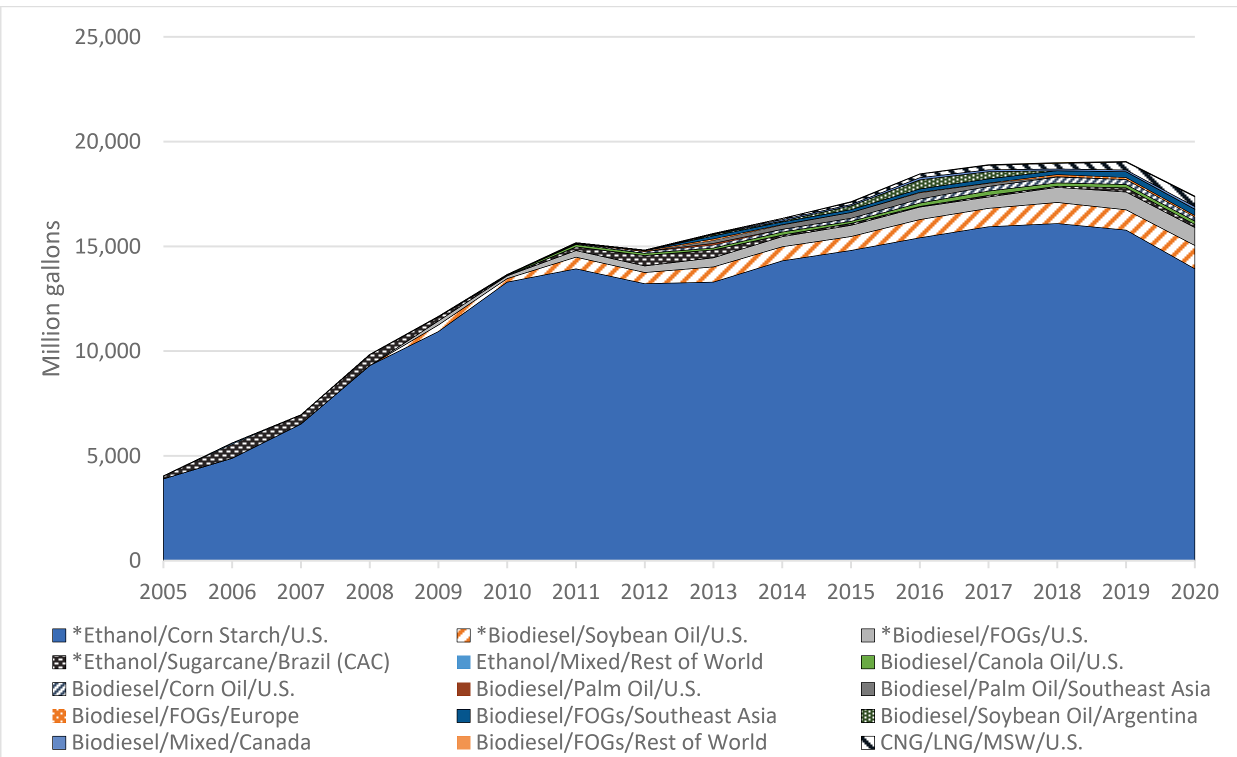

Figure IS.1. The estimated volumes of biofuel (million gallons) imported or domestically produced from individual biofuel/feedstock/region combinations from 2005 to 2020 (same information as Chapter 2, Table 2.1[7]). All combinations are mentioned in the RtC3 but the four dominant biofuels (*) are emphasized. Note that biodiesel also includes renewable diesel.[8]

[7] For Figure IS.1, sugarcane ethanol from Central America and the Caribbean (CAC) was combined with Brazil because, as explained in Chapter 2, most of the ethanol imported from the CAC actually originated in Brazil.
[8] Details on the sources of information for Table IS.1 are in Chapter 2 and Appendix B. CNG/LNG-MSW stands for compressed natural gas (CNG) or liquified natural gas (LNG) from municipal solid waste (MSW).

The statutory language in Section 204 of EISA establishes the general environmental and resource conservation issues to be addressed in the reports. The authors interpret and define terms in the statutory language based on technical knowledge of the subject matter. From this, the categories listed in the statutory language were reorganized into groups that are more consistent with the scientific literature (Table IS.1).

In addition to what is included in the statutory language of EISA Section 204, what is not included in Section 204 helps to limit the scope. GHGs and climate change are not mentioned in EISA Section 204, and thus are not explicitly addressed in this report (but see Chapter 2, Box 2.2 for a brief overview). GHGs are explicitly addressed in EISA Section 202 which modified the RFS Program, and are evaluated during the biofuel pathway analysis conducted by EPA as part of the ongoing implementation of the RFS Program. EPA maintains a summary of lifecycle GHG intensities estimated for the RFS Program, which are available in spreadsheet form in a document titled "Summary Lifecycle Analysis Greenhouse Gas Results for the U.S. Renewable Fuels Standard Program."[9] EPA's

Table IS.1. Mapping of statutory language in EISA Section 204 and the RtC3

EISA Section 204(a) statutory language	RtC3 chapter number (and title)
Environmental [. . .] and Resource [C]onservation [I]ssues	Chapters contained in Part 3
[A]ir quality	Chapter 8 (Air quality)
[E]ffects on hypoxia	Chapter 13 (Aquatic ecosystems)
[P]esticides, sediment, nutrient, and pathogen levels in waters	Chapter 10 (Water quality)
[A]creage and function of waters	Chapter 11 (Water availability)
[S]oil environmental quality	Chapter 9 (Soil quality and conservation)
[S]oil conservation	Chapter 9 (Soil quality and conservation)
[W]ater availability	Chapter 11 (Water availability)
[E]cosystem health and biodiversity	Chapter 12–14 (separated by ecosystem type for terrestrial [12], aquatic [13], and wetlands [14])
[I]mpacts on forests	Chapter 12 (Terrestrial ecosystems)
[I]mpacts on [. . .] grasslands	Chapter 12 (Terrestrial ecosystems)
[I]mpacts on [. . .] wetlands	Chapter 14 (Wetlands)
The growth and use of cultivated invasive or noxious plants and their impacts on the environment and agriculture.	Chapter 15 (Invasive species)
[T]he annual volume of imported renewable fuels and feedstocks for renewable fuels, and the environmental impacts outside the United States of producing such fuels and feedstocks.	Chapter 16 (International effects)

[9] This document is available on EPA's website at https://www.epa.gov/fuels-registration-reporting-and-compliance-help/lifecycle-greenhouse-gas-results. This summary is also available in docket EPA-HQ-OAR-2021-0324.

analyses of the lifecycle assessment (LCA) of various pathways are also published online.[10] A list of pathways that have been approved by regulation can also be found at 40 CFR 80.1426(f)(1). This approach of omitting GHGs in the RtC3 is consistent with the RtC1 and RtC2.

Comparison of Overall Conclusions Between the RtC2 and RtC3

This section presents the overall conclusions from the RtC2 (literature review cutoff date of April 2017) and discusses any different or new conclusions in RtC3 versus the earlier report. Overall conclusions from the RtC2 were:

- Disregarding any effects that biofuels have on displacing other sources of transportation energy, evidence since 2011 indicates the specific environmental impacts listed in EISA Section 204 are negative. The environmental and resource conservation impacts, whether positive or negative, related to displacement of other transportation energy sources by biofuels were not assessed.
- Literature published since 2011 supports the conclusion of the potential for positive and negative effects. Available information suggests, without accounting for the environmental effects of displacing other sources of transportation energy, the specific environmental impacts listed in EISA Section 204 are negative in comparison to the period prior to enactment of EISA.
- Evidence continues to support the conclusion that biofuel production and use could be achieved with reduced environmental impacts. The majority of biofuels continue to be produced from corn grain and soybeans, with associated impacts that are well understood. Cellulosic and other feedstocks remain a minimal contributor to total biofuel production.

The RtC3 reaffirms the conclusions in the RtC2. The RtC2 reported that there were land use change trends observed that were consistent with a potential effect from the RFS Program (e.g., increases in corn acreage and total cropland). However, there was not enough information available at the time to separate the effects of biofuels generally from the effects of the RFS Program specifically (see RtC2 page ix). The RtC3 advances the knowledge in this important area. The RtC3 reaffirms the conclusion that biofuels have the potential for positive and negative effects, and that the majority of impacts to date come from lifecycle effects from corn ethanol and soybean biodiesel. The RtC3 does not focus on comparing the impacts from biofuels to those of conventional fossil fuels, as Section 204 does not address fossil fuels' impacts. Part 3 of this report (Chapters 8–16) includes limited comparisons where the scientific literature is available. Additionally, related material comparing biofuels to their fossil fuel counterparts on

[10] See https://www.epa.gov/renewable-fuel-standard-program/approved-pathways-renewable-fuel and https://www.epa.gov/renewable-fuel-standard-program/other-actions-renewable-fuel-standard-program

a per-megajoule basis is presented from established lifecycle models (i.e., Greenhouse Gases, Regulated Emissions, and Energy Use in Technologies [GREET]), and from other approaches and models. The RtC3 focuses on estimating the impacts from the RFS Program, though impacts from biofuels more broadly are also discussed as important context. Overall conclusions from the RtC3 are:

- The overall effect of the RFS Program on biofuels depends on the biofuel being discussed and is dynamic through time because of several co-occurring market and non-market factors. The RFS Program itself played a relatively minor role in the increase in corn ethanol in the United States, but has played a more significant role for other biofuels.
- The volume of domestic corn ethanol consumption estimated to be attributable to the RFS Program historically suggests that a maximum of 0–1.9 million acres of cropland expansion (roughly 0–20% of the estimated *increase* in cropland, and 0–0.5% of all cropland) and 0–3.5 million acres of corn expansion (roughly 0–35% of the observed *increase* in corn acreage, and 0–3.7% of all corn acreage) are estimated to be attributable to the RFS Program.
- As the historical effect of the RFS Program on domestic corn ethanol production and consumption and associated land use changes varies through time and includes zero in the range of estimates each year, estimates of environmental impacts also vary through time and include zero each year. This holds for most end points examined, with small but negative potential impacts nationally on soil quality, water quality, biodiversity, and other effects. Local impacts may be larger in some areas for some effects, but this could not be quantified for the RtC3.
- Though adoption of conservation practices is improving, additional conservation measures—such as further adoption of conservation tillage and cover crops—would help reduce the impacts of biofuels generally and the potential RFS Program specifically on the environment.
- Consistent with the RtC1 and RtC2, the RtC3 does not estimate or assess the impact of increased renewable fuel consumption on conventional fossil fuel consumption, nor does it assess the environmental impacts of changes in of fossil fuel production or consumption.

The following sections discuss specific conclusions from chapters in the RtC3 on the impacts to date, the likely future effects, uncertainties and limitations, and recommendations.[11]

[11] Specific conclusions from Chapters 1–4 are not presented in the Integrated Synthesis as these are more background material for the RtC3.

Specific Conclusions: Impacts to Date

Domestic Land Cover and Land Management [Chapter 5]

Land use change from all causes shows a steady increase in total cultivated cropland and corn/soy acreage since 2007. Based on the 2012, 2015, and 2017 U.S. Department of Agriculture (USDA) National Resource Inventory (NRI), from 2007 to 2017 there has been a 10 million-acre increase in cultivated cropland coinciding with a 15 million-acre decline in perennially managed land (i.e., sum of lands in the Conservation Reserve Program [CRP],[12] pasture, and noncultivated cropland). This increase in cultivated cropland was largely driven by a 26.5 million-acre increase in corn and soybeans with small grains and hay in rotation decreasing by 16.5 million acres. Results from other federal datasets such as the Cropland Data Layer (CDL) and the Census of Agriculture are consistent with the NRI when harmonized appropriately. Thus, after decades of decline in cultivated cropland since at least the 1980s, increases have been recorded in multiple federal datasets using a variety of methodologies following the 2007 to 2012 period. More than half of the corn and soybean increase has come from other cultivated cropland (56%), while the rest has come from approximately equal proportions of pasture (13%), noncultivated cropland (20%), and CRP (11%). Many of these changes are taking place throughout the Midwest, with hotspots in northern Missouri, eastern Nebraska, North and South Dakota, Kansas, and parts of Wisconsin. Lands enrolled in the CRP have steadily decreased since 2007; and, although these decreases are likely due to Farm Bill policies and not directly to biofuels, how these lands are managed after leaving the CRP are likely influenced by biofuels and the RFS Program. More recently, the Agriculture Improvement Act of 2018 increased maximum allowable CRP land to 27 million acres in 2023 and enrolled acreage has increased significantly. Data to assess the effect of the RFS on CRP enrollment under this new allotment is not currently available.

Attribution: Corn Ethanol and Corn [Chapter 6]

Multiple lines of evidence suggest the RFS Program itself played a relatively minor role in the growth of corn ethanol in the United States (0–1.0 billion gallons per year from 2002–2011) and may have played a more important role more recently since reaching the E10 blend wall (0–2.1 billion gallons per year from 2012–2018).[13] Many factors overlap with and predate the RFS Program. Principal among these was the need of a replacement for methyl-tert-butyl-ether (MTBE) as an

[12] https://www.fsa.usda.gov/programs-and-services/conservation-programs/conservation-reserve-program/

[13] The E10 blend wall commonly describes the amount of ethanol that can be blended into the gasoline pool at 10% by volume. Above this limit, higher amounts of ethanol consumption domestically would have to come from higher blends where it faces greater economic challenges. E15 is approved for use in vehicles manufactured after 2000 but remains limited in availability nationally [see Chapters 2 and 3].

oxygenate[14] in gasoline for areas with smog concerns administered under the Reformulated Gasoline Program (RFG). From 2003 to 2006, largely before the RFS Program, roughly a third of the national gasoline pool needed a substitute for MTBE because of growing concerns, ongoing litigation, and individual states addressing the environmental issues associated with MTBE. At the time, that substitute was ethanol from corn grain. Ethanol is an oxygenate, and ethanol from corn grain was estimated at the time to be the only substitute available in the quantities needed that did not require expensive refinery retrofitting that other petroleum-based alternatives may have needed. Furthermore, ethanol did not have the same potential water quality concerns as other petroleum-based substitutes for MTBE (e.g., ethyl tertiary butyl ether, or ETBE). The logistical barriers that had previously limited ethanol consumption to the Midwest had to be overcome to provide ethanol to the largely coastal and urban areas that were administered under the RFG.

Gasoline used to be produced as "finished gasoline" (E0) ready for sales at gas stations. This gasoline met all the necessary standards under the Clean Air Act (CAA) for transportation fuels. To make E10 in these early years, E0 was "splash blended" with ethanol often at the gas station or terminal. Splash blending refers to mixing ethanol with finished gasoline to reach 10% ethanol by volume. Between 2005 and 2010, refineries invested in switching to "match blending," whereby refineries utilized the higher octane in ethanol in their processes to target a specific octane rating in the finished product. To carry out match blending, refineries switched to producing Blendstocks for Oxygenate Blending (BOBs), which are "unfinished gasoline" that can only be legally sold at the pump (i.e., meeting all applicable CAA standards) after an oxygenate is added. These BOBs were then mixed with ethanol at the refinery or terminal to produce E10. BOBs are cheaper to produce because they require less refining and take advantage of the higher octane value of the oxygenate. They rely on changes to refinery operations and the downstream distribution and blending network. As a result of these changes, it would be difficult and costly to revert back to the production of finished gasoline.

Once the supply chains were in place, and with the construction boom in ethanol biorefineries in 2006 and 2007, ethanol in the United States was poised to quickly reach market saturation at 10% of the gasoline pool. By 2006 (the first year of the RFS Program), ethanol consumption far outpaced the RFS1 mandates and had already increased to 40% of the E10 blend wall. By 2010—the first year of the RFS2—ethanol consumption was nearing 93% of the E10 blend wall, and the volume of ethanol production either operating or under construction was already 13.4 billion gallons. Record high oil prices in this period,

[14] Octane enhancers are added to transportation gasoline to avoid engine knock. Octane enhancers may be oxygenates (i.e., contain oxygen, such as MTBE and ethanol) or not (e.g., tetra-ethyl lead, or "lead"), and may be petroleum-based (e.g., MTBE) or renewable (e.g., corn ethanol). Octane enhancers used in U.S. gasoline has changed through time, from lead in the 1920s–1980s, to MTBE in the 1980s–2000s, to ethanol from the 2000s to the current day.

beginning in 2005, also made gasoline with 10% ethanol cheaper to produce than gasoline without ethanol, and so the market responded with increased ethanol consumption also in non-RFG areas. If these factors had not been in place, the RFS Program likely would have had a stronger and more direct effect in encouraging the growth of corn ethanol in the United States.

More recently, the RFS Program may be playing a more significant role in the continued production and consumption of corn ethanol. Market and policy conditions have changed with the expiration of the Volumetric Ethanol Excise Tax Credit (VEETC, 2004–2011), the drop in oil prices after 2015, and the decrease in consumption from the global COVID-19 pandemic starting in 2020. Therefore, the effect of the RFS Program in sustaining production may be more important in recent years compared with historically. However, there remains uncertainty surrounding the recent influence of the RFS Program because refineries have already made costly investments to switch to match blending, and retrofitting refineries to produce gasoline without ethanol could be cost prohibitive.

The RFS Program is a policy applied to a dynamic market, and therefore the effect of the policy is also dynamic through time. RIN prices for renewable (D6) fuels provide evidence that the RFS Program increased U.S. consumption of renewable biofuels in 2009 (and late 2008) and from 2013 to 2019. Higher D6 RIN prices after reaching the E10 blend wall in 2013 are likely not indicative of an effect on corn ethanol, because of the nested nature of the RFS standards. They are indicative of an effect on total renewable fuel. Nonetheless, estimates from simulation models, the observed overproduction of ethanol domestically compared to the RFS standards, and other sources suggest that from 2006 to 2011 the RFS Program—in isolation—accounted for 0–1 billion gallons of ethanol. This effect in the earlier years appears to be due to contributions to market certainty and encouragement of capital investment from EISA, and to a lesser extent by stabilization of demand during the Great Recession of 2008–2009. In other years of this period, the RFS Program is estimated to have had no effect on ethanol production, with other factors having more influence (Figure IS.2). From 2012 to 2018, there is a wider range of estimates of the effect of the RFS Program than in the 2006–2011 period, as other contributing factors diminished (e.g., oil prices declined after 2015, VEETC expired at the end of 2011, MTBE transition had already occurred). From 2012 to 2018, annual estimates of the range of impacts of the RFS Program vary from year to year. The minimum estimated effect is zero for every year examined, and the maximum varied from year to year and was highest in 2016 at 2.1 billion gallons. (Figure IS.2). The low end of this range is driven by a thorough state-by-state analysis of the relative economics for refiners for match blending 10% ethanol into gasoline, taking into consideration its considerable octane value. In addition, even where the economics for blending ethanol may not have been favorable for some gasoline grades in some states, a strong "lock-in effect" from the transition to match blending was presumed to prevent reversion. The high end of this range is from economic modeling of the biofuels industry that includes key factors such as the price of oil, MTBE, and the potential octane value of ethanol. This report focuses on this historical period when the growth in domestic ethanol production occurred.

Combining these estimated volumes attributable to the RFS Program with literature reviews and a recent statistical analysis suggests that additional corn and new cropland areas, with estimates ranging from zero to as high as 3.5 ± 1.0 million acres of corn, and from zero to as high as 1.9 ± 0.9 million acres of cropland expansion may be attributable overall to the RFS Program from direct and indirect effects.[15] Though small relative to total cropland (0–0.5%) and total corn acreage (0–3.7%), this corresponds to 0–20% of the *increase* in cropland and 0–35% of the *increase* in corn acreage from 2008 to 2016.

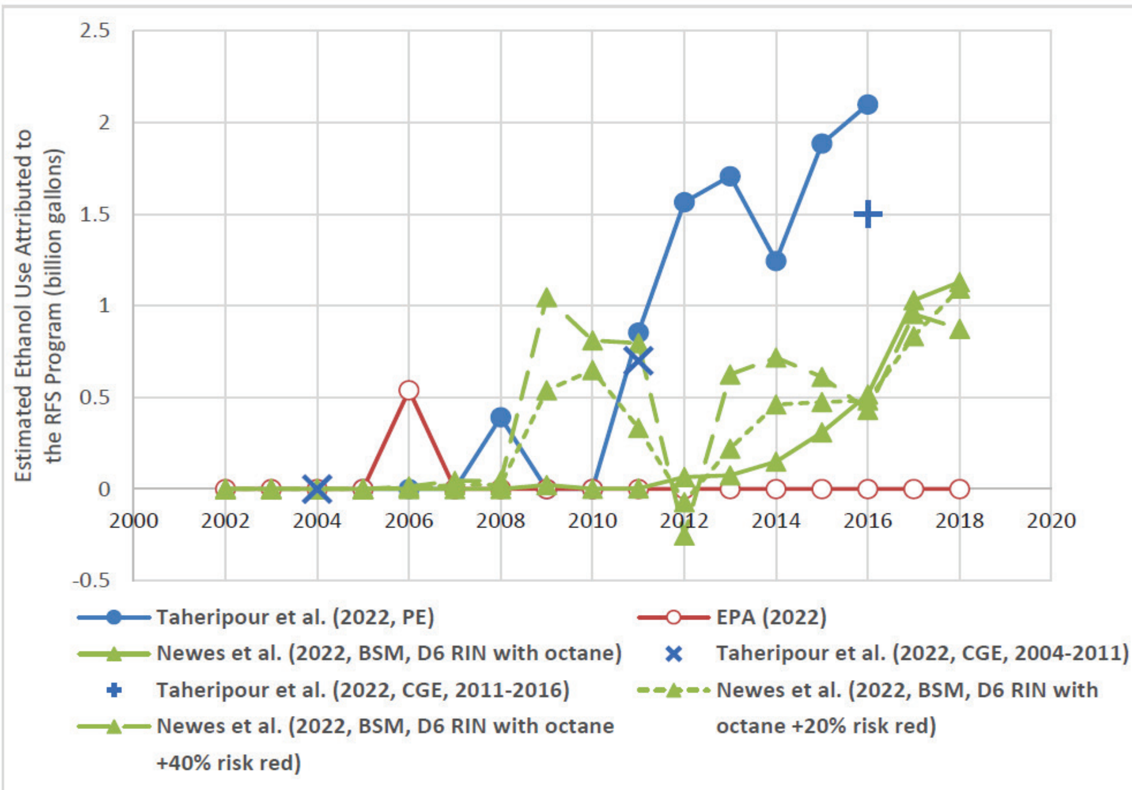

Figure IS.2. Comparison of estimates of ethanol use attributed to the Renewable Fuel Standard (RFS) Program from key studies in Chapter 6. Shown are estimates from recent models that separate estimated RFS effects from other key factors (e.g., oil price, MTBE, transition to match blending). These include: the annual partial-equilibrium (PE) model in Taheripour et al. 2022 (AEPE, blue line, circles); the two general equilibrium (GE) periods in Taheripour et al. 2022 (GTAP-BIO; 2004–2011, blue "x"; and 2011–2016, blue "+"); Newes et al. 2022 using the Biomass Scenario Model (BSM, green line, triangles); and an economic analysis from EPA 2022 (red line, circles). Note the estimate in 2006 from EPA 2022 is driven more by the MTBE phaseout than the RFS Program (see section 6.3.5). The estimates from Newes et al. (2022) show those that include the estimated effect the RFS Program had on increasing market certainty (dashed green lines, two levels of 20% and 40% risk reduction), and those that do not include this effect (solid green line).

[15] Note that the additional corn could have come all from existing cropland, or up to 1.9 million acres of it could come from new cropland. This result simply means that it is estimated that there are 0–3.5 million more acres of corn and 0–1.9 million more acres of total cropland than would have occurred absent the RFS Program.

There are many uncertainties associated with this estimate of the volume of ethanol attributable to the RFS Program. There is no single study that used robust methodologies across all steps of the analysis; thus, this estimate represents a synthesis from the best available studies to date. Disentangling the effect of the RFS Program, as required under EISA Section 204, is difficult given the many co-occurring factors that affect biofuels in the United States. As a mandate, the RFS Program created a guaranteed market demand for biofuels in the United States that certainly *could have* driven the increase in ethanol production and consumption. However, this potential market demand from the RFS Program also existed for cellulosic biofuels that have not seen a similar increase; thus, clearly other factors must also align. There are many factors not included or not investigated in depth in this analysis, including a more detailed examination on the effect the RFS Program had in influencing investor confidence and infrastructure buildout before the mandates were in full effect, the costs or willingness of refiners to switch back to producing finished gasoline if the RFS Program were no longer in effect, and others. These factors are difficult to quantify. Furthermore, as events played out, non-RFS factors that are quantified and known to influence the market were favorable and appear to sufficiently explain much or all of the increase in corn ethanol production and consumption in the United States. Thus, though notwithstanding several uncertainties, these represent the best estimate based on currently available information for the effect of the RFS Program on corn ethanol and the associated effects on cropland in the United States.

These RFS effects, though smaller than anticipated by many studies discussed in Chapters 4 and 6, may still have implications on the nation's air, land, and water, and have more significant effects locally. However, specific areas where environmental effects may have occurred cannot be quantified with confidence due to the vast quantity of potential cropland in the United States and the multitude of factors that contribute to an individual farmer's decision whether to bring additional land into crop production. The more likely hotspots of increased cropland and corn/soy acreage have been identified throughout the country (Chapter 5, section 5.3.1).

Attribution: Biodiesel and Renewable Diesel [Chapter 7]

The RFS Program, especially after the expansions in the RFS2, likely always played an important role in supporting the production and consumption of biodiesel and renewable diesel, in contrast to corn ethanol. However, separating the effect of the RFS Program quantitatively from other factors remains difficult. Before 2010 and the RFS2, the RFS Program had little effect on biodiesel because there was no separate biodiesel or advanced mandates, and domestic corn ethanol and imported Brazilian sugarcane ethanol[16] were the most cost-effective way to meet the total renewable fuel

[16] Imports from Brazil were largely temporary, limited to a few early years before U.S. production had grown, and to a few later years when drought occurred that lowered U.S. production.

standards under the RFS1. Other factors such as the Biodiesel Tax Credit (BTC) and state incentives were especially influential in these earlier years for biodiesel. Once there existed a biodiesel mandate with the RFS2, the RFS Program and other policies played an important role in the increased production and consumption of biodiesel and renewable diesel. Biodiesel and renewable diesel demand is fundamentally different from ethanol—biodiesel and renewable diesel were not incentivized by the need for a substitute for MTBE in gasoline, and oil prices were not ever high enough to make biodiesel competitive with diesel on the basis of price alone. Thus, the RFS Program created an important added incentive beginning with the RFS2 in 2010. Advanced (D5), biomass-based diesel (D4), and cellulosic (D3) RIN prices provide evidence that the RFS2 increased U.S. consumption of advanced, biomass-based diesel, and cellulosic biofuels. Aside from observed RIN prices and a handful of studies, there is much less quantitative information in the peer-reviewed literature on the effects of the RFS Program on biodiesel compared with effects on corn ethanol, and none of the studies assessed included other factors such as FOGs, the BTC, or state biofuels mandates. The handful of economic models suggest a strong effect from the RFS Program—with a 1 billion gallon increase in the RFS biodiesel standard inducing an increase in biodiesel consumption by 0.6–1.1 billion gallons. Comparison of state and federal mandates suggest that while roughly 0–30% of biodiesel consumption may be due to state programs (e.g., mandates and low carbon programs like the California Low Carbon Fuel Standard, LCFS), the remaining 70–100% may be attributable to a combination of other factors, primarily the RFS Program and the BTC. The effects of the RFS Program on the historical period cannot be isolated at this time because most studies do not separate the RFS from other important factors that occurred at the same time such as the BTC and state programs. Although multiple lines of information suggest a sustained effect of the RFS Program since 2010 on supporting biodiesel production and consumption, the effects from other factors such as the BTC and state incentives cannot be quantitatively separated from the effects of the RFS Program at this time. Thus, instead of a volumetric and acreage-based estimate of attribution in the RtC3, a more general synthesis is provided.

Air Quality [Chapter 8]

The RtC3 reiterates the conclusions from the RtC1 and RtC2 on air quality, concluding that emissions of nitrogen oxides (NO_x), sulfur oxides (SO_x), carbon monoxide (CO), volatile organic compounds (VOCs), ammonia (NH_3), and fine particulate matter ($PM_{2.5}$) can be impacted at each stage of biofuel production, distribution, and usage. EPA's "anti-backsliding" study (see section 8.3.2.2) examined the impacts on vehicle and engine emissions and air quality from two different fuel scenarios for calendar year 2016. Specifically, the study compared air quality impacts of actual renewable fuel volumes in 2016 to a scenario with renewable fuel use approximating the 2005 levels before the RFS was enacted. The anti-backsliding study, which is not a full lifecycle assessment but focused on vehicle and engine emissions, found atmospheric concentrations of ozone and $PM_{2.5}$ can increase or decrease depending on location, and in general, NO_2 and acetaldehyde concentrations increase, while CO and benzene concentrations decrease. Lifecycle analyses conducted by the Argonne National Lab using GREET indicate that on a per unit energy basis many non-GHG emissions, including of several criteria air pollutants, are higher for biofuels per unit energy than their petroleum counterparts. However, the location of emissions from biofuel production tends to be in more rural areas where there are fewer people. How this translates to health effects on communities is complex, as it depends not only on the number of people, but on the dose-response relationship (e.g., possibly fewer people in rural areas but receiving higher or lower doses), their demographics and vulnerability (e.g., elderly or other at-risk populations), as well as other factors. Other modeling approaches support these findings, but also show that biofuels are improving as industries mature and practices improve. These analyses, though state-of-the-art, may not reflect some recent improvements in biorefining, are not spatially resolved enough to be directly linked with exposure, and do not account for many large-scale events associated with oil and gas exploration that may affect the overall results (e.g., oil spills).

Soil Quality [Chapter 9]

Effects on soil quality to date, as with effects detailed in other chapters, continue to be primarily from the cultivation of corn and soybean feedstocks. The soil quality effects of these crops are well established in the scientific literature, yet the amount attributable to biofuels and the RFS Program specifically remains less understood. Soil quality impacts are highest when land in perennial cover is converted to annual crop production. Simulations using the EPIC (Environmental Policy Integrated Climate) model estimate that satellite-derived conversions of 4.2 million acres of grassland to various assumed agricultural scenarios negatively affected soil quality across a 12-state U.S. Midwestern region, increasing erosion by -0.9–7.9%, nitrogen loss by 1.2–3.7%, and soil organic carbon loss by 0.8–5.6%. The range in losses depended upon the assumed tillage practices, with no-till at the low end and conventional tillage at the high end of the range of effects. As noted above from Chapter 6, an estimated 0 to 20% of cropland expansion is estimated to be associated with corn ethanol production from the RFS Program historically, with larger attributable effects if other biofuels (e.g., soybean biodiesel) were included quantitatively and smaller effects in years with smaller effects from the Program. Nevertheless, applying these percentages to the modeling results yields estimates from zero to relatively small negative soil quality effects. Thus, the effects of the RFS Program on soil quality are likely comparatively small in magnitude relative to that of cropland over a large, multistate region or the contiguous United States, yet may be more important at local scales. Additional conservation measures—such as further adoption of conservation tillage and cover crops—would help reduce the impacts on soil quality of biofuels generally and the potential impacts of the RFS Program specifically.

Water Quality [Chapter 10]

As with soil quality, effects on water quality continue to be from cultivation of corn and soybean, with well established relationships between water quality and these crops generally, and less established relationships with biofuels and the RFS Program specifically. Trends in total nitrogen (TN) and total phosphorus (TP) from the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) from 2002 to 2012 show that both are likely decreasing in the central Midwest where conservation tillage practices have increased, and both are likely increasing in the areas of cropland expansion in western and northern Midwest where such practices are less common. Although TN and TP concentrations may be improving in some locations, trends in nutrient condition[17] from the EPA's comprehensive National Aquatic Resource Surveys (NARS) are less conclusive, with little change in stream TN condition and many areas worsening in stream TP condition. Simulations using the Soil & Water Assessment Tool (SWAT) in the Missouri River Basin estimated that for TN and TP loads and concentrations, satellite-derived grassland conversion to continuous corn would result in the greatest increase in TN and TP loads (6.4% and 8.7% increase, respectively); followed by conversion to corn/soybean rotation (TN increased 6.0% and TP increased 6.5%); and then conversion to corn/wheat rotation (TN increased 2.5% and TP increased 3.9%). As with soil quality, the effects from cropland expansion potentially attributable to the RFS are estimated to be roughly 0–20% of these. These estimated increases are relatively small on an absolute basis considering this basin is already intensively cultivated but aggravate impacts in watersheds already affected by nutrients. Lifecycle potential eutrophication effects for both corn ethanol and soybean biodiesel are higher than their fossil fuel counterparts (gasoline and diesel, respectively) per unit energy and in total in most cases, although these analyses do not include many factors and may underestimate the effects from petroleum.

Water quality considerations are not just from farming activities, but also from potential leakages from underground storage tank (UST) systems, which may be affected by increased concentrations of biofuels. Most older, and even some newer, existing UST systems are not fully compatible with higher blends of ethanol (e.g., E15, E85) and may require modification before storing them. For example, the actual tank is often compatible with E15, but some of the other system components may not be.

[17] While nutrient concentration is the estimated concentration of nutrients in the water, nutrient condition refers to the concentration relative to region-specific reference water bodies that are relatively unpolluted. Nutrient condition in the NARS is often categorized as "good", "fair", and "poor." Thus, nutrient concentration may improve, but not enough to change nutrient condition classes.

Water Use and Availability [Chapter 11]

National-level impacts to date on water use and availability may be relatively limited as only 10–14% of soybean and corn acreage is irrigated, but those impacts may be important regionally and are an additional pressure on already stressed water resources such as the High Plains Aquifer (HPA). Most water withdrawals in the United States are for thermoelectric power (41%) followed by irrigation (37%). And, while most corn and soybean acreages are rainfed (86% and 90%, respectively), nearly 40% of water withdrawals for irrigation are for these two crops. Almost all of the irrigated corn is in the western corn belt where much of the observed cropland expansion has occurred. Water use and water availability impacts related to biofuels are primarily due to irrigation of feedstocks (88–99% across the lifecycle), while water use in biorefineries represents a small (1–9%) and declining percentage of lifecycle water use as biorefinery production efficiencies improve. Nevertheless, lifecycle estimates suggest that corn ethanol requires an average of 13 times more water per gallon of fuel produced compared to gasoline, ranging from roughly break-even with gasoline (at 8.7 gallons per gallon fuel) under rainfed conditions and efficient conversion facilities, to greater than 100 times more water requirements under irrigated and less efficient conversion facilities.

Terrestrial Ecosystem Health and Biodiversity [Chapter 12]

Effects on terrestrial ecosystems, particularly terrestrial biodiversity and possibly threatened and endangered species, continue to be primarily from corn and soybean feedstock production, with the two main drivers of effects being shifts in perennial cover to corn and soybeans and associated agronomic practices. The USDA NRI estimates that almost half of the lands shifting to corn and soybeans from 2002 to 2017 were previously under perennial cover (e.g., grasses on CRP land, pasture). Satellite-derived data suggest grasslands account for 88% of land in perennial cover that were converted to annual crops between 2008 and 2016, while 3% and 2% were from wetlands and forests, respectively. These shifts in perennial cover may negatively impact grassland birds, bats, pollinators and other beneficial insects, and plants, including threatened and endangered species. Across the contiguous United States, 27 terrestrial threatened and endangered species had an estimated 10 acres or more of non-cropland conversion to corn or soybeans within 1-mile of its critical habitat between 2008 and 2016. Of those, six threatened and endangered species had estimated conversion of 10 acres or more within their designated critical habitat. Ancillary datasets such as from the USDA National Agriculture Imagery Program (NAIP) are needed to verify these estimates. These impacts are from land conversion to agriculture and cannot be attributed to the RFS Program specifically because the range of the impact of the RFS Program on corn ethanol consumption includes zero and because methods to explicitly link the RFS Program with individual parcels of land do not currently exist. Overall, the range of possible impacts from the RFS Program likely spanned from no effect to a negative effect on terrestrial biodiversity historically (2008 to 2016). The magnitude of any impacts is uncertain and may be relatively small compared to that of total U.S. cropland, but may still be important for locally endemic species and other important local environmental resources. It is unknown whether these relatively small changes in land cover and land management may or may not cross ecological thresholds for various habitats and species. Whether these effects historically were adverse or not in the context of the ESA is unknown. Notably, these findings do not necessarily apply for years beyond 2016, when the effects of the RFS Program on corn ethanol and soy biodiesel production may have changed.

Aquatic Ecosystem Health and Biodiversity [Chapter 13]

As with other environmental effects, the primary impacts to date on aquatic ecosystems are from the conversion of grasslands to corn and soy production, which often lead to increased sediment, pesticide, and nutrient loads to aquatic ecosystems. Although the estimated effects from the RFS Program are not likely to shift current biological conditions, they are estimated to be an additional stress on already stressed ecosystems. As reported in the water quality chapter [Chapter 10], although nutrient concentrations and loads in certain areas of the Upper Midwest are estimated to be improving from the USGS NAWQA, these improvements do not appear to be sufficient to lead to improvements in stream biological conditions (e.g., fish, macroinvertebrates). For pesticides, potential harm to aquatic life was indicated by exceedances of benchmarks for several pesticides used in row crop production, especially neonicotinoid insecticides widely used as coatings on corn seeds. Based on data from nationally representative surveys of the nation's wadeable stream miles in 2004 and about 10 years later in 2013–2014, biological condition generally worsened between the two surveys, although there was wide regional variation in the response. In the SWAT study in the Missouri River Basin (MORB) introduced in Chapter 10, the flow-weighted nutrient concentrations increased by less than 5% on average across the MORB from estimated agricultural expansion from 2008 through 2016. Thus, increases in nutrient concentrations that may be attributable to the RFS Program are unlikely to result in *new exceedances* of current state numeric nutrient criteria (where available) in agricultural regions of the United States. However, most watersheds already experience exceedances of multiple stressors; thus, additional nutrients aggravate stream condition even if only by a small amount. For example, according to national surveys conducted by the EPA, 67% of the wadeable streams in the United States were already in poor or fair biological condition as of 2004. Many states have no numerical criteria with which to compare these concentrations. This SWAT analysis did not assess pesticides which are difficult to accurately characterize due to the large variety of pesticides to potentially model. Total effects may be larger or smaller because this study only included effects from agricultural expansion (expected to be the largest source) and not agricultural intensification or recent improvements in tillage practices. Nonetheless, the potential effects from the RFS Program may be contributing to additional strain to aquatic ecosystems, potentially exacerbating harmful algal blooms and hypoxia events. There were 78 aquatic threatened and endangered species that had an estimated 10 acres or more of non-cropland conversion to corn or soybeans within 1 mile of their critical habitat between 2008 and 2016. As discussed in Chapter 12, these cannot be attributed to the RFS Program specifically; thus, the range of possible impacts from the RFS Program likely spanned from no effect to a negative effect on aquatic biodiversity historically (2008 to 2016).thus, the range of possible impacts from the RFS Program likely spanned from no effect to a negative effect on aquatic biodiversity historically (2008 to 2016).

Wetland Ecosystem Health and Biodiversity [Chapter 14]

Although cropland expansion from 2008 through 2016 is estimated to be mostly of grasslands and not of wetlands, some additional losses of wetland acreages are estimated in ecologically sensitive areas which had already experienced significant losses before the inception of the RFS Program. Since 2007, the nation has lost 120.3 thousand acres of palustrine (marsh-like) wetlands and gained 205.9 thousand acres of lacustrine (lake-like) wetlands in the conterminous United States. The diverse wetlands within these broad classes support different species and perform different ecosystem functions. Lacustrine habitats are generally deeper, less vegetated, and more permanently ponded, providing ecological functions similar to lake ecosystems. Palustrine habitats, on the other hand, are shallower, have dense emergent vegetation, generally greater biodiversity, and undergo periodic drying that enhances biogeochemical processes such as denitrification. In the palustrine class, small, seasonal wetlands are being lost at a faster rate, though the direct effect from biofuels generally or the RFS Program specifically cannot be determined from available surveys. Although cropland expansion from 2008–2016 was mostly from conversion of grassland (88%), 3% was estimated from reclamation of wetlands, totaling nearly 275,000 acres of wetlands concentrated in the Prairie Pothole Region. A percentage of this (0–20%) may be attributable to the RFS Program. This acreage is small compared with historical losses of wetlands but could have cumulative environmental effects or landscape level effects in some areas. For example, by 2004—the year before enactment of the Energy Policy Act—over half of the historical wetlands in the lower 48 states had already been lost (>100 million acres lost), with several Midwestern states losing more than 80% of their historical wetlands. Given currently available datasets, which wetlands specifically may have been converted as a result of the RFS Program cannot be accurately estimated. Unlike other waterbird species, commercially valued waterfowl (ducks, geese, swans) as a group have not experienced national declines over the past decade, possibly due to a positive response to availability of food (grains) and habitat from interspersed lake-like wetlands and agricultural fields along migration routes. While national trends in status of wetland resources document large-scale transitions from palustrine wetlands toward more lake-like, lacustrine conditions, federal and state programs are having a positive influence on wetland conservation. The USDA has multiple programs focused on conserving and enhancing wetlands on agricultural lands, including the USDA Natural Resources Conservation Service's (USDA-NRCS) Agricultural Conservation Easement Program and the North American Wetlands Conservation Act (NAWCA) grant program, which has contributed to the protection, restoration, and enhancement of approximately 30.7 million acres total of wetlands and associated upland habitats since 1991.

Invasive or Noxious Plant Species [Chapter 15]

Impacts to date on the environment from the cultivation of invasive or noxious plant species as biofuel feedstocks have not been observed, but cultivation practices of corn and soybean feedstocks have likely contributed to the increasing incidence of herbicide-resistant weeds. Currently, most biofuel is produced from a small number of non-invasive feedstock species (i.e., corn, soybean) and therefore do not pose risk of invasion directly. However, impacts from the cultivation practices of corn and soybeans on the evolution of herbicide-resistant weeds do exist, although it is unclear to what extent impacts can be attributed to corn and soybeans grown to meet either biofuel demand generally or the specific requirements of the RFS Program. While potential impacts have been identified using weed risk assessment for some newer feedstocks being considered, none are currently used to produce biofuels and there are practices available for their mitigation (e.g., registration, reporting, and record keeping requirements).

International Effects [Chapter 16]

Direct international effects from the RFS Program attributable to biofuel imports from other countries could not be quantified in the RtC3. Although the United States imported biofuels from several regions that are biologically diverse, these amounts of imported biofuel were small and relatively short-lived, with the United States transitioning to being a net exporter of biofuels that may actually reduce environmental effects overseas. It does not necessarily follow that overall international effects of the RFS Program have been small, as research has shown the indirect effects of increased biofuel production on feedstock commodity trade flows could be substantial. Combining published simulation modeling estimates of the non-U.S. land use change effects of biofuels and estimates for the effect of the RFS Program on corn ethanol (Chapter 6) yields an illustrative range of the effect of the RFS Program on non-U.S. cropland area of 0 to 1.6 million acres. The estimated effect of the RFS Program does not yet include effects on soy biodiesel. As more data become available and are analyzed, historical relationships among U.S. biofuel policies, production, trade, environmental indicators, and other variables may be clarified and uncertainties reduced. Estimating indirect land use change (ILUC) overseas remains one of the most challenging areas of biofuels research. Most simulation models may capture U.S. policies adequately, but offer only a simplified view of the dozens of biofuel policies in other countries, and are based on many parameters that have not been thoroughly and transparently evaluated. The Model Comparison Exercise (MCE) conducted by OTAQ may help improve these models and their estimates of ILUC by identifying priority areas for research.[18] The United States was a net importer of ethanol from 2004 to 2007, mostly but not entirely originating from Brazil. The United States transitioned to a net ethanol exporter as the domestic biofuel industry matured. For biodiesel the trends were different. After a period of little biodiesel trade from 2002 to 2006, the United States was a net exporter of biodiesel from 2007 to 2012, and since has transitioned to be a net importer after ethanol reached the E10 blend wall in roughly 2013 and the advanced biofuel mandate continued to increase. Biodiesel imports from 2013 to 2017 were primarily of soybean biodiesel from Argentina, and to a lesser extent from FOGs and palm oil from Southeast Asia, and biodiesel from Canada. After 2017, total biomass-based imports of biodiesel have declined significantly and stopped from Argentina, and since then are predominantly biodiesel and renewable diesel from Southeast Asia and Canada. There are important uncertainties that remain, for example surrounding the potential for low-cost palm oil from ecologically sensitive areas in Southeast Asia to "backfill" diverted soybean oil from international vegetable oil markets, and especially if RFS Program total biofuel mandates increase in the future. These effects from the RFS Program, however, may be small, as palm oil is affected by many regions and markets, predominantly developing Asian markets, only a fraction of which directly intersect with the U.S. biofuels industry.

Specific Conclusions: Likely Future Effects

EISA requires the EPA to also examine the "likely future" effects of the RFS Program, which for this report is interpreted out to roughly 2025, presuming current likely future technologies, rates of market penetration, current policy, and market dynamics. The likely future effects from the RFS Program were published in the Final Set Rule for 2023–2025 (docket # EPA-HQ-OAR-2021-0427) and projected an increase in 2025 due to the RFS Program over the baseline (with no RFS Program) of 3.9 billion gallons (Chapter 6, Table 6.12). This increase in 2025 from the RFS Program is primarily from increases in biodiesel and renewable diesel from soybean oil (+1.5 billion gallons), increases in cellulosic biofuel from CNG-LNG biogas (+932 million gallons), and increases in corn ethanol (+787 million gallons).

[18] See the docket #EPA-HQ-OAR-2021-0427 and the EPA workshop on GHG modeling (https://www.epa.gov/renewable-fuel-standard-program/workshop-biofuel-greenhouse-gas-modeling) for additional information.

Domestic production and consumption of other biofuels are expected to change fairly little by comparison. These estimated effects in 2025 from the RFS Program are different from the trends through time from 2022 to 2025. Though highly uncertain, EPA determined in its ESA biological evaluation for the Set Rule that the RFS-attributable volumes could potentially lead to an increase of up to 2.65 million acres of cropland by 2025. While the projected cropland expansion for 2023–2025 is slightly larger than the estimated historical cropland expansion from the RFS Program (2.65 versus 1.9 million acres), there is uncertainty whether any particular area will be impacted, due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes. Other sources of uncertainty also remain, including ongoing recovery from the global COVID-19 pandemic, uncertainty in the penetration of E15 in the marketplace, uncertain growth of cellulosic ethanol production from agricultural or marginal lands, and complex transportation market dynamics, among other factors [Chapter 2, section 2.3.2; Chapter 6, section 6.5]. As policy and market conditions change, so may the factors to consider and the estimate of the likely future effects of the RFS Program.

Uncertainties and Limitations

Although much information is presented in this report on the impacts to date and likely future impacts from the RFS Program, there are many uncertainties and limitations that remain. These come in many forms, including data limitations, modeling limitations, and other sources of uncertainty. Data and modeling limitations are numerous and include a current inability to link RFS-attributable biofuels to specific areas, as well as a lack of detailed data through time (e.g., annually) and space (e.g., county-level or smaller) on many practices such as conservation tillage, cover crops, pesticide application rates, and others. Other data limitations include a lack of data to track crops from the farm to the biorefinery, which often travel through an intermediary (e.g., a grain elevator or crusher). Another central limitation is the inherent difficulty of using remote sensing data to correctly assign grass-covered land to the correct land use (e.g., pasture, CRP, grassland, idle), as well as the various and often inconsistent definitions of different land covers among available datasets and publications. The models available also have important limitations, including a lack of industry detail both domestically and abroad, a lack of contributing biofuel and agricultural policies in other countries in driving biofuel production around the world, and a lack of examination of many biofuels that are actually emerging as opposed to the biofuels that were anticipated to emerge following EISA (e.g., cellulosic biofuels on marginal lands). There are other limitations as well, including the inherent challenges of projecting into the future. Notwithstanding these significant uncertainties and limitations, this report makes significant progress towards a better understanding of the impacts to date and likely future impacts of the RFS Program to a broad range of environmental and resource conservation issues.

Recommendations

- Additional research is needed to link the quantities of biofuels estimated attributable to the RFS Program in this report to specific local changes in land cover and land management. This linkage would enable more explicit quantification of the impacts to date of the RFS Program and facilitate informed assessments of the likely future effects of the RFS Program.
- Conservation practices exist to offset many of the environmental effects from the cultivation of conventional biofuel feedstocks (e.g., corn, soybean) and agricultural effects more generally; and, while some of these have been widely adopted (e.g., conservation tillage), some have not (e.g., cover crops). A sustained effort to deploy these practices across a wider area, especially in areas of recent cropland expansion may be needed to offset the potential negative effects from the RFS Program specifically and biofuels more generally.
- Additional research is needed to better understand the several other complex uncertainties that remain, including the effects from the RFS Program on biofuels other than corn ethanol, the potential for palm oil and other low-cost oils to "backfill" soybean oil diverted toward biofuels, improvements in the skill of many remote-sensing datasets in quantifying grassland conversion, better data on where and which conservation practices are in place across the landscape, and others discussed above and in more detail throughout this report.
- More research overall on the environmental effects from the emerging biofuels is needed given that the mix of emerging biofuels may not have the same effects as the biofuels that were historically dominant.

Part 1

1. Introduction

1.1 Legislative and Regulatory Background

In August 2005, Congress enacted the Energy Policy Act of 2005 (EPAct),[1] which included the creation of the Renewable Fuel Standard (RFS) Program to be administered by the Environmental Protection Agency (EPA). The RFS Program required that the amount of biofuel mixed into the gasoline pool in the United States be 4 billion gallons in 2006 and increase to 7.5 billion gallons by 2012. In December 2007, Congress enacted the Energy Independence and Security Act (EISA) with the stated goals of "mov[ing] the United States toward greater energy independence and security [and] to increase the production of clean renewable fuels."[2] In accordance with these goals, Section 202 of EISA revised the RFS Program to increase the volume of renewable fuel required to be blended into transportation fuel to 36 billion gallons per year by 2022. The two versions of the RFS Program under the EPAct and EISA are commonly called the "RFS1" and "RFS2," respectively.[3] In addition, EISA created a new requirement under Section 204 for EPA to examine the environmental and resource conservation impacts of the RFS Program. The purpose of this report is to meet the requirements of Section 204. Section 204 states:

> *"(a) In General. Not later than 3 years after the enactment of this section and every 3 years thereafter, the Administrator of the Environmental Protection Agency, in consultation with the Secretary of Agriculture and the Secretary of Energy, shall assess and report to Congress on the impacts to date and likely future impacts of the requirements of Section 211(o) of the Clean Air Act on the following:*
>
> 1. *Environmental issues, including air quality, effects on hypoxia, pesticides, sediment, nutrient and pathogen levels in waters, acreage and function of waters, and soil environmental quality.*
> 2. *Resource conservation issues, including soil conservation, water availability, and ecosystem health and biodiversity, including impacts on forests, grasslands, and wetlands.*
> 3. *The growth and use of cultivated invasive or noxious plants and their impacts on the environment and agriculture.*

[1] Energy Policy Act, 2005

[2] Energy Independence and Security Act, 2007

[3] The RFS1 was in effect from 2006 to 2008 and the RFS2 was in effect from 2010 to current. 2009 was a transition year between programs, where the RFS2-volumes were applied, but to a single total renewable fuel standard like the RFS1. For convenience, because the RFS2 volumes applied, 2009 is denoted as being under the RFS2.

In advance of preparing the report required by this subsection, the Administrator may seek the views of the National Academy of Sciences or another appropriate independent research institute. The report shall include the annual volume of imported renewable fuels and feedstocks for renewable fuels, and the environmental impacts outside the United States of producing such fuels and feedstocks. The report required by this subsection shall include recommendations for actions to address any adverse impacts found.

This text defines the statutory scope of the Section 204 report series, both in terms of what is included and what is omitted, in particular in relation to greenhouse gases (GHGs). The details of how this statutory language is translated into a scientific report is discussed in Chapter 2.

To better understand an assessment of the environmental and resource conservation impacts of the RFS Program, it is helpful to briefly introduce some basic components of the Program, which include concepts and terms that will be referred to throughout the report. EPAct established in RFS1 a single total annually increasing renewable standard, or volume requirement, for 2006 through 2012 (Table 1.1). Actual biofuel production greatly outpaced the RFS1 standards, and more ambitious standards were established in the RFS2 by EISA (Table 1.1, Figure 1.1).[4] In contrast to a single renewable fuel standard under the RFS1, EISA created renewable standards for four categories of biofuel that began in different years: cellulosic biofuel (2010–2022), biomass-based diesel (2009–2012), advanced biofuel (2009–2022), and total renewable fuel (2006–2022). To implement the RFS1 and RFS2, each year EPA has promulgated annual rules that translate the renewable standards into annual percentage standards.[5] These annual percentage standards are called Renewable Volume Obligations (RVOs) and indicate the volume of biofuel that refiners or importers of gasoline or diesel must blend into transportation fuel in a given year.[6] RVOs may be different from the statutory renewable standard for many reasons, such as changes in market conditions, waivers, or other reasons.

[4] Clean Air Act, 2022f
[5] The CAA requires that EPA set percentage standards in advance of the year they apply (Clean Air Act, 2022a). For various reasons, some RVOs were finalized after the year to which they applied. *See* “Renewable Fuel Standard Program: Standards for 2014, 2015, and 2016 and Biomass-Based Diesel Volume for 2017,” 80 FR 77420 (Dec. 14, 2015).
[6] Obligated parties (refiners or importers of gasoline or diesel) may also demonstrate compliance by purchasing Renewable Identification Numbers (RINs), which function as credits that correspond with produced volumes of biofuel.

Table 1.1. Annual biofuel volumes in the statutes and final rules through time (billion gallons). For the RFS2 these are set for cellulosic biofuel (CB), biomass-based diesel (BBD), advanced biofuel (AB), and total renewable fuel (TRF). Also shown is the implied standard for conventional biofuel (CVB, gray shading), which is mostly corn ethanol in the United States. CVB is the difference between total and advanced biofuels (i.e., TRF – AB).

			RFS2									
	RFS1		EISA					Final Rule				
Year	EPAct	Final Rule	CB	BBD	AB	TRF	CVB	CB	BBD	AB	TRF	CVB
2006	4	4[a]										
2007	4.7	4.7										
2008	5.4	5.4	NA	NA	NA	9.0	NA					
2009	6.1	NA	NA	0.5	0.6	11.1	10.5	NA	NA	NA	11.1[b]	11.1
2010	6.8	NA	0.1	0.65	0.95	12.95	12.0	0.007	1.15[c]	0.95	12.95	12.0
2011	7.4	NA	0.25	0.8	1.35	13.95	12.6	0.00	0.80	1.35	13.95	12.6
2012	7.5	NA	0.5	1.0	2.0	15.2	13.2	0.00	1.00	2.00	15.2	13.2
2013			1.0	[d]	2.75	16.55	13.8	0.001	1.28	2.75	16.55	13.8
2014			1.75	[d]	3.75	18.15	14.4	0.033	1.63	2.67	16.28	13.61
2015			3	[d]	5.5	20.5	15.0	0.123	1.73	2.88	16.93	14.05
2016			4.25	[d]	7.25	22.25	15.0	0.230	1.90	3.61	18.11[e]	14.5[e]
2017			5.5	[d]	9.0	24.0	15.0	0.311	2.00	4.28	19.28	15.0
2018			7.0	[d]	11.0	26.0	15.0	0.288	2.10	4.29	19.29	15.0
2019			8.5	[d]	13.0	28.0	15.0	0.418	2.10	4.92	19.92	15.0
2020			10.5	[d]	15.0	30.0	15.0	0.51[f]	2.43	4.63[f]	17.13[f]	12.5[f]
2021			13.5	[d]	18.0	33.0	15.0	0.56[f]	2.43	5.05[f]	18.84[f]	13.79[f]
2022			16.0	[d]	21.0	36.0	15.0	0.63[f]	2.76[f]	5.63[f]	20.63[f]	15.0[f]
2023[g]								0.84	2.82	5.94	20.94[f]	15.0[f]
2024[g]								1.09	3.04	6.54	21.54	15.0
2025[g]								1.38	3.35	7.33	22.33	15.0

[a] EPA promulgated a direct final rule on December 30, 2005 (U.S. EPA, 2005) which implemented the statute's default 2.78% standard on a collective compliance basis, or 4 billion gallons.
[b] In 2009 EPA set annual volumes only for total renewable fuels.
[c] The 2009 and 2010 BBD volume requirements were combined.
[d] To be determined by EPA each year, but no less than 1.0 billion gallons.
[e] EPA used its various waiver authorities to reduce the required volume of total renewable fuel for 2016. In *Americans for Clean Energy v. EPA*, 864 F.3d 691, (2017) the court held that EPA had improperly exercised its general waiver authority and remanded the 2016 rule to EPA.
[f] EPA issued a final rule for these standards on June 3, 2022 (EPA-HQ-OAR-2021-0324). EPA also established a 250-million-gallon "supplemental obligation" to the volumes finalized for 2022 to address the remand of the 2014–2016 annual rule by the DC Circuit Court of Appeals in *Americans for Clean Energy v. EPA*. (U.S. EPA, 2022) That supplemental obligation was also applied in 2023.
[g] CAA 211(o) does not impose any statutorily specified volumes after 2022. (2022b) EPA issued required volumes in "Renewable Fuel Standard (RFS) Program: Standards for 2023-2025 and Other Changes" on June 21, 2023 (docket EPA-HQ-OAR-2021-0427).

Because EISA was enacted in December 2007, the standards for 2007 and 2008 were still based on the RFS1 volume targets and set to 4.7 and 5.4 billion gallons, respectively (Figure 1.1). The first applicable standard promulgated under RFS2 was in 2009 and increased total renewable fuel significantly from 6.1 to 11.1 billion gallons.[8] The four standards are nested in a way such that one gallon of a specific type of biofuel can contribute towards meeting multiple standards (Figure 1.2). However, corn ethanol is prohibited by statute from being considered an advanced biofuel,[9] and thus can only be considered a "conventional" renewable fuel, which must meet the minimum 20% required reduction in lifecycle GHGs relative to petroleum to qualify as a renewable fuel under the RFS Program.[10] Most corn ethanol is produced from facilities that were constructed or began construction prior to December 17, 2007,[11] and so were "grandfathered" as qualifying as a renewable fuel even if they do not meet the 20% lifecycle GHG reduction threshold.[12] In comparison, cellulosic biofuel must meet a 60% lifecycle GHG reduction

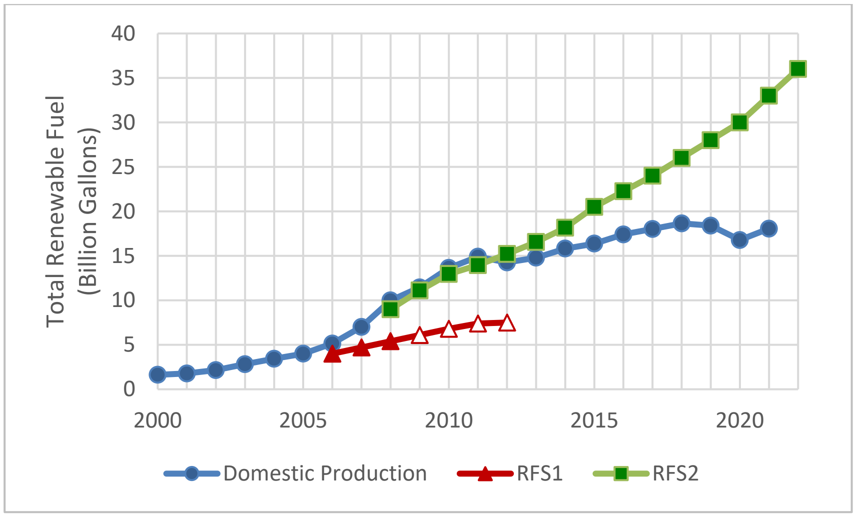

Figure 1.1. RFS1 and RFS2 legislative mandates. Shown are the statutory volume requirements from the RFS1 and RFS2 for total renewable fuels, compared to actual total renewable fuel production from 2000–2022. Sources: EIA and EPA for actual production,[7] EPAct and EISA for RFS1 and RFS2, respectively. Closed circles for RFS1 and RFS2 indicate the year that version of the RFS was in effect, open circles represent a year that version of the RFS was superseded by the other version.

[7] Total renewable fuel here is the sum of ethanol, biodiesel, renewable diesel, biogas, and other biofuels. Data sources for individual biofuels are described in Figure 1.3.
[8] See U.S. EPA, 2008 and Clean Air Act, 2022e.
[9] Clean Air Act, 2022c
[10] Clean Air Act, 2022d; Detailed assessment of the GHG balance of corn ethanol and other biofuels are not in the scope of the RtC3. This is discussed further in Chapter 2, Box 2.2, and see Federal Registry (FR) FRL–9307–01–OAR, and https://www.epa.gov/renewable-fuel-standard-program/workshop-biofuel-greenhouse-gas-modeling.
[11] EPA tracks compliance with the RFS Program through Renewable Identification Numbers (RINs), which are credits used for compliance. EPA batch data show that in 2018, 88% of RIN-generating corn ethanol production (13.2 billion gallons of 15 billion gallons total) was produced under the grandfathering provisions. For more information on RINs see https://www.epa.gov/renewable-fuel-standard-program/renewable-identification-numbers-rins-under-renewable-fuel-standard
[12] Clean Air Act, 2022d; Biofuel production facilities that commenced construction prior to December 19, 2007 are generally exempt from the lifecycle GHG reduction requirements and may generate conventional biofuel RINs even if they do not meet the required GHG reductions. Any new facilities or facility expansions that commenced construction after this date generally are subject to the GHG reduction requirements to generate RINs.

threshold and advanced biofuel and biomass-based diesel must meet a 50% lifecycle GHG reduction threshold.[13] These GHG emissions estimates are based on a lifecycle assessment that includes both direct emissions and indirect emissions, such as from land use change. EPA is currently reviewing the technical methodology used as an ongoing part of its responsibilities in administering the RFS Program.[14]

No annual volume standards are set for conventional renewable fuel, but the maximum quantity of conventional biofuel that can contribute towards the total renewable fuel volume is the difference between the total renewable fuel standard and the advanced biofuel standard (Figure 1.2). Because the RFS Program is not explicit in setting volume requirements for conventional biofuel, the conventional biofuel volume is commonly called an "implied standard." The volumes of corn ethanol consumed in years when the RFS2 applies are thus indirectly capped rather than mandated in a standard (Table 1.1, CVB columns).

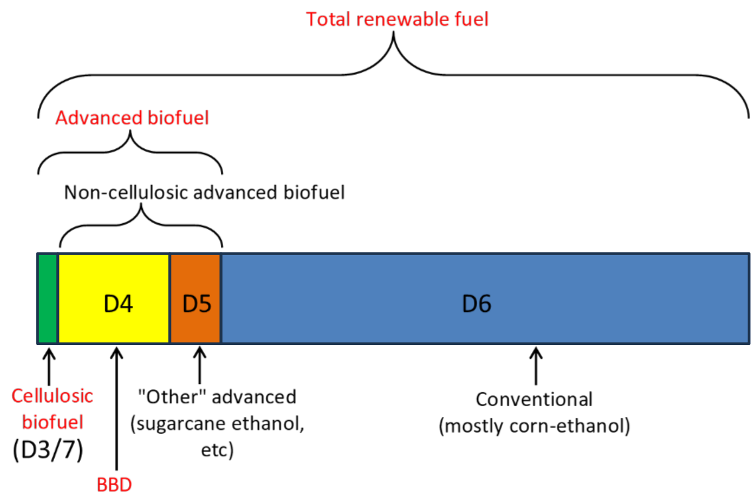

Figure 1.2. The nested structure of the RFS2 standards. Shown are the four volumetric standards under the RFS2 (red text: biofuels for which EPA annually set standards) and other "implied" volumetric standards (black text) in the RFS2, along with the "D-code" for Renewable Identification Numbers (RINs) used to track compliance.

As noted above, while the values in Table 1.1 are expressed as biofuel volumes, the RVOs that apply to obligated parties (refiners or importers of gasoline or diesel)[15] under the RFS Program are percentage standards. The volumes in Table 1.1 are converted to a percentage of total gasoline and diesel production, and each individual obligated party is required to use that percentage of renewable fuels to

[13] Clean Air Act, 2022a
[14] EPA hosted a virtual public workshop on biofuel GHG modeling February 28 - March 1, 2022. The purpose of this workshop was to solicit information on the current scientific understanding of GHG modeling of land-based crop biofuels used in the transportation sector. The meeting was conducted by EPA's Office of Transportation and Air Quality (OTAQ) in consultation with the Department of Agriculture and the Department of Energy. As a next step in this process, EPA will be proceeding with a model comparison exercise. The model comparison exercise will feature models discussed in the workshop and explore further details about these tools. Further details from the model comparison exercise may be found in the Final RFS Standards for 2023, 2024, and 2025 (EPA-HQ-OAQ-2021-0427).
[15] To whom does the Renewable Volume Obligation apply?, 2021

demonstrate compliance with the program. Obligated parties can either blend the renewable fuels themselves or obtain credits (RINs) from other parties that did so.

1.2 Prior Biofuel Reports to Congress

The first triennial report to Congress was completed in 2011 (hereafter the "RtC1") and provided an assessment of the environmental and resource conservation impacts associated with increased biofuel production and use. (U.S. EPA, 2011)Although many impacts had been anticipated by the July 2010 publication cutoff date for the RtC1, few impacts had been actually reported in available peer-reviewed literature at the time. Furthermore, although EISA was passed in 2007, the RFS2 did not fully go into effect with the four renewable fuel standards until March 2010. Thus, the first report was largely forward-looking and evaluated the potential impacts of several assumed future scenarios that were common in the literature. The overarching conclusions of the RtC1 were (1) the environmental impacts of increased biofuel production and use were likely negative but limited in impact; (2) there was a potential for both positive and negative impacts in the future; and (3) EISA goals for biofuels production could be achieved with minimal environmental impacts if best practices were used and if technologies advanced to facilitate the use of second-generation biofuel feedstocks (e.g., corn stover, perennial grasses, woody biomass, algae, waste such as municipal solid waste).

The second report to Congress was completed in 2018 (hereafter the "RtC2") and reaffirmed the overarching conclusions of the RtC1. (U.S. EPA, 2018) The RtC2 noted that the biofuel production and consumption that led to the conclusions of the RtC1 had not materially changed, and that the production of biofuels from cellulosic feedstocks anticipated by EISA had not materialized. Noting observed increases in acreage for corn and soybean production in the period prior to and following implementation of the RFS2, the RtC2 concluded that the environmental and resource conservation impacts associated with land use change were likely due, at least in part, to the RFS Program and associated production of biofuel feedstocks.

This third report to Congress (hereafter the "RtC3") builds from the RtC1 and RtC2 to provide an update on the impacts to date and likely future impacts of the RFS Program on the environment. There are new additions and approaches in the RtC3 relative to earlier reports that are discussed in Chapter 2.

1.3 Biofuel Production, Consumption, and Trade

1.3.1 Biofuel Production and Consumption

Ethanol and biodiesel are the types of biofuels produced in the largest quantities in the United States. However, in recent years the production of other biofuels, such as renewable diesel[17] and biogas used as transportation fuel, have increased. Domestic ethanol production increased rapidly from 2000 to 2011, reaching nearly 14 billion gallons in 2011 (Figure 1.3). Almost all the domestic ethanol production and consumption is of corn starch ethanol. Ethanol production was slightly lower in 2012 and 2013, likely due to decreased corn production in 2012 as the result of drought conditions in much of the United States (Rippey, 2015). Domestic ethanol production once again increased from 2014 through 2018, albeit at a slower rate, reaching just over 16 billion gallons in 2018. Ethanol production decreased to 13.9 billion gallons in 2020, largely due to the COVID-19 pandemic, and increased to 15.0 billion gallons in 2021. One likely reason for the slower rate of growth of ethanol production in the United States from2014–2019, relative to 2000–2011, are challenges associated with the E10 blend wall.[18] The Energy Information Administration (EIA) reported that the E10 blend wall was reached in approximately 2015,[19] though different regions of

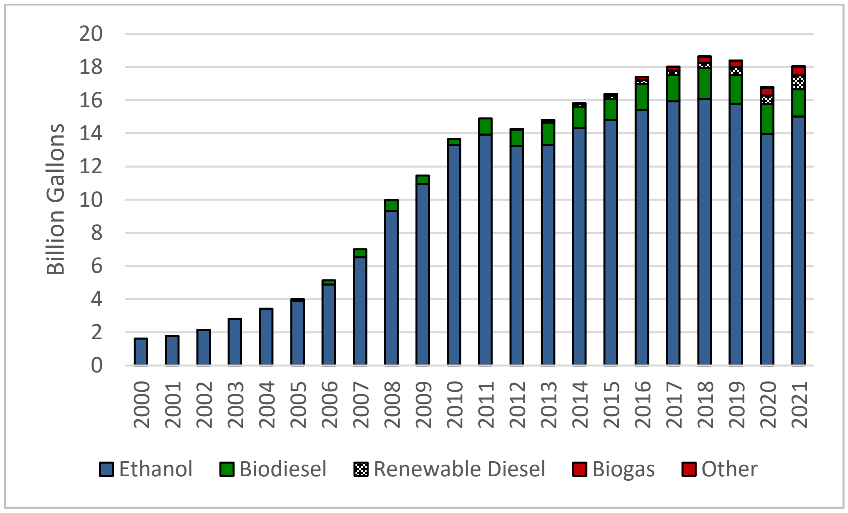

Figure 1.3. Domestic biofuel production from 2000 to 2021.[16]

[16] Data for ethanol and biodiesel from USDA ERS US Bioenergy Statistics (USDA, 2022); ethanol data available in Table 2 and biodiesel data available in Table 4. Renewable diesel, biogas, and other data from EPA's public data for the Renewable Fuel Standard program (USDA, 2022).

[17] Biodiesel is a fatty acid ester generally produced by transesterifying renewable fats or oils. Renewable diesel is a hydrocarbon generally produced by hydrotreating renewable fats or oils. Both fuels are diesel replacements and can qualify as biomass-based based diesel under the RFS program. See the Alternative Fuels Data Center for more information (https://afdc.energy.gov/fuels/emerging_hydrocarbon.html).

[18] The E10 blend wall describes the amount of ethanol that can be blended into the gasoline pool at 10% by volume. Above 10% of the gasoline pool by volume, higher amounts of ethanol consumption domestically would have to come from higher blends where it faces greater economic challenges. E15 (i.e., 15% ethanol by volume) is approved for use in vehicles manufactured after 2000, but remains limited in availability nationally (see Chapters 2 and 3). E85 is widely available but the vehicular fleet that can use E85 remains small, and many owners of flex-fuel vehicles (FFVs) choose to refuel with E10. For example, according to the 2019 edition of EIA's Annual Energy Outlook, total energy consumption in all FFVs in 2018 was 1.56 Quad Btu, while that from E85 used in FFVs was only 0.05 Quad Btu. (DOE, 2019) Thus, E85 use in FFVs represents about 3% of all fuel used in FFVs.

[19] EIA, 2022

the country reached the E10 blend wall at different times. Furthermore, ethanol consumption nationally approached the blend wall as early as 2010 (Figure 1.4). Since then, nearly all gasoline consumed in the United States has contained at least 10% ethanol. The primary markets for increased ethanol production beyond that which can be consumed in E10 blends in the United States have been in foreign countries (i.e., ethanol exports), as volumes of ethanol sold in blends that contain higher levels of ethanol, such as E15 or E85 have remained fairly small.

Domestic production of biodiesel has increased steadily since 2000 (Figure 1.3). In 2018, biodiesel production in the United States reached a record high of 1.86 billion gallons, before declining to 1.64 billion gallons in 2021. Similarly, domestic production of renewable diesel has increased each year since 2012, with production reaching a record high of approximately 840 million gallons in 2021. The feedstocks used for biodiesel and renewable diesel are much more varied than ethanol, with soybean oil, fats/oils/greases (FOGs), and corn oil making up most of the domestic production. Other biofuels have a much smaller share of total production. More details on different feedstocks and pathways are provided in Chapters 2 and 3.

Since 2000, the consumption of biofuels in the United States has grown significantly, rising from less than 2 billion gallons in 2000 to approximately 17.3 billion gallons in 2019, decreasing to 15.62 billion gallons in 2020, and recovering to 17.2 billion gallons in 2021 (Figure 1.4). During this time period ethanol and biodiesel were the types of biofuels consumed in the largest quantities in the United States, with smaller volumes of renewable diesel, compressed natural gas (CNG)/liquified natural gas (LNG) derived from biogas, and other renewable fuels also being consumed as transportation fuel.

Domestic ethanol consumption increased rapidly from 2000 to 2010, reaching nearly 13 billion gallons in 2010 (Figure 1.4). Domestic ethanol consumption continued to increase but at a slower rate from 2011 through 2016, reaching a total of approximately 14.4 billion gallons in 2016. Ethanol consumption remained fairly stable at 14.4–14.5 billion gallons from 2016 through 2019, declined to 12.7 billion gallons in 2020, and increased to 13.9 billion gallons in 2021. The rate of growth in domestic ethanol consumption decreased significantly as total ethanol consumption approached the E10 blend wall between 2011–2013. Domestic consumption of biodiesel has increased through 2016, from approximately 100 million gallons in 2005 to just over 2 billion gallons in 2016 (Figure 1.4). Since 2016, consumption of biodiesel decreased slightly to approximately 1.8 billion gallons in 2021. Consumption of other types of renewable fuels are smaller, but have generally increased from 2000 through 2021. Renewable diesel consumption increased steadily, from approximately 80 million gallons in 2012 to approximately 960 million gallons in 2021. Consumption of CNG/LNG derived from biogas as transportation fuel increased from approximately 30 million gallons in 2013 to over 560 million gallons in 2021.

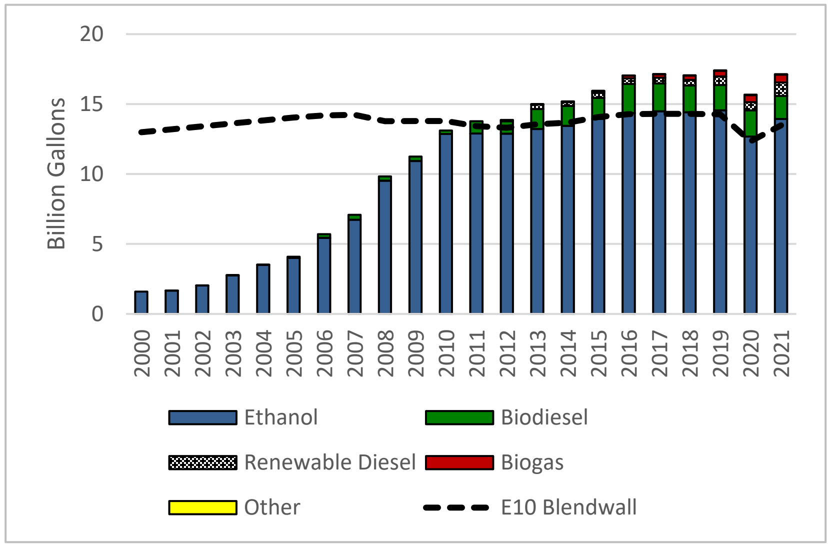

Figure 1.4. Biofuel consumption (bars) from 2000 to 2021 and the estimated E10 blend wall (dashed line). Data sources same as Figure 1.3, E10 blend wall estimated as 10% of the transportation gasoline consumed in that year.[20]

1.3.2 Biofuel Imports and Exports

Biofuel imports into the United States since 2000 have been highly variable, both in terms of the volume of imported biofuel and the types of biofuels that are imported (Figure 1.5). Ethanol imports were relatively high from 2004 through 2008 when domestic consumption typically exceeded domestic production. Ethanol imports decreased significantly thereafter as domestic production of ethanol increased to meet consumption. Ethanol imports increased again in 2012–2013 likely due to the 2012 drought that reduced domestic production. (Rippey, 2015) Imports of ethanol have been relatively low since 2014, as the market has generally looked to non-ethanol biofuels (such as biodiesel and renewable diesel) to satisfy the increasing advanced biofuel requirements of the RFS Program above the E10 blend wall that are not met with domestic production. Prior to 2013, imports of biodiesel and renewable diesel were

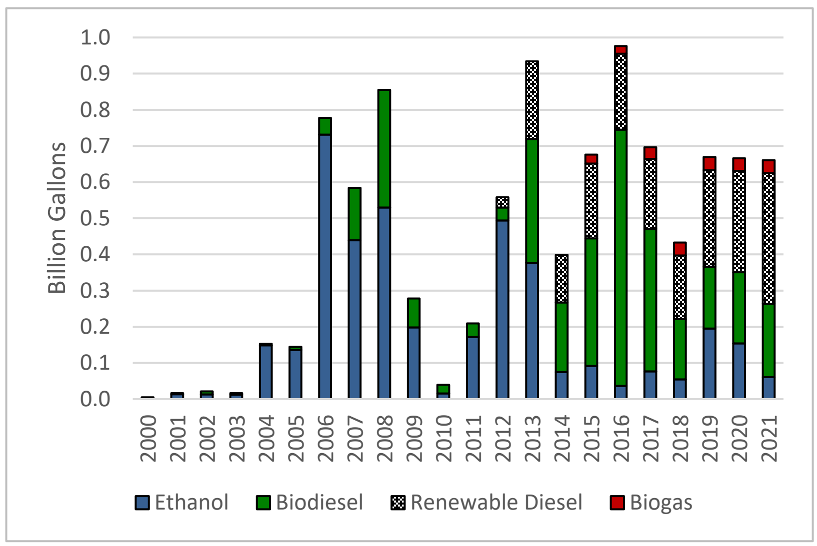

Figure 1.5. Biofuel imports from 2000 to 2021. Data sources same as Figure 1.3.

[20] This estimate may be biased low given that some ethanol is consumed as E15 and E85, but alternatively some consumers prefer E0 which would bias the estimate high. These factors are likely small and thus do not affect the broad trends.

small, with the exception of 2007 through 2009 (see Chapters 7 and 16 for more on temporary trade dynamics of biodiesel for this period). Imports of biodiesel and renewable diesel have been relatively high since 2013. Biodiesel imports have been sourced from a variety of regions, including Argentina, Canada, Europe, and southeast Asia. The vast majority of renewable diesel imports (over 95% from 2012–2021), have been imported from southeast Asia.[21] After reaching its highest levels in 2016, biodiesel imports decreased in response to tariffs on biodiesel imported from Argentina and Indonesia first announced in August 2017.

Biofuel exports have generally increased from 2007 through 2018, before decreasing slightly from 2019 to 2021 (Figure 1.6). Biofuel exports are driven largely by increasing domestic production in excess of domestic consumption (Figure 1.7). Ethanol exports increased rapidly in 2010 and 2011 as domestic ethanol production exceeded domestic ethanol consumption. Ethanol exports dropped in 2012 and 2013 in response to lower domestic ethanol production from the 2012 drought. Ethanol exports increased from 2013 through 2018 before dropping slightly from 2018 through 2021. Biodiesel exports, which were low through 2006, increased significantly in 2007 and 2008 in response to temporary trade factors discussed in Chapters 7 and 16.

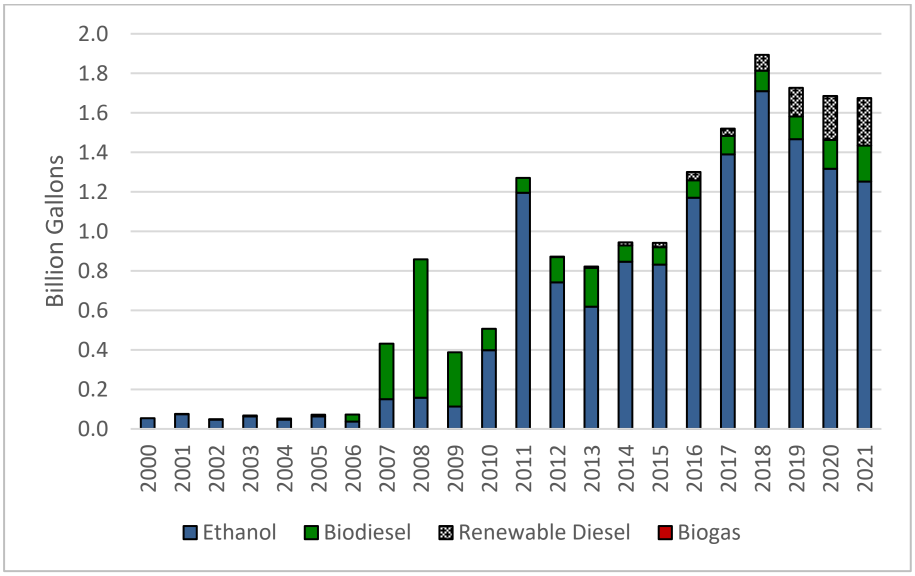

Figure 1.6. Biofuel exports from 2000 to 2021. Data sources same as Figure 1.3.

All of these processes—the production, consumption, and trade of biofuels—are dynamic, responding simultaneously to a large number of policy, economic, and environmental drivers. Examining all of these processes for the two major biofuel types (ethanol and biodiesel/renewable diesel) in the same graphic clearly shows some of the aforementioned trends, including for ethanol the early imports when domestic production had not yet caught up with consumption demand (2004–2008, Figure 1.7) and more recent exports of ethanol when domestic production exceeded consumption after reaching the E10 blend wall (2011, 2014–2021, Figure 1.7). These also show that trade is a relatively small proportion of the

[21] Data on renewable diesel imports (labeled "other renewable diesel" by EIA) from country of origin sourced from EIA (2023).

ethanol dynamics in the United States, as are temporary trade phenomena such as the increases in imports and exports of biodiesel in 2007–2009 (Figure 1.8). Also shown is the drought of 2012 causing increases in imports and decreases in exports (2012, 2013, Figure 1.7, Figure 1.8).

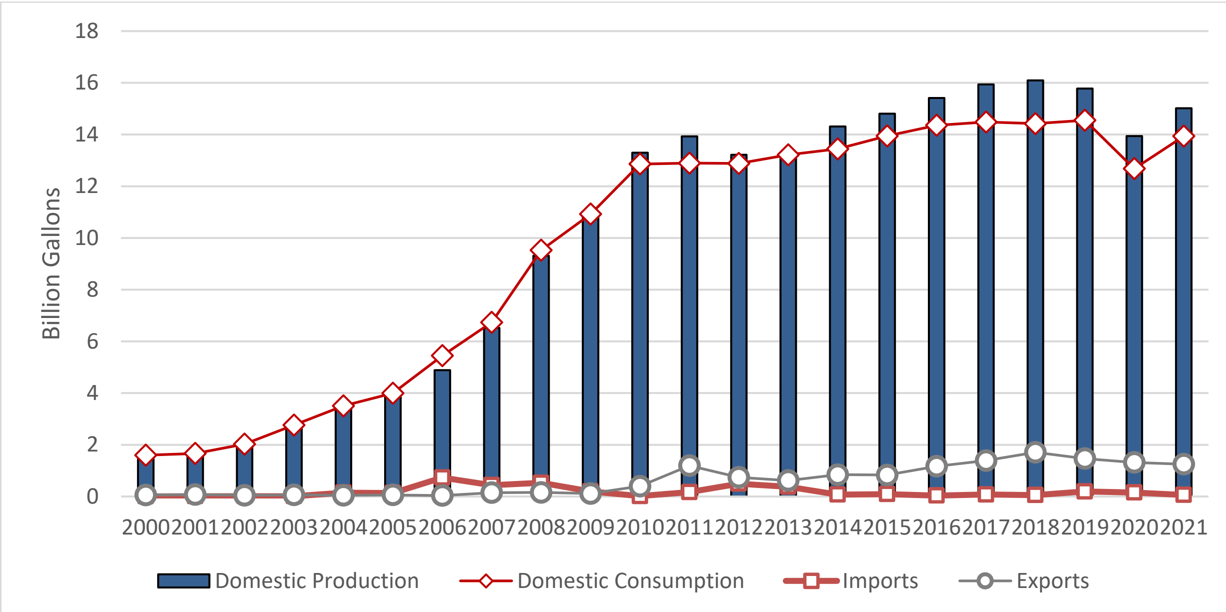

Figure 1.7. Ethanol production, consumption, imports, and exports. Data sources same as Figure 1.3.

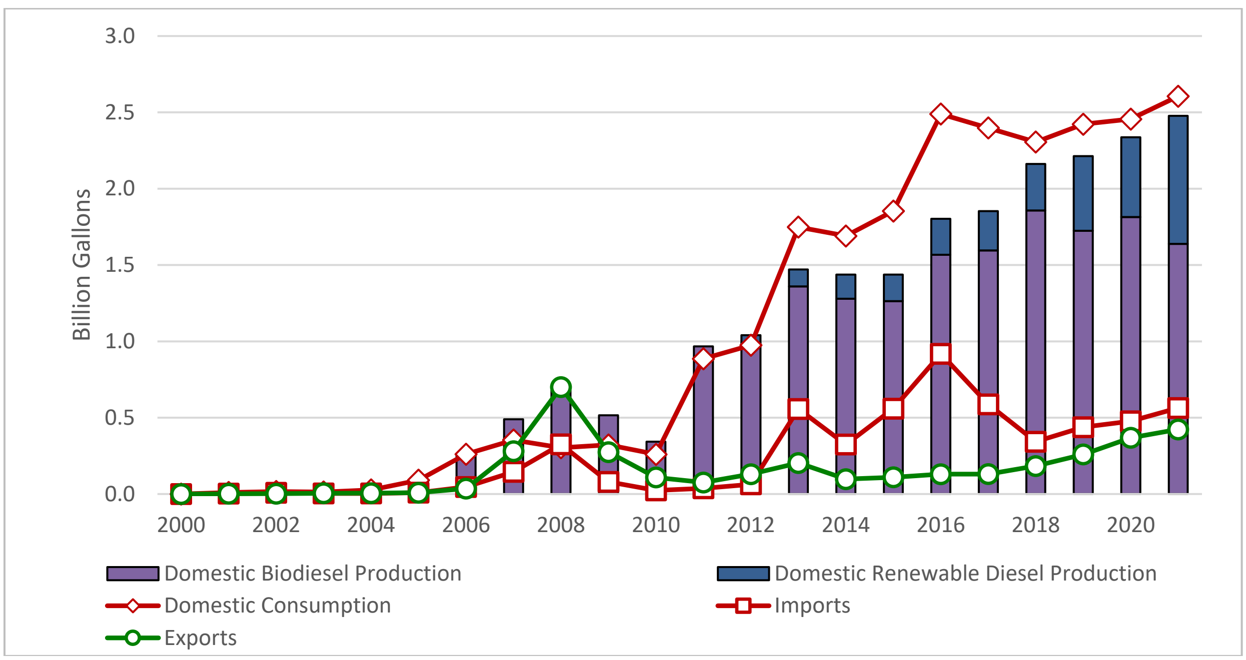

Figure 1.8. Biodiesel and renewable diesel production, consumption, imports, and exports. Data sources same as Figure 1.3.

1.4 Approach of the RtC3

The approach to and organization of the RtC3 is different from the RtC1 and RtC2 in several ways, and it is helpful to understand the reasoning behind these distinctions. As explained previously, the RtC1 came out in 2011 and was largely forward-looking because many of the environmental implications of the program, although anticipated, had not yet materialized (U.S. EPA, 2011). In the RtC2, EPA sought to identify any major changes that had occurred since the RtC1 and to lay the foundation for the RtC3. The emphasis in the RtC2 was on whether there were observed, substantive changes since 2011 in the major drivers of environmental impacts, including feedstock volumes and types. The RtC2 presented considerable new information that drew from literature and governmental reports published since 2011 regarding observed shifts in land use change domestically, given the importance of land use change on the environmental and resource conservation effects listed in EISA. In the RtC2, EPA found that a critical knowledge gap existed in understanding the attribution of land use changes and other environmental effects to the RFS Program itself, as opposed to biofuels and agricultural markets generally. Many past and ongoing changes in the linked agricultural-biofuel industry are due to other factors in addition to the RFS Program. Confidently assessing the effects of the RFS Program—as required under Section 204 of EISA—requires better understanding of the portion of land use change and other environmental changes that are attributable to the RFS Program specifically. The issue of attribution is one new major focus of the RtC3 (see Part 2 Chapters 6 and 7). However, attribution is not the only focus of the RtC3, as updating the state-of-knowledge with respect to the environmental and resource conservation effects specified in EISA Section 204 is also required.

The RtC3 also includes a wide range of co-authors from across the Federal Government, including researchers from the EPA, the U.S. Department of Agriculture (USDA), National Labs of the Department of Energy (DOE), and Department of Interior/U.S. Geological Survey (DOI/USGS).

The analytical approach to the RtC3 is also different from earlier reports. The RtC1 and RtC2 relied entirely on existing peer-reviewed literature to assess the environmental and resource conversation effects of biofuels. The RtC3 reviews the existing literature, but also includes new analyses to fill key knowledge gaps identified in the RtC2 and in the drafting of the RtC3. These new analyses have been peer reviewed, most of which are now available in scientific journals. The small subset of those new studies not yet through the peer review process and published are clearly identified in the text.

To prepare for the RtC3, the authors used the Health and Environmental Research Online (HERO[22]) database to assemble all publications that cited any of the 365 references in the RtC2. There were 14,513 references that cited one or more references in the RtC2. The cutoff date for this initial draw

[22] https://hero.epa.gov/

of publications was September 2019. These were then screened by reviewing the titles and abstracts using SWIFT Active Screener (see Appendix A for details). Of these, 1,555 were identified as relevant for the RtC3 and were sent to Chapter Leads for potential inclusion in the RtC3. Many papers may have cited one or more papers in the RtC2 but were not relevant because of any number of possible factors, including the evaluation of environmental end points not specifically included, a focus on biofuels in other countries, or any number of other possible reasons. More recent literature was included based on co-authors' expertise of the subject area. Additional literature was considered if recommended following the expert panel and the public comment period of the External Review Draft (closed March 6, 2023).

In addition to the literature review, there were new analyses conducted that support the RtC3. These are listed below and referenced to the relevant subsection for more information:

- Supplemental literature review on land use change attributable to biofuels and the RFS Program (see section 6.4).
- Analyses by the National Renewable Energy Lab (NREL) using the Biomass Scenario Model (BSM) to examine the marginal effect of various factors driving increased ethanol growth (see section 6.3.4).
- Economic analysis of ethanol blending through time on a state-by-state basis (see section 6.3.5).
- Assessment of the effects of conversion of non-cropland to cropland from 2008–2016 on losses of nitrogen (N), phosphorus (P), and total suspended sediment (TSS) from fields across a 12-state area in the Midwest (see section 9.3.2)
- Assessment of the effects of conversion of non-cropland to cropland from 2008–2016 on stream water quality in the Missouri River Basin (see section 10.3.2)
- Lifecycle assessment by NREL of non-GHG environmental effects using the Bioeconomy Environmentally extended Input-Output Model (BEIOM), comparing soybean biodiesel with diesel (see sections 8.5, 10.5, and 11.5).

Thus, the RtC3 combines a synthesis of new literature published since the RtC2 with targeted new analyses focused on critical knowledge gaps to advance the current understanding and to lay the foundation for future reports.

1.5 Organization of the Report

Chapters in Part 1 of the RtC3 (Ch. 1–5) present background and scoping information to help the reader understand the biofuels industry and the RtC3. Chapter 2 discusses the scope of the RtC3, explaining the rationale for topics that are addressed and those that are not. Chapter 3 provides background information on the supply chain for biofuels included in the scope of the RtC3. Chapter 4 describes the economics of this agro-industrial system. Chapter 5 discusses background information on large-scale trends in land cover and land management in the United States.

Chapters in Part 2 (Ch. 6 and 7) discuss and assess the issue of attribution, which focuses on what fraction of biofuel and feedstock production are estimated to be attributable to the RFS Program—the focus of this report as specified in EISA Section 204—as opposed to other potential driving factors. Attribution is a necessary prerequisite to understanding the potential effects of the RFS Program on the environment as opposed to the potential effects from the broader biofuels industry. Chapter 6 focuses on corn ethanol and corn, while Chapter 7 focuses on biodiesel and renewable diesel from soybean.

Part 3 (Ch. 8–16) constitutes the core environmental chapters of the RtC3, which separately address the impacts to date and likely future impacts on various environmental and resource conservation issues specified in Section 204. These chapters discuss both the general environmental effects from agriculture and the biofuels industry, and the subset of those effects that may be attributable to the RFS Program. How the statutory language in EISA Section 204 is translated to scientific language for the RtC3 impacts chapters is explained in Chapter 2, but includes effects on air quality (Chapter 8), soil quality (Chapter 9), water quality (Chapter 10), water availability (Chapter 11), terrestrial ecosystems (Chapter 12), aquatic ecosystems (Chapter 13), wetlands (Chapter 14), and invasive species (Chapter 15). Chapter 16 discusses international effects.

The RtC3 then ends with Part 4, which presents the Key Findings from individual chapters (Chapter 17), and Part 5 which contains all the supporting Appendices. A graphical abstract for the RtC3 is shown in Figure 1.9 to help orient the reader.

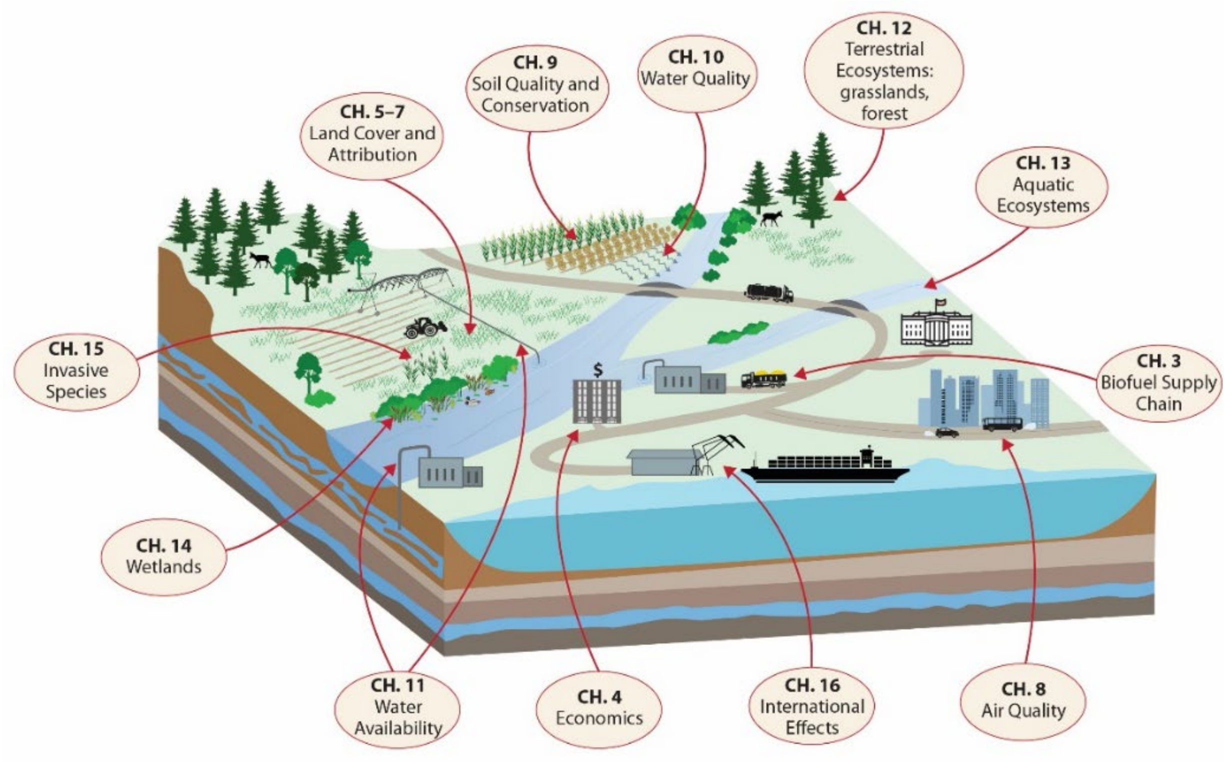

Figure 1.9. Graphical abstract for the RtC3. Included are caricatures for each of the chapters in the RtC3 to describe this complex system.

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2. Scope of the Report

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Key Points

- The EISA Section 204 reports are intended to examine the "impacts to date" and "likely future effects" of the RFS Program. This may include contextual information on the environmental or resource conservation impacts of biofuel production or agricultural activities more generally, but those subjects are not the intended focus of this report series.
- The authors interpret the impacts to date as the historical effects of the RFS Program from 2005 to about 2020, and interpret the likely future as what may be considered relatively likely to occur over the near term, to approximately 2025, considering current market and technology conditions and trends.
- There were 17 biofuels screened for potential inclusion in the RtC3 based on unique combinations of fuel, feedstock, and production region (e.g., biodiesel-soybean-Argentina). This report focuses on any biofuels that dominated the total U.S. pool from 2005 to 2020 to examine those that potentially have a material effect on the environment. This yielded four biofuels for emphasis in the RtC3: (1) domestic corn ethanol from corn starch, (2) domestic biodiesel from soybean oil, (3) domestic biodiesel from fats, oils, and greases (FOGs), and (4) imported ethanol from Brazilian sugarcane.
- Although these four biofuels are the focus of the RtC3, other biofuels (biodiesel and renewable diesel from other feedstocks, cellulosic biofuels, algae-based fuels, etc.) and considerations are also discussed where appropriate.
- All of the environmental and resource conservation effects specified in EISA Section 204 are included. Effects omitted from EISA Section 204 or covered elsewhere in EISA (e.g., greenhouse gases [GHGs] are addressed in Section 201) are not a focus of this report.

Chapter terms: Compressed natural gas and liquified natural gas (CNG/LNG), Conservation Reserve Program (CRP), E10, E15, E85, Fats, oils, and greases (FOGs), Reid Vapor Pressure (RVP), Renewable Volume Obligation (RVO)

2.1 Background

As discussed in Chapter 1, the scope of the RtC3 is based on the statutory language in EISA Section 204 ("Section 204 Report"), which directs EPA to consider the environmental and resource conservation impacts of "the requirements" of CAA Section 211(o), that is, the requirements of the RFS Program. EISA introduced the Section 204 Report, yet the RFS Program predates EISA. Thus, EPA interprets "the requirements" of the RFS Program to include both the requirements of the Program from

2005 through 2008 under the EPAct (RFS1), as well as the requirements of the RFS Program as modified by EISA from 2009 to present day (RFS2).[1]

The reports required by EISA Section 204 are required to assess the effects of the RFS Program on environmental and resource conservation issues. They are not required to assess biofuels as an industry generally, nor all biofuel policies in the United States (including other federal policies and state policies), nor the environmental effects of agricultural production generally or even of the environmental effects from the cultivation of feedstocks used for biofuels (e.g., corn, soybean). However, these and other factors are critical in understanding the effects of the RFS Program in the context of other factors. The statutory focus on the effects of the RFS Program independent of other important factors (e.g., market and non-market factors) is difficult given the overlapping nature of many of these factors in a dynamic market and policy environment (see Chapters 6 and 7). In some publications, environmental effects are ascribed to agriculture generally, biofuels generally, or the RFS Program specifically. However, in many cases there is an assertion or assumption of attribution to the RFS Program that is made, rather than explicitly being evaluated or demonstrated. As discussed in Chapters 4 and 6, many publications assume the RFS Program drove the increase in biofuels in the United States without explicitly evaluating whether that assumption is accurate. It is a straightforward assumption to make, but one that this report series is required to evaluate. This is where the RtC2 left off, concluding that the literature indicated that there were environmental changes taking place that were consistent with anticipated impacts of the RFS Program, but that explicit attribution to the RFS Program, and where and when these effects occurred, had not yet been precisely quantified. These issues are discussed further in Chapters 6 and 7.

This chapter discusses the scope of the RtC3 and the rationale for those decisions. The scope is primarily defined by the time horizon (section 2.2), the biofuels and feedstocks (section 2.3), the spatial extent (section 2.4), and the associated environmental end points (section 2.5). Section 2.6 describes some emerging issues not addressed in the RtC3 and the rationale for that omission. These issues may be covered in future reports.

2.2 Time Horizon

EISA Section 204 states that the report must assess and report the "impacts to date" and "likely future impacts" of the RFS Program. With respect to "impacts to date," the period beginning in roughly 2000–2005 up to about 2020 is considered.[2] This approach is consistent with that of the RtC1 and the

[1] The timing of the two versions of the RFS Program are discussed in Ch. 1, section 1.1.
[2] Unlike the RtC1 and 2, which were primarily literature reviews up to a cutoff date that was 1–1.5 years prior to actual publications, the RtC3 includes studies published in 2023 and with public datasets that are continually being updated as they become available. Different datasets are updated at different intervals, so there is not a single cutoff date for the RtC3.

RtC2 and covers the impacts since the enactment of the RFS Program by Congress (RFS1 in 2005 with EPAct). In addition, a few years prior to 2005 are considered to place whatever changes that occurred beginning in 2005 in the context of contemporaneous trends.

The term "likely future" in Section 204 is ambiguous. The authors of this report interpret the term "likely future" to mean a future impact that is reasonably certain to occur. Given the ongoing recovery from the global COVID-19 pandemic, and the fact that many of the datasets and reports relied upon in this report do not account for this global event, any future impacts are especially uncertain (see sections 2.3.2, and 2.6.1 for more information). The term in the statute is not "possible future" or "potential future," which would expand the scope for considering a much broader range of future outcomes. Other reports discuss these potential futures, which may include biofuels and technologies not yet widely available in the market or which are still under development in research labs across the country. For example, the Department of Energy's 2016 "Billion Ton Report" (DOE, 2017, 2016) and the associated series are useful for decision makers considering the potential of biofuels and bioproducts in the United States under an aspirational future; but, they are not a prediction of a likely future[3] (see Box 2.1: The 2016 Billion Ton Study). Thus, part of the "likely future" interpretation includes how realistic the assumptions are in any particular study in terms of representing current or likely future markets and policy realities.

Another part of interpreting the "likely future" impacts involves how far into the future is projected. Because projections further into the future are inherently less certain than nearer-term projections, the use of "likely future impacts" statutory language restricts the time horizon to be somewhat near term. Because of this, and because the Section 204 reports are required every three years, a relatively short time horizon is used such that the likely future may change from one report to the next as the industry evolves and conditions change. Based on this reasoning, and consistent with the RtC2, "likely future impacts" is interpreted as encompassing near-term future impacts presuming current likely future technologies and rates of market penetration, and current policy and market dynamics, out to approximately 2025.[4]

[3] The disclaimer of the 2016 Billion Ton Report (BT16) states: "BT16 volume 2 is not a prediction of environmental effects of growing the bioeconomy, but rather, it evaluates specifically defined biomass-production scenarios to help researchers, industry, and other decision makers identify possible benefits, challenges, and research needs related to increasing biomass production."
[4] This approach is consistent with the RtC2, which addressed "anticipated over the next three to five years." The RtC1 did not specify a timeline that was "likely future," and instead relied on information in the peer-reviewed literature to specify the time period.

Box 2.1. The 2016 Billion Ton Study

With the goal of understanding the potential of a growing bioeconomy, the U.S. Department of Energy (DOE), national laboratories, and U.S. Forest Service research laboratories, together with academic and industry collaborators, undertook a study in 2016 (termed the "BT16") to estimate the potential biomass available in the United States and the potential environmental effects from utilizing that biomass under specific sets of assumptions. Volume 1 developed county-level biomass-production scenarios for 2017 and 2040 using the agro-economic model Policy Analysis System Model (POLYSYS) under a range of economic and agronomic assumptions (DOE, 2016). Volume 2 used these county-level biomass-production scenarios to estimate environmental effects across multiple metrics (DOE, 2017).

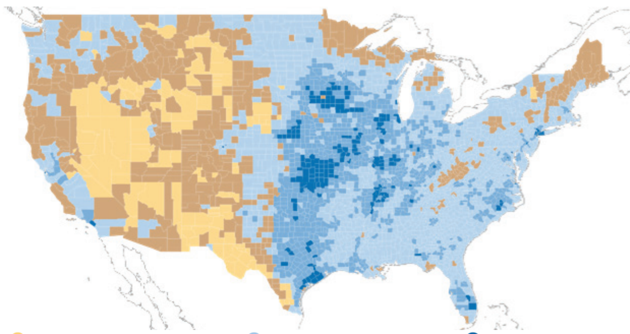

Figure B.2.1. Combined potential supplies in 2040 from forestry, wastes, and agricultural resources, base case.

Volume 1 assessed the biomass availability at three price points ($40, $60, and $80 per dry ton), under a low- and high-yield assumption, for various biomass sources (DOE, 2016). These included eight agricultural energy crops (switchgrass, Miscanthus, biomass sorghum, energy cane, coppice wood from willow and eucalyptus, and non-coppice wood from poplar and pine), five crop residues (corn stover, wheat straw, oat straw, barley straw, and sorghum stubble), four different forestry feedstocks (logging residues, thinnings, whole trees, and other removals), and dozens of sources of waste material from agricultural practices, municipal solid waste, forestry, and other industries. Algae was also included under scenarios of co-location with industrial sources of CO_2. The biomass produced in the scenarios was high, and sources were diverse, compared to current biomass produced in the United States. DOE (2016) estimated that 0.8 billion dry tons of additional biomass could potentially be available annually by 2040 at $60 per dry ton or less (Figure B.2.1), with 1.1 billion dry tons potentially available under a high-yield production scenario.

Volume 2 then assessed the environmental effects of a subset of those scenarios (DOE, 2017). The types of potential effects investigated included changes in soil organic carbon (SOC), GHG emissions, water quality and quantity, air emissions, and avian biodiversity. Most analyses did not estimate benefits from displacing fossil fuels or other products, with the exception of a few illustrative cases on potential reductions in GHG emissions and fossil energy consumption associated with using biomass supplies.

Most analyses showed potential for a substantial increase in biomass production with minimal or negligible environmental effects. However, it is important to note that modeling assumptions were developed to minimize land-use transitions of highest concern and to assume the use of best management practices (BMPs) that promote environmental quality (e.g., reduced tillage, minimal irrigation, no cropland expansion on Conservation Reserve Program [CRP] land). The use of these constraints reduced the potential biomass supply, as well as potential adverse environmental effects of that supply, compared to a scenario without those BMPs.

Compared to fossil fuels, cellulosic biomass showed improvements in certain environmental indicators. The scenarios showed national-level net SOC gains, and in scenarios that expanded the system boundary to biomass end use, reductions in GHG emissions resulted. Analyses of water quality revealed tradeoffs between biomass productivity and some water quality indicators. Biodiversity analyses showed possible habitat benefits to some species, with potential adverse effects to others. Increasing productivity of algae can reduce GHG emissions and water consumption associated with producing algal biomass, though the effects of water consumption are likely of greater concern in some regions than in others. Key research gaps and priorities included actions that can enhance benefits and reduce potential for negative effects.

Box References

DOE (U.S. Department of Energy). (2016). 2016 billion-ton report: Advancing domestic resources for a thriving bioeconomy. Volume 1: Economic availability of feedstocks. (ORNL/TM-2016/160). Oak Ridge, TN: Oak Ridge National Laboratory. https://dx.doi.org/10.2172/1271651.

DOE (U.S. Department of Energy). (2017). 2016 billion-ton report: Advancing domestic resources for a thriving bioeconomy. Volume 2: Environmental sustainability effects of select scenarios from volume 1. (ORNL/TM-2016/727). Oak Ridge, TN: Oak Ridge National Laboratory. https://dx.doi.org/10.2172/1338837.

2.3 Biofuels and Feedstocks

Central to the scope of the RtC3 are the biofuels and feedstocks considered. Biofuels and feedstocks included in the RtC3 are any that had either a material impact to date or are anticipated to have a material impact in the likely future on the environment. This does not include biofuels under development in industries and labs across the country that may eventually be on the market or that are currently produced at volumes too low to have a material environmental impact. This report is required every three years, so as technologies change, so will the composition of biofuels and feedstocks included in the reports.

Because the environmental effect of a biofuel depends on what the biofuel is (e.g., ethanol vs. biodiesel), what feedstock is used to produce that biofuel (e.g., corn vs. sugarcane), and where that feedstock is grown (e.g., soybean in the United States vs. soybean in Argentina), unique biofuel-feedstock-region combinations were identified to consider for the RtC3. There are other factors that also influence the effect of a biofuel-feedstock-region combination (e.g., how it is grown, including different tillage and fertilizer practices), but these are not used in defining the scope of this report and are addressed in other chapters where they are relevant. Furthermore, because the magnitude of any potential environmental effect is partly dependent on the volumes of biofuel produced and consumed, only biofuel-feedstock combinations with any meaningful contribution to the total U.S. biofuel pool from 2005 to 2020 are included (see IS, Figure IS.1). Here the total U.S. pool refers to the domestic production plus imports because production and imports affect the potential environmental effect (e.g., more so than just consumption). Nonetheless, individual chapters in Part 3 have a "Horizon Scanning" section that considers other biofuels that may make up a smaller portion of the domestic pool but still merit discussion (e.g., switchgrass, palm oil). Use of these biofuels in the United States likely has had a much smaller effect on the environment than the more common biofuels. Biofuel-feedstock-region combinations that demonstrate sustained use in the United States are emphasized, rather than one-year use that may not lead to long-term environmental effects. For example, biodiesel from soybean cultivated in Argentina was imported in significant quantities (e.g., 100 million gallons or more) and only periodically (e.g., 2015–2017) because of the United States imposing additional duties on biodiesel imports from Argentina (discussed in Chapters 7 and 16) (USDA, 2018).

2.3.1 *Historical Period*

There were 17 biofuel-feedstock-region combinations evaluated for the historical period considered for this report, which represent the majority of biofuels produced in or imported into the United States according to various sources (Table 2.1, Figure 2.1). Five of these dominated the U.S. pool from 2005 to 2020: (1) corn ethanol from domestically grown corn starch, (2) sugarcane ethanol imported from Brazil, (3) sugarcane ethanol imported from Central America and the Caribbean (CAC), (4) biodiesel from domestically grown soybean, and (5) biodiesel from fats, oils, and greases (FOGs) produced domestically. For the environmental and resource conservation effects associated with imported ethanol, the focus is on Brazil rather than the CAC because of economic and trade factors that suggest that most of the ethanol imported from the CAC actually originated in Brazil (U.S. EPA, 2021; Yacobucci, 2008).[5] Thus, the remainder of the RtC3 focuses on these remaining four biofuel-feedstock combinations: (1) corn ethanol from domestically grown corn starch, (2) biodiesel from domestically grown soybean, (3) biodiesel from FOGs produced domestically, and (4) sugarcane ethanol imported from Brazil.

[5] See Appendix B for further discussion.

Table 2.1. The estimated volumes of biofuel (million gallons) imported or domestically produced from individual biofuel-feedstock-region combinations from 2005 to 2020. Also shown in the percent of the total summed across 2005–2020. Note that biodiesel also includes renewable diesel.[6]

Fuel	Feedstock	Region/Country	Source	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	%
Ethanol	Corn Starch	U.S.	1	3,904	4,884	6,521	9,309	10,938	13,298	13,929	13,218	13,293	14,313	14,807	15,413	15,936	16,091	15,776	13,926	87.4
Ethanol	Sugarcane	Brazil	2	35	453	185	203	5	-	101	404	322	56	88	36	77	53	195	144	1.1
Ethanol	Sugarcane	Central Am./Car.	3	98	228	243	320	182	2	69	82	50	11	-	-	-	1	-	-	0.6
Ethanol	Mixed	Rest of World	4	3	49	8	6	11	13	2	8	5	8	3	1	-	-	-	-	0.1
Biodiesel	Canola Oil	U.S.	5	-	-	-	-	-	35	113	105	85	144	101	160	205	159	164	166	0.6
Biodiesel	Corn Oil	U.S.	6	-	-	-	-	13	16	40	86	141	134	143	185	223	278	234	202	0.8
Biodiesel	Palm Oil	U.S.	7	-	-	-	-	-	-	-	-	83	-	-	-	-	-	-	0	<0.1
Biodiesel	Soybean Oil	U.S.	8	-	-	-	-	309	161	553	537	726	670	665	865	878	1,004	971	1,118	3.8
Biodiesel	FOGs	U.S.	9	-	-	-	-	194	131	320	313	437	480	533	594	542	722	848	853	2.7
Biodiesel	Palm Oil	Southeast Asia	10	-	-	-	-	-	-	-	-	101	54	161	178	2	-	-	-	0.2
Biodiesel	FOGs	Europe	11	-	-	-	-	-	-	11	34	70	17	3	24	19	85	76	59	0.2
Biodiesel	FOGs	Southeast Asia	12	-	-	-	-	-	-	7	13	139	129	138	165	197	185	286	307	0.7
Biodiesel	Soybean Oil	Argentina	13	-	-	-	-	-	-	-	-	65	48	183	435	341	-	-	-	0.5
Biodiesel	Mixed	Canada	14	-	-	-	-	-	-	23	19	22	66	57	102	96	83	83	123	0.3
Biodiesel	FOGs	Rest of World	15	-	-	-	-	-	-	3	1	2	-	-	1	-	-	-	-	<0.1
CNG/LNG	MSW	U.S.	16	-	-	-	-	-	-	1	3	26	52	112	165	208	269	371	467	0.7
CNG/LNG	MSW	Canada	17	-	-	-	-	-	-	-	-	-	-	25	21	32	36	35	35	0.1
Total				4,040	5,614	6,956	9,838	11,652	13,657	15,173	14,823	15,614	16,331	17,134	18,466	18,897	19,000	19,052	17,400	100

[6] Details on the sources of information for Tables 2.1 and 2.2 are in Appendix B. In brief, source #1 (i.e., ethanol-corn starch-U.S.) is calculated from the USDA ERS assuming all domestic ethanol production is from corn. Sources #2–9 are calculated from EIA data. Sources #10–15, imported volumes of biodiesel and renewable diesel by country, are estimated using data from EIA and the EPA Moderated Transaction System (EMTS). Domestic and imported biogas (#16 and #17) are from EMTS. CNG/LNG refers to Compressed Natural Gas and Liquified Natural Gas and MSW refers to Municipal Solid Waste. In two cases (ethanol from the rest of the world and biodiesel from Canada), there was not sufficient data to determine the feedstocks used to produce these fuels. Based on the authors' understanding of the production processes used in these areas to produce ethanol and biodiesel respectively, it is likely that these fuels were produced from a variety of different feedstocks. See Appendix B for additional details.

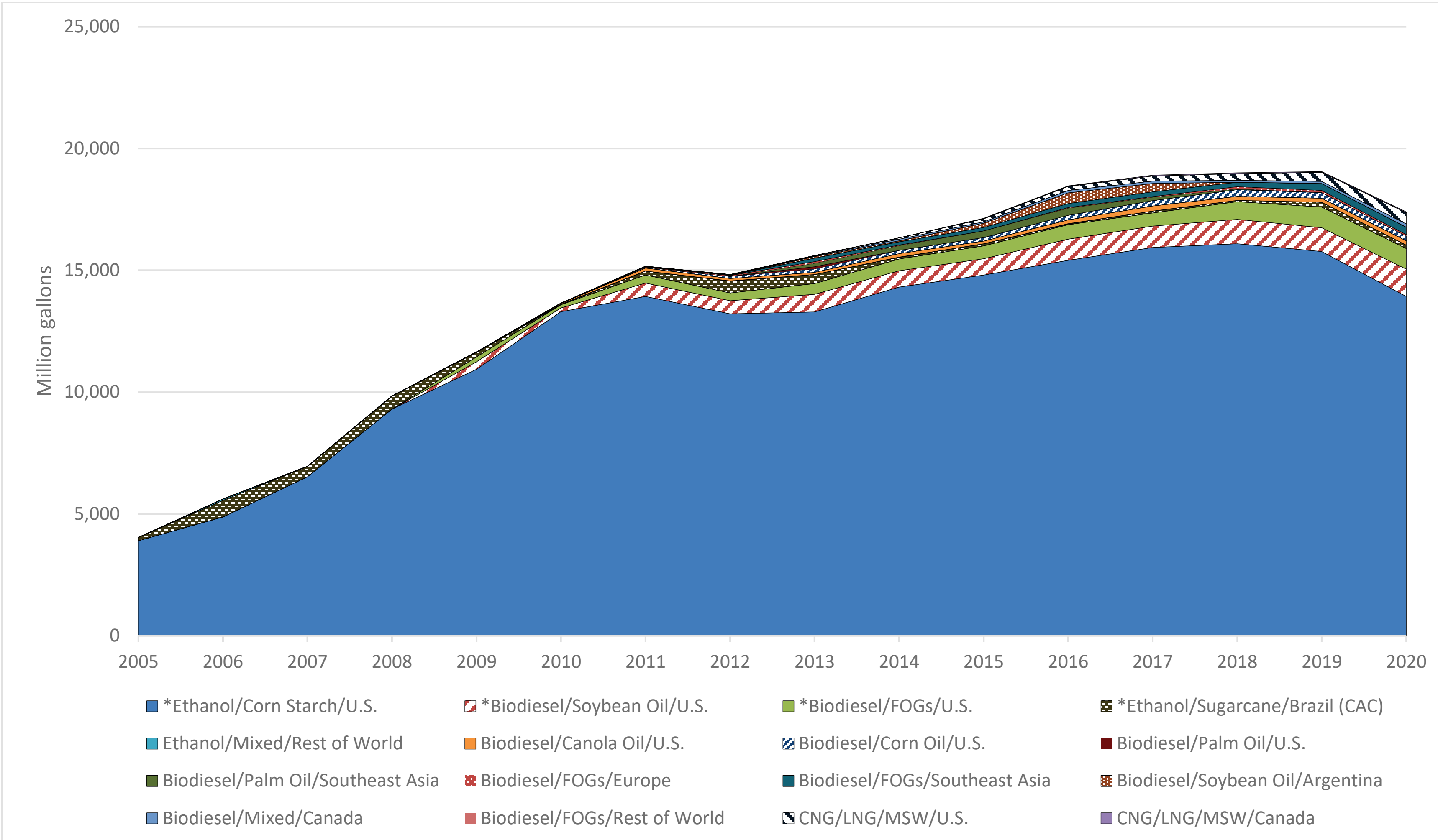

Figure 2.1. The estimated volumes of biofuel (million gallons) imported or domestically produced from individual biofuel-feedstock-region combinations from 2005 to 2020. Same information as Table 2.1**.** Biofuels with an asterisk are the focus of the RtC3, representing most of the biofuels produced and consumed in the United States.

Table 2.2. The percentage on a volumetric basis of total biofuel imported or domestically produced from individual fuel-feedstock-country combinations from 2005 to 2020. This table has the same structure and source material used as in Table 2.1.

Fuel	Feedstock	Region/Country	Source	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020
Ethanol	Corn Starch	U.S.	1	96.6%	87.0%	93.7%	94.6%	93.9%	97.4%	91.8%	89.2%	85.4%	88.4%	87.0%	84.0%	85.0%	84.8%	82.9%	80.0%
Ethanol	Sugarcane	Brazil	2	0.9%	8.1%	2.7%	2.1%	0.0%	0.0%	0.7%	2.7%	2.1%	0.3%	0.5%	0.2%	0.4%	0.3%	1.0%	0.8%
Ethanol	Sugarcane	Central Am./Car.	3	2.4%	4.1%	3.5%	3.3%	1.6%	0.0%	0.5%	0.6%	0.3%	0.1%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
Ethanol	Mixed	Rest of World	4	0.1%	0.9%	0.1%	0.1%	0.1%	0.1%	0.0%	0.1%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
Biodiesel	Canola Oil	U.S.	5	0.0%	0.0%	0.0%	0.0%	0.0%	0.3%	0.7%	0.7%	0.5%	0.9%	0.6%	0.9%	1.1%	0.8%	0.9%	1.0%
Biodiesel	Corn Oil	U.S.	6	0.0%	0.0%	0.0%	0.0%	0.1%	0.1%	0.3%	0.6%	0.9%	0.8%	0.8%	1.0%	1.2%	1.5%	1.2%	1.2%
Biodiesel	Palm Oil	U.S.	7	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.5%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
Biodiesel	Soybean Oil	U.S.	8	0.0%	0.0%	0.0%	0.0%	2.7%	1.2%	3.6%	3.6%	4.7%	4.1%	3.9%	4.7%	4.7%	5.3%	5.1%	6.4%
Biodiesel	FOGs	U.S.	9	0.0%	0.0%	0.0%	0.0%	1.7%	1.0%	2.1%	2.1%	2.8%	3.0%	3.1%	3.2%	2.9%	3.8%	4.5%	4.9%
Biodiesel	Palm Oil	Southeast Asia	10	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.6%	0.3%	0.9%	1.0%	0.0%	0.0%	0.0%	0.0%
Biodiesel	FOGs	Europe	11	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.1%	0.2%	0.4%	0.1%	0.0%	0.1%	0.1%	0.4%	0.4%	0.3%
Biodiesel	FOGs	Southeast Asia	12	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.1%	0.9%	0.8%	0.8%	0.9%	1.0%	1.0%	1.5%	1.8%
Biodiesel	Soybean Oil	Argentina	13	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.4%	0.3%	1.1%	2.4%	1.8%	0.0%	0.0%	0.0%
Biodiesel	Mixed	Canada	14	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.2%	0.1%	0.1%	0.4%	0.3%	0.6%	0.5%	0.4%	0.4%	0.7%
Biodiesel	FOGs	Rest of World	15	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
CNG/LNG	MSW	U.S.	16	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.2%	0.3%	0.7%	0.9%	1.1%	1.4%	1.9%	2.7%
CNG/LNG	MSW	Canada	17	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.1%	0.1%	0.2%	0.2%	0.2%	0.2%

- indicates a biofuel that was <0.1% of the total.

2.3.2 *Future Period*

As in previous years, production and use of biofuel in the United States in future years is expected to be impacted by a broad range of economic and regulatory factors, including the RFS Program. EPA published the estimated likely future effects of the RFS Program for 2023-2025 as a part of the Final Set Rule (Docket: EPA-HQ-OAR-2021-0427). These volumes were proposed on November 30, 2022, and finalized on June 21, 2023.[7] The Final Set rule volumes are presented in Table 2.3.

Table 2.3. Final volume targets (billion RINs)[a] from the Set Rule.

Fuel type	2023	2024	2025
Cellulosic biofuel	0.84	1.09	1.38
Biomass-based diesel[b]	2.82	3.04	3.35
Advanced biofuel	5.94	6.54	7.33
Renewable fuel	20.94	21.54	22.33
Supplemental standard	0.25	n/a	n/a

[a] One RIN (Renewable Identification Number, discussed in Chapter 1) is equivalent to one ethanol-equivalent gallon of renewable fuel. Throughout the Final Set Rule, RINs are generally used to describe total volumes in each of the four categories shown above, while gallons are generally used to describe volumes for individual types of biofuel such as ethanol, biodiesel, renewable diesel, etc. Exceptions include biomass-based diesel (BBD), which is always given in physical volumes, and biogas and electricity, which are always given in RINs.
[b] The BBD volumes are in physical gallons (rather than RINs).

For the purpose of analyzing the projected impacts of the RFS Program volume requirements for 2023–2025 EPA developed "candidate volumes." These candidate volumes, shown in Table 2.4 and detailed in Chapter 3 of the Set rule RIA, represent the mix of renewable fuel types and feedstocks that are projected to be used to meet the volumes for each year. The RIA also contains the analyses that form the basis for these projected candidate volumes.

Relative to the volumes of renewable fuel consumption in 2022, EPA projected an increase of approximately 1.57 billion gallons of renewable fuel consumption in the U.S. in 2025 (Table 2.4). The increase was primarily increases CNG/LNG derived from biogas (+634 million gallons) and renewable diesel from soybean oil (+737 million gallons), FOGs (+295 million gallons), and canola oil (+291 million gallons). EPA projected a slight decrease in the supply of biodiesel (-113 million gallons) due to competition with renewable diesel producers for qualifying feedstocks and a reduction in corn ethanol volumes (-255 million gallons) driven by a reduction in gasoline consumption.

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[7] On July 26, 2022, the United States District Court for the District of Columbia filed a consent decree, which after stipulations from the parties, required EPA to sign a notice of proposed rulemaking (NPRM) to establish 2023 volumes for the RFS Program on or before November 30, 2022, and to sign a notice of final rulemaking finalizing 2023 volumes by June 21, 2023. *Growth Energy v. Regan et al.*, No. 1:22-cv-01191, Document Nos. 12, 14, 15. EPA's proposed RFS volumes for 2023-2025, 87 FR 80582 (proposed and signed on November 30, 2022 and published in the Federal Register on December 30, 2022) available in Docket No. EPA-HQ-OAR-2021-0427 at https://www.regulations.gov, were finalized on June 21, 2023, and published in the Federal Register on July 12, 2023 (88 FR 44468).

Table 2.4. Projected renewable fuel use in 2023–2025 (million gallons) vs. 2022.

Fuel type	2022	2023	2024	2025	2025–2022
Cellulosic biofuel (D3/D7)	666	838	1,090	1,376	710
CNG/LNG – from biogas	665	831	1,039	1,299	634
Ethanol from Corn Kernel Fiber	1	7	51	77	76
Total Biomass-based diesel (D4)[a]	3,113	3,710	3,846	4,239	1,273[a]
Biodiesel	1,737	1,710	1,667	1,624	-113
Soybean oil	995	982	967	953	-42
FOG	346	321	303	285	-61
Corn oil	130	115	89	63	-67
Canola oil (treat as soy oil)	267	292	307	323	56
Renewable Diesel	1,361	1,986	2,165	2,601	1,240
Soybean oil	293	457	641	883	590
FOG	859	1,108	1,074	1,154	295
Corn oil	209	205	239	272	63
Canola oil (treat as soy oil)	0	216	182	291	291
Jet fuel from FOG	14	14	14	14	0
Other Advanced (D5)	247	232	232	232	-15
Renewable Diesel from FOG	73	61	61	61	-12
Sugarcane Ethanol	81	95	95	95	14
Domestic Ethanol from waste ethanol	29	27	27	27	-2
Other[b]	65	49	49	49	-16
Conventional (D6)	14,034	13,845	13,955	13,779	-255
Ethanol – corn	14,034	13,845	13,955	13,779	-255
Renewable diesel - palm oil	0	0	0	0	0
Total Renewable Fuel	18,060	18,625	19,123	19,626	1,566

[a] Excludes BBD (147 million gallons) in excess of the candidate volume for advanced biofuel. The excess would be used to help meet the candidate volume for conventional renewable fuel.
[b] Composed of non-cellulosic biogas, heating oil, and naphtha.

For further context, biofuel production and consumption projections from the Energy Information Administration's Annual Energy Outlook 2023 (EIA, 2023) were considered. The AEO 2023 was released in March 2023, and contains projected production and net imports of ethanol, biodiesel, and renewable diesel through 2050. The projected domestic production and consumption from the AEO for 2023 through 2030 are shown in Figure 2.2 (ethanol) and Figure 2.3 (biodiesel and renewable diesel). The AEO 2023 projects that domestic ethanol production will increase to about 15.7 billion gallons in 2024 and then slowly decline to about 15.5 billion gallons in 2030. Ethanol consumption is projected to increase to about 13.9 billion gallons in 2024 and then steadily decline to about 13.3 billion gallons in

2030. Biodiesel and renewable diesel production are less consistent, but both are expected to reach a maximum by 2024 (about 1.6 billion gallons and 2.5 billion gallons respectively), and then decrease by 2030 (1.2 billion gallons and 2.0 billion gallons respectively). Similarly, biodiesel and renewable diesel consumption are both expected to reach a maximum by 2024 (about 1.7 billion gallons and 3.1 billion gallons respectively), and then decrease by 2030 (1.4 billion gallons and 2.6 billion gallons respectively). Biodiesel consumption for 2023-2025 is in the AEO compared with the EPA estimate; while conversely, renewable diesel consumption is higher in the AEO compared with the EPA estimate.

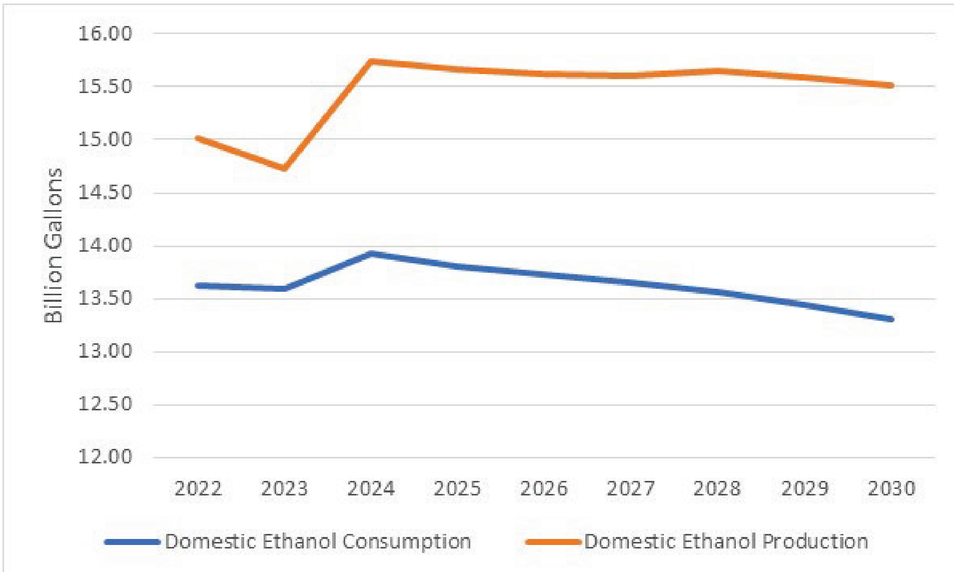


Figure 2.2. Projected ethanol domestic production and consumption from the AEO 2023.

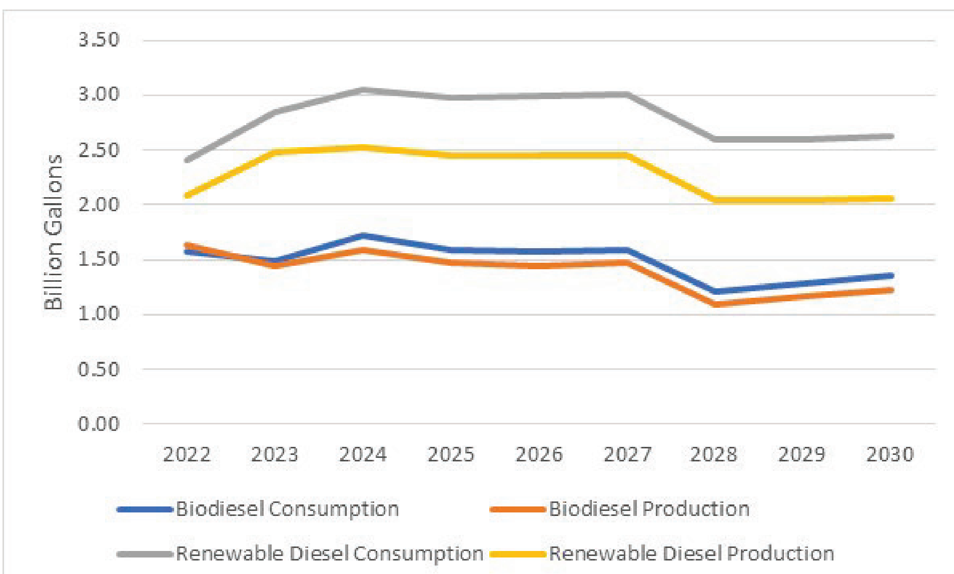


Figure 2.3. Projected biodiesel and renewable diesel production and consumption from the AEO 2023.

2.4 Spatial Extent

While the spatial extent of EISA Section 204 is not defined in the statutory text, the report is clearly meant to include, at a minimum, "impacts" domestically on environmental and resource conservation issues explicitly identified in EISA Section 204(a)(1)-(3) and "impacts" abroad of imported renewable fuels. As EISA Section 204(a) does not define "impacts," but does include many specific "effects" in the lists of environmental and resource conversation issues in EISA Section 204(a)(1)-(3), EPA interprets "impacts" to include both effects *directly* influenced by the RFS Program (e.g., by the cultivation of feedstocks for use as a biofuel as a result of the RFS Program) as well as those *indirectly* influenced by the RFS Program (e.g., the displacement of other crops by corn, leading to greater cultivation of those crops elsewhere to meet the market gap). This dichotomy of direct versus indirect effects is a common theme in the biofuels literature and is discussed in Chapter 5, 6, 16, and elsewhere (NASEM, 2022; Taheripour et al., 2010; Keeney and Hertel, 2009).[8] Additionally, EISA Section 204(a), after setting out specific environmental and resource conversation issues in (a)(1)-(3), further requires EPA to look at the "environmental impacts outside the United States." Due to this placement of the additional requirement to look at international impacts after EISA Section 204(a)(1)-(3), EPA interprets the scope of the list of specific environmental and resource conservation effects in EISA Section 204(a)(1)-(3) to be limited to domestic impacts, while the scope of the international effects are more general in nature.

The spatial extent required for the Section 204 Report does not include environmental effects from the production of all crops or even the production of all biofuel feedstocks (e.g., all corn and soybeans). Rather, EISA Section 204 specifies that the scope of the report is the environmental and resource conservation impacts caused by the RFS Program itself.

As mentioned previously, the statute requires EPA look at international impacts: "The report shall include the annual volume of imported renewable fuels and feedstocks for renewable fuels, and the environmental impacts outside the United States of producing such fuels and feedstocks" (EISA Section 204(a)(1)). It is a reasonable reading of the language that this sentence is likewise being limited to the impacts of the RFS Program itself because "[t]he report" makes that distinction. Secondly, the requirement to look at the "annual volume" of imported renewable fuels and feedstocks for renewable fuels can reasonably be interpreted to refer to the annual volume requirements of the RFS Program.

[8] In the context of this report, "direct effects" means the effects from the primary activities associated with biofuel production and consumption along its entire lifecycle (e.g., farming, conversion, combustion), whereas "indirect effects" mean the secondary effects that may be triggered from these primary activities (e.g., price effects that may affect land use change).

Therefore, consistent with the statute, this report includes the international impacts associated with U.S. imports of Brazilian sugarcane ethanol that were triggered by the RFS Program.

The nuance between biofuels generally and the RFS Program specifically is important to the Section 204 reports but is very difficult to estimate (see Chapters 6 and 7 on Attribution). However, EPA does in some instances provide analysis of impacts from biofuels or agriculture generally, as appropriate, to provide context for those impacts that are estimated to have been caused by the RFS Program specifically.

2.5 Environmental End Points

The statutory language in Section 204 of EISA establishes the general environmental and resource conservation issues to be addressed in the reports. In refining the scope of the report, the authors interpret and define terms in the statutory language. Based on technical knowledge of the subject matter, the categories listed in the statutory language were reorganized into chapters that are more consistent with the scientific literature (Table 2.5). For example, soil environmental quality and soil conservation are inherently linked phenomena (e.g., the latter contributes to the former); these comprise the "Soil Quality" chapter (Chapter 9). As another example, "pesticides, sediment, nutrient and pathogen levels in waters" are different aspects of water quality; thus, these are combined into a single chapter on "Water Quality" (Chapter 10). Further mapping of the statutory language in EISA Section 204 and the RtC3 is shown in Table 2.3. The definitions of these terms are included in the Glossary and described in the individual chapters where they are discussed.

In addition to what is included in the statutory language of EISA Section 204, what is not included in Section 204 helps to further define the scope. Greenhouse gases (GHGs) and climate change are not mentioned in EISA Section 204, and thus are not explicitly addressed in this report (see Box 2.2. Greenhouse Gas Emissions). GHGs are explicitly addressed in EISA Section 201,[9] which modified the RFS Program, and are evaluated during the biofuel pathway review by EPA's Office of Transportation and Air Quality (OTAQ).[10] This approach is consistent with the RtC1 and RtC2, in which the reasons for excluding GHG emissions from the report are further discussed.[11] Exports also are not explicitly mentioned in the statutory language. However, given that U.S. exports have recently increased, and these

[9] Energy Independence and Security Act of 2007, Pub. L. No. 110-140, § 202, 121 Stat. 1492, 1521-28 (2007) (codified as amended at 42. U.S.C. § 7545(o)).
[10] A "pathway" is a unique combination of feedstock, biofuel type, and production process that is evaluated by EPA to determine if it qualifies for Renewable Identification Number (RIN) generation under the RFS Program. For more information, see https://www.epa.gov/renewable-fuel-standard-program/fuel-pathways-under-renewable-fuel-standard.
[11] In the RtC1 this is discussed on page 1-2 and in the RtC2 on pages 3-4.

may indirectly reduce environmental impacts in other countries for which the statute does call for evaluation, exports are included briefly in the RtC3 (see Chapter 16).

Table 2.5. **Mapping of statutory language in EISA Section 204 and the RtC3.**

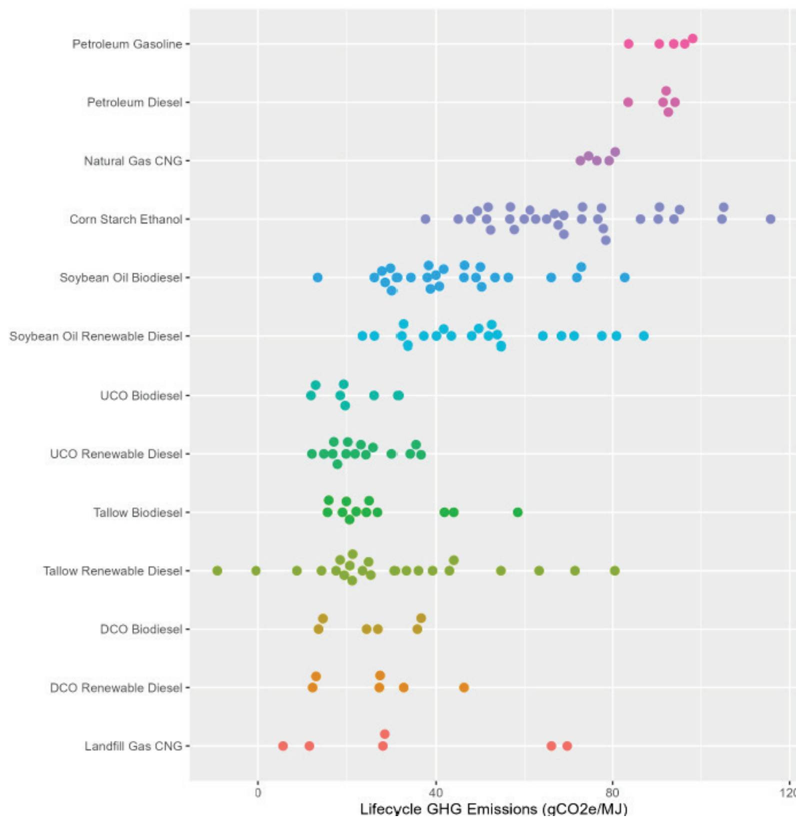
EISA Section 204(a) statutory language	RtC3 chapter number (and shorthand title)
Environmental [. . .] and Resource [C]onservation [I]ssues	Chapters contained in Part 3
[A]ir quality	Chapter 8 (Air quality)
[E]ffects on hypoxia	Chapter 13 (Aquatic ecosystems)
[P]esticides, sediment, nutrient and pathogen levels in waters	Chapter 10 (Water quality)
[A]creage and function of waters	Acreage in Chapter 11 (Water availability), function in Chapters 10 (Water quality),13 (Aquatic ecosystems) and 14 (Wetlands).
[S]oil environmental quality	Chapter 9 (Soil quality)
[S]oil conservation	Chapter 9 (Soil quality)
[W]ater availability	Chapter 11 (Water availability)
[E]cosystem health and biodiversity	Chapter 12–14 (separated by ecosystem type for terrestrial [12], aquatic [13], and wetlands [14])
[I]mpacts on forests	Chapter 12 (Terrestrial ecosystems)
[I]mpacts on [. . .] grasslands	Chapter 12 (Terrestrial ecosystems)
[I]mpacts on [. . .] wetlands	Chapter 14 (Wetlands)
The growth and use of cultivated invasive or noxious plants and their impacts on the environment and agriculture.	Chapter 15 (Invasive species)
[T]he annual volume of imported renewable fuels and feedstocks for renewable fuels, and the environmental impacts outside the United States of producing such fuels and feedstocks.	Chapter 16 (International effects)

Box 2.2. Greenhouse Gas Emissions

As has been noted above, analyses of lifecycle GHG emissions from biofuels are addressed by EPA elsewhere and are not analyzed in this report. Importantly, and in recognition of the need to update EPA's analytical work in this area, the Agency has initiated work to develop a revised modeling framework that would be applied to analyze the GHG impacts associated with biofuels. In consultation with USDA and DOE, EPA hosted a virtual public workshop on biofuel GHG modeling on February 28 and March 1, 2022 to gather input on the current scientific understanding and how this information can be applied to a range of current and future actions.[12] As part of the Additional Resources published in support of the Set Rule for 2023-2025 (https://www.epa.gov/renewable-fuel-standard-program/final-renewable-fuels-standards-rule-2023-2024-and-2025), EPA published the Model Comparison Technical Document, which compared the simulated results from a range of models. The land use change results from the document are summarized in Chapter 6 (domestic) and 16 (international). EPA expects to share additional information on this work either in Federal Register notices or in upcoming RFS rulemakings. While that work progresses, however, and given the importance of lifecycle GHG emissions from biofuels, a brief summary of published estimates is provided here for context for the various topics covered in this report.

The figure below summarizes lifecycle GHG estimates from the scientific literature, shows the ranges of estimates in the scientific literature, and illustrates the level of variability across these estimates. The figure includes the pathways within the scope of the RtC3 (see section 2.3) as well as petroleum diesel and gasoline for comparison. It excludes sugarcane ethanol from Brazil as this literature review did not include this pathway due to time constraints.

Figure B.2.2. Lifecycle GHG estimates from a review of published literature.



Notes: CNG = compressed natural gas, DCO = distillers corn oil, UCO = used cooking oil. Other than reporting all estimates in gCO2e/MJ, no effort has been made to harmonize estimates. Estimates for CNG produced from manure digester biogas are excluded, as some of the estimates for this pathway (e.g., -533 gCO2e/MJ) are so low they would skew the rest of the chart.

[12] Description of the workshop and the presentations are available at https://www.epa.gov/renewable-fuel-standard-program/workshop-biofuel-greenhouse-gas-modeling. EPA maintains a summary of lifecycle greenhouse gas intensities estimated for the RFS Program at https://www.epa.gov/fuels-registration-reporting-and-compliance-help/lifecycle-greenhouse-gas-results.

The figure presents data from studies that were published after the March 2010 RFS2 rule, as that rule considered the available science at the time. In cases where multiple studies include updates to the same general model and approach, only the most recent study was included. However, the authors also included a subset of older estimates that are still used for major regulatory programs or that continue to be widely cited for other reasons. Estimates of the average type of each fuel produced in the United States are presented. For studies that included sensitivity analysis, the authors include representative high and low estimates. For example, when studies report a 95% confidence interval the central estimate is used (usually the default, mean or median estimate) along with the estimates at the top and bottom of the confidence interval. The charts report lifecycle GHG emissions as carbon dioxide-equivalent (CO_2e) emissions per megajoule (MJ) of fuel consumed. All CO_2e estimates are based on 100-year global warming potential (GWP) from the IPCC. This allows comparison across all the estimates on a gCO_2e/MJ of fuel basis. Importantly, the studies in this chart do not consistently align in terms of their scope, system boundaries, time horizon, year of analysis, or other factors. Therefore, the estimates shown in this figure give a sense for the range of estimates for each pathway, but should not be used for rigorous comparison of estimates.

As mentioned above, the science associated with the lifecycle assessment (LCA) of biofuels continues to evolve. Significant analytical work has been undertaken since EPA laid out its lifecycle methodology in the 2010 RFS rulemaking, with work in this area continuing. For example, the National Academies of Science, Engineering and Medicine started an assessment entitled "Current Methods for Life Cycle Analyses of Low-Carbon Transportation Fuels in the United States." This study, released in 2022, assesses the current methods of estimating lifecycle GHG emissions associated with transportation fuels used in a potential national low-carbon fuels program. This work provides useful insight into estimations of GHG emissions over each part of the lifecycle of a given fuel, indirect GHG emissions, and data quality and quantity. EPA is evaluating the results of this work as it will be a useful additional set of information to add to the feedback EPA received on lifecycle assessment through its LCA workshop. EPA also notes the Administration, as part of its Sustainable Aviation Fuel Grand Challenge, has created a workgroup between DOE, EPA, FAA, and USDA to look at LCA methodologies and data needs specifically related to renewable aviation fuel, which will also be a useful platform in assessing LCA capabilities and uncertainties. Data and findings from these ongoing assessments in addition to EPA's modeling comparison exercise will help inform EPA's specific next steps on updating its methodology.

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2.6 Emerging Issues Not Addressed in the RtC3

2.6.1 *COVID-19*

The COVID-19 pandemic has had a significant impact on the production and consumption of transportation fuels in the United States and around the world. While overall demand for gasoline and diesel in the United States has been stable over the past several years, demand for these fuels dropped significantly in response to the COVID-19 pandemic. This reduction in transportation fuel demand affects demand for renewable fuels, particularly ethanol, because the volume of ethanol that could be blended with gasoline (e.g., at a 10% rate) was significantly lower especially in 2020 than in other years (see Chapter 1, Figure 1.4). In addition, the pandemic reiterated the opportunity for additional uses for ethanol outside of the traditional transportation fuel markets, such as the production of pharmaceutical grade ethanol that is used as a component of many hand sanitizers.[13] These alternative markets, however, are small domestically by comparison with liquid transportation fuels. At this time, the impact of the ongoing recovery from the pandemic on renewable fuel production and gasoline and diesel demand is highly uncertain, as it is unclear the degree to which peoples' commuting and driving habits may or may not return to pre-pandemic levels.

2.6.2 *Focus on Emerging Issues as Horizon Scanning*

The requirement to identify "likely future" impacts results in a focus on near-term changes in biofuel production, use, and impacts as noted previously in this chapter. Nevertheless, it is important to recognize longer-term trends that have the potential to change the environmental and resource conservation effects related to the RFS Program to inform the development of future Section 204 reports. Therefore, the "horizon scanning" section, though not strictly necessary, is helpful to include but is intentionally brief. Its focus is on identifying issues of potential importance in the near term that may be relevant in future reports. The trends identified in the "horizon scanning" sections are likely to have modest impact, at most, on likely future consequences within the timeframe of the RtC3 out to 2025.

2.6.3 *Long-Term Changes in Demand*

The primary driver of ultimate environmental impacts from biofuels is the volume of biofuel annually produced and consumed, although the specific impact types and magnitudes will be strongly influenced by feedstock, production practices, and conversion technology, among other factors discussed in this report. Currently, ethanol consumption is dominated by use in light-duty vehicles, while biodiesel consumption is dominated by use in heavy-duty vehicles. Large-scale changes in vehicle technologies, policies, and driving patterns will affect that consumption and the subsequent demand for biofuels.

As mentioned above in section 2.3.2, an increase in E15 and E85 consumption would support increased biofuel use, while an increase in electric-capable (both battery electric and hybrid electric) vehicles would tend to reduce liquid fuel consumption (including biofuels). As seen in Figure 2.4, the number of E85-capable vehicles (flex-fuel vehicles, FFVs) sold in the United States has declined each year since 2013. FFVs were overtaken by hybrid electric vehicles for the first time in 2021. Furthermore, as noted in Chapter 1 (section 1.3.1), since flex-fuel vehicles (FFVs) tend to refuel with E10, the impact of FFVs on biofuel consumption is much smaller than their potential. The trend for hybrid electric and battery electric vehicles is increasing and could reach levels that have a significant effect on biofuel

[13] "More ethanol plants help produce hand sanitizers." 2019. Ethanol Producer Magazine. https://ethanolproducer.com/articles/-17045.

consumption in future years, but they are not anticipated to have significant impacts on biofuel demand out to 2025 compared with gasoline vehicles.

Large-scale policy drivers can also affect both fuel efficiency and vehicle type. Policies designed to reduce emissions of air pollutants and CO_2 at the federal, state, or urban level may result in lower total fuel consumption through increased use of mass transit (and lower vehicle miles), increased fuel efficiency, or growth in the number of alternative fuel vehicles. Such policies will likely have some impact on national fuel (and therefore, biofuel) consumption, although substantial changes are likely to occur only over time periods longer than those examined in this report. However, given the uncertainties in future biofuel consumption in the United States from all the factors discussed above, specific assumptions about long-term demand in the RtC3 are not made.

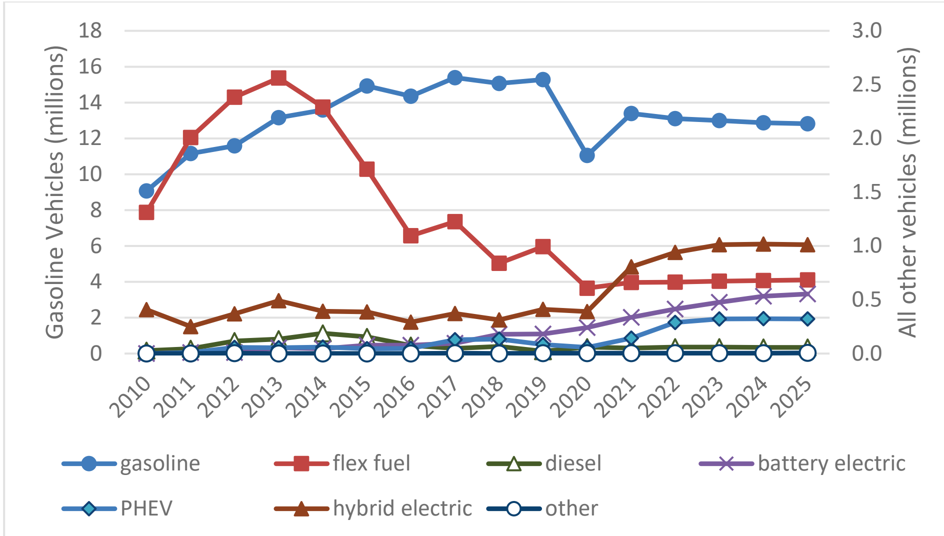

Figure 2.4. Number of gasoline, FFV, diesel, battery electric, plug-in hybrid vehicles (PHEV), hybrid electric, and other vehicles (in millions) sold in the United States from 2010 to 2025.[14] Note the differences in the y-axes between the numbers of gasoline vehicles (left axis) and all other vehicle types (right axis).

2.6.4 *Development Status of Advanced Pathways and Processes*

The substantial majority of biofuel volume remains in the form of corn-starch-based ethanol (Table 2.1 and Figure 2.1). The large-scale development of cellulosic ethanol or other biofuels from renewable feedstocks has not developed as anticipated either in the United States or the rest of the world (Padella et al., 2019). The vast majority of cellulosic biofuel in the United States is currently CNG/LNG derived from biogas, with a smaller volume of cellulosic ethanol produced from corn kernel fiber produced at facilities also producing ethanol from corn and other grains.[15] As of the time of writing, there were no large-scale cellulosic biorefineries producing other liquid cellulosic biofuels in the United States.

[14] Data are from the 2022 Annual Energy Outlook, slide 82 of the Full Chart Library. (https://www.eia.gov/outlooks/aeo/, downloaded 6/17/2022). Projections begin after 2021.
[15] See Cellulosic Biofuel RIN generation data at https://www.epa.gov/fuels-registration-reporting-and-compliance-help/rins-generated-transactions.

Several large-scale cellulosic biorefineries are scheduled to begin production in the next 2–5 years; however, their ability to successfully reach commercial-scale production volumes remains uncertain. While there have been considerable efficiency improvements along the conventional corn ethanol supply chain (discussed further in Chapter 3, and see Box 2.3. Innovation in Ethanol Production), the shift toward other feedstocks with potentially lower environmental impacts has yet to occur at significant scales. Challenges to the commercialization of cellulosic biofuel include the readiness of conversion technology, high capital costs of cellulosic production facilities, and the availability of feedstocks at a price and quality to enable the production of cellulosic biofuel at competitive prices (Padella et al., 2019). Cellulosic biofuels have contributed less than 0.1% of total biofuel volumes as measured by Renewable Identification Number (RIN) generation, and no more than 0.2% of cellulosic biofuel production levels anticipated in EISA.[16] Cellulosic biofuel volume data and the current state of cellulosic biofuel production technologies do not suggest that cellulosic biofuel production will increase to an extent that would have a material effect on environmental or resource conservation impacts within the timeframe of the RtC3.

Removal of woody biomass from the nation's forests instead of burning it as a strategy to reduce wildfire risk and access potential biofuel feedstocks is also being explored (e.g., see Box 2.1). Biomass removal is completed through a variety of tools, including both mechanical thinning and prescribed fire management methods. Where vegetation must be physically cut and removed from the forest, it is most typically accomplished by hauling it off-site. When full removal is not viable because there are no markets, material is cut and piled for on-site burning. All options associated with removal present significant administrative, logistical, environmental, and financial challenges when most of the material is non-marketable. Under the Forest Service's Wildfire Crisis Strategy, the USFS is removing hazardous fuels from 50 million acres over 10 years, which will generate millions of tons of non-marketable biomass each year. This material presents a serious disposal and management challenge. The Renewable Fuel Standard may offer a potential to reduce that challenge by financially incentivizing utilization of this material for higher value end uses, though the types of woody biomass that qualify as renewable biomass under the RFS program are limited by the Energy Independence and Security Act. Production of renewable fuel from forestry feedstock has been very limited in the RFS Program to date, and appreciable volumes of these fuels are not projected through the 2025 time period covered by this report under current market and policy conditions.

[16] https://www.epa.gov/renewable-fuel-standard-program/renewable-identification-number-rin-data-renewable-fuel-standard

2.6.5 *Climate Change and Extreme Weather Events*

Climate change and extreme weather events can affect feedstock production and possibly production of biofuels. Over the next few decades, the impacts of climate change are expected to result in more frequent and more severe extreme weather conditions and events, including flooding, drought, storms, and excessive heat (Hayhoe et al., 2018) that would tend to reduce feedstock production, leading to feedstock supply constraints and the potential for higher fuel prices. The 2012 drought in the central United States, for instance, resulted in significant reductions in corn production (Rippey, 2015). However, the occurrence of such events or the magnitude of their impacts on biofuels cannot be predicted with confidence. As mentioned above, GHGs and climate change are not mentioned in EISA Section 204, and thus are not a focus of this report. Although not a point of emphasis, climate change is addressed in the "Horizon Scanning" sections of several chapters where appropriate.

Box 2.3. Innovation in Ethanol Production

Production of ethanol from corn has become more efficient since the establishment of the RFS2 in 2010 (Rosenfeld et al., 2020). Larger facilities, more efficient production processes, and marketing a portion of distillers grains in wet rather than dry form are contributing to these improvements. Carbon capture and storage (CCS) is a potentially promising recent innovation that has recently been deployed at commercial scale to reduce the GHG footprint of a facility.

Ethanol fermentation produces 0.96 pounds of concentrated, high-purity CO_2 for each pound of ethanol (E100). This concentrated CO_2 stream is easier to capture than the dilute CO_2 stream produced by the combustion of fuels such as natural gas for heat or electricity generation (NETL, n.d.). Roughly one-fifth of all ethanol plants currently capture CO_2 for use in food and beverage production and other industrial uses (Phipps, 2022). This CO_2 could instead be permanently stored in geologic reserves.

The U.S. has one commercial-scale ethanol CCS facility, and another, smaller, facility is under development. Archer Daniels Midland's ethanol plant in Decatur, Illinois has a production capacity of 375 million gallons of ethanol per year, comparable to other mid-to-large biofuel facilities (Ethanol Producer Magazine, 2022). The Illinois Industrial Carbon Capture and Storage (ICCS) facility, partly funded by DOE, is located next door and has been operational since 2017 (DOE, 2022). The ICCS facility has capacity to sequester 1 million metric tons of CO_2 per year, equivalent to the annual production of CO_2 from the ethanol plant's fermenters. Since the ICCS facility began operations, roughly 3.5 million metric tons of CO_2 have been permanently stored in a saline aquifer about 7,000 feet underground.

Red Trail Energy owns a 64-million-gallon-per-year ethanol plant in Richardton, North Dakota. In September 2021, Red Trail secured a loan from USDA to construct a CCS facility onsite. Red Trail's fermenters produce 176,000 metric tons of CO_2 at full annual output. Red Trail plans to store this CO_2 approximately 6,300 feet below the surface.

Innovations like CCS are not yet common across the ethanol industry, but with added investment and adoption they point toward a future that could be positive for the environmental performance of the industry. That said, there remain many uncertainties with CCS, including economic and logistical challenges associated with CO_2 separation, capture, and transport, and in the development of geologic sinks that are safe, effective, and economical.

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3. Biofuel Supply Chain

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Key Findings

- The supply chain of the major biofuels in the RtC3 involve feedstock production (corn and soybean) and collection (fats, oils, and greases [FOGs]), logistics and transport to biorefineries, biofuel production, biofuel logistics, blending and distribution to point of dispensation, and biofuel end use.
- During feedstock production, fertilizers and pesticides are used for corn and soybean cultivation. On a per acre basis, corn typically uses more nitrogen and phosphorus fertilizer than many other crops, including soybean. Corn grown in rotation with soybean requires less nitrogen fertilizer than when not in rotation.
- Adoption of conservation practices has been steadily increasing since the 1990s. From the most recent estimates available, conservation tillage is practiced on 65% of corn and 70% of soybean acres, while other conservation practices have been less widely adopted (e.g., cover crops are approximately 5–6% of cropland, but are slowly increasing). The extent to which conservation practices are used from one year to another on individual lands is largely unknown.
- Although in early years of the biofuels industry wet- and dry-mill processing were comparable in magnitude, dry-mill operations now make up 91% of the ethanol biorefineries. The production of distillers' grains (DGs) for animal feed through either process is a significant coproduct from ethanol production, which mitigates the effect of ethanol demand on demand for corn which is also used for animal feed.
- FOGs are collected from many different types of operations as a waste product or coproduct (e.g., food-processing or livestock production) and typically purified at rendering facilities into useful commodities that are then processed into fuel or for other purposes.
- Ethanol refineries are concentrated in the Midwest nearer to the major feedstock (corn), whereas biodiesel refineries are smaller and more distributed due to the more diverse number and distribution of feedstocks (e.g., soybean oil, FOGs).
- In the early years of ethanol blending, ethanol was "splash blended" with finished gasoline at the gasoline terminal. For at least the last decade ethanol is now blended into gasoline blendstocks which cannot be legally sold at the pump without the addition of an oxygenate such as ethanol.
- Although the number of E15, E85, and B20 stations are increasing in the United States, they remain a small fraction of total fuel stations and thus are not as widely available as E10 or diesel.

Chapter Terms: Anhydrous ethanol, B5, B20, Conservation tillage, Continuous corn, Continuous saccharification, Coproduct, Crop residue, DDGS, Double cropping, E10, E15, E85, Fuel terminal, Lignin, Lignocellulosic biomass, Mid-level ethanol blend, Mulch-till, No-till, Post emergent, Saccharification, Sterols, Tillage, Transesterification, Transloader, Transmodal facility

3.1 Introduction

The biofuel supply chain includes several discrete phases in a long series of activities, ranging from feedstock production decisions to how biofuels are used. This chapter documents the biofuel supply chain in the United States and describes each phase in the supply chain to provide context for the discussion of drivers and impacts of biofuel production and use found elsewhere in this report.[1]

As with the second Triennial Report to Congress in 2018 (i.e., "RtC2"), the main biofuel feedstocks remain corn and soybeans. Therefore, much of this section will focus on the supply chain of the biofuels made from these two feedstocks (Figure 3.1). However, this report and this chapter will also address fats, oils, and greases (FOGs) since FOGs have emerged as an important feedstock in the United States (see Chapter 2, Table 2.2). The supply chain for Brazilian sugarcane ethanol is discussed briefly in Chapter 16. This chapter presents data from roughly 2000 to present, or whatever is the most recent year in which data are available, to characterize the baseline conditions prior to the implementation of the RFS Program.[2]

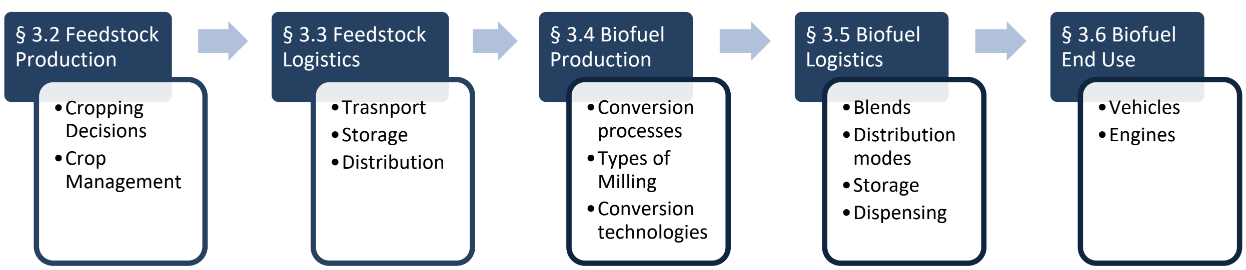


Figure 3.1. Biofuel supply chain. The five major steps in the simplified biofuel supply chain, associated sections in this chapter where they are discussed, and brief examples of topics covered.

[1] Related to this Chapter is Executive Order (EO) 14081 on Advancing Biotechnology and Biomanufacturing Innovation for a Sustainable, Safe, and Secure American Bioeconomy. As part of EO 14081 the Administration is directed to assess the biomass supply chains to advance biotechnology and biomanufacturing towards innovative solutions in health, climate change, energy, food security, agriculture, supply chain resilience, and national and economic security. For more information, see https://www.whitehouse.gov/briefing-room/presidential-actions/2022/09/12/executive-order-on-advancing-biotechnology-and-biomanufacturing-innovation-for-a-sustainable-safe-and-secure-american-bioeconomy/.
[2] See Chapters 1 and 6 for more details on the sequence of acts and regulations associated with the RFS Program.

3.2 Feedstock Production

3.2.1 Crop Feedstocks: Corn and Soybean

The term *feedstocks* in this report refers to crop or non-crop materials that are used to produce biofuels. As mentioned previously, in the United States the two most commonly utilized feedstocks are corn and soybeans. The type of corn discussed in this chapter is referred to as "field corn," which is commonly used for animal feed and for ethanol production.[3] This section provides a broad overview of trends in corn and soybean production practices. The organization of the section reflects the progression of the growing season. First, crop choice and crop rotations are discussed. Next, trends in planting dates and seeding rates are explored. The section then discusses seed choices, pest management decisions, fertilizer use and then harvest. The section concludes with an overview of how corn and soybean are used after harvest.

3.2.1.1 Crop Planting and Production

The number of planted corn acres generally increased from 2000 to 2021, from just under 80 million acres in 2000 to just over 93 million acres in 2021 (Figure 3.2). Corn acres planted reached a peak in 2012 of over 97 million acres. The number of planted soybean acres also increased, from almost 75 million acres in 2000 to just under 87 million acres in 2021, after falling to just over 76 million acres in 2019. Soybean acres planted reached a peak in 2017 of just over 90 million acres. Greater discussion of the general land use trends in the United States associated with agriculture are in Chapter 5.

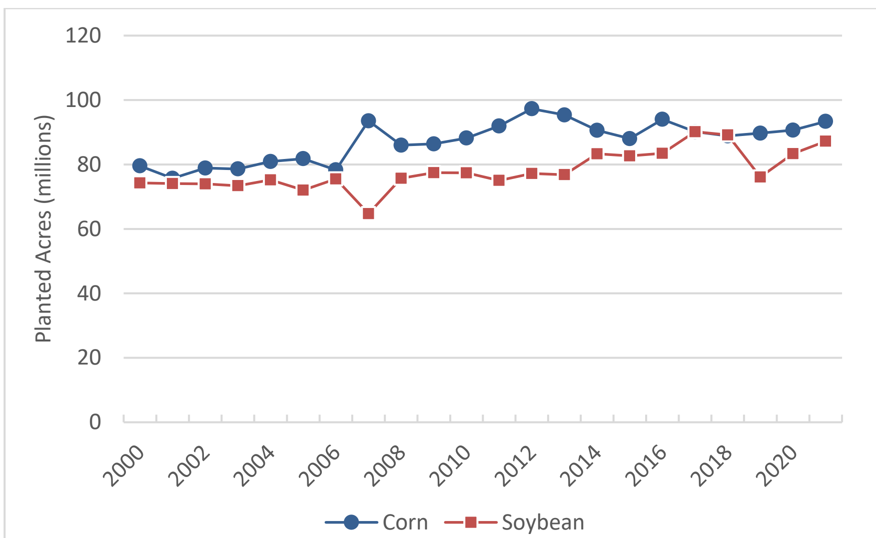


Figure 3.2. Planted acres of corn and soybeans (2000–2021). Source: (USDA-NASS (2022)).

The total production and yields of corn and soybeans also generally increased from 2000 to 2020 from a variety of factors including improvements in genetics, planting methods, and other agronomic

[3] This is in contrast with "sweet corn," which may be directly consumed by humans. Sweet corn only makes up roughly 1% of corn production.

improvements (Figure 3.3). Corn production increased from less than 10 billion bushels in 2000 to over 15 billion bushels in 2021, close to its high point in 2016. Soybean production increased from less than 3 billion bushels in 2000 to just under 4.5 billion bushels in 2018, the year with the highest soybean production, before falling to just over 3.5 billion bushels in 2019 and recovering to just under 4.5 billion bushels in 2021. These increases in corn and soybean production were due to the combination of increased planting acreages of corn and soybeans (Figure 3.2) as well as generally increasing yields for both corn and soybeans during this time period (Figure 3.3). Corn yields increased from 124 bushels per acre in 2000 to 177 bushels per acre in 2021, and soybean yields increased from 37 to 51 bushels per acre over the same period. The decrease in soybean planting and production in 2019 was probably due to a combination of several factors, including depressed soybean prices from tariffs on soybeans exported to China and record acreages of crops prevented from planting due to extreme precipitation in the spring of 2019.[4]

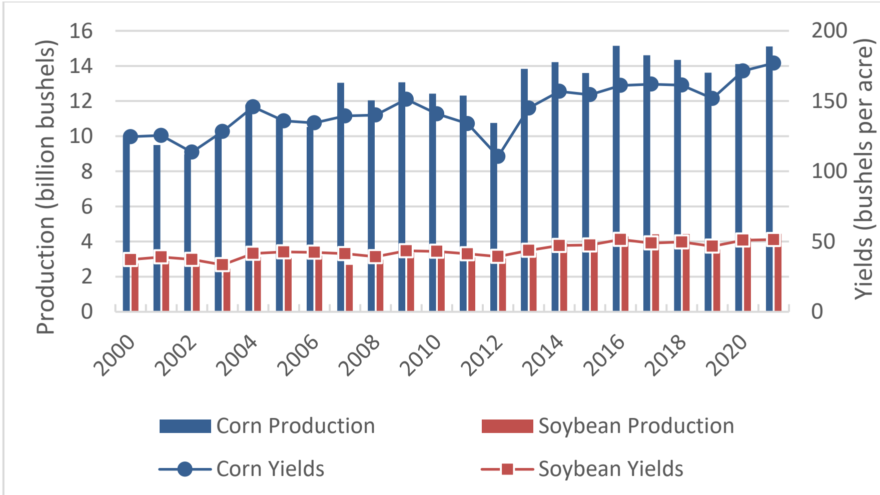


Figure 3.3. Corn and soybean production and yields (2000–2021). Source: (USDA-NASS (2022)).

3.2.1.2 Cropping Decisions: Crop Selection and Rotations

Prior to each planting season, farmers make decisions about which crops to grow. This decision is dependent on many factors, including the anticipated relative profitability of different crops under consideration, current market conditions, the benefits of crop rotation, and historical management practices. Crop rotation can have many benefits. For instance, rotation can reduce pest and disease pressure, while improving soil health and fertility (Congreves et al., 2015; Metcalf and Flint, 1967). Some crops, like soybeans, sequester atmospheric nitrogen, which is then potentially available to more nutrient intensive crops in the following year such as corn.

[4] Trade dynamics for soybean are discussed further in Chapters 7 and 16. Prevent-planting is when an insured crop was prevented from being planted due to extreme weather. In 2019 there were roughly 4.5 million prevented acres soybean and 11.4 million prevented acres of corn. The 2019 USDA report on this is found at https://www.usda.gov/sites/default/files/documents/NASSandFSAacreage_08222019.pdf.

One of the most common crop rotations in the United States is a rotation between corn and soybeans. In 2018, approximately 72% of the soybean fields planted were rotated with corn (Figure 3.4). In 2016, approximately 61% of the corn fields planted were rotated with soybeans.[5] The balance of these rotations has been relatively stable since 2000 at a national level, with some increases in the frequency of corn in rotation. Notably, corn and soybeans are not always rotated on an annual basis. For instance, corn may be planted in multiple consecutive seasons if market conditions warrant. Corn was planted in consecutive years on approximately 28% of U.S. corn fields in 2015 and 2016. In some cases, planting the same crop year after year or in rotation is planned well in advance. In other cases, crop planting plans may change as market conditions change (Wallander et al., 2011). For example, farmers may normally rotate corn and soybeans, but if corn prices are high relative to soybean prices that may incentivize them to plant corn in consecutive years. Corn and soybeans are also rotated, though less so, with small grains and/or other crops, including wheat, barley, sorghum, cotton, hay, and alfalfa (Figure 3.4, "Other"). These rotations with other crops are slightly more common on corn fields than on soybean fields, and rotations with a fallow period are slightly more common on soybean fields than corn fields (Ebel, 2012).

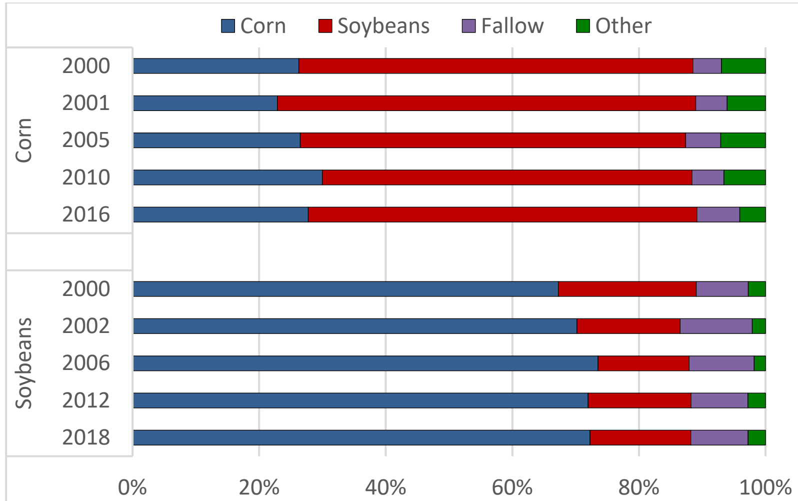


Figure 3.4. Previous crop for corn and soybeans (2000–2018). Source: USDA Agricultural Resource Management Survey (2000, 2001, 2002, 2005, 2006, 2010, 2012, 2018).

On some corn and soybean fields, particularly in southern regions where the growing season tends to be longer, two crops may be grown in one year, one following the other. In cases where both crops are harvested, this practice is called double cropping. Both soybeans and corn tend to be double cropped with winter wheat, though corn is also double cropped with rye. Generally, soybeans are more frequently double cropped than corn (Borchers et al., 2014). However, double cropping is an uncommon practice on U.S. cropland and generally only occurs on 2.2% of cropland acres (Borchers et al., 2014). In some instances, fields are planted to reduce soil erosion and improve soil health. While this cover

[5] Note that the USDA Agricultural Resource Management Survey (ARMS), which is the source of this rotational information, is not collected on the same crops each year. Thus, the different years presented in Figure 3.4. For more information on ARMS see https://www.ers.usda.gov/data-products/arms-farm-financial-and-crop-production-practices/documentation/.

cropping is increasingly prevalent in the United States, it remains relatively uncommon overall (i.e., roughly 5-6% of cropland, USDA, 2022; Wallander et al., 2021; Baranski et al., 2018). Certain federal and state government programs, such as the Environmental Quality Incentives Program (EQIP) or the Conservation Stewardship Program (CSP), incentivize cover cropping.

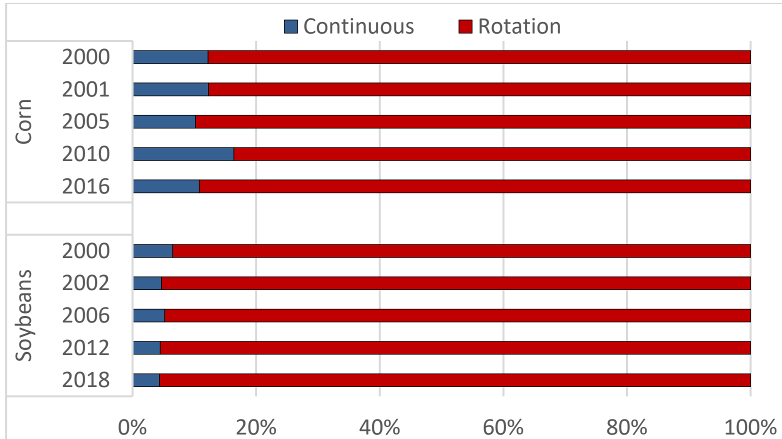


Figure 3.5. Continuous cropping and rotations for corn and soybeans (2000–2018). Source: USDA Agricultural Resource Management Survey (2000, 2001, 2002, 2005, 2006, 2010, 2012, 2018).

On a small fraction of fields, corn and soybeans are rarely (if ever) rotated. These fields are often referred to as being in "continuous" corn or soybean plantings. While it varies over time, approximately 11% of corn fields in the United States have historically been continuously planted with corn (Figure 3.5). The prevalence of fields in continuous corn plantings was relatively stable between 2000 and 2016, with a high point in 2010 (Figure 3.5). Continuous soybean plantings are relatively uncommon due to pest pressures that build up over time. Between 2015 and 2018, approximately 4% of soybean fields were planted continuously.

3.2.1.3 Tillage Decisions

After deciding what to plant, farmers may choose to till the soil. This decision, along with crop type and other decisions, may change from year to year based on several considerations (Sawadgo and Plastina, 2022; Wallander et al., 2021). Tillage is the practice of agitating and aerating the soil in order to incorporate nutrients, bury weeds, warm up the soils in spring, and reduce soil compaction (e.g., plowing). Prior to the development and commercialization of herbicides, tillage was the primary weed control method in corn and soybeans operations. Over time, farmers have become less reliant on tillage for weed control and have reduced the use of conventional tillage equipment. However, tillage continues to be used to control weeds when herbicides are not fully effective.

Tillage practices can be categorized in a variety of ways. Conservation tillage is often defined as any tillage practice leaving at least 30% of the soil surface covered by crop residues. Tillage practices can also be characterized based on their Soil Tillage Intensity Ratings (STIR[6]) which is designated in part by

[6] STIR is a numerical index that represents the type and severity of disturbance caused by tillage operations. The STIR value incorporates the type, speed, depth, and degree of disturbance caused by tillage management decisions. The STIR is the sum of STIR values of individual field operations.

the area of soil surface disturbed (Baranski et al., 2018). No-till, a type of conservation tillage, disturbs the soil only marginally by cutting a narrow planting strip and surface residue is left primarily undisturbed. Mulch and zone tillage are other types of conservation tillage, intermediate in disturbance between no-till and conventional tillage (Claassen et al., 2018).

The prevalence of conservation tillage has increased in both corn and soybeans since 1988 (Mohinder, 1997). In 2016, conservation tillage was used on a majority (65%) of corn fields, especially mulch tillage (Baranski et al., 2018; Claassen et al., 2018) (Figure 3.6).[7] Less than half of the conservation tillage fields were no-tilled. In 2012, 70% of soybean fields were in conservation tillage, more than half of which were not tilled at all (Figure 3.6). Overall, rates of no-till are higher in soybeans (40% in 2012) than in corn (27% in 2016). Recently, there appears to have been a slight decrease in the use of no-till on soybean fields (Figure 3.6). This may be because of the evolution of resistance to glyphosate in weed populations, which first developed in the years following the commercialization of genetically engineered, glyphosate-resistant seeds (see Pest Management, section 3.2.1.5). This emergence of glyphosate-resistant weeds would incentivize the use of other practices for weed control, including potentially more intense tillage practices. Tillage practices are also not universally applied either through time or within an area or even field from year to year (Sawadgo and Plastina, 2022).

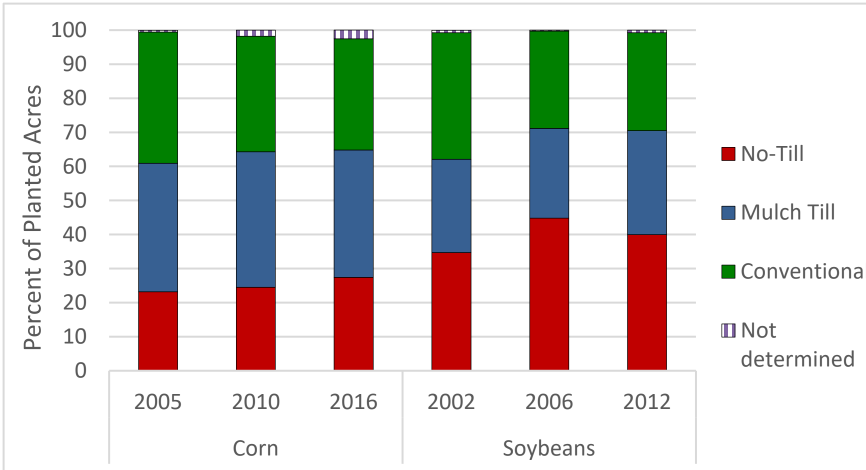


Figure 3.6. Tillage in corn and soybean. Mulch-till—A type of conservation tillage where soil is tilled (for example with a chisel or disk) but soil disturbance is low (STIR less than 80). No-till—The practice of refraining from tilling the soil from harvest of the previous crop to harvest of the current crop. Conventional tillage—A combination of tillage management practices that result in a STIR of greater than 80. Source: Claassen et al. (2018).

In parallel with these shifts in conservation tillage practices for corn and soybean, there are broader shifts in the use of these and other conservation practices in U.S. agriculture generally. The USDA Conservation Effects Assessment Program (CEAP) is a comprehensive examination of

[7] In this example conservation tillage includes both no-till and mulch-till operations.

conservation efforts on cropland in the United States. The second such report (CEAP-2) was released in March 2022, and compares conservation trends on all cropland between 2003–2006 (CEAP-1) and 2013–2016 (CEAP-2) (USDA NRCS, 2022). The CEAP-2 report shows that between 2003–2006 and 2013–2016 conservation tillage increased nationally, and conventional tillage decreased by roughly the same amount (Table 3.1). The largest type of increase of conservation tillage was from continuous no-till. Structural conservation practices[8] also increased, largely along with conservation tillage, with the largest increase in the use of field borders. Thus, there were large decreases in the total cultivated cropland with no conservation tillage or structural conservation practices. However, there were still 61.1 million acres (19%) of cropland with neither management practice in 2013–2016 (down from 100.7 million acres in 2003–2006). The CEAP-2 report also described cover crop adoption, which only made up about 6% of cropland as mentioned in section 3.2.1.1, despite an increase from 2 million to 19 million acres of adoption.

In addition to the total acreage in tillage and other conservation practices nationally and regionally, it is important to document the trends through time because the tillage decisions on a parcel of land may change from year to year, which could have environmental effects (Sawadgo and Plastina, 2022). For example, the emissions from one year of conventional tillage could negate the carbon gains in the soil from several years of conservation tillage (Conant et al., 2007). (Sawado and Plastina, 2022) found that the rate of adoption and disadoption within a region was higher for tillage than it was for

Table 3.1. Tillage groups and classes between CEAP-1 (2003–2006) and CEAP-2 (2013–2016). Shown are the total acreages (in thousands of acres) the percent of total, and the change between CEAP-1 and CEAP-2. Source: USDA NRCS (2022).

Tillage Group/ Tillage Class	CEAP-1		CEAP-2		CEAP-2 minus CEAP-1		Percent Change in Acres Relative to CEAP-1
	Acres (1,000s)	Percent	Acres (1,000s)	Percent	Acres (1,000s)	Percent	
Conservation Tillage	**157,124**	**50**	**210,532**	**67**	**53,408**	**17**	**34**
Continuous mulch	50,631	16	60,212	19	9,581	3	19
Seasonal no-till	44,941	14	47,211	15	2,271	1	5
Continuous no-till	61,553	20	103,108	33	41,556	13	68
Conventional Tillage	**155,941**	**50**	**104,771**	**33**	**-51,169**	**-17**	**-33**
Continuous conventional	62,922	20	42,052	13	-20,869	-7	-33

[8] See Box 1 in USDA NRCS, 2022. Structural conservation practices were broken down into five types in CEAP: (1) field border (e.g., strips of permanent vegetation [grasses, legumes, forbs, or shrubs] established on one or more sides of a field), (2) edge-of-field buffering and filtering (e.g., riparian forest buffers, riparian herbaceous buffers, filter strips, critical area planting), (3) wind erosion control (e.g., windbreaks or shelterbelts, herbaceous wind barriers, hedgerow plantings), (4) concentrated flow control (e.g., grassed waterways, grade stabilization structures, diversions, structures for water control), and (5) overland flow control (e.g., terraces, contour buffer strips, contour farming, stripcropping, in-field vegetative barrier).

Table 3.1. Tillage groups and classes between CEAP-1 (2003–2006) and CEAP-2 (2013–2016). Shown are the total acreages (in thousands of acres) the percent of total, and the change between CEAP-1 and CEAP-2. Source: USDA NRCS (2022).

Tillage Group/ Tillage Class	CEAP-1		CEAP-2		CEAP-2 minus CEAP-1		Percent Change in Acres Relative to CEAP-1
	Acres (1,000s)	Percent	Acres (1,000s)	Percent	Acres (1,000s)	Percent	
Seasonal conventional	93,019	30	62,719	20	-30,300	-10	-33

cover crops. For further discussion of how tillage and other conservation practices affect soil health and water quality, see Chapter 9 (Soil Quality) and Chapter 10 (Water Quality) of this report.

3.2.1.4 Planting Dates and Seeding Rates

After making a crop selection, farmers must choose a planting date. Corn planting dates have consistently moved earlier each successive year, as technology improves and weather patterns change (Abendroth et al., 2017). Corn seed requires a soil temperature of at least 50°F for optimal germination. Corn planting can begin as early as March 1 in extreme southern regions and run as late as early June in more northern areas. Planting dates are broken out by states with the most planted acres of corn in Table 3.2.

Soybean seed requires different planting conditions than corn seed. Young soybean seedlings are more sensitive to environmental conditions after emergence than corn seedlings. Corn seedlings do the majority of growing underground, allowing the plant to sustain cold temperatures with minimal long-term freeze damage. Soybeans, however, are sensitive to cold freeze and require a soil temperature of 55°F before planting (Andales et al., 2000). Seeding dates will vary across the United States, starting in late March in the South to mid-July in the Northeast. General planting dates are listed in Table 3.3 for the states with the majority of soybean planted acreage.

Table 3.2. Planting dates for the top five corn states ordered by rank. Source: (USDA-NASS (2010)).[9] Field Crops: Usual Planting and Harvesting Dates.

State	Planting Dates		
	Begin	Most Active	End
Iowa	Apr 19	Apr 25–May 18	May 26
Illinois	Apr 14	Apr 21–May 23	Jun 5
Nebraska	Apr 19	Apr 27–May 15	May 21
Minnesota	Apr 22	Apr 26–May 19	May 29
Kansas	Apr 5	Apr 15–May 15	May 25

[9] For Tables 3.2 and 3.3, 2010 is the most recent planting dates available from NASS. More updated information may be available from state cooperative extensions.

Table 3.3. Planting dates for the top five soybean states ordered by rank. Source: (USDA-NASS (2010)) [9]. Field Crops: Usual Planting and Harvesting Dates.

State	Planting Dates		
	Begin	Most Active	End
Illinois	May 2	May 8–Jun 12	Jun 24
Iowa	May 2	May 8–Jun 2	Jun 16
Minnesota	May 2	May 8–Jun 2	Jun 13
North Dakota	May 7	May 14–Jun 3	Jun 11
Indiana	May 1	May 5–Jun 10	Jun 25

Seeding rates, in pounds of seed per acre, for corn have increased in recent years partly due to the improved stress tolerance of newer seed hybrids. Seed companies provide recommended seeding rates based on the physical traits of the hybrid and its response to stress. Producers calibrate the recommended seeding rate by the physical characteristics of the field such as soil type, irrigation status, and row width, as well as production considerations like yield expectations and chemical inputs (Reeves and Cox, 2013). A typical seeding rate for a non-irrigated corn field is approximately 30,000 seeds per acre (USDA-NASS, 2022). Planting too many seeds per acre can lead to deficiencies in nutrients or water, which can cause reductions in yield. Corn seed generally germinates at the rate of about 95% and will typically lose 5–10% of the plant population to insects, disease, or other pests (Wright et al., 2004).

The optimal yield for soybeans depends partially on seeding rate but is also influenced by plant genetics and planting date. Seed is one of the most expensive inputs for soybean growers and using an optimal seeding rate minimizes input costs and increases profitability. High plant populations can have advantages for limiting weed competition, however, yield does not always increase as plant population increases (Cox et al., 2010). Seeding rate recommendations are generally between 90,000 and 120,000 seeds per acre to achieve maximum profitability (Kratochvil et al., 2004).

3.2.1.5 Pest Management

U.S. crop producers employ a variety of practices to mitigate potential yield losses from pests. To maintain an optimal yield, producers may alter their crop choices, adjust the planting date, and rotate crops to limit the emergence and spread of weeds, insects, and fungi. As noted earlier, producers may also use mechanical methods such as tillage to manage weeds. Some may release beneficial organism in fields, especially when managing insect pests. Producers may also apply chemical pesticides, including herbicides, insecticides, and fungicides, to control pest populations and mitigate yield losses. The use of agricultural pesticides can impact surface and groundwaters, as pesticides and pesticide residues may be transported from the point of on-field application to nearby waters via runoff, leaching/tile drainage, spray drift, and other transport mechanisms.

3.2.1.5.1 Chemical Pesticides by Crop

In support of the RtC3, the Biological and Economic Analysis Division (BEAD) of the Office of Chemical Safety and Pollution Prevention (OCSPP) provided information on the pesticide usage for corn and soybean, and also for cotton and wheat, which have been observed to be commonly displaced by these biofuel crops (see Chapter 5, section 5.3.1). Pasture and hay, which also may be replaced with corn and soybean, may receive pesticides as well but these are typically at much lower rates. The top pesticides used on field corn, soybeans, cotton, and wheat from 2005 through 2020 were analyzed in terms of base acres treated (BAT), the number of unique acres of a crop treated with a pesticide in a year, and percent crop treated (PCT), the BAT divided by the number of crop acres grown (CAG) in that year.[10] Quantitative seed treatment data are not available for most pesticide types (e.g., neonicotinoids), therefore, this analysis focused on soil and foliar-applied pesticide uses exclusively (but see later in this section). The tables below present usage rates for the first five years (2005–2009) and last five available years (2016–2020) for each crop focusing on the top 15 pesticides by BAT for each period. There are many more pesticides potentially used on each of these crops some for specialty needs, but these top 15 represent the most common and widely used in each period. Longer usage timeseries (1998–2020) are discussed in section 3.2.1.5.2. The potential subsequent environmental and ecological effects of the usage of these pesticides are discussed in Chapters 10, 12, and 13.

The top 15 pesticides in terms of PCT applied to corn between 2005 and 2009 (Table 3.4) were dominated by herbicides, with a single fungicide, pyraclostrobin, and a single insecticide, cyfluthrin, near the bottom of the list. The top 5 pesticides, all herbicides, remained consistent between 2005–2009 and 2016–2020, although the relative proportion of corn acres treated with each herbicide changed. A large increase in glyphosate usage occurred between the periods, moving it to the most widely used pesticide in corn by a large margin. The proportion of acres treated with atrazine remained stable, while the

[10] Nationally representative estimates of active ingredient usage for this analysis were obtained from Kynetec USA, Inc., a proprietary source of pesticide usage data derived from annual grower surveys. Kynetec is one of several sources of data that EPA uses to estimate pesticide usage. EPA also relies on publicly available data from USDA and the State of California Pesticide Use Reporting (PUR). The proprietary data from Kynetec are preferred here because those data are the only national-level, annual pesticide usage data available. The NASS Chemical Use and Agricultural Resource Management Surveys capture pesticide usage on some of the same crops as the Kynetec survey, but they are generally conducted only every 2–5 years. The California PUR data are mandatory reporting for agricultural and other pesticide applications made by professional applicators but are limited to pesticide applications made within California. There are numerous other state-level pesticide reporting databases; however, disparities in the depth, breadth, and methodologies of those surveys make national-level or even regional estimates from data available from multiple states impractical. Thus, the proprietary data available from Kynetec provides more complete spatial/temporal pesticide usage information than is available from other sources alone or in combination.

proportion of acres treated with mesotrione nearly doubled. Usage of metolachlor/S-metolachlor and acetochlor also increased, albeit more modestly.

The increase in corn acres treated with glyphosate was likely due in large part to the increasing adoption of glyphosate-tolerant corn, which is now widespread. (Livingston et al., 2015) offer that although glyphosate-resistant weeds have become problematic, glyphosate resistance is managed in corn through co-application of glyphosate with partner herbicides that offer control of the resistance. This provides a likely explanation for the rise in usage of many of the herbicides within and outside those reported in Table 3.4.

Table 3.4. Percent of corn area treated (PCT) and base acres treated (BAT) for the 15 most common pesticides. PCT and BAT are averaged for 2016–2020 and 2005–2009 and ordered by BAT in 2016–2020. NA indicates that the pesticide was not in the top 15 for the period reported.[11]

Top Active Ingredients (AIs)	Pesticide Type	2016–2020		2005–2009	
		Avg Annual PCT	Avg Annual BAT	Avg Annual PCT	Avg Annual BAT
GLYPHOSATE	Herbicide	76%	69,000,000	58%	49,700,000
ATRAZINE	Herbicide	60%	54,200,000	60%	50,900,000
MESOTRIONE	Herbicide	39%	35,600,000	18%	15,400,000
METOLACHLOR/S-METOLACHLOR	Herbicide	30%	27,300,000	23%	19,700,000
ACETOCHLOR	Herbicide	30%	27,100,000	22%	18,600,000
DICAMBA	Herbicide	18%	15,900,000	9%	7,600,000
CLOPYRALID	Herbicide	17%	15,000,000	NA	NA
2,4-D	Herbicide	14%	12,200,000	7%	6,200,000
FLUMETSULAM	Herbicide	11%	9,600,000	4%	3,400,000
ISOXAFLUTOLE	Herbicide	9%	8,100,000	6%	4,900,000
THIENCARBAZONE-METHYL	Herbicide	8%	7,200,000	NA	NA
TEMBOTRIONE	Herbicide	8%	7,000,000	NA	NA
BICYCLOPYRONE	Herbicide	7%	6,800,000	NA	NA
AZOXYSTROBIN	Fungicide	7%	6,600,000	NA	NA
PROPICONAZOLE	Fungicide	7%	6,400,000	NA	NA
RIMSULFURON	Herbicide	NA	NA	6%	5,000,000
NICOSULFURON	Herbicide	NA	NA	6%	4,700,000
PYRACLOSTROBIN	Fungicide	NA	NA	5%	4,500,000
DIMETHANAMID/DIMETHANAMID-P	Herbicide	NA	NA	5%	4,300,000
GLUFOSINATE	Herbicide	NA	NA	5%	4,100,000
CYFLUTHRIN	Insecticide	NA	NA	4%	3,300,000

[11] Source for Tables 3.4–3.7 is the aforementioned Kynetec dataset in the footnote above.

Like corn, the top pesticides applied to soybeans were mostly herbicides, with one insecticide, lambda-cyhalothrin, and one fungicide, pyraclostrobin, in the top 15. Glyphosate was the dominant pesticide in both the 2005–2009 and 2016–2020 (Table 3.5) intervals, with approximately 95% and 81% of acreage treated, respectively. However, glyphosate usage decreased somewhat in recent years, and usage of other herbicides increased dramatically. In particular, dicamba, metolachlor/S-metolachlor, sulfentrazone, fomesafen, 2,4-D, and metribuzin acres treated all more than doubled relative to the earlier period. This was likely due to grower attempts to rotate herbicide chemistries and use combinations of chemicals as a partner with glyphosate to combat glyphosate-resistant weeds (Livingston et al., 2015).

Table 3.5. Percent of soybean area treated (PCT) and base acres treated (BAT) for the 15 most common pesticides. PCT and BAT are averaged for 2016–2020 and 2005–2009 and ordered by BAT in 2016–2020. NA indicates that the pesticide was not in the top 15 for the period reported.

		2016–2020		2005–2009	
Top Active Ingredients (AIs)	Pesticide Type	Avg Annual PCT	Avg Annual BAT	Avg Annual PCT	Avg Annual BAT
GLYPHOSATE	Herbicide	81%	68,200,000	95%	69,100,000
DICAMBA	Herbicide	24%	20,000,000	NA	NA
METOLACHLOR/S-METOLACHLOR	Herbicide	22%	18,300,000	NA	NA
SULFENTRAZONE	Herbicide	21%	17,900,000	2%	1,700,000
FOMESAFEN	Herbicide	21%	17,900,000	3%	2,200,000
2,4-D	Herbicide	19%	16,400,000	8%	5,700,000
METRIBUZIN	Herbicide	19%	16,100,000	NA	NA
GLUFOSINATE	Herbicide	16%	13,600,000	NA	NA
FLUMIOXAZIN	Herbicide	13%	10,900,000	4%	3,100,000
CHLORIMURON	Herbicide	13%	10,600,000	6%	4,400,000
PYROXASULFONE	Herbicide	13%	10,600,000	NA	NA
CLETHODIM	Herbicide	12%	10,500,000	4%	2,600,000
IMAZETHAPYR	Herbicide	11%	9,200,000	4%	2,600,000
CLORANSULAM-METHYL	Herbicide	10%	8,800,000	3%	2,100,000
LAMBDA-CYHALOTHRIN	Insecticide	9%	7,800,000	8%	5,900,000
CHLORPYRIFOS	Insecticide	NA	NA	6%	4,300,000
PYRACLOSTROBIN	Fungicide	NA	NA	5%	3,600,000
PENDIMETHALIN	Herbicide	NA	NA	4%	2,600,000
TRIFLURALIN	Herbicide	NA	NA	2%	1,800,000
AZOXYSTROBIN	Fungicide	NA	NA	2%	1,700,000

Pesticide usage on cotton was dominated by herbicides and plant growth regulators in both the 2005–2009 and 2016–2020 periods; however, insecticides also had a noticeable presence. Glyphosate was the predominant herbicide applied to cotton in both periods (Table 3.6), being applied to around 85% of cotton acres annually. Glyphosate also had minor usage as a growth regulator, but other active ingredients, particularly mepiquat and ethephon, were much more commonly applied for that purpose. Herbicides and growth regulators accounted for the vast majority of acres treated. However, insecticides were among the top 10 pesticides used on cotton, with aldicarb and acephate used on an average of approximately a quarter of cotton acres from 2005-2009, and acephate being used on a similar fraction of cotton acreage in the 2016 to 2020 period.

Table 3.6. Percent of cotton area treated (PCT) and base acres treated (BAT) for the 15 most common pesticides. PCT and BAT are averaged for 2016–2020 and 2005–2009 and ordered by BAT in 2016–2020. NA indicates that the pesticide was not in the top 15 for the period reported.

Top Active Ingredients (AIs)	Pesticide Type	2016–2020		2005–2009	
		Avg Annual PCT	Avg Annual BAT	Avg Annual PCT	Avg Annual BAT
GLYPHOSATE	Herbicide/Growth Regulator	85%	10,200,000	84%	9,700,000
MEPIQUAT	Growth Regulator	54%	6,500,000	60%	7,000,000
ETHEPHON	Growth Regulator	52%	6,300,000	49%	5,600,000
DICAMBA	Herbicide	40%	5,000,000	NA	NA
PARAQUAT	Growth Regulator/ Herbicide	35%	4,100,000	15%	1,600,000
THIDIAZURON	Growth Regulator	32%	3,800,000	26%	3,100,000
DIURON	Growth Regulator/ Herbicide	29%	3,500,000	19%	2,300,000
GLUFOSINATE	Herbicide	26%	3,100,000	NA	NA
ACEPHATE	Insecticide	26%	3,100,000	25%	2,800,000
TRIBUFOS	Growth Regulator	24%	2,900,000	28%	3,300,000
ACETOCHLOR	Herbicide	22%	2,700,000	NA	NA
TRIFLURALIN	Herbicide	22%	2,600,000	28%	3,300,000
2,4-D	Herbicide	20%	2,500,000	11%	1,200,000
FLUMIOXAZIN	Herbicide	20%	2,500,000	NA	NA
METOLACHLOR/S-METOLACHLOR	Herbicide	18%	2,200,000	NA	NA
ALDICARB	Insecticide/Nematicide	NA	NA	24%	2,800,000
PENDIMETHALIN	Herbicide	NA	NA	18%	2,000,000
DICROTOPHOS	Insecticide	NA	NA	17%	1,900,000
PYRITHIOBAC-SODIUM	Herbicide	NA	NA	11%	1,300,000
BACILLUS CEREUS	Growth Regulator	NA	NA	9%	1,200,000

For wheat, herbicides accounted for the vast majority of pesticide usage during the 2005–2009 and 2016–2020 periods (Table 3.7). 2,4-D and glyphosate were the predominant herbicides used in wheat cultivation over the 2005–2009 period. Annually, those active ingredients were each applied to approximately one-fifth of the acres on which wheat was grown between 2005 and 2009, with a variety of other herbicides also being applied to lesser extents. Fungicides were also among the pesticides with the highest reported usage in wheat, but neither of the two most used fungicides reached 10 PCT in the 2005–2009 period. In the most recent period, 2016–2020, the average annual percentage of acres of wheat planted that were treated with an herbicide increased; however, the average number of acres on which wheat was grown decreased between the periods. Thus, the average number of herbicide-treated acres remained relatively static. In contrast the percentage and absolute number of acres of wheat treated with fungicides increased markedly, with propiconazole and tebuconazole usage having the greatest increases, more than doubling the number of acres treated with fungicides between 2005–2009 and 2016–2020.

Table 3.7. Percent of wheat area treated (PCT) and base acres treated (BAT) for the 15 most common pesticides. PCT and BAT are averaged for 2016–2020 and 2005–2009 and ordered by BAT in 2016–2020. NA indicates that the pesticide was not in the top 15 for the period reported.

		2016–2020		2005–2009	
Top Active Ingredients (AIs)	**Pesticide Type**	**Avg Annual PCT**	**Avg Annual BAT**	**Avg Annual PCT**	**Avg Annual BAT**
GLYPHOSATE	Herbicide	25%	11,000,000	17%	9,700,000
2,4-D	Herbicide	23%	10,000,000	23%	14,000,000
PROPICONAZOLE	Fungicide	21%	9,600,000	9%	5,100,000
FLUROXYPYR	Herbicide	17%	7,700,000	8%	4,900,000
BROMOXYNIL	Herbicide	16%	7,300,000	11%	6,100,000
TEBUCONAZOLE	Fungicide	15%	7,000,000	NA	NA
METSULFURON	Herbicide	14%	6,300,000	14%	8,100,000
MCPA	Herbicide	12%	5,500,000	13%	7,400,000
TRIBENURON METHYL	Herbicide	12%	5,300,000	12%	6,700,000
THIFENSULFURON	Herbicide	12%	5,300,000	12%	7,200,000
DICAMBA	Herbicide	11%	4,800,000	7%	4,200,000
CLOPYRALID	Herbicide	10%	4,400,000	6%	3,700,000
PYRASULFOTOLE	Herbicide	9%	4,200,000	NA	NA
PROTHIOCONAZOLE	Fungicide	8%	3,700,000	NA	NA
CHLORSULFURON	Herbicide	6%	2,500,000	8%	4,800,000
FENOXAPROP	Herbicide	NA	NA	6%	3,700,000
PYRACLOSTROBIN	Fungicide	NA	NA	5%	2,700,000
CLODINAFOP	Herbicide	NA	NA	4%	2,100,000

There are other pesticides used on crops that are not able to be quantified with the same level of confidence. As noted above, the neonicotinoid insecticides, including imidacloprid, thiamethoxam, and clothianidin, are not reported in the available datasets and are also important to consider. Neonicotinoids' effective application rates are not reported by these sources, in part because of gaps in, and difficulties associated with the collection of data regarding these chemicals' primary method of application (i.e., via treated seed) (Hitaj et al., 2020). Neonicotinoids are important additions to the list of pesticides of potential concern in the corn belt, in part because of their ecotoxicological properties (see Chapters 10 and 13) and because their usage as seed coatings has increased dramatically over the past two decades, partly as replacements for organophosphate and carbamate insecticides (Chrétien et al., 2017; Hladik et al., 2014). By 2008, neonicotinoids accounted for an estimated 80% of the insecticide-treated seed market (Hitaj et al., 2020), and by 2011 approximately 34–44% of soybean acreage and 79–100% of corn acreage in the United States were treated with neonicotinoid-coated seed (Douglas and Tooker, 2015). Neonicotinoids are highly water soluble, hydrolytically stable compounds with half-lives up to hundreds to thousands of days in soil and water (Bonmatin et al., 2015; Morrissey et al., 2015). These issues are discussed further in Chapters 10 and 13.

3.2.1.5.2 Chemical Pesticides Trends

As a compliment to the information presented in section 3.2.1.5.1, BEAD also provided annual usage over a longer period (1998–2020) to show any annual trends that may not have been apparent with the 5-year averages, and for trends that may have predated the RFS Program. Annual usage for 2,4-D, acetochlor, atrazine, dicamba, dimethenamid and dimethenamid-P, glyphosate, metolachlor and s-metolachlor, and paraquat are discussed below alphabetically by active ingredient (AI), and a subset of these are shown in Figure 3.7.

Usage of 2,4-D in corn and soybeans was relatively similar in terms of reported PCT and BAT up to 2019 (Figure 3.7a). Usage of 2,4-D on these crops generally increased in the late 2000s with both BAT and PCT approximately doubling for these crops in the 2010s relative to the 2005–2009 period. 2,4-D PCT and BAT increased from an average 7% and 6.2 million acres to 14% and 12.2 million acres, respectively. A similar trend was noted in cotton, albeit the number of acres was lower, changing from approximately 1.2 to 2.5 million acres. In contrast, usage on wheat remained relatively constant around 20–25% in terms of PCT, although the absolute number of BAT decreased from approximately 15 million to 10 million acres between 1998 and 2020 (Figure 3.7b).

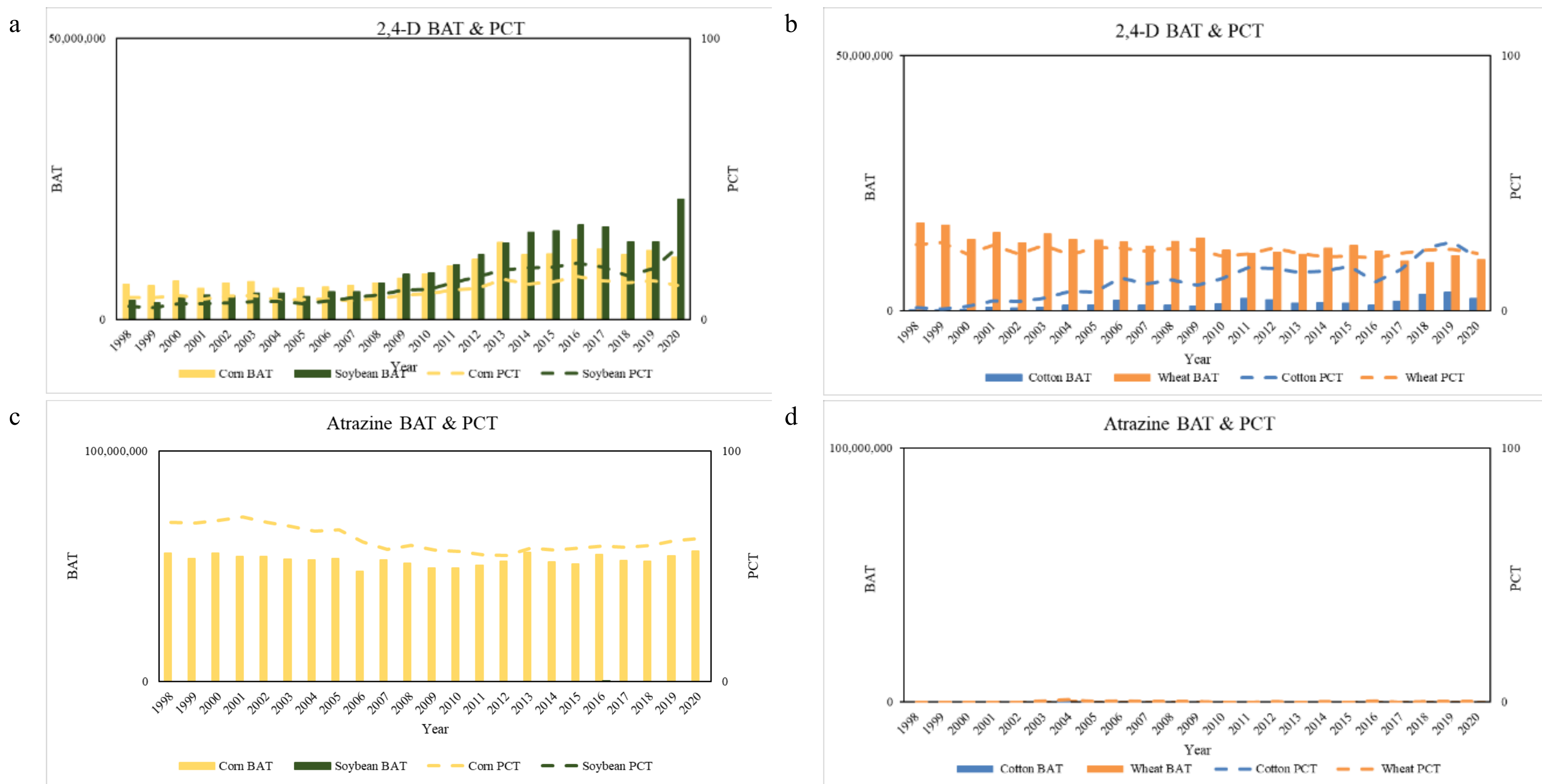


Figure 3.7. Annual BAT and PCT from 1998-2020 on corn and soybean (left column) or cotton and wheat (right column) for 2,4-D (a-b), atrazine (c-d), dicamba (e-f), glyphosate (g-h), and combined metolachlor and s-metolachlor (i-j).[12]

[12] Values are not able to be shown with quantitative precision due to data-use agreements with Kynetec. Hence, the y-axes with only the maximum identified.

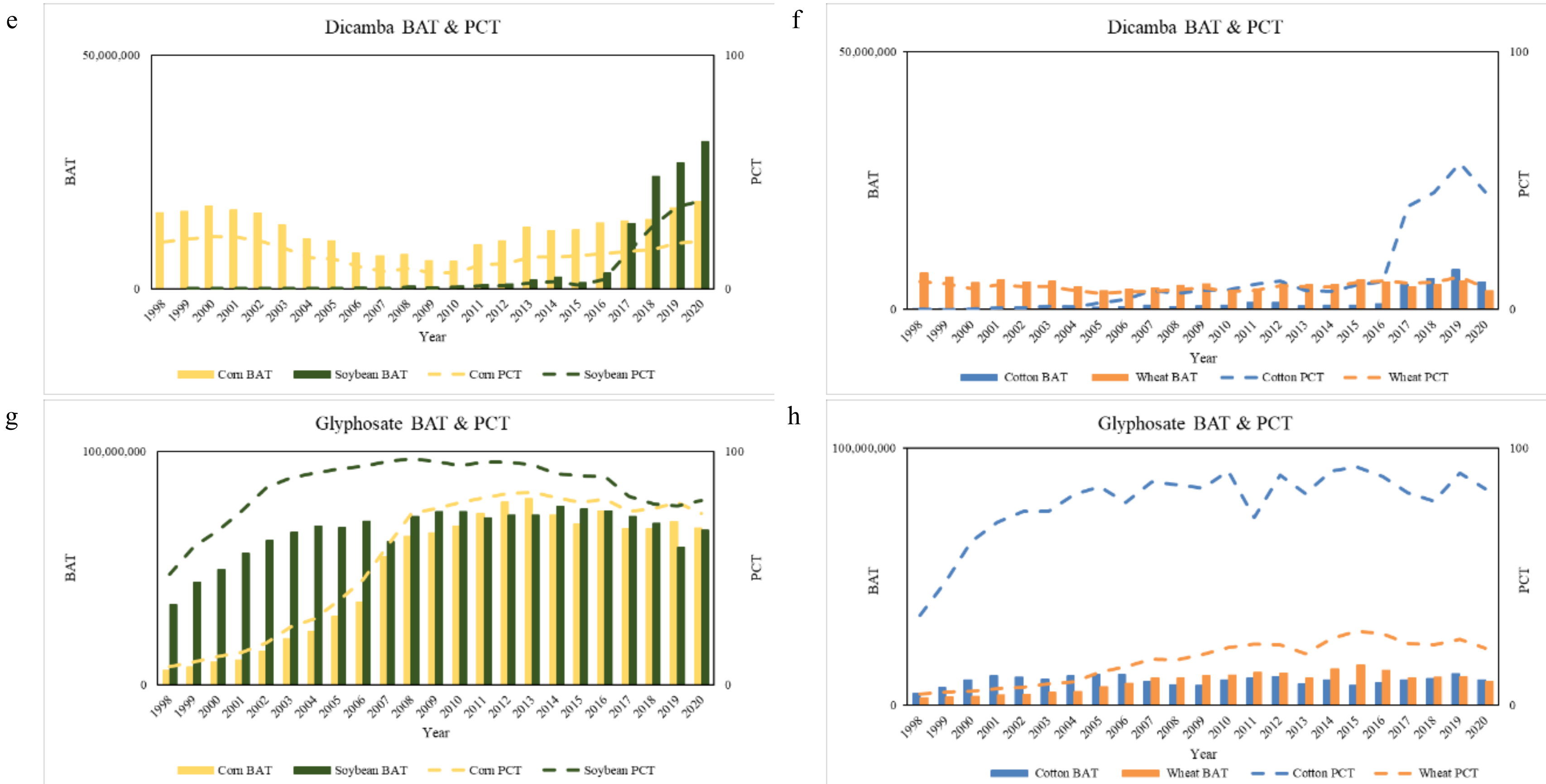
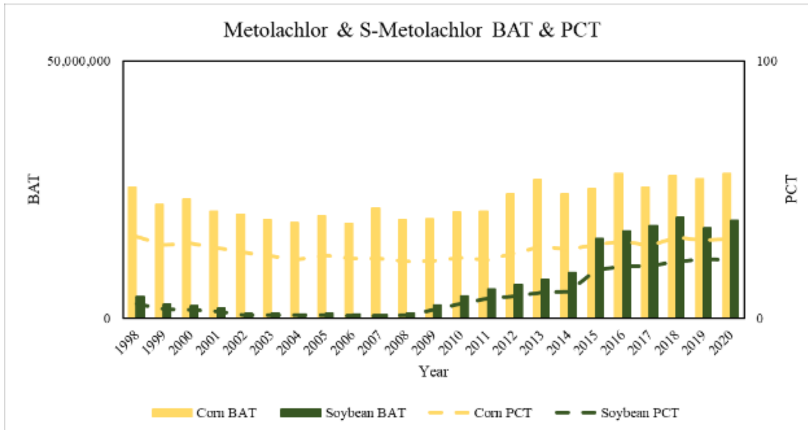


Figure 3.7 (continued). Annual BAT and PCT from 1998-2020 on corn and soybean (left column) or cotton and wheat (right column) for 2,4-D (a-b), atrazine (c-d), dicamba (e-f), glyphosate (g-h), and combined metolachlor and s-metolachlor (i-j).[12]

i



j

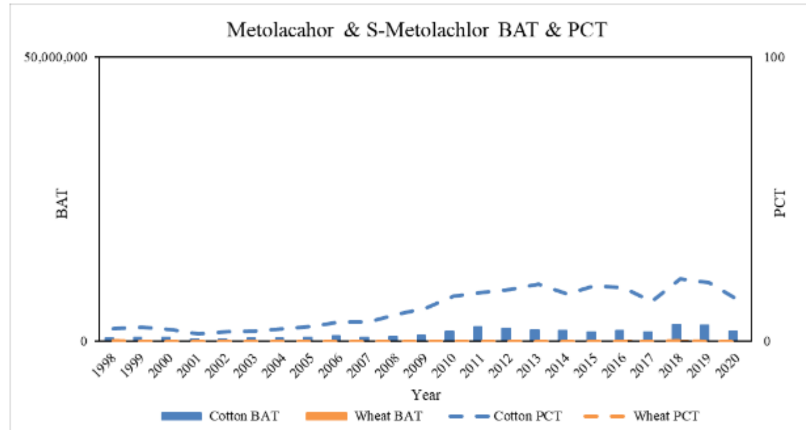


Figure 3.7 (continued). Annual BAT and PCT from 1998-2020 on corn and soybean (left column) or cotton and wheat (right column) for 2,4-D (a-b), atrazine (c-d), dicamba (e-f), glyphosate (g-h), and combined metolachlor and s-metolachlor (i-j).[12]

Usage of acetochlor in corn was consistently reported in the 20–25 PCT range from 1998 into the early 2000s with a steady increase observed from 2011 and 2020. No substantive usage of acetochlor in soybean was reported prior to 2011. Since then, usage steadily increased, but remained below 10 PCT. Usage of acetochlor on cotton was not reported prior to 2010, but since 2013 its usage has hovered around 20 PCT and 2 to 3 million acres. Usage of acetochlor was not reported on wheat between 1998 and 2020.

Atrazine usage on corn was consistently around 50 million acres per year from 1998 to 2020 (Figure 3.7c). Due to increases in the number of acres of corn grown, this resulted in a slight decrease in the annual PCT of atrazine over the years reported. The average atrazine PCT in corn in the earliest years reported was approximately 70 PCT, while more recently the annual PCT was closer to 60%. In wheat, atrazine was consistently reported, but the estimated number of acres treated annually was somewhat variable and extremely low, averaging approximately 100,000 acres. This translated to a very small percentage of wheat acres (i.e., <1 PCT, Figure 3.7d). Substantive usage of atrazine on soybean and cotton was not reported (Figure 3.7c, d).

Usage of dicamba on corn was relatively consistent prior to 2004, with approximately 16 million base acres treated or 20 PCT, but usage dipped in the early 2000s at the time glyphosate-tolerant corn gained popularity (Figure 3.7e). More recently, dicamba usage increased to the previous level. Dicamba usage in soybean and cotton steadily increased over the observed period until 2017, when dramatic increases were observed (Figure 3.7e, f). The observed increase was consistent with the timing of commercialization of dicamba-tolerant soybeans and cotton (Wechsler, 2018). Dicamba usage on wheat was relatively consistent from 1998 to 2020, with an average of approximately 10 PCT and 5 million acres treated.

Usage of dimethenamid and dimethenamid-P[13] were relatively low for corn and soybean over the observed interval, with average PCTs below 10%. Usage on wheat was not reported for the entire period, while usage on cotton was not reported until 2015. Dimethenamid and dimethenamid-P usage below minimal levels was first reported in 2017 and increased over the last 4 years. In the most recent 4 years, the average annual PCT was approximately 7%, which was approximately 800,000 acres treated annually.

Glyphosate was one of the top two, and often the most used active ingredient in corn, soybean, cotton, and wheat (Figure 3.7g, h). In corn, glyphosate usage increased over the late 1990s and into the mid-2000s following introduction of glyphosate-tolerant corn in 1996, with usage stabilizing around 80 PCT and 70 million acres annually between 2008 and 2020. Usage of glyphosate in soybeans was relatively high of the entire period examined, although usage showed steady increases from the earliest

[13] Dimethenamid is a racemic mixture containing the biologically active dimethenamid-P. Dimethenamid and dimethenamid-P are each registered as a pesticide active ingredient. Because the biological activity comes from the presence of dimethenamid-P, their usage was combined.

years reported, 1998 to 2003, with largely stable usage from 2004 to the mid-2010s, followed by a decrease in the PCT in the most recent 5 years of data. The recent decrease is likely attributable to the introduction of competing herbicide-tolerant systems and the emergence of glyphosate resistance in some weed species. Similarly, glyphosate usage in cotton steadily increased since 1998 from to the mid-2010s and has maintained a high level (i.e., average annual PCT of approximately 85% and BAT of 10 million acres; Figure 3.7). Wheat also showed increased glyphosate adoption, with increasing usage from 1998 to 2008, followed by relatively consistent usage of approximately 25 PCT applied to 12 million acres.

Some variation was observed, but metolachlor & S-metolachlor usage on corn was relatively consistent with an annual average BAT of 25 million acres and 25 PCT across the entire time period (Figure 3.7i).[14] In contrast, reported usage on soybeans for these AIs was generally limited (i.e., <5 PCT, 5 million acres) prior to 2010. Between 2010 and 2015 usage increased and usage of metolachlor and S-metolachlor on soybean averaged around 20 PCT and 19 million acres annually. Much like soybeans, usage of these AIs early in the period (i.e., 1998 and 2005 was limited, but trended upward into 2010, followed by fairly stable usage around 18 PCT and 2 million base acres treated. Usage of metolachlor and S-metolachlor on wheat was minimal, with only 2 years of minimal reported usage during the entire period (Figure 3.7j).

Usage of paraquat on corn was reported over the entire 23-year period, but was below 5 PCT in each year and the number of acres never exceeded 3 million BAT. Paraquat usage in soybeans was similarly low through 2013. From 2014 to 2020, paraquat usage on soybean increased to an annual average of nearly 6 PCT and 5 million base acres treated. In contrast, paraquat usage on cotton was variable, but annual average PCT and BAT were approximately 25% and 3 million acres, respectively. Paraquat usage was reported on wheat, but the average annual PCT was below 1% and the number of BAT averaged approximately 200,000 annually.

3.2.1.5.3 Biotechnology for Pest Management

Genetically engineered (GE) crops with pest management traits were first commercialized in 1996, after a sustained and expensive effort to develop profitable crop biotechnologies throughout the 1980s. Subsequently, a variety of GE crops have been commercially introduced. Generally, however, two types of GE crops dominate domestic markets: those that are resistant to herbicidal active ingredients (such as glyphosate or glufosinate) and those with tissues containing insecticidal substances (which the EPA refers to as “plant incorporated protectants”). The first of these GE varieties is commonly referred to as herbicide-tolerant (HT), the second is referred to as insect resistant (IR). IR crops are often referred to

[14] Metolachlor is a racemic mixture containing the biologically active S-metolachlor. Metolachlor and S-metolachlor are each registered as a pesticide active ingredient. Because the biological activity comes from the presence of S-metolachlor, their usage was combined to produce the usage trends for corn, soybean, cotton, and wheat.

as Bt crops because genes from the soil bacterium *Bacillus thuringiensis* were used to produce insect-resistance in the earliest varieties of insect resistant crops that were commercialized.

Adoption rates for HT and Bt crops differ by crop, and over time, because of differences in output prices, input prices, pest pressure, and the number and effectiveness of alternate pest control options. For instance, adoption rates for HT soybeans increased more quickly than adoption rates for HT corn (Figure 3.8). In part, HT soybeans may have been adopted more quickly than HT corn because the widespread use of herbicides, called ALS inhibitors, led to the evolution of resistance in troublesome weed species in soybean operations (Heatherly et al., 2009). HT seeds enabled soybean farmers to use herbicides, such as glyphosate, that weeds had not developed resistance to yet after crop emergence.

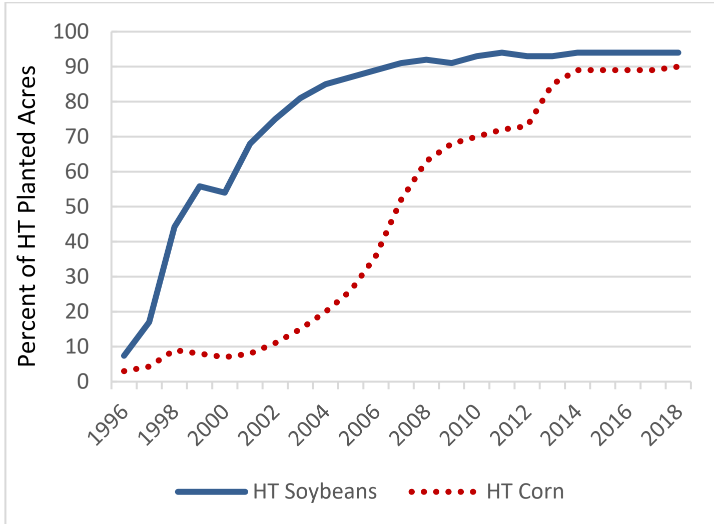


Figure 3.8. Herbicide-tolerant (HT) crops were adopted more quickly in soybeans than in corn. Source: Wechsler (2018)**.**

Adoption rates for HT corn may have increased slowly because domestic corn farmers were able to control weeds using atrazine (which was registered for use in 1958). Though weed resistance to atrazine was identified in 1968, atrazine-resistant weeds tended to be less difficult to manage and less competitive than atrazine-susceptible weed species. Post-emergent applications of atrazine do not damage corn because atrazine kills broad-leaf (dicot) plants, and corn is a grass (monocot). Atrazine cannot be used in soybean production because soybeans are broad-leaf plants, and thus are susceptible to damage by the herbicide.

Insofar as IR crops are concerned, though Bt corn was commercialized in 1996, Bt soybeans were not. In part, this may be because insects tend to be more problematic in corn than in soybean production. The percentage of corn acreage cultivated with Bt seeds was relatively low from 1996 through the turn of the century (Figure 3.9). However, there were increases in adoption rates across corn states from 2006 to 2008 and in 2013 (Dodson, 2020). These increases may be due to the commercial introduction of a trait targeting below-ground pests called corn rootworms in 2003 (prior varieties only targeted aboveground pests, such as the European corn borer), and the commercialization of new seed varieties, called SmartStax seeds, in 2009.

Not surprisingly, the widespread adoption of GE crops has had substantive impacts on the herbicide and insecticide use of domestic corn and soybean farmers. Early varieties of HT corn were tolerant of herbicide products containing either the active ingredient glyphosate or the active ingredient glufosinate. Glyphosate was less expensive than glufosinate, and easier to use. Consequently, glufosinate-tolerant seed use was relatively rare. In fact, glufosinate-tolerant soybean seeds were not commercialized until over a decade after they were developed and approved. As adoption rates of HT corn and soybeans increased, application rates of glyphosate increased, while application rates of other herbicides fell (Figure 3.10).

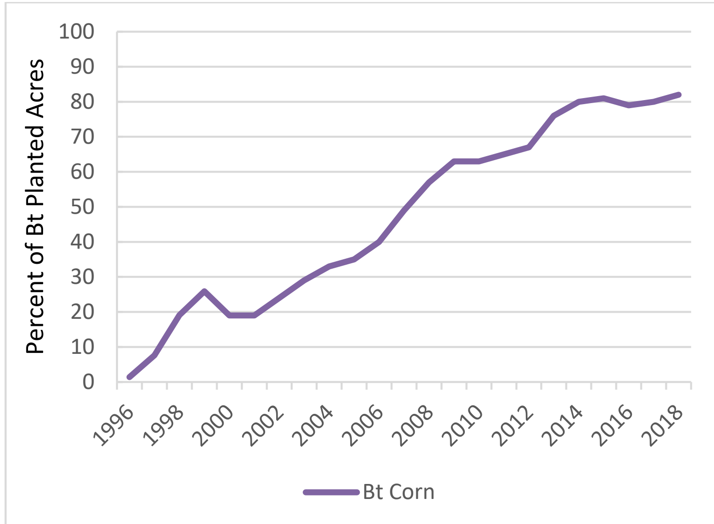


Figure 3.9. Adoption rates for corn with genetically engineered insect-resistant (Bt) traits has increased over time. Source: Wechsler (2018).

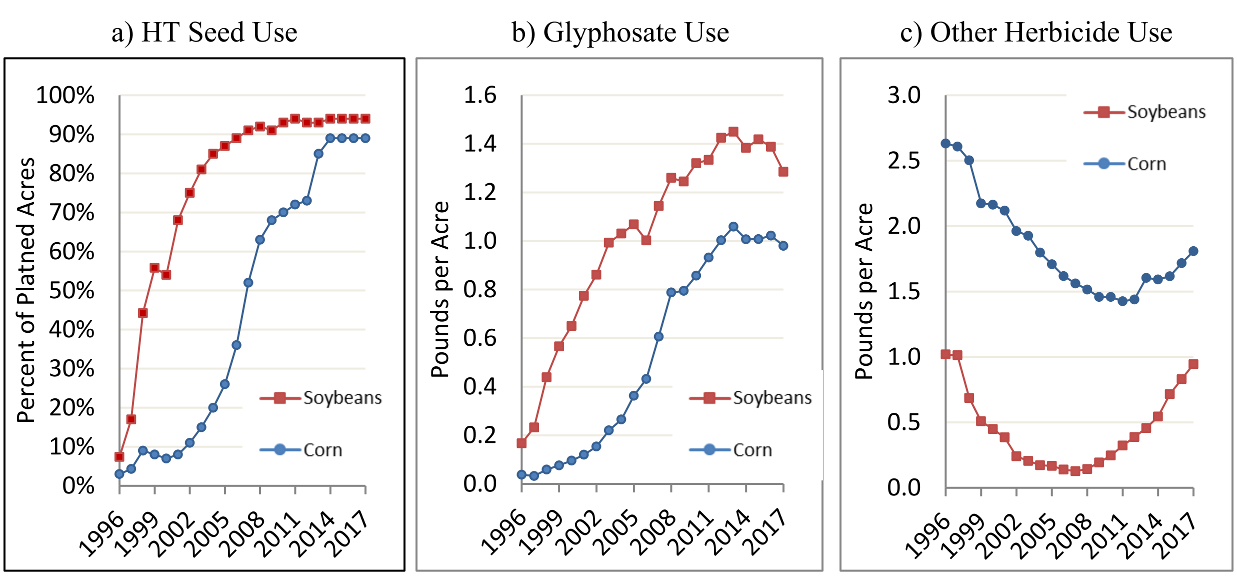


Figure 3.10. Increases in herbicide-tolerant (HT) seed use are associated with increases in glyphosate use and decreases in the use of herbicides other than glyphosate. Figure 3.10a is the same information as Figure 3.8 and is repeated for ease of comparison. Source: Wechsler (2018).

Many scientists estimate that this shift toward more glyphosate and less of other herbicides may have had net environmental and human health benefits because glyphosate is less toxic than the herbicides it replaces (Duke and Powles, 2008). However, domestic farmers' heavy reliance on glyphosate has also led to the evolution of glyphosate-resistant weeds (Duke and Powles, 2008). The evolution and spread of glyphosate-tolerant weeds has led to increases in the use of glyphosate and other herbicides.

Recently, new corn and soybean varieties have been developed that are genetically engineered to be tolerant of other herbicides such as 2,4-D or dicamba. The development and commercialization of these varieties has led to increases in 2,4-D and dicamba use (Wechsler, 2019). However, glyphosate is still regarded as less toxic than either 2,4-D or dicamba (EXTOXNET, 1996a, b, c). Dicamba is also prone to volatilization and off-field movement, particularly late in the corn and soybean growing seasons, when temperatures are high. This off-field movement, commonly referred to as "drift," has caused damages to non-genetically engineered soybeans, trees, shrubs, and other cultivated crops that may be nearby (Wechsler, 2019).

Insofar as insecticide use is concerned, the adoption of Bt corn decreased application rates of synthetic foliar and soil-applied insecticides (Figure 3.11). This decrease has had environmental and human health benefits, particularly because Bt toxins are very selective (i.e., non-toxic to non-target organisms), and many soil-applied/foliar insecticides are not. One potentially confounding, but relatively under-documented trend, is the increase in seed-applied insecticides, or insecticidal seed treatments, over time (Hitaj et al., 2020). Recent evidence suggests that many farmers are not

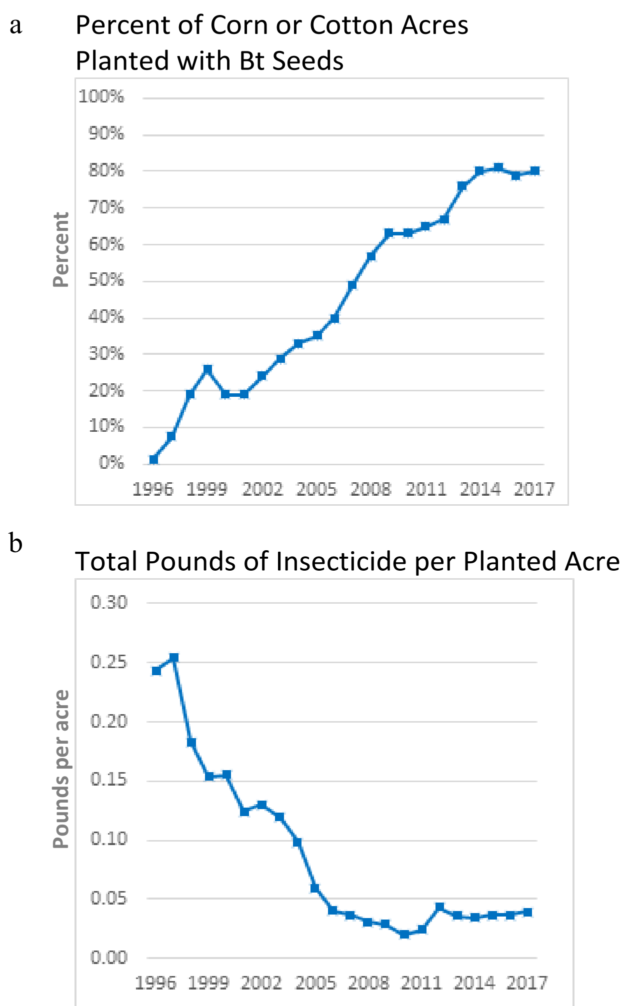


Figure 3.11. Increases in genetically engineered insect resistant (Bt) adoption rates are associated with decreases in insecticide use. Figure 3.11a is the same information as Figure 3.9 and is repeated for ease of comparison. Source: (Wechsler, 2019).

well-informed about active ingredients that are seed applied, and that most farmers are not able to purchase biotech traits unless they purchase treated seeds (Hitaj et al., 2020).

3.2.1.6 Fertilizer Use

Seed choices directly affect yield goals, which also influence farmers' fertilizer use decisions. Commercial fertilizers are a blend of nutrients containing elements such as nitrogen, phosphorus, and potassium that are necessary for plant growth. Applied annually, these nutrients are absorbed by the crop, but they are also lost to the environment through volatilization into the air, leaching into groundwater, emission to the air, and runoff into surface water as explained in subsequent chapters of this report (Ribaudo et al., 2011). These losses can be reduced by adopting best management practices (BMPs) that increase nutrient accessibility and enhance plants' ability to uptake the nutrients, and more closely match nutrient applications with agronomic needs.

Total commercial fertilizer consumption of nitrogen, phosphorus, and potassium has increased as more acres are devoted to high-yielding crop varieties and as new hybrids respond well to the more intensive use of commercial fertilizer (U.S. EPA, 2021b). Total applications of nitrogen, phosphorus, and potassium on corn and soybeans have steadily increased since 2000, reflecting expanded acreage, increases in application rates, and a higher share of acres receiving fertilizer (Figure 3.12). The sharp drop in nutrient application in total 2009 (seen in Figure 3.12b, d) can be attributed to the global financial crisis and a drastic price increase for fertilizer inputs (Roberts, 2009). Overall, fertilizer use has fluctuated over time in line with changes in cropping system implementation and fertilizer/crop prices and has shown a persistent upward trend.

The increases in fertilizer application rates, however, may or may not be associated with increased nutrient losses and volatilization, because plant yields are also increasing, thereby sequestering more of the nutrient in the plant biomass. In a recent assessment of corn and seven other crops, Zhang et al. (2021) found that the efficiency of nitrogen use (NUE, nitrogen use efficiency) was increasing for corn since roughly 2012 leading to less surplus leftover in the environment for potential loss (Figure 3.13).[15] That said, surplus nitrogen (N) for corn was still very high in 2019 (>1 teragram[16] N), and higher than all other crops evaluated. For a greater discussion of these effects see Chapter 10.

Corn and soybeans have different fertilization requirements (Figure 3.12 and Figure 3.14) partly because soybeans are able to sequester atmospheric nitrogen (N) due to close associations with bacteria (termed "N fixation"). Thus, while soybean producers add some commercial nitrogen fertilizers, corn producers apply substantially more to their crop. Fertilizer nutrient requirements for corn are based on

[15] Zhang et al. did not evaluate soybean, likely because it fixes N from the atmosphere.
[16] 1 teragram = 10^{12} grams.

expected yield and soil nutrient availability. There are many management decisions involved in the use of nitrogen fertilizers, the most important of which is selecting a rate that will maximize profit while minimizing environmental effects (Dobermann et al., 2011). The choice of an appropriate rate can be difficult due to the transient nature of nitrogen in soils. After nitrogen, phosphorus (P) is the nutrient most likely to be deficient for corn and soybean production and thus applied in fertilizers. In most cases, soils have adequate levels of sulfur, zinc, and iron to support corn production. However, in some cases, the application of these micronutrients can be yield enhancing.

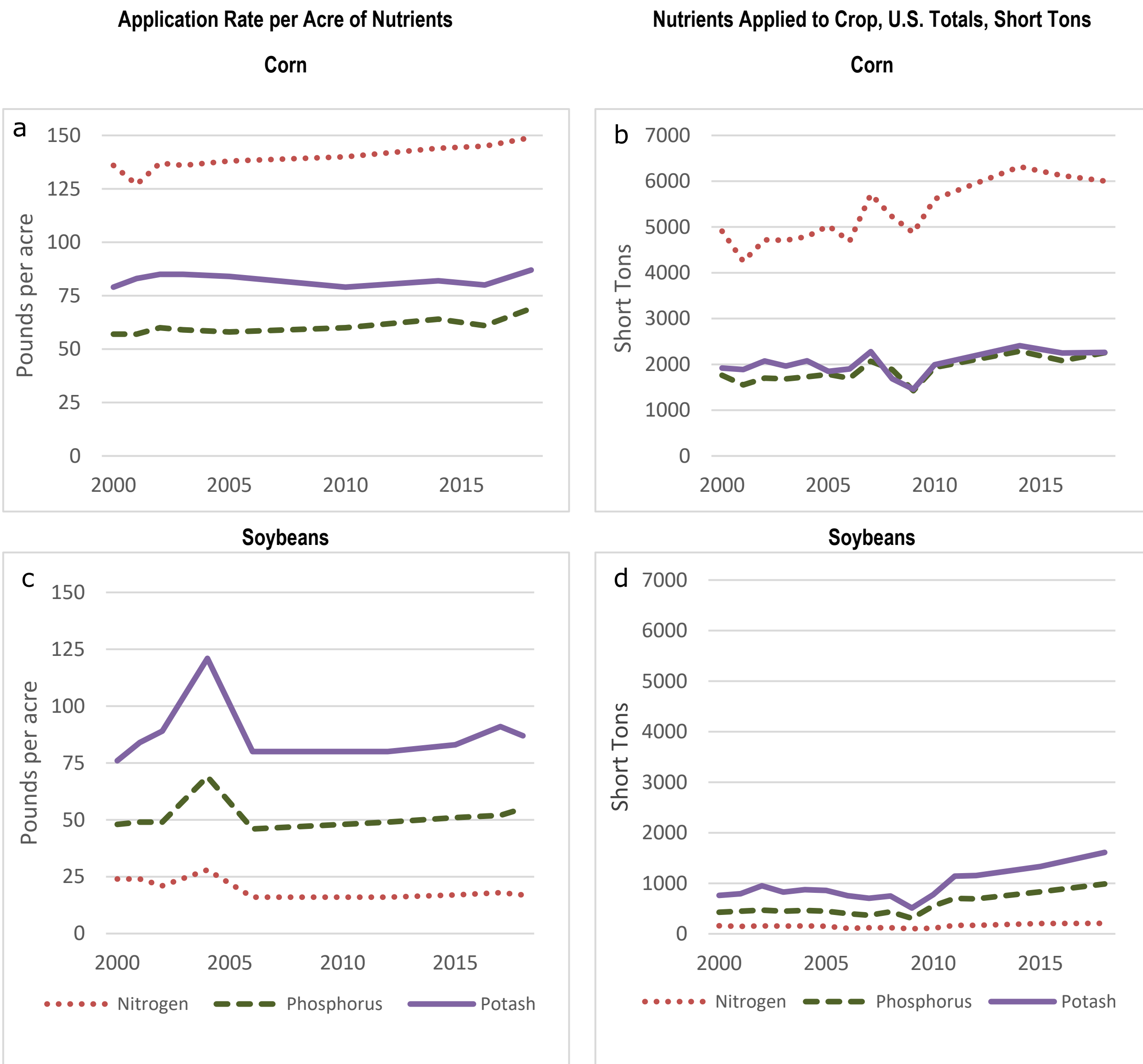


Figure 3.12. Nutrient application in corn and soybean production (1 short ton equals 2,000 pounds). Source: USDA ERS.[17]

[17] Fertilizer information are from the USDA ERS "Fertilizer Use and Price" available at https://ers.usda.gov/data-products/fertilizer-use-and-price/. Corn data are from Tables 9 and 10 and soybean data are from Tables 21 and 22.

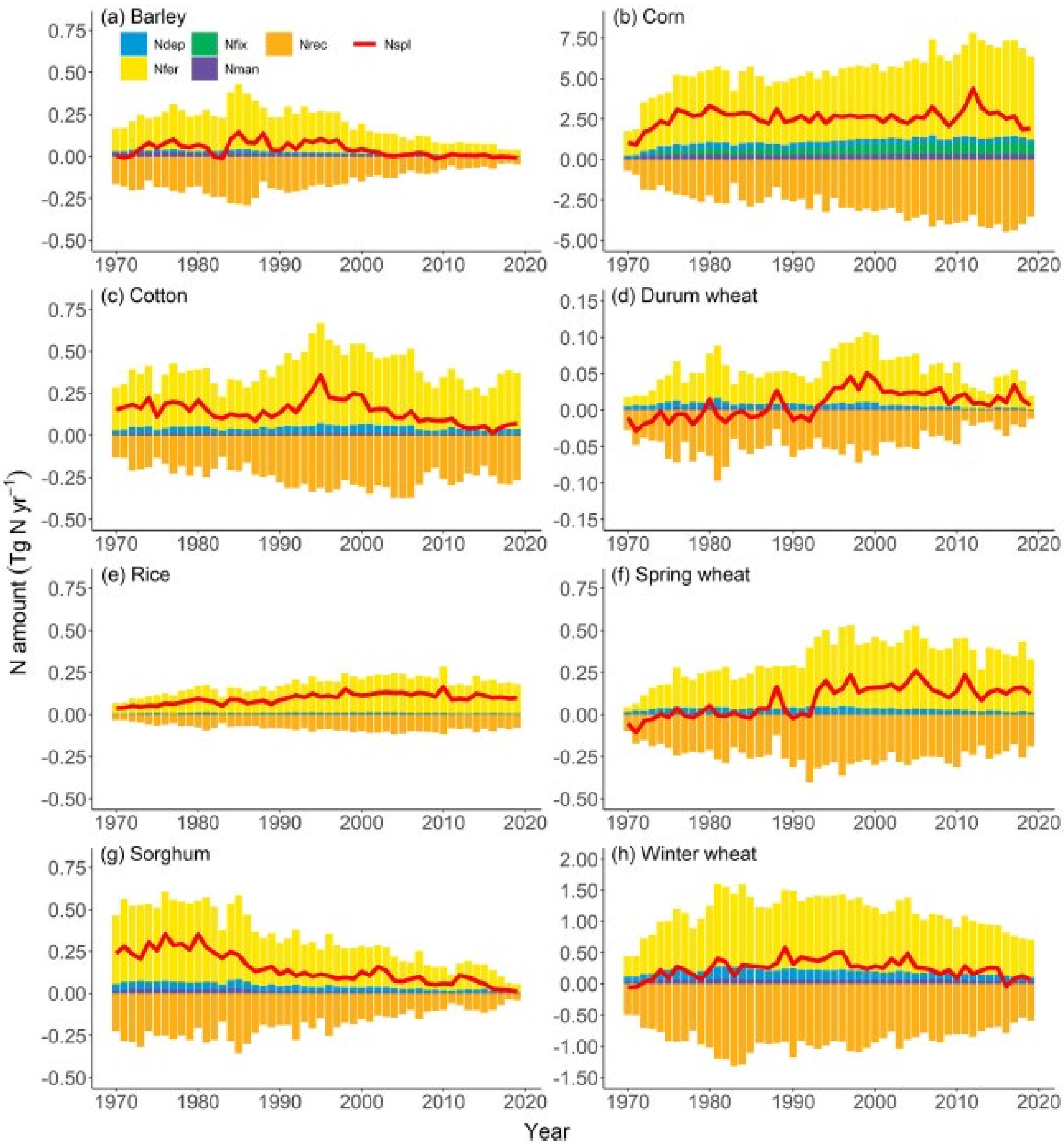


Figure 3.13. Crop-specific nitrogen budget in the United States from 1970 to 2019 for eight dominant crops (a-h). N inputs include manure (Nman), atmospheric deposition (Ndep), fertilizer (Nfer), and biological N fixation (Nfix). N output includes crop recovered N (Nrec). The timeseries of nitrogen surplus (Nspl) is shown by the red curve. The data are aggregated from the county-level crop-specific N budget. Note the y axis scale for corn, is higher than for other crops. Source: Zhang et al., 2021 (Creative Common License, CC BY-NC-ND 4.0 license).

The environmental implications of changes in fertilizer practices from biofuel crops depends on which crops are being replaced. As noted, corn generally receives more nitrogen and phosphorus than other crops that it often replaces (e.g., wheat and cotton, trends in crop switches discussed in Chapter 5), but soybean receives substantially less nitrogen fertilizer than other crops because it forms associations with bacteria that fix atmospheric nitrogen (Figure 3.13). On the other hand, soybean receives more phosphorus than both wheat and cotton (Figure 3.14). Fertilizer application rates for haylands are low by comparison with these crops.

The optimal rate of nitrogen application on a corn field depends on the soil type of the field, whether or not the field is irrigated, the crop rotation history, and the price of nitrogen. For example, in Minnesota, a nitrogen recommendation for dryland (non-irrigated) corn grown in non-sandy soil with a continuous corn rotation is between 152 to 180 pounds of nitrogen per acre. Corn grown in rotation with soybeans requires less nitrogen fertilizer, approximately 120 to 145 pounds per acre (Kaiser et al., 2011). Table 3.8 provides nitrogen recommendations for a subset of states.

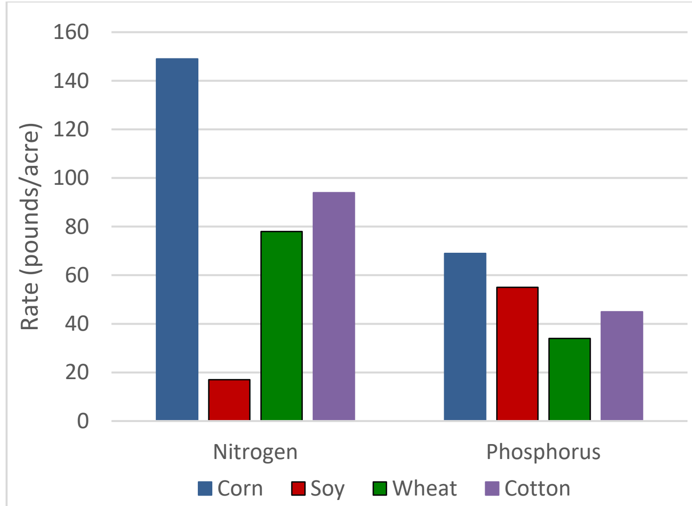


Figure 3.14. Nitrogen and phosphorus fertilizer application. Fertilizer application rates for four common crops in the Midwest (corn, soy, wheat, cotton) for nitrogen (left bars, in pounds of nitrogen per acre, lbs/acre) and phosphorus (right bars, in pounds of phosphate per acre).[18]

Table 3.8. Corn fertilizer recommendations.

	Nitrogen Recommendations in Pounds of N per Acre[1]			
	Corn Following Soybeans		Corn Following Corn	
State	Rate[2]	Range[3]	Rate	Range
Iowa	140	126–153	188	174–204
Illinois	180	166–194	193	184–210
Minnesota	130	120–145	165	152–180
Michigan	151	137–161	156	151–163
Ohio	180	164–196	189	172–206

Source: Corn Nitrogen Calculator (ISU Extension and Outreach, 2022).
[1] Recommended nitrogen amounts refer to a price ratio of 0.10, where the dollars per pound of nitrogen is divided by the bushel sale price of corn.
[2] Rate is the pounds of nitrogen per acres that provides the maximum return to nitrogen.
[3] Range is the range of the most profitable nitrogen rates that provide a similar economic return.

[18] Data are from USDA ERS for all crops available in the Fertilizer Use and Price Dataset (https://www.ers.usda.gov/data-products/fertilizer-use-and-price.aspx). Data are averages for the U.S. for the most recent year available at the time of writing (i.e., 2017 for wheat and cotton, 2018 for corn and soybean).

The USDA CEAP-2 report previously discussed also reported on trends in conservation practices associated with nutrient management for U.S. agriculture, though not specific to individual crops (USDA NRCS, 2022). The CEAP-2 report found that adoption of variable rate technology (VRT) increased from 12.6 to 51.2 million acres between 2003–2006 and 2013–2016, and use of enhanced efficiency fertilizers (EEF) increased from 11.7 to 74.1 million acres over the same period, making up over a quarter of cultivated cropland by CEAP-2.[20] The largest increases in total acres for both technologies were in the Midwest. On the other hand, rates of average nitrogen application rates on cultivated cropland increased by 7% (from 73 to 78.5 pounds per acre per year) between CEAP-1 and CEAP-2, and rates of average phosphorus application increased by 15% (from 16.2 to 18.6 pounds per acre per year). This could be partly due to the increase in corn acreage and soybean (phosphorus) nationally over the period. There were also shifts in the method and timing of fertilization between CEAP-1 and CEAP-2. There was a decrease in methods that incorporate nutrients in the soil (e.g., knifing, injection) for both nitrogen (by 29%)

Box 3.1. Innovation in Crop Production

There have been significant advances in agricultural production systems over the past decade that can improve overall environmental performance and lower the overall environmental footprint of biofuels. Technologies and practices such as conservation tillage or no-till, enhanced efficiency fertilizers (EEFs), cover crops, buffer strips, and other conservation practices form the core of "climate-smart" farming systems for U.S. row crops.

EEF is a term for new formulations that control fertilizer release or alter reactions in the soil that reduce nutrient losses to the environment. EEFs and other next generation product technology innovations are an important component of a system of conservation practices that may help reduce the impacts from row crop agriculture on the environment, while maintaining or increasing agricultural productivity and profitability. EEFs may improve water quality through reduced nitrogen leaching and reduce emissions from nitrous oxide—a powerful GHG and the largest source of GHG emissions from the U.S. agriculture sector—by providing nitrogen more efficiently to meet plant nutrient demands. Recent data from the USDA's CEAP show promising trends. CEAP recently reported increased adoption of EEFs between 2003–2006 and 2013–2016, with the acreage where these innovative fertilizers are applied increasing from 4% of cropland in 2003–2006 (11.7 million acres) to 26% (74.1 million acres) in 2013–2016 (USDA NRCS, 2022).

Similarly, as discussed in section 3.2.1.2, the application of cover crops is increasing, albeit slowly. Cover crops, which are typically added to a crop rotation in between two commodity or forage crops, provide living, seasonal soil cover with a variety of on-farm benefits, such as increased soil moisture capacity, improved nutrient cycling, and weed suppression. Cover crops can also reduce sediment loss, nutrient runoff, and leaching; reduce flooding; and store carbon in the soil (USDA NRCS, 2022). For the most recent available estimates, (Wallander et al., 2021) found that cover crops were used on corn and soybean only 1 out of 4 years on a majority of acres.

USDA recently launched a $1 billion initiative to spur continued advances and expansion of these technologies.[19] Growing interest in "Climate-Smart" commodity production could facilitate the transformation of row crop production, improving overall environmental performance of row crop production while simultaneously lowering the environmental footprint of derived products such as biofuels.

[19] https://www.usda.gov/media/press-releases/2022/02/07/usda-invest-1-billion-climate-smart-commodities-expanding-markets

[20] VRT uses precision guidance systems to allow for improved placement of nutrients and the ability to apply nutrients to actively growing crops. EEFs are fertilizers (often N-based) that use often biochemical or physical

and phosphorus (by 24%), which potentially could lead to larger losses. There was also a decrease in the nitrogen and phosphorus applied at planting, and an increase in pre- and post-plant, representing a shift away from the largest plant demand and thus a higher potential for loss. Fertilizers applied as manure were also increasing, though was often used in combination with commercial fertilizers and at higher rates. Acres receiving manure and commercial fertilizer were reported to have nutrient application rates nearly twice that of acres receiving only commercial fertilizers, and almost a third higher than acres receiving manure alone.

3.2.1.7 Harvest Dates

Field corn used in biofuel production is harvested at a later stage in plant development than sweet corn used for human consumption. Corn kernels achieve physiological maturing when a black film develops at the tip of a kernel (known as black layer). The black layer formation usually indicates a kernel moisture of approximately 30% (Daynard and Duncan, 1969). Corn can be harvested any time after the black layer formation, but it cannot be stored until kernel moisture is under 15%.

Farmers may choose to harvest wetter corn at 20 to 30% moisture and use a grain dryer to dry the corn down to an acceptable storage moisture level. If the farmer chooses to let the corn dry in the field, there is a risk of additional yield loss before harvesting the crop (Wright et al., 2004). The typical time frame for corn harvest in several high corn-producing states is available in Table 3.9.

Soybean harvest is also determined by crop moisture. Soybeans contain 45 to 55% moisture when mature and must dry down before being harvested. When the soybean plant is mature, the seeds, pods and stem turn yellow. Seed moisture begins to drop and reaches approximately 33% when the plant turns brown and the leaves drop from the plant. The plant is typically ready for harvest four to five days later, when the soybeans are between 13 and 15% moisture. Harvesting at 13 to 15% moisture maximizes weight of the crop and minimizes harvest losses (Rahman et al., 2004). The typical time frame for soybean harvest in several high soybean producing states is available in Table 3.10.

approaches to slow the release of N in the soil so that more is available to the plant and less is lost to the environment. See Box 3.1: Innovation in Crop Production.

Table 3.9. Corn harvest dates for top 5 corn states (planted acreage). Source: (USDA-NASS, 2010).

State	Harvest Dates		
	Begin	Most Active	End
Iowa	Sep. 21	Oct. 5–Nov. 9	Nov. 21
Illinois	Sep. 14	Sep. 23–Nov. 5	Nov. 20
Nebraska	Sep. 19	Oct. 4–Nov. 10	Nov. 20
Minnesota	Sep. 27	Oct. 8–Nov. 8	Nov. 23
Kansas	Sep. 1	Sep. 10–Oct. 25	Nov. 10

Table 3.10. Soybean harvest dates for top 5 soybean states (planted acreage). Source: (USDA-NASS, 2010).

State	Harvest Dates		
	Begin	Most Active	End
Illinois	Sep. 19	Sep. 26–Oct. 26	Nov. 7
Iowa	Sep. 20	Oct. 1–Nov. 1	Nov. 10
Minnesota	Sep. 20	Sep. 27–Oct. 20	Oct. 31
North Dakota	Sep. 17	Sep. 30–Oct. 31	Nov. 5
Indiana	Sep. 20	Oct. 1–Nov. 1	Nov. 10

3.2.1.8 Crop Use

Corn and its derivatives are used in many products. Historically, corn was primarily used as feed for livestock, and other industrial uses such as components in soft drinks, cereal, crayons, and other commercial goods (Figure 3.15). Corn used for ethanol production has increased significantly since 1999/2000 (0.57 billion bushels), but remained relatively steady from 2010/2011 through

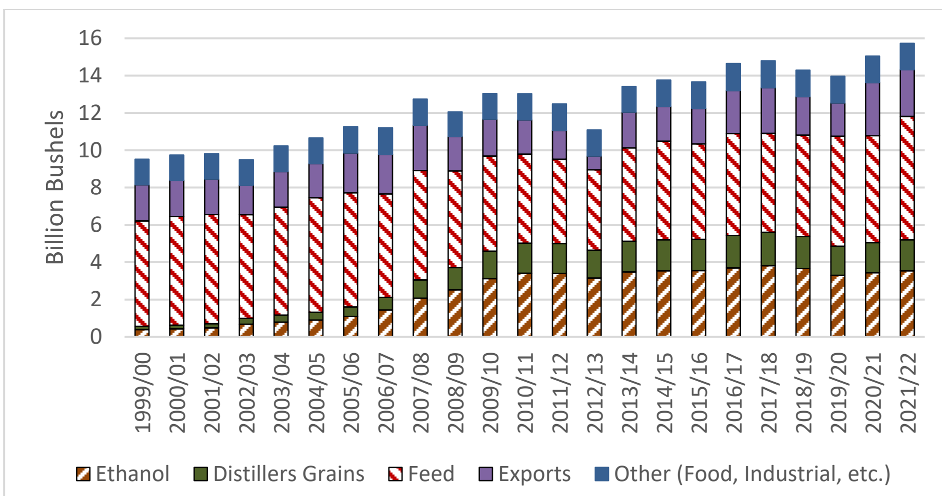


Figure 3.15. Corn end use by marketing year from 1999/2000 to 2020/2021.[21]

[21] Marketing years are typically reported for crop end use, are crop-specific, and often span calendar years. Corn and soy marketing years are the same, from September 1 to August 31 (https://www.ers.usda.gov/data-products/feed-

2018/2019, before decreasing slightly in 2019/2020. Since 2013/2014 corn used for ethanol production (including distillers' grains) accounted for a low of 4.9 billion bushels of corn in 2019/2020 and a high of 5.4 billion bushels in 2017/2018. Corn used for ethanol production as a percentage of overall corn production increased significantly since 1999/2000 (6%) but has remained relatively steady since 2009/2010 (35% to 43%) when the E10 blend wall was approached (see Chapter 1 Figure 1.4). Corn used for feed varied from 1999/2000 to 2019/2020, with a low of 4.32 billion bushels of corn used for feed in 2012/2013 and a high of 6.14 billion bushels used for feed in 2004/2005. Ethanol production facilities generally produce both ethanol and distillers' grains, along with other co-products such as corn oil and carbon dioxide (see section 3.4.1.1). These numbers reflect the total quantity of corn used at ethanol production facilities. Therefore, they may overstate the quantity of corn used for ethanol production and underestimate the quantity of corn used for feed. Corn exports and other uses have been relatively stable from 1999/2000 to 2019/2020. All uses of corn decreased in 2011/2012 and 2012/2013 due to relatively low corn production in 2012/2013 caused by drought (Rippey, 2015).

Soybeans also are used directly for animal feed as well as processed for use in a wide variety of products. Soybeans used in the United States are generally first processed at crushing facilities to separate the oil from the meal. Soybean meal is mostly used for livestock feed and as a supplement for food products. Soybean oil is used primarily for food, feed, and other industrial uses, while the remainder of the oil is used for biofuel production (primarily biodiesel and renewable diesel) or for export. The use of soybeans increased in most sectors from 1999/2000 to 2020/2021 (Figure 3.16). On an absolute basis, the largest contributor to this was from an increase in whole bean exports around the world—particularly to China. The use of soybean oil to produce biofuel increased from 0.09 billion pounds in 2001/2002 (the first year USDA reports data for soybean oil used for biofuel production) to approximately 8.85 billion pounds in 2020/2021. Soybean oil used for biofuel production as a percentage of overall soybean oil domestic consumption increased from less than 1% in 2001/2002 to 36% in 2020/2021. Domestic consumption of soybean oil for reasons other than biofuel production has been relatively stable between 13 and 15 billion pounds from 2008/2009 to 2020/2021. Soybean meal production increased from approximately 38 million tons in 1999/2000 to approximately 51 million tons in 2020/2021. Exports of

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grains-database/documentation/). Data for domestic corn end use and share of soybean oil used for biodiesel production from USDA ERS U.S. Bioenergy Statistics, 2022; corn use data available in ERS Table 5 and share of soybean oil used for biodiesel production from Table 6. Data for domestic soybean end use from USDA ERS Oil Crops Yearbook (USDA, 2021); soybean end use from ERS Table 3. Note that the ERS data does not list corn used to produce ethanol and distillers' grains separately as both are produced from the same process. This graphic assumes that 68% of the corn used by ethanol production facilities is used to produce ethanol and 32% of corn used by ethanol plants produces distillers' grains. This proportion is based on the quantity of distillers' grains produced per bushel of corn processed at a dry mill corn ethanol production facility (18 pounds of distillers' grains per bushel of corn), and is shown in shown in Figure 3.15 to provide an indication of the quantity of corn processed at an ethanol plant that is available to the livestock market in the form of distillers grains.

whole soybean increased from 1 billion bushels (approximately 30 million tons) in 1999/2000 to 2.3 billion bushels in 2020/2021 (approximately 69 million tons), while the use of soybeans for seed and feed was relatively stable from 1999/2000 through 2020/2021.

3.2.2 Non-Crop Feedstocks: Fats, Oils, and Greases (FOGs)

FOG is a descriptive term that covers animal byproducts and grease from food-handling operations and are typically processed at rendering facilities for use in various industries. FOGs include animal fats (e.g., tallow, white grease, poultry fat) obtained from slaughterhouse and livestock farm waste, used cooking oil (UCO) generated at commercial and industrial cooking operations, and trap/interceptor grease recovered from traps installed in the sewage lines of restaurants/food-processing plants and wastewater treatment plants. FOGs may have highly complex and varying supply chains depending on the identity of the FOGs. There were no other non-crop biofuels that dominated the U.S. pool over the historical period (Chapter 2, section 2.3, but see Box 3.2 and Box 3.3). The rest of the section briefly summarizes the production and logistics for FOGs.

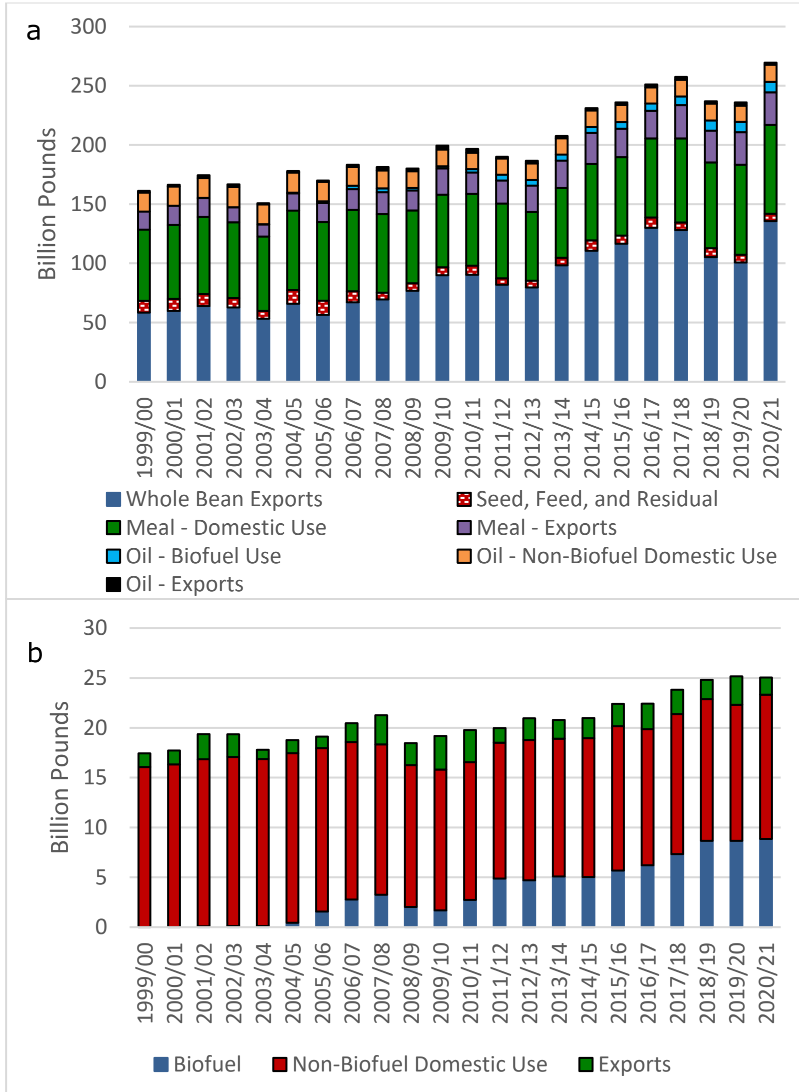

Figure 3.16. Soybean end uses by marketing year from 1999/2000 to 2020/2021. Shown are the various uses for soybean (a) and for soybean oil (b). See source information for Figure 3.14.

FOGs are conventionally managed by animal rendering operations and collection companies (haulers) that remove it from commercial, institutional, or industrial food-processing facilities, slaughterhouses, and farms. Typically, FOGs are not used in their raw form—industries purchase purified material from rendering plants. The rendering plants convert raw material (animal byproducts and

Box 3.2. Biogas

Biogas is the gaseous product of anaerobic digestion, a biological process in which microorganisms break down biodegradable material in the absence of oxygen. Biogas is comprised primarily of methane (50–70%), carbon dioxide (30–40%), and trace amounts of other compounds or elements such as water, nitrogen, hydrogen, and others. Biogas is produced from many sources. These include organic material disposed of at landfills, animal manure, wastewater sludge, and food waste. Biogas can also be produced from lignocellulosic material (e.g., crop residues and herbaceous energy crops) through either dry fermentation or thermochemical conversions that have limited application in the United States. Biogas is used primarily to produce heat and electricity. It is also upgraded to pipeline quality gas to substitute for fossil natural gas in residential, commercial, and industrial applications, as well as a transportation fuel in the form of compressed or liquefied renewable natural gas (CNG/LNG, see Chapter 2, Tables 2.1 and 2.2) used in natural gas vehicles. The leftover liquid and solid digested materials (digestate) may be used as a soil conditioner or compost.

As of August 2020, there were about 565 operating landfill gas (LFG) projects in the United States. Most of these facilities (about 88%) are producing electricity or using biogas directly on site while the remaining 12% of facilities are producing CNG/LNG from biogas for pipeline injection or local use. As of September 2020, there were about 282 manure digesters currently operating in the United States. Most of these plants produce electricity and heat. As of February 2021, about 26 of those plants produce compressed renewable natural gas (RNG) as a transportation fuel and 14 plants inject RNG into the pipeline. Of the 1,269 wastewater treatment plants using anaerobic digestion on site for sludge treatment, only around 860 use the generated biogas. Most of these plants produce electricity and heat for onsite use and very few produce CNG/LNG from their biogas. About 198 food waste digesters were identified in 2018. These were either stand-alone digesters processing only food waste or codigestion facilities processing manure or sludge as well. Most of these facilities use the generated biogas to produce electricity and heat and few generate transportation fuel.

The use of biogas as a transportation fuel has increased substantially over the past 11 years from nearly none in 2011 to just over 665 million ethanol-equivalent gallons in 2022 (Chapter 2, Table 2.4). The use of biogas as transportation fuel is projected to increase to approximately 1.3 billion ethanol-equivalent gallons in 2025 (Chapter 2, Table 2.4). This is due to biogas qualifying as a feedstock for cellulosic biofuel under the RFS Program in 2014. California's Low Carbon Fuel Standard (LCFS), designed to decrease the carbon intensity of California's transportation fuel pool and provide an increasing range of low-carbon and renewable alternatives, is another driver for increased use of CNG/LNG from biogas as a transportation fuel. Currently, most of the cellulosic biofuel volumes under the RFS Program are being met through the use of CNG/LNG from biogas. Although biogas did not meet the criteria for inclusion in the RtC3, if trends continue it may be included more substantially in future reports.

cooking/trap greases) into valuable products (e.g., yellow grease [rendered UCO], brown grease [rendered trap grease], and animal fats) used by various industries (e.g., animal feed, pharmaceuticals, cosmetics, lubricants, plastics, biofuels). While animal fats and UCO are primarily processed at rendering plants, trap grease is handled in various ways. In addition to rendering, trap grease is landfilled, incinerated, anaerobically digested, or composted. It is these processed FOGs that ultimately are transported to biorefineries to produce biodiesel, renewable diesel, and jet fuel. It has been estimated that about 5.9 million tons of inedible FOGs (excluding edible fats such as edible tallow and lard) are produced in the United States annually (Milbrandt et al., 2018). Animal fats contribute more than 50% of the total inedible FOG production, brown grease contributes about 28%, and yellow grease about 19%. The geographic distribution of yellow and brown grease follows animal populations—highly populated areas are also locations of large grease production. The top five states for animal fat production are Nebraska, Texas, Kansas, Iowa, and North Carolina. These states have the highest production and slaughter of cattle, hogs, or poultry in the country. Naturally, there is a high concentration of rendering plants in these and other states with relatively high animal production.

Regarding current utilization of FOGs in the United States, it has been estimated that a significant amount of yellow grease, poultry fat, inedible tallow, and choice white grease (inedible pork fat) is currently used by various industries (including for biofuel production) and for export, while brown grease is largely underutilized. From 2009 through 2020 the U.S. Energy Information Administration (EIA) reported quantities of FOGs used for biodiesel production. Consumption of yellow grease for biodiesel production has increased from about 78,000 tons in 2009 to a high of about 720,000 tons in 2019. Animal fat use for biodiesel production has increased from 530,000 tons in 2009 to about 600,000 tons in 2020. These numbers do not include yellow grease and animal fats used to produce renewable diesel or other biofuels. Starting in 2021 EIA began reporting quantities of FOGs used to produce all biofuels, including both biodiesel and renewable diesel. In 2021 about 1.7 million tons of yellow grease was used to produce biofuel and about 1.2 million tons of animal fats were used to produce biofuels. Brown grease is generally not used in biofuel production due to high water and free fatty acid content.

3.3 Crop Feedstock Logistics

Crop feedstock logistics encompass the steps involved with getting the feedstock material from where they are produced to the biorefinery.[22] For agricultural commodities used to produce fuel, this means from the farm to the biorefinery. In most cases, the systems utilized for the transportation and processing of the agricultural feedstocks when utilized to produce biofuel are basically the same as the logistics system utilized when the crops are utilized for food or livestock feed. In general, the operations that are covered under the umbrella of feedstock logistics are harvest, storage, and transport. However, in some cases additional preprocessing is needed before the material can be utilized in the fuel conversion process.

3.3.1 Corn Grain for Ethanol

The corn ethanol logistics system begins with the harvest of the grain. Harvests are carried out using machinery called combines. Combines are separated into five classes based on the engine horsepower (HP); however, combines are no longer produced in the smaller size classes (i.e., below 340 HP). Typically, the combines are fitted with "headers" to harvest multiple rows, with the most common being either 6, 8, or 12 row harvesting headers.

[22] There is no logistics section here for FOGs because that was already discussed as part of the collection process in section 3.2.2.

In addition to the combine, grain carts are also employed in the harvest of corn. The purpose of the grain cart is to increase the efficiency of the harvest by allowing the combine to be unloaded without stopping. Additionally, the grain carts are used to transport the corn from the field and load the trucks.

Upon harvest the corn grain is moved from the field to either short- or long-term storage before being moved into the market. Approximately 60–70% of the annual harvest is placed in storage, while the remaining 30–40% is marketed directly off the field during the harvest months. The storage of the corn grain is undertaken for the primary reason of maintaining market flexibility, as the potential uses and prices fluctuate throughout the year. The production of ethanol has necessitated farmers to utilize more storage capacity in order to access this market. In general, ethanol producers would rather not store large volumes of the grain on premises but rather purchase the grain as they need it throughout the year. In order to increase access to storage, the farmers either use on-farm storage or rent storage space from neighbors or commercial elevators.

In general, corn ethanol plants are built in areas that have a large amount of feedstock resource. This results in a large portion of the processing and conversion happening near the fields in which the grain is harvested. As on-farm storage increases, much of the corn is transported by truck directly to the biorefinery plants, bypassing local grain elevators. This is a departure from more traditional uses of the grain, where multi-modal transportation is commonplace, using rail, barge, and truck. The impact of ethanol production and the increase of local processing of the corn grain is evident in the 7% reduction in rail use for domestic transportation from 2007 to 2010, while truck transport increased by 3% over the same period.

3.3.2 Soybean for Biodiesel

The soybean for biodiesel logistics system also begins with the harvest of the grain. Similar to the harvest of corn, soybeans are also harvested with a combine. However, instead of the harvesting headers being sized based on row spacing, they are classified based on their width. The width of soybean harvesting headers range from 20 to 40 feet, with the vast majority being over 30 feet wide. Also like the harvest of corn, soybean harvest also takes advantage of the efficiency gains that come from using grain carts.

After the soybeans are harvested there are five primary destinations when the soybeans leave the field: on-farm storage, elevator, barge terminal, shuttle elevator,[23] or crushing plant (Informa Economics, 2016). During the harvest approximately 25% of the soybeans are shipped off farm, with the remaining 75% of the harvested soybeans placed in storage on the farm or sold to an elevator (Informa Economics, 2016). Like the corn grain logistics system, on-farm storage provides a way to harvest faster without

[23] Shuttle elevator is a term for larger grain elevator that is serviced by rail as opposed to smaller non-shuttle elevators that are often serviced by truck (Ndembe and Bitzan, 2018).

needing to wait for trucks to haul the material. Instead, the soybeans are stored temporarily on the farm to act as a buffer between the harvest and transportation to either offsite storage or into the market. Also, like corn, on-farm storage is also used to try to take advantage of the dynamic market.

Biodiesel production requires that the beans be crushed and pressed in order to extract the oil contained within. Prior to crushing, the soybeans must be transported from either the farm or other storage facilities to the crushing facilities. Approximately 51% of the total soybean harvest is sent to crushing facilities, with 10% of the soybeans sent directly from the farm during harvest and the remaining 41% finding their way from the various off-farm elevators and terminals (Informa Economics, 2016). The transport of soybeans from the farm is carried out almost entirely by trucking, using either straight trucks or tractor-trailers. As operations have gotten bigger and more sophisticated, the size of truck has increased. Currently, approximately 80% the transportation from the farm is carried out by tractor-trailer. The movement of soybeans from off-farm sites to crushing facilities uses a more diverse set of transportation options. The transport from elevators to crushing facilities uses primarily truck with a portion of the transport being made by rail. However, shuttle elevators utilize railways exclusively while the aptly named barge terminals use barges exclusively. Like corn, transportation of soybeans is seeing a transition to more local processing for fuel production and the use of rail and barge are reducing with time.

3.4 Biofuel Production

3.4.1 Ethanol Production

The total ethanol production capacity in January 2022 in the United States was 17,380 million gallons per year (MGY) and there was a total of 192 ethanol facilities in operation.[24] With both new facilities and the expansion of existing facilities, the total ethanol production capacity will likely increase. The average capacity per corn ethanol biorefinery has increased from 31.9 MGY to 79.5 MGY between 1998 and 2020 (RFA, 2020). Ethanol biorefineries vary widely in their size, technology, and energy sources, and thus so does their efficiency of conversion of corn to ethanol. They are distributed around the country but are concentrated in the corn-producing regions of the Midwest (Figure 3.17a). Almost all corn ethanol produced is from corn starch, though there is increasing effort to produce ethanol from the corn fiber and biodiesel/renewable diesel from corn oil as well (see Box 3.3).

[24] https://www.eia.gov/petroleum/ethanolcapacity/.

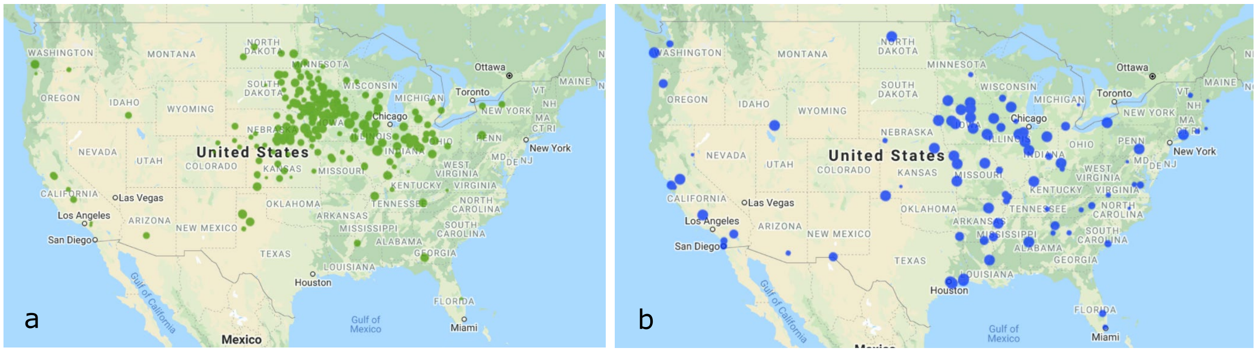

Figure 3.17. Map of ethanol refineries (a, green dots) and biodiesel refineries (b, blue dots) in the United States. Dot size corresponds to capacity. Maps are from the NREL Biofuels Atlas (https://maps.nrel.gov/biofuels-atlas).

Box 3.3. Gen 1.5. - Corn Fiber Ethanol and Distillers Corn Oil Biodiesel/Renewable Diesel

With projections that 1.5 billion gallons of ethanol could be produced from the 12 million tons of corn fiber currently available in U.S. dry-mill facilities, many companies have been working to develop this technology.

Conversion of corn fiber to cellulosic ethanol requires pretreatment of the fiber to modify the underlying structure and convert the sugars to ethanol. Due to the very low lignin content in corn fiber, pretreatment conditions are milder than those for treating other cellulosic feedstocks, like lignocellulosic biomass, and require lower concentrations of enzymes. Unlike starch ethanol production, corn fiber has more complicated sugar content and conversion of all types of sugars is necessary in the ethanol production process. To date this technique is not widespread.

Distillers' corn oil (DCO) is a co-product of the corn-based ethanol production process used as an animal feed ingredient or biodiesel feedstock. About 90% of ethanol plants in the United States have added dry fractionation technology at the front end of their plant to extract nonedible corn oil at a rate of about 0.6 pounds/bushel.[25] DCO production in 2022 was about 2.1 million tons.[26] It is the most-used domestic feedstock for biodiesel production after soybean oil and FOGs, with 1.5 million tons consumed in 2022.[27] Given its lower-than-soybean-oil carbon intensity (CI) score under the LCFS (31 vs 55)[28] and relatively low price, DCO is an attractive feedstock not just for the biodiesel industry but also for the developing renewable diesel industry in the United States. Both biodiesel and renewable diesel (as well as renewable jet fuel and heating oil) produced from DCO are approved pathways for biomass-based diesel (D-code 4) or advanced biofuel (D-code 5) under the RFS program.

3.4.1.1 Types of Milling (Dry vs Wet Milling)

More than 91% of the U.S. fuel ethanol is produced using the dry-mill process (with the remaining 9% coming from wet mills).[29] The main difference between the two processes is in the initial treatment of the grain (RFA, 2020). Dry milling is a process that grinds corn grain into flour and ferments the starch component of the corn flour into ethanol with coproducts of distillers' grains (DG), distillers' corn oil,[30] and carbon dioxide (CO_2). Wet-mill plants primarily produce corn grain sweeteners, along with

[25] https://www.nrel.gov/docs/fy20osti/75776.pdf
[26] https://ethanolrfa.org/markets-and-statistics/feedstocks-and-co-products
[27] https://www.eia.gov/biofuels/update/
[28] https://vitalbypoet.com/stories/voila-poet-rolls-out-co-product-for-renewable-diesel-feedstock
[29] Estimates from the USDA NASS's "Grain Crushings and Co-Products Production" available at https://usda.library.cornell.edu/concern/publications/n583xt96p.
[30] Distillers' corn oil has become an important coproduct and is often used to produce biodiesel (see Box 3.3).

ethanol and several other coproducts (such as edible corn oil and starch). Wet mills separate starch, protein, and fiber in corn grain prior to processing these components into ethanol and other products. Dry grind processes are less capital and energy intensive than their wet mill counterparts. However, they also produce fewer products. Wet mills are structured to produce a number of products, including starch, high fructose corn syrup, ethanol, corn gluten feed, and corn gluten meal. As a result, ethanol yields from wet mills are slightly lower (2.5 gallons per bushel) than from dry mill processes (2.9 gallons per bushel).

3.4.1.1.1 Dry Milling

In a dry-mill process (shown in Figure 3.18), the corn or other grain is conveyed to the grain-cleaning equipment, and debris (such as tramp metal and rocks) are removed. The corn is then milled, and after milling the corn meal is sent to a continuous liquefaction tank and mixed with hot evaporator condensate and enzyme. Continuous saccharification is carried out in a stirred tank by adding enzymes with sulfuric acid. Starch is converted to glucose using enzymes. These enzymes have improved over the years, and now convert essentially 100% of the starch to glucose, provided that the corn is finely ground and properly cooked. The saccharification reaction converting starch to glucose is typically 6 hours with addition of a caustic chemical to optimize pH. Fermentation of glucose to ethanol takes about 60 hours using yeast. Urea is provided to the yeast fermentation as a nitrogen source (Kwiatkowski et al., 2006).

The raw fermentation beer contains ethanol, water, carbon dioxide, glucose, protein, non-fermentative solids, and organic acids. In the dry-mill process ethanol concentrations by weight in the beer are typically 14–20 weight percent (Tao et al., 2014; Lualdi et al., 2011). The ethanol is concentrated and purified through a series of distillation and molecular sieve dehydration steps to produce anhydrous ethanol that can be blended with gasoline. The solid byproducts of the ethanol conversion process are dewatered and dried through a series of centrifugation, evaporation, and drying steps, in order to produce distillers' dried grains and solubles (DDGS). Most of the carbon dioxide is removed and fed to a water scrubber along with the vent streams from fermentation (McAloon et al., 2000).

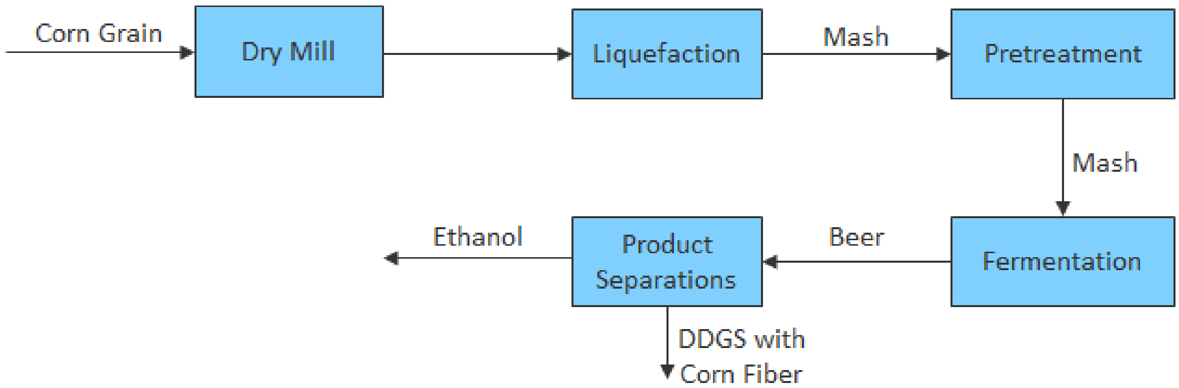

Figure 3.18. Block flow diagram of corn ethanol production from corn grain. Source: Modified from Tao et al. (2017a) (used with permission).

3.4.1.1.2 Wet Milling

In wet milling, the corn kernels are first soaked in a mixture of water and sulfur dioxide (SO_2) in a process known as "steeping" in order to allow for separation of the kernel components. Germ, fiber, gluten, and starch are separated from one another through a series of screens, cyclones, presses, and other equipment. Oil can be further extracted from the germ. The liquified mash then goes through enzymatic hydrolysis converting starch to fermentable sugars, and the sugars are fermented to ethanol, similarly to dry grind processing as described above.

3.4.1.1.3 Distillers' Grains Ethanol Production Coproducts

Growth in U.S. ethanol production has created an increased supply of its feed coproducts, DGs, DDGSs, corn gluten feed, corn gluten meal, and distillers' corn oil. Production of these feed coproducts has lessened the potential impact of corn's removal from the feed supply to produce ethanol. However, this initial growth surge of both ethanol and its feed coproducts has plateaued in recent years relative to the initial growth from 2002 to 2012 (Figure 3.19).

Distillers' grains are a byproduct of alcohol production. DGs can take many forms depending on their moisture content. As they are removed from the distillation process, DGs have high moisture content and are like a mash in consistency. In this form they can be sold for consumption by livestock within a few miles of a facility, but long-distance shipping is not economical due to their high weight and short shelf life. Dried distillers' grains (DDGs) are distiller's grains dried to a moisture content of roughly 10%. In this form they can be economically shipped long distances either by truck, rail, barge, or container. In addition, condensed distillers' solubles can be recovered from the refining process, dried, and either sold as a livestock feed supplement or added to DDGs to produce DDGS, a nutrient-rich form of DDGs. DDGSs, because of their low moisture content, are also easily shipped and stored. DGs have been used as animal feed since humans produced alcohol. Until the 2000s and the advent of large-scale use of ethanol

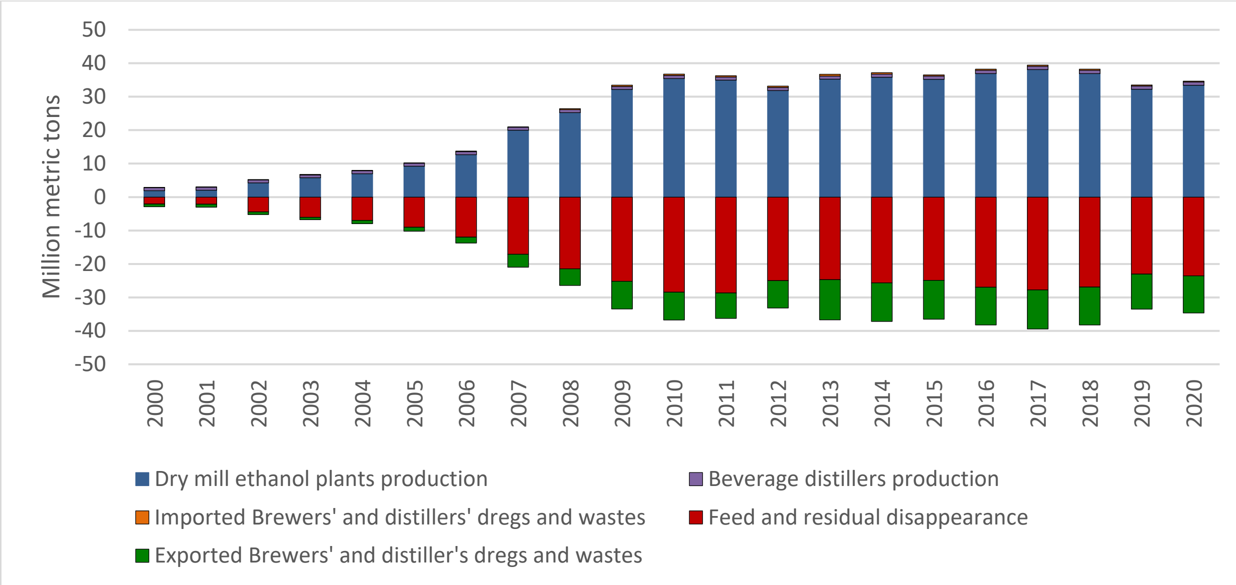

Figure 3.19. DDGS supply (positive) and disappearance (negative) from 2000 to 2020.[31] Source: USDA ERS using data from USDA, Office of the Chief Economist, World Agricultural Supply and Demand Estimates, updated April 2021.

in gasoline in the United States, the smaller volume of DDGS produced by beverage distilleries and brewers limited its use as a feedstock. Since the mid-1990s, production of DGs from non-beverage refiners has exceeded that from beverage producers (Figure 3.19). Currently, one bushel of corn processed in a dry mill produces approximately 2.9 gallons of ethanol and 15.9 pounds of DDGS (RFA 2020). The use of DGs and DDGS offset a portion of the corn production needed by the livestock industry thereby reducing the potential environmental footprint of corn ethanol. However, substituting DDGS for grain as feed for livestock is not without environmental effects (see Box 3.4). The economics of DDGS are further discussed in Chapter 4 section 4.5.1.

Box 3.4. Environmental Challenges of Feeding Ethanol Coproducts

Distiller's grains are cereal grain coproducts from ethanol production. Of the approximately 42 million metric tons of ethanol coproducts produced in 2018, 77% was consumed by beef or dairy (RFA, 2019). Ethanol coproducts are an affordable and nutritious feed ingredient for both beef and dairy cattle; 70.8% of U.S. beef feedlots report the use of wet distiller's grains in finishing diets (Samuelson et al., 2016). When DGs are used as a partial replacement for grain in cattle diets, the nitrogen, phosphorus and sulfur contents of the diets are higher than the animals' nutrient requirements (NASEM, 2016). Thus, cattle consume more of these nutrients than they can utilize, and excess nitrogen, phosphorus, and sulfur are excreted in feces and urine, creating some environmental considerations when DGs are fed. This match/mismatch of nutrient requirements of the livestock with the nutrient content of the feed is not unique to feed with DGs and is always carefully managed regardless of the source of the feed. For more details on the environmental impacts of DDGS, see the Air Quality (Chapter 8) and Water Quality (Chapter 10) chapters of this report.

[31] Marketing year September – August. Distillers' spent grains do not account for non-corn spent grains from dry or wet ethanol plants. Assumes brewers' spent grains are minor and may contain non-corn brewers' and distillers' dregs and wastes. Data for 2005-2007 table is computed from estimates contained in the WASDE and Feed Grains Database for the month prior to the update date.

3.4.2 Biodiesel

Biodiesel and renewable diesel refineries are also concentrated in the Midwest, but less so due to the more diverse feedstocks used in their production (e.g., FOGs, Figure 3.17b). Biodiesel is a renewable fuel produced through transesterification to produce chemical compounds known as fatty acid methyl esters. Biodiesel is the name given to these esters when they meet fuel quality specifications from ASTM International. Biodiesel is used in blends with petroleum diesel. Fats are main constituents of the oil feedstocks, which also contain sterols, water, odorants, and other impurities. Because of these impurities, the oil cannot be used as fuel directly. During the transesterification process, approximately 100 pounds of oil or fat are reacted with 10 pounds of a short-chain alcohol (usually methanol) in the presence of a catalyst (usually sodium hydroxide [NaOH] or potassium hydroxide [KOH]) to form 100 pounds of biodiesel and 10 pounds of glycerin (or glycerol, Figure 3.20). Glycerin, a coproduct, is a sugar commonly used in the manufacture of pharmaceuticals and cosmetics.[32] The biodiesel yield ranges from 7.3 to 7.4 pounds of feedstock per gallon of biodiesel.[33]

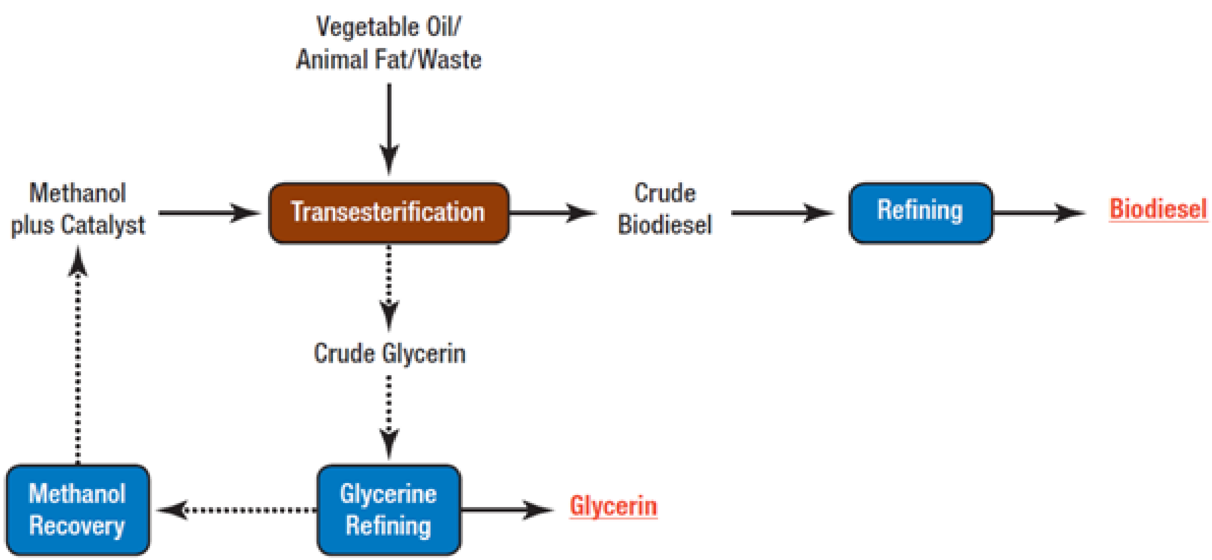

Figure 3.20. Block flow diagram of biodiesel production process. Source: (AFDC, 2022a).

3.4.3 Renewable Diesel

Renewable diesel is made through a variety of biological, thermal, and chemical processes and from a variety of biomass sources (like UCO, animal fats, algae, and vegetable oils) (Moriarty et al., 2020). Renewable diesel and petroleum diesel meet the same fuel quality ASTM specification. Since the fuels meet the same specification, renewable diesel can be used in existing infrastructure and vehicles/engines without modifications.

[32] DOE (AFDC, 2022b)
[33] Estimates available from https://afdc.energy.gov/files/pdfs/3229.pdf .

Renewable diesel production technologies (e.g., hydrotreating, deoxygenation, isomerization, hydrocracking) are at a relatively high maturity level and are commercially available and are only briefly summarized here (Figure 3.21). These processes are commonly used in today's refineries to produce transportation fuels and have been used by petroleum refineries for some time but are relatively new to the production of renewable diesel. First, catalytic hydrogenation could be used to convert liquid-phase unsaturated fatty acids or glycerides into saturated ones with the addition of hydrogen. The next step is to cleave the propane and produce free fatty acids. These reactions require high temperatures (250–260°C) and high pressure to maintain the reactants in liquid phase. To meet the fuel specification, it is often required to hydrocrack and hydroisomerize the hydrotreated vegetable oils to meet the specifications for renewable diesel. Hydrocracking reduces the length of the carbon chains to lengths typically found in diesel fuel and isomerization takes the straight-chain hydrocarbons and turns them into the branched structures to reduce the freeze point of the finished fuel. The hydroisomerization and hydrocracking processes are followed by a fractionation process to separate the mixtures to paraffinic kerosene (HRJ SPK), paraffinic diesel, naphtha, and light gases.

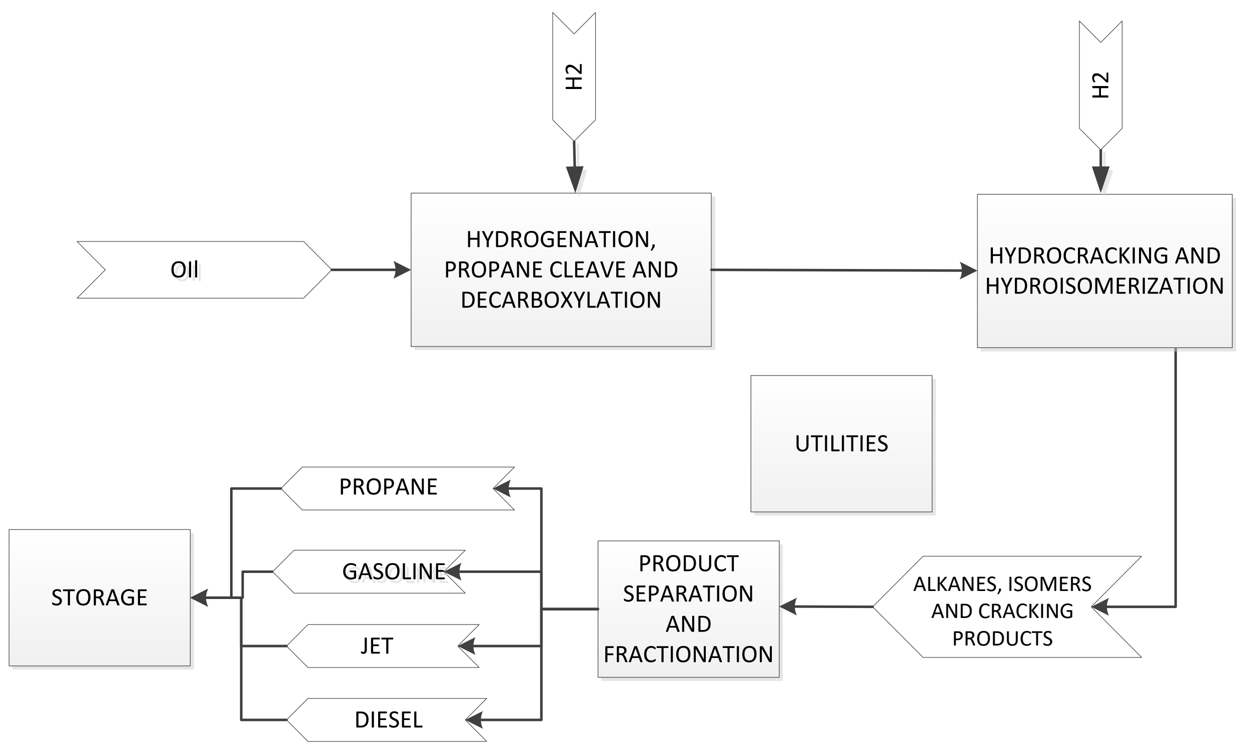

Figure 3.21. Block flow diagram of renewable diesel production process. Source: Tao et al. (2017b) (Creative Commons license, https://creativecommons.org/licenses/by/4.0/).

3.5 Biofuel Logistics

3.5.1 Distribution: From the Biorefinery to the Retail Station

The method of moving fuels throughout the country depends on the location of production, fuel type, and volume. The modes of transport for fuels include barge/ship, pipeline, rail, and truck (Figure 3.22). Petroleum products and ethanol move through the supply chain differently—where 70% of petroleum products are transported by pipeline and 70% of ethanol is transported by rail (Figure 3.23a, b). This is primarily due to the location of petroleum refineries along the Gulf Coast and ethanol's affinity for water, which could lead to corrosion in pipelines. Early research by DOE and others into the possibility of an ethanol pipeline concluded a dedicated pipeline would not be not economical and thus other modes of transport filled that need (DOE, 2010) and see Chapter 6 section 6.2.3).

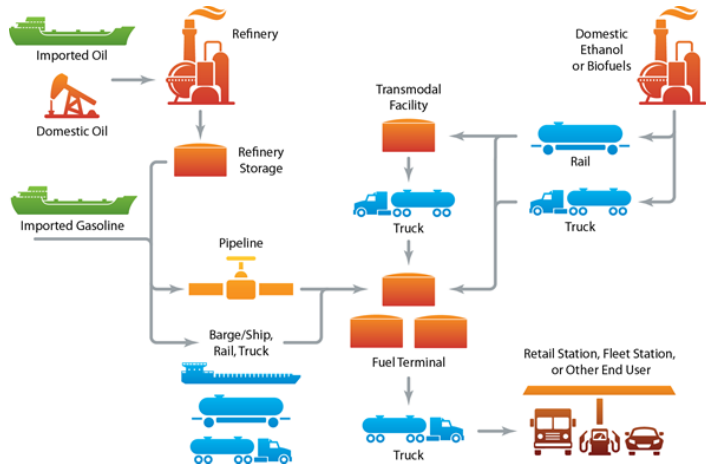

Figure 3.22. Liquid fuel delivery transportation modes. Source: Modified from Moriarty and Kvien (2021).

According to the National Biodiesel Board, biodiesel moves from production facilities to terminals and end users by truck (55%), rail (40%), barge (3%) and pipeline (2%) (Figure 3.23c). Emerging biofuels, such as renewable diesel, are typically moved by truck until volumes are accumulated

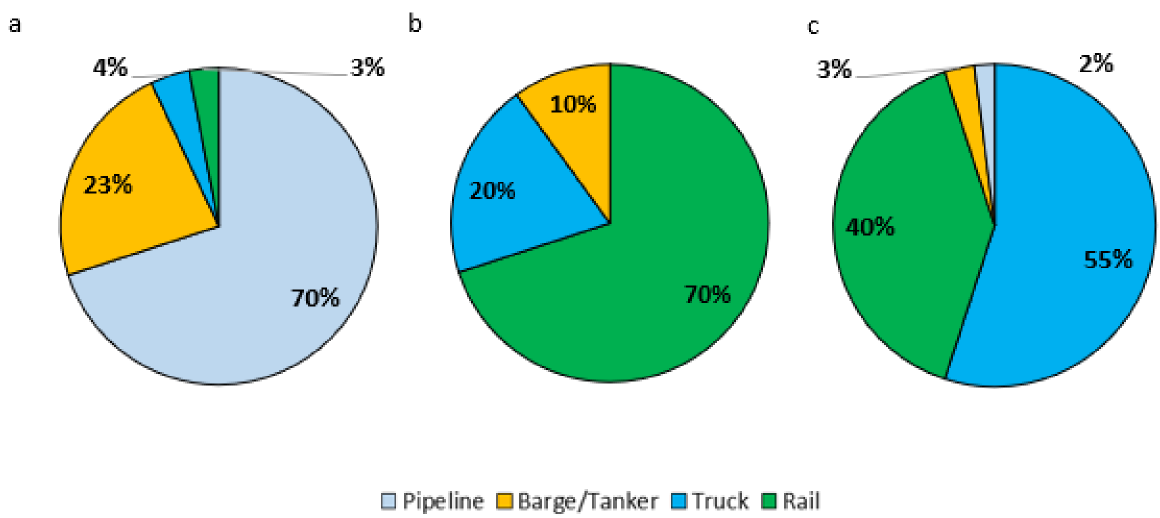

Figure 3.23. Logistics for crude oil and petroleum (a), ethanol (b), and biodiesel (c) volume shipments by mode. Data for (a) are from Conca for the year 2018, (b) are from (Bevil, 2011) for the year 2011, and (c) are from the National Biodiesel Board.

and then are moved by rail. Biofuels are sometimes delivered to a transmodal facility, which can receive large trains whereas few terminals have the infrastructure to accommodate large trains. Rates to ship fuels vary based on mode and distance with discounts for large-volume shippers. Pipelines offer the lowest costs for transporting fuels, followed by barge and rail. Trucking is cost effective for short distances from terminals to gas stations. The average cost between 2016 and 2018 to ship petroleum products via pipeline was 5.24 cents per gallon; petroleum products/ethanol via barge was 7.07 cents per gallon, and ethanol via rail was 21.95 cents per gallon.[34]

Generally, fuels are moved to fuel terminals and stored separately (Figure 3.22). Then, they are blended into a fuel delivery truck for delivery to a retail station. Early in the growth of the industry ethanol was "splash blended" with finished gasoline at the fuel terminal. Since then, the industry has shifted to "match blending" where the ethanol is blended with a suboctane blendstock for oxygenate blending (BOB, the primary petroleum-based component of gasoline, see Chapter 6 section 6.2.3 for more discussion of BOBs). Nearly all terminals currently store ethanol and are capable of blending ethanol and BOBs. BOBs cannot be sold as finished fuel without the addition of an oxygenate, which currently is 10% ethanol or more. Historically, stations received E10 and most E85 from a terminal. Some stations receive E85 directly from ethanol plants if they are nearby.

3.5.2 Dispensing: At the Retail Station

At the retail level, renewable fuels are nearly always sold in blends with petroleum fuels. Ethanol blends include E10, E15, E85, and less common intermediate blends. E10 is dispensed through conventional equipment intended for petroleum products and is available nationwide. Blends above E10 are dispensed through equipment specifically approved for its use (Figure 3.24a,b).

As E15 has entered the market in the last few years it was initially provided by a station storing E10 and E85 in separate underground storage tanks and using a blender pump to create E15. As the market for E15 has grown, terminals have started offering E15. It is estimated that more than 95% of stations cannot currently store E15 and would need to replace fueling equipment prior to doing so (U.S. EPA, 2020). Most of the existing E85 stations should have compatible equipment for E15, but stations not selling E15 or E85 likely do not have required equipment that is fully compatible with E15 because it costs more to install equipment compatible with E15, and most stations would not install it if they had not planned to sell E15.

While consumption of E15 in the United States has been limited, recent actions such as USDA's funding of infrastructure compatible with E15 at retail stations could result in some increased

[34] From the dataset, Argus Petroleum Transportation in North America. 2019. https://www.argusmedia.com/en/crude-oil/argus-petroleum-transportation-north-america?page=1. Purchased annually by NREL.

consumption of E15 in the future. As of April 2022, industry data show that 2,667 stations nationally were registered to sell E15.[35] These industry estimates are close to, but slightly higher than, EPA estimates.[36] However, the precise number is not critical given that 2,667 retail stations represent less than 2% of all retail stations in the United States.[37] The majority of these E15 stations were likely funded by the USDA Biofuel Infrastructure Partnership (BIP). By 2019, BIP had supported the upgrading of 1,486 stations to be able to sell E15 with $100M in matching grants from 2015 to 2019. This represents 99% of the E15 stations registered with the EPA in 2020. The successor to the BIP is the Higher Blends Infrastructure Incentive Program (HBIIP), which received the same amount of funding (i.e., $100M) in 2020. Given the same funding levels for the HBIIP as BIP, it is reasonable to estimate a continuation of trends observed under BIP for the numbers of E15 stations. This trend would roughly double the number of stations that sold E15 in 2020 by 2025, representing roughly 4,000 stations using industry estimates. This number of stations represents less than 3% of retail stations, suggesting that consumption of E15, though increasing, may not have a large effect on total ethanol consumption to 2025 unless even larger investments occur.[38]

Because gas stations are designed for long lifespans and underground storage tank systems are normally not frequently replaced, the total number of stations capable of using E15 is unlikely to change significantly without infrastructure funding programs, and thus terminal growth offerings are expected to remain limited (see Chapter 2 section 2.3.2 for more discussion of E15).

Storing and dispensing fuel containing more than 10% ethanol (more than E10) at gas stations with equipment that is not compatible with higher blends of ethanol fuel can result in leaks and releases that contaminate land and groundwater (see Chapter 10 section 10.3.1.8). Most existing underground storage tank systems (UST systems), which include but are not limited to the tanks, pumps, ancillary equipment, lines, gaskets, and sealants, are not fully compatible with these fuels and require modification before storing them. For example, although the tank is often compatible with E15, some of the connectors and pump components may not be. That can lead to leaks. Dispensers face the same compatibility concerns and are a critical part of the fueling system.

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[35] Estimates from Growth Energy, https://ethanolrfa.org/retailers/e15/.

[36] In the latest year where both EPA and industry estimates are available (2020), EPA estimated 1,445 E15 stations and industry estimated just over 2000. EPA estimates were from the RFG Survey Association collected under 40 CFR 80.68 and 69 as well as 40 CFR 1090.925 and industry estimates are from Growth Energy, https://ethanolrfa.org/retailers/e15/.

[37] According to the Transportation Energy Data Book: Edition 40 (February 2022), there were approximately 142,000 conventional refueling stations in the United States in 2020 (Table 4.24).

[38] Since drafting the report, the Inflation Reduction Act added an additional $500 million for infrastructure to support higher blends of ethanol (greater than 10%) and biodiesel (greater than 5%). Assuming similar rates of increase, this would add an additional 10,000 retail stations (14,000 total, or 10% of the total number of retail stations).

Common biodiesel blends include B5, which is allowable for use in conventional infrastructure and vehicles and engines, and B20, which may require some upgraded infrastructure equipment and approval for use in vehicles and engines. Biodiesel was available at roughly 5000 terminals as of June 2022 (Figure 3.24). The high concentration of B20 stations in North Carolina is primarily related to FOGs and the large number of rendering facilities in the state. Large diesel retailers such as truck stops have also invested in biodiesel blending infrastructure.

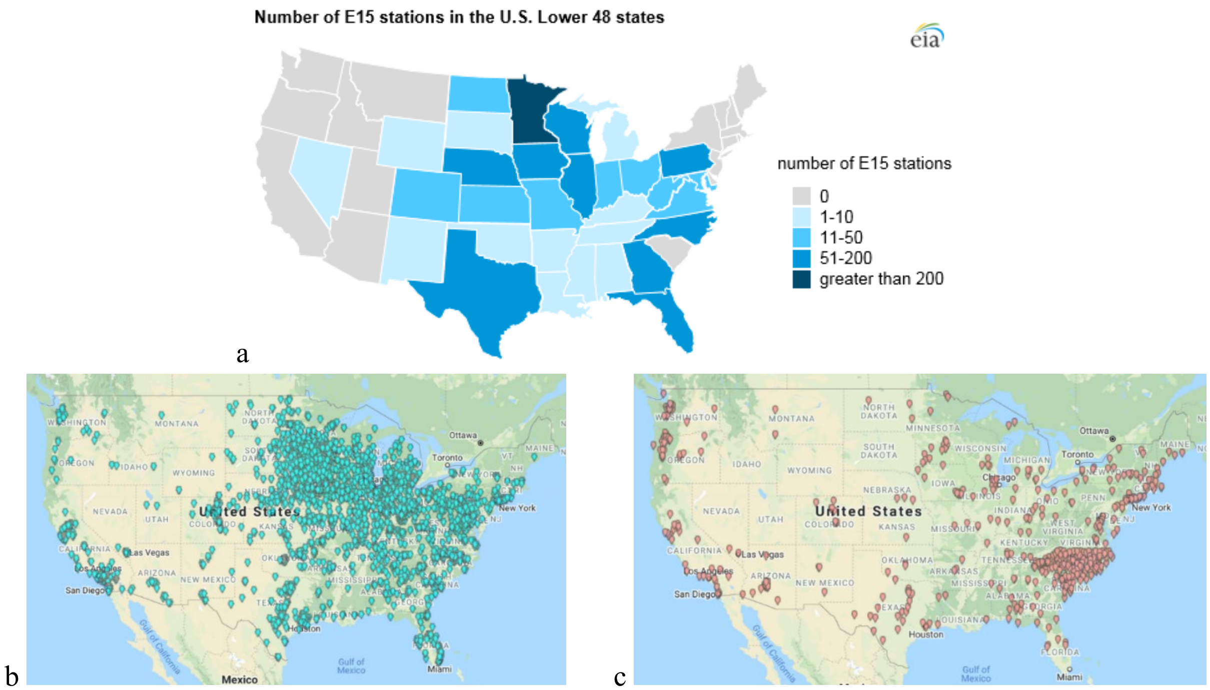

Figure 3.24. Stations offering E15 (a), E85 (b), and B20 (c). Sources: (a) EIA (2019), (b and c) NREL Biofuels Atlas (https://maps.nrel.gov/biofuels-atlas).

3.6 Biofuel End Use

3.6.1 Ethanol

The primary end use of ethanol is on-highway transportation fuel. Chapter 1 (Figures 1.3 and 1.4) highlights the growth in production and consumption of ethanol and other biofuels over time. Ethanol production has exceeded consumption in recent years leading to exports. Ethanol exports reached a high of 1.7 billion gallons in 2018 before declining to 1.25 billion gallons in 2021. In recent years the largest export markets for U.S. ethanol have been Brazil, Canada, and India (see Chapter 1 section 1.3.2 and Chapter 16 for more information).

Nearly all (98%) gasoline sold in the United States contains ethanol, and nearly all ethanol in gasoline is sold as E10 (RFA, 2017). In 2011, EPA approved E15 for use in model year (MY) 2001 and newer vehicles. At the end of 2017, 94% of the gasoline light-duty truck and vehicle population was MY

2001 and newer. However, manufacturers did not begin to warrant their vehicle for E15 use until much later, and several still do not.[39] E85 contains 51% to 83% ethanol, depending on geography and season and can be used in flex fuel vehicles (FFVs). The number of E85 stations has been increasing through time (Figure 3.25), but still represents a small fraction of the roughly 150,000 retail stations selling fuels. More than 21 million FFVs were registered nationwide as of the end of 2017, which is approximately 8% of the light-duty gas vehicle market (Figure 3.26). Thus, while E15 and E85 station numbers are growing, neither fuel is widely available compared with E10 (see Chapter 2 section 2.3.2).

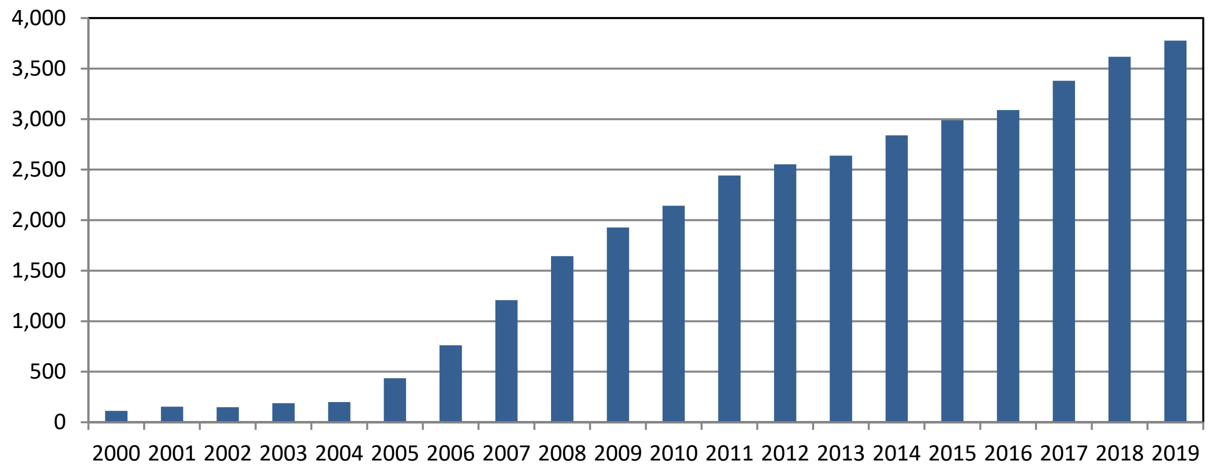

Figure 3.25. U.S. historical E85 stations. Source: (AFDC, 2022b).

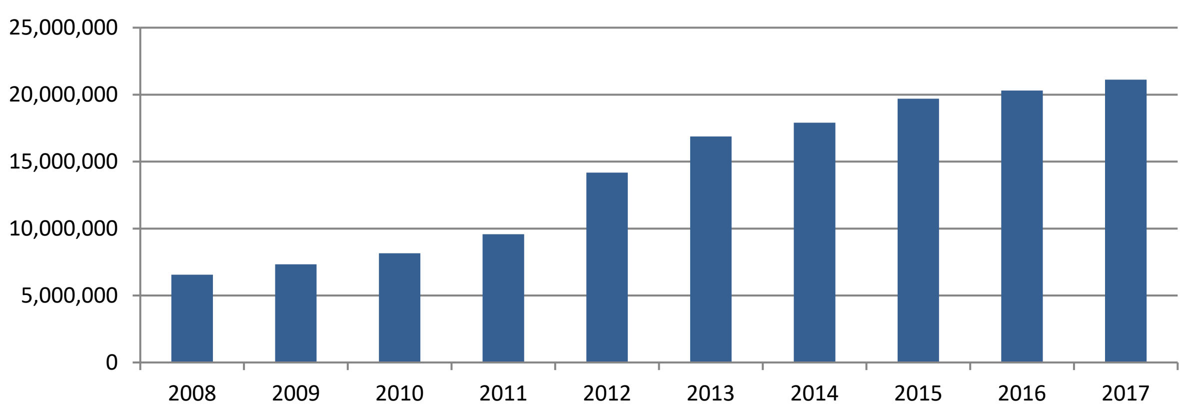

Figure 3.26. U.S. historical FFVs stock. Source: IHS Automotive.[40]

[39] Vehicle populations were determined using 2017 IHS Automotive (formerly Polk) vehicle registration data purchased by NREL.
[40] Proprietary data from IHS automotive purchased annually by NREL (https://ihsmarkit.com/products/automotive-market-data-analysis.html).

3.6.2 *Biodiesel*

Any blends of B5 (5% biodiesel, 95% petroleum diesel) or below that meet ASTM fuel quality specifications for conventional diesel fuel can be used in existing infrastructure and any compression-ignition engine intended for petroleum diesel. There are also ASTM specifications that describe the properties of B6 to B20 blends. B20 is the most common higher-level biodiesel blend, and engines operating on B20 have similar fuel consumption, horsepower, and torque to engines running on petroleum diesel (B0). Some, but not all, engine and diesel vehicle manufacturers warrant the use of B20.[41] Biodiesel is used predominately for on-highway transportation; however, there is also a growing market for use in home heating and non-road applications. Though small by comparison with ethanol, biodiesel production and consumption have expanded over the past decade, reaching a total production of over 1.8 billion gallons in 2020 (Chapter 1, Figures 1.3 and 1.4). Diesel use is predominately related to the trucking industry's consumption pattern and not personal vehicles. This is why many retail stations offering diesel are located along major trucking routes. This is also the reason stations selling B20 are located primarily in urban centers and along major highways. Those outside of these locations are typically private stations serving the fleets of the U.S. Department of Defense, other federal agencies, and local governments. Of the 610 refueling stations offering B20 in 2019, roughly 190 are open to the public (Figure 3.27 and Figure 3.24c).

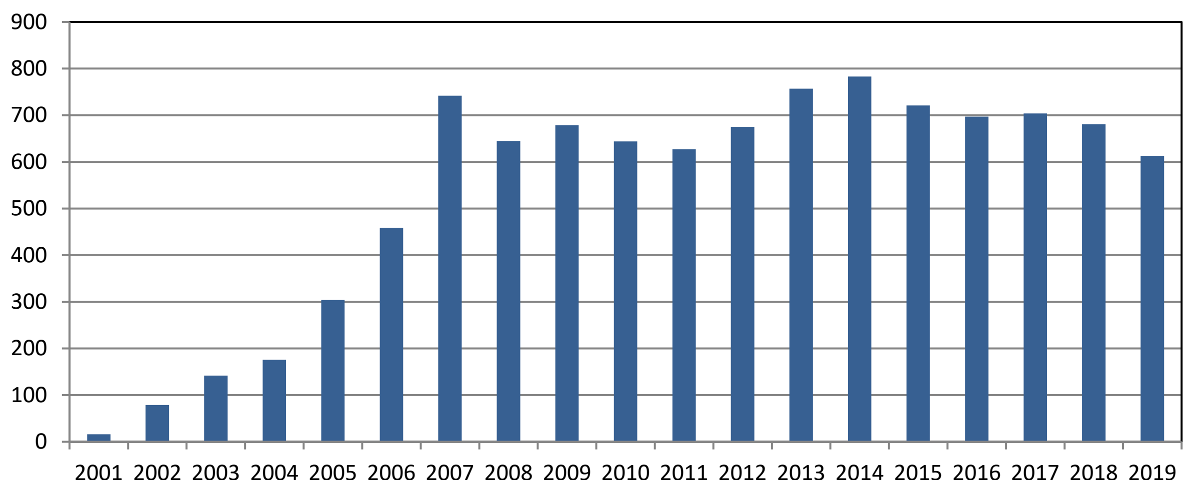

Figure 3.27. U.S. historical biodiesel (B20) refueling stations. Source: (AFDC 2022b).

3.6.3 *Renewable Diesel*

Domestic production of renewable diesel has increased each year from 2012 through 2019, with larger annual increases in recent years (see Chapter 1, Figures 1.3 and 1.4). Relative to biodiesel, renewable diesel is cheaper to transport and faces fewer concerns related to compatibility with diesel

[41] For a full list of auto manufacturers who warrant B20, visit https://www.biodiesel.org/docs/default-source/fact-sheets/oem-support-summary.pdf?sfvrsn=4e0b4862_10 (NBB, 2020).

engines, especially at higher blend levels (discussed above in sections 3.4.2 and 3.4.3). These advantages along with others (discussed in Chapter 7) have contributed to the significant increases in renewable diesel production and use in the United States in recent years at the same time that domestic biodiesel consumption increased more slowly. Further, a number of new renewable diesel projects have been announced in trade magazines (Bryan, 2021). These facilities include both new production facilities and conversions of petroleum refineries to process renewable feedstocks.

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4. Biofuels and Agricultural Markets

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Key Findings

- Renewable Identification Number (RIN) prices for renewable (D6) fuels provide evidence that the Renewable Fuel Standard (RFS) Program increased U.S. consumption of renewable biofuels in 2009 (and late 2008) and from 2013 to 2019.
- Advanced (D5), biomass-based diesel (D4), and cellulosic (D3) RIN prices provide evidence that the RFS2 increased U.S. consumption of advanced, biomass-based diesel and cellulosic biofuels in every year of RFS2 for which standards had been set for these fuels (i.e., starting in 2010).
- Studies estimated that the RFS Program could increase corn ethanol production between 0 and 5 billion gallons under scenarios with relatively high oil prices (greater than $60 per barrel in 2018 prices). Oil prices were greater than $60 per barrel for much of the period of growth in the corn ethanol industry.
- A meta-analysis of studies published between 2007 and 2014 on the impact of biofuels estimated that for every billion-gallon increase in corn ethanol production between 2010 and 2019, corn prices would increase about 3–5%.
- Studies of the impact of RFS2 estimated that the Program could increase biomass-based diesel consumption 0.6–1.1 gallon for every gallon in the biomass-based diesel volume obligations. This is equivalent to an increase in biomass-based diesel consumption of 0.4–0.7 gallons for every ethanol equivalent gallon in the advanced volume obligations.
- Studies of the impact of biofuels estimated that for every billion-gallon increase in biomass-based diesel production, soybean prices increased 1.8–8.9%.
- RFS2 was estimated to have a limited impact on soybean meal production (decrease of 1.2% per billion gallons of biodiesel) and put downward pressure on soybean meal prices (decrease of 1–4.1% per billion gallons of biodiesel).
- On average, production decreases in beef, milk, pork, and poultry were estimated to be less than 0.5% per billion gallons of corn ethanol. Producer price increases in these livestock commodities were estimated to be less than 1 cent per pound per billion gallons of corn ethanol. The impact on consumer prices would likely be less than this.
- On average, an estimated 1 million acres of additional corn would be produced, and cropland would expand an estimated 0.7 million acres for each billion-gallon increase in corn ethanol production.

Chapter Terms: D3 RIN, D4 RIN, D5 RIN, D6 RIN, EPA Moderated Transaction System (EMTS), fats, oils, and grease (FOGs), general equilibrium (GE) models, partial equilibrium (PE) models, Renewable Identification Number (RIN), Renewable Volume Obligation (RVO)

4.1 Introduction

The Renewable Fuels Standard (RFS) is a tradable credit program that uses market signals (i.e., prices) to create incentives to cost-effectively blend a mandated quantity of biofuels into the transportation fuel supply. Increasing the use of biofuels has economic impacts throughout the agricultural and fuel sectors, affecting prices and quantities of many commodities. Figure 4.1 illustrates the flow of goods among the markets closely linked to biofuels and examined in this chapter. Impacts in these markets can involve changes in the use of land and other resources, with eventual effects on environmental quality. Therefore, a discussion of economics of these markets is important to understand the environmental impacts of the RFS Program.

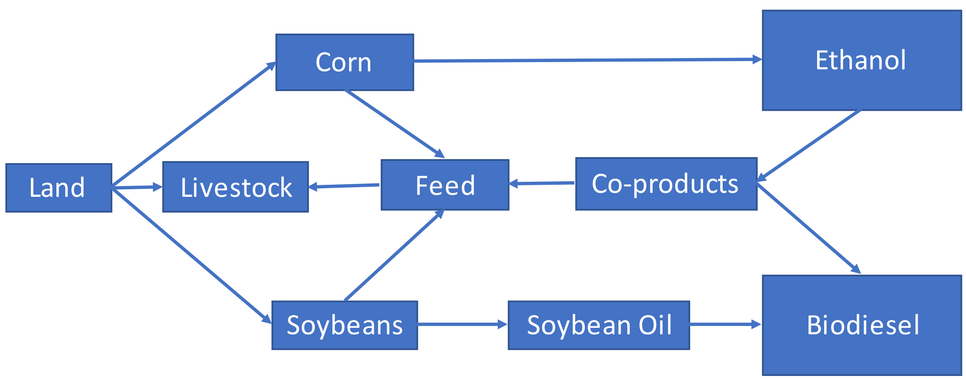

Figure 4.1. Conceptual diagram of the flow of goods in the biofuel and agricultural markets examined in this chapter.

This chapter explores the economic effects of ethanol and biodiesel in agricultural markets, in particular their impact on production and prices of corn and soybeans (Figure 4.2), the dominant feedstocks to date. The large body of literature and the associated syntheses and meta-analyses that exist for corn serve as the foundation for the assessment of corn and corn ethanol (section 4.3). Much less literature is available for soybean, and nearly no syntheses or meta-analyses have been conducted; therefore, this chapter reviews individual studies for the assessment of soybean and soybean biodiesel. The goal is not to provide a comprehensive discussion of all economic effects but only economic aspects and market links that result in environmental and resource impacts. Economic impacts on food prices and welfare effects on food producers and consumers are not mentioned in Section 204 and thus are outside of the scope of this report. Assessment of the economics of the other two biofuels that dominated the U.S. pool are discussed in Chapter 16 for Brazilian sugarcane (i.e., International Effects) and Box 4.1 for fats, oils, and greases (FOGs).

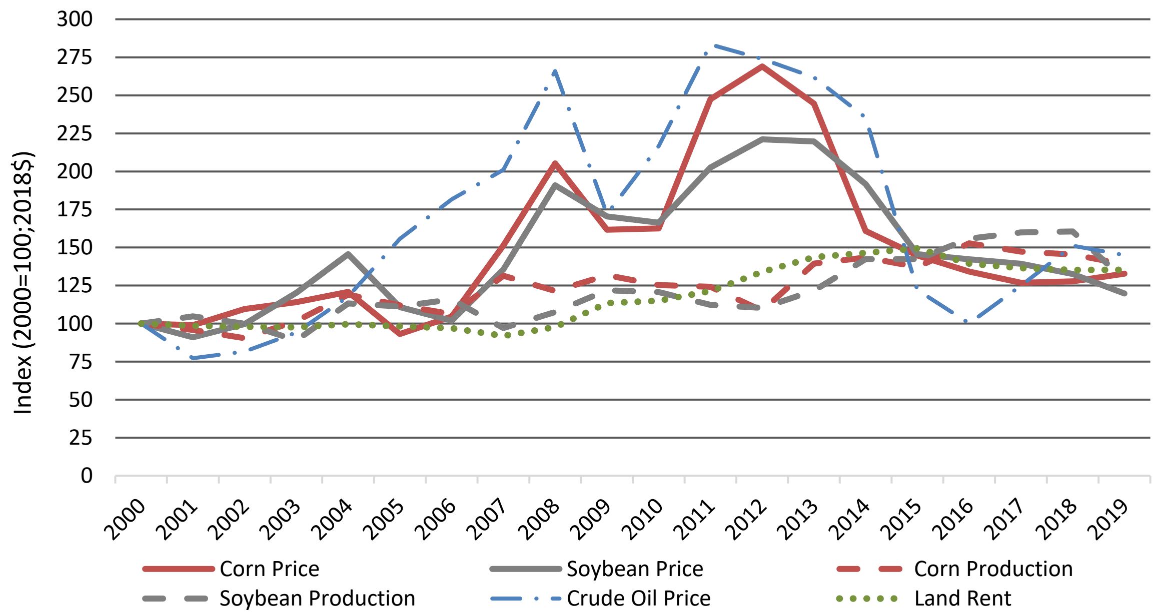

Figure 4.2. U.S. corn, soybean, crude oil, and land price and corn and soybean production indices (year 2000=100; 2018$).[1]

4.2 Renewable Identification Number (RIN) Markets

The RFS Program relies on a market-based approach to compliance. Obligated parties—typically petroleum refiners or importers of gasoline and diesel—submit credits to the EPA to demonstrate they have met their obligations under the RFS Program. These credits are designated in terms of RINs. Each gallon of qualifying biofuel that is produced domestically or imported is assigned a RIN, which is separated from that gallon when it is blended with motor fuel.[2] Once separated, a RIN can be banked or bought and sold independently until it is retired to meet an obligated party's renewable volume obligation.[3] The four standards are nested within each other: cellulosic biofuel and biomass-based diesel RINs also satisfy the advanced biofuel standards, and all types of RINs, including those for conventional biofuel (e.g., corn ethanol) can be used to meet the total renewable fuel standard.[4] Beginning in July 2010, EPA introduced an electronic system to manage all RIN transactions,[5] which allowed for increased transparency regarding RIN trading activity and prices.[6]

[1] Sources: USDA (2019c); inflation adjusted based on Consumer Price Index (U.S. BLS, 2019); monthly imported crude oil prices from the Energy Information Administration (EIA) Short-Term Energy Outlook (EIA, 2022).
[2] The lifecycle of a RIN is summarized at https://www.epa.gov/renewable-fuel-standard-program/renewable-identification-numbers-rins-under-renewable-fuel-standard.
[3] Exporters also need to retire RINs within a certain amount of time after the renewable fuel has been exported to demonstrate compliance.
[4] See Chapter 1, section 1.2 for more information on the nested design of the standards.
[5] This system is called the EPA Moderated Transaction System (EMTS).
[6] Thus, from the beginning of the RFS Program in 2006 to 2010, EPA did not digitally record RIN prices or transactions, although a few private companies began recording data on daily spot market transactions in mid-2008.

RIN prices reflect the cost of producing renewable fuels, the demand for renewable fuels, and the blending of renewable fuels into the fuel supply. Demand for RFS RINs exists solely to meet RFS Program requirements (see Chapters 2 and 6). Thus, lower RIN prices reflect a relatively low cost of compliance (e.g., because biofuels can compete with petroleum or there are ample RINs available for purchase). In this case, the RFS Program is forcing little additional biofuel consumption beyond what is driven by other market or policy forces (e.g., demand for fuel oxygenates). A RIN price near zero indicates that the RFS Program has no effect on biofuel consumption (i.e., it is not "binding" in economic parlance): the cost of compliance is near zero (other than administrative costs). Higher RIN prices indicate a higher cost of compliance to obligated parties. The higher price encourages them to find cheaper ways to meet the standard (Pouliot and Babcock, 2013; McPhail and Babcock, 2012). If other policies simultaneously create incentives for biofuel production, such as the volumetric ethanol excise tax credit (VEETC) that ended in 2011, the Biodiesel Tax Credit (BTC), and the various state low-carbon fuel standards and incentive programs, then RIN prices will be lower and the RFS Program is less likely to cause increased biofuel consumption (Babcock, 2012).

The nested nature of the RFS Program standards (see Chapter 1, Figure 1.2) has implications for RIN prices, with prices of advanced RINs at or above the price of renewable RINs (Whistance and Thompson, 2014). In addition, the wider the gap between the prices of advanced renewable and total renewable RINs, the more binding the advanced component relative to the total component (Paulson and Meyer, 2012). RINs are also tradable. RINs become separated from the biofuel when they are blended with gasoline or diesel. These RINs can then be purchased by obligated parties.[7] Given that RINs can also be banked or borrowed, they provide obligated parties with a way to anticipate and buffer anticipated future costs through arbitrage (Zhou and Babcock, 2017; Whistance et al., 2016). RIN prices respond to these expectations, expressed through changes in demand for RINs. For example, there were more than 1 billion RINs carried over each year from 2011 through 2019, constituting roughly 10–25% of the annual obligation (termed a Renewable Volume Obligation, RVO, Figure 4.3).

[7] While most RINs are privately transacted, they must be registered in the EPA system. This, along with prosecution of fraudulent RINs under its Clean Air Act authority, ensures that RINs are valid and the system as a whole is a reliable way to measure compliance with the RFS (Yacobucci, 2013).

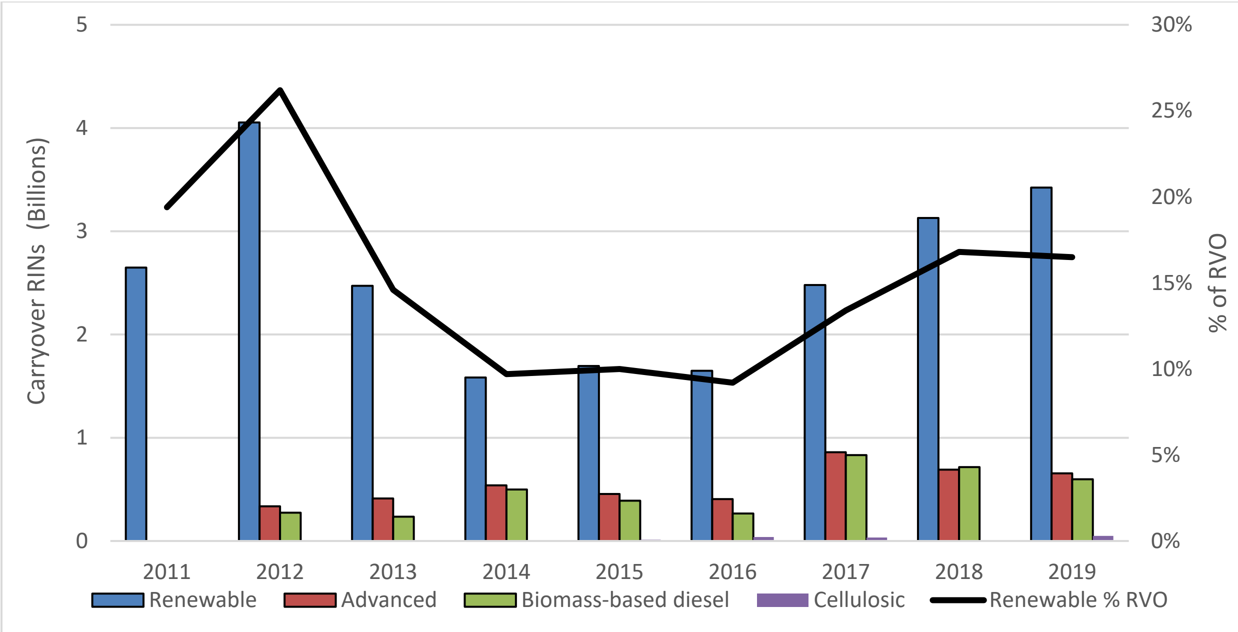

Figure 4.3. RIN banking. Shown are the carryover RINs from 2011 through 2019 (left axis and bars) and the percent carryover relative to the total annual volume obligation (i.e., Renewable Volumetric Obligation, RVO, right axis and line). Since RINs were not electronically tracked until 2010, the first year of carryover is 2011. Carryover RINs for advanced and biomass-based diesel are not reported for 2011 due to transition from RFS1 to RFS2. Cellulosic RINs were first generated in 2012 and so 2013 was the first year of carryover.[8]

RIN prices fluctuate over time both within and across years (see Figure 4.4). For instance, in the year 2013 alone, weekly RIN prices for renewable fuels (D6), such as those derived from conventional sources (e.g., corn), ranged from 5 cents at the beginning of January to a high of $1.05 by August. Weekly RIN prices for cellulosic ethanol (D3) also demonstrate substantial variation over a longer time frame.[9] D6 RIN prices were relatively low until 2013, aside from a small increase in late 2008 and 2009 (see Figure 4.4 and Figure 4.5).

Whistance and Thompson (2014) examined RIN price behavior between January 2009 and May 2013 and found that 94% of the time they responded in ways consistent with expectations (based on whether a mandate was binding, the hierarchical nesting structure of the different RFS Program standards, and RIN vintage).[10] Focusing only on biomass-based diesel, Irwin et al. (2020) found that RIN prices also largely adhered to expectations.

[8] Data from the EMTS.
[9] See https://www.epa.gov/fuels-registration-reporting-and-compliance-help/rin-trades-and-price-information for RIN transaction data.
[10] Whistance and Thompson (2014) relied on RIN prices from the Oil Price Information Service (OPIS), which are available beginning in 2008. Recall, EPA began electronically tracking RIN transactions in July 2010. The hierarchical nesting structure of the RFS standards implies that the price for broader RINs acts as a price floor for narrower RINs (since excess RINs from narrower volume standards can be used to satisfy broader volume standards). RIN vintage implies that older RINs act as a price floor for newer vintage RINs (since older RINS have a limited potential for use relative to newer RINs).

Another factor that affects RIN prices is the E10 blend wall. Above 10% ethanol, the economics and logistics of blending ethanol change dramatically (U.S. EPA, 2022). As consumers purchase less gasoline, for instance due to high oil prices or increases in fuel efficiency (U.S. EPA, 2022), the point at which the E10 blend wall becomes binding also decreases, (i.e., since there is less gasoline into which ethanol volumes can be blended), leading to higher D6 RIN prices at lower overall ethanol levels. Working papers by Burkholder (2015), de Gorter and Drabik (2015), and Meiselman (2016) explore how the blend wall affected RIN markets in 2013. They point to the potential expansion of E85 (85% ethanol) for flex-fuel vehicles, but more importantly in practice the use of biodiesel RINs—beyond what is needed to meet the biomass-based diesel mandate, but nested within the advanced mandate—to meet RFS Program requirements in the face of the E10 blend wall constraints.

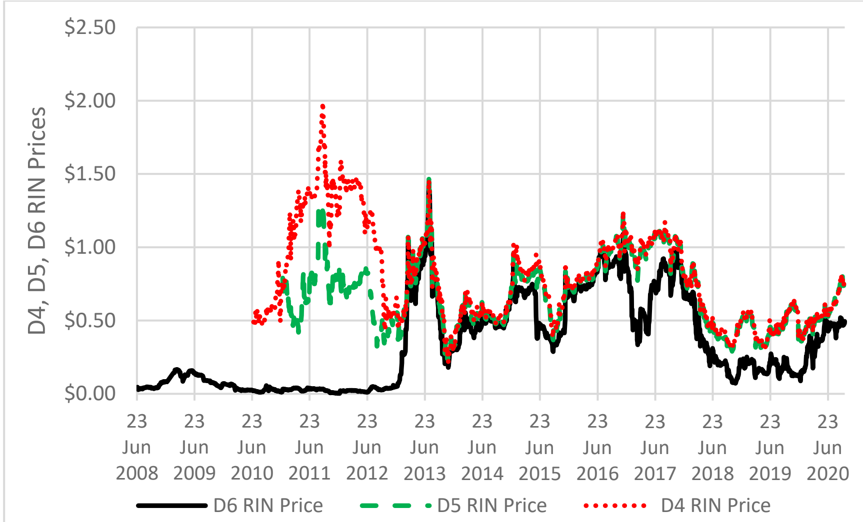

Figure 4.4. Daily RIN prices (June 23, 2008–2020). Biomass-based diesel (D4). Advanced (D5), and Renewable (D6) RIN prices. Source: ARGUS (2022).

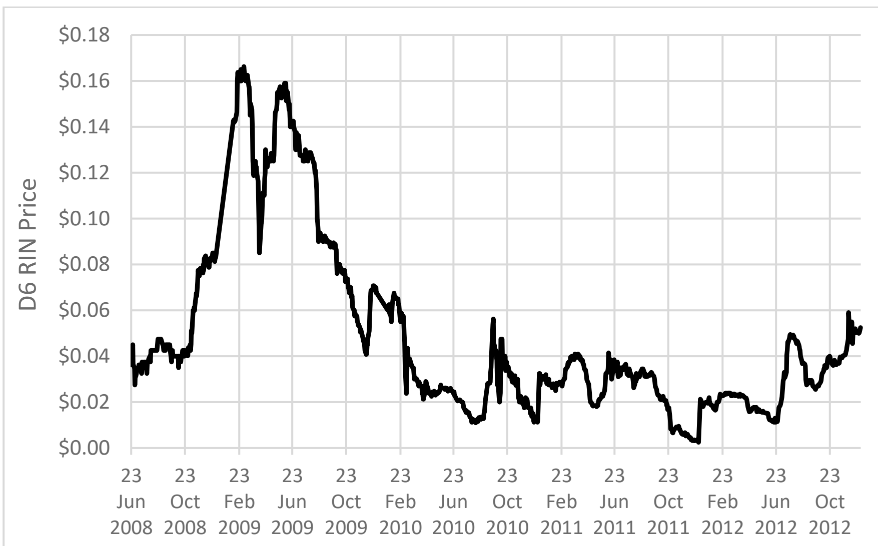

Figure 4.5. Renewable (D6) RIN prices (June 23, 2008–December 23, 2012). Note the difference in y-axis compared with Figure 4.4. Source: ARGUS (2022).

4.3 Corn Markets

In the years since the passage of the Energy Independence and Security Act (EISA) in 2007, the field of economics has generated a multitude of studies that evaluate the impacts of biofuel expansion.

Most studies have examined the impacts of expanded corn ethanol use on corn markets in response to U.S. or international biofuel policies and drivers. A more limited set of studies has considered other biofuels, such as soy biodiesel and sugarcane ethanol, as well as impacts on livestock or other commodity markets. These studies are largely prospective—predicting impacts of biofuel expansion in future years before they have occurred—and so are based on simulations rather than a retrospective analysis of observed data. This literature is useful for understanding the expected impacts of increases in biofuel production and consumption, regardless of the driver, on agricultural commodity prices and outputs. However, it should not be confused with an assessment of what actually occurred, which is the focus of Chapter 6.

This literature typically relies on simulation models (i.e., models informed by economic theory) that provide either a simplified representation of the global economy as a whole (general equilibrium models, GE) or a more detailed representation of the agriculture and/or fuel sectors (partial equilibrium single or multimarket models, PE). These types of models include a series of equations that specify how responsive the quantity of each good is to different prices. Other key factors that might affect supply or demand, such as technological change, input costs, or preferences for related goods, are also specified, sometimes by exogenously imposing a given assumption; sometimes by explicitly modeling these relationships. In the biofuels market, production and price are determined by many factors, such as weather, oil prices, and the demand for food and competing uses of biofuel feedstocks.

The supply and demand equations are used to determine equilibrium prices and quantities within the model (endogenously) for certain goods. Prices and quantities of other goods can also be specified outside the model (exogenously). For example, oil prices may be specified exogenously based on Energy Information Agency (EIA) forecasts or determined endogenously by modeling supply and demand in the fuel market. Regardless, researchers use a data-driven approach to assess current and future market conditions, as well as the responses of markets to changes in these conditions. One advantage of these types of simulation models is that they can be used to prospectively evaluate factors and policies under conditions not well represented in the historical data, such as the potential effects of introducing a new policy that might impact renewable fuel prices and quantities and corn markets under different oil price scenarios.

4.3.1 Overview of Corn Markets

This discussion highlights direct demand-side and supply-side factors that impact corn production and prices, though indirect impacts can affect the market as well. Direct demand-side factors are summarized by the share of the market used for different purposes, which are referred to as utilization. As discussed in Chapter 3, the two largest uses of the corn in recent years have been for ethanol and animal feed, but there have been changes in utilization over time. Between the 1999/00 and 2017/18 marketing years,[11] the utilization of corn for ethanol grew from just 6% to 38% of corn production (Figure 4.6). Conversely, the market share of animal feed has decreased. From the 1999/00 to 2018/19 marketing years, animal feed's share of the market fell from 60% to 38%. However, some corn that goes to ethanol production still contributes to animal feed markets in the form of distillers' grains (DGs), a byproduct of corn ethanol production (see section 4.5 for more details).

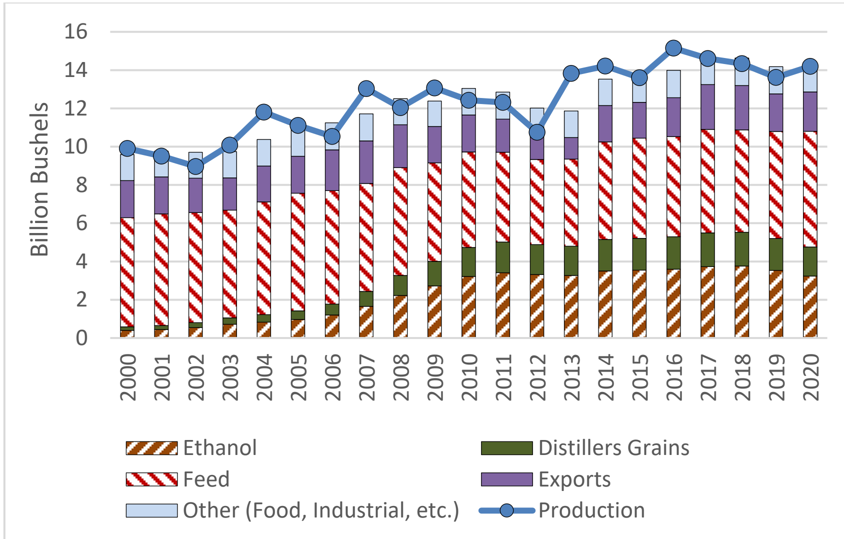

Figure 4.6. Corn production and use.[12]

Direct supply-side factors in corn markets can be weather related. Cold temperatures, excessive rainfall and soil moisture, flooding, and dry and hot conditions can all reduce yields or cause crop failure.

There are also non-weather-related supply-side factors. These factors vary in importance according to their share of production cost. Land and machinery are two of the highest costs, each of which are about a quarter of the cost of production (Table 4.1). Fuel, fertilizers, pesticides, and other chemicals, which are heavily dependent on energy prices, make up about 26% of the cost of production. Therefore, increases in oil prices are transferred to increased production costs.[13] The final major share of costs is seed, at about 14%.

[11] Marketing year is generally the 12-month period following harvest during which a commodity may be sold domestically, exported, or put into reserve stocks. The year varies by country and commodity, and often includes months from two calendar years. Corn and soy marketing years are the same, from September 1 to August 31.

[12] Source: Use from USDA (2022b) and production from NASS. Note this is the same use data (based on market year) presented in Chapter 3 (Figure 3.15), but it has been aligned with calendar year here to better align with the production data (based on calendar year) assuming that two-thirds of the use occurred in the dominant year (i.e., for calendar year 2013, two-thirds of use is from MY2012/2013, one-third from 2013/2014).

[13] Oil prices can also have secondary links with commodity crop prices. For example, changes in energy markets impact incomes of oil-producing countries and, in turn, purchasing power for U.S. commodity crops (Hart and Zhang, 2016). Oil prices can also influence ethanol prices (and vice versa) because of their potential as substitutes in the fuel market, which in turn can impact agricultural commodity prices (Zafeiriou et al., 2018; Papiez, 2014). Chiou-Wei et al. (2019) found that oil prices and agricultural commodity prices tended to move together during 2005–2017. Corn prices responded positively to lagged increases in ethanol and natural gas prices but negatively to lagged increases in oil and soybean prices.

Table 4.1. Share of cost of production for corn and soybeans in 2019.

Crop	Land	Machinery	Fuel, fertilizers, pesticides, and other chemicals	Seed	Other
Corn	23%	24%	26%	14%	13%
Soybeans	33%	26%	14%	13%	14%

Source: USDA (2020c)

Production of corn was trending upwards between the 1999/00 and 2015/16 marketing years (Figure 4.6). The more than 50% increase in production during this period was driven by increases in yields and acres harvested (Figure 4.7). However, in some years, such as 2002 and 2012, drought decreased yields and production.

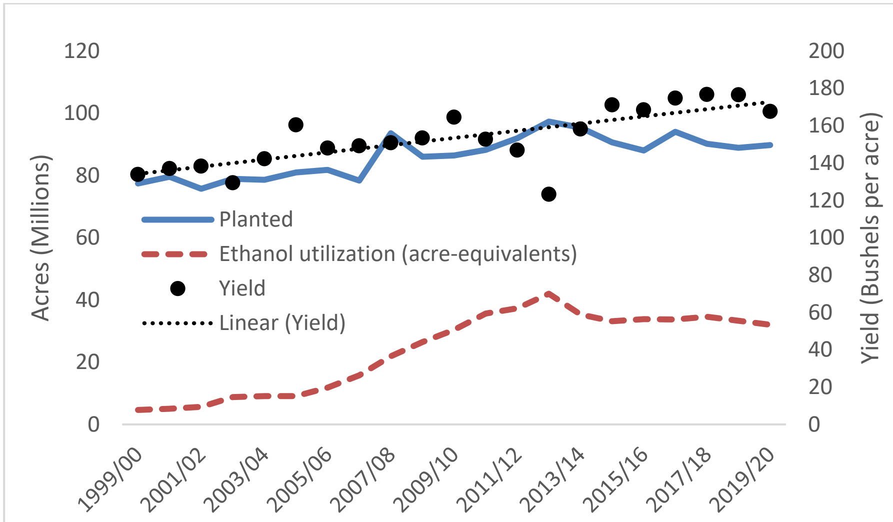

Figure 4.7. U.S. corn acreage and yields from USDA NASS. Biofuel utilization is calculated by dividing the quantity utilized for biofuels by the average corn yields in that year.

After harvest, corn is stored and then these stores are drawn down throughout the year. In recent years, about 15% of production is still in storage when the corn crop is harvested USDA (2020a). If demand goes up after harvest, corn stocks will be drawn down faster. If demand decreases, stocks are drawn down slower. In 2012, a drought year, corn stocks fell to the lowest levels since 2000. These dwindling stocks put upward pressure on prices. Storage can also be used across years to dampen supply shocks. Zhou and Babcock (2017) constructed a model of the U.S. corn, ethanol, and gasoline markets to illustrate how the ability to store corn over time reduces corn price volatility associated with supply and demand shocks. For instance, they found that a weather-induced shock to corn yields had a relatively modest impact on corn prices, as the ability to store corn absorbed much of the impact.

4.3.2 Corn Price Impacts from Corn Ethanol Policies

It is common practice in the economics literature to use simulation models to examine the expected impacts of changes in ethanol and biofuel use that might result from various policy and market drivers—including the RFS Program—on future corn prices.[14] A comprehensive review of this literature is outside the scope of the present report. Where possible, existing reviews that synthesize the many available discussion papers, reports, and peer-reviewed articles are relied upon. Because many of these studies provide insights about how increasing corn ethanol production is likely to affect corn prices, even when they do not specifically focus on the RFS Program, a broader range of studies that examine the effects of changes in ethanol volumes on corn prices is considered.

Early reviews of the literature by the NRC (2011) and Zhang et al. (2013) highlighted the large disparity in estimated impacts across studies, but neither formally evaluated the role different factors played in driving those differences. The NRC (2011) discussion of the 2007–2009 food price spike included estimates from eleven studies ranging from 17 to 70% for the proportion of price increases during this period attributable to increased biofuel production generally (not specific to the RFS Program). The Zhang et al. (2013) review found projections for the effect of biofuel policies on corn prices in 2015 varied from 5% to 53% across nine studies. Both reviews acknowledged that the wide variation in price impacts across studies resulted in part from the different domestic and international biofuel policies and scenarios analyzed. The Zhang et al. (2013) review also discussed the importance of differences in modeling framework (GE versus PE) and assumptions about biofuel trade, land supply, crude oil prices, and ethanol co-products (such as dried distillers' grains [DDGs] used as animal feed) across studies. Both reviews noted that differences in modeling choices and scenarios across studies made it very difficult to compare results or glean definitive conclusions regarding the relative contribution of biofuel policies to food price increases from them.

More recent reviews of the literature have systematically assessed what has driven the wide variation in results across a larger number of studies. Persson (2016)'s analysis of over 100 studies emphasized the importance of supply and demand elasticities. Condon et al. (2015)'s meta-analysis of over 150 estimates from 29 published papers found that studies using GE models, projecting results several years into the future, and accounting for the use of ethanol co-products in livestock markets yielded relatively smaller estimates of corn price impacts. They also found that studies using PE models, those including other biofuels in addition to corn ethanol, and those assuming higher oil prices estimated larger corn price impacts from biofuel policies.[15] Hochman and Zilberman (2018)'s meta-analysis of 273 estimates drawn from 41 studies showed that analyses incorporating energy market linkages and

[14] Other policy instruments considered in this literature include tax credits, low-carbon fuel standards, carbon taxes, and varying levels of mandates, both in the United States and other countries.

[15] Condon et al. (2015) employed both random and fixed effects models where standard errors were clustered by study. Condon et al. (2015) found that factors in their meta-regression models explained 65% to 89% of the spread across estimates. In addition to the factors discussed above, they examined the role of the type of scenario, policy instrument, corn yield, and whether corn was included in a more aggregate commodity bundle or examined separately. The results regarding the role of these variables were either inconclusive or relatively small.

modeling mandates (instead of other policy instruments) yielded smaller results, while studies with higher demand elasticities and calibrated to a later year (thus implying larger mandated ethanol volumes) generated larger impact estimates.[16]

Condon et al. (2015) and Hochman and Zilberman (2018) reported the average impact of corn ethanol expansion on corn prices across studies included in their analyses. Condon et al. (2015) found a 17.8% increase in corn prices across studies, and Hochman and Zilberman (2018) found a 13.0 to 13.7% increase in corn prices on average across studies.[17] However, these averages are difficult to interpret given that the underlying studies differed considerably in the increase in corn ethanol production projected to occur in response to the policies examined. In addition, the underlying studies varied along several other dimensions noted above, including time period of analysis and modeling approach.

To allow for more direct comparisons across studies, both Persson (2016) and Condon et al. (2015) normalized corn price impacts per unit increase in corn ethanol production. Both found that, taking the average across studies included in their respective reviews, the impact of an additional billion gallons of corn ethanol production was a 3 to 5% increase in corn prices.[18,19]

The meta-regression results from Condon et al. (2015) can be used to estimate the corn price impact associated with a marginal change in corn ethanol production. For example, if RFS2 resulted in an additional 1 billion gallons of corn ethanol production annually from 2010 through 2019,[20] then results from the Condon et al. (2015) meta-analysis imply that corn prices would have been 3–5% higher on average, given annual oil prices and corn yields during this time period.[21] With corn prices averaging

[16] Hochman and Zilberman (2018) used both ordinary least squares and frequency weighted regression approaches to explore the effects of corn ethanol on food prices. Other factors included in these regressions were the assumed supply elasticity, the period of analysis, and a fuel market dummy. None of these variables was significant.
[17] The 13.0% result represents a simple average, and the 13.7% result is the authors' preferred meta-analytic average.
[18] Persson (2016) normalized by exajoule (EJ) of energy, finding an average corn price increase of 32 to 36% per EJ depending on the weighting approach, which converts to 2.81 to 3.16% per billion gallons.
[19] An unpublished working paper by Thompson et al. (2016) reviewing 66 published and unpublished studies found results similar to Persson (2016) and Condon et al. (2015) for the median change in corn prices across studies per billion gallon increase in corn ethanol when focusing on observations that focused exclusively on changes in corn ethanol volumes (with no other biofuel quantity changes). Thompson et al. (2016) reported a median corn price change per billion gallons corn start ethanol of $0.19 per bushel (reported in Table 4, first row). This represents a 4% increase in corn prices relative to the average U.S. corn prices of $4.73 during 2010–2019. Unlike the other reviews, Thompson et al. (2016) also emphasized that short-run studies—which do not allow for a supply response—of the effect of biofuel policies during drought periods yield larger estimated impacts than studies focused on medium- to long-term estimates. Excluding short-run observations yielded a study-weighted average of $0.18 per bushel and a study-weighted median corn price increase of $0.15 per bushel per billion-gallon increase in corn ethanol.
[20] See Chapter 6 for the actual estimates, this is a hypothetical illustration.
[21] The estimates from Condon et al. (2015)'s random effects price change per billion-gallon increase model (Table 4, column 3) were used to generate this result because the random effects model allows for projections out of sample. The low end (3%) of the range assumes a general equilibrium modeling framework, and the high end (5%) assumes a partial equilibrium modeling framework.

$4.73 per bushel (USDA, 2020b), converted to 2018$ using CPI during this period, this percentage impact represents an average increase in corn prices from what they would have been without corn ethanol of $0.14 to $0.24 per bushel per billion gallons of corn ethanol (2018 dollars).

Condon et al. (2015)'s results can also be extrapolated to estimate the effects of total U.S. corn ethanol production that occurred during the time period of RFS2—not just the portion of ethanol production *attributable* to the RFS Program—on corn prices. Their results suggest that U.S. corn ethanol production during 2010–2019, which increased from about 13 billion to 16 billion gallons and averaged 14.6 billion gallons annually, increased corn prices by 32 to 53%.[22]

4.3.3 Corn Production Impacts from Corn Ethanol Policies

The amount of ethanol produced per bushel of corn has increased over time as production processes have become more efficient. In 2019, each bushel yielded about 2.8 gallons of corn ethanol, meaning that about 360 million bushels of corn were needed to produce 1 billion gallons of corn ethanol (EIA, 2019). However, total corn production need not increase by this full amount in response to each billion-gallon increase in corn ethanol production. Increased corn prices spurred by higher demand for corn ethanol can have a combination of two effects: (1) corn production can increase, and (2) substitution can occur from other feedstock uses (e.g., corn previously used for animal feed can be diverted to ethanol). In addition, DGs are co-produced with ethanol, so each bushel of corn used for ethanol production yields some livestock feed (section 4.5.1 provides more discussion of DGs.) The only review of the literature to the authors' knowledge that specifically examined how corn production responds to ethanol production increases is by Thompson et al. (2016). This literature review is a report produced under contract to the USDA, and has not yet been published in a refereed journal but includes refereed journal articles in its review and summary. Using a weighted average of eight corn production estimates from studies that focused exclusively on corn ethanol and allowed for a long-term supply response, the review authors found that only 100 million additional bushels of corn would be produced for each additional billion gallons of corn ethanol, holding other supply and demand drivers constant. This result implies that the remaining 260 million bushels required to produce a billion gallons of corn ethanol would be derived from redistributing domestic uses among feed and other industrial uses, though a substantial portion of this redistributed corn would ultimately be returned to the feed market in the form of DG co-products.

[22] The estimates from Condon et al. (2015)'s absolute price change model (Table 4, column 1) were used to generate this result. This model does not impose a linear effect of corn ethanol on corn prices. Rather, it finds that the marginal effect of corn ethanol quantity on corn prices diminishes as the total corn ethanol quantity increases. The low end of the range assumes a general equilibrium modeling framework, and the high end assumes a partial equilibrium modeling framework.

4.3.4 Corn Ethanol Production Impacts from the RFS Program

The previous sections discuss corn market impacts from increases in corn ethanol production generally, resulting from a variety of policy and market drivers. A smaller set of studies has attempted to isolate the expected impacts of the RFS Program from those of other policy or market drivers. This literature is useful for understanding the likely effect of the RFS Program on the volume of corn ethanol production under different economic conditions.

Using economic simulation models to estimate the incremental contribution of the RFS Program to total corn ethanol production requires comparing a projection of the world with the RFS Program to a projection of the world without it but with all other policy and market drivers in place, such as oil prices, oxygenate requirements, ethanol tax credits and tariffs, and state biofuel policies (also referred to as the counterfactual). Chapter 6 addresses attribution of impacts to the RFS Program in detail using a variety of approaches. This section briefly summarizes results from the economic simulation literature.

Because most of the studies included in this section conducted prospective assessments, the authors had to make assumptions about key parameters affecting fuel and agricultural markets, such as global oil prices, food demand, weather shocks, and other policies. While the studies are data-driven exercises based on historical information and expectations about future trends, projections often differed from what actually occurred in subsequent years. In particular, few projections from these studies anticipated the expiration of the VEETC in 2011 or the fall in oil prices to an average of around $50 per barrel in the second half of the 2010s after averaging around $100 per barrel in the early 2010s.[23]

The U.S. EPA (2010)'s analysis of the final regulation implementing RFS2 compared the impacts of the full conventional biofuels mandate of 15 billion gallons to an estimate of U.S. corn ethanol production developed by the EIA for the 2007 Annual Energy Outlook (AEO). The 2007 EIA assumed that MTBE would continue to not be used as an oxygenate in the fuel supply, the VEETC would continue indefinitely, and oil prices would rise from $98 per barrel in 2010 to $142 per barrel in 2022, but it did not account for either the RFS expansion (i.e., RFS2, EISA passed in December 2007) or corporate average fuel economy (CAFE) standards required by EISA. Based on these assumptions, the EIA projected that the United States would produce 12.3 billion gallons of corn ethanol in 2022. Using EIA's AEO 2007 projection as the baseline scenario, the USDA (2019c) estimated that the conventional biofuels mandate would therefore result in a 2.7 billion-gallon increase in corn ethanol production in 2022 to reach the mandated level.

Babcock (2012) used the Food and Agricultural Policy Research Institute (FAPRI) agricultural sector model to examine the effects of U.S. ethanol policies using both prospective and retrospective

[23] All prices reported in 2018 dollars. For publications that did not report the dollar year, the year of publication was used to convert to 2018 dollars.

analyses. The retrospective analysis evaluated the joint impact of the RFS Program and the VEETC during 2005–2009 compared to a scenario without either policy but accounting for actual oil prices, which ranged from $62 to $107 per barrel. Babcock (2012) motivated the analysis by explaining that because high oil prices during this period boosted ethanol profits, it is not immediately clear what incremental role these policies played in encouraging ethanol expansion. Babcock estimated that the two ethanol policies jointly caused an increase in U.S. corn ethanol production of 1.3 billion gallons per year, on average, during this period. The effect of the RFS Program alone without the VEETC was not evaluated but presumably would have been smaller. The prospective analysis generated predictions of the effects of the RFS Program and VEETC both jointly *and separately* for the year 2011 using a stochastic PE model. The analysis projected an average increase in U.S. corn ethanol production of 1.57 billion gallons from the policies together, with 0.92 billion gallons attributable to the RFS Program alone, assuming oil prices of $137 per barrel.

Pouliot and Babcock (2013) applied a similar model to project how the RFS Program might affect markets under different assumptions about corn acreage, gasoline prices, and whether the fuel industry values the oxygenate and octane content of ethanol in addition to its energy content.[24] They found that if the fuel sector valued the octane and oxygenate content of ethanol, the impact of the RFS Program was expected to be relatively small or even nil (0 to 0.3 billion gallons) under $3 per gallon gasoline prices (corresponding to an oil price of $93 per barrel). Under $2 per gallon gasoline ($50 per barrel), Pouliot and Babcock (2013) projected that the effect of the RFS Program would be a more substantial 1.8 to 2.4 billion gallons, because ethanol usage in the absence of a mandate would have been lower.

Bento and Klotz (2014) examined the joint and separate effects of the VEETC and the RFS-implied ethanol mandate using a multimarket economic model representing the agricultural and fuel markets. Accounting for the fact that the VEETC was in place until 2011 and assuming that oil prices would grow from $95 to $108 per barrel, the authors estimated that U.S. corn ethanol production without the RFS Program would have been 11 to 12 billion gallons during 2011 to 2015. Therefore, they projected the incremental contribution of the RFS Program to be 1 to 3.5 billion gallons. Bento and Klotz (2014) also estimated that repealing the VEETC during this time frame would have had minimal impacts on corn ethanol production. However, they also considered a hypothetical scenario in which the VEETC was phased out in 2004. In this situation, Bento and Klotz (2014) projected that baseline U.S. corn ethanol production would have ranged from only 5 to 7 billion gallons from 2011 to 2015. Therefore, the RFS Program would have had a much larger incremental impact of 5 to 7 billion gallons during this period if

[24] Oxygenates are fuel additives that enhance combustion and reduce carbon monoxide emissions. Fuel additives such as ethanol can also increase the octane rating, which is the ability of the fuel to withstand compression before detonating.

the VEETC had not been in place in after 2004, even though letting it expire later was estimated to have minimal impact once expanded ethanol production capacity was in place.

Meyer et al. (2013) used a stochastic simulation model to examine the hypothetical effects of eliminating the RFS Program during 2017–2021 accounting for uncertainty in the distribution of crop yields, non-biofuel crop demands, and oil prices. At average oil prices of $99 per barrel, they estimated that eliminating the RFS Program would be expected to reduce U.S. corn ethanol production by about 1.5 billion gallons from an average level of 15.8 billion gallons with the RFS Program, whether or not corn yields were assumed to improve. They found that corn ethanol production often exceeded the mandated level because the conventional ethanol mandate was not binding under many of the simulations.[25]

Tyner and Taheripour (2008a) and Tyner et al. (2010) used a multimarket model of agricultural and fuel markets to examine the effects of different policy instruments and oil price scenarios. These studies did not model a particular year but calibrated their model mainly to 2006 data to compare the effects of the 2015 15 billion-gallon conventional biofuel level under the RFS Program with other policy instruments. Tyner and Taheripour (2008a) estimated that a fixed per-gallon ethanol subsidy (comparable to the VEETC), but no RFS mandate, would result in 3.3 billion gallons of U.S. corn ethanol production at $47 per barrel oil prices, 10 billion gallons at $70 per barrel, 13.7 billion gallons at $94 per barrel, and 16 billion gallons at $117 per barrel assuming no demand shock. These results suggested that a 15 billion-gallon conventional ethanol mandate would have had no incremental impact on U.S. corn ethanol production (i.e., it would not have been binding) with an oil price of $117 and the VEETC in place. They also examined the expected effects of corn yield increases and projected that with a hypothetical corn yield increase of 30%, the RFS Program would not have had an effect at oil prices as low as $70 per barrel. Tyner et al. (2010) also found that the oil price was a critical driver of ethanol production levels absent the mandate; they estimated that the 15 billion-gallon conventional ethanol mandate would not have been binding at $92 or more per barrel oil prices with a fixed ethanol subsidy similar to the VEETC.[26]

Figure 4.8 illustrates the role of oil prices in determining the incremental impact of the RFS Program on U.S. corn ethanol production. Using the relevant scenarios from the prospective analyses discussed above, the figure plots the difference in ethanol quantities between the RFS and no-RFS

[25] Debnath et al. (2017) used the same model to examine the effect of low and high oil prices on corn ethanol production in the 2023–2025 timeframe assuming the RFS Program continues. Under low oil prices (about $50 per barrel), the RFS Program is binding, but under high oil prices (about $220 per barrel), over 18 billion gallons of corn ethanol would be produced in the U.S., exceeding mandated levels.

[26] In a recent publication, Lark et al. (2022) assessed the effects of the RFS2 mandates on corn and corn ethanol production. However, due to several underlying assumptions, this study is better characterized as an estimate of the effect of the increase in corn ethanol demand above the 2005 RFS1 mandates from many factors, including, but not limited to, the RFS2. See Chapter 6, section 6.3.3 for additional discussion of this study.

scenarios at different reported crude oil prices (converted to 2018 dollars). As results across and within studies generally show, higher oil prices are expected to lead to higher corn ethanol production even absent the RFS Program. (The exception is Bento and Klotz (2014), who assumed that oil price increases would occur simultaneously with increases in the RFS required volumes.) Most studies projected that the incremental impact would be modest or even nil at oil prices similar to the levels actually seen in the early 2010s (>$90/barrel in real 2018$, see Chapter 6, Figure 6.4). Note also the large difference between estimates at a given range of oil prices of $40-$55 if a study included the octane value of ethanol (Pouliot and Babcock, 2013) or not (Tyner and Taheripour 2008, Tyner et al. 2011).

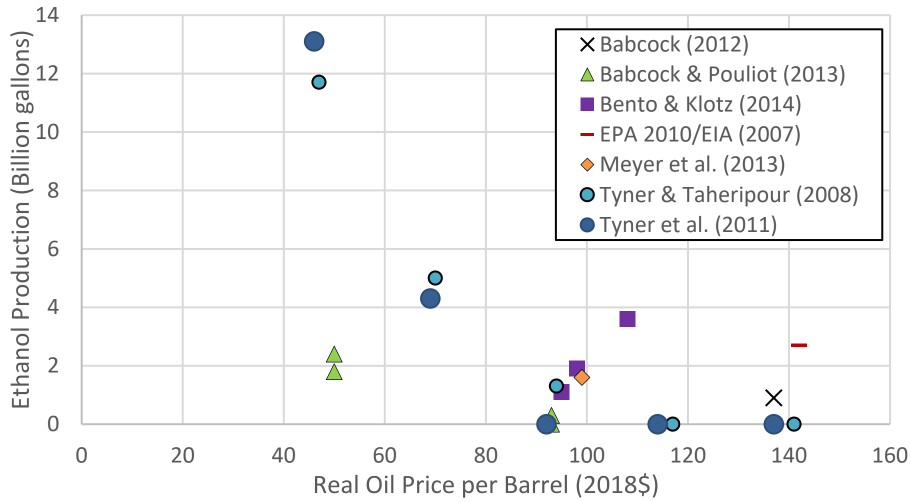

Figure 4.8. Incremental effect of RFS on U.S. corn ethanol production.[27]

4.4 Soybean Markets

Soy biodiesel is made from the oil extracted after crushing the soybean into meal. This intermediate market mediates the effects of the RFS Program between the biodiesel and soybean markets. When demand for soy biodiesel increases, the vegetable oil market may respond by increasing the production of soybean oil (or other vegetable oils), by substituting away from soybean oil to other oils, or some combination of these two responses. For this reason, it is important to understand the market dynamics of both the soybean oil and soybean markets. For more details on the soybean oilseed crushing process and the production of soybean oil and meal, see Chapter 3.

[27] Estimates are from Babcock (2012)'s forward-looking analysis of 2011 impacts; Pouliot and Babcock (2013)'s projections for 2014 using a demand curve reflecting oxygenate and octane value and 85 and 90 million harvested acres; Bento and Klotz (2014); EPA's comparison of RFS2 with the 2007 U.S. EPA (2010); Meyer et al. (2013)'s no corn yield improvement scenario during 2017–2021; Tyner and Taheripour (2008b)'s RFS and fixed subsidy with no demand shock scenarios; and Taheripour et al. (2011)'s RFS and fixed subsidy scenarios.

4.4.1 Overview of Soybean Oil Markets

As discussed in Chapter 3, there are two primary uses of soybean oil in the United States: domestic vegetable oil and biodiesel production. Since 2004, domestic utilization of soybean oil in biodiesel has steadily grown while other domestic uses have declined (Figure 4.9). In the 2019/20 marketing year, biodiesel production used approximately 32% of soybean oil production.

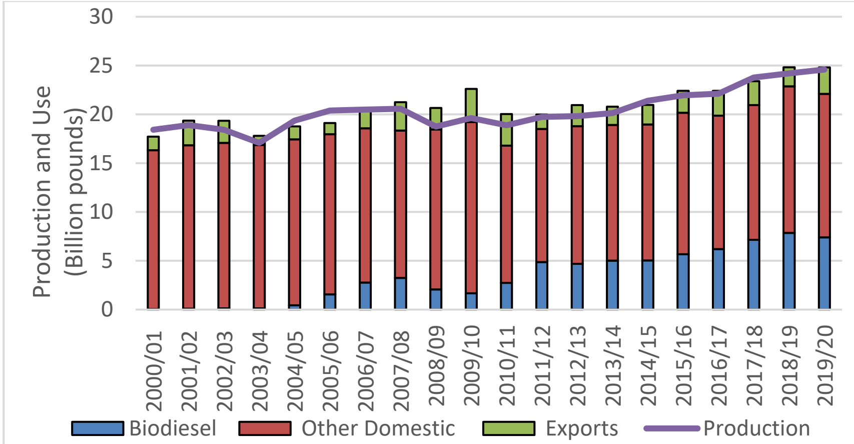

Figure 4.9. Soybean oil production and uses (2000/01 to 2019/20 marketing year). Quantities are reporting by marketing year. Marketing year runs from October to September. Source: USDA (2022a).

Production of soybean oil has risen steadily since the 2010/2011 marketing year. Soybean oil yields, that is the amount of soybean oil extracted from a bushel of soybeans, have not changed over this time period (USDA, 2021). Soybean oil prices peaked in 2004, 2008, and 2011 (Figure 4.10). These three peaks coincided with crude oil price spikes (Figure 4.2), large increases in utilization of soybean oil for biodiesel (Figure 4.9), lower crop yields, and/or increases in soy biodiesel and soybean prices (Figure 4.10). Since 2011, both soybean and soybean oil prices have trended downward even as demand for biodiesel increased. During this period soybean prices have also trended downwards.

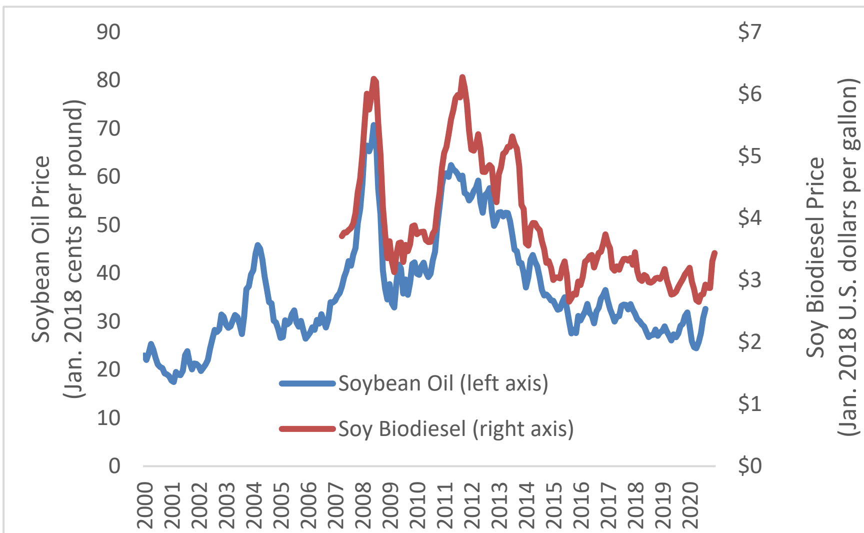

Figure 4.10. Inflation-adjusted soybean oil and soy biodiesel prices (2000–2020). Soybean oil prices are for crude, tanks, freight on board (FOB) central Illinois. Soy biodiesel prices are B-100 (soy methyl ester 2) FOB at IL, IN, and OH. Vertical axes are scaled to show approximate relative value.[28]

[28] Source: Soybean oil prices are from USDA (2021) and soy biodiesel prices are from USDA (2022a).

Box 4.1. Fats, Oils, and Greases (FOGs)

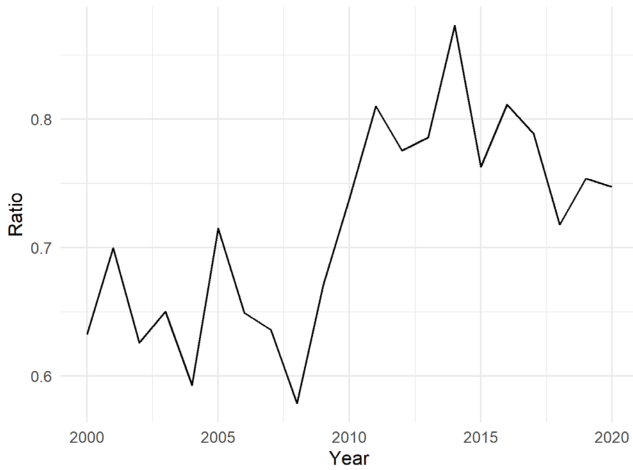

Figure B.4.1. U.S. annual yellow grease to soybean oil price ratio.[1]

As discussed in Chapter 3 (section 3.2.2), FOG is a term that covers animal byproducts and grease from food-handling operations and are often processed at rendering facilities for use in animal feed, pharmaceuticals, cosmetics, lubricants, plastics, and biofuels. FOG includes animal fats (e.g., tallow, white grease, and poultry fat) obtained from slaughterhouses and livestock farm waste, used cooking oil (UCO) generated at commercial and industrial cooking operations, and grease recovered from traps installed in the sewage lines of restaurants/food processing plants and wastewater treatment plants. Biodiesel produced from FOGs (hereafter "FOG biodiesel") increased from 1.7% in 2009 to 4.4% in 2019 (See Table 2.2).

Typically, FOGs are not used in their raw form—industries purchase purified material from rendering plants. Peer-reviewed and public data on the economics of FOGs and FOG biodiesel are limited, but the substitutability of FOGs with other oils is common. Since the implementation of biofuel policies, many FOGs have been exchanged in Europe at prices that are only slightly discounted from virgin oils (Chudziak and Haye, 2016). In the United States, yellow grease sold for 60–70% the price of soybean oil from 2000 to 2010, but since the implementation of RFS2 in 2010, yellow grease sold in the range of 80% the price of soybean oil (Figure B.4.1). The effect of biofuel policies on FOG prices and other market effects is an area for further research, though the relative lack of data is a challenge. Because FOGs are often considered a waste product or byproduct of some other primary product or activity, the upstream environmental effects of the FOGs are often allocated to the primary product rather than to FOGs. However, given the not-insignificant prices paid for FOGs (Figure B.4.1), this assumption may be inappropriate, and other studies treat FOGs as a co-product. This is an active area of research.

[1] Data from USDA-AMS (2021). Prices = High Bid (cwt). For yellow grease data: Location = "Minneapolis - Duluth, MN", Delivery Period = "20 Day Delivery Period", 2015 and after Pricing Point = "Mills and Processors." For soybean oil data: for 2006 and later Location = "Minnesota, MN"; for 2005 and before Location = "Decatur-Central Illinois, IL." These choices were made to align yellow grease and soybean oil data to the extent possible.

4.4.2 Overview of Soybean Markets

Soybean markets depend on many of the same supply-side and demand-side factors that influence corn markets. However, the market share of these factors and therefore their importance in production and prices differ from corn markets.

Soybeans are not a direct feedstock into biofuels production at the biorefinery. It is the oil that is used as the feedstock to soy biodiesel production. In the 2019/20 marketing year almost half of the utilization of soybeans was for crushing into oil and meal domestically (see Figure 4.11). Soybean oil is a more highly valued product by weight than soybean meal. Even though only 19% of the soybean is comprised of oil, the value of that oil per bushel of soybeans is around a third of the total value of the crush (see Figure 4.12).

Weather is an important supply-side factor in soybean markets. National-level impacts of weather in soybean markets can best be observed in deviations from expected national average yields (Figure 4.13). Bad weather years, such as the droughts of 2003 and 2012, are below the 2000–2020 trend line.

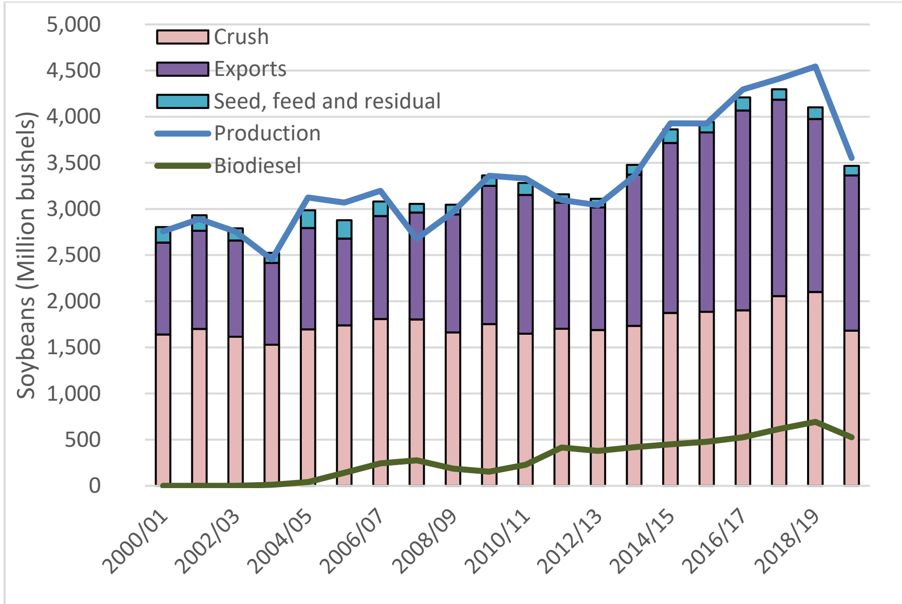

Figure 4.11. Soybean production and utilization. The biodiesel line represents the quantity of soybeans that would need to be crushed in order to extract oil equal to that utilized for biodiesel. Source: USDA (2021).

Other supply-side factors are best summarized by production cost shares (Table 4.1). The largest share is for land, comprising almost 33% of the cost of production (USDA, 2022a). The second largest share is machinery costs at 26% of production cost. Fuel, fertilizers, pesticides, and other chemicals, all of which are dependent on oil prices, constitute only 14% of production cost. This share is lower than for corn because soybean do not require nitrogen fertilizer. Finally, seed costs are about 13% of the cost of production.

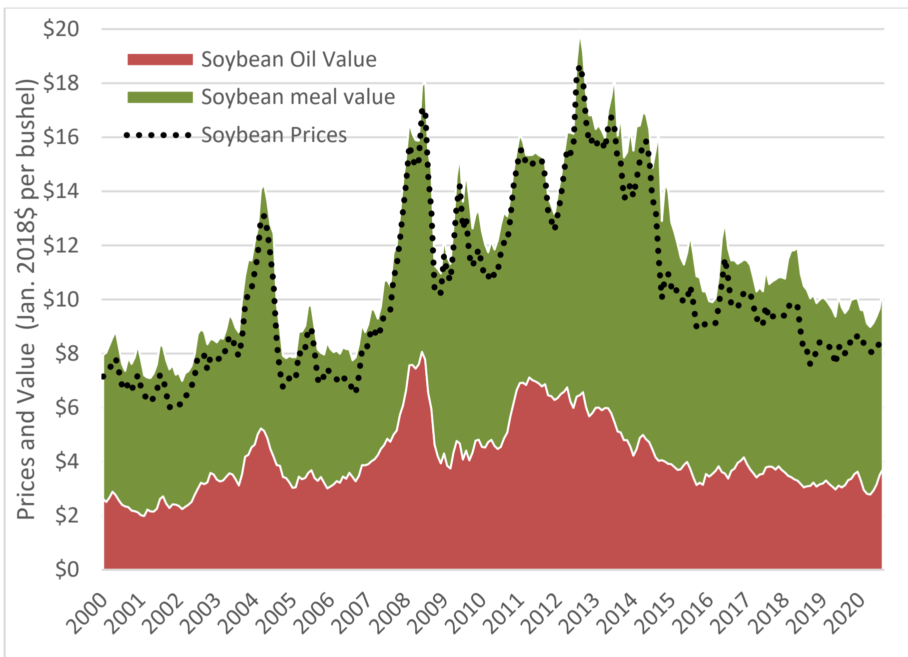

Figure 4.12. Soybeans and related products prices and value. The soybean oil and soybean meal values are stacked to show the total value of the products produced when crushing soybeans. Source: USDA (2021).

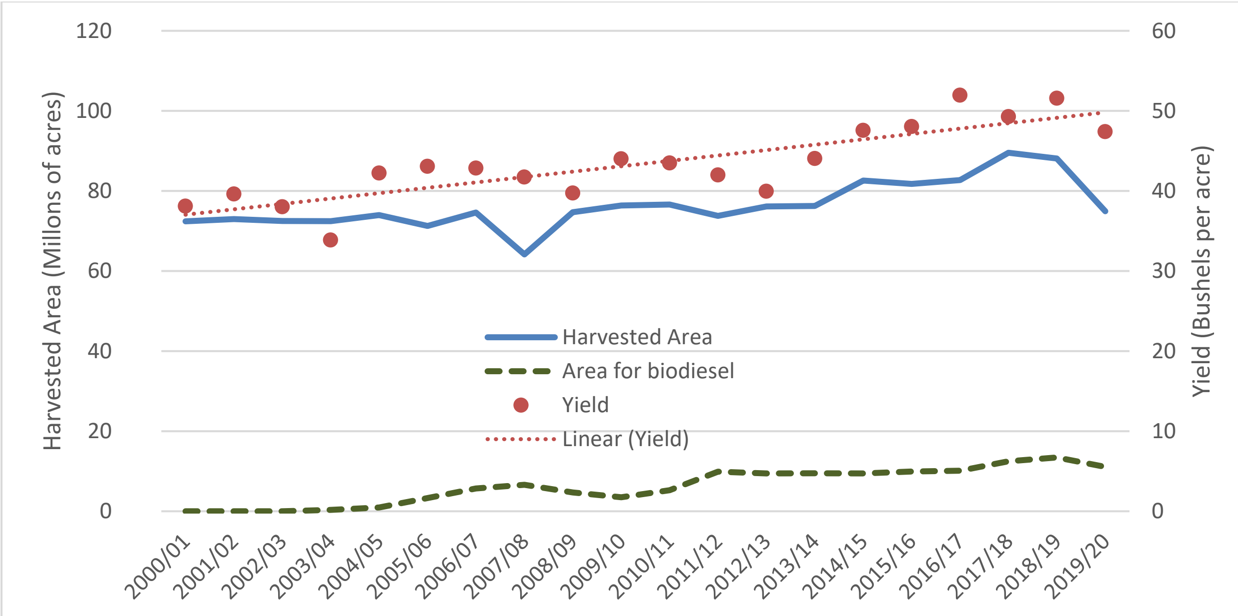

Figure 4.13. Soybean yields and acreage. Area for biodiesel is calculated by dividing the utilization of soybean oil for biofuels by the average soybean oil yields. Source: USDA (2021).

When soybeans are harvested in the fall they are stored until they are utilized. Year-end stockpiles can be replenished in good years and drawn down in bad years. In the 2019/20 marketing year, stocks ended at 15% of production. This level of storage is higher than in most years due to record production in the 2016/17 through 2018/19 marketing years and a decrease in exports in the 2018/19 and 2019/20 marketing years. The lowest level of storage since 2000 was in the 2014/15 marketing year at just 2% of production. This low level of storage was a result of lower-than-expected yields in 2011, 2012 and 2013.

4.4.3 Soybean Price and Production Impacts from Biodiesel Policies.

There is a small but growing body of literature that attempts to analyze the impact of the RFS Program on the soybean and biodiesel markets. However, to the authors' knowledge there is no comprehensive review or meta-analysis of this literature. Therefore, relevant individual papers and their estimates of the impacts in soybean markets are discussed.

Like the corn ethanol literature, most of these papers rely on prospective mathematical modeling to make future predictions of impacts. Therefore, the actual impacts depend on realized economic conditions, including RFS Program volume obligations. For example, these papers often use a biomass-based diesel volume obligation of 1 billion gallons, which is the minimum volume in RFS2. However, the actual RFS Program biomass-based diesel and advanced biofuel volume obligations set by EPA have exceeded 1 billion gallons, with the advanced biofuel volume obligation being the more stringent of the two (see Chapter 1, Table 1.1 for details on annual obligations).

Most of these studies estimate the joint impact of both the implied conventional ethanol and the biomass-based diesel RFS Program volume obligations (in some cases the cellulosic mandates are also included). Due to the nested standards, biodiesel and renewable diesel supply the vast majority of the advanced biofuel volume obligation. Furthermore, the nested nature of the RFS Program allows biodiesel to be used to backfill for shortfalls in conventional biofuel to meet the total renewable fuel standard. Thus, the total renewable fuel volume obligations also impact the soybean market. In addition, the total renewable fuel mandate also indirectly impacts the soybean market through the increased competition of corn production with soybean production.

As noted earlier, in 2010 EPA conducted a regulatory impact analysis for the RFS Program, which includes estimates of the economic impacts (U.S. EPA, 2010). The impacts of achieving the RFS Program volumes were estimated with two different models: the first being the Forest and Agricultural Sector Model (FASOM) and the second being the FAPRI model. Both these models use EIA's 2007 AEO as the reference case (EIA, 2006). The FASOM model estimated an 7.9% increase in soybean prices per billion gallons of biodiesel. However, after accounting for international trade, the FAPRI model estimated a lower 1.8% increase in soybean prices per billion gallons of biodiesel.

Using the Center of Agricultural and Rural Development (CARD) variant of the FAPRI model (FAPRI-CARD), Hayes et al. (2009) examined the impact of biofuel support on agricultural markets. Their estimates of the impact of the removal of biofuel supports includes the elimination of tax credits[29] and the RFS Program volume mandates, including cellulosic mandates, as well as import tariff and duties. They estimated that these biofuel supports increased soybean prices by 11.6% per billion gallons of biodiesel. Even with an increased demand for biodiesel, Hayes et al. (2009) estimated that soybean production would decline 3.9% per billion gallons. While this might seem counterintuitive, it is likely due to the increased competition for land for the production of corn for ethanol.

Babcock (2012) used the FAPRI-CARD model in a retrospective analysis of the expansion of ethanol production and a prospective analysis of the market impacts of the RFS Program in the United States. In the retrospective analysis Babcock found that even when holding the biomass-based diesel mandate fixed, increasing the production of corn ethanol drove up soybean prices (soybean production was not reported). The reason for this is that increasing corn production and acreage to meet ethanol demands reduces the supply of some inputs (e.g., land) for soybeans, driving up soybean prices. In the prospective analysis, Babcock analyzed the impact of both the implied conventional and biodiesel mandates. Babcock estimated a 3.9% increase in soybean prices per billion gallons of biodiesel.

[29] This paper analyzed the impact of two biofuel tax credits: ethanol ($0.51 per gallon) and biodiesel ($1.00 per gallon) blenders' tax credits. Biofuel tax credits can be used by blenders to reduce the cost of biofuels.

Meyer et al. (2013) used the University of Missouri (MU) variant of the FAPRI model (FAPRI-MU) to examine the impacts of the U.S. biofuel mandates (i.e., implied conventional, cellulosic, and biodiesel). They found that the RFS Program volumes would increase the price of soybeans by 8.5% per billion gallons of biodiesel on average in 2017–2021. They also found that the RFS Program volumes would increase soybean production by 1.4% per billion gallons of biodiesel.

In a global analysis using a modification of the Global Trade Analysis Project (GTAP) model, Huang et al. (2012) estimated the impact of government mandates (including the RFS Program) and outcomes in the United States, European Union, and Brazil. They ran four scenarios that made two assumptions about the elasticity of substitution between fossil fuels and biofuels and two assumptions about energy prices. They found soybean price impacts of between 3.9 and 6.5% per billion gallons of biodiesel and soybean production impacts from 4.5 to 4.6% per billion gallons of biodiesel. When the energy prices are low ($60 per barrel in 2004 dollars), the U.S. impact of government mandates are higher than when energy prices are high ($120 per barrel in 2004 dollars). Notably, when there is a high rate of substitutability between fossil fuels and biofuels and energy prices are high, the mandates have no impact on soybean markets.

Cui and Martin (2017) used a stylized multi-market equilibrium model of the agricultural and oil refining sectors and fuel and food consumption to analyze the impacts of a biodiesel mandate. The model includes two regions, one for the United States and the other for the rest of the world. The baseline consumption of biodiesel was 1.55 billion gallons. The study analyzed biodiesel mandates of 2.0 and 3.4 billion gallons. They found that the mandates increased soybean oil demand for biodiesel. This demand was met by decreasing soybean oil exports from the United States and not increased production of soybean oil or soybeans (0.2% per billion-gallon increase in the mandate). The largest impacts on prices were for soybean oil (8.3–8.9% per billion-gallon increase in the mandate). The study did find that increasing the biodiesel mandates increased soybean prices by 1.9–2.0% per billion gallons.

Moschini et al. (2017) utilized an economic model of U.S. supply of corn, soybeans, and crude oil; biofuel refineries; crude oil refineries; U.S. demand for food and fuel; and rest of the world demand to analyze the impacts of the actual 2015 and projected 2022 volume obligations. They found soybean price increases of 8.2–8.9% per billion gallons of biodiesel and soybean production decreases of 0.9-3.5% per billion gallons of biodiesel. The reason for the decrease in soybean production was due to the increased demand for corn production and corn ethanol.

To summarize, the literature estimates a wide range of impacts of the RFS2 biomass-based diesel volume obligations on soybean markets (Table 4.2). This is in part because these studies estimate the impact of a variety of different policy combinations. Only one of the studies (Cui and Martin, 2017) separated out just the impact of the RFS2 biomass-based diesel volume obligations. Ethanol volume obligations could impact soybean markets even in the absence of a biomass-based diesel obligation due to increased competition for inputs such as land. The largest impacts are estimated when biomass-based diesel obligations are modeled jointly with the implied conventional and cellulosic ethanol obligations. Given that the actual cellulosic ethanol obligations have been much lower than those commonly modeled, the studies that model only an implied conventional ethanol obligation should be preferred. These studies find that the impact of biodiesel is to increase soybean price by 1.8 to 8.9% per billion gallon increase of biomass-based diesel. These studies also found mixed evidence of soybean production impacts. This result may be due to the variation in the additional polices included in the analysis. Corn ethanol mandates in the absence of biodiesel mandates would likely decrease soybean production, whereas, biodiesel mandates would increase soybean production. The net effect of both mandates depends on a variety of factors.

Table 4.2. Soybean market impacts from biodiesel.

Study	Model	Prices (% per billion gallons produced)	Production (% per billion gallons)	Policies included in addition to biomass-based diesel mandates
U.S. EPA (2010)	FASOM	7.9%	NA	Conventional and cellulosic ethanol mandates
U.S. EPA (2010)	FAPRI	1.8%	NA	Conventional ethanol mandates
Hayes et al. (2009)	FAPRI-CARD	11.6%	-3.9%	Conventional and cellulosic ethanol mandate and VEETC
Babcock (2012)	FAPRI-CARD	3.9%	NA	Conventional ethanol mandate
Meyer et al. (2013)	FAPRI-MU	8.5%	1.4%	Conventional and cellulosic ethanol mandates
Huang et al. (2012)	GTAP	3.9–6.5%	4.5–4.6%	U.S. conventional mandate and EU and Brazil mandates
Cui and Martin (2017)	Cui et al. (2011)	1.9-2.0%	0.2%	No explicit policies but calibrated to 2014 when renewable fuel mandates were in effect.
Moschini et al. (2017)	Moschini et al. (2017)	8.2-8.9%	-0.9 to -3.5%	Conventional ethanol mandates

4.4.4 Biodiesel Production Impacts from the RFS Program

The consumption of biodiesel in U.S. diesel transportation blends has become increasingly important. The legislated minimum volume of biomass-based diesel in the RFS Program was 1 billion gallons in 2012 (EISA, 2007). After that time, the biomass-based diesel volume obligations continued to be a minimum of 1 billion gallons, but EPA could increase volume obligations. Biomass-based diesel is the only biofuel for which EPA has this authority prior to 2023, and EPA has used it to steadily increase the volume obligations from year to year, reaching 2.43 billion gallons in 2020. However, in practice EPA used the advanced biofuel standard to drive up biomass-based diesel volumes to higher levels to meet the statutory obligations for that standard, and instead set the biomass-based diesel standard at a level that would guarantee at least a certain portion of the advanced biofuel standard would be met with biomass-based diesel. Blending biomass-based diesel was one of the lowest-cost biofuels to meet the advanced mandate as evidenced by similar prices for advanced and biomass-based diesel RIN prices (see sections 4.2 and 4.3 for more details). Recently, after ethanol reached the E10 blend wall, biodiesel has also become the lowest-cost fuel to meet the total renewable mandate above the blend wall. Therefore, biomass-based diesel is being blended to meet three out of the four mandates. Chapter 7 addresses attribution of impacts to RFS in detail using a variety of approaches. This section reviewed the literature on the potential impacts of RFS Program on biodiesel.

The EPA RFS2 Regulatory Impact Analysis (RIA) (U.S. EPA, 2010) estimated a 1.3 billion-gallon increase in biodiesel production to 1.7 billion gallons in 2022. In the reference case, with no RFS Program mandates, the total production of biodiesel was estimated to be 0.4 billion gallons with 75% of that assumed to come from FOGs. In the RIA it was assumed that roughly 0.6 billion gallons would come from corn oil extracted from a byproduct of the production of corn ethanol and 0.4 billion gallons would come from FOGs in 2022. Soy biodiesel was assumed to contribute another 0.6 billion gallons.

Hayes et al. (2009) estimated that without tax credits, the RFS Program mandate, and import tariffs and duties, production of biodiesel would be limited to 0.3 billion gallons. They estimated that the biofuels support (i.e., tax credits, the RFS Program mandate, and import tariffs and duties) would increase production by 0.9 billion gallons with most of this increase coming from the production of biodiesel from soybean oil. This increase in production was to meet the consumption mandate of 1 billion gallons. The remaining 0.2 billion gallons would be exported. Notably they did not estimate that corn oil would be used for biodiesel production, which has been increasing in production since 2009 (Chapter 2, Table 2.1).

Babcock (2012) estimated that with no mandate biodiesel production would be very low (0.04 billion gallons). The biomass-based diesel mandate in the RFS Program was therefore assumed to increase production of biodiesel by 0.9 billion gallons in 2011. In that year, the biomass-based diesel volume obligation was 0.8 billion gallons.

Meyer et al. (2013) estimated that the biodiesel RFS2 mandate would increase biodiesel production by 0.9 billion gallons. Without the mandates this study estimated that production of biodiesel would be less than 0.4 billion gallons. Similar to many of the other studies, Meyer et al. (2013). assumed that the biomass-based diesel consumption mandate would be held constant at 1 billion gallons after 2012.

The global analysis by Huang et al. (2012) estimated that without mandates and with low energy prices ($60 per barrel), the production of biodiesel would be low (0.2 billion gallons). With low energy prices, the mandate would be binding and increase the production of biodiesel in the United States by 1.6–1.7 billion gallons (6–6.3 million tons of biodiesel). Notably, these authors set the U.S. biodiesel mandate at 1.9 billion gallons (6.9 million tons). When the energy price was high ($120 per barrel), the impact of the mandate was dependent on the substitutability of fossil fuels with biofuels. When the substitutability remained at historic levels, the mandates increased biodiesel production by 1.2 billion gallons (4.2 million tons). However, under the scenario with substantially increased substitutability, the U.S. biodiesel mandate was not binding, and there was no impact on production.

Moschini et al. (2017) estimated that biodiesel consumption without the RFS scenario would have been 0.7 billion gallons (model calibrated to 2015 as the benchmark year). This estimate is higher than the other studies in this chapter. Notably this study found that without the RFS Program no soybean oil would be used for biodiesel. Therefore, the feedstock must have come from some other source, such as FOGs or corn oil. This study estimated both actual 2015 and projected 2022 mandates. It found that the RFS Program increased the quantity of biodiesel by 1.1 billion gallons with the 2015 mandate and 2.6 billion gallons with the projected 2022 mandate.

To summarize, these studies find that production of biodiesel would have been low (0.2–0.7 billion gallons) without the RFS Program mandates and most of this biodiesel production would have come from FOGs (Table 4.3). These studies estimate that biodiesel production would have increased by 0.6–1.1 billion gallons per billion-gallon mandate for biomass-based diesel. Studies that modeled corn oil production from corn ethanol byproducts found that 0.6 billion gallons of corn oil biodiesel would be produced.

Table 4.3. Summary of estimates of biodiesel production with and without RFS Program and consumption volume obligations (billion gallons).

Study	Production without RFS	Production with RFS	Consumption Volume Obligation	Increase in Production with RFS (per billion gallons)
U.S. EPA (2010)	0.4	1.7	1.8 (includes biodiesel to meet advanced)	0.7
Hayes et al. (2009)	0.3	1.2	1.0	0.9
Babcock (2012)	0.04	0.9	0.8	1.1
Meyer et al. (2013)	0.4	1.3	Not reported	0.9
Huang et al. (2012)	0.2	1.9	1.9	0.9
Moschini et al. (2017)	0.7	1.8 (2015 RFS) 3.3 (2022 RFS)	1.8 (2015 RFS) 3.3 (2022 RFS)	0.6 (2015 RFS) 0.8 (2022 RFS)

4.5 Feed and Livestock Markets

Biofuels and the RFS Program may affect feed and livestock markets in addition to fuel and crop markets because the feedstocks for biofuels are also used for feed and livestock. Thus, understanding the impact of the RFS Program on feed markets requires an understanding of many agricultural market interactions. This includes the increased utilization of corn for ethanol, the increased supply of DGs and soybean meal into the feed market, and the competition of corn and soybean production for land, among other factors. Data from the USDA provide insights into these interactions between biofuels and feed markets. Given its crucial role in these interactions, the data on distillers' dried grains with solubles (DDGS) supply, disposition, and prices are first highlighted. DDGS are the dominant form of DGs in the United States and thus are emphasized here. Impacts of the RFS Program on livestock supply, demand, and prices occur primarily through the land and feedstock markets. However, changes in these two markets do not necessarily translate into proportional changes in livestock markets and associated consumer products due to many adjustment options along the supply chain. These adjustment options broadly include potential changes in the total and mix of livestock inventory, and changes in the total supply and mix of consumer livestock products.

4.5.1 Overview of Distillers Grains Markets

Growth in U.S. ethanol production has created an increased supply of its feed co-products, DGs, DDGS, corn gluten feed (CGF), corn gluten meal (CGM), and distillers corn oil.[30,31] Corn is a major ethanol feedstock source, and the production of these feed co-products has lessened the impact of corn's removal from the feed supply to produce ethanol. However, this initial growth surge of both ethanol and its feed co-products has slowed in recent years.

DGs are a byproduct of alcohol (e.g., ethanol) production. As they are removed from the distilling process, DGs have high moisture content and are mash in consistency. In this form they can be sold for consumption by livestock within a few miles of a facility, but long-distance shipping is not economical due to their higher moisture content contributing to greater weight and shorter shelf life. DDGs are distillers' grains dried to a moisture content of roughly 10%. In this form they can be economically shipped long distances either by truck, rail, barge, or container. In addition, condensed distillers solubles can be recovered from the refining process, dried, and either sold as a livestock feed supplement or added to DDGs to produce DDGS, a nutrient-rich form of DDGs.

DDGS are a co-product of ethanol production that can be used as an economical animal feed that provide both energy and protein. In many rations, each unit of DDGSs displaces approximately 1.2 units of corn or soybean meal (Hoffman and Baker, 2011). Because of their high nutrient value, they are an

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[30] The term "distillers' grains" refers to co-products generated by dry-mill ethanol plants, including distillers' wet grains (DWG), distillers' dried grains (DDG), distillers' wet grains with solubles (DWGS), distillers' dried grains with solubles (DDGS), and condensed distillers' solubles (CDS). Unless otherwise specified for the remainder of this report, the term distillers' grains will mean distillers' dried grains with solubles (DDGS), the most common form of distillers' grains fed to livestock.

[31] Ethanol production can also yield non-feed co-products, such as high-grade biogenic carbon dioxide used in food and beverage processing and other industrial uses (Xu et al., 2010).

important factor in the profitability of ethanol production facilities. Most DDGS are produced in the dry mill process, a production process that involves grinding the whole corn kernel and fermenting the resultant corn meal without separating out the component parts. Wet mills produce ethanol and other products by soaking the corn kernels and then separating the components to produce products such as ethanol, starch, and sugars. Since most fuel ethanol is produced using the dry mill process, the focus is on dry mill production of DDGS.

Currently, one bushel (56 pounds) of corn processed in a dry mill produces approximately 2.92 gallons of ethanol and 15.9 pounds of DDGS (RFA, 2020). An ethanol mill not only receives a return on the ethanol but also on the DDGS, which can be priced above corn on a weight basis.

DGs have been used as an animal feed since humans have been producing alcohol from corn. Until the 2000s and the advent of large-scale use of ethanol in gasoline in the United States, the smaller volume of DDGS produced by beverage distilleries and brewers limited its use as a feedstock. As production of ethanol ramped up significantly an increasing amount of DDGS were produced. From 2000 through 2019, DDGS production increased twelvefold. DDGS production from ethanol is projected to reach roughly 42 million tons (38 million metric tons) in 2018/19 (Figure 4.14).

While supply has grown significantly, the lack of a decline in prices suggests that demand has kept pace with supply. Real prices for DDGS have in most months ranged between \$100 and \$200 per metric ton (January 2018 dollars) with prices above \$200 per metric ton in 2008, 2011–2013 and 2015. Prices of DDGs are correlated with prices of other feed substitutes such as soybean meal and corn (Figure 4.15).

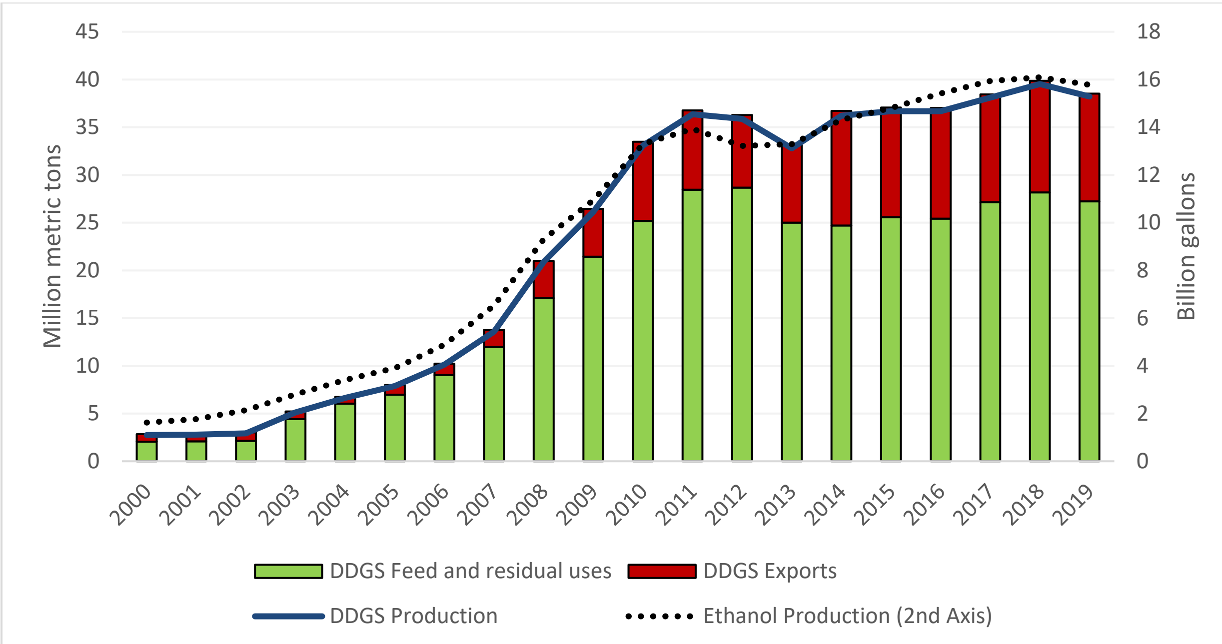


Figure 4.14. U.S. dried distillers' grain with solubles (DDGS) production and utilization. Source: USDA (2020b).

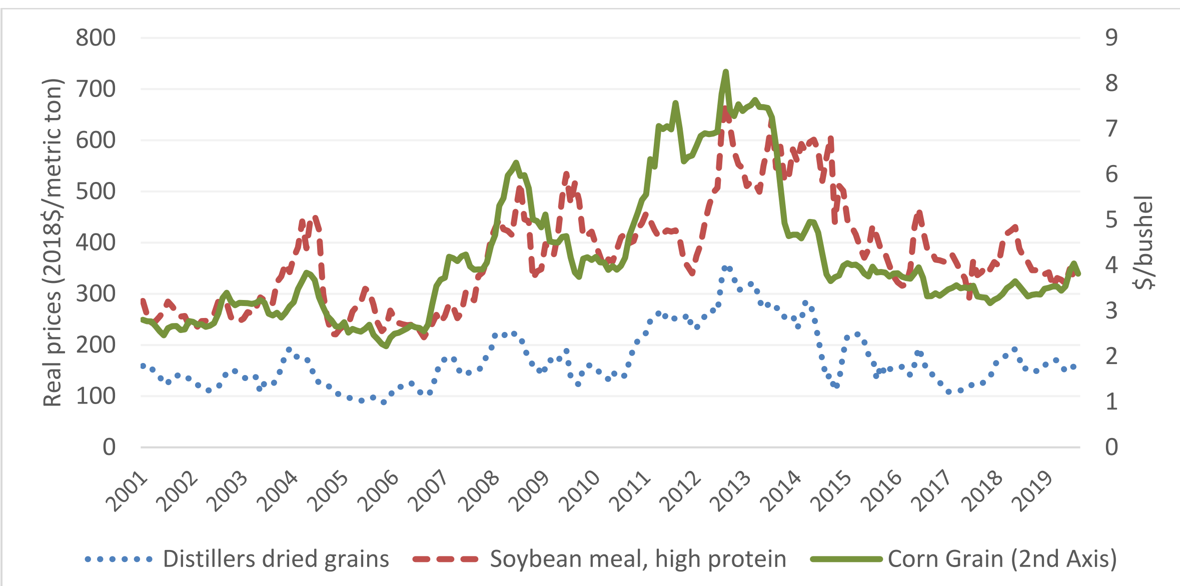


Figure 4.15. Monthly U.S. dried distillers' grains (DDGs), soybean meal (high-protein grade), and corn grain prices. Source: USDA (2020b).

Initially, as production of DDGS rose in the early 2000s most of it was utilized domestically. Beginning in the late 2000s and early 2010s a larger share of the production of DDGS were being exported. DDGS have become a significant agricultural export, accounting for over $2.2 billion in sales during 2019. This compares with $7.9 billion for corn grain exports.

4.5.2 *Overview of Feed Markets*

Figure 4.16 shows the use of major feed grains for livestock production in the United States. Following increases in 2003 and 2004, consumption of feed grains for livestock essentially flattened until 2007 with a significant drop in 2006. Total livestock feed grains consumption decreased from 2007 to 2012 and has increased steadily since 2013. Figure 4.16 shows the increasing role of DDGS in livestock feed as biofuel production increased, along with slight increases in oilseed meals (mainly soybean meal) and corn gluten feed. The significant drop in livestock feed grain uses between 2007 and 2012 is notable, but the role of biofuel is difficult to discern due to several reasons. Most of the large increases in ethanol production occurred by 2009, with much smaller increases since 2010. In addition, the Great Recession of 2008 and 2009 and its aftermath, as well as major drought conditions, occurred within this period, leading to significant impacts on global commodity markets only partly related to biofuels. Riley (2015) examined the data on hay and silage, which are the two other main livestock feeds apart from feed grains, and found that hay production peaked in 2004, declining significantly from 2004 to 2012. Riley (2015) suggested that the 9 million-acre decrease in hay production between 2002 and 2011 can be explained by increases in corn and soybean acreage largely due to increases in uses for biofuels during these years. Corn silage was found to increase slightly between 2004 and 2008.

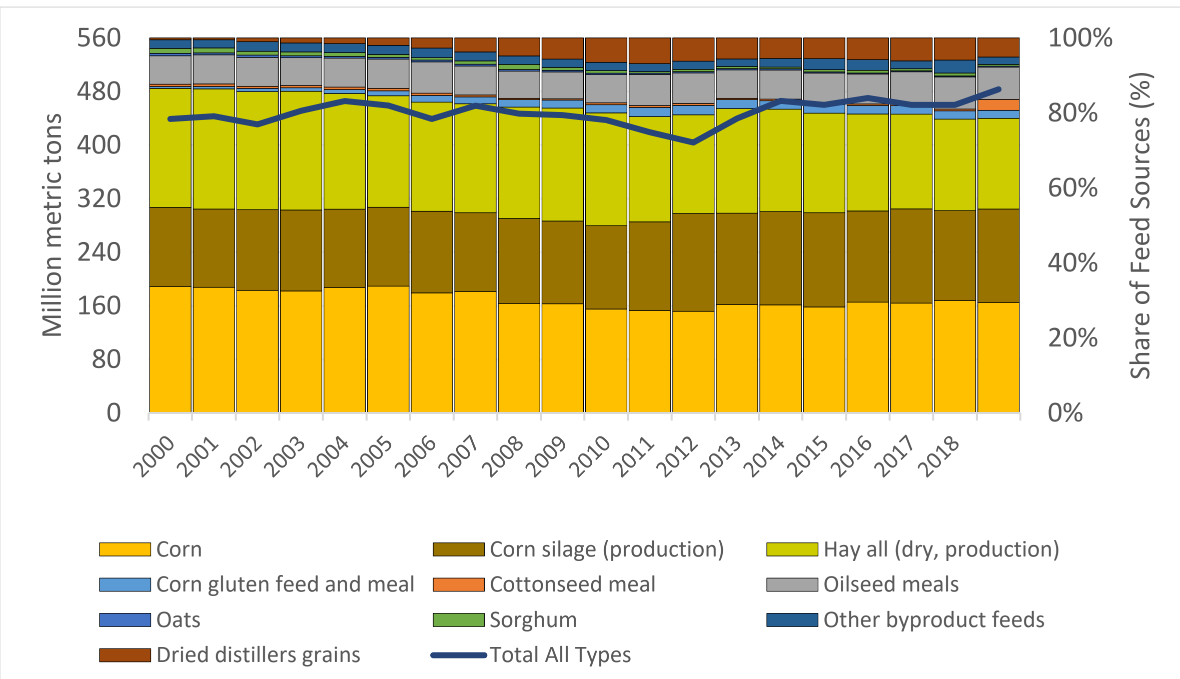


Figure 4.16. U.S. livestock grain-based feed use and production of hay and corn silage. Source: USDA (2020b).

4.5.3 *Feed Market Impacts from Biofuel Policies*

Many studies assess the market impacts of biofuels with mathematical models but few studies explicitly model or report on feed market impacts. In their prospective study on the U.S. agricultural sector, Hayes et al. (2009) estimated the feed market impacts of potential ethanol volumes from the RFS Program mandates. They found that production of DDGS increased by 11.7% per billion gallons of ethanol and that the price of DDGS increased by 2.8% per billion gallons of ethanol. The reason for this price increase, despite the increase in supply, is likely due to the increased utilization of corn for ethanol and the increase in the price of corn (2.9% per billion gallons of ethanol), a substitute for DDGSs. Hayes et al. (2009) found limited impact of the RFS Program volume mandates on soybean meal production (-1.2% per billion gallons of biodiesel). The reason for this is that they estimated that the RFS Program biodiesel volumes have no impact on soybean oil production even though it does impact soybean oil use. However, they found that it puts downward pressure on soybean meal prices (4.1% decrease per billion gallons of biodiesel). Moschini et al. (2017) and Cui and Martin (2017) found similar results for soybean meal prices (1–3% and 1.3–1.4% decrease per billion gallons of biodiesel respectively).

4.5.4 Overview of Livestock Markets

Livestock markets can be impacted both by the production of corn ethanol and soy biodiesel through feed markets. Feed markets further mediate the impact of biofuel production because livestock farms can substitute among the different feeds and adjust animal production. Rather than discuss each animal separately, this section presents an overview of the aggregate livestock sector. The relative numbers of USDA animal units by the type of feed consumption are shown in Figure 4.17. The variables are useful for comparing changes in animal inventory across different types of livestock in feed-weighted units (Grain Consuming Animal Units – GCAU; Grain and Roughage Consuming Animal Units – GRCAU, High-Protein Consuming Animal Units – HPCAU and Rough Consuming Animal Units – RCAU). An animal unit is based on the dry-weight quantity of a given feed type (i.e., grains, high protein, roughage, or composite) consumed by livestock. A set of factors or weights is developed for each type of livestock and poultry by relating consumption of the given feed for each type

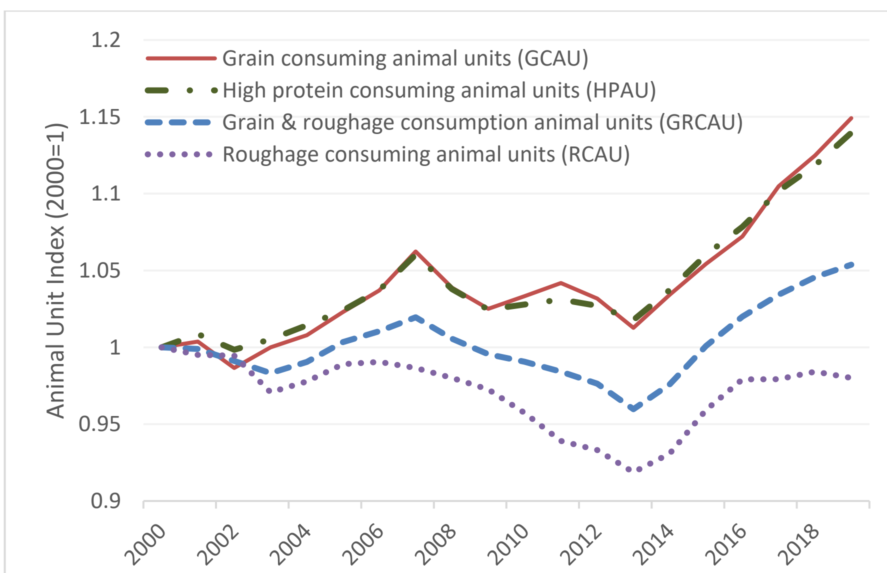


Figure 4.17. Quarterly U.S. livestock animal units (2000=1). Source: USDA (2020b).

of livestock to the feed consumed by the average milk cow.[32] Riley (2015) evaluated the aggregate and individual livestock GCAU between 2000 and 2013. Given the close relationship between the GCAU and HPCAU in Figure 4.17, the GRCAU, which is a composite of the GCAU and RCAU, provides an overall summary of feed-weighted animal inventory changes in the United States. The GRCAU index fell below 1 from 2000 to 2004, rising from 0.98 in 2003 to 1.02 in 2007, then falling to about 0.96 in 2013. It has risen steadily since 2013 with a value of about 1.05 in 2019. The near-steady decline in the RCAU in Figure 4.17 supports the conclusion in Riley (2015) that the decline in forage production led to a higher dependence of livestock on grain feeding even as biofuel demand for corn and soybean increased. Thus, the use of high-protein DDGs for livestock feed appears to have enabled the shift of about 9 million acres from hay to corn and soybean production, as noted above, to be accommodated without major impacts on total livestock inventory in the United States.

Figure 4.18 shows monthly price series for livestock and feed markets. The price ratios are in $ per 100 pounds to $ per bushel of corn and provide measures of the value of livestock in corn terms (bushels per 100 pounds). Declines in these price ratios mean that the corn price is rising faster than the livestock price, and vice versa. The price ratios in Figure 4.18 appear to move generally in opposite directions to the price of corn, with only a few exceptions. Of the nearly 300 corn price changes in Figure 4.17, the steer and heifer-corn price ratio moved in the same direction only 56 times and the hog-corn price ratio 93 times, with no apparent differences between the 2003–2012 period and the rest of the periods. Although feed is a significant cost factor for livestock

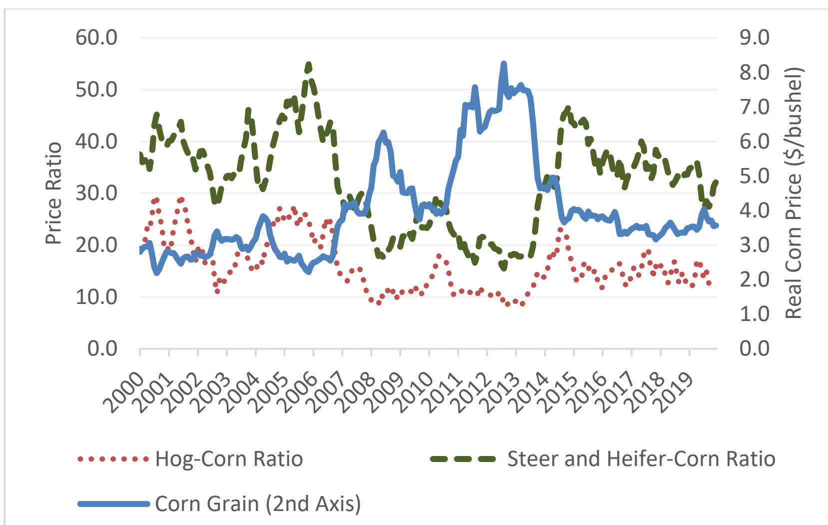


Figure 4.18. Monthly livestock-corn price ratios and corn price. Source: USDA (2020b).

[32] These data are calculated by USDA to evaluate aggregate livestock feed uses. A feed unit is equivalent to the feeding value of a pound of corn with 78.6% total digestible nutrients and is normalized by the feed consumed by the average milk cow. Four different animal units are calculated: (1) grain-consuming animal units based on consumption of concentrate feeds; (2) roughage-consuming animal units based on consumption of hay, pasture, and other forage; (3) grain-and-roughage consuming animal units combine livestock and poultry numbers on the basis of total feed of all kinds; and (4) high-protein animal units based consumption of only protein rich feeds. https://www.ers.usda.gov/data-products/feed-grains-database/documentation/.

production, these observations suggest that livestock prices do not respond quickly to changes in feedstock prices.

Meat production between 2000 and 2019 increased gradually (Figure 4.19), with the exception of 2008–2010. The pattern of increases in domestic consumption of these meat products is similar to production from 2000 to 2007 and since 2014, diverging significantly between 2008 and 2013. Prices of meat and other livestock products are shown in Figure 4.20, showing that, except for the 2003 and 2004, fluctuations in prices were largely consistent between 2000 and 2006. In 2007, when most global commodity prices rose rapidly, there were also considerable jumps in U.S. livestock product prices, except beef prices, which remained largely flat until 2009. Similarly, livestock product prices, except beef, appear to be affected by the price collapse in 2008 and 2009.

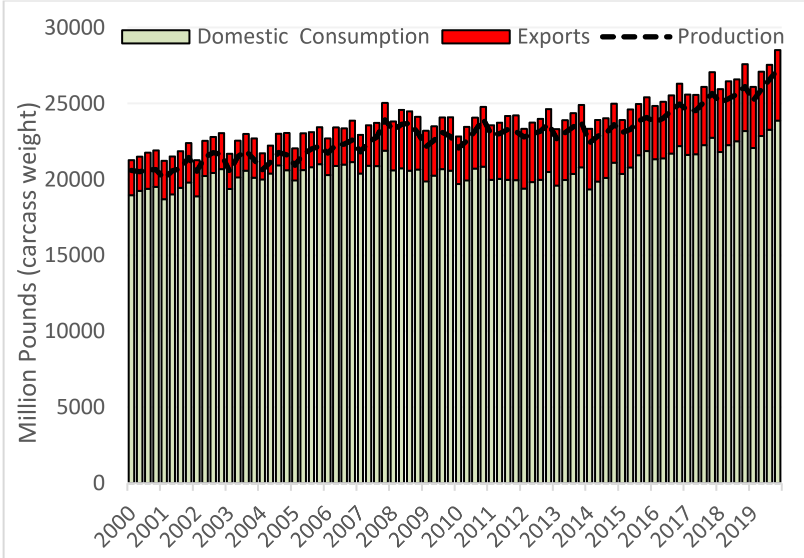
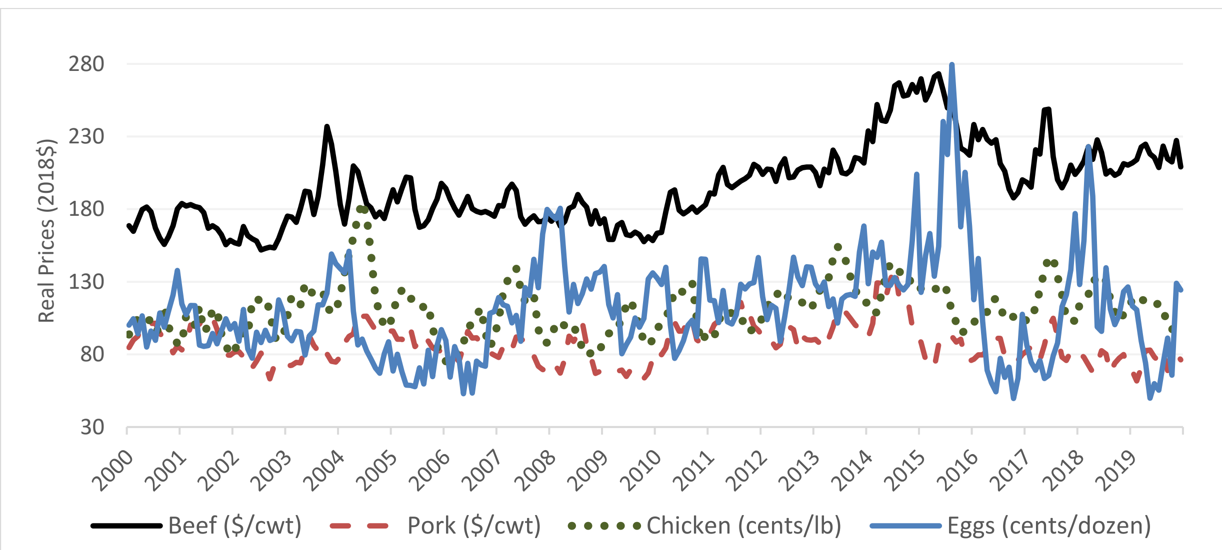


Figure 4.19. Quarterly U.S. red meat and poultry production and use (million pounds, carcass weight). Source: USDA (2019b).



cwt = carcass weight; lb = pounds

Figure 4.20. Monthly U.S. beef, pork, and poultry prices. Beef is central U.S. boxed choice 1–3, 600–900 pounds; Pork is central U.S. cutout composite; Chicken is Northeast breast with ribs; Eggs is combined regional. Source: USDA (2019b).

4.5.5 *Livestock Market Impacts of Biofuel Policies*

Although receiving less attention than other market impacts, there are a few studies evaluating the impacts of biofuels on livestock markets and one unpublished review. In addition, the land market in many PE and GE models of biofuels often include pastureland as one of the potential sources of cropland for corn and soybean production. While less studied than corn market impacts, changes in livestock markets have important implications for land use and emissions, as discussed in Chapters 5 and 8.

Babcock (2011) used the FAPRI-CARD model to evaluate the impact on livestock prices of holding ethanol production at marketing year 2004/2005 levels from 2005/2006 to 2009/2010. The 2004/2005 marketing year ethanol production was about 3.7 billion gallons and the simulations imply reductions in ethanol production of about 18% in 2005/2006 and 70% in 2009/2010. Prices for beef, pork, broilers, and eggs did not change significantly in 2005/2006 but increased for all other years. Egg prices were most affected with average price change per billion gallons change in corn ethanol production of 0.38%, 0.037%, 0.045%, and 0.079% for eggs, beef, pork and broilers, respectively.

Thompson et al. (2012) used the FAPRI model to evaluate several scenarios of biofuel waivers under the 2012 drought. They evaluated a "conventional gap" scenario that waives the implied specific corn starch portion of the RFS Program mandate but keeps the overall mandate, allowing corn ethanol to be voluntarily used to meet the requirement. Under the RIN stock rollover case of the "conventional gap" scenario, U.S. corn ethanol production declines by about 160 million gallons in 2012 (1.3%) and 980 million gallons in 2013 (6.6%). Price changes for beef, pork, and chicken were negligible overall but include a mix of increases and decreases, with the magnitude of retail price impacts less than 0.005% per billion-gallon change in corn ethanol.

Mosnier et al. (2013) used the global partial equilibrium model (GLOBIOM) to examine ±50% change in the total RFS2 mandates in 2030, keeping the proportion of different biofuels the same. Normalizing by the change in total biofuel quantity, the results imply that livestock prices in 2030 would decrease by 0.06% to 0.17% per billion-gallon increase in biofuel quantity under the +50% scenario and by 0.17% to 0.22% per billion-gallon increase in biofuel quantity under the -50% scenario.

Gehlhar et al. (2010) used the U.S. computable general equilibrium model USAGE to evaluate the impacts of increasing corn ethanol production to 15 billion gallons in 2022 from a baseline case of 8 billion gallons in 2022. Six scenarios combining oil prices (low, high) with U.S. ethanol tax credits (full, half, none) were simulated. The average percentage changes in output were -0.05% for dairy, -0.07% for beef, and -0.13% for other livestock per billion gallons of corn ethanol. Average changes in prices per billion gallons of corn ethanol are estimated at 0.06% for meat, 0.07% for fluid milk, and 0.05% for cheese.

Variants of the global general equilibrium model GTAP have also been used to examine U.S. RFS Program mandates. Hertel et al. (2010) evaluated the impacts of U.S. and EU biofuel policies between 2006 and 2015 using the static form of the GTAP general equilibrium model. Although the policies included both ethanol and biodiesel production within the United States and EU, the increase in biofuels was mostly due to an 184% or a nearly 10 billion-gallon increase in U.S. corn ethanol production. The impacts on livestock production was -0.06% per billion-gallon increase in U.S. corn ethanol production. Taheripour et al. (2011) used a slightly modified version of the GTAP model to examine similar scenarios as in Hertel et al. (2010) with a focus on the global livestock industry. Although separating the livestock sector into six categories, including three processed livestock products, the estimated impacts on U.S. outputs are similar to those in Hertel et al. (2010). Taheripour et al. (2011) provided total price impacts for the six livestock industries, with larger increases of just above 2% for U.S. dairy, other ruminants, and non-ruminant sectors, and less than 1% for the corresponding processed livestock sectors. Using the same change in ethanol volume as in Hertel et al. (2010), this translate to price increases of about 0.02% and 0.01%, respectively, per billion-gallon increase in U.S. corn ethanol. Oladosu et al. (2012) used a recursive dynamic GTAP model and estimated changes in U.S. livestock production due to the RFS2 mandates of -0.004% dairy farms, -0.035% cattle and ruminants, and +0.009 non-ruminants per billion-gallon increase in U.S. biofuels production to meet the RFS Program mandates in 2022.

The literature review by Thompson et al. (2016) also provided a summary of the livestock market impacts of the biofuels, including most of the studies noted above. The average increase in the prices of beef, milk, pork, and poultry per billion-gallon increase in corn ethanol across refereed studies focused on ethanol were 0.8 cents per pound of beef, 0.2 cents per pound of milk, 1.2 cents per pound of pork, and 1.1 cents per pound of poultry production, which all represent price changes of around 1% or less. Further, these are wholesale price impacts; wholesale food prices typically increase retail food prices by less than 10% (Leibtag, 2008). Similarly, the average decreases in production per billion-gallon increase in corn ethanol were 0.1% for beef, 0.4% for milk, 0.5% for pork, and 0.2 % for poultry.

4.6 Land Markets

4.6.1 Overview of Land Markets

Land is a primary and important input into both corn and soybean production. In addition, as discussed in Chapter 3 corn and soybean production compete for, and are commonly rotated in, the same fields. For these reasons, rather than discuss the impacts of biofuels on corn and soybean acreage separately, this section jointly discusses corn and soybean acreage and the land market. For a more detailed discussion of land use change see Chapter 5.

While the domestic supply of land is generally constant over time, the supply of cropland is flexible. Land can go in and out of crop production based on economic conditions. For example, when crop prices are low and crop production is not profitable, farmers might choose to idle a field or convert to pasture or grassland. The overall trend between 2000 and 2019 has been a slight decline in field cropland acreage (see Figure 4.21), although trends for intervals of this period show different trends (see Chapter 5). Over this same period, the total acreage of corn and soybeans has increased. Therefore, the acreage of other field crops (e.g., wheat) has been declining at a faster pace than the overall field crop acreage. See Chapter 5 for a more thorough discussion of trends in land cover and land management.

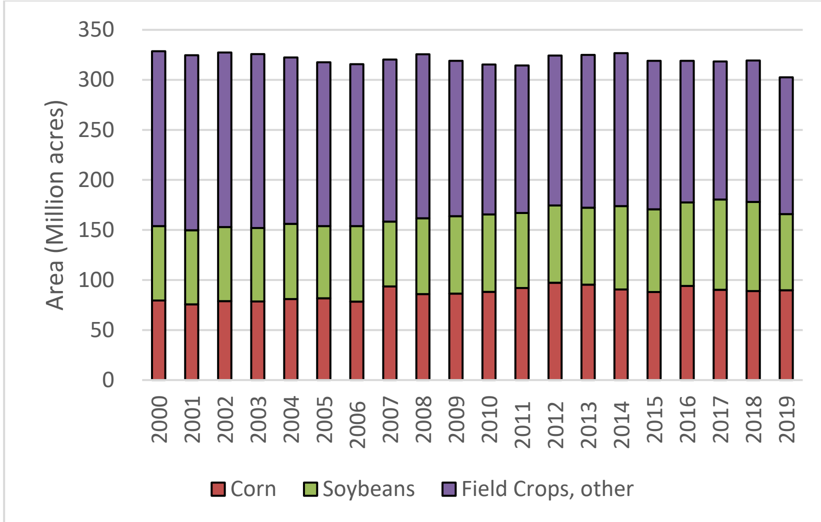


Figure 4.21. Field cropland acreage. Source: USDA (2019a).

Over this period, the value of cropland has also increased (see Figure 4.22). There are two basic measures of cropland value. One is the cash rental rate, which measures the price farmers pay annually to rent an acre of land. The other is the farmland value, which is the price farmers must pay to purchase an acre of land. These two are related. The farmland value (termed "value" in Figure 4.22) is just the value today of the expectations of future cash rents. Therefore, the cash rental rate can be thought of as the value of land in the current market and the farmland value as expectations about where the market is headed. Between 2000 and 2014, farmland value more than doubled, due in part to declining interest rates and strong farm earnings over this period (Nickerson et al., 2012). Higher

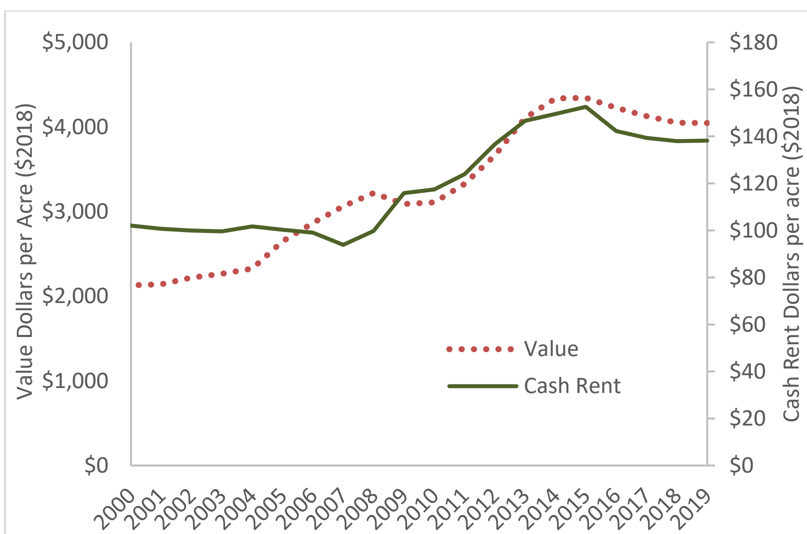


Figure 4.22. Average inflation-adjusted U.S. cropland prices (2001–2019). Source: USDA (2019c).

values may have contributed to increased interest from large investors seeking to diversify portfolios through farmland ownership (Ouma, 2020, 2018; Fairbairn, 2014). Cash rental rates dropped slightly between 2000 and 2007 despite the concurrent increase in farmland values. The difference in trends may be due to interest rate declines over this time period (interest rate on the 10-year Treasury bond declined from approximately 6% to 4% between 2000 and 2007), which puts upward pressure on farmland prices due to the reduced cost of ownership. Both cash rental rates and farmland value increased every year until 2015. This corresponds with a growth in the use of corn and soybeans for biofuels and increases in exports of soybeans. Since 2015, both the farmland value and cash rental rate have been declining. This corresponds with declining corn and soybean prices.

4.6.2 Land Market Impacts from Biofuel Policies

An increase in corn production for ethanol may come from a combination of yield and acreage increases, with implications for use of land and other agricultural inputs. Based on 14 studies that focused on corn ethanol and allowed for long-term supply response (7 refereed journal articles), the Thompson et al. (2016) review found that an estimated 1 million additional U.S. corn acres were used for each billion-gallon increase in corn ethanol production, on average. The same study found that based on 12 studies (9 refereed journal articles), cropland increased by 0.7 million acres for each billion-gallon increase in ethanol. Given trends in U.S. corn acreage during 2010 to 2018, an increase of 1 million U.S. corn acres represents an increase of slightly more than 1% in total corn acreage and about 3% of the acreage needed to supply corn ethanol utilization. Thus, to supply an additional 15 billion gallons of ethanol, corn and crop acreage would be projected to increase by 15 and 10.5 million acres, respectively.

Thompson et al. (2016) noted that estimated world crop acreage impacts are considerably larger, though fewer studies reported these impacts. Out of five refereed journal articles that modeled indirect international land use change, Thompson et al. (2016) calculated a weighted average increase of 25.4 million acres of cropland per dollar increase in corn price per bushel. Paired with Thompson et al. (2016) findings of a $0.15 median increase in corn prices per billion gallons of corn ethanol among studies allowing for a long-term supply response, this result implies a roughly 3.8 million-acre increase in crop area globally for each billion-gallon increase in corn ethanol production.

4.7 Conclusions

RIN Markets

- Renewable Identification Number (RIN) prices for renewable (D6) fuels provide evidence that the Renewable Fuel Standard (RFS) Program increased U.S. consumption of renewable biofuels in 2009 (and late 2008) and from 2013 to 2019.

- Advanced (D5), biomass-based diesel (D4), and cellulosic (D3) RIN prices provide evidence that the RFS2 increased U.S. consumption of advanced, biomass-based diesel and cellulosic biofuels in every year of RFS2 for which standards had been set for these fuels (i.e., starting in 2010).
- The close tracking of renewable (D6) and advanced (D5) RIN prices with biomass-based diesel (D4) RIN prices and the nested structure of the standards provides evidence that U.S. ethanol fuel consumption hit the E10 blend wall in 2013 and at that point biomass-based diesel became the lowest-cost marginal renewable fuel to meet all three of these volume standards. Therefore, due to the E10 blend wall and the RFS Program, renewable and advanced volume obligations increased consumption of biomass-based diesel (D4) in 2013–2019.

Corn Markets

- Studies estimated that the RFS Program could increase corn ethanol production between 0 and 5 billion gallons under scenarios with relatively high oil prices (greater than $60 per barrel in 2018 prices). Oil prices were greater than $60 per barrel for much of the period of growth in the corn ethanol industry.
- Even though it takes 360 million bushels of corn on average to produce a billion gallons of ethanol, the available estimates suggest that only about 100 million additional bushels of corn would be produced for each additional billion gallons of corn ethanol on average, holding other supply and demand drivers constant. The remaining 260 million bushels required to produce a billion gallons of corn ethanol are derived from redistributing domestic uses among feed and other industrial uses.
- A meta-analysis of studies published between 2007 and 2014 on the impact of biofuels estimated that for every billion-gallon increase in corn ethanol production between 2010 and 2019, corn prices would increase about 3–5%.

Soybean Markets

- Prospective studies suggests that the RFS2 increased biomass-based diesel consumption 0.9–1 gallons for every gallon in the biomass-based diesel volumetric standards. This is equivalent to an increase in biomass-based diesel consumption of 0.6–0.7 gallons for every gallon in the advanced volume obligations.
- For a prospective study that assessed the joint impact of the corn ethanol and biomass-based diesel volume standards, soybean production was estimated to increase 4.5–4.6% per billion-gallon increase in biomass-based diesel production.

- Studies of the impact of biofuels estimated that for every billion-gallon increase in biomass-based diesel production, soybean prices increased 1.8–8.9%.

Feed and Livestock Markets

- A review of studies of increased ethanol volumes estimated that the production of dried distillers grains (a byproduct of the ethanol production process) increased by 11.7% per billion gallons of corn ethanol. This helped to offset the displacement of corn from the feed markets to produce corn ethanol.
- RFS2 was estimated to have a limited impact on soybean meal production (decrease of 1.2% per billion gallons of biodiesel) and put downward pressure on soybean meal prices (decrease of 1–4.1% per billion gallons of biodiesel).
- On average, production decreases in beef, milk, pork, and poultry were estimated to be less than 0.5% per billion gallons of corn ethanol. Producer price increases in these livestock commodities were estimated to be less than 1 cent per pound per billion gallons of corn ethanol. The impact on consumer prices would likely be less than this.

Land Markets

- On average, an estimated 1 million acres of additional corn would be produced, and cropland would expand an estimated 0.7 million acres for each billion-gallon increase in corn ethanol production.

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5. Domestic Land Cover and Land Management

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Key Findings

- After decades of decline, increases in cultivated cropland have been recorded in multiple federal datasets, using a variety of methodologies, following the 2007 to 2012 period. This increase ranges from 6 to 10 million acres. The portion of this increase estimated attributable to the RFS Program is discussed in Chapters 6 and 7. Despite these recent increases, the extent of current cultivated crop acreage for this period is still below historic levels of crop cultivation.
- Based on the 2012, 2015, and 2017 National Resource Inventory (NRI), there has been a steady increase in agricultural intensity from 2007 to 2017 with a 10 million-acre increase in cultivated cropland coinciding with a 15 million-acre decline in perennially managed land (i.e., sum of lands in Conservation Reserve Program [CRP], pasture, and noncultivated cropland). This increase in cultivated cropland was largely driven by a net 26.5 million-acre increase in corn and soy with small grains and hay in rotation decreasing 16.5 million acres.
- More than half of the corn and soybean increase has largely come from other cultivated cropland (56%), while the rest has come from approximately equal proportions of pasture (13%), noncultivated cropland (20%), and CRP (11%). Corn likely has larger environmental effects than hay, pasture, and other crop types because corn typically uses more fertilizer, pesticides, and other inputs than other crops.
- Many of these changes are taking place throughout the Midwest, with hotspots in northern Missouri, eastern Nebraska, the Dakotas, Kansas, and parts of Wisconsin.
- Based on both the National Agricultural Statistics Service (NASS) and NRI, crop production is becoming less diverse in the United States as cultivated cropland, besides that of the increasing corn/soy acreage, continued to decline from 2000 to present.
- These changes in cultivated cropland acreage have coincided with increased corn and soybean yields and increasing adoption of a variety of best management practices like conservation and no-till practices.
- After short-term disruptions from weather and trade disputes with China, the USDA Long Term Agricultural Projections (LTAP) suggest that corn acreage and corn used for ethanol will remain relatively stable from 2020 to 2025, declining slightly thereafter. This projected decline is driven by increases in fuel efficiency decreasing total gasoline consumption, increasing crop yields, and E10 blend wall issues further exacerbated by slow growth in E15 and E85 consumption. Likewise, soybean acreage is projected to remain stable due to

increased yields meeting both domestic and international demand, especially to meet growing international meat consumption.

Chapter Terms: Census of Agriculture (Census), Cropland Data Layer (CDL), Cropland Reporting Districts (CRD), cultivated cropland, direct land cover and land management change, extensification, indirect land cover and land management change, intensification, land cover and land management (LCLM), land use, Long Term Agricultural Projections (LTAP), Major Land Use (MLU), National Agricultural Statistics Service (NASS), National Resource Inventory (NRI),

5.1 Introduction

Land cover and land management (LCLM) is defined as the physical cover of the land (e.g., corn, grass), and how that land is managed for a particular use (e.g., for corn cultivation in rotation with soy, for hay).[1] LCLM is not explicitly identified in Section 204 of EISA as one of the factors that EPA must analyze. However, LCLM is foundational to many of the other impacts that EPA is required to analyze (e.g., water quality, habitat of grassland); thus, this chapter provides a discussion of spatiotemporal trends in LCLM in the United States. International changes in LCLM are discussed in Chapter 16. As mentioned in section 2.1, because the intended focus of this report is on the effect of the RFS Program, the temporal period of emphasis is roughly 2005 (beginning with the Energy Policy Act) to present, with some years prior also provided for context.

Section 5.1.1 gives an overview of the drivers of change in LCLM, and a brief discussion of the general outcomes that may occur as a prelude to the environmental and resource conservation effects in Part 3. Section 5.1.2 gives an overview of various concepts, terms, and datasets that are used to assess LCLM in the United States. Section 5.2 summarizes the major findings on LCLM from the RtC2. Section 5.3 then updates this information on trends in domestic LCLM, with separate subsections for trends to date versus likely future trends. Thus, this chapter describes spatiotemporal trends in LCLM across the contiguous United States (CONUS) but does not attribute observed trends to the RFS Program or any other factor (see Chapters 6 and 7 for information on attribution). Conclusions, uncertainties, and recommendations are then presented in section 5.4. The trends in LCLM presented here, irrespective of cause, are then used in Chapters 6 and 7 to compare the magnitude of LCLM change attributable to the RFS Program with overall changes in LCLM, and in Part 3 to compare the environmental and resource conversation effects attributable to the RFS Program with the environmental and resource conversation effects from many causes.

[1] The term LCLM is used intentionally in the RtC3 as opposed to the more common land use change (LUC) or other terms because the latter terms are often poorly defined and not used consistently in the literature. This is discussed further in section 5.1.2 and in the second triennial report to Congress on biofuels (RtC2, U.S. EPA, 2018)

5.1.1 *Overview of Drivers and Outcomes*

LCLM is a complex phenomenon that is affected by a variety of market and non-market factors. It sits at the intersection of "drivers" in Part 1 and "effects" in Part 3, as it is both a driver of environmental impacts and an environmental effect that itself is directly affected by other drivers discussed above (U.S. EPA, 2018). Farmers generally make decisions based on the expected return for growing various potential crops (Walsh et al., 2003). Farmers, however, do not consider all potential crops that could be grown each year, as they have invested time and resources in the cultivation of particular crops in particular regions. In the context of biofuels, farmers generally make decisions based on the relative margins between corn and soybean, and what they grew the previous year.[2]

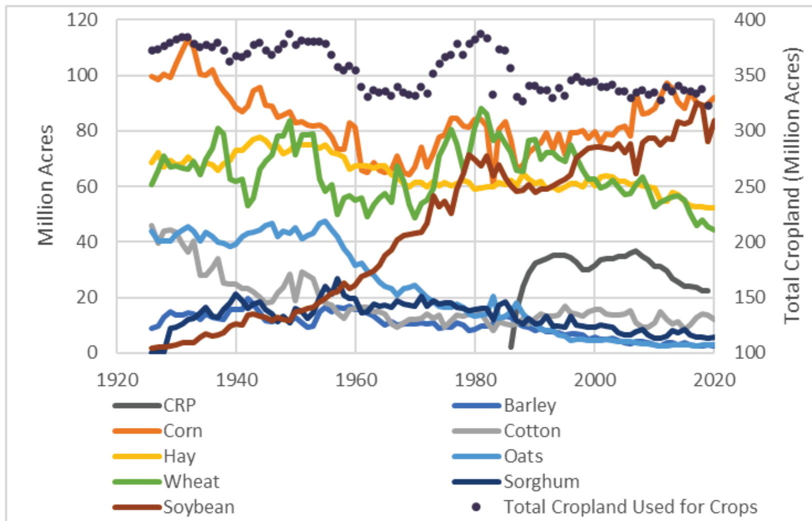


Figure 5.1. Long-term trends in major crops and other categories of agricultural LCLM from 1926 to 2020. Major crop types and CRP acreage is associated with the main y-axis (left), and total cropland acreage is tied to the secondary y-axis (right). Note the difference in scale and increments between left and right axes. Total Cropland Used for Crops only goes thru 2018. Data from USDA NASS, MLU, and CRP Statistics Databases.[3]

Before discussing the LCLM that occurred over the focal period of interest for the RtC3 (i.e., ~2005–current), it is important to understand the longer trends on LCLM in the United States (Figure 5.1). Individual crop acreages rise and fall from year to year based on a complex combination of climate and economic factors. From 1925 to 2018, total cropland acreage ranged from 330 to 390 million acres. Generally, total cropland acreage ranged from 360 to 390 million acres from the 1920s to early 1950s, declined through the 1950s, then increased from the early 1970s to early 1980s, peaking at 387 million acres in 1981. After this peak, total acreage declined 40 to 50 million acres to approximately 330 million acres followed by a 12-million-acre increase from 2011 to 2012 (328 to 340 million acres) that slowly declined through 2018 to 338 million acres (Figure 5.1). Corn decreased from the 1920s to the 1960s, and then began steadily increasing from the mid-1980s to the current day. Soybean has shown a relatively steady increase

[2] See Chapter 3 for more information on agronomic practices and Chapter 4 for more information on the economics of corn and soybean markets.
[3] Total Cropland Used for Crops and CRP acreage was available only through 2018 and 2019, respectively. Individual crops are from USDA NASS (2020c), total cropland is from the USDA MLU (2020b), and CRP is from the CRP Statistics database (USDA-NASS, 2019).

throughout the period of record, but at a greater rate post-2000, increasingly grown in rotation with corn as a feed grain, for its nutrient capture, and ability to limit pests on corn such as the western corn rootworm (Levine et al., 2002). Wheat has varied over the period of record but began a steady decline in the 1980s that began to level off after 2020. Cotton as well as other principal, small grains (i.e., oats, sorghum, barley) have generally decreased over the period of record. The rate of decline in harvested hay acreage increased in the 2000s after four decades of stable acreage, coinciding with the increased rate of corn and soybean planted acreage (Figure 5.1). The Conservation Reserve Program (CRP), which did not exist until 1985 (though similar programs were in operation starting in 1956), has varied over time, but had been experiencing a steady decrease since 2007 coinciding with reductions in national acreage constraints as specified in successive Farm Bills. More recently, the Agriculture Improvement Act of 2018 (2018 Farm Bill) increased the CRP acreage limit from 24 million acres to 24.5 million acres in 2020, 25 million acres in 2021, 25.5 million acres in 2022, and 27 million acres in 2023 (see section 5.4.1.5 for more information on CRP trends).

This longer-term context is important when interpreting the changes from 2005 to current, which are the focus of the RtC3. However, even though LCLM changes from 2005 to current may or may not be small relative to some earlier periods, that does not preclude their potential impacts on the environment as the cultivation of various crop types have varying environmental impacts. Details of these recent trends are discussed further below in section 5.3.1.

5.1.2 Definitions and Datasets

There are many terms in this scientific domain, several of which overlap, are poorly defined, or are used somewhat interchangeably. This section attempts to clarify some of these concepts. Land cover (LC) strictly describes the physical cover of the land surface (e.g., grassland) irrespective of what it is used for (e.g., pasture). Land management (LM) describes how the land is managed, which may include many factors which may be agronomic (e.g., fertilizer application, irrigation), or in some cases even geopolitical (e.g. zoning, land rights). Many studies including the RtC2, have used the term "land use and land use change" (LULUC) as a general term to describe these and other processes. The variety of definitions in this space is summarized in the RtC2 and the peer-reviewed literature (USDA, 2018; Nickerson et al., 2015). It is not the purpose of the RtC3 to resolve this ambiguity, but it is important to understand and communicate it when drawing from multiple sources that may use these terms differently.

This ambiguity in term definition, and variety of usages across studies, contributes to confusion and perceived differences among studies (Nickerson et al., 2015). Different studies can lead to different conclusions simply because the same concept is defined and thus quantified differently. Furthermore, land is not *used* like a feedstock, which is physically and chemically converted into something else like a biofuel. Land is *managed* for a particular use. In many cases, farmers are not directly involved with or in many cases even know the ultimate use of their product. Corn farmers often sell their corn to an intermediary like a grain elevator, which then sells that grain for its ultimate use as either feed, fuel, or both (see Chapter 3 for more details).

There are several additional terms common in the literature that are important to clarify (also see Glossary and NASEM (2022)). Change in LCLM is often separated into groups to describe different reasons for or manners of change, including: (1) extensification versus intensification, (2) direct versus indirect, and (3) domestic versus international. Extensification is the expansion of agricultural activities onto previously uncultivated land while intensification is increased production from the land without an increase in cropland acreage (Babcock, 2015; Lark et al., 2015). Intensification can come from a variety of changes in agronomic practices, including crop genetic improvements, double cropping, irrigation, and changes in fertilizer or other chemical inputs. Direct LCLM change in the context of biofuels is any LCLM change that occurs to produce biofuels (Gnansounou and Pandey, 2016). As mentioned earlier, because farmers usually do not know the ultimate use of a given crop, direct land use change is difficult to quantify. Indirect LCLM occurs when, following the diversion of some crop production to the new biofuel market, there is an unmet demand left in the market, which may stimulate additional LCLM change to meet that deficit (Fritsche et al., 2010). It should be noted that indirect changes in LCLM are similar to the less precise concept of indirect land use change (ILUC) used in previous reports. However, for similar reasons of using LCLM rather than land use change described above, greater efforts to better distinguish shifts in both LC and LM will be emphasized in this report. These indirect effects may occur in the immediate vicinity of biorefinery plants or not, depending on complex market interactions. Domestic LCLM in the context of this report series occurs within the United States, while international LCLM occurs outside the United States (discussed in Chapter 16).

In the context of extensification, it is important to understand how long an area was uncultivated (e.g., how long has this field been in pasture?). Outside of protected areas like national parks, there are very few areas in the U.S. lower 48 states that were *never* cultivated (i.e., natural areas never cultivated/ urbanized (Krech, 1999). And even areas that may not have been cultivated were likely managed in some ways such as prescribed burning (Krech, 1999). Nevertheless, lands and more specifically croplands that are set aside for years can accumulate carbon, become suitable habitat for many species, and thus can begin to provide many ecosystem services over time (Johnson et al., 2016). Agricultural lands also provide ecosystem services (e.g., carbon sequestration), though the magnitude and composition of these services differ from unmanaged lands. Thus, agricultural expansion onto lands that were once cultivated at some point may incur similar kinds of environmental effects as expansion onto pristine habitat, albeit at lower levels.

For the major federal efforts that quantify LCLM in the United States, agricultural land is defined and categorized in slightly different ways (see definitions in Box 5.1. Definitions from the NRI; and the Glossary). For example, in the USDA National Resource Inventory (NRI) (USDA, 2020d, 2018, 2015), cropland is divided into two categories: (1) cultivated cropland and (2) uncultivated cropland. Cultivated cropland includes what is commonly considered cropland, row crops, and other land used in rotation with row crops, while uncultivated cropland may include many other types of LCLM including permanent hayland and horticultural crops (Box 5.1. Definitions from the NRI). The USDA Census of Agriculture (Census) (USDA-NASS, 2019, 2014), on the other hand, includes five categories within total cropland: (1) harvested cropland; (2) other pasture and grazing land that could have been used for crops without additional improvements; (3) cropland on which all crops failed or were abandoned (4) cropland in cultivated summer fallow, and (5) cropland idle or used for cover crops or soil improvement but not harvested and not pastured or grazed.[4] The "other pasture and grazing land that could have been used for crops without additional improvements" is essentially potential cropland that is not used to grow crops. These differences in categories and definitions are not constrained just to the NRI and Census, which further contribute to confusion on the trends of changes in LCLM in the United States.

As detailed in the RtC2 (U.S. EPA, 2018), the best data for assessing trends in agriculture depends on the specific trends of interest. For annual information on individual crops at county scales or larger, the best dataset is from the USDA National Agricultural Statistical Survey (NASS) (USDA, 2020c). Relying on annual survey data as well as the Census of Agriculture, NASS provides objective and unbiased statistics of crop acreage, production of food and fiber, and other economic and demographic information important for tracking the status of American agriculture. For total acreage in cropland, the best data is from the USDA NRI (USDA, 2020d). The NRI is a formal statistical sample of LCLM in the United States assessed from over 800,000 point locations across the country generally every 3–5 years. The NRI is backward casted with each new version so that trends through time are internally consistent with each vintage of the report, and not conflated with methodological or sampling changes that may occur from one period to the next. This is especially important in light of the methodological changes that occurred in several key supporting data sources over the time period coinciding with the RFS Program.

[4] Categories 3–5 are often combined into "Other Cropland" in the USDA Census. Refer to the USDA Glossary for further information (https://www.ers.usda.gov/data-products/major-land-uses/glossary/).

Box 5.1. Definitions from the NRI.[5]

Below are some of the major categories in the NRI and their associated definitions. (see Glossary for more information.)

- Cropland: Two subcategories of cropland are recognized: cultivated and noncultivated.
  - Cultivated cropland: Cultivated cropland comprises land in row crops or close-grown crops and also other cultivated cropland, for example, hayland or pastureland that is in a rotation with row or close-grown crops.
  - Noncultivated cropland:
    - Hayland: Land managed for the production of forage crops that are machine harvested. The crop may be grasses, legumes, or a combination of both. Hayland also includes land in set-aside or other short-term agricultural programs.
    - Horticultural cropland: Land used for growing fruit, nut, berry, vineyard, and other bush fruit and similar crops. Nurseries and other ornamental plantings are included.
- CRP land: Only acres that have been enrolled in CRP general sign-up are included in the CRP land cover/use category. CRP continuous sign-up lands must be suitable to serve as one of a number of conservation practices, such as a wetland restoration, filter strip, riparian buffer, or field windbreak. These lands are included in the NRI under their respective land cover/use.
- Pastureland: A land cover/use category of land managed primarily for the production of introduced forage plants for livestock grazing. Pastureland cover may consist of a single species in a pure stand, a grass mixture, or a grass-legume mixture. Management usually consists of cultural treatments: fertilization, weed control, reseeding or renovation, and control of grazing. For the NRI, includes land that has a vegetative cover of grasses, legumes, and/or forbs, regardless of whether or not it is being grazed by livestock.
- Rangeland: A land cover/use category on which the climax or potential plant cover is composed principally of native grasses, grass-like plants, forbs or shrubs suitable for grazing and browsing, and introduced forage species that are managed like rangeland. This would include areas where introduced hardy and persistent grasses, such as crested wheatgrass, are planted and such practices as deferred grazing, burning, chaining, and rotational grazing are used, with little or no chemicals or fertilizer being applied.
- Forest Land: A land cover/use category that is at least 10% stocked by single-stemmed woody species of any size that will be at least 4 meters (13 feet) tall at maturity. Also included is land bearing evidence of natural regeneration of tree cover (cut over forest or abandoned farmland) and not currently developed for non-forest use. Ten percent stocked, when viewed from a vertical direction, equates to an areal canopy cover of leaves and branches of 25% or greater. The minimum area for classification as forest land is 1 acre, and the area must be at least 100 feet wide.
- Other rural land: A land cover/use category that includes farmsteads and other farm structures, field windbreaks, barren land, and marshland.
- Developed land: A combination of land cover/use categories, large urban and built-up areas, small built-up areas, and rural transportation land.
- Water areas and Federal land: Water areas are a land cover/use category comprising water bodies and streams that are permanent open water. Federal land is a land ownership category designating land that is owned by the Federal Government. It does not include, for example, trust lands administered by the Bureau of Indian Affairs or Tennessee Valley Authority (TVA) land. No data are collected for any year that land is in this ownership.

[5] These are slightly abbreviated definitions, see the source material for the full definitions.

Between 2007 and 2012, there were several changes in both the Census[6] and the U.S. Forest Service (USFS) Forest Inventory and Analysis[7] (FIA) that could affect the trends of cropland over this interval reported in several reports, including the Census, USDA Major Land Use Series (MLU) (Bigelow and Borchers, 2017) (USDA, 2020b), and the U.S. conterminous Wall-to-Wall Anthropogenic Land Use Trends database (NWALT) (Falcone, 2015). Because of the importance of 2007–2012 in this report series given the focus on 2005–current study period, estimates derived from the USDA Agricultural Census and MLU sources are less certain due to the methodological issues described above. The NRI is unaffected by these methodological changes and thus is preferred as described in the RtC2 (U.S. EPA, 2018).

5.2 Review of Major Findings from the RtC2

In the RtC2, EPA extensively reviewed the published literature on the trends to date on LCLM in the United States, including an assessment of the strengths and weaknesses of individual studies, and came to the following conclusions:

- Biofuel feedstock production is responsible for some of the observed changes in land used for agriculture, but the amount of land with increased intensity of cultivation and the portion of crop land expansion that is due to the market for biofuels cannot be quantified with precision.
- Recent research and anticipated updates to data are expected to improve the ability over the next three years to quantify the fraction of land use change attributed to biofuel feedstock production in the United States.
- Evidence from multiple sources demonstrates an increase in actively managed cropland in the United States since the passage of EISA by roughly 4–7.8 million acres, depending upon the source.

[6] There were changes in the 2007 and 2012 Census that affect trends in land use change of grassland and cropland in the Census, and thus the MLU and the NWALT, which partially rely on the Census. From the MLU: "Cropland pasture estimates, one of two nonpermanent grazing uses tracked in MLU, declined nearly 80% in the past 10 years (2002-12) after exhibiting relative stability for more than 50 years. This decline is largely attributable to methodological changes [*i.e. change in wording and location of the question in the Census, emphasis added*] in the collection of cropland pasture data in the [2007 and 2012] Census of Agriculture, the data source of the cropland pasture category…While there is no way to definitively determine the extent of the effects of changes in the placement and wording of the cropland pasture question, it seems likely, given the relatively stable cropland pasture acreage trend from 1949 to 2002, that the changes contributed to the large decrease between 2002 and 2012" (Bigelow and Borchers, 2017).

[7] The MLU partly attributed the increase in grassland between 2007 and 2012 to a methodological change in the USFS FIA. In the FIA, large areas of chaparral and shrubland, which were originally classified as forests because of the presence of tree cover, were reclassified as woodland or grasslands because the relatively sparse tree cover meant the lands were more likely used as grassland and rangeland than for timber production (Bigelow and Borchers, 2017). This partly contributed to an increase in grassland pasture in the MLU.

- Much of this increase is likely occurring in the western and northern edges of the corn belt with reductions of pasture and grassland, but also through infilling of already agricultural areas.
- Thus, intensification likely dominates in already agricultural areas and extensification dominates in less agricultural areas.

The RtC2 focused on five major national efforts: (1) the USDA 2012 National Resources Inventory (2015), (2) the USDA 2012 Census of Agriculture (2014), (3) the USDA's Major Uses of Land in the United States, 2012 (USDA, 2020b; Bigelow and Borchers, 2017), (4) the U.S. Geological Survey (USGS) U.S. Conterminous Wall-to-Wall Anthropogenic Land Use Trends (Falcone, 2015), 1974–2012, and (5) a pair of studies from the University of Wisconsin and the University of Minnesota (Wright et al., 2017; Lark et al., 2015). These efforts vary in their approaches and definitions, making direct comparisons difficult. In the RtC2, EPA harmonized the many definitions among studies to the degree possible in order to focus on changes in land actively used to grow crops.[8]

Once this harmonization was completed, all studies showed a relatively consistent trend of an increase in actively managed cropland in the United States up to 2012 (Table 5.1). This trend is different from that of total agricultural land, which includes land areas that are not used to grow crops but could be used to grow crops (e.g., pasture, fallow fields). Total agricultural land had been steadily decreasing in the United States since the 1970s, mostly as a result of urbanization, increasing crop yields, and agricultural abandonment (Falcone, 2015). The changes in LCLM that is most relevant to the EISA Section 204 Report Series are those pertaining to any lands that went into production either directly to support the production of feedstocks used for biofuels (i.e., direct changes in LCLM), or indirectly because of cascading effects from the diversion of existing crops to this new market (i.e., indirect changes in LCLM). It is very difficult to isolate the subset of LCLM attributable to the RFS Program using these reports (but see Part 2: Chapters 6 and 7 for an assessment of attribution). Thus, as noted above, these reports are more useful for describing the broader trends in agricultural land, some of which might be attributable to biofuels and/or the RFS Program.

Thus, the RtC2 concluded that the five major national-scale studies available suggested that actively managed cropland had increased in total acreage in the United States by 4–7.8 million acres between 2007–2008 and 2012. For context, 4.7 million acres is approximately the land area of the state of New Jersey. This had been primarily a conversion of grassland or pasture to corn, soybeans, and wheat, along the extensive agricultural margin, and through infilling of previously uncultivated areas, prior to 2007 or 2008, in the central Midwest (U.S. EPA, 2018).

[8] See section 2.4 in the RtC2 for a full discussion of this harmonization.

Table 5.1. Comparison of major national studies on land use change from the RtC2. Shown are the source publication, the comparable term(s) and definition(s), years assessed, and the change in acreage in millions of acres (and % from study-specific reference, copied from the RtC2, (U.S. EPA, 2018), Chapter 2, Table 4).

Study	Comparable term(s)	Definition(s)	Years reported	Change million ac (%)
USDA NRI (2015)	Cultivated cropland	Cultivated cropland comprises land in row crops or close-grown crops and also other cultivated cropland, for example, hayland or pastureland that is in a rotation with row or close-grown crops.	2007–2012	+4.3 (1.4%)
USDA Census (2014)	Harvested cropland + failed/ abandoned + summer fallow	Harvested cropland—This category includes land from which crops were harvested and hay was cut, land used to grow short-rotation woody crops, Christmas trees, and land in orchards, groves, vineyards, berries, nurseries, and greenhouses. No separate definition for failed/abandoned, or summer fallow cropland.	2007–2012	+7.8 (2.4%) [a]
USDA MLU (2020b)	Cropland used for crops	Three of the cropland acreage components—cropland harvested, crop failure, and cultivated summer fallow—are collectively termed cropland used for crops, or the land used as an input to crop production.	2007–2012	+5 (1.5%)
Falcone (2015)	Crops	Areas used for the production of crops, such as corn, soybeans, wheat, vegetables, or cotton, as well as perennial woody crops such as orchards and vineyards. Includes cultivated crops, row crops, small grains, and fallow fields.	2002–2012	+3.9 (1.2%)
Lark et al. (2015)	Net cropland	Net cropland increases (gross expansion - gross abandonment) of lands in the lower 48 states that have no evidence of cultivation since 1992.	2008–2012	+3 (1%)[b]
Wright et al. (2017)	Net cropland	Net cropland increases (gross expansion - gross abandonment) of lands within 100 miles of a biorefinery that have no evidence of cultivation since 1992.	2008–2012	+4.2 (NA)[c]

[a] Harvested cropland, failed/abandoned cropland, and summer fallow cropland changed by +5.4, +4.0, and -1.5 million acres, respectively between 2007 and 2012 according to the Census.
[b] Estimates from Lark et al. (2015) are likely to be lower because they focus on a subset of lands that had no evidence of cultivation for 20 years or more.
[c] Estimates from Wright et al. (2017) are likely to be lower because they focus on a subset of lands that had no evidence of cultivation for 20 years or more as in Lark et al. (2015), and on lands within 100 miles of a biorefinery. The percent increase from Wright et al. (2017) could not be calculated here because the 2008 baseline acreage within 100 miles of a biorefinery was not reported.

5.3 Domestic Trends in Land Cover and Land Management

The following subsections highlight the trends to date and future trends for LCLM domestically building from the RtC2. The domestic trends to date are primarily based on insights from the NRI as recommended in the RtC2, but other ancillary sources of information from other federal studies, federal databases, and peer-reviewed publications identified in the literature review for the RtC3 are reported to compliment insights from the NRI and to highlight uncertainties (see Appendix A). Future trends domestically draw mainly from the USDA Long Term Agricultural Projections (2020e) and any short-term projections from the literature review.[9]

[9] The Energy Information Administration reports discussed in Chapter 2 for future biofuel production does not include estimates of croplands so is not the preferred source for this chapter on LCLM (see Chapter 2, section 2.3.2).

5.3.1 *Trends to Date Domestically*

5.3.1.1 *Major Land Classes from Multiple Federal Sources*

The 2015 and 2017 NRIs were released in September of 2018 and 2020, respectively (USDA, 2020d, 2018) and both reports demonstrate that the trends since 2007 in cropland acreage reported in the RtC2 have continued. After a 25-year or longer decline of actively managed cropland in the United States from 1982 to 2007 (71 million acres, Figure 5.1 and 5.2), there has been an increase by about 10 million acres in cultivated cropland and total cropland that began in roughly 2007 (Figure 5.2).[10] On net, cultivated crop acreage is still 60 million acres less than levels observed in 1982 (Figure 5.2). Cultivated cropland increased by 4.5 million acres between 2007 and 2012, by an additional 4.5 million acres between 2012 and 2015, and an additional 0.9 million acres between 2015 and 2017, or an average of approximately 1 million acres per year over the 8–10-year interval (~9–10 million acres). Noncultivated cropland increased from 1982 to 2002, remained stable from 2002 to 2007, and has been relatively steady since 2012 at roughly 52 million acres.

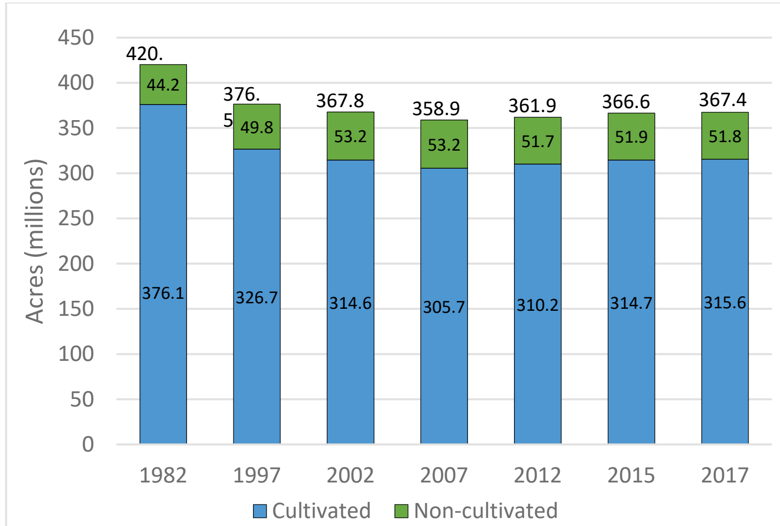


Figure 5.2. Trends in cropland from 1982 to 2017 from the 2017 NRI (in millions of acres). Cropland in the NRI includes cultivated and noncultivated cropland. The 2015 values are from the 2015 NRI since the 2015 estimate was not reported in the 2017 NRI (USDA, 2020d).

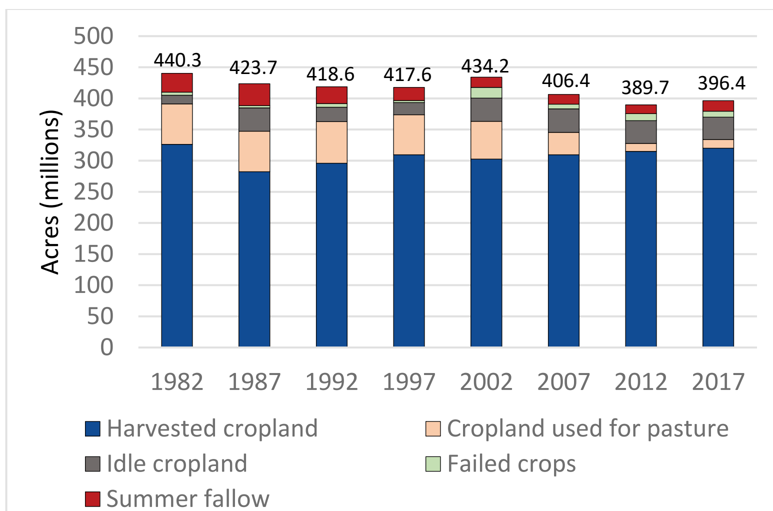


Figure 5.3. Changes in total cropland and its five components from 1982 to 2017 from the Census.

[10] Note that estimates for prior intervals (e.g., 2007–2012) are updated with each new report as methods improve, so estimates on the same interval may not be identical between different NRI Reports.

For context, the 2017 Census of Agriculture reported for the first time in this series for over a decade, an increase in total and harvested cropland between 2012 and 2017 (USDA-NASS, 2019). Total cropland was estimated to have increased by 6.7 million acres, while harvested cropland increased by 5 million acres (Figure 5.3). This development followed a longer-term decrease in total cropland between 2002 to 2012 yet an increase in harvested cropland over the same period. Both of these changes appeared to come from large decreases in land that could be used as crops but was not.[11] Thus, harvested cropland is making up a larger proportion of total cropland over time (Figure 5.3). Annual estimates of total cropland used for crops, drawn from the USDA ERS Major Land Use Database (MLU), were largely stable for the entirety of 2002 to 2018 (USDA, 2020b). But, like the Census of Agriculture, the timeseries displays an initial decline then increase in the latter half of the time period after 2011 (Figure 5.1).

Thus, although the potential agricultural land base may be declining over time (Figure 5.1–5.3), the area used to actually grow crops and/or hay in rotation has been increasing since 2007 according to the NRI (i.e., cultivated cropland, blue bars, Figure 5.2), since 2012 according to the Census (i.e., primarily harvested cropland, blue bars, Figure 5.3), and since 2011 according to the MLU (i.e., total cropland used for crops, orange line, Figure 5.1). Differences in the actual year of increase are likely due to definitional differences among the three reports, different dates and methods of sampling, measurement error, and aforementioned changes in methodology over the period.

5.3.1.2 Detailed Trends from the NRI

5.3.1.2.1 National Trends

The published 2015 and 2017 NRIs report large categories of LCLM in the final report (e.g., total cropland and Figure 5.2), but the dataset can be parsed out in finer detail. USDA separated the total cropland category in the NRI into four requested subcategories for the EPA in support of the RtC3: corn, soybean, other cultivated cropland, and noncultivated cropland. For evaluating transitions among these categories, corn and soy were combined into a single category because these crops are often grown in rotation and both are used for biofuels (Table 5.2). Changes in annual rotation patterns between these two crops may be important to examine but are not appropriate to examine with the NRI, which only comes out every 3–5 years (see 5.3.1.3 for additional information on crop rotations).

Many of the LCLM classes may be managed similarly (e.g., pasture and noncultivated cropland), and thus transitions between them may not have large environmental implications. One of the primary drivers of potential environmental effects is whether the lands are managed annually or perennially.

[11] Formally, this category is called "Other pasture and grazing land that could have been used for crops without additional improvement." Between 2007 and 2012 there was reported a 23 million acre decrease in this land category, which largely drove the reduction in total cropland (USDA 2014). Hereafter this land is called "cropland used for pasture" for brevity.

Annually managed systems include annual cropland (e.g., all primary crops) and may be tilled, fertilized, or otherwise managed, on an annual basis. Land managed as a perennial system (e.g., pasture, CRP, noncultivated cropland) may receive some annual amendments like fertilized hayland, but they are often at lower rates compared with row crops. Hay crops, like alfalfa and other forages, may also be irrigated and cut multiple times in a growing season; thus, it should not be construed that perennial systems are passively managed. However, given that perennially managed systems are not tilled annually and generally have continuous vegetation coverage, they typically facilitate accumulations of root biomass and soil carbon more so than annually managed systems (especially those that are conventionally tilled). Changes in perennial cover may be an indicator of potential environmental effects. Thus, a perennial agricultural LCLM class is sometimes reported rather than reporting all three individual perennially managed LCLM types for succinctness.

Leveraging estimates from the 2015 and 2017 NRI reports, acreage differences for the periods of 2002 to 2015 and 2002 to 2017 highlight the potential changes in acreage. Rather than solely relying on a snapshot in the difference between time periods, using both of these recent reports provide further confidence in the directionality of acreage changes across space and time. The first period covers the interval from before the RFS Program and the increase in biofuels in the United States (Chapter 1, Figure 1.3) until after ethanol production reached the E10 blend wall (Chapter 1, Figure 1.4). The second period extends this information to the most recent data in the NRI.[12]

Table 5.2 shows that between 2002 to 2015 or 2002 to 2017 acreage devoted to corn and soybean increased by roughly 21–32 million acres, about 13–20%, and other rural and developed land also increased, by roughly 14 million acres (~10% increase). Consistent with other studies (Johnston, 2014; Wallander et al., 2011), large decreases occurred for other cultivated cropland (-21 to -31 million acres) and CRP (-13.5 to -15.5 million acres). Other land classes changed much less by comparison. The decrease in CRP is likely due primarily from decreases in acreage caps from updates to the Farm Bill (see section 5.3.1.5). Thus, although the total acreage of the CRP Program is not directly linked to biofuels or the RFS Program, what is grown on those lands after leaving the CRP Program is due to many market and non-market factors including the RFS Program. Recent remote-sensing efforts suggest that approximately 40% of expired CRP land in the Midwest from 2010–2013 went into cultivated cropland (Morefield et al., 2016). From 2013 to 2016, almost 80% of non-reenrolled CRP land was converted to some type of crop production across the United States (Bigelow et al., 2020). Thus, a large fraction of expired CRP is likely cultivated for crops after leaving the CRP Program. Crop-specific acreage data from NASS is also

[12] A third period is also shown in Table 5.2 (2007–2017), which approximates the period of the RFS2 that was created with EISA in 2007. This period is used for other purposes later to overlap with the period of national CDL datasets (e.g., 2008–2016 in Lark et al. (2020).

consistent with the NRI (Figure 5.1). Corn and soybean acreage has consistently increased over the period of record (~18 and 27 million acres from 2002 to 2015 and 2002 to 2017, respectively, Figure 5.1) with corresponding declines in small grains, cotton, and hay (~25–36 million acres from 2002 to 2015 and 2017, respectively). NASS acreage data suggests that these trends for soybean and corn as well as other crop types are maintained through 2020 (see section 5.3.1.3).

It is tempting to assume that the increases in corn/soy came entirely from other cultivated cropland given the close correspondence of the increase in the former with the decrease in the latter. Table 5.2, however, only shows the net change through time after all the individual inputs and outputs from other groups are accounted for. To examine which lands contributed to the increases and decreases, USDA also provided "transition matrices" that explicitly track which lands are moving from one group to another group. Following publication of the 2017 NRI, sequential transition periods of 2002–2007, 2007–2012, and 2012–2017 were provided by USDA to track gross and net changes in land cover and land management. The 2002 NRI is leveraged because 2007 was coincident with the RFS Program and a large increase in corn acreage (see Figure 5.1 and section 5.3.1.3). Examining these three intervals (i.e., 2002–2007, 2007–2012, and 2012–2017) approximates comparisons for a period before the RFS Program (i.e., 2002–2005), with the rapid expansion of biofuel production (i.e., 2004–2012), and a period after the E10 blend wall was reached and corn ethanol production was comparatively stable (i.e., 2013–2020).[13]

At the national level, there was an increase in corn/soy by almost 5.2, 15.3, and 11.2 million acres for the transition periods of 2002–2007, 2007–2012, and 2012–2017 (Figure 5.4a, Net Total), respectively for an overall increase of 31.7 million acres from 2002 to 2017 (Figure 5.4 and Table 5.2). Most of the 31.7 million-acre increase in corn/soy acreage came from other cultivated cropland (~18 million acres from 2002–2017, 56%), followed by noncultivated croplands (6.7 million acres from 2002–2017, 20%), pastureland (4.2 million acres from 2002–2017, 13%), and CRP (3.6 million acres from 2002–2017, 11%). Since CRP, uncultivated croplands, and pastureland are generally managed as perennial cover, in total 45% of the conversion to corn/soy came from lands formerly in perennial cover, while the rest came from lands already managed annually for other cultivated crops. As mentioned above, conversion from perennial to annual cover is expected to have larger negative environmental effects than conversion from different types of annual cover.

[13] For a more detailed discussion of the timing if these events, the E10 blend wall, and other factors, see Chapter 6.

Table 5.2. Trends in major land classes from the 2017 NRI (in millions of acres). Note the 2015 values are from the 2015 NRI because this year was not reported in the 2017 NRI.

	Land Class	2002	2007	2012	2015	2017	Change (2015–2002)	Change (2017–2002)	Change (2017–2007)
Cropland	Corn and Soybeans	156.8	162	177.3	178	188.5	21.2	31.7	26.5
Cropland	Other Cultivated Cropland	157.8	143.6	132.8	136.7	127.1	-21.1	-30.7	-16.5
Cropland	Noncultivated Cropland	53.2	53.2	51.2	51.9	51.8	-1.3	-1.4	-1.4
CRP (general signup)		31.4	32.5	23.7	17.9	15.9	-13.5	-15.5	-16.6
Pastureland		120	120.9	122.9	121.7	121.6	1.7	1.6	0.7
Rangeland		407.5	406.6	405.1	404.4	403.9	-3.1	-3.6	-2.7
Forest Land		415.8	415.7	416.4	415.9	417.5	0.1	1.7	1.8
Other Rural and Developed Land		146.1	153.2	156.9	160.3	160.1	14.2	14	6.9
Water Areas & Federal Land		455.4	456.4	457.2	457.3	457.6	1.9	2.2	1.2

Nationwide, other cultivated cropland decreased by 14.2, 10.8, and 5.7 million acres for the transition periods of 2002–2007, 2007–2012, and 2012–2017, respectively (Figure 5.4b, Net Total), or by approximately 30% from 2002 to 2017 (Table 5.2). These results are consistent with the continued increase in corn and soy acreage, and decrease in small grains, cotton, and hay in the NASS surveys (Figure 5.1). Most of the decrease from 2002 to 2017 came from conversion to corn/soy (-18 million acres, 58% of the net decline), noncultivated cropland (-7.2 million acres, 23% of the net decline), and pasture (-6.2 million acres, 20% of the net decline), offsetting increases from CRP being cultivated as other cropland (+2.0 million acres, offsetting the net decrease by 7%). Contributions to and from other land classes were small by comparison and largely offset (Figure 5.4b).

In contrast to the large declines in other cultivated cropland acreage, noncultivated cropland decreased by only 1.4 million acres from 2002 to 2017 (Figure 5.4c, Net Total), or by only 1.4% (Table 5.2). Most of the decrease came from conversion to corn/soybean (-6.6 million acres) and pasture (-2.2 million acres) offsetting increases in noncultivated from other cultivated cropland (7.1 million acres) and CRP (1.5 million acres). Contributions to and from other land classes were small by comparison.

For CRP, the 15.5 million-acre or 49% decline between 2002 and 2017 led to large conversions to all classes, including pastureland (6 million acres), corn/soy (3.7 million acres), other cultivated cropland (2.0 million acres), and noncultivated cropland (1.6 million acres). Once again, net conversions to other land classes were small by comparison (Figure 5.4d). Nationally, 36% of CRP went in cultivated cropland production which is consistent with a 12-state analysis in the Midwest that highlighted about 30% of expiring CRP land went into five principle crops (corn, soy, winter and spring wheat, and sorghum (Morefield et al., 2016).

Figure 5.4. Net change in major land classes from 2002–2007, 2007–2012, 2012–2017, and 2002–2017 from the NRI (in thousands of acres). Changes are shown from corn/soy (a), other cultivated cropland (b), noncultivated cropland (c) and CRP (d). Note "Perennial Ag Land" is the summed acreage of CRP, pastureland, and noncultivated cropland, and net total is summed change in the major land class across all periods. Positive numbers indicate a net increase and negative numbers indicate a net decrease in that class overall. Black dashed line indicates perennial agricultural land and net total are combinations of individual categories to the left (USDA, 2020d).

In Figure 5.4, "all other" LCLM classes include forestland, developed urban and rural land, and water area/federal land, and they all increased from 2002 to 2017 (Table 5.2). On a net basis and amounting to roughly 5 million acres from 2002 to 2017, small amounts of acreage of corn/soy (0.8 million acres), CRP (2.2 million acres), other cultivated cropland (1.4 million acres), and noncultivated cropland (1.2 million acres) were converted to these other LCLM classes from 2002 to 2017 (Figure 5.4).

In summary at a national level there were many changes in LCLM from 2002 to 2017. First, on an acreage basis, crop production in the United States is becoming less diverse despite the observation that cultivated acreage (including corn/soy and other cultivated cropland) has begun to increase since 2007—reversing a general long-term decline (Figures 5.1–5.3). This observation is reinforced by the fact that acreage in 2012, 2015, and 2017 is increasing year after year with each NRI survey (Figure 5.2 and Table 5.2). This increased cultivation was driven by a large increase in corn/soybean acreage, roughly 56% of which was from other cropland, 33% from pasture or noncultivated cropland, and 11% from CRP (Figure 5.4). Because other cropland and especially pasture and noncultivated cropland are less intensively managed than corn, these shifts generally represent a form of agricultural intensification due to increased fertilizer and pesticide use (see Chapter 3). This is supported by recent publications illustrating large relative increases in nitrogen and phosphorus fertilizer use nationwide between 2002 and 2012 (Sabo et al., 2021; Sabo et al., 2019); and these increases were primarily concentrated in the cereal crop producing regions of the Midwest.

5.3.1.2.2 Regional Trends

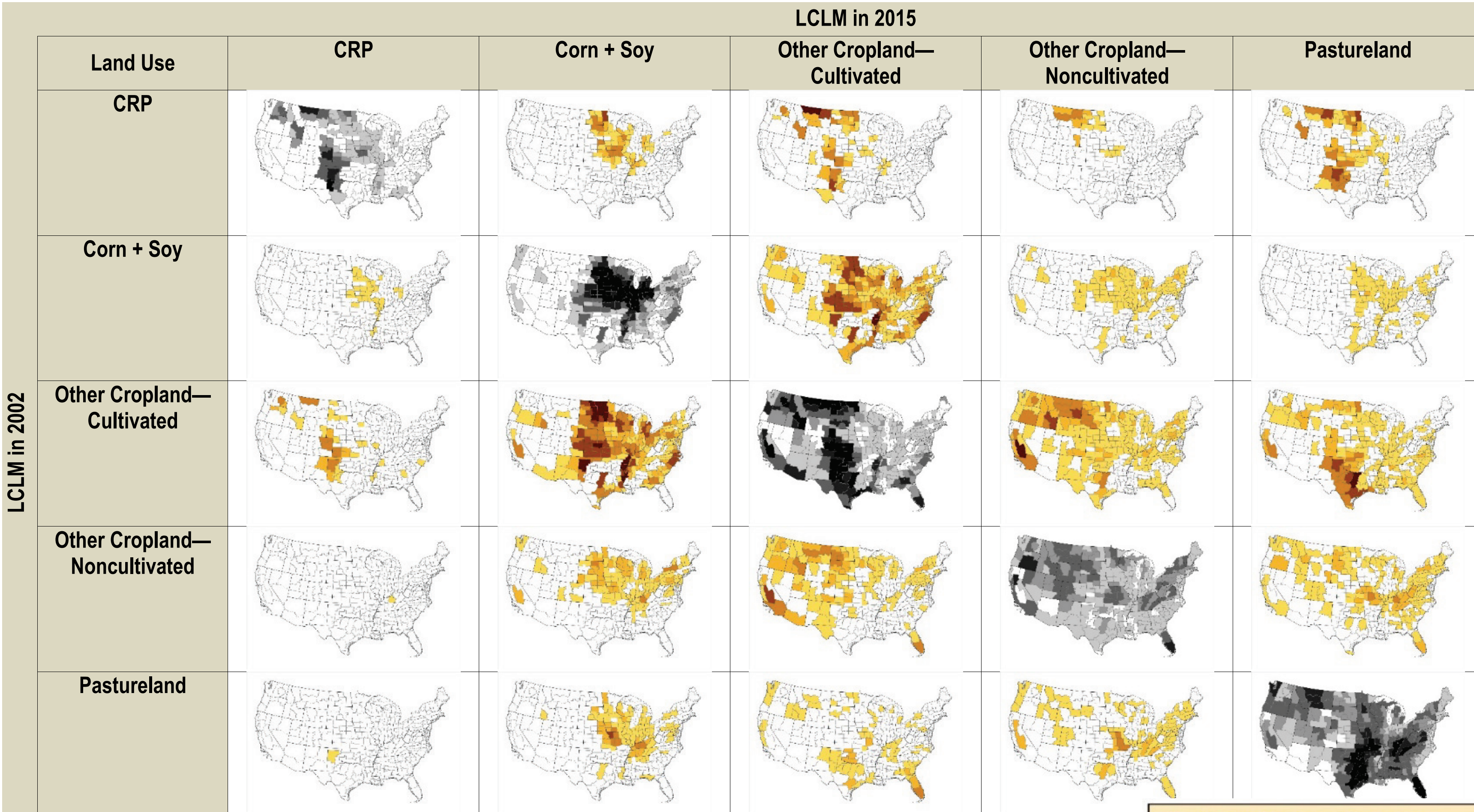
Evaluation of changes in LCLM only at the national level may fail to capture important state and region-specific shifts relevant for local-level environmental impacts. To illustrate the general changes in LCLM at the regional level (i.e., this chapter uses Cropland Reporting Districts, CRDs [14]), gross changes in LCLM from 2002 to 2015 (Figures 5.5 and 5.6) are illustrated. Here this chapter focuses only on the classes that contributed most to increases or decreases in corn and soybean: other cultivated cropland, noncultivated cropland, CRP, and pasture, which accounted for more than 95% of the changes. Contributions to these biofuel feedstocks from other classes were small by comparison. As shown in the different legends for the diagonal cells (gray scale for acreage remaining) versus off-diagonal cells (brown scale change in acreage), much more land generally stays in a given class than moves between classes (Figure 5.5). Interestingly, almost as much corn/soy transitioned to other crops, on a gross basis,

[14] The NRI can be aggregated or disaggregated to a variety of scales (e.g., county, state). At smaller scales, there is more uncertainty in the estimate, at larger scales, there is less information on where the changes occurred (USDA, 2020d). The CRD level, which is roughly the size of a few counties, was chosen as a balance of somewhat fine-grain information, and lower uncertainty in the estimates at a larger scale.

between 2002 and 2015 (22.1 million acres, Figure 5.5) as the reverse (30.6 million acres), illustrating the dynamic nature of crop planting for farmers. Most of the losses of other cultivated cropland were to corn/soy throughout the Midwest and to noncultivated cropland and pasture in the West and Texas, respectively. Large amounts of pastureland, often ranging from 50,000 to 500,000 acres per CRD, transitioned to corn/soy in Missouri and along the western fringe of the corn belt in the Dakotas, Nebraska, and Kansas. CRP went mostly to pasture and other cultivated cropland in northern Montana, North Dakota, and Minnesota as well as the south in Texas and New Mexico, but to corn/soy in the central Midwest including southern Minnesota and the Dakotas, and to other crops in the western Midwest, following dominant cropping patterns (Figure 5.5).

On a percentage basis (Figure 5.6), many of these individual transitions were small relative to the total size of the CRD. Only transitions from other cultivated cropland to corn/soy exceeded 10% in any CRD (Figure 5.6), and these occurred in the Dakotas, Nebraska, and Lower Mississippi River Basin (oftentimes >500,000 acres, Figure 5.5). However, gross changes to corn/soy were also consistently more than 5% or 200 to 500 thousand acres in CRDs of eastern Kansas (Figures 5.5 and 5.6). Consistent with these changes, though smaller in a relative sense, 5–100 thousand acres of noncultivated cropland were cultivated as corn and soy in other CRDs along this western fringe of the cereal crop producing regions of the midwestern United States (an additional 1–5% to the CRDs). Likewise, pasture also shifted to corn/soy throughout the Midwest and the western fringe of the corn belt (5–200 thousand acres), with a larger shift in northwestern Missouri (50–200 thousand acres). These acreage changes accounted for an additional 1–5% increase to corn/soy in many of the CRDs (Figure 5.6). Compared to other cultivated cropland and corn/soy, little land was transitioned to noncultivated cropland or pasture in the Midwest, especially the Missouri River Basin where these LCLM shifts are further explored in Chapters 9 and 10. Overall the western expansion of corn/soy at the expense of other cultivated cropland and pasture was highlighted in RtC2 from 2007 to 2012, and this trend continues.

The gross change maps highlight a snapshot in LCLM between two years (e.g., 2002–2015, Figures 5.5 and 5.6), so it is unclear if the perceived western expansion of corn/soy is simply an artifact of the choice of years. Thus, NRI-estimated net changes in corn/soy were also estimated, as well as shifts in perennial agricultural land, for multiple 5-year transition periods from 1992 to 2017 to establish if this expansion is consistent through time (Figure 5.7). Consistent with the 2002-2015 gross change maps (Figure 5.5), corn/soy acreage by and large had the greatest increases in North Dakota, South Dakota, Nebraska, and Kansas with smaller, corresponding increases in Iowa, Minnesota, and Wisconsin (Figure 5.7). Both Missouri and Arkansas had smaller year-to-year changes in corn/soy acreage, but these states saw consistent increases after 2007 (Figure 5.7). The large increases in corn acreage in the Dakotas, Kansas, and Nebraska since 1997 seemingly offset declines in corn cultivation in the eastern United



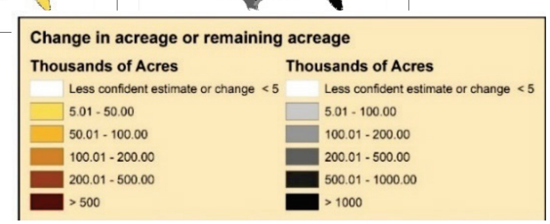
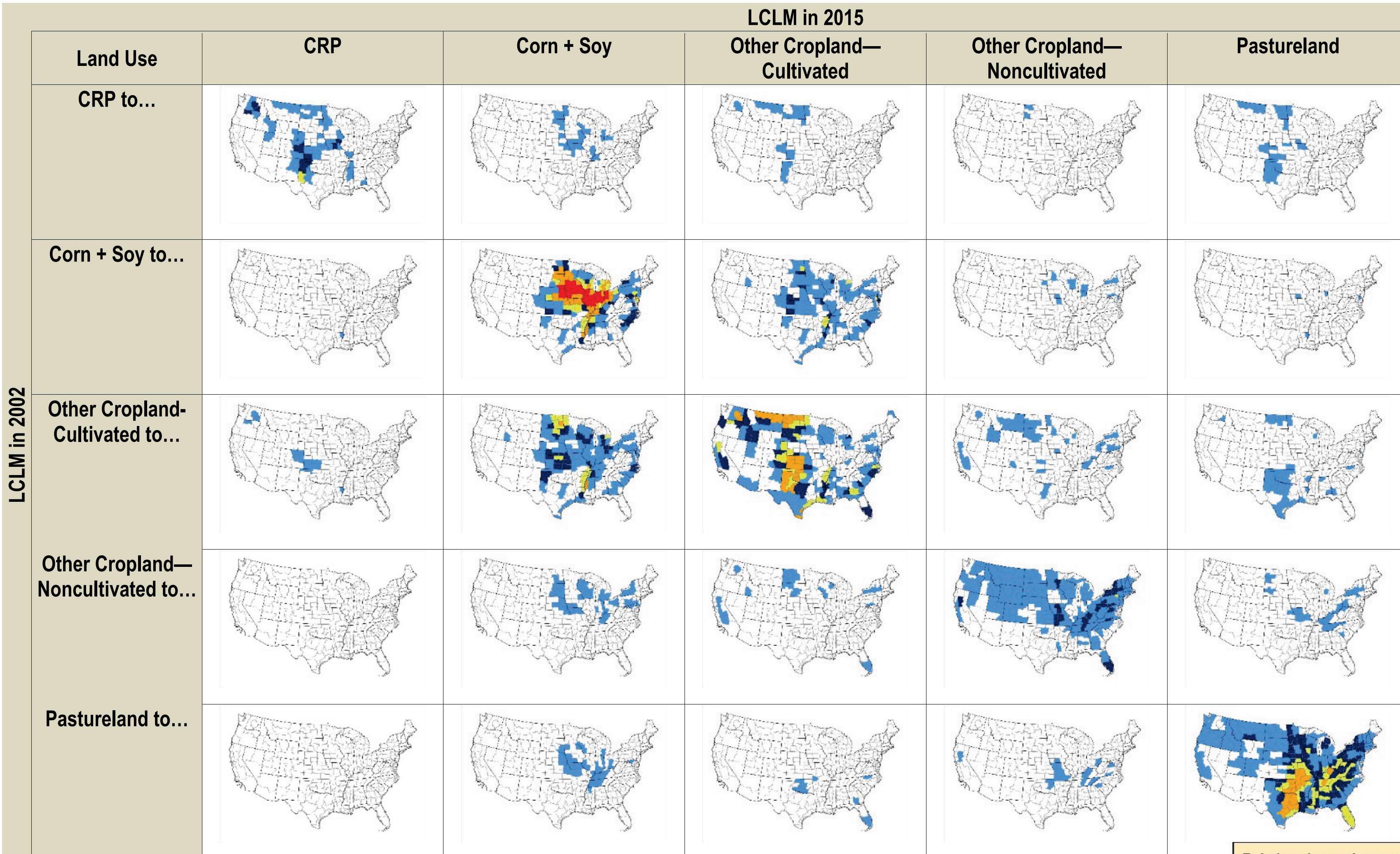


Figure 5.5. Gross estimates of gross land use change between 2002 (rows) and 2015 (columns) at the CRD level among five major land use classes according to the NRI. Gray scale highlights acreage remaining a given land use from 2002 to 2015, whereas brown scale highlights the magnitude of changes (all values are positive). Only changes that were relatively confident are displayed.[15]

[15] The NRI is a statistically based sample from individual re-measured points which means that there is an estimate and a standard error in the estimate. As the spatial scale of inquiry increases (e.g., county to state), there are more NRI points included in each estimate, which often leads to lower standard errors associated with a given estimate. Here the scale of a Crop Reporting District (CRD, approximately 4-5 counties) is mapped because this retains some of the spatial granularity of information, while reducing the error commonly seen when mapped at smaller scales such as at the county. When the estimate of error is larger than the estimate, the 95th confidence interval includes zero (i.e., no change). CRDs where the 95% confidence internal includes zero are omitted.



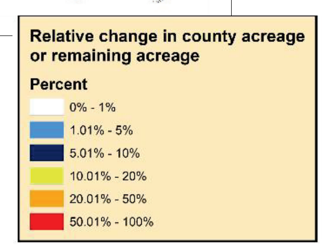


Figure 5.6. Gross estimates of relative land use change as a percentage of the CRD between 2002 (rows) and 2015 among five major land use classes according to the NRI. Only changes that were relatively confident are displayed **(see footnote for Figure 5.5).**

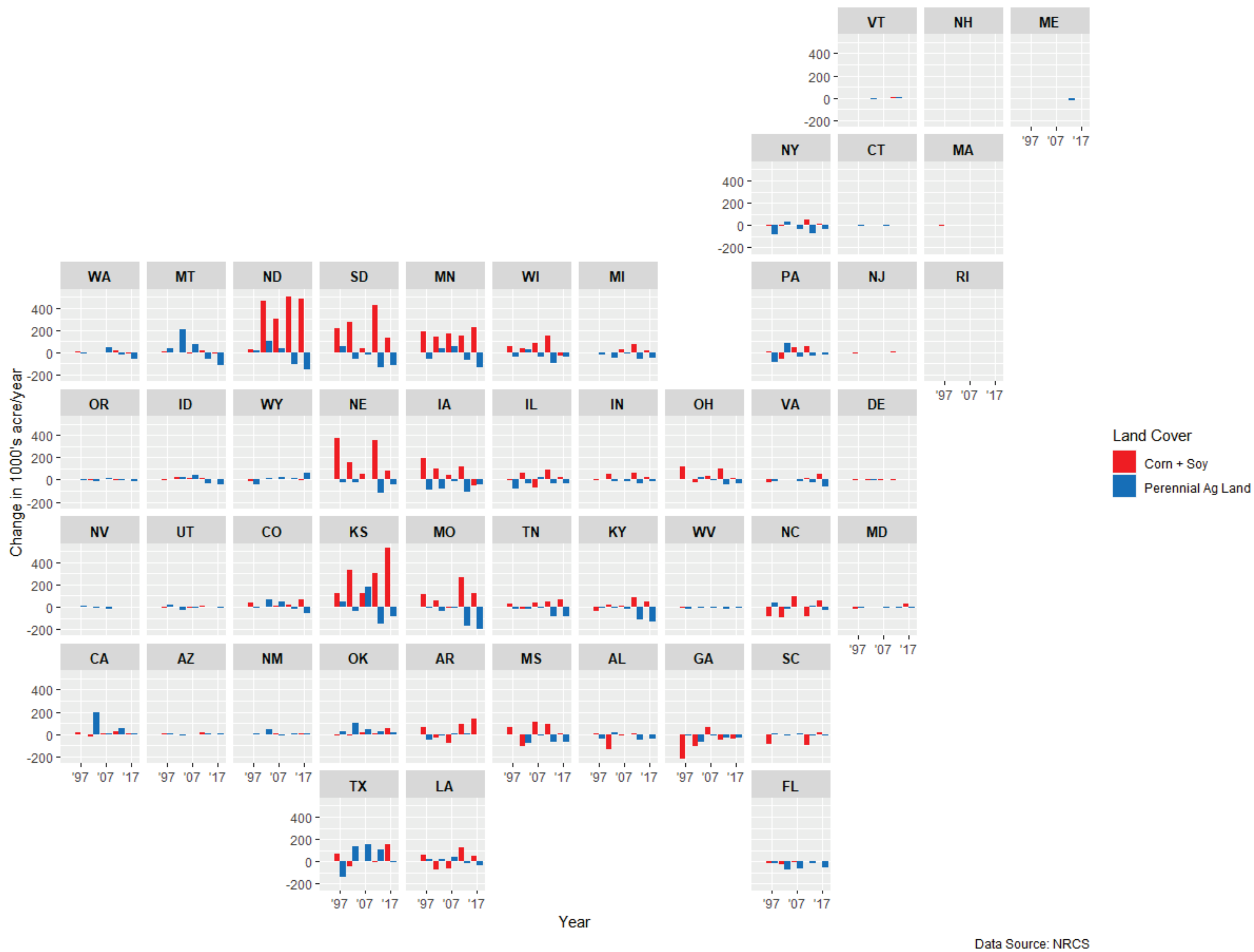


Figure 5.7. NRI estimated net change in perennial agricultural land (i.e., sum of CRP, pastureland, and noncultivated cropland) and corn+soy acreage by state for five 5-year transition periods from the NRI beginning from 1992 to 2017 (1992-1997, 1997-2002, 2002-2007, 2007-2012, 2012-2017). The first bar on the left within a state graphic represents the difference in acreage between 1997 and 1992, so a positive number indicates an increase in acreage (USDA, 2020d).

States, thus partly explaining the national stability of corn/soy acreage in the 1990s seen in NASS (Figure 5.7, Figure 5.1). Outside of further confirming the western expansion of corn/soy as revealed by Figure 5.5 and 5.6, this analysis clearly highlights that increases in corn acreage were occurring in the western fringe of traditional cereal crop producing regions prior to 2002 and certainly before the RFS1 was enacted legislatively in 2005 and the RFS2 in 2007. This observation emphasizes the importance of complementing national level trends (Figures 5.1–5.3, Table 5.2) with regional trends as the national level timeseries provided no indication of these regionally offsetting trends in corn/soy acreage prior to 2007. It should be noted, however, that the net loss of perennially managed agricultural land in the midwestern states cannot alone account for the increase in corn/soy acreage (though the balance in Missouri is close, Figure 5.7). Regional studies based on independent datasets (i.e., the National Agricultural Imagery Program, NAIP) have also reported increases in cropland and decreases in grassland in both South Dakota and Nebraska (Joshi et al., 2019; Reitsma et al., 2015). Based on the nationwide analysis of net transitions and the gross transition estimates at the CRD level (Table 5.2, Figures 5.4–5.6), other cultivated cropland was where the slight majority of acreage being transitioned to corn/soy came from.

5.3.1.3 Individual Crops

The best continuous, annual data for trends in individual crop acreages is the USDA NASS NASS June Area Survey (Figure 5.1). These data are collected at the end of every season and give a snapshot of crop acreages, production, and other information at the county scale in the United States. As reported in the RtC2, the NASS data demonstrate that corn planting area nationally was relatively flat from 2000 to 2006 at roughly 80 million acres (Figure 5.8). This was followed by a large jump in 2007 to 93.5 million acres, which stabilized after that to roughly 90 million acres. Thus, there was roughly a 10 million-acre increase in corn acreage

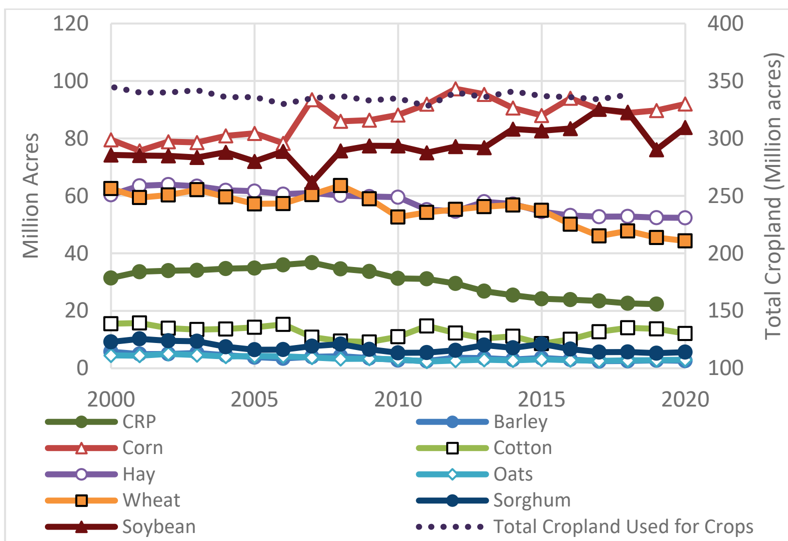


Figure 5.8. Changes in major cultivated crop types from 2000 to 2020 without total cropland (same timeseries from Figure 5.1, but focused on 2000–2020). Major crop types and CRP acreage are associated with the main y-axis (left), and total cropland acreage is tied to the secondary y-axis (right). Note the difference in scale and increments between major crop types and CRP acreage and total cropland acreage. The subset of major crop type timeseries does not sum to total cropland used for crops since the latter estimates comes from separate data source and includes other crops.

planted between the periods of 2000–2006 and 2007–2020 (Figure 5.8). This obviously coincides with the RFS1 (2006–2008) and EISA (2007), but many other factors were also occurring in this period (see Chapter 6), including the phaseout of methyl tert-butyl ether (MTBE) as an oxygenate, substantial variation in federal direct farm payments (USDA Economic Research Service, 2022), shifts in refining practices, and other factors like shifts in precipitation patterns, changing ownership structure, and improved technology (Reitsma et al., 2015). These attributional issues for corn are discussed in Chapter 6. Furthermore, as discussed in Chapter 3 the increased planting of corn is coincident with the increased adoption of no-tillage and conservation tillage practices throughout the Mississippi River Basin. In the late 1980s, the proportion of no-tilled acres made up only 7% of cropland acres, but in 2017 that value has increased to 46% with greater cultivation of herbicide-resistant corn and other crop types (Sabo et al., 2021; USDA-NASS, 2014; Baker, 2011).

Soybeans showed a different pattern over the same period. After a period that was relatively flat between 2000 and 2006 at just under 80 million acres, soybean decreased to 64.7 million acres in 2007. The decrease in 2007 has been extensively examined and appeared to be mostly from farmers shifting existing crops to corn (Wallander et al., 2011); but, section 5.3.1.2 suggests that conversion of perennial lands was also occurring (e.g., blue bars in Figure 5.4a, Figure 5.7). After 2007, the soybean trends were not flat like corn, but rather increased, especially in 2014 (+5 million acres from 2013), and again in 2017 (+8 million acres from 2016) (Figure 5.8). These increases in soybean are likely due to many factors, including increased international trade (especially with China) and because corn and soybean are historically grown in rotation, and after a period of growing more corn, farmers returned to the historical rotations but at a higher level of combined corn/soy acreage. Additionally, domestic demand due to increases in livestock and poultry populations as well as increased production of soybean-based biodiesel may also partly explain the increase in soy acreage (Sabo et al., 2021; Sabo et al., 2019). The western expansion of corn and soy acreage over the past three decades (illustrated in Figures 5.5–5.7), in particular, has been thought to be at least partly a result of breeding improvements that led to better adaptation in this region, increases in precipitation, and improvements in technology and cultivation techniques (Hoffman et al., 2020; Gale et al., 2019; Gale, 2015; Clay et al., 2012). In 2017 and 2018, planted soybean was at an all-time high of roughly 90 million acres (Figure 5.8), 15 million acres more than the 2000–2006 period. There was a notable decrease in soybean in 2019 due largely to trade tensions with China and poor planting conditions in the spring (USDA, 2020c), but this decrease seems temporary as acreage began to increase again after 2019.

Other major crops showed notable trends over the recent period as well. Wheat hovered around 60 million acres from 2000 to 2008, but then decreased from 2008 to 2020 to just over 40 million acres in 2020. Cotton was relatively stable over the entire period, 15.5 million acres in 2000 and 13.8 million acres in 2020 with two small dips in 2007–2010 and in 2015. Hay harvested decreased over the same time period, from 60.4 million acres in 2000 to 52.8 million acres in 2020. Overall declines in small grains (e.g., oats, barley, wheat, sorghum) as well as hay (~28 million acres) have been only partly offset by increases in corn and soy acreage (~22 million acres) from 2005 through 2020. However excluding hay, which does not fall under cultivated cropland in NRI, the increase in corn/soy acreage exceeds declines in small grains by roughly 4 million acres for the 2005–2020 period, which is 6 million acres less than if comparing the difference between the years 2005 and 2017. As mentioned above, the NASS acreage data was consistent with trends in NRI through at least 2017 and more recent NASS survey data (post-2017) suggests diversity in agricultural crop production is continuing to decline as corn/soy increasingly occupy a greater proportion of cultivated cropland.

A recent study leveraged the USDA Cropland Data Layer (CDL) and National Land Cover Dataset (NLCD) to ascertain spatiotemporal patterns of cropland expansion and abandonment for the 2008–2016 period (Lark et al., 2020). In addition, they identified crop types that were the first to be cultivated on areas previously uncultivated for at least 20 years (defined further below). The CDL is a remote sensing-based data product produced by the USDA that identifies cropland areas as well as specific crop types at 30m resolution across the United States. Recent critiques of CDL suggest this product may be limited in its utility for such timeseries analysis (Copenhaver et al., 2021; Dunn et al., 2017), but Lark et al. (2017) have argued that, with appropriate adjustments, the CDL can provide meaningful information. A recent assessment of the CDL suggests that with the appropriate aggregations and processing the skill for identifying cropland is 97% or higher (Lark et al., 2021). It is beyond the scope of this report to resolve this debate, and these findings are reported as an additional, though possibly less certain, line of evidence when compared to NRI, to track changes in LCLM through time. Nonetheless the findings from the CDL broadly agree with other datasets (see RtC2 Table 4 and below).

If a given area in the nationwide analysis (1) was classified as noncropland for at least 6–10 years prior to conversion to cropland, (2) remained cropped for at least 2 years, and (3) never transitioned back to noncropland over the 2008-2016 period, then that area would have been considered a conversion to cropland from noncropland (i.e., cropland expansion). Since 2008, large relative increases in cropland area have occurred in southern Iowa, the Dakotas, eastern Nebraska, and northern Missouri (oftentimes >5%, Figure 5.9). These relative increases in cropland are consistent with the NRI-derived estimates of perennially managed agricultural land (pasture, CRP, and noncultivated cropland) converting to either other cultivated cropland or corn and soy (Figures 5.5 and 5.6) or net loss of perennially managed acreage after 2007 in the same states (Figure 5.7). Cropland abandonment was particularly concentrated in the coastal plain and piedmont regions of the Chesapeake Bay watershed, and southeastern North Carolina (Figure 5.9). These patterns are not as apparent in the NRI gross and net change maps (Figures 5.5–5.7), though it may be partly a function of binning and the scale of the analysis. Overall and on a net basis across the United States, the CDL suggests cultivated cropland expanded by about 6.6 million acres from 2008 to 2016 (gross increase of 10.1 minus gross decrease of 3.5 million acres). This net change from 2008 to 2016 is 3.4 million acres less than the net 10 million acres increase in corn/soy and other cultivated cropland which NRI estimated from 2007 to 2017 (i.e., Table 5.2). Despite varying definitions of cultivated cropland and methodologies, the NRI, Census, MLU, and this CDL analysis all suggest a 6.6 to 10 million-acre increase in cultivated cropland post-2007.

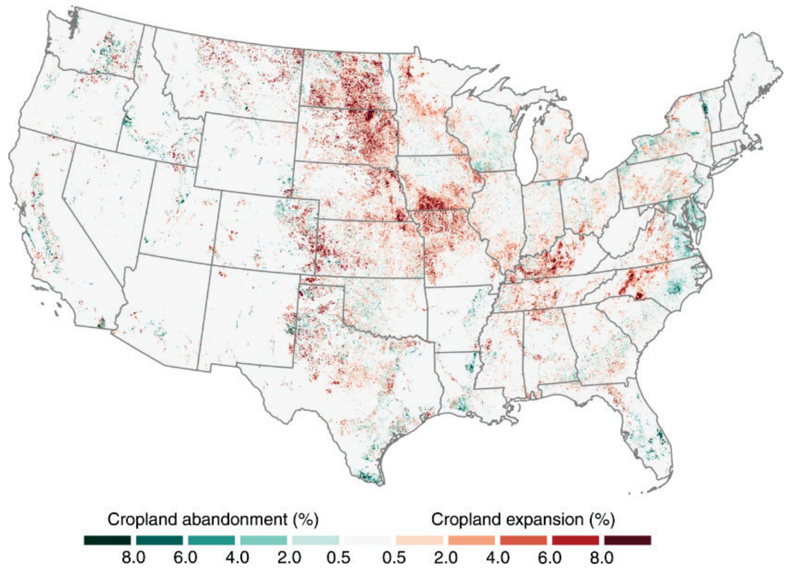


Figure 5.9. Using the USDA Cropland Data Layer, relative estimates of net cropland conversion from 2008 to 2016. Displayed as a percentage of total land area within a non-overlapping 3 x 3 km block, net cropland conversion is calculated as net cropland expansion minus gross abandonment. Source: Lark et al. (2020) (Creative Commons license, https://creativecommons.org/licenses/by/4.0/).

In addition to general cropland and noncropland categories to characterize cropland expansion and abandonment, Lark et al. (2020) identified the specific crops that were planted on newly cultivated land. By and large, corn and soybean were the predominant crops cultivated on new cropland with the majority concentrated once again in the Dakotas, Iowa, Missouri, Nebraska, and Kansas (Figure 5.10). Notably, corn and soy were also planted on newly cultivated cropland in Tennessee and Kentucky, and this increase in acreage is consistent with the regional NRI analysis (Figures 5.5–5.7). The detected increase in other cultivated cropland and the loss of perennially managed agricultural acreage in Montana in the NRI (Figures 5.5–5.7) is also consistent with large increases in wheat cultivation on newly cultivated land from Lark et al. (2020) (Figure 5.10). Complimenting spatiotemporal insights from the NRI, this CDL analysis suggests that newly cultivated cropland has been largely planted with corn and soy with a coincident northwestern shift of wheat cultivation to North Dakota and Montana. The rate of cropland expansion appears to have peaked in 2011, decreasing from 2011 to 2013, and then stabilizing after the E10 blend wall was approached in 2013 (see Chapter 1, Figure 1.4), with further decreases from 2013 to 2016 (Figure 5.10, Lark et al. (2020)).

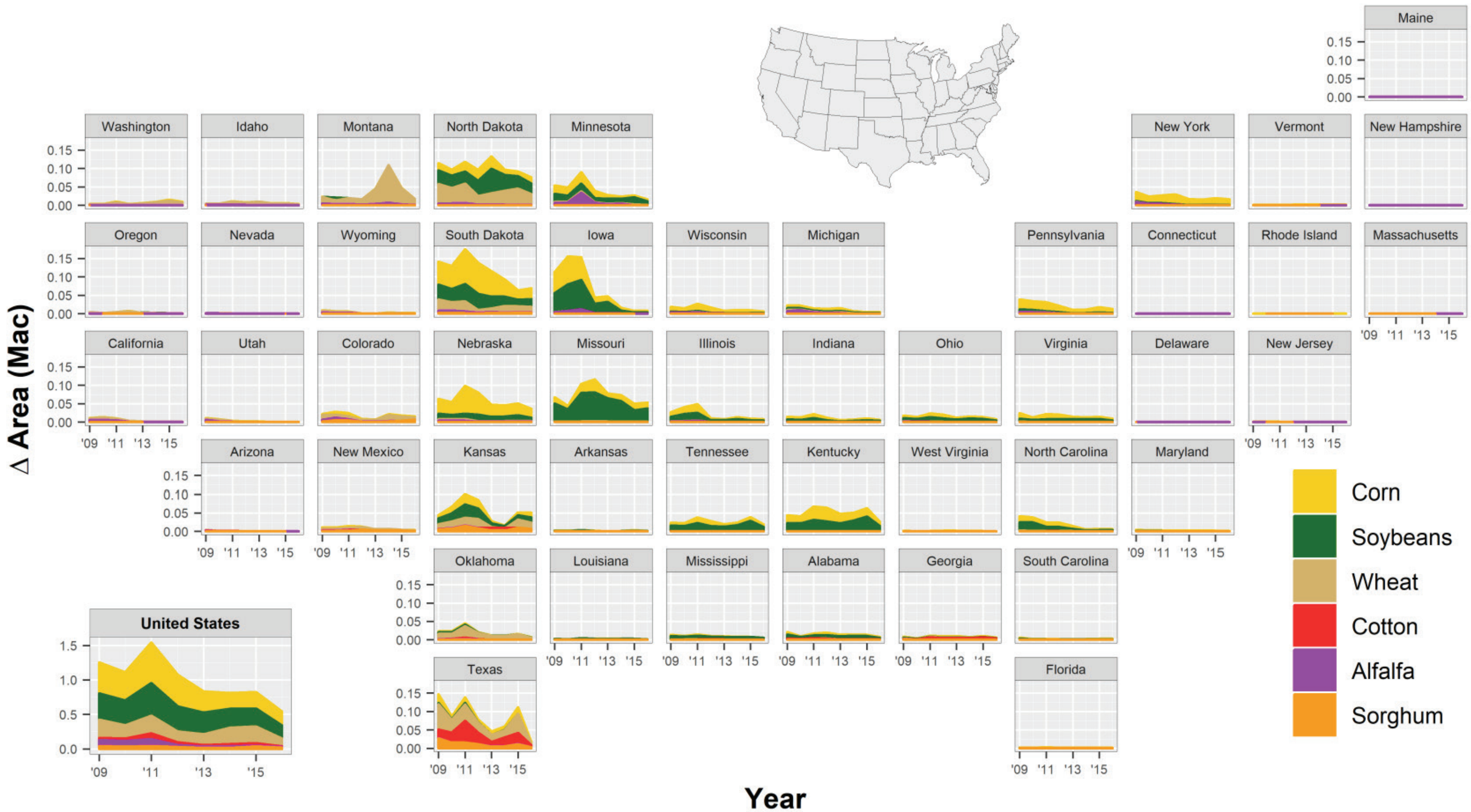


Figure 5.10. By state and year, identification, and acreage (million acres) of the first crop type planted on newly cultivated land from 2008 to 2016. First crop type was identified using the USDA Cropland Data Layer. Source: Lark et al. (2020) (Creative Commons license, https://creativecommons.org/licenses/by/4.0/).

5.3.1.4 Crop Rotations and Double Cropping

Changes in crop rotations and double cropping may also be meaningful in the context of the environmental and resource conservation effects in this report. Crops are not all managed the same, so shifts toward or away from more intensively managed crops can have implications on the environment. Corn receives more fertilizer and pesticide than many other crops (see Chapter 3, section 3.2.1); thus, shifts toward more corn likely has implications on the environment. For example, the total mass of nitrogen fertilizer applied to an acre of corn in the United States far exceeds cotton, soybeans, or wheat (USDA, 2019) (also see Chapter 3, Figure 3.13). Corn also requires substantial nitrogen, phosphate, and potash application to maintain increasingly high yields. If increased nutrient application exceeds local corn crop needs due to overapplication of fertilizer/manure nutrients or depressed yields due to drought (e.g., Sabo et al. (2019) and Sabo et al. (2021)), these applications may result in large deleterious environmental effects for local air and water quality as well as downstream systems. There is some emerging evidence that cropland nutrient use efficiency and declines in surplus are occurring nationwide, especially for phosphorus, but these trends vary across states (e.g., Zhang et al. (2021); TFI (2023)). Overall increased nutrient application will not necessarily result in increased environmental degradation, but inefficient application of nutrients may greatly exacerbate nutrient pollution if nutrient inputs greatly exceed crop uptake rates. In addition to larger nutrient application, corn acreage received 39.5% of total pesticide application, primarily glyphosate, despite only making up approximately 30% of total cultivated cropland acreage (Fernández-Martínez et al., 2017) (also see Chapter 3, section 3.2.1).

As reported in the RtC2 (U.S. EPA, 2018), double cropping is not widely adopted in the United States. A recent NASS report shows that double cropping only occurred on roughly 2% of total cropland (roughly 8 million acres) for most years between 1999 and 2012 and did not show a consistent trend for any of the seven regions examined (Borchers et al., 2014), and this proportion did not change from 2005 to 2019 (USDA, 2020b). Thus, there do not appear to be trends in double cropping associated with the RFS Program.[16] There has been no nationwide assessment on changes in crop rotations, but region-specific studies suggest that more rotations with corn are occurring in Iowa (Ren et al., 2016) and in other parts of the Midwest (Plourde et al., 2013). Furthermore, the total increases in corn and decreases in other crops discussed in sections 5.3.1.2 and 5.3.1.3, and the higher input rates for corn compared with most other crops, suggest that effects from rotations toward corn may be occurring as well.

[16] There are, however, industry reports of the potential for more double cropping in the south to capitalize on canola for biodiesel (e.g., https://www.biobased-diesel.com/post/corteva-bunge-and-chevron-collaborate-to-produce-winter-canola-as-feedstock-for-biobased-diesel).

5.3.1.5 Trends in CRP

While this chapter does not attribute drivers of LCLM changes across the United States, the dramatic decline in CRP acreage and its association with the maximum allowed acreage, or caps, legislated by the Farm Acts will be summarized to provide further context when interpreting trends in CRP (Coppess, 2017). It should be emphasized that similar programs to CRP existed prior to the 1980s and 1990s (Figure 5.11a). Up to 40 million acres were taken out of production during the New Deal era (1930s and 1940s) and up to 75 million acres were removed via soil bank and set aside programs thru the early 1990s (Figure 5.11a). Similar to these predecessor programs, the primary objective of CRP was originally to protect highly erodible and otherwise environmentally sensitive cropland and pasture, thus the concentration of CRP land in the more arid western plains (e.g., North Texas, Figures 5.6–5.7). However, the CRP's influence and goals have changed over time (Hellerstein, 2017), in turn leading to a need to acquire land that can be acquired outside of the

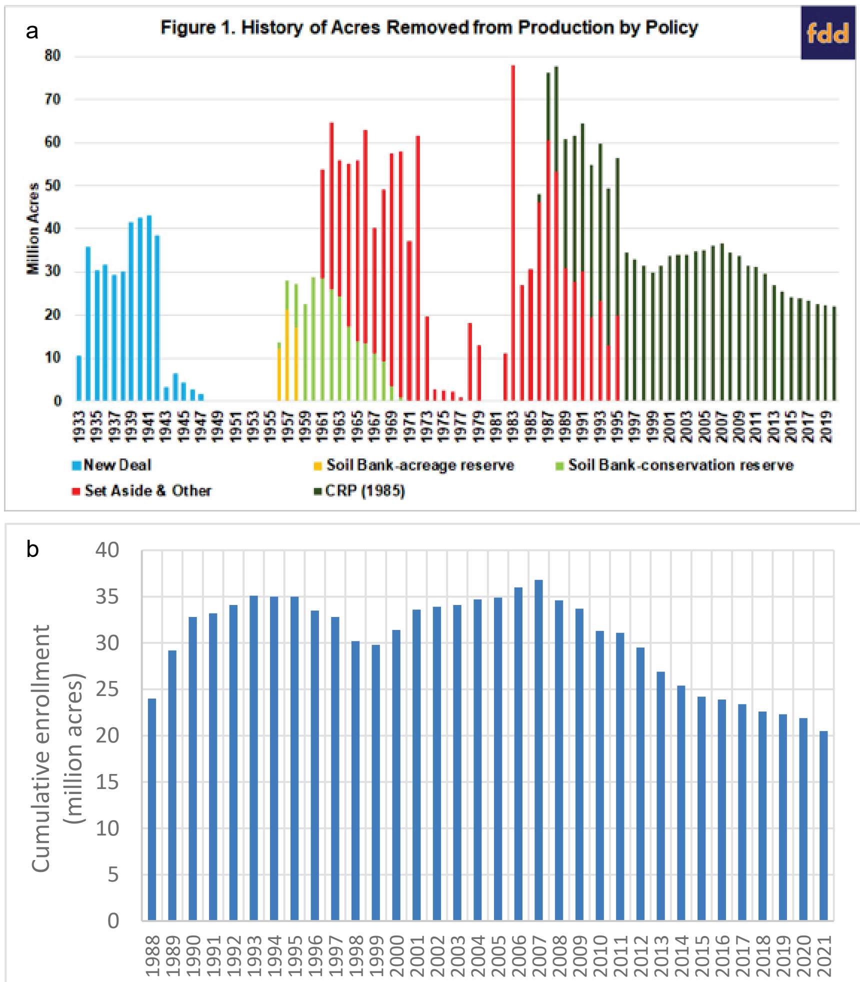


Figure 5.11. Federal policies that removed cultivated cropland acreage from production since 1933 (a), and total CRP land (general enrollment + continuous enrollment) from 1988 to 2021 (b). Source: (a) Coppess et al. (2020) (used with permission); (b) Data from USDA (2020a).[17]

[17] Note that the lower acreages in Table 5.2 compared with here in Figure 5.11 is because Table 2 (NRI) only has general enrollment CRP, whereas Figure 5.11 has both general and continuous enrollment (i.e., total enrollment). While farmers were largely limited in their ability to sign up for general enrollment after 2007, there was some opportunity to maintain continuous enrollment. Thus, general enrollment decreased ~50% (Table 5.2, partly because of limited opportunities) yet total CRP acreage only declined ~30% (Figure 5.11). Note that the GCRP was

general enrollment process. The general enrollment process involves farmers auctioning/bidding for highly erodible land up to the cap, but CRP developed a process to continuously add cropland acreage for specific restoration projects to help meet other environmental goals (e.g., filter strips, wetland restoration). From 2002 to 2007 there was an increase in total and general CRP up to about 37 and 32 million acres (Figure 5.11b), respectively, which is a little less than the 39.2 million-acre cap set by the 2002 Farm Security and Rural Investment (FSRI) Act. The next Farm Bill, the Food, Conservation and Energy (FCA) Act of 2008, set a lower cap of 32 million acres and both general and total CRP land decreased steadily, falling below the cap in 2010 and ending at roughly 25 and 30 million acres in 2012, respectively. The Agricultural Act of 2014 set the next CRP acreage cap down to 27.5 million acres for 2014 and established that the next cap would be reduced continuously to 24 million acres by 2018. During this period CRP acreage decreased by 2 million acres per year to about 24 million acres in 2015 (Figure 5.11b) and this continued through 2020. Although leveling off, the CRP is at its lowest acreage historically. It is also important to note that in addition to the cap being lowered, opportunities for general sign ups were limited in the years spanning 2014–2019. The majority of CRP acreage is enrolled through this general sign up. Thus, farmers had little opportunity to re-enroll expiring CRP acreage into the general program and could essentially only reenroll in the continuous CRP sign-up.[18] It should be emphasized that data regarding why these land management decisions by individual farmers were made are not available, and comments about CRP enrollment caps in this section should not be construed as explaining why CRP acreage has declined. In the most recent Farm Bill, the Agriculture Improvement Act of 2018 increased maximum allowable CRP land to 27 million acres in 2023. The response of CRP acreage to this new allotment will be assessed in the future when more data become available.

Also introduced in the 2018 Farm Bill was the Grassland subprogram of the CRP (GCRP). The GCRP is a voluntary, working lands program that contracts with agricultural producers to enhance the sustainability of their grazing operations and to protect grazing lands under threat of conversion. As of October 2023, the GCRP is the single largest CRP subprogram (by acreage, 35%), followed by continuous signup (second) and general (third). Because GCRP targets grazing land, there is no crop

introduced in the 2018 Farm Bill and is not shown in Figure 5.11. The acreage in GCRP in fiscal years 2018, 2019, 2020, and 2021 was 0.5, 0.9, 0.9, and 1.8 million acres. By 2023, GCRP has become the largest subprogram in the CRP (https://www.fsa.usda.gov/programs-and-services/conservation-programs/reports-and-statistics/conservation-reserve-program-statistics/index).

[18] The CRP historically (until 2018) had two enrollment types: general and continuous. Under general enrollment, producers have the opportunity to offer land for CRP general enrollment annually during announced enrollment periods. Offers for CRP contracts are ranked according to the Environmental Benefits Index (EBI). Under continuous enrollment, environmentally sensitive land devoted to certain conservation practices may be enrolled in CRP at any time. Certain eligibility requirements still apply, but offers are not subject to competitive bidding. Many of the lands that enrolled during the general signup were not eligible for the continuous signup. Since 2018, the Grassland subprogram of the CRP was added, introducing lands to the CRP that were not required to have a history of cropping.

history requirement and the number of total CRP acres associated with converted cropland has decreased over time.[19]

5.3.2 Future Trends Domestically

The future trends for LCLM for the RtC3 are derived from the most recent USDA Long Term Agricultural Projections (LTAP) available at the time of writing, which were released in February 2021 and cover until 2030 (IAPC, 2021). These future trends are broad in nature, and do not specifically account for the final volumes proposed under the Set Rule for 2023–2025.[20] For the estimated likely future changes in LCLM attributable to the RFS Program for 2023–2025, see Chapter 6, section 6.5. Thus, the future trends discussed here are meant to provide broader context on the specific trends discussed in Chapter 6, and focus on the near-term LTAP projections out to 2025. USDA clearly states that the LTAP estimates are not a prediction of future events, instead they are an estimate of what is expected to happen under a continuation of current policies and economics, and assuming no unusual weather, geopolitics, or other factors. Nonetheless, they represent the USDA's best estimate of the likely future in the agricultural sector. There are many other future projections in the peer-reviewed literature, but the majority of these are focused on either longer-term projections beyond 2025, or hypothetical scenarios (e.g., large increases in cellulosic production) that have not yet become a "likely future" in EPA's estimation.

The LTAP reports a wealth of information on U.S. and global production of commodity crops, trade, and other factors under a predefined set of assumptions. The assumptions cover a range of topics, and those most relevant for the RtC3 are provided below (Table 5.3). For a full list of assumptions see IAPC (2021).

[19] See data published by FSA for more detail (https://www.fsa.usda.gov/programs-and-services/conservation-programs/reports-and-statistics/conservation-reserve-program-statistics/index).

[20] On July 26, 2022, the United States District Court for the District of Columbia filed a consent decree, which after stipulations from the parties, required EPA to sign a notice of proposed rulemaking (NPRM) to establish 2023 volumes for the RFS Program on or before November 30, 2022, and to sign a notice of final rulemaking finalizing 2023 volumes by June 21, 2023. Growth Energy v. Regan et al., No. 1:22-cv-01191 (D.D.C.), Document Nos. 12, 14, 15. EPA's proposed RFS volumes for 2023-2025, 87 FR 80582 (proposed and signed on Nov. 30, 2022, and published in the Federal Register on Dec. 30, 2022) available in Docket No. EPA-HQ-OAR-2021-0427 at https://www.regulations.gov, were finalized on June 21, 2023.

The 2021 LTAP projects relatively stable levels of all commodity crops after short-term effects due to trade disruptions and weather occur (Figure 5.12, Table 5.4). Corn is expected to increase slightly as corn/soy farmers opt to grow corn given trade tensions with China (USDA, 2020e). This increase is projected to be short term as increasing demand for soybean meal due to higher meat consumption globally restores the demand for U.S. soybean despite trade tensions. Soybean, after dropping sharply in 2019 due to weather-related planting issues and trade tensions with China, is projected to rebound and remain relatively steady. For wheat, after a long historical decline

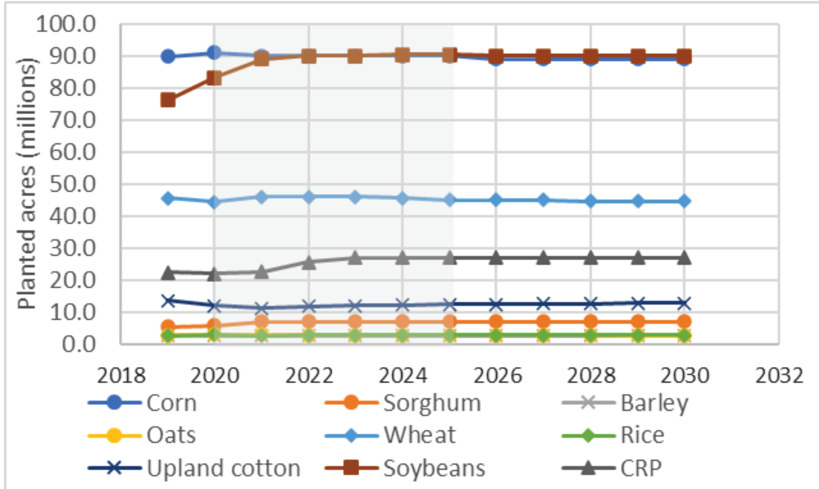


Figure 5.12. Trends in eight principal crops and CRP from 2019 to 2030. Shaded in gray is the interval of interest for the RtC3 (2020–2025)[21] (IAPC, 2021).

[21] Note that projections in the 2020 LTAP begin in 2019, with historical data used for 2018. Differences in 2019 between projections and observed data are very small.

Table 5.3. Key assumptions in the USDA 2021 Long Term Agricultural Projections.

Topic	Assumptions
Global Economics and Energy Prices	▪ Global real economic growth is projected to average 2.7% annually over the next decade, 2020–2029. The United States is expected to average 1.8% growth annually, while developed countries as a group are expected to experience an average of 1.5% annual growth. Meanwhile, growth in the developing countries remains faster than the global average, but declines from 4.8% annual average growth during 2010–2019, to 4.3% during 2020–2029. ▪ As global economic activity improves, crude oil prices are assumed to increase from their recent lows (under $40 per barrel in 2016 for the first time since 2004).
Agricultural Policy	▪ The Agriculture Improvement Act of 2018 is assumed to be in effect through the projection period. Similarly, the trade tariffs in place as of October 2019 are assumed to remain in effect throughout the next 10 years. The projections only include policies in place or already expected to be implemented as of October 2019. Recent trade agreements or discussions including the Phase One deal with China, the USMCA agreement, and a Japan-U.S. free trade agreement were not considered for these projections. ▪ Acreage enrolled in the CRP is assumed to rise to nearly 27 million acres, which is the maximum level legislated by the 2018 Farm Act.
U.S. Biofuels	▪ Final renewable fuel standards for cellulosic biofuel, advanced biofuel, and total renewable fuel for 2020 were announced by EPA on December 19, 2019. The biomass-based diesel (BBD) standard for 2020 and 2021 was also set in December 2019. These projections were completed before any subsequent volume requirements were established by EPA. ▪ Over the baseline period, corn use for ethanol production is projected to increase in most years, rising 5% over the baseline period. Ethanol exports are assumed to account for the gain in use, and imports remain mostly flat. Demand for corn to produce ethanol continues to have a strong presence in the sector, accounting for at least one-third of total U.S. corn use. ▪ Underpinning the projections are declines in overall gasoline consumption in the United States. The 10% ethanol blend wall is assumed to constrain domestic ethanol use over the next decade. Most gasoline in the United States continues to be a 10% ethanol blend (E10). Some growth in the E15 (15% ethanol blend) market will occur with the approval of year-round blending, but infrastructure and other constraints limit growth. The E85 (51 to 85% ethanol blend) market remains small. The impact of Small Refinery Exemptions on biofuels consumption is expected to diminish. ▪ According to EIA data, motor gasoline prices will increase 12% over the baseline period. This, combined with a more efficient vehicle fleet, will have a dampening effect on gasoline consumption, which limits ethanol consumption. ▪ The BBD use volume requirement, as administered by the EPA under the Renewable Fuels Standard, was 2.1 billion gallons for 2019, is raised to 2.43 billion gallons for 2020 and 2021 and is assumed to continue at that level.
International Policy and Biofuels	▪ During 2018, China imposed retaliatory tariffs of 25% or more on nearly all U.S. agricultural commodities. The projections to 2029 assume these tariffs remain in effect throughout the projection period. ▪ Global production of biofuels is projected to continue to increase during the next decade, although at a slower pace than over the previous half-decade. This slowdown, in part, reflects crude oil prices, that despite their projected growth, are expected to remain below the levels reached earlier in the decade. In addition, of the countries with biofuel programs, blending growth is likely to slow as many have already reached or approached their biofuel use targets. The remaining countries with larger gasoline fuel pools that have not yet adopted a fuel ethanol program are unlikely, in most cases, to do so over the baseline period as alternative sources of engine power (electric, natural gas) gain ground and transportation habits change (e.g., greater use of public transport and ride-sharing).

Table 5.4. Annual planted acreages (millions of acres) for the eight principal crops and CRP from 2019 to 2030 (USDA, 2020e).

Year	Corn	Sorghum	Barley	Oats	Wheat	Rice	Upland Cotton	Soybeans	CRP
2019	89.7	5.3	2.8	2.8	45.5	2.5	13.5	76.1	22.3
2020	91.0	5.8	2.6	3.0	44.3	3.0	11.9	83.1	22.0
2021	90.0	7.0	2.6	2.9	46.0	2.6	11.2	89.0	22.6
2022	90.0	7.0	2.6	2.8	46.0	2.7	11.8	90.0	25.5
2023	90.0	7.0	2.6	2.8	46.0	2.7	12.0	90.0	27.0
2024	90.0	7.0	2.6	2.8	45.5	2.7	12.2	90.5	26.9
2025	90.0	7.0	2.6	2.8	45.0	2.7	12.3	90.5	27.0
2026	89.0	7.0	2.6	2.8	45.0	2.7	12.4	90.0	26.9
2027	89.0	7.0	2.6	2.8	45.0	2.7	12.5	90.0	26.9
2028	89.0	7.0	2.6	2.8	44.5	2.7	12.6	90.0	26.9
2029	89.0	7.0	2.6	2.7	44.5	2.7	12.7	90.0	26.9
2030	89.0	7	2.6	2.7	44.5	2.7	12.8	90	26.9

(Figure 5.1), acreages are expected to remain relatively stable. Acreage enrolled in CRP is assumed to rise to nearly 27 million acres, which is the maximum level legislated by the 2018 Farm Act, up from the 2014 Farm Act cap of 24 million acres. The 2018 Farm Act largely remains in force through 2023. Farmers have historically found that enrollment in the CRP is an attractive low-risk way to get revenue from lower quality lands (Coppess, 2017; Hellerstein, 2017).

It is important to reiterate the wide fluctuations in actual historical plantings compared with the relatively stable plantings in future projections (Figure 5.13). This is not unexpected, as the steady-state assumptions in the modeling will almost certainly not occur in the real world. Nevertheless, these estimates are not expected to be biased high or low and reflect reasonable expectations in the absence of shocks to the agricultural system.

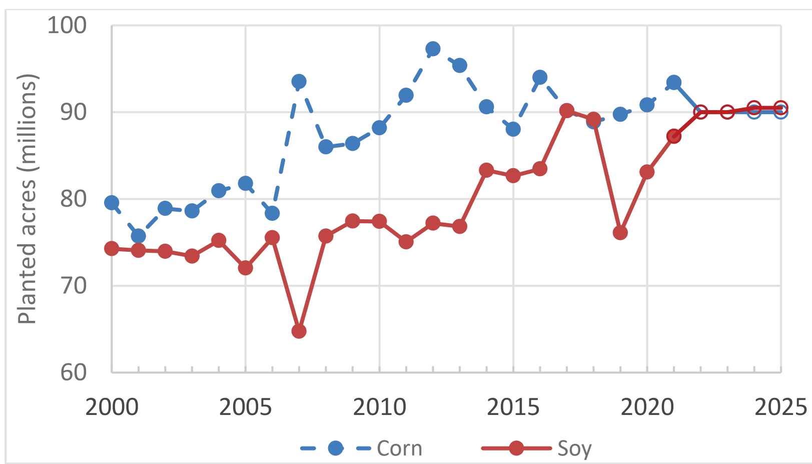


Figure 5.13. Actual plantings (closed circles) for corn (blue) and soybean (red) from 2000–2021 from NASS, compared with projected plantings from 2020–2025 in the LTAP (actual and projected plantings for 2020 are on top of one another). Source: NASS (https://quickstats.nass.usda.gov/) and (USDA, 2020e).

Corn yields are expected to continue to increase over the projected period from 178.4 bushels per acre in 2020 to 198.5 in 2030 (Table 5.5). Corn supply is expected to continue to come predominantly from annual production, comprising roughly 86% of annual supply on average over the period compared with beginning stocks (~14%) and imports (<1%) (Table 5.5).

Table 5.5. Corn yields (bushels per acre [bu/ac]), supply, and use from 2019 to 2030 from the LTAP (supply and use are in millions of bushels).

Beginning Market Year (MY)	Yield (bu/ac Harvested)	Supply (Million Bushels)			Use (Million Bushels)			
		Beginning Stocks	Production	Imports	Feed and Residual	Food, Seed, and Industrial	Ethanol and Byproducts	Exports
2019	167.5	2,221	13,620	42	5,827	6,282	4,852	1,778
2020	178.4	1,995	14,722	25	5,775	6,475	5,050	2,325
2021	180.5	2,167	14,890	25	5,950	6,550	5,125	2,325
2022	182.5	2,257	15,055	25	6,050	6,545	5,125	2,375
2023	184.5	2,367	15,220	25	6,200	6,545	5,125	2,425
2024	186.5	2,442	15,385	25	6,275	6,540	5,125	2,475
2025	188.5	2,562	15,550	25	6,400	6,540	5,125	2,525
2026	190.5	2,672	15,525	25	6,425	6,535	5,125	2,575
2027	192.5	2,687	15,690	25	6,500	6,535	5,125	2,625
2028	194.5	2,742	15,850	25	6,575	6,555	5,150	2,675
2029	196.5	2,812	16,015	25	6,725	6,555	5,150	2,725
2030	198.5	2,847	16,180	25	6,850	6,550	5,150	2,775

Corn use is projected to be relatively stable as well (Figure 5.14), with 32% on average used for food, seed, and industrial uses, 31% for feed and residual uses, 25% for ethanol and byproducts (e.g., distillers dried grains with solubles [DDGS]), and 12.0% for exports. Thus, compared with the large increase in corn use for ethanol from 2005 to 2013 (see Chapter 3, Figure 3.9), projections from 2020 to 2025 are expected to be relatively stable.

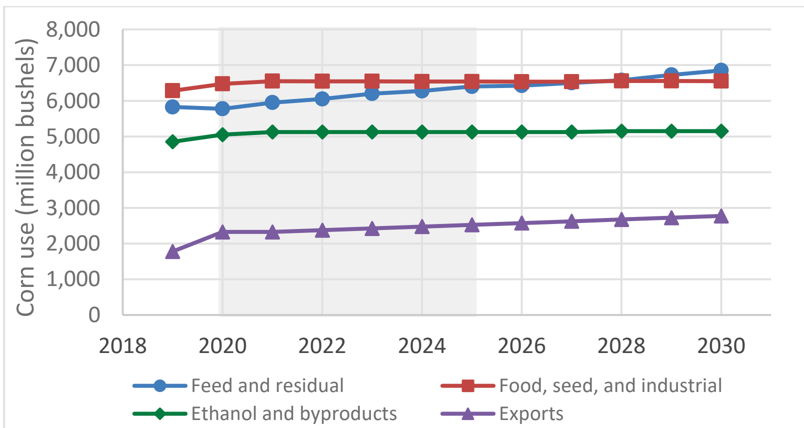


Figure 5.14. Trends in projected uses of corn from 2019 to 2030 (IAPC, 2021). Shown are market years labeled by the starting year. Shaded in gray is the interval of interest for the RtC3 (2020–2025).

Soybean yields are expected to increase over the projected period from 51.9 bushels per acre in 2020 to 55.6 in 2030 (Table 5.6). Soybean supply is expected to continue to come predominantly from annual production, comprising roughly 92% of annual supply on average over the period compared with beginning stocks (~7%) and imports (<1%) (Table 5.6). Soybean also has many uses in the economy, with the original harvest used predominantly for either crush (49%) or export (48%) (Table 5.6). The crush is then separated into either soybean oil (for biodiesel, food, feed, and other industrial uses [including renewable diesel[22]], and exports), or soybean meal (for feed and exports) (Table 5.7). As

[22] In the 2030 LTAP (IAPC, 2021) renewable diesel was a part of the "Food, Feed, and Other Industrial" category.

illustrated in Figure 5.15, uses of soybean as biodiesel are projected in the LTAP to be flat because USDA assumed the EPA rulemaking from December 2019 would remain flat throughout the period of study (Table 5.3). However, domestic uses, and oil use in food, feed, and other industrial uses are all projected to increase from 2020 to 2025, while exports are projected to decrease slightly before rebounding.

Table 5.6. Soybean yields (bushels per acre [bu/ac]), supply, and use from 2019 to 2030 from the LTAP (supply and use are in millions of bushels).

Beginning Market Year (MY)	Yield (bu/ac Harvested)	Supply (million bushels)			Use (million bushels)		
		Beginning Stocks	Production	Imports	Crush	Seed and Residual	Exports
2019	47.4	909	3,552	15	2,165	112	1,676
2020	51.9	523	4,268	15	2,180	136	2,200
2021	50.6	290	4,465	15	2,200	140	2,175
2022	51.2	255	4,565	15	2,230	140	2,195
2023	51.7	271	4,610	15	2,260	141	2,210
2024	52.3	285	4,685	15	2,290	141	2,255
2025	52.8	299	4,735	15	2,315	141	2,290
2026	53.4	303	4,760	15	2,345	141	2,295
2027	53.9	297	4,810	15	2,375	142	2,305
2028	54.5	300	4,855	15	2,405	142	2,330
2029	55	294	4,905	15	2,435	142	2,345
2030	55.6	291	4,955	15	2,460	143	2,375

Table 5.7. Projected supply and uses of soybean oil and meal from the crush from the LTAP (IAPC, 2021).

Beginning Market Year (MY)	Soybean Oil (million pounds)						Soybean Meal (thousand short tons)				
	Supply			Use			Supply			Use	
	Beginning Stocks	Production	Imports	Biodiesel	Food, Feed, and Other Industrial	Exports	Beginning Stocks	Production	Imports	Domestic	Exports
2019	1,775	24,890	325	7,850	14,600	2,800	402	51,028	620	37,750	13,900
2020	1,740	25,265	350	8,100	14,900	2,600	400	51,400	400	38,300	13,500
2021	1,755	25,520	450	8,150	15,350	2,500	400	51,975	400	38,725	13,650
2022	1,725	25,880	450	8,200	15,800	2,400	400	52,625	400	39,225	13,800
2023	1,655	26,240	450	8,250	16,025	2,300	400	53,300	400	39,750	13,950
2024	1,770	26,600	350	8,300	16,250	2,400	400	53,975	400	40,275	14,100
2025	1,770	26,900	350	8,350	16,450	2,400	400	54,650	400	40,800	14,250
2026	1,820	27,260	350	8,400	16,650	2,500	400	55,325	400	41,325	14,400
2027	1,880	27,620	350	8,450	16,850	2,650	400	56,000	400	41,850	14,550
2028	1,900	27,980	350	8,500	17,050	2,750	400	56,675	400	42,375	14,700
2029	1,930	28,345	350	8,550	17,250	2,900	400	57,350	400	42,900	14,850
2030	1,925	28,645	350	8,600	17,450	2,950	400	58,025	400	43,425	15,000

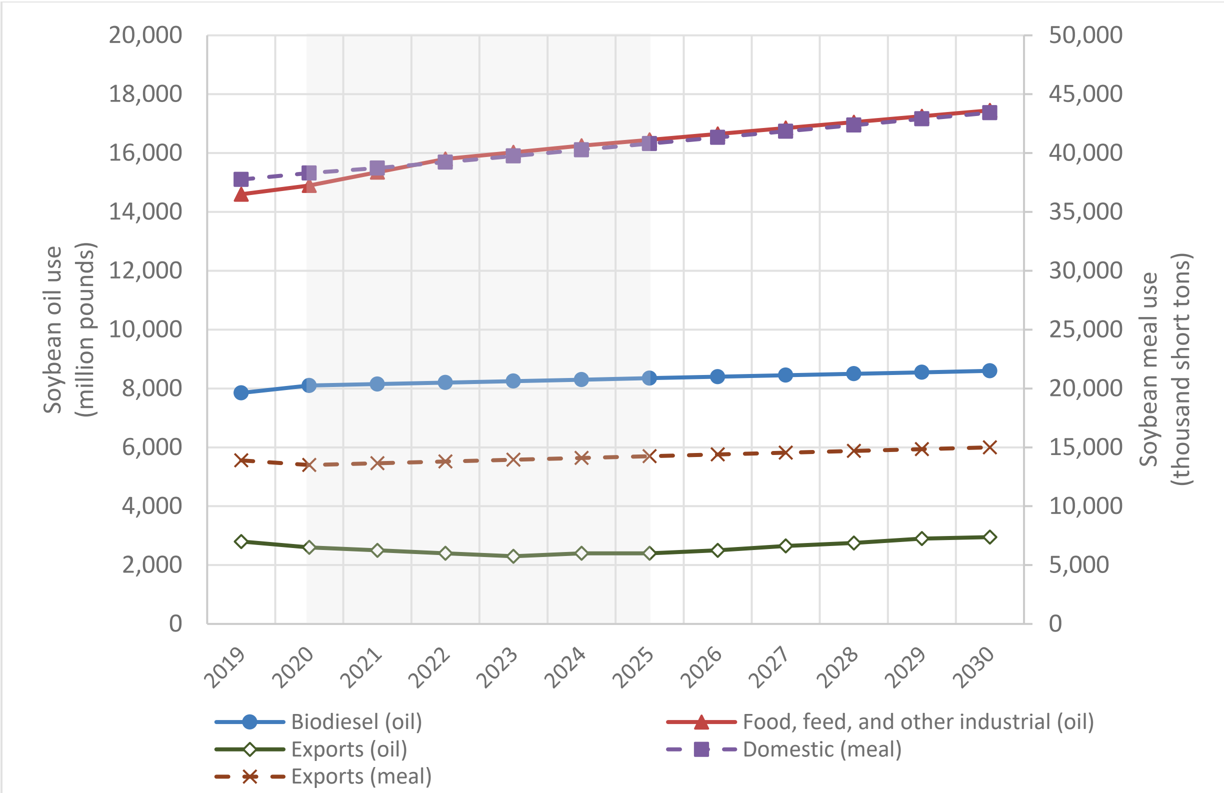


Figure 5.15. Trends in uses of soybean oil (left axis, solid lines) and meal (right axis, dashed lines) from 2019 to 2030. Shown are market years labeled by the starting year. Shaded in gray is the interval of interest for the RtC3 (2020–2025) (IAPC, 2021).

5.4 Synthesis

5.4.1 Chapter Conclusions

- After decades of decline, increases in cultivated cropland have been recorded in multiple federal datasets, using a variety of methodologies, following the 2007 to 2012 period. This increase ranges from 6 to 10 million acres.
- Based on the 2012, 2015, and 2017 National Resource Inventories (NRIs), there has been a steady increase in agricultural intensity from 2007 to 2017 with a 10 million-acre increase in cultivated cropland coinciding with a 15 million-acre decline in perennially managed land (i.e., sum of lands in Conservation Reserve Program [CRP], pasture, and noncultivated cropland). This increase in cultivated cropland was largely driven by a net 26.5 million-acre increase in corn and soy with small grains and hay in rotation decreasing 16.5 million acres.
- More than half of the corn and soybean increase has come from other cultivated cropland (56%), while the rest has come from approximately equal proportions of pasture (13%), noncultivated cropland (20%), and CRP (11%). Corn likely has larger environmental effects

than hay, pasture, and other crop types because corn typically uses more fertilizer, pesticides, and other inputs than other crops.

- Many of these changes are taking place throughout the Midwest, with hotspots in northern Missouri, eastern Nebraska, the Dakotas, Kansas, and parts of Wisconsin.
- Based on both the National Agricultural Statistics Service (NASS) and the NRI, crop production is becoming less diverse in the United States as cultivated cropland, besides that of the increasing corn/soy acreage, continue to decline from 2000 to present.
- These changes in cultivated cropland acreage have coincided with increased corn and soybean yields and increasing adoption of a variety of best management practices like conservation and no-till tillage practices.
- After short-term disruptions from weather and trade disputes with China, the USDA Long Term Agricultural Projections (LTAP) suggest that corn acreage and corn used for ethanol will remain relatively stable from 2020 to 2025, declining slightly thereafter. This projected decline is driven by increases in fuel efficiency decreasing total gasoline consumption, increasing crop yields, and E10 blend wall issues further exacerbated by slow growth in E15 an E85 consumption. Likewise, soybean acreage is projected to remain stable due to increased yields meeting both domestic and international demand, especially to meet growing international meat consumption.

5.4.2 Conclusions Compared to Last Report to Congress

The RtC3 generally shows that the conclusions drawn in the RtC2 still hold, and that the general trends observed in the previous report have continued. The RtC2 reported a roughly 4–8 million-acre increase in cultivated cropland from 2007 to 2012, and this report highlights that this expansion has likely continued with increases now ranging from 6 to 10 million acres from 2007 through 2017. Overall cultivated cropland increased at a rate of roughly 1 million acres per year from 2007 to 2017 if solely relying on inferences from the NRI (though these rates agree with other sources). The RtC2 highlighted much of the increase in cultivated cropland acreage is occurring in the western and northern edges of the corn belt, and this report confirms those same regions in the Dakotas, eastern Nebraska, and Kansas still as major hot spots for cultivated cropland expansion through 2017. Expansion of cultivated cropland, driven by net increases in corn and soy acreage, came largely at the expense of perennially managed land (sum of pasture, noncultivated cropland, and CRP), consistent with previous findings from the RtC2. This report highlights, however, that corn and soy are replacing other cultivated crops, which are in decline, in turn making crop production less diverse in the United States. Increased cultivation of corn potentially has larger environmental effects compared to other crops since it typically requires more fertilizer, pesticides, and other inputs to maximize crop yields.

5.4.3 Uncertainties and Limitations

- It is clear that different datasets (i.e. Census, NRI, NASS, MLU) yield slightly different projections of land use change as well as the timing of changes, though it is unclear exactly what drives these differences.
- Although in aggregate the projections from different datasets appear unbiased once suitable adjustments and definitional reconciliations are made, large amounts of scatter prevent estimating where and when these transitions occurred at fine scales, which is critical for environmental assessments.

5.4.4 Recommendations

- Improvements in the skill of satellite-derived data to successfully characterize grassy habitats (e.g., grassland, pasture, CRP) and the ability to track implementation of conservation tillage, cover crops, and other conservation practices are needed.
- Standardized and repeatable trend assessment approaches of LCLM in the United States, including standardized and consistent land cover and land management classification nomenclature, that integrate the USDA datasets relied upon in this report need to be conducted (i.e., NASS, NRI, Census, CRP, MLU).
- Estimates of LCLM trends for policy decisions in the lower 48 states should preferentially be based on the NRI, complemented by continuous annual survey data such as NASS, though for research efforts other datasets may be suitable or even preferred.
- Research is needed to assess the influence of increasing crop yields on past shifts in cultivated and noncultivated crop acreage.
- Development of spatial datasets at fine resolutions (e.g., county or smaller) tracking the implementation of best management practices are needed to account for efforts that may offset negative environmental effects associated with more intensive management of cropland.

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Part 2

6. Attribution: Corn Ethanol and Corn

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Key Findings

- Many factors have impacted ethanol production and consumption in the United States historically, including higher prices of oil and gasoline, the replacement of methyl tert-butyl ether (MTBE) in reformulated gasoline (RFG) areas, the RFS Program, the Volumetric Ethanol Excise Tax Credit (VEETC), the octane value of ethanol, state programs, and air emission standards.
- The period of rapid growth in the ethanol industry was from 2002 to 2010, and nearly 40% of the increase in ethanol consumption had already occurred by 2006 (the first year of the RFS Program, RFS1[1]), and over 90% of the increase had already occurred by 2010 (the first year of the RFS2).
- Because the factors that affect ethanol production and consumption—including the RFS Program—change through time, so too does the estimated effect of the RFS Program on ethanol production and consumption.
- Evidence from simulation models, observed RIN prices, production exceeding consumption from the RFS standards, and other sources suggest that from 2006 to 2011 the RFS Program—in isolation—accounted for 0–1 billion gallons of ethanol per year, primarily by encouraging market growth and capital investment from the Energy Independence and Security Act (EISA) and to a lesser extent by stabilizing demand during the Great Recession of 2008–2009. In other years of this period, the RFS Program is estimated to have had no effect on ethanol production, with other factors having more influence.
- The synthesis of evidence suggests a dynamic range of effects from the RFS Program from 2012 to 2019 as well, with the largest effect in 2016 (0–2.1 billion gallons per year) primarily due to the RFS Program supporting the industry after other factors had either phased out (e.g., VEETC, MTBE) or diminished in effect (e.g., high oil prices).
- In sum over the entire period assessed, the RtC3 estimates that 0–9% of corn ethanol production and consumption is likely attributable to the RFS Program historically. Lower estimated effects of the RFS Program occur if the effect on market certainty is not considered, or if MTBE replacement by ethanol and transitions to match blending are assumed to be independent of, but coincident with, the RFS Program; larger effects occur if market certainty is included, or if these other factors are omitted or ascribed to the RFS Program.

[1] The RFS1 and RFS2 are described further in Chapters 1 and 2 and refer to the different versions of the RFS Program enacted under the Energy Policy Act of 2005 (RFS1) or the Energy Independence and Security Act of 2007 (RFS2).

- Combining these estimated volumes attributable to the RFS Program with literature reviews and a recent statistical analysis suggests that the RFS Program may be attributable for cropland expansion of zero to 1.9 ± 0.9 million acres, and additional acres of corn of zero to 3.5 ± 1.0 million acres, with the largest potential effect estimated in 2016.
- These best available estimates from econometrics of observed trends are consistent with other econometric studies once appropriate adjustments are made and are consistent with estimates from simulation models.
- The likely future effect of the RFS Program was estimated by EPA in the Final Set Rule on June 14, 2023, which estimated 787 million gallons of corn ethanol consumption in 2025 to be due to the RFS Program, potentially inducing up to 0.46 million acres of cropland expansion. These estimates are highly uncertain, due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes.
- Uncertainties in the estimated effect of the RFS Program on ethanol production remain, including the effect of the RFS Program in establishing market certainty before the mandates were in full effect, the cost, ability, and willingness of refiners to switch back to producing finished gasoline without ethanol if blending ethanol were no longer economical, and others. However, these factors are difficult to quantify with current tools available.
- The RFS Program created a guaranteed market demand for biofuels in the United States that certainly could have helped drive much of the increase in ethanol production and consumption in the United States. However, as events played out, non-RFS factors that also affect ethanol production and consumption (e.g., oil prices, octane value, MTBE bans, tax incentives, state programs) were favorable, and appear to sufficiently explain much or all of the increase in ethanol production and consumption historically in the United States.
- A modest effect from the RFS Program on corn ethanol does not preclude a larger effect on other biofuels (see Chapter 7) as the RFS Program affects many biofuels in addition to corn ethanol.

Chapter Terms (see Glossary): Clean Air Act (CAA), D6 RIN, distillers dried grains with solubles (DDGS), E0, E10, E15, E85, ethanol consumption, ethanol production, match blending, methyl-tert-butyl-ether (MTBE), octane value, oxygenate, Reformulated Gasoline (RFG) Program, Renewable Identification Number (RIN), splash blending, Volumetric Ethanol Excise Tax Credit (VEETC).

6.1 Introduction

This chapter discusses the effects of the RFS Program on historical production and consumption of corn ethanol and corn. These estimates of attribution are then used in Part 3 chapters to guide the assessment of the impacts to date of the RFS Program on environmental and resource conservation effects. An assessment of attribution may be a retrospective or prospective undertaking. The focus of this chapter is on the past, to lay the foundation for a better understanding of the effect of RFS Program on the environment. In order to differentiate effects attributable to the RFS Program from effects attributable to other factors, the RFS Program must be examined in the context of the many factors that may affect ethanol production and consumption in the United States. These include other federal and state policies, economic considerations, and infrastructure, to name a few. Section 6.2 reports the historical trends for major factors affecting ethanol production and consumption in the United States as context. The subsequent sections discuss evidence of effects of the RFS Program on the production and consumption of corn ethanol (6.3) and its feedstock corn (6.4) historically. Section 6.5 briefly summarizes the likely future effects of the RFS Program as estimated in the Final Set Rule.[2] Section 6.6 then presents conclusions from this material. Supporting information and additional details are in Appendix C.

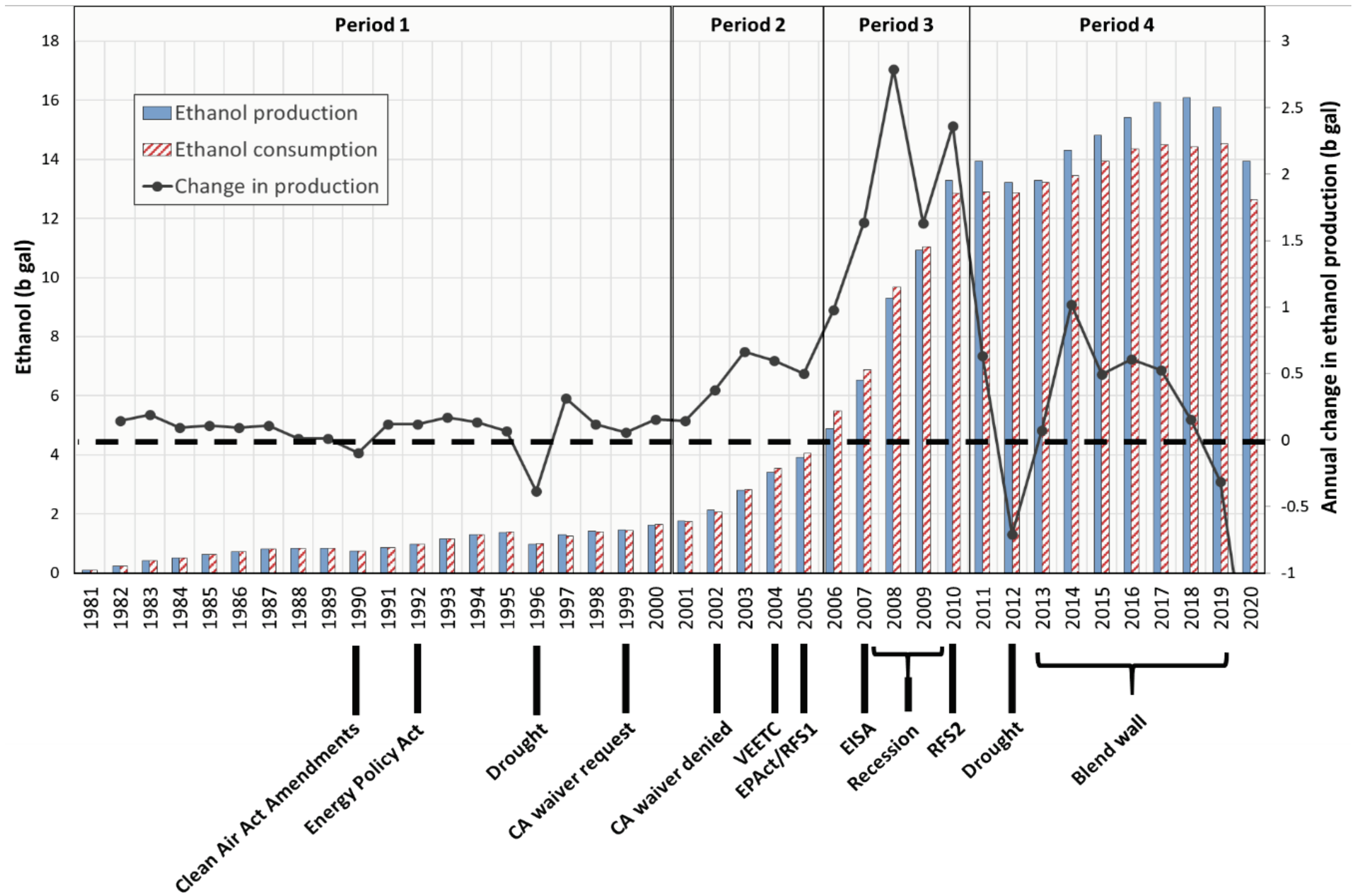
6.2 Historical Trends and Factors Potentially Affecting Corn Ethanol Production and Consumption in the United States

Ethanol as a component of transportation fuel has a long history in the United States. Beginning with a Clean Air Act waiver in 1978, ethanol was permitted to be blended into gasoline (Duffield et al., 2015). Examining the growth of the industry through time, many experts have noted distinct periods in its evolution (Taheripour et al., 2022a; Duffield et al., 2015; Dirks et al., 2012). Similar to these, the timeline is divided into four periods linked with changes in the annual rate of growth of the industry: (1) 1980–2000, (2) 2001–2005, (3) 2006–2010, and (4) 2011–2019 (Figure 6.1).

6.2.1 Period 1: 1980–2000

From 1980 to 2000, there was a slow increase in ethanol production and consumption in the United States with annual increases in both averaging roughly 80 million gallons per year over this period (Figure 6.1). Many pieces of legislation spurred ethanol production and consumption in the United States over this period (Duffield et al., 2015) (Table 6.1). A major update to the Clean Air Act occurred with the Clean Air Act Amendments of 1990 (CAAA), which established the Oxygenated Fuels Program and the Reformulated Gasoline (RFG) Program to control carbon monoxide and ozone, respectively, in areas around the country that were judged to be in non-attainment.[3] At the time, methyl-tertiary-butyl-ether (MTBE), which is a fossil fuel product, was the preferred oxygenate because it was less expensive than

[2] See docket. EPA-HQ-OAR-2021-0427.
[3] Non-attainment means that the area in question does not meet federal air quality standards for a particular pollutant.



b gal = billion gallons

Figure 6.1. Annual production and consumption of ethanol in the United States from 1981 to 2020 (left axis, blue and red-hatched bars, respectively) and the change in production from one year to the next (right axis and solid line, dashed line at zero change). Data from EIA in billions of gallons.[4] The vertical lines separate periods that coincide with different rates of growth in the industry, and key events discussed in the text are highlighted below the timeline.

[4] Downloaded 9/9/2020 from https://www.eia.gov/totalenergy/data/monthly/index.php#renewable.

Table 6.1. Summary of major legislation related to ethanol from 1978-2000 (modified from Duffield et al., 2015).

Title of Legislation	Description
National Energy Act of 1978	The first major piece of legislation related to ethanol that gave ethanol blends of at least 10% a $0.40/gallon exemption from the federal motor fuels tax. Due to changes in excise taxes on motor fuels in 1983, the tax exemption for ethanol increased to $0.50/gallon.
Energy Security Act of 1980	Offered insured loans to small ethanol plants producing less than 1 million gallons per year. The U.S. Secretaries of Agriculture and Energy were ordered to prepare a plan that would increase ethanol production to at least 10% of total gasoline supply by the end of 1990.
Crude Oil Windfall Profit Tax Act (1980)	Extended the motor fuels tax exemption through 1992 and provided blenders the option of receiving the same tax benefits by using an income tax credit instead of the fuel tax exemption.
Omnibus Reconciliation Act of 1980	Established a 2.5% ad valorem tariff and an import duty on ethanol of $0.54/gallon.
Caribbean Basin Initiative (1983)	Shortly after Congress first adopted the motor fuel tax credit, it also enacted a duty on fuel ethanol imports to offset the value of the federal tax exemption, so foreign ethanol producers could not benefit from the exemption. Duty-free treatment for ethanol was granted to 22 Caribbean Basin countries and territories in January 1984, under the Caribbean Basin Initiative.
Deficit Reduction Act of 1984	The ethanol tax exemption and blenders income tax credit were raised to $0.60/gallon.
Alternative Motor Fuels Act (1988)	Provides credits to automakers towards meeting their corporate average fuel efficiency (CAFE) standards for manufacturing alternative-fueled vehicles, including flex-fuel vehicles (FFVs) capable of running on E85.
Omnibus Budget Reconciliation Act (1990)	Lowered the ethanol tax exemption and blenders income tax credit to $0.54/gallon. The expiration date for the new tax rates was extended to 2002. The Act also provided a $0.10/gallon payment to small ethanol producers with a capacity of 30 million gallons or less. Producers could receive the tax credit up to 15 million gallons of production annually.
Clean Air Act Amendments of 1990 (CAAA)	Provisions of the CAAA established the Oxygenated Fuels Program and the Reformulated Gasoline (RFG) Program to control carbon monoxide and ozone problems in certain urban areas around the country. The Oxygenated Fuels Program required gasoline to contain 2.7 weight percent oxygen (equivalent to 7.7 volume percent ethanol) in its covered cities. The RFG Program required gasoline to contain 2.0 weight percent oxygen (equivalent to 5.7 volume percent ethanol) in its covered cities. While most of the market utilized MTBE to meet the oxygenate requirements, ethanol was also often used at concentrations up to 10 volume percent.
Energy Policy Act of 1992 (EPAct)	EPACT extended the fuel tax exemption and the blenders' income tax credit to two additional blend rates containing less than 10% ethanol, effective January 1, 1993 (National Agricultural Law Center). The two additional blend rates were for gasoline with at least 7.7% ethanol and for gasoline with 5.7% ethanol. These additional blends were added to encourage blending of ethanol to make oxygenated gasoline in the Oxygenated Fuels Program, requiring 7.7% ethanol, and in the Reformulated Gasoline (RFG) Program, which requires 5.7% ethanol. This Act also required federal agencies to purchase a certain percentage of alternative-fuel vehicles, including FFVs.
Transportation Equity Act for the 21st Century (1998)	Reduced the ethanol tax exemption and blenders' income tax credit to $0.53 starting January 2001, reducing it further to $0.52 in January 2003 and to $0.51 in January 2005. Both tax credits were extended to the end of 2007.
California Banned MTBE (1999)	MTBE was banned in California at the earliest possible date, but no later than December 31, 2002. This date was amended in March 2002, to December 31, 2003. Following California's lead, at least 24 other states also banned MTBE, allowing ethanol to become the dominate fuel in the oxygenate market.

ethanol on a volumetric basis, could be shipped in existing pipelines, and had no impact on the Reid Vapor Pressure (RVP)[5] of gasoline (Duffield et al., 2015; California Energy Commission, 1999; U.S. EPA, 1999). To help ethanol compete with MTBE, the Energy Policy Act of 1992 extended the fuel tax exemption and the blenders' income tax credit,[6] which encouraged blending of ethanol to make oxygenated gasoline in the Oxygenated Fuels Program. At the end of this period there was growing concern about the environmental effects of MTBE mainly in the context of groundwater contamination resulting from leaking underground storage tanks. The California Air Resources Board (CARB) made a formal request to EPA in 1999 for a waiver from the requirement to use oxygenates in reformulated gasoline. The governor of California issued an executive order in March 1999 to ban MTBE in the state's gasoline by the end of 2002; and, by 2000, the replacement of MTBE with ethanol was underway in California (Anderson and Elzinga, 2014; GAO, 2002). Although the CARB request to EPA was ultimately denied in 2001, it was in the context of consideration of that waiver request that the EPA announced in 2000 that it intended to impose a nationwide ban on the use of MTBE in gasoline.[7,8] The replacement of MTBE in the gasoline pool, though beginning in 1999–2002 administratively and legislatively in California and other states, would not actually occur in the gasoline pool until 2003 in California and until after the passage of the EPAct in 2005 across the rest of the country (discussed in the next section). At the end of 2000, the concentration of ethanol in gasoline was 1.27% (Figure 6.2), mostly from ethanol blending in the Midwest where corn ethanol was a preferred oxygenate because of abundant corn, co-located biorefineries, and fewer fuel transport barriers due to the proximity of producers and consumers (Duffield et al., 2015).

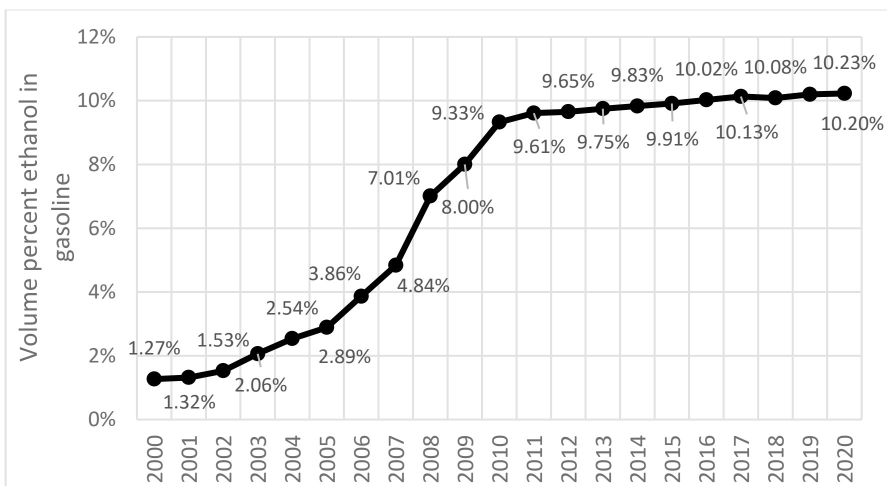


Figure 6.2. Ethanol concentration in consumed gasoline. Source: EIA Monthly Energy Review, Tables 10.3 (Ethanol in thousand barrels) and 3.5 (Gasoline in thousand barrels per day).

5 The RVP describes the volatility of gasoline to evaporative emissions.
6 The blenders' credit was updated in different legislations and had different values and forms in different periods, being $0.60 per gallon from 1984 to 1990, $0.51–$0.54 through 2008, and $0.45 from 2009 to 2011 (USDA ERS, Bioenergy statistics Table 15, https://www.ers.usda.gov/data-products/us-bioenergy-statistics/).
7 "Clinton-Gore Administration Acts To Eliminate MTBE, Boost Ethanol," EPA News Release, March 20, 2000.
8 "Advance Notice of Proposed Rulemaking to Control MTBE in Gasoline," EPA Regulatory Announcement EPA420-F-00-012, March 2000. This action did not ultimately become a final rule.

6.2.2 *Period 2: 2001–2005*

From 2001 to 2005, domestic ethanol production increased from 1.8 to 3.9 billion gallons per year, for an average rate of increase of roughly 450 million gallons per year (Figure 6.1). This rate was over five times the average annual rate of increase from 1980 to 2000. Ethanol increased as a percentage of gasoline in the fuel supply from 1.3% to 2.9% (Figure 6.2), mostly in areas where it was already in use, like the Midwest (Duffield et al., 2015) and in California, which saw a sharp decline in MTBE at the end of 2002 and the end of 2003 (Anderson and Elzinga, 2014) (Figure 6.3, PADD 5).[9,10] By the end of July 2005, before the passage of the Energy Policy Act in August of that year, 17 states had some form of partial or complete ban on MTBE use (Duffield et al., 2015; U.S. EPA, 2007). These states represented 41% of domestic gasoline consumption in 2005. At the federal level, multiple bills banning MTBE were considered by Congress, but none were ultimately adopted.[11] At the same time, Congress also considered providing liability protection for refiners using MTBE under the premise that refiners had no choice but to use an oxygenate in the RFG and Oxyfuels Programs, and that the EPA had implicitly approved MTBE's use inasmuch as EPA knew MTBE was a primary option when the RFG Program was originally implemented.[12,13] The potential for some sort of liability protection, as well as the lack of sufficient infrastructure for distributing and blending ethanol to coastal urban areas during this period (Duffield et al., 2015), may have encouraged refiners to continue producing and using MTBE despite state bans and concerns expressed by the EPA and the public.

Around this time many substitutes were considered for replacing MTBE, renewable and non-renewable, and even the elimination of requirements for oxygenates altogether. The California Energy Commission (CEC) published a report in 1999 examining several possible substitutes for MTBE, including ethanol, tertiary-butyl-alcohol (TBA), ethyl-tertiary-butyl-ether (ETBE), and tertiary-amyl-methyl-ether (TAME) (California Energy Commission, 1999).[14] The substitutes considered in the CEC

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[9] The California ban was originally scheduled to go into effect December 31, 2002, but was extended by one year to give industry more time. Some companies converted by the original timeline while others converted under the new timeline. Summary here: https://www.icis.com/explore/resources/news/2006/07/05/1070674/timeline-a-very-short-history-of-mtbe-in-the-us/.

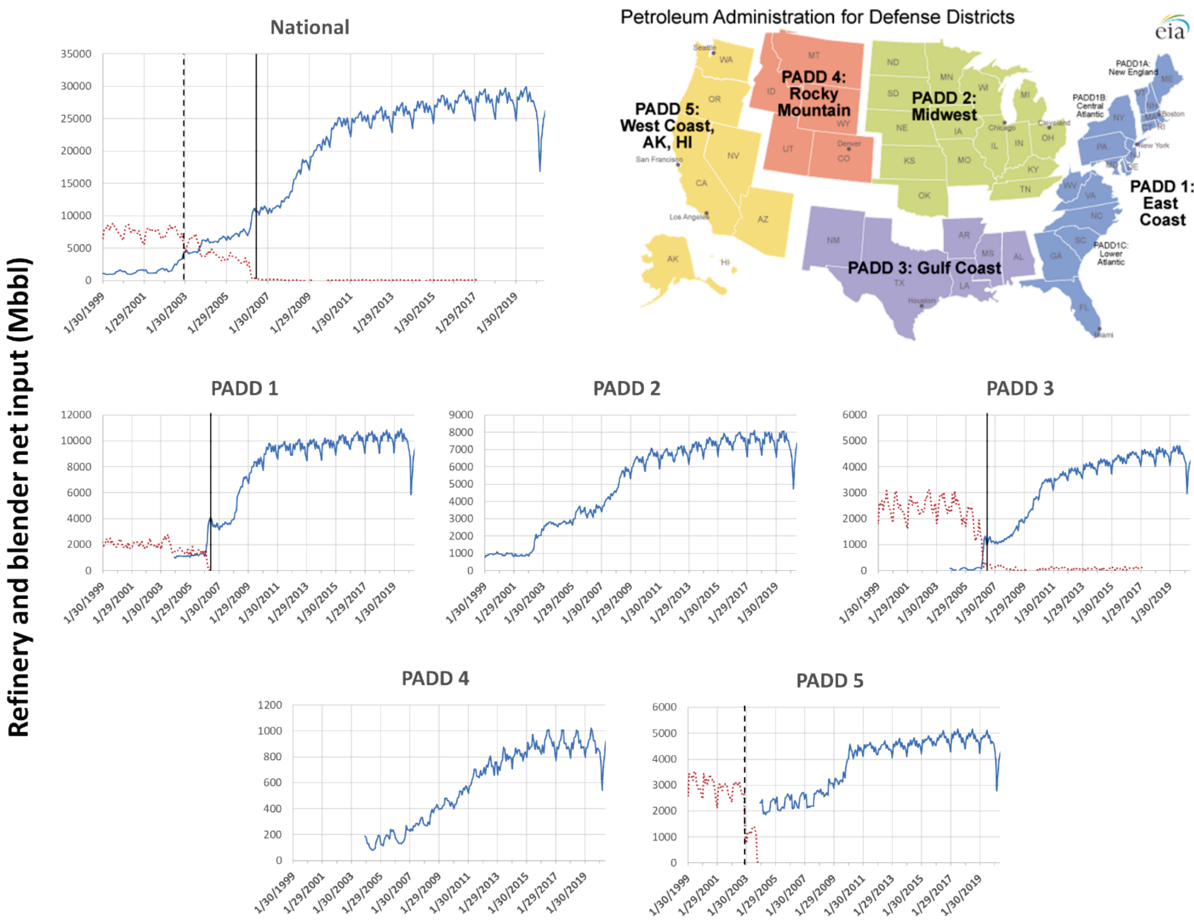
[10] The Petroleum Administration for Defense Districts (PADDs) are geographic aggregations of the 50 States and the District of Columbia into five districts (Figure 6). Originally for rationing purposes during World War II, the districts are now used for analysis, data collection, and monitoring (https://www.eia.gov/todayinenergy/detail.php?id=4890#).

[11] For example, Section 833 of the Energy Policy Act of 2002 (not signed into law) stated that "Congress has reconsidered the relative value of MTBE and decided to eliminate use of MTBE as a fuel additive."

[12] See discussion of liability protection in "CRS Report for Congress - Renewable Fuels and MTBE"

[13] Within Section 833 of the Energy Policy Act of 2002 there was acknowledgement that Congress was aware of the potential for significant use of MTBE to meet the fuel oxygen standard and the potential consequences on water quality.

[14] The reason for reviewing this older period is to understand whether non-ethanol substitutes for MTBE may have emerged in the absence of the RFS Program.



Mbbl = million barrels

Figure 6.3. Monthly volume of MTBE (red, dotted line) and ethanol (blue, solid line) blended by refineries nationally and by PADD from 1993 to 2020. Dashed vertical line is the original date of the California state ban (December 31, 2002; National and PADD 5 panels), and the solid vertical lines were the dates when MTBE was phased out in the EPAct (May 6, 2006; National and PADD 1 and 3 panels). Note y-axes differ, MTBE was not blended in PADD 2 or 4. Data from EIA, https://www.eia.gov/dnav/pet/pet_pnp_inpt_dc_r50_mbbl_m.htm, thousands of barrels.

report were used as fuel additives at the time in varying amounts to increase the octane[15] of gasoline and/or meet the oxygenate requirements. MTBE was the dominant additive nationally as an oxygenate and octane enhancer because of its lower cost and compatible blending properties, although other materials were used at the time regionally, like ethanol in the Midwest (Duffield et al., 2015). The CEC report examined three timelines for replacement of MTBE: 1-year, 3-years, and 6-years, with ethanol, TBA, ETBE, TAME, or a mixture. The report found that only ethanol was available in sufficient quantities for the 1-year timeline, but that such a rapid new demand from California would likely disrupt national ethanol markets and increase prices significantly. Modifications to MTBE plants to produce

[15] In this chapter "octane" refers to "octane rating," rather than the molecule octane (i.e., C_8H_{18}). The octane rating describes the fuel's ability to resist auto-ignition, which can cause engine knocking. It is most typically presented as the (R+M)/2 value, the average of the research (R) and motor (M) octane numbers.

ETBE or TBA were estimated to take 12–24 months. Thus, ETBE, TBA, and ethanol were all projected to be available in sufficient quantities under the 3- and 6-year timelines. TAME was not estimated to be available at sufficient quantities under any scenario unless mixed with other oxygenates. Under the 3- and 6-year timelines, the cost increases to gasoline for replacing MTBE with ethanol were higher (+1.9 to 6.7 ¢/gal) than for ETBE (+0 to 2.5 ¢/gal) or TBA (+0.3 to 1.4 ¢/gal). However, the same water quality issues associated with MTBE were thought to be a potential concern for ETBE and TBA (California Energy Commission, 1999; U.S. EPA, 1999). Recent studies have verified these and other potential risks from ETBE, TAME, and TBA (Dietrich and Burlingame, 2020; van Wezel et al., 2009; Fischer, 2003).

An EPA Blue Ribbon Report in 1999 also examined several options, including all the substitute oxygenates in the CEC report, plus no oxygenates at all, as well as "Other Alternatives" like alkylates, reformate, aromatics, and others (U.S. EPA, 1999). The Blue Ribbon Report came up with some similar conclusions, but also suggested alkylates as a viable alternative, along with the removal of the 2% oxygen requirement established by the CAA for RFG areas. When EPA declined the request by CARB to waive the oxygenate requirement in 2002, and with the CAA requirements still in place and the original governor's deadline looming in December 31, 2002, the replacement of MTBE with ethanol began in earnest in California (Figure 6.3, PADD 5). This is also visible in the increase in the price of ethanol from mid-1999 to mid-2001 (Figure 6.4) as predicted by the CEC report with California refineries having to outbid Midwestern blenders to acquire ethanol from refineries because of the state ban on MTBE that was originally scheduled to go into effect at the end of 2002.

Oil prices have complex and important associations with many kinds of economic activity, including gasoline, ethanol, and corn production (Babcock, 2013; Tyner et al., 2010).[16] Oil prices, which had been low from 1990 to 2003 ($20–50/barrel, Figure 6.4), began to increase during this time, reaching levels that had not been seen in years toward the end of this period (e.g., above $69/barrel by 2005 in 2018-adjusted dollars). Furthermore, from 2003 through 2006 the price of oil was increasing, while the price of corn was relatively stable (Figure 6.4). This has implications for the economics of ethanol as a blend in gasoline and likely played a role in the phase out of MTBE (U.S. EPA, 2022). Since ethanol is blended with gasoline to make E10, it becomes less expensive to make a given volume of gasoline with ethanol than gasoline without ethanol as gasoline prices increase relative to ethanol.

[16] Often oil prices are presented as opposed to gasoline prices for convenience. Although gasoline without ethanol (i.e., "E0") is the substitute for ethanol in the market when blending E10, gasoline prices track oil prices very closely (see Appendix C). Furthermore, whereas there are representative oil prices (e.g., Cushing, West Texas Intermediate, Figure 6.4), gasoline prices vary more widely across the country.

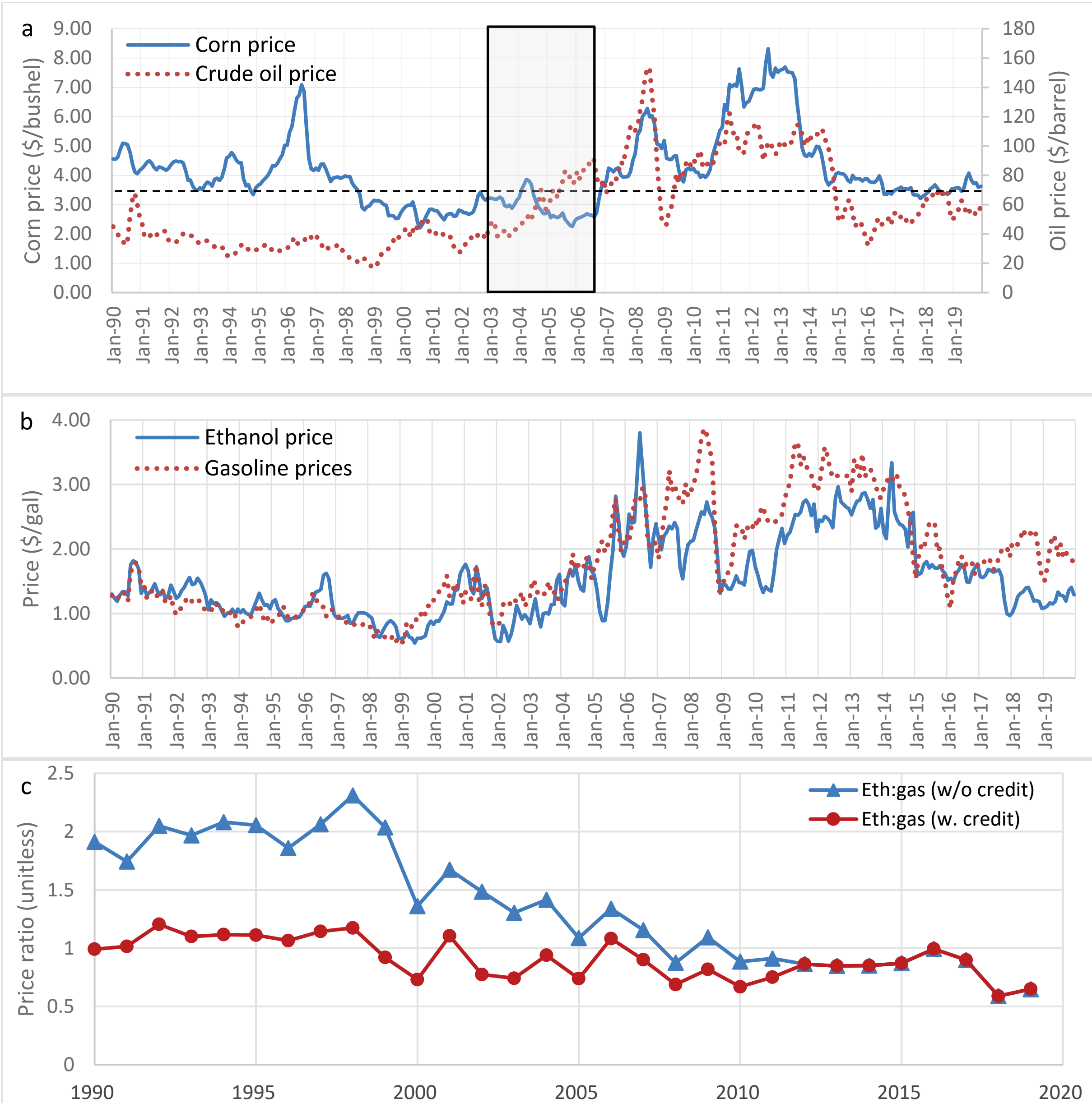


Figure 6.4. Monthly prices from 1990 to 2019 for feedstocks (a: corn and crude oil, left and right axes, respectively) and refinery products (b: ethanol and gasoline). Shown in (c) is the ratio of annual ethanol to gasoline price (ethanol/gas) with and without the blenders' tax credit through time (expired at the end of 2011, shown are market years identified by ending year). Ethanol prices in (b) include the blenders' credit. Prices for corn, ethanol, and gasoline from USDA ERS Biostatistics, Table 14, downloaded 9/9/2020. Prices for crude oil from EIA spot prices for Cushing, OK West Texas Intermediate (WTI) Spot Price FOB, downloaded 9/9/2020). In (a), added for reference is a box for January 2003 to November 2006, and a dashed line for the estimated break-even point from Tyner et al. (2010) ($69/barrel of oil with no blenders subsidy or RFS, updated to $2018). All prices in a and b are in real 2018 dollars.

The next major federal policy in this period was the Volumetric Ethanol Excise Tax Credit (VEETC) in the Jobs Creation Act (JCA), which was signed into law October 22, 2004. VEETC changed the form of the tax subsidy from an excise tax exemption to a tax credit, and extended the credit to ethanol use in concentrations higher than 10%. These changes provided revenue to the ethanol producers and allowed the tax credit to be claimed for ethanol used in higher level blends (Duffield et al., 2015). The JCA also extended the expiration date of the tax credit from 2007 to 2010, which was extended again and eventually expired at the end of 2011. As discussed earlier, an ethanol tax credit of one form or another makes gasoline with ethanol more competitive than it would be without the credit (Figure 6.4c). Even so, it was not until after 2000 when oil prices began to increase that ethanol was cost-competitive for several years in a row, and was even cost-competitive without the credit in 2008 and after 2010 (Figure 6.4c). This would have potential implications on infrastructure buildout (discussed in the next section).

The passage of the Energy Policy Act (EPAct) of 2005, which included the RFS1 along with many other provisions, was signed into law on August 8, 2005, and effectively (though not by mandate) ended all use of MTBE in U.S. gasoline. Included in the EPAct, though not a part of the RFS Program, was the elimination of the oxygen requirement in RFG areas. Even though the EPAct went into effect in 2005, the first year in which the volume requirements of the RFS Program applied was 2006. Thus, 2006 is the first year that the RFS Program per se could have a direct material effect. Although the EPAct did not include a nationwide ban on the use of MTBE as had previous bills that Congress considered, neither did it include any form of liability protection that had been sought after by refiners who blended MTBE into gasoline. Instead, the EPAct eliminated the oxygen requirement for federal RFG and created the RFS Program (RFS1). Although the oxygen requirement was removed, the emission standards under the CAA were neither eliminated nor modified, and the use of an oxygenate continued to be the most economical way to meet those emission standards. EPA batch data shows no change in oxygenate use in RFG areas despite the removal of the 2% oxygenate requirement (see Appendix C, Figure C.17). Other substitutes for MTBE either were associated with similar water quality concerns as MTBE (e.g., ETBE, TBA, TAME), or were aromatics that did not satisfy the emission requirements for RFG under the CAA. The combination of these changes in the EPAct, in addition to the lack of any explicit or implicit liability protection, meant that refiners had little incentive to continue using MTBE and significant incentive to use ethanol. The result was that MTBE use in the remaining federal RFG areas outside of California dropped by nearly 80% between 2005 and 2006 (Figures 6.3 and 6.5). Demand from RFG areas constituted 34% of all gasoline nationwide because of the larger populations in the often coastal areas covered by RFG.[17] Transporting ethanol to meet demand in RFG areas thus needed to overcome logistical limitations (Duffield et al., 2015) (discussed in more detail in the next section). MTBE was almost gone from all gasoline by May 2006[18] and replaced by ethanol, the first year of actual volumes under the RFS1 (Chapter 1, Table 1.1). This transition is observable in the conventional gasoline pool (CG[19]) as well, where ethanol was redirected from CG areas likely in the Midwest to non-CA RFG areas from 2005 to 2006 (Figure 6.5) in order to make up for the shortfall in supply. Afterwards, ethanol in the CG pool rebounded as demand in RFG areas was satisfied and ethanol production continued to increase (Figure 6.5). At the end of this period the concentration of ethanol in the gasoline pool was 2.9% (Figure 6.3).

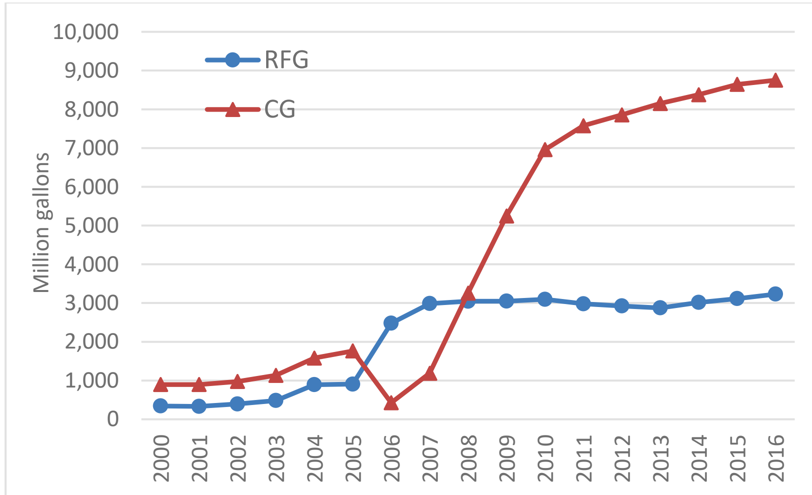


Figure 6.5. Consumption of ethanol in reformulated gasoline (RFG) and conventional gasoline (CG) outside of California. Source: EPA batch report data (required under 40 CFR 80.75 and 80.105. See https://www.epa.gov/fuels-registration-reporting-and-compliance-help/public-data-gasoline-fuel-quality-properties).

6.2.3 Period 3: 2006–2010

This is the period of most dramatic growth in the production and use of ethanol in the United States, from roughly 5 billion gallons produced in 2006 to 13 billion gallons in 2010, an annual increase averaging 1.9 billion gallons per year (Figure 6.1). By the end of 2006—the first year of the RFS1—the percentage of ethanol in gasoline was 3.9% (Figure 6.2). Oil prices continued to increase (Figure 6.4a), reaching record levels in 2008, which then crashed with the 2009 recession, recovering at levels that were still historically high in 2010 (~$80/barrel). There was a large buildout in ethanol production capacity beginning in 2006 and peaking in 2007–2008 (Figure 6.6), corresponding with the historically high corn and oil prices that likely influenced the economics of ethanol blending (see sections 6.3.3 and 6.3.5).

[17] In 2005, petroleum supply made up 1,132,692 thousand barrels in RFG areas as opposed to 2,210,440 thousand barrels in CG areas. Thus, RFG represented 34% of total in 2005 (Petroleum Supply Annual, Excel file: "U.S. Supply and Disposition", Worksheet "Data 5 - Finished Products" https://www.eia.gov/dnav/pet/pet_sum_snd_d_nus_mbbl_m_cur.htm.)
[18] EPAct included an effective date for removal of the 2% oxygen requirement at 270 days after enactment (i.e., May 5, 2006). EPA finalized the rule removing the 2% oxygen requirement on May 8, 2006, coinciding with the large drop in Figure 6.5.
[19] Conventional gasoline areas are effectively areas not in the RFG Program.

Even as early as 2007, the same year that the Energy Independence and Security Act (EISA) passed in December, total ethanol capacity in operation and under construction was already 12 billion gallons, which increased to roughly 13.4 billion gallons by 2010—the first full year of the RFS2 (Figure 6.6). Given that EISA passed in December, it is unlikely that refinery construction in 2007 was due to the RFS Program, though construction in 2008 was still substantial and certainly could be. Many additional state-level policies were enacted in this period, including ethanol mandates (e.g., HI, OR, MO, WA) and the Low Carbon Fuel Standard (LCFS) in CA (Duffield et al., 2015). The large increase in ethanol use in federal RFG in summer 2006 due to the replacement of MTBE may have contributed to the large increase in corn price that began in the winter of 2006 (Figure 6.7). Corn prices had been roughly $2 per bushel for many years,[20] and increased to $3.50 in the winter of 2006.

Important infrastructure changes were also taking place during this time interval. USDA reports demonstrate that once it became clearer that MTBE was likely to be replaced with ethanol, infrastructure quickly developed to distribute ethanol to new markets (Duffield et al., 2015; Denicoff, 2007). This included increases in rail tank cars, unit trains, blending terminals, and hubs for ethanol storage (Duffield et al., 2015; Denicoff, 2007). As discussed in Chapter 3, most ethanol is transported by rail, and orders for

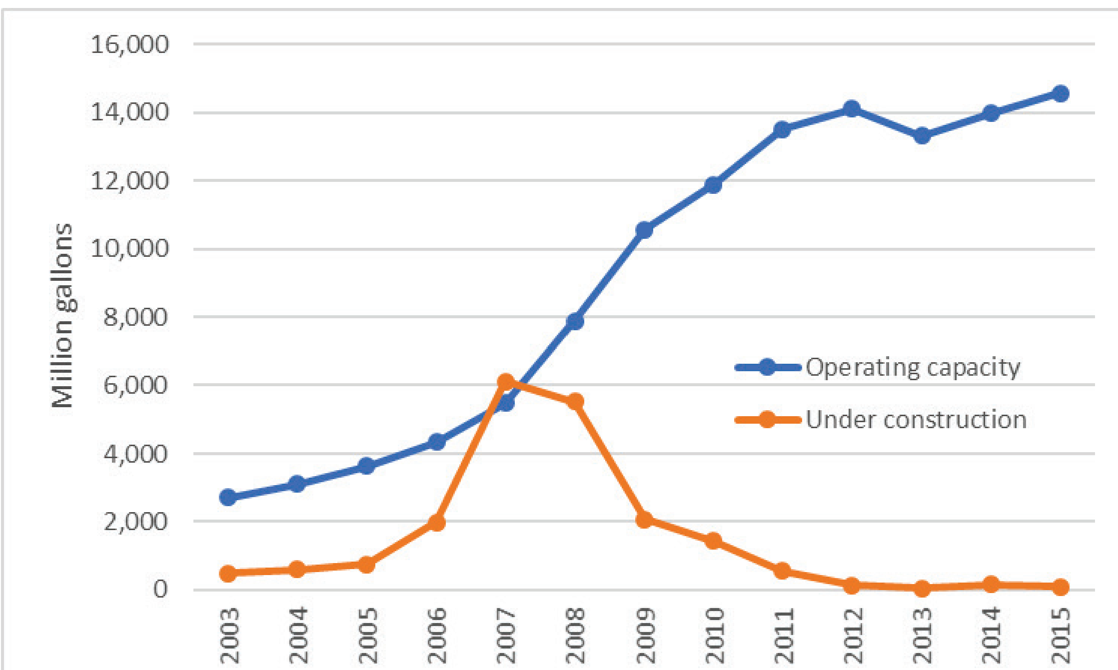


Figure 6.6. Corn ethanol production capacity in operation and under construction from 2003 to 2015. Source: Renewable Fuel Association's annual "Ethanol Industry Outlook," https://ethanolrfa.org/publications/outlook/. There is no parallel government dataset to the authors' knowledge.

[20] Monthly corn prices received by farmers varied from roughly $1.50 to $3.00 per bushel from January 2000 (first month of the dataset) to August 2006, averaging $2.10 per bushel.

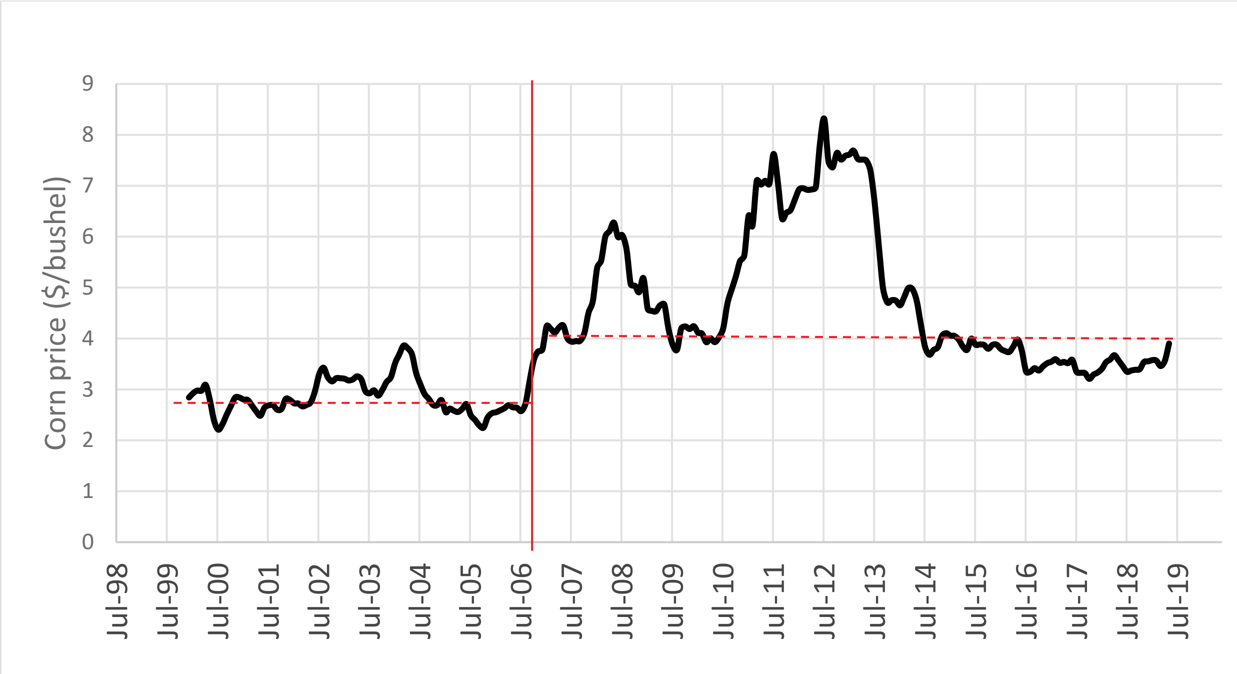


Figure 6.7. Monthly prices (in real 2018 dollars per bushel) received by farmers in the United States from 1990 to 2019.[21] November 2006 is shown for reference (vertical red line), along with the historical prices of $2.75 and $4.00 (horizontal dashed red lines).

new rail tank cars, 75% of which were estimated to be for ethanol use, started to increase in 2005 and continued to increase through 2006, creating a substantial backlog (Figure 6.8) (Denicoff, 2007). Similar increases for "Jumbo Hopper Cars"[22] that transport distiller's dried grains with solubles (DDGS) were reported over the same period (Denicoff, 2007). A dedicated ethanol pipeline was also considered, but was determined to not be economically competitive with other options without increasing to E15, more consumption of E85, or significant

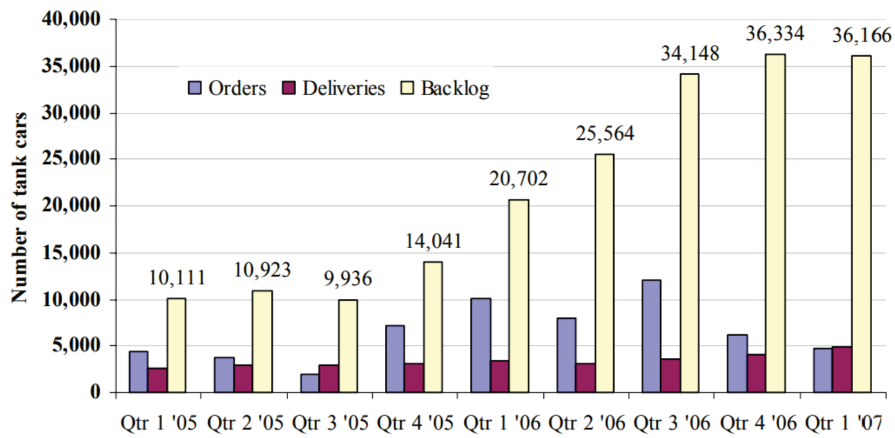


Figure 6.8. New rail tank car orders, deliveries, and backlog (from Denicoff (2007) citing monthly reports from the Rail Supply Institute)**.**

[21] Data are from the USDA ERS, specifying the "Prices received by farmers" for "Corn grain" on a "Monthly" basis from "2000-2019". Source: https://data.ers.usda.gov/FEED-GRAINS-custom-query.aspx#.
[22] Jumbo hopper cars have wider openings than standard hopper cars that are better suited for DDGS, which tend to cake and bridge between particles.

additional incentives (Duffield et al., 2015; DOE, 2010).[23] Since rail tank cars and trucks had already been transporting ethanol for several years by 2010, they became the preferred mode that remains to this day (Duffield et al., 2015).

Additional important infrastructure changes were taking place at the refinery as well during this period. In areas where CG was sold, oil refineries transitioned between approximately 2005 and 2010 from producing "finished gasoline," which could be sold at a retail station (e.g., 87 octane gasoline) or mixed with oxygenates to make higher octane blends, to producing "unfinished gasoline," which needed to be "match blended" with an oxygenate to be legally sold at a retail station (Duffield et al., 2015) (See Box 6.1. Blending 101). This had been occurring in RFG areas for years, but the practice expanded to refineries supplying the CG pool. With match blending, refineries could produce cheaper unfinished gasoline—called a Blendstock for Oxygenate Blending, or BOB, often at 84 octane—which would then be mixed with ethanol at the terminal to raise the octane value to 87 or higher. The cost of production of a BOB was lower than the cost of finished gasoline because of lower refining necessary, which allowed refiners producing unfinished gasoline to reduce the price of the gasoline they produced. Once investments had been made to convert refineries to match blending, it would be costly to revert back to production of finished gasoline. It is unclear when precisely transitions to match blending occurred, but trade groups suggest it was roughly between 2005 and 2010 in most areas.

Box 6.1. Blending 101

Here finished gasoline means gasoline that can be sold to at a retail station. Unfinished gasoline cannot be legally sold at a retail station until additional processing occurs, which in this case is the addition of an oxygenate like ethanol. Match blending is the process by which a lower oxygenate blendstock (called a Blendstock for Oxygenate Blending or BOB) is mixed with an oxygenate to meet the CAAA emissions requirements and to increase the octane value to 87 in order to be legally sold. BOBs are less expensive to make than finished gasoline because of lower refining. Under match blending, the terminal operator would mix the cheaper 84 octane BOB with ethanol: 90% x 84 octane BOB + 10% 115 octane ethanol = E10 at 87 octane. Match blending is differentiated from "splash blending," which occurred first in the industry. Under splash blending, the retail station would mix the more expensive 87 octane finished gasoline with ethanol: 90% x 87 octane gasoline + 10% x 115 octane ethanol = E10 at 90 octane. Thus, splash blended E10 is more expensive than match blended E10 because the finished gasoline is more expensive to make than the BOB.

EISA was enacted on December 19, 2007, and replaced the RFS1 with the RFS2. EISA contained volume requirements for four nested categories of biofuel: cellulosic biofuel, biomass-based diesel, advanced biofuel, and total renewable fuel (see Chapter 1, Table 1.1, Figure 1.2). These are tracked through Renewable Identification Numbers (RINs), which are created by biorefineries, and may or may not be traded (McPhail et al., 2011). RINs are ultimately retired by obligated parties each year to show

[23] The 2010 DOE Report to Congress for a dedicated pipeline between the East Coast and the Midwest concluded that the expected ethanol demand (2.8 b gallons per year) was well below the demand required (4.1 b gallons per year) for the pipeline to be economically viable. At the estimated demand, the pipeline would have to charge an average tariff of $0.28/gal, which was 47% higher than the average cost across other modes of transport ($0.19/gal, on average for rail, barge, and truck).

that the fraction of renewable fuel blended into the domestic gasoline pool meets the RFS2 mandates on a party-by-party basis.[24] The industry had been producing more ethanol than was mandated in all years of the RFS1, leading to an accumulation of banked RINs (see Appendix C). Detailed information on the magnitude of banked RINs is lacking in these early years from 2006 to 2010, but given that there were more than 2.5 billion carryover RINs in 2011 (the first year of records, mostly D6 RINs) and that ethanol production exceeded the RFS1 mandates for every year, there was likely an excess of RINs for the entire period.[25] Total volumes of renewable fuel required by the RFS2 were much higher than those required by the RFS1 (Table 1.1). Because EISA was not passed until December 2007, the annual mandates in 2008 were still set by the RFS1. The RFS2 went into effect in 2009, but only for total renewable fuel; thus, it was not until 2010 that the RFS2 with the four volumetric standards went into effect. By then the percentage of ethanol in gasoline was already at 9.3%, near the "E10 blend wall."

The E10 blend wall is a term for the amount of ethanol that can be blended into gasoline if every gallon of gasoline contains 10% ethanol. Thus, it is a function of the total amount of gasoline consumed, which changes as vehicle fuel efficiencies increase and people's driving habits change. Ethanol consumption can increase beyond the E10 blend wall through blending of higher level ethanol blends such as E15 and E85; however, those face greater logistical and economic challenges. Higher consumption of E15 has been limited in the past by availability of retail stations that sell E15, legal concerns regarding liability, and challenges related to using higher ethanol blends in the summer months in CG areas,[26] among other factors (Duffield et al., 2015). Higher consumption of E85 has been limited in the past by limited sales of flex-fuel vehicles (FFVs), consumer choice to refuel with E10 rather than E85,[27] and potentially the nested structure of the RFS Program among other factors (Zhong and Khanna, 2022; Duffield et al., 2015). Thus, historically the E10 blend wall has represented a challenge to increased domestic consumption of ethanol, but it does not directly limit production or exports. Although the EIA announced the United States had reached the blend wall nationally in May 2016,[28] examining Figure 6.2 demonstrates that the United States was close (e.g., ethanol > 9% of gasoline) as early as 2010. By the end of this period, the concentration of ethanol in the gasoline pool was approximately 9.3% (Figure 6.2).

[24] RINs existed under the RFS1 but were not tracked digitally by EPA nor differentiated by renewable fuel type.
[25] Each D6 RIN represents one gallon of conventional biofuel, which for the most part is corn ethanol in the United States (see Chapter 1 and 2). So 2.5 billion D6 RINs represent 2.5 billion gallons of ethanol, which could be blended to produce 25 billion gallons of E10 gasoline. So large backlogs of carryover RINs affect the potential binding effect of the RFS Program in any given year. RINs may be carried over one year for compliance.
[26] This was because the 1psi RVP waiver for E10 only explicitly applied to E10. This extension to E15 was not granted until June 2019. It was later revoked and reinstated on an emergency basis.
[27] See Chapter 1 section 1.3.1 for a brief discussion on this consumer choice.
[28] https://www.eia.gov/todayinenergy/detail.php?id=26092

6.2.4 Period 4: 2011–2019

Annual growth in production of ethanol dramatically decreased from an average of 1.9 billion gallons per year from 2006 to 2010 to 275 million gallons per year from 2011 to 2019 (Figure 6.1). This occurred even though the RFS2 standards for the four renewable fuels were fully in effect, and the RFS-implied volume requirements for conventional biofuel increased through 2015. The California LCFS, enacted legislatively in 2007, went into full effect in 2011. The E10 blend wall was slowly approached over this time period, with ethanol percentages in gasoline increasing from 9.6% in 2011 to 10.2% in 2019 (Figure 6.2). This resulted in modest growth in domestic ethanol consumption in these years, with an average annual increase of just over 180 million gallons per year from 2011 to 2019. In 2012, a significant drought in the Midwest was associated with a 1.5 billion bushel reduction (12%) in corn production with impacts on ethanol production in 2012 and 2013 (Rippey, 2015).[29] Corn production recovered in 2013 and ethanol production recovered in 2014 (Figure 6.1). Exports of ethanol increased rapidly in 2010–2011, decreased with the drought in 2012–2013, and have generally increased since 2015 (Figure 6.9). Imports of ethanol, primarily sugarcane ethanol from Brazil (Table 2.1 and Chapter 16 section 16.3), increased when domestic production was lowered by the drought (2012–2013, Figure 6.9, Figure 6.1). These import levels were similar to those in 2004–2006, when domestic ethanol production was not yet fully mature to meet the growing domestic demands.[30] Because much of the growth of the industry had already occurred by the early portion of this period, the review here is less detailed. However, other factors during this period (e.g., Small Refinery Exemptions [SREs]) are still discussed where appropriate in the sections that follow.

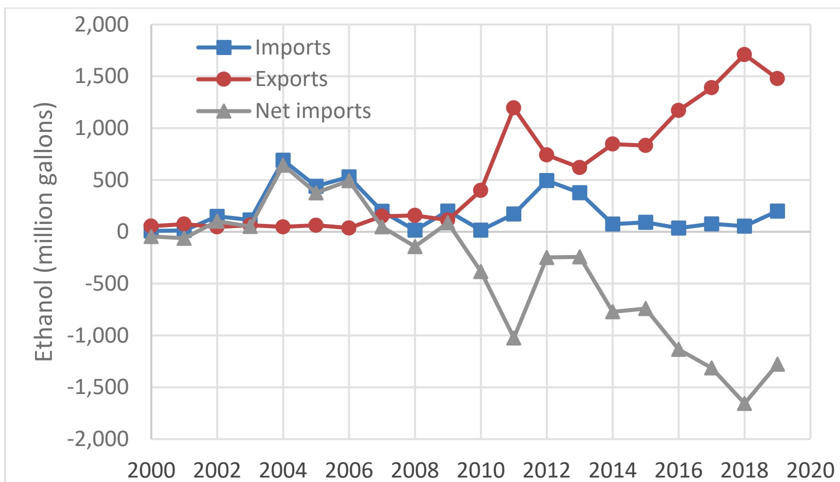


Figure 6.9. Imports, exports, and net imports of ethanol. Source: USDA ERS Bioenergy Statistics, Table 2, accessed 8/31/2020, https://www.ers.usda.gov/data-products/u-s-bioenergy-statistics/.

[29] Data from NASS, https://www.nass.usda.gov/Charts_and_Maps/Field_Crops/cornprod.php, accessed 9/30/2020.
[30] See Chapter 16 on International Effects for a discussion of trends of and potential impacts from these imports.

6.2.5 Factors Affecting Ethanol Production and Consumption in the United States

The historical record described in sections 6.2.1–6.2.4 clearly demonstrates that many factors—including the RFS Program—potentially influence the production and consumption of corn ethanol in the United States (Table 6.2). For example, the interruptions in ethanol growth trends observed in Figure 6.1 can be attributed to specific drivers. Annual change in production was negative in 1996 and 2012 due to significant droughts, and the sudden decline in growth in 2009 is attributed to the recession. The factors in Table 6.2. are not an exhaustive list. Other factors contributing to changes in corn ethanol production include land management and human behavior changes, such as urbanization, commuting practices, and dozens of agronomic factors affecting corn production. Rather, the historical record provides a key subset of factors to consider when seeking evidence from the peer-reviewed literature regarding the extent to which the RFS Program caused changes in corn ethanol production. To understand conclusions from the literature about the role of the RFS Program, how well those studies control for the multiple factors that can affect ethanol production must be understood.

6.3 Evidence of the Impact to Date of the RFS Program on Corn Ethanol Production and Consumption

Five main sources of information are used in sections 6.3.1–6.3.5 to assess the effect of the RFS Program on corn ethanol production and consumption in the United States: (1) comparison of the annual RFS mandates with consumption, (2) observation of RIN prices, (3) results from the peer-reviewed literature, (4) new analyses by the National Renewable Energy Laboratory (NREL) using the Biomass Scenario Model (Peterson et al., 2019), and (5) new analyses by EPA's Office of Transportation and Air Quality (OTAQ). These lines of evidence are discussed in turn and expanded upon in Appendix C.

Table 6.2. Some of the major factors that affect ethanol production and consumption in the United States, ordered roughly by the year of first instance. Also see Table 6.1.

Name	Description	Years in effect
Federal RFG and Oxyfuel Programs	For areas in non-attainment of O_3 (RFG) and CO (Oxyfuel), an oxygenate was required (2% for RFG and 2.7% for Oxy).	1980s (Oxy) and 1990s (RFG), to current
MTBE phaseout/ Octane demand	The loss of MTBE as an additive in gasoline created the need for a substitute to increase the octane rating.	1999–2003 to current
Oil/gas prices	Gasoline (E0) is the substitute for ethanol in the production of E10, thus as gasoline price increases relative to ethanol, blending ethanol is favorable.	All years (esp. 2006–2015)
Distribution costs	Differences in distribution costs from region to region significantly impact blend rates over time in different parts of the country. The development of unit trains and associated facilities to the East and West Coasts lowered distribution costs and increased ethanol consumption.	2010 to current
VEETC	Lowered the cost for blenders to mix ethanol into gasoline.	2004–2011
Match blending for octane value	Once the upfront capital investments were made in the gasoline production and distribution systems to switch to match blending, this factor then created a significant economic incentive to maintain E10 blending regardless of short-term economic factors and preventing any reversion back to E0.	All years (esp. 2010 to current)
RFS1	Created annual standards for renewable fuel that would have to be met by obligated parties through submitting RINs.	2006–2008
RFS2	Created annual standards for four nested renewable fuels that would have to be met by obligated parties through submitting RINs.	2009 to current
MSAT	The requirement to reduce benzene and aromatics in gasoline as a means for reducing toxic emissions created an incentive to use ethanol, since the octane in ethanol helped to replace some of the octane lost through lower benzene and aromatics.	2011 to current
E10 blend wall	The E10 blend wall is commonly defined as the volume of ethanol that can be "readily" blended into the gasoline pool at 10 volume percent, and is a function of the total gasoline consumed.	~2011-2013 to present
Other state programs	Many such as the CA LCFS, state mandates, tax incentives, etc.	Many (summarized in text and detailed in Appendix C)
Weather/climate	Weather and climate affect the cultivation of corn, which affects feedstock availability and price.	Many (esp. 2012)
Other factors affecting corn production	Many factors affect including land, production costs, land rental rates, farm subsidies, Conservation Reserve Program (CRP) policy shifts, etc.	All years

6.3.1 Mandate Versus Consumption Levels

Comparing the level of the RFS-implied mandate for corn ethanol[31] and the consumption level provides initial information about the potential "binding" effect of the Program. The Program is binding if consumption would not have occurred at that level without the mandate, and the Program is not binding if consumption would have occurred at those levels under market conditions regardless of the mandate.

[31] As described in Chapter 1, there is no explicit corn ethanol standard, only the four renewable fuels (i.e. total, advanced, biomass-based diesel, cellulosic) of which corn ethanol is a subset. Thus, the corn ethanol standard is termed an "implied standard" as it makes up the bulk of the conventional biofuel in the United States and is the difference between two regulatory standards (total renewable fuel – advanced biofuel). See Chapter 1 for more details.

When consumption is higher than the mandate, that is evidence that the RFS Program was not binding in that year (Taheripour et al., 2022a; Tyner et al., 2010). When consumption is close to the mandate, the RFS Program may or may not be binding, and more information is required to determine the binding effect (e.g., RINs, section 6.3.2). Due to the ability for parties to carryover RINs, the RFS Program could still be driving consumption above the required volumes in any given year so that parties can use those RINs in subsequent years. Similarly, since it requires considerable time and capital investment to switch gasoline refining and distribution into and out of ethanol blending, market factors could still be driving consumption in any given year even when short-term economics might suggest otherwise. As shown in Figure 6.10, consumption exceeded the mandate by a wide margin of 2–5 billion gallons for all years the RFS1 was in effect (i.e., 2006–2008), indicating the RFS1 may not have been binding in these years. Consumption and the implied ethanol mandate were very close for all years of the RFS2 (2009–2018), which may or may not indicate a binding effect.

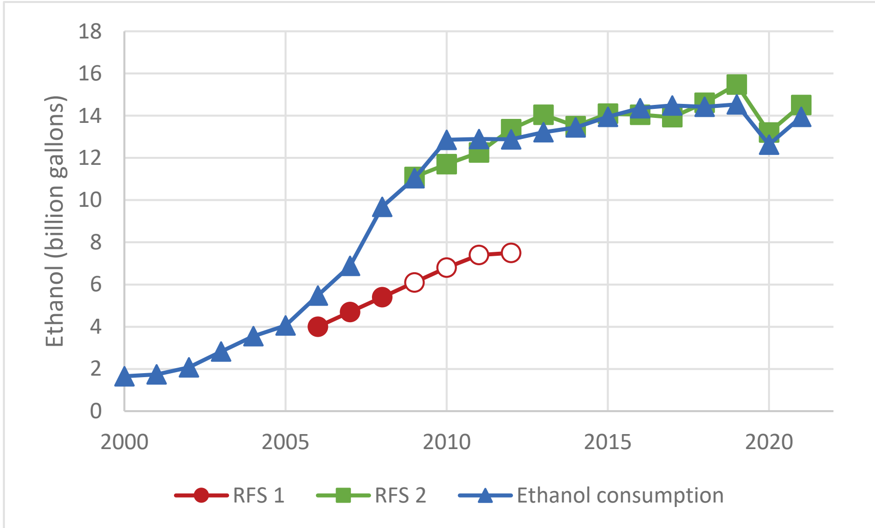
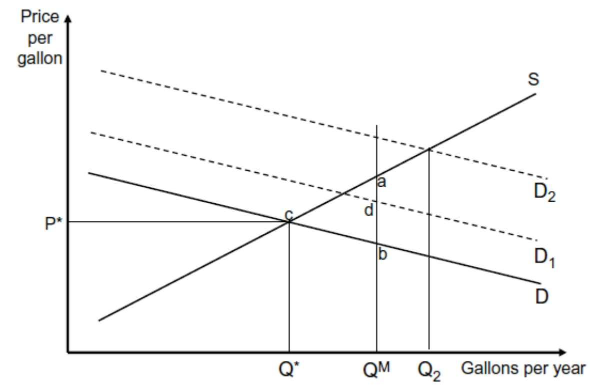


Figure 6.10. Ethanol consumption versus the RFS1 and RFS2 mandates (2000-2021). Annual consumption is from EIA Monthly Energy Review (Table 10.3). RFS1 mandates in the EPA Final Rules and EPAct were equal, and mandates for the RFS2 are "ex-post" volumes (termed "Reported Volume Obligations" after accounting for any administrative adjustments for that year). Open circles indicate years where there was a standard by statute that was not in effect by rule (e.g., 2009-2012 for the RFS1).

6.3.2 D6 RIN Prices

The primary means through which the volume requirements under the RFS Program affect production and consumption of renewable fuels is through RINs. RINs are the means through which producers and importers of gasoline and diesel demonstrate compliance with the volume mandates. Essentially, RIN price can be thought of as representing the difference between the supply and demand given all available subsidies, carryover effects, and any other market and policy factors (for more information on RINs see Chapter 4, McPhail et al. (2011) and Box 6.2. What are RINs?). When RIN prices are near zero, the RFS is said to be non-binding. In reality, RIN prices are never precisely zero, as all parties who own or trade RINs must expend administrative resources (e.g., employee time) in meeting the regulatory recordkeeping and reporting requirements as well as transaction cost incurred in trading RINs. When RIN prices are above these transactional or administrative costs,[32] the RFS is said to be "binding," which means that the RFS Program may be partly responsible for the ethanol consumption in that year. Above the E10 blend wall, the effect of the mandate on RINs becomes more complicated (Burkholder, 2015). Given the nested structure of the standards and the ability for obligated parties to meet multiple standards with different RIN types, an increase in one standard (e.g., total renewable fuel) may have an effect on nested biofuels (e.g., biodiesel), since they may be used to demonstrate compliance with multiple standards (Burkholder, 2015). Furthermore, above the E10 blend wall the D6 RIN price may increase, but that does not necessarily indicate that E10 blending is not economical without the RFS Program, but rather that the standard may be binding for just the marginal increase of biofuel consumption beyond the volume of ethanol that can be consumed as E10. For the most part, however, this chapter focuses on the RIN effects below the blend wall since this is the period of most of the growth of the industry (Figures 6.1 and 6.2).

Box 6.2. What Are RINs?



The theoretical relationship between the supply and demand of ethanol, tax credits and mandates, and their effects on RINs is shown in the figure above. With no tax credit in place, the demand and supply of ethanol is given by lines D and S and the market-clearing price and quantity are given by P* and Q*, respectively, that corresponds to point c. With a mandate of Q^M, above Q*, the price needed by producers increases to a, but the market value drops to b. The price gap, a - b, is made up by the price of RINs. If a blender tax credit is enacted, that increases demand to D_1. The vertical distance between D_1 and D equals the tax credit. With this tax credit, the mandate still binds (i.e.., $Q^M > Q^*$) so the tax credit has no impact on ethanol production. All it does is decrease the RIN price from a - b to a - d. If the mandate is below Q*, or if the credit is large enough to push demand to D_2, there is no value to the RIN above transactional costs. Small refinery exemptions (SREs) can be thought of as decreasing Q^M by the SRE amount, thus also affecting the potential price gap (modified from Babcock (2012)). Note that this representation is a simplistic graphical illustration of the biofuel market for illustrative purposes, and does not show many effects such as indirect effects of the RINS or tax credit in other markets (e.g., gasoline).

As explained in Chapter 1, ethanol produced from corn starch can only generate D6 RINs, and thus the D6 RIN is the relevant RIN to track. EPA did not begin tracking RIN prices digitally until

[32] Transactional or administrative costs are difficult to precisely quantify, but are reportedly only a few cents per RIN (Brown-Hruska et al., 2018).

2010,[34] with only paper records available before that. This means the EPA digital RIN price record begins when the concentration of ethanol in gasoline was already at 9.3% (Figure 6.2) or almost at the E10 blend wall. However, private companies were tracking RINs digitally beginning in 2008, and these data suggest that D6 RIN prices remained low from 2008 until 2013, with a small increase in late 2008 and into 2009 (Figure 6.11).[35] As noted in Chapter 4, D6 RIN prices provide evidence that the RFS Program increased U.S. consumption of renewable biofuels in late 2008 and 2009, as well as from 2013 to 2019. It is important to note that although RIN prices give some information on the effect of the RFS Program on the *existing* fleet of biorefineries, they do not adequately capture the effect the RFS Program may have on influencing investment in new biorefineries. Low RIN prices from 2008 to 2013 indicate that existing ethanol refineries could produce ethanol at a low enough cost that blenders would likely use it without the RIN subsidy. The increase in late 2008 and 2009 was coincident with the large decrease in oil prices with the Great Recession in late 2008 (Figure 6.4). This association is expected, as lower oil prices would reduce the economic incentive to blend ethanol into gasoline, which in turn would drive RIN prices up to increase the incentive to blend ethanol and/or other renewable fuels into transportation fuel at RFS levels. This event was short lived, however, as oil prices again increased in early 2009 (Figure 6.4). In 2013 there was a large increase in D6 RIN prices that EPA reports to be associated with the first year that the implied corn ethanol mandate[36] was above the E10 blend wall (Burkholder, 2015).

Figure 6.11. Historical weekly nominal D6 RIN prices for conventional renewable fuel (predominantly corn ethanol in $/gallon). Data from ARGUS (2008–2020) and EPA (2010–2020).[33]

[33] The RIN system did not exist prior to September 1, 2007, so the earliest date shown is for early 2008. As noted in the text, EPA did not begin digitally tracking RINs until 2010 although private companies like ARGUS and OPIS began earlier. All three sources show the same general trends through time.
[34] Available at https://www.epa.gov/fuels-registration-reporting-and-compliance-help/rin-trades-and-price-information.
[35] By November 2009 D6 RINs had declined back to a few cents per gallon.
[36] See Chapter 1 section 1.1. for information on the implied corn ethanol mandate.

Observations of D6 RIN prices suggest that the total renewable fuel standard of the RFS Program may have been binding in late 2008 and 2009 and after 2013, and thus had some effect increasing corn ethanol production and consumption in those years. RINs did not exist for 2006, the first year of the RFS Program. However, even without the RIN incentives provided by the RFS Program, use of corn ethanol in the United States in 2006 far exceeded the mandated volume (Figure 6.10), suggesting that the RFS Program was not binding in 2006. Recent economic analysis using a partial equilibrium model suggests non-binding effects in both 2005 and 2006 (see section 6.3.3 for more information) (Taheripour et al., 2022a). This agrees with estimates that 2005 and 2006 were favorable years for profit margins for ethanol producers (Figure 6.12; Babcock (2012)) and for potential investors in biorefinery plants (Schmit et al., 2011).

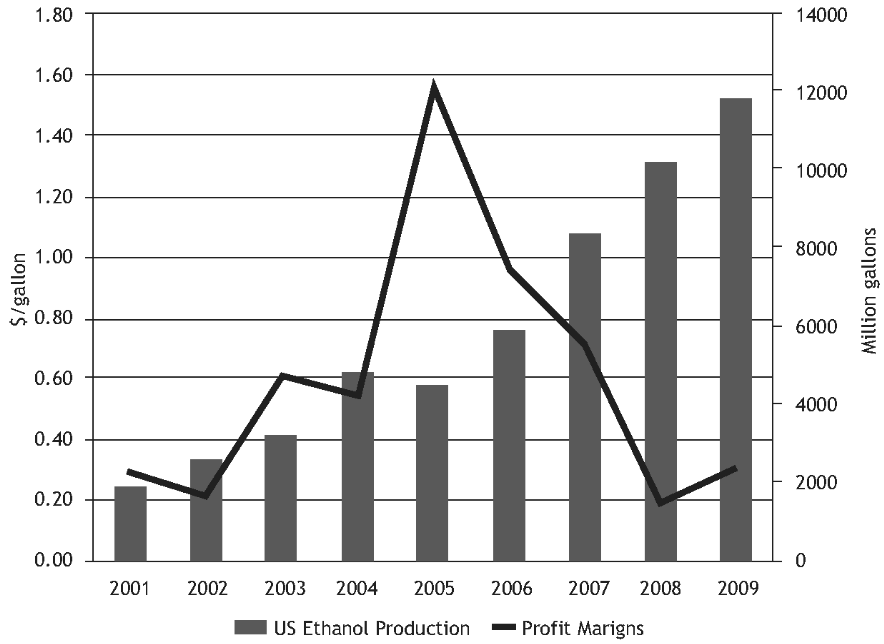


Figure 6.12. Ethanol production (bars) and estimated profit margins (line) from 2001 to 2009. Source: Babcock (2011) (Creative Commons license, https://creativecommons.org/licenses/by-nc-nd/3.0/; no changes made).

It is important to note that at the time the RFS was originally drafted and then enacted with EPAct in 2005 and EISA in 2007, *projected* oil prices were much lower than what ultimately occurred. These anticipated oil prices at the time were comparable to what oil prices had been historically from 1990 to 2004.[37] Thus, at these lower projected oil prices the RFS Program was anticipated to be necessary to spur the development of the industry, as well as provide a guaranteed market in years in which crude oil prices

[37] AEO oil price projections in 2003–2005 were for $20–30/barrel out to 2016, and AEO projections in 2006–2007 were for $40–55/barrel, all much lower than what occurred (Figure 6.4). Even AEO's 2008 report projected decreases in oil price back to $50–60/barrel (see Appendix C for further information).

were low. Small refinery exemptions (SREs) may also be important in interpreting RIN prices, but likely did not have a significant effect during the growth of the industry up to 2013.[38]

Evidence from comparing the mandate with consumption (section 6.3.1) and D6 RINs (section 6.3.2) agree for 2006–2007 (non-binding) and disagree for late 2008. In 2008, there were conflicting indicators of whether the RFS Program was binding. The rise in RIN prices at the end of 2008 suggests the RFS Program may have been binding as oil prices had crashed due to the recession. It is unclear whether the crash in oil prices was too short lived to influence ethanol production and consumption, but it may have had an effect. A further complication with this period is market expectations for future market conditions and the ability to carry over RINs from one year to the next. RIN prices increased after the EPA announced the 2009 standards (November 2008), which were based on the required volumes in EISA and were much higher than those required under EPAct and the RFS1. The market may have expected the RFS Program to be binding in 2009, causing RINs to temporarily increase in price. However, the observation that ethanol consumption exceeded the mandate in 2008 by a considerable margin suggests that the RFS Program may not have been binding in that year after all. Additional information is needed to shed light on the binding nature of the RFS Program in late 2008 and into 2009 (see sections 6.3.3 and 6.3.4).

6.3.3 Subset of Peer-Reviewed Literature

The peer-reviewed literature also provides many analyses of the historical effect of the RFS Program on corn ethanol production. Recall this chapter is not an assessment of the *potential effect* of the RFS Program, which as a mandate is the full volume consumed in any given year, but rather of the *actual effect as events occurred historically*. Thus, in leveraging the literature it is important to consider the quality, quantity, and agreement among studies. In terms of quality, many factors are considered, including whether the literature was retrospective or prospective in nature, and whether individual studies included factors known to affect biofuel production, in addition to the RFS Program (Table 6.2). Thus, only a subset of the broader literature that is reviewed in Chapter 4 is relevant for this chapter.[39]

[38] SREs may have the effect of lowering the standard for that year. However, since the SRE obligations prior to 2011 were all reallocated by EPA to larger refineries, there was no change in the total required volumes prior to 2011, and thus no effect from SREs prior to 2011. The majority of the growth in domestic ethanol consumption had already occurred by then. In 2010, the United States produced 13.3 billion gallons and consumed 12.9 billion gallons (Figure 6.1), and the nationwide average ethanol concentration was 9.3% (Figure 6.2). Thus, although SREs have likely played a role in the effect of the RFS Program later, they played little role in the growth of the industry up to and near the E10 blend wall.

[39] A broader review of lifecycle models is presented in section 6.4.1 and focused on effects on land-cover-land-management (LCLM). It is not included here because most of those models do not have adequate industry detail to separate the effects from the RFS Program from the effects of biofuels more generally. Their strengths, rather, are in the integration of effects across sectors and regions to ultimately influence lifecycle GHG emissions from many factors that affect biofuel production and consumption.

Retrospective studies are useful in that they evaluate the effect of the RFS as conditions actually occurred (e.g., trade, oil prices, droughts). Prospective studies are useful in that they estimate the future effect of the RFS Program under a specified set of assumptions about the future at the time. If those assumptions are representative of what actually occurred, the predictions of those studies may be insightful. No individual study likely accounts for all possible factors that affect ethanol production and consumption, but collectively the literature may be informative.

Primary among the factors to consider is the relative price of E0 gasoline to ethanol, which influences the basic economics of whether blending ethanol into gasoline is favorable or not. Most studies that include energy costs (e.g., for corn cultivation and gasoline production) use oil prices as a surrogate for this economic effect. Gasoline prices, however, track oil prices very closely since gasoline is refined from oil; and, whereas gasoline prices vary widely by region, oil prices have standard reference prices (e.g., West Texas Intermediate [WTI] from Cushing, OK, Figure 6.4). When oil prices are high enough, finished gasoline with ethanol tends to be less expensive to produce than finished gasoline without ethanol. Tyner et al. (2010); Tyner and Taheripour (2008) estimated that ethanol is profitable with nominal oil prices above \$60/barrel (\$69/barrel in 2018 dollars), a level that was not experienced from 1990 through 2004. However, oil prices were generally above these levels from the middle of 2005 until the end of 2014, aside from the crash during the Great Recession (Figure 6.4). This \$69/barrel threshold in 2018 dollars is not absolute, but rather a useful heuristic, as the actual threshold is dynamic as other factors change in the marketplace (e.g. match blending, prices of corn, technology of refining and blending). Nonetheless, Babcock (2012) report that profitability of ethanol was more than \$0.40/gal from 2003 to 2007, peaking in 2005 (Figure 6.12), with similar related findings in other studies (Taheripour et al., 2022a; Schmit et al., 2011). What drove these increases in oil prices in the mid-2000s is outside the scope of this report, but was not driven by the RFS Program, and instead by many geopolitical factors including increased demand from rapidly growing Asian markets, decreased production in non-OPEC countries (e.g., hurricanes Katrina and Rita in the United States), and instability in some OPEC counties (e.g., Iraq, Venezuela), among others (EIA, 2007).

Chapter 4 provides a review and synthesis of much of the peer-reviewed literature in this specific area. It concludes that prospective studies that isolated the expected impact of RFS on corn ethanol production under scenarios with relatively high oil prices (i.e., greater than \$60 per barrel in 2018 prices, which are most consistent with actual market conditions), estimated that the RFS Program would increase corn ethanol production between 0 and 5 billion gallons. Section 4.3.4 reviewed this subset of seven prospective studies that estimated the incremental effect of the RFS Program on ethanol production while controlling for the price of oil and possibly other factors (Figure 6.13) (Bento and Klotz, 2014; Babcock, 2013; Meyer et al., 2013; Babcock, 2012; Tyner et al., 2010; U.S. EPA, 2010; Tyner and Taheripour, 2008). These studies are not all directly comparable, as some attempt to isolate the effect due to the RFS Program from other factors like VEETC, and others include effects like the octane value of ethanol while others do not (Table 6.3). But together they suggest a strong influence from oil price on the incremental effect of the RFS Program, with only a potential small effect when oil prices are above roughly $90 per barrel in 2018 dollars (Figure 6.13, roughly 0-2 billion gallons). Only two studies reviewed estimated the effect of the RFS Program with oil prices in the $40–80 per barrel range in 2018 dollars, a key range covering most of the period of growth from 2005 to 2011 (Figure 6.4). The incremental effect of the RFS Program at these moderate oil prices depends critically on whether the industry is assumed to value octane in ethanol or not. Tyner and Taheripour (2008) and Tyner et al. (2010) did not include the potentially significant value of octane and as a consequent estimated a large effect of the RFS Program (i.e., 11.8–13 billion gallons, blue and teal dots in Figure 6.13), while Babcock (2013) included this factor and estimated a much smaller effect of the RFS Program (i.e., 2.2–3 billion gallons, green triangles in Figure 6.13). It is likely that the industry would value the octane in ethanol given there was a need to replace MTBE (discussed in section 6.2). However, to realize the full economic value of ethanol's high-octane value, it required refiners to make changes to their operations to switch to match blending ethanol instead of simply splash blending it. This change may have been influenced by the RFS Program, especially in non-RFG areas. Thus, the lower end of this range, 2.2–3 billion gallons (Babcock, 2013), is used as a more credible estimate of the incremental effect of the RFS Program according to these prospective studies. Clearly, because of the potential moderating effect from crude oil prices on the effect of the RFS, and the dynamic nature of oil price through time, the effect of the RFS (Babcock, 2013), is

Table 6.3. Summary of assumptions or omissions from the subset of prospective studies that did not assume a binding effect of the RFS Program and included the effect of oil price on corn ethanol production.

Study	Year(s) Modeled	Octane Value of Ethanol Included	RFS Isolated from Other Effects (e.g., VEETC)	MTBE Phaseout Explicitly Included[1]
Tyner et al. (2010); Tyner and Taheripour (2008)	Model calibrated to 2006 data to examine 2015–2022 mandate levels		X	
Babcock (2012)[2]	2011		X	
Babcock (2013)	2014	X	X	
Bento and Klotz (2014)	2004–2015		X	
Meyer et al. (2013)	2017–2021		X	

[1] Many studies implicitly include MTBE phaseout in that it is included in the baseline, but few studies explicitly include this factor.

[2] This study used a generic partial equilibrium (PE) model representing various markets associated with ethanol (Babcock et al., 2010) and also included a retrospective analysis that used a modified CARD-FAPRI model to examine 2005–2009. However, that analysis did not separate the RFS from VEETC, and thus the results are not discussed.

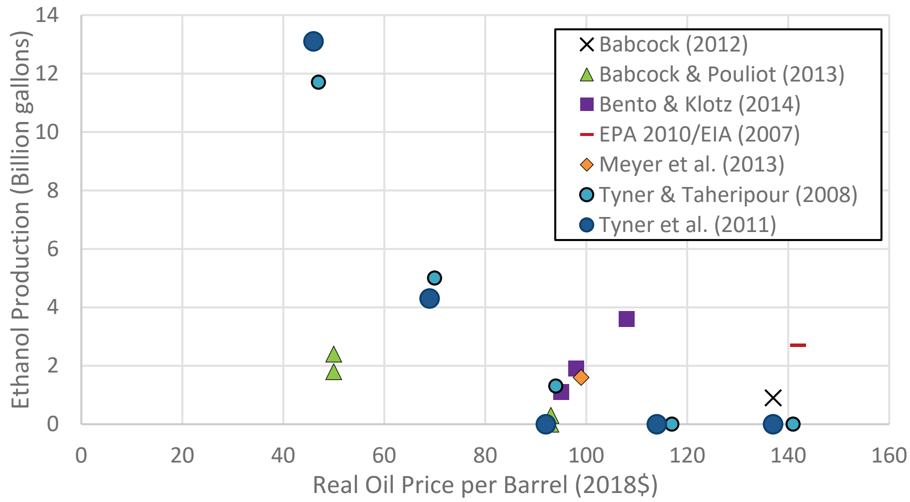


Figure 6.13 (from Chapter 4, section 4.3.4). Incremental effect of RFS on U.S. corn ethanol production. Estimates are from Babcock (2012)'s forward-looking analysis of 2011 impacts (black X); Babcock (2013) projections for 2014 using a demand curve reflecting oxygenate and octane value and 85 and 90 million harvested acres (green triangles). Circles highlight the large difference in estimated effect among studies at lower oil prices ($40-60 per barrel) that included and omitted the octane value of ethanol. Bento and Klotz (2014) (purple squares); EPA's comparison of RFS2 with the 2007 AEO projection for 2022 (2010) (red dash); Meyer et al. (2013)'s no corn yield improvement scenario during 2017–2021 (yellow-orange diamond); Tyner and Taheripour (2008)'s RFS and fixed subsidy with no demand shock scenarios (small teal circles); and Tyner et al. (2010)'s RFS and fixed subsidy scenarios (larger blue circles).

used as a more credible estimate of the incremental effect of the RFS Program according to these prospective studies. Clearly, because of the potential moderating effect from crude oil prices on the effect of the RFS, and the dynamic nature of oil price through time, the effect of the RFS Program varies from year to year. Furthermore, because of the potential differences among RFG and non-RFG areas in the value of octane until the CG market also transitioned to match blending around the 2010 timeframe, the effect of the RFS Program may vary regionally as well. Studies that do not explicitly include octane often implicitly assume that any effect of ethanol's octane value is instead due to the RFS Program. These studies should re-evaluate how gasoline would have been manufactured in the United States without MTBE if it were not with octane from ethanol. These two effects (i.e., ethanol replacing MTBE and the RFS Program), though coincident in the same statute of the EPAct of 2005, are not actually the same driving factor.

Retrospective analyses may be more helpful in assessing the effect of the RFS Program, but few such analyses that control for other important factors that could influence ethanol production are available. A recent assessment by Taheripour et al. (2022a) fills this critical gap and is one of the only published retrospective economic modeling to date that included factors such as oil price, octane, and MTBE, and estimate the annual effect of the RFS Program through the entire period from 2004 to 2016 (but see section 6.3.5).[40] They used a partial equilibrium (PE) model, Agricultural Energy Partial Equilibrium (AEPE), to estimate the annual effect of the RFS Program from 2005 to 2016, and a Computable General Equilibrium (CGE) model, GTAP-BIO, to estimate the long-run equilibrium effect for two time periods: 2004–2011 and 2011–2016. The combination of PE and CGE models in a single study is useful in that the PE approach includes annual estimates as conditions change (e.g., prices) and industry detail known to be important in biofuel markets (e.g., octane, oil, MTBE, corn price), but PEs do not have feedback loops to the global economy. The CGE approach examines global economy-wide feedbacks but with less industry and temporal detail. The PE model included the effect of octane, MTBE phaseout, and actual oil prices over the interval. The CGE model did not explicitly include these factors but did separate the effect of the ethanol mandate in the RFS from that of the effect of growth in ethanol production more generally.

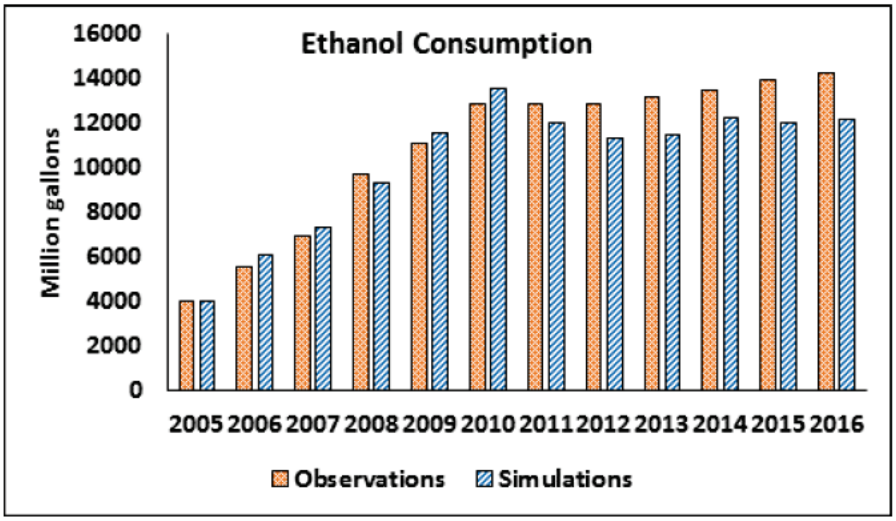


Figure 6.14. Partial equilibrium modeling results using AEPE. Observed ethanol consumption ("Observations", red bars) and consumption absent the RFS mandate ("Simulations", blue bars) are shown from 2005 to 2016. If the blue bar is below the red bar, the Program was estimated to be binding for that year. Source: (Taheripour et al., 2022a) (used with permission).

Using the PE model, Taheripour et al. (2022a) found the RFS was not binding in 2005–2007 and 2009–2010, and was binding in 2008 and 2011–2016 (Figure 6.14). In 2008 the incremental effect of the RFS Program was to increase ethanol consumption by 0.4 billion gallons, and for 2011–2016 to increase consumption by roughly 1–2 billion gallons each year. Thus, during the period when ethanol concentrations in gasoline increased from 1.5% to 9.3% from 2002 to 2010, Taheripour et al. (2022a) estimate the RFS was binding in one year, 2008, the year that oil prices crashed (Figure 6.4). This conclusion for 2008 coincides with the observed small increase in D6 RIN prices in 2008–2009 (Figure 6.11). A binding effect after 2011 in the model was only manifested in higher D6 RIN prices after 2013 (Figure 6.11), likely due to the expiration of VEETC in 2011 and the omission of other factors (e.g.,

[40] An earlier retrospective study using CARD-FAPRI also found small effects from biofuel policies relative to market factors, but because this study did not separate VEETC from the RFS or include the octane value of ethanol, these results are not highlighted here in the assessment of the RFS Program (Babcock, 2012).

carryover RINs) in the model that may have prevented a binding effect. For example, there were over 2.5 billion banked RINs in each of 2011 and 2012 that could have kept RIN prices lower. Oil prices were still high from 2011 to 2014 and probably do not explain the estimate of a binding effect in 2011–2012, which did not manifest in higher D6 RIN prices (Figure 6.4).

Using the CGE model, Taheripour et al. (2022a) report that from 2004 to 2011 the RFS Program (both mandates examined) increased ethanol production by 0.7 billion gallons in 2011 relative to 2004 (for an annual estimated increase of 0.1 billion gallons), and by 1.5 billion gallons in 2016 relative to 2011 (for an annual estimated increase of 0.3 billion gallons).[41] Thus, the RFS Program is estimated to have a smaller effect in the earlier time period when many other factors contributed (e.g., VEETC, octane value, high oil prices, MTBE phaseout), and a larger effect in the later period when fewer other factors contributed (e.g., VEETC expired on December 31, 2011, lower oil prices after 2015).

A recent publication by Lark et al. (2022b), based on economic analyses in Carter et al. (2017) and Pates and Hendricks (2021), assessed the effects of corn and corn ethanol production on a range of environmental end points relevant to the RtC3. This study provides a useful analysis of the effects from corn and corn ethanol broadly, rather than from the RFS Program specifically, estimating an increase in corn ethanol production of 5.5 billion gallons each year. These estimates are close to, but above, the range of estimates from the broader literature reviewed in Chapter 4 that account for the effect from oil price (0-5 billion gallons). They are also higher than other studies that include other relevant factors because of several assumptions in the underlying economic model (Carter et al., 2017) that increase the estimated effect of the RFS Program. First, it assumes the RFS2-effect began in 2006, which is actually prior to EISA (signed into law December 2007) and well prior to promulgation of the rule (March 2010). Second, it assumes beginning in 2006 a baseline without the RFS Program to be the annual mandates under the RFS1. Given that actual production far exceeded the RFS1 mandates in every year that the RFS1 volume targets applied (Figure 6.10), it is unrealistic to assume zero growth above the RFS1 mandates absent RFS2. Third, two key factors that affect corn ethanol production and consumption were omitted from the underlying economic model: the effects from MTBE replacement on corn price and the effects of a switch from splash blending to match blending. In Lark et al. (2022) the authors acknowledge the latter[42]. Either

[41] Taheripour et al. (2022a) estimate the effects of the RFS Program separately from the effects of two mandates, an ethanol mandate and a biodiesel mandate. From 2004 to 2011, the effect of the ethanol mandate is to increase ethanol consumption by 0.6 billion gallons and the effect of the biodiesel mandate is to further increase ethanol consumption by 0.1 billion gallons through cross-market effects on feed and livestock, for an overall increase of 0.7 billion gallons. From 2011 to 2016, the effect of the ethanol mandate is to increase ethanol consumption by 1.8 billion gallons and the effect of the biodiesel mandate is to decrease ethanol consumption by 0.3 billion gallons, for an overall increase of 1.5 billion gallons. The other two RFS mandates were not modeled.

[42] The paper states, “These outcomes approximate the contribution of the RFS policy specifically, although other factors including changes in fuel blending economics that favored 10% ethanol as an octane source in gasoline (E10) may also have contributed.”

one of these factors could have a large effect on the estimated effect from the RFS Program. Assuming that all effects after and including 2006 are due to the RFS Program implicitly ascribes potential effects from MTBE phaseout and match blending to the RFS Program. The large decrease in MTBE and concurrent increase in ethanol, outside of California, occurred in summer 2006 (Figure 6.3), which aligns well with the increase in corn price in the winter of 2006 (Figure 6.7)—before EISA. Thus, Lark et al. (2022b) may provide useful estimates of the potential effects from corn ethanol broadly, which is affected by many market and non-market factors, more so than an estimate of the effect of the RFS Program specifically.[43]

6.3.4 Biomass Scenario Model

In preparation for the RtC3, EPA collaborated with NREL to develop scenarios for the Biomass Scenario Model (BSM) to inform understanding of how various factors, including the RFS Program, influenced ethanol production. The BSM has been developed over more than 10 years to include many policy and economic drivers that are relevant to biofuels and other biomass-based products, so that decision-makers can better understand the estimated implications of policy decisions under consideration (Peterson et al., 2019; Newes et al., 2015; Vimmerstedt et al., 2015; Vimmerstedt et al., 2012; Newes et al., 2011). The BSM has been used internally by DOE to evaluate various "what if" scenarios of biofuels and other biomass-based products. The BSM is not an economic model with a series of markets that must be cleared to produce an optimum configuration of prices and quantities; rather, the BSM is a system dynamics model that includes a series of 10 dynamically interconnected modules with linear programming submodules for feedstock supply, feedstock logistics, feedstock conversion, inventory and pricing (of biofuels), distribution logistics, dispensing stations, fuel use, vehicles, biofuel imports/exports, and the interaction between the biofuels and petroleum industries. These modules receive and react to information in a complex, nonlinear fashion that depends on, among other things, industrial learning, project economics, installed infrastructure, consumer choices, and investment dynamics. Much of the logic and information underpinning the BSM has been developed with industry and federal input through an iterative process of refinement. The BSM is not a predictive model; rather it represents the dynamic behavior of the biofuel and bioproducts industry according to present understanding.

Typically, the BSM has been used prospectively to examine the estimated effects of different hypothetical policies, although the logic and architecture of the model support retrospective analyses as

[43] The Lark et al. (2022b) estimate likely reflects a combined effect from MTBE and the RFS Program. Once these attributional differences are accounted for, the changes in cropland estimated in Lark et al. (2022b) and this report are relatively close (discussed in the next section 6.4.3). Technical aspects of the Lark et al. (2022b) study remain actively debated (Alarcon Falconi et al., 2022; Lark et al., 2022a; Taheripour et al., 2022b, c); however, much of that debate is outside of the peer-reviewed literature. Most of these technical issues surround the GHG emissions, which are out of scope of this report, and the assumed attribution to the RFS, which is discussed above.

well. NREL developed a retrospective version of the BSM focused on corn ethanol in support of the RtC3 to evaluate the effect of various policies on domestic ethanol production from 2002 to 2019 (Newes et al., 2022). NREL examined the estimated effect of five different factors across a combination of seven scenario runs (Table 6.4). This sequential approach in scenarios A through G leads to "priority" given to factors already included, but also better represents actual historical effects as events unfolded. For example, the actual effect of the RFS Program was in addition to whatever effect VEETC already had since VEETC preceded the RFS Program.[44] It is important to note that the BSM estimates the effect of the RFS Program through observed RIN prices, which began in 2008 and are exogenous to the model. Thus, the BSM can only estimate the marginal effect of the RFS Program after accounting for other factors that may influence ethanol production and thus RIN prices.

Table 6.4. Potential drivers of changes in ethanol production evaluated in the BSM and how they are combined in each of seven BSM scenarios (years active, "X" indicates the factor is included).

Driver	Mechanism of inclusion (years active)	A	B	C	D	E	F	G
Oil prices	As oil prices increase relative to ethanol prices, it is increasingly attractive to blend ethanol into gasoline up to the blend wall (all years).	-	X	X	X	X	X	X
MTBE phaseout	Replacement of MTBE with ethanol as an oxygenate to satisfy Clean Air Act requirements (2002–2008).	-	-	X	X	X	X	X
Blenders' credit (e.g., VEETC)	Incorporate blenders tax credit (2002–2011a).	-	-	-	X	X	X	X
RFS Program	Use historical D6 RIN values to estimate the marginal effect of the RFS (2008–2019).	-	-	-	-	X	-	X
Octane	Account for industry transition to match blending to take advantage of ethanol as an octane enhancer (2005–2019).	-	-	-	-	-	X	X

[a] The effects of the MTBE phaseout and VEETC extend past 2008 and 2011, respectively, even though the model does not explicitly include these drivers past those years. The blenders' tax credit varied through time with different policies (e.g., VEETC from 2004 to 2011): $0.60 per gallon through 1990, $0.51 through 2008, $0.45 from 2009 through December 31, 2011 (USDA ERS Biostats Table 14).

The BSM results for simulations A through G demonstrate that the incentive to produce and blend ethanol from oil price alone (Figure 6.15, scenario B) increased ethanol production from 2002 to 2014, but not enough to match observed production (Figure 6.15, observed). Production under scenario B also lagged observed production by several years at specified levels (e.g., observed production crossed 10 billion gallons by 2009 but scenario B did not reach this threshold until 2013). The addition of the MTBE phaseout on top of the effect from oil prices increased production further by 0–2 billion gallons and accelerated it to earlier years (Figure 6.15, scenario C), but still not to observed levels. It was not until VEETC and MTBE phaseout were included with oil price that the simulated ethanol production levels from 2002 to 2011 matched that of observed (Figure 6.15, scenario D). For this scenario, however, there

[44] And indeed the blenders' tax credit in some form dated back to the 1980s (see Table 6.1).

was a large simulated drop in ethanol production in 2012 due to the expiration of VEETC and omission of other factors that could have buffered this effect.[45] This was simulated to occur because the MTBE effect was largely over by 2007 (Figure 6.5) and oil prices were not simulated to be enough on their own to maintain ethanol production in the absence of VEETC. The addition of the RFS Program through observed D6 RINs (Figure 6.15, scenario E) was estimated to have little effect in this early period when D6 RIN prices were still low and thus did not buffer the system from the estimated drop in ethanol production in 2012. However, RINs did help maintain higher production after 2013 (Figure 6.15, scenario E) when RIN prices were higher (Figure 6.11). The addition of an octane value for ethanol also buffered the simulated drop in ethanol production in 2012, with or without the RFS Program (Figure 6.15, scenario G and F for scenarios with an octane value, and with versus without the RFS, respectively). Scenarios with an octane value for ethanol but without the RFS Program closely matched production except in the later years from 2016 to 2018, when the RFS likely provided additional support (Figure 6.15, scenario F). The addition of the RFS on top of an octane value for ethanol (i.e., all factors, Figure 6.15, scenario G) closed the remaining gap in 2016–2018, with simulated and observed production matching over most of the period.

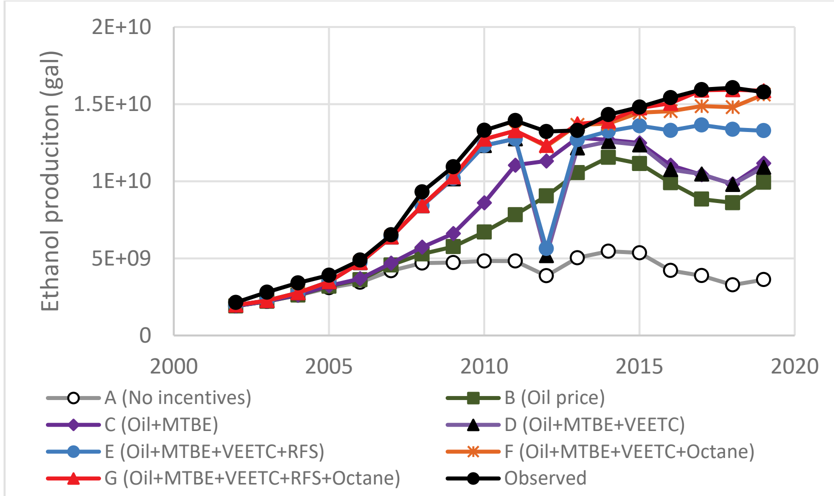


Figure 6.15. Simulated ethanol production from 2002 to 2019 using the BSM, assuming chronological addition of five potential drivers: Oil prices, MTBE phaseout, blenders' tax credit (VEETC), RFS Program, and octane. Observed production from EIA added for reference. Source: Newes et al. (2022) (Creative Commons license, https://creativecommons.org/licenses/by/4.0/; graph colors and symbols adjusted).

From this simulation modeling, the marginal effect of the RFS Program as captured through D6 RINs may be estimated from differences among scenarios. The effect of the RFS Program, including the octane value in ethanol, is represented by scenarios G minus F, and excluding the value of octane is scenarios E minus D.[46] These differences (Figure 6.16) show that the RFS Program is estimated to have

[45] This includes the omission in scenario D of other factors such as the RFS and match blending, as well as a lack of foresight in the model, each of which could have buffered the system from the effect of VEETC expiration.
[46] It is likely more realistic that the runs with octane value of ethanol represent how the industry unfolded, given that the octane was needed to meet CAAA requirements and the infrastructure was largely in place after roughly 2007–2008 (see section 6.2.3).

had a small effect from 2002 to 2011 with oil prices, MTBE phaseout, and VEETC supporting production. The small increase in D6 RINs in 2008/2009 was not estimated to have much of an effect in the BSM. After 2013 the effect of the RFS Program increased as D6 RIN prices increased. This effect peaked after 2015 when oil prices had come down, with the maximum incremental effect of the Program estimated at 1.1 and 3.6 billion gallons, depending on whether the octane value of ethanol was included or not, respectively. The observed D6 RIN record may not be an ideal estimate of the effect of the RFS Program in absence of other factors because there is only one observational record, which included all the other factors. The BSM model would need to be redesigned to internalize RIN prices. Absent that improvement, an alternate possibility for estimating the effect of the RFS with the BSM could be the difference between the observed ethanol production and the estimated effect of all non-RFS factors (i.e., observed minus scenario F). The issue with this approach is that, as has been done in other studies discussed above, this implicitly lumps all effects not included in the model together and ascribes them to non-RIN effects of the RFS. Since there are many factors not included in the model (e.g., state mandates, California LCFS, trade), this was not considered an appropriate method for estimating the effect of the RFS Program using the BSM.

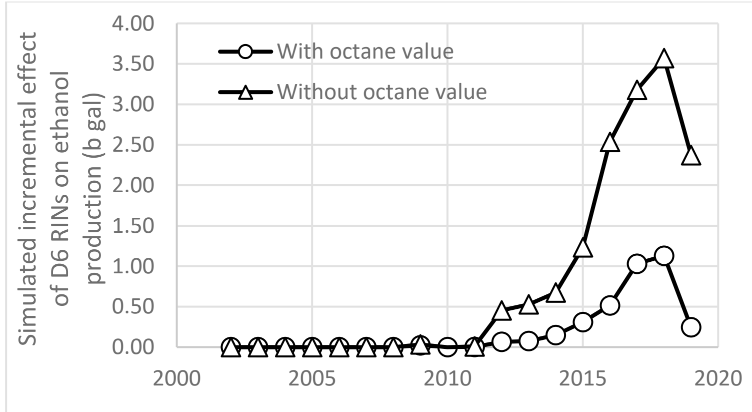


Figure 6.16. Simulated incremental effect of the RFS Program from the BSM using several approaches. Estimated effects from D6 RINs including the octane value of ethanol (black line, circles, scenario G – F), and excluding it (black line, triangles, scenario E – D). See Table 6.4 for scenarios and Newes et al. (2022) (Creative Commons license, https://creativecommons.org/licenses/by/4.0/).

NREL supplemented these runs with estimates of the effect of the RFS Program on reducing risk and establishing market certainty. This factor is commonly cited in the literature as important but is exceedingly difficult to quantify. Building off the results from Figure 6.16, which isolate the effect of the RFS Program, and assuming octane had a value to refiners, two risk reduction levels are used from Newes et al. (2022), 20% and 40%.[47] Addition of this factor has a significant effect, increasing buildout in the early years and production in 2009-2011 relative to the scenario without this factor (Figure 6.17). The effect on production does not materialize until 2009 because, although the risk reduction begins with EPAct in 2005, it takes roughly 3 years for new plants to be built and begin to produce ethanol. Thus,

[47] Various risk reduction levels were considered (0%, 20%, 40%, 60%), but Newes et al. (2022), concluded that risk reduction by 20-40% was more supported by the observed trends than higher levels.

effects in 2006–2008 were generally small (5–45 million gallons). The model also shows that the scenarios with and without risk reduction converged by 2016, suggesting that the biorefinery capacity triggered by the RFS would have been built eventually. This suggests that the risk reduction factor was important in accelerating the early growth of the industry, but it did not influence the ultimate size of the industry given the other factors that also influence ethanol production.

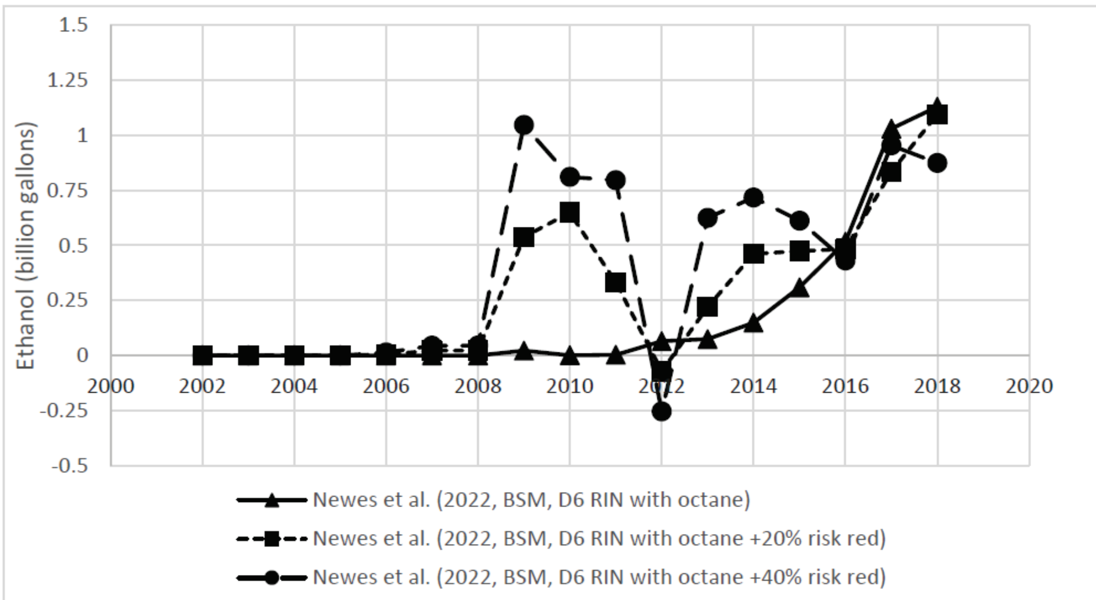


Figure 6.17. Simulated ethanol production including risk reduction from the RFS. Shown are the isolated effect from the RFS Program assuming no reduction in risk from the RFS Program (solid line, same as Figure 6.16), and a 20% (dotted line) and 40% (dashed line) reduction in risk (from Newes et al. (2022)) (Creative Commons license, https://creativecommons.org/licenses/by/4.0/).[48]

There is a vast literature on the economic and welfare effects of various biofuel policies (e.g., de Gorter and Just (2009), Schmit et al. (2011), Lapan and Moschini (2012), Ghoddusi (2017), Thome and Lawell (2022)). Much of this literature is reviewed in Chapter 4. Surprisingly few studies examined the role of the RFS Program in reducing investment risk and thus the boom in construction observed in 2007 and 2008. There are many studies on the role of policy uncertainty generally or in other markets (Handley and Limao, 2015; Dixit and Pindyck, 1994), but it is not clear whether these lessons require modification for application to first-generation biofuels in the United States. McCarty and Sesmero (2015) found that price risk was a significant barrier to market entry for second-generation corn stover biorefineries, but it is not clear whether these findings would apply to first-generation biorefinery plants. The BSM analysis above suggests that roughly 1 billion gallons may be associated with the RFS Program through a

[48] All simulations include an octane value. The crash in 2012 was due to the aforementioned lack of foresight in the model and the expiration of VEETC. Simulations were cut off at 2018 for brevity and because almost all biorefineries had been built by then.

reduction in investment risk. Thome and Lawell (2022) used an econometric approach to assess the effect of various factors on the entry of new ethanol refineries from 1996 to 2008 (e.g., RFS1, RFS2, MTBE, other refineries in the county, other refineries in adjacent counties, tax credits, various prices and other factors). They found a strong negative effect from existing ethanol refineries (competition), and a weak effect from the RFS Program, which was not significant in 8 out of 10 specifications with a linear year trend.[49] When the ratio of ethanol to gasoline price was included and time was handled as a linear trend, the effects of both the RFS1 and 2 were significant, but small relative to the effect of existing plants. The MTBE ban was not significant but was only modeled at the state level so the effective national ban in 2006 was not captured in the analysis. The response in Thome and Lawell was the number of new biorefineries, not volume of ethanol produced, thus it cannot readily be converted to a volume attributable to the RFS Program. Schmit et al. (2011) use a real options analysis (ROA) to examine the effect of ethanol policies on plant investment decisions. They showed that an expansion period was favored in late 2005 to early 2007, which emerged in the observed construction data roughly one year later (Figure 6.6). This was also shown in Babcock (2011) (Figure 6.12). Schmit et al. (2011) projected no expansion after 2007, which too was reflected in the observed data after roughly 2009. Policies were shown to have a strong influence on expansion period primarily through revenues, but individual policies were not separated in the analysis to account for the various interacting federal and state policies.

Newes et al. (2022) also separately estimate the *potential* effect of the various drivers, from the *actual* estimated effects (Table 6.5). They report that the potential effect of the RFS Program (RFS1 and RFS2) was 0–6.2 billion gallons, in line with Lark et al. (2022b) and several other studies that do not account for many co-occurring factors; but, that the actual effect as events unfolded was estimated to be much smaller. Newes et al. (2022) estimate the actual effect of RFS1 was roughly 0–2 billion gallons, and the actual effect of the RFS2 depended on whether match blending was included, with an effect of 0–1.1 billion gallons if match blending occurred prior to RFS2 and 0–3.6 billion gallons if match blending occurred after the RFS2.[50] The remainder was estimated to be attributable to the replacement of MTBE (0–3.2 billion gallons) and VEETC (0–3.7 billion gallons) and the transition to match blending (0–6.6 or 0–6.7 billion gallons depending on whether RFS2 effects occurred after, or before match blending effects, respectively).[51] See Newes et al. (2022) for more details.

[49] Only specifications with a linear year trend are relevant because using annual effects for year confounded the time variable with the RFS terms.
[50] As discussed above in section 6.2.2., there is no official data tracking the transition to match blending, but it likely predated RFS2, which did not go into effect until 2009 (for total renewable fuels) or 2010 (for the four individual standards).
[51] Note that the 5.5 billion-gallon effect from the RFS Program in Lark et al. (2022b) is similar to the combined effect of MTBE and the RFS Program without ethanol valuation in the BSM (0-6.8 billion gallons), as expected from the design of the Lark study.

Table 6.5. Metric of Modeled Effects by Driver.

Driver	Rationale	Range of Estimated *Potential* Effect (2002 to 2019) Annual EtOH Production above Baseline (billion gallons)	Range of Estimated *Actual* Effect (2002 to 2019) Annual EtOH Production above Baseline (billion gallons)
Price competition	With higher oil prices, blending ethanol into gasoline, up to applicable constraints, becomes economically advantageous. There is no range, as this was the baseline assumption; however, simulations estimated price competition to account for at least 50% of observed production.	Baseline	Baseline
MTBE replacement as part of the California and federal RFG programs	MTBE was the preferred oxygenate before 2001, but concerns about groundwater contamination and associated liabilities created a need for a substitute. The federal RFG Program, which has been in effect since 1995, created a demand for oxygenate in O_3 non-attainment areas. The California RFG Program, which has been in effect since 1996, created a demand for oxygenate in O_3 nonattainment areas.	0.0–3.2	0.0–3.2
Federal ethanol tax subsidy	This tax subsidy, which went into effect in 2004 and expired at the end of 2011, lowered the cost for blenders to mix ethanol into gasoline.	0.0–3.8	0.0–3.7
RFS1 standards	The RFS1 standards, which were in effect from 2005 to 2008, created or contributed to demand for biofuels.	0.0	0.0–2.0[52]
RFS2 standards	The RFS2 standards, which have been in effect since 2009, created or contributed to demand for biofuels. D6 RINs are used as the market mechanism for RFS2. They are not an independent driver but are evidence of an effect from RFS2.	0.0–6.2	0.0–1.1 (assuming RINs implemented *after* match blending) 0.0–3.6 (assuming RINs implemented *before* match blending)
Transition to match blending	The octane value of ethanol allowed refineries to transition from producing 87 octane BOBs to cheaper 84 octane BOBs.	0.0–6.5	0.0–6.6 (assuming RINs implemented *after* match blending) 0.0–6.7 (assuming RINs implemented *before* match blending)

NREL also supplemented these runs on how the industry evolved with additional hypothetical RIN and oil prices to estimate the effect the RFS Program *may have had* under different conditions. RIN prices were assumed to vary from $0 to $1.00 representing the range of D6 RIN prices observed from

[52] Note that the upper value here (2.0 billion gallons) is from the maximum risk reduction effect of 60%. Newes et al. (2022) suggested that 20-40% were more in line with observations (Figure 6.21). Newes et al. (2022) also ascribed the risk reduction effect to RFS1, but the factor likely is better ascribed to EISA, which created the RFS2. Either way, it is only counted once to avoid double counting.

2008 to 2019,[53] and oil prices were assumed to vary from $25 to $100/barrel, representing much of the range of oil prices over the same period (Figure 6.4).[54] These simulations show that the RFS had little effect from 2005 to 2007, regardless of these scenarios, because many of the changes over this period were dominated by MTBE phaseout and VEETC. Scenarios began to separate beginning in 2008 (Figure 6.18). Figure 6.18a shows that with oil prices at $25/barrel, which was typical for the 1990s and up until the early 2000s, and no octane value of ethanol,[55] RIN prices between $0.25 and $0.75 were sufficient to drive ethanol production near observed levels. This range of D6 RINs is not unrealistic and has been observed in the D6 RIN record (Figure 6.11), suggesting these levels may have occurred had oil prices remained low. After roughly 2015 even $1 RINs were not sufficient and ethanol production decreased for lower valued RINs. Figure 6.18b shows that with oil at $25/barrel and a value of octane, again prices between $0.25 and $0.75 were sufficient to drive ethanol production near observed levels for the entire period and were resilient to the shocks of VEETC expiration. With oil prices at $75/barrel and above (Figure 6.18c–d), simulated production exceeded observed production for any RIN value as suggested in Tyner et al. (2010).

6.3.5 Economic Analysis of Blending Ethanol

To conduct a thorough evaluation of the economics of ethanol blending, OTAQ analyzed ethanol's blending economics on a state-by-state basis for each year from 2000 to 2020, for blending ethanol up to the E10 blend wall (U.S. EPA, 2022). A prime motivation for the analysis was to better understand the potential effects from factors that are omitted from the broader literature and economic studies. The analysis used output from a linear program refinery cost optimization model to estimate ethanol's full replacement value. The model accounts for ethanol's high octane, but also accounts for replacing ethanol's volume, its very low sulfur and zero benzene content, and its volatility cost when blending into RFG, which does not receive a 1 psi waiver. The analysis accounted for the VEETC, state ethanol blending mandates, and distribution costs for transporting the ethanol to individual states. The analysis further enhanced its granularity by modeling ethanol's relative blending value into different gasoline types, conventional and RFG, and different gasoline grades, regular and premium, during each of the two annual gasoline blending periods, summer and winter.

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[53] The highest D6 RIN price observed from 2008 to 2018 was $1.05 on October 5, 2013 (Figure 6.10 and https://www.epa.gov/fuels-registration-reporting-and-compliance-help/rin-trades-and-price-information).
[54] Only results for $25 and $75/barrel oil are shown, see Newes et al. (2022) for more details.
[55] This scenario could also reflect limited availability to capitalize on the octane value of ethanol due to infrastructure limitations.

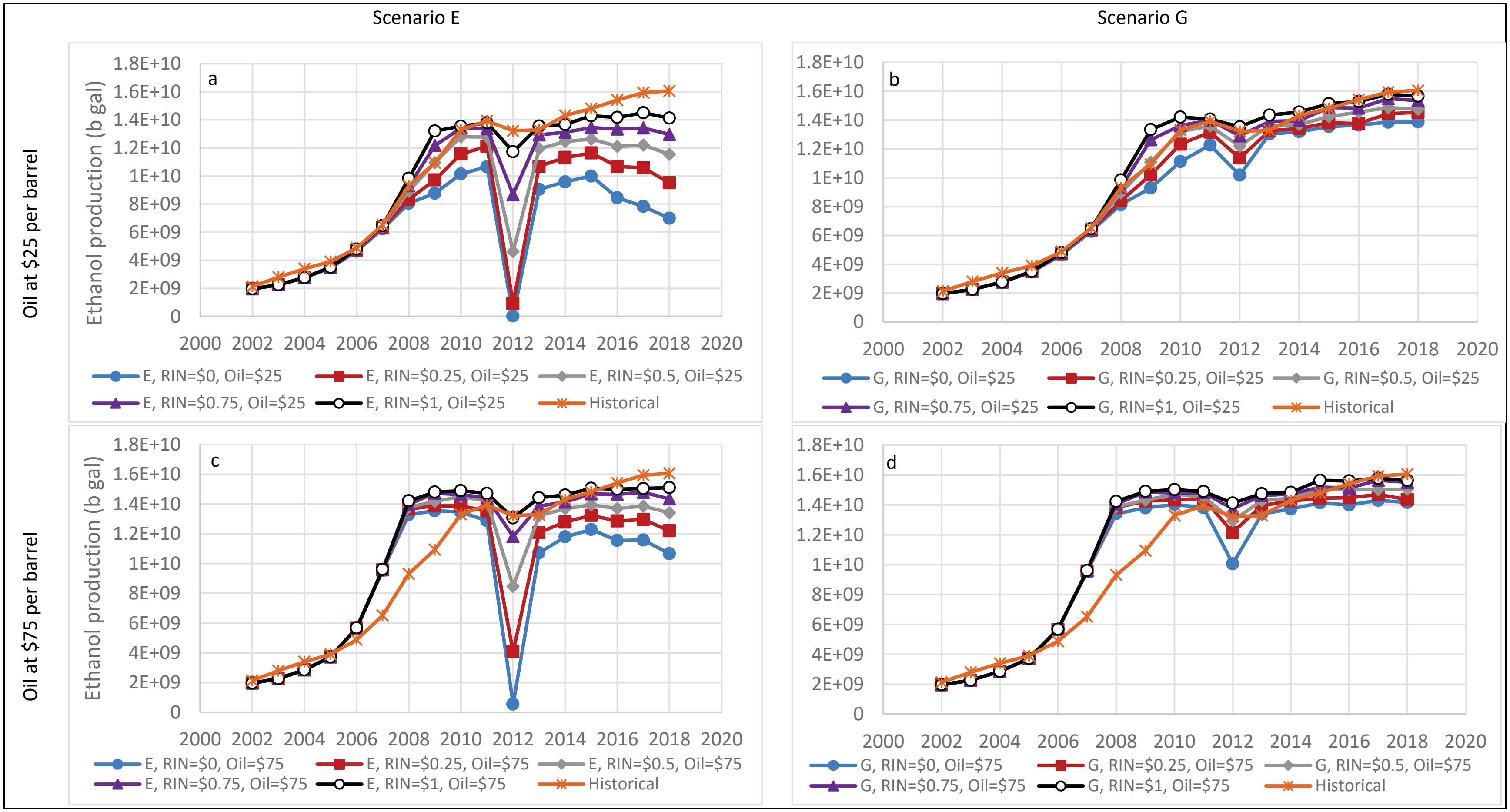


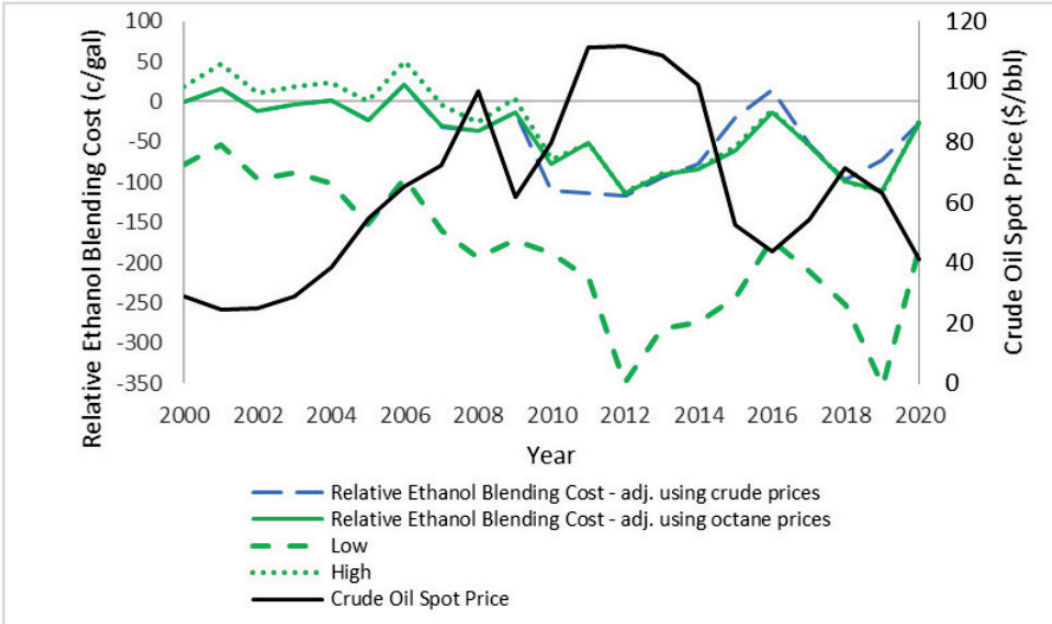
Figure 6.18. Simulated ethanol production from 2002 to 2018 using the BSM for scenarios E (a, b; all factors except octane) and G (c, d; all factors). Simulations were run assuming different D6 RIN values (i.e., $0, $0.25, $0.50, $0.75, $1.00) and oil prices (i.e., $25, $50, $75, $100; only $25 and $75 are shown here). Observed production added for reference (orange with asterisks).[56]

[56] Only oil prices of $25 and $75 are shown for brevity. Similar results for $50 and $100 are reported in Newes et al. (2022).

The analysis considered ethanol's economics in two different ways: (1) based on ethanol plant gate spot prices reported by USDA, and (2) estimated cost of ethanol production based on the prices of corn, natural gas, and other inputs and other co-products. These two price measures are of interest for different reasons. The spot price of ethanol, after accounting for distribution costs, is representative of the price that blenders would pay for ethanol to blend into gasoline to make E10. The cost of ethanol production is an estimate of the cost to produce the ethanol and differs from the spot price by eliminating market factors which skew the ethanol plant gate prices. The cost of ethanol production relative to the spot price, while not the only factor that impacts investment decisions and profitability, can provide insights about potential investment decisions in ethanol production facilities in certain times. If the estimated cost of production plus transport to point of sale (i.e., the ethanol blender) is lower than the price difference between gasoline and the spot price of ethanol, potential investors could reasonably expect demand for ethanol. Additionally, the difference between the ethanol spot price and the estimated cost of production provides an estimate of the profitability of ethanol production on a per gallon basis for an average ethanol production facility.

This retrospective analysis concluded that even without the RFS Program, the economic and market factors were more than sufficient to drive the expansion of corn ethanol as E10 since the mid-2000s (Figure 6.19). With the exception of 2008–2009, this finding agrees with the observation of low D6 RIN prices up to the E10 blend wall in roughly 2013 as discussed in section 6.3.2 (Figure 6.11). While the analysis showed that ethanol was not cost-competitive with gasoline in 2006, this was likely due to the sudden demand increase for ethanol to replace MTBE in RFG to comply with the RFG Program requirements (see section 6.2.3 and Figure 6.3). This sudden demand increase resulted in a spike in its spot price in 2006. From an ethanol manufacturer's perspective, ethanol's production profitability peaked in 2005–2007, as also reported elsewhere (Taheripour et al., 2022a; Babcock, 2012) (Figure 6.12), and immediately preceded the large increase in construction of biorefineries in 2007 and 2008 (Figure 6.6). Notably this increase in profitability was before the RFS2 was in effect and when production already exceeded the RFS1 mandates by a large margin. The ethanol blending analysis in U.S. EPA (2022) additionally shows that despite the much lower crude oil prices beginning in 2015, ethanol was estimated to be cost-competitive with gasoline even after 2015, with the exception a very small portion of the gasoline pool in 2016 (Figures 6.19 and 6.20).

One way that match blending would likely incentivize refiners to continue to blend ethanol is because they would not have the time to make the necessary capital investments to replace ethanol's volume, octane, and other of ethanol's favorable properties. The results of a refinery modeling study conducted by MathPro under subcontract for work conducted for EPA showed that to remove ethanol from the entire conventional gasoline pool in 2020 would require investments in 3 to 6 million barrels per



c/gal = cents per gallon; $/bbl = dollars per barrel

Figure 6.19. Relative ethanol blending cost (i.e., ethanol – gasoline) at actual ethanol volumes (left axis, green lines) and crude oil prices (right axis, black line); the min and max reflects the best and poorest blending markets across states for ethanol in the United States, respectively (2000–2018). Source: U.S. EPA (2022). Negative numbers indicate it was cheaper to make gasoline with ethanol at 10% volume than without.

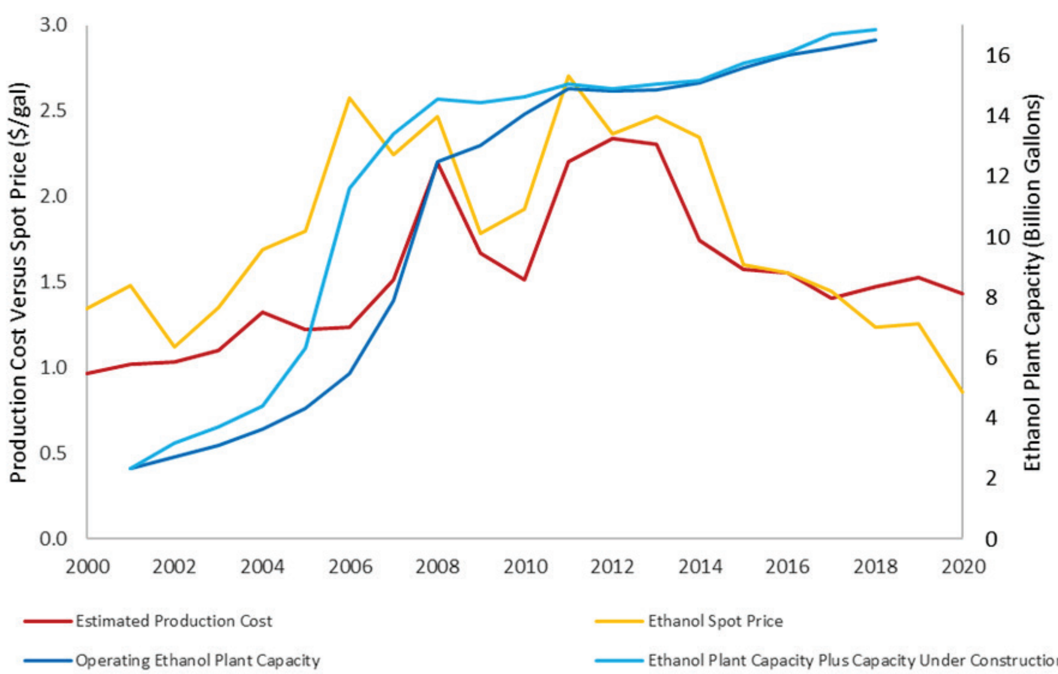


Figure 6.20. Comparison of estimated production cost to ethanol spot price and ethanol plant capacity increases, 2000 to 2018 (OTAQ model). Source: U.S. EPA (2022).

day of new refinery unit additions.[57] Before refiners would have considered making these new refinery capital investments on a large scale—knowing that it would require at least several years to implement—they would need some confidence that crude oil prices would remain low for at least several more additional years (i.e., at least 7 years total) to pay off the investments.[58] However, crude oil prices were not projected to remain this low. The EIA projected in its Annual Energy Outlook (AEO) that crude oil prices would immediately begin increasing again with crude oil prices reaching $70/barrel within only three years, providing insufficient time to even complete their capital investments, and certainly not enough time to pay off these investments. Another analysis conducted in 2016–2018 under subcontract for EPA examined the effect in 2020 of the hypothetical removal of the RFS Program in 2016 (Appendix D).[59] In the "No-RFS case" the use of ethanol and biodiesel and renewable diesel fuel in each PADD was governed not by the RFS mandates, but by the economics of gasoline, diesel fuel, and biofuel production and by state and local mandates for ethanol, biodiesel, and renewable diesel fuel use. The report found that after 2016 the volume of ethanol used in gasoline in the No-RFS case was estimated to be the same as in the Reference case, in which the RFS is in place. This result held for all PADDs and for all grades of gasoline. Biodiesel, on the other hand, was much more sensitive without the RFS Program (see Chapter 7). See Appendix D for further details.

This study (Appendix D) sought to understand if refiners would have continued to blend ethanol regardless of the RFS Program in periods of low crude oil prices when corn ethanol might look less economical. The study concluded that refiners would likely continue to blend ethanol even if ethanol was more expensive to blend, regardless of the RFS Program, due to the logistical inertia to continue to blend ethanol into gasoline. To revert back from producing BOBs to producing E0 would likely require coordination downstream of distribution system parties in a particular area that share the same gasoline distribution system. This effort would likely have to be coordinated among all these parties because terminals, at both refineries and parties downstream of refineries, do not have enough storage tanks to store both finished E0 and BOBs at the same time. Thus, all the associated refiners and downstream parties would need to agree to switch their gasoline production and tank storage from sub-octane BOBs to

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57 "Analysis of the Effects of Low-Biofuel Use on Gasoline Properties," MathPro Inc., June 7, 2019, prepared for ICF Incorporated, LLC under EPA Contract No. EP-C-16-020.

58 The refinery model found that these costs investments were necessary when refineries were operating near capacity. In the case that fuel demand is reduced, some and perhaps many of these investments might not be needed for refiners to produce gasoline without ethanol, which could only be determined by additional refinery modeling for the particular fuel demand scenario.

59 Modeling a No-RFS Case; ICF Incorporated; Work Assignment 0,1-11, EPA contract EP-C-16-020; July 17, 2018. Docket number: EPA-HQ-OAR-2019-0136, https://www.regulations.gov/document/EPA-HQ-OAR-2019-0136-2147.

finished gasoline at the same time. This is a challenging change and would require a significant amount of time to coordinate.

6.3.6 Synthesis of Evidence for the Effect of the RFS Program on Ethanol Production and Consumption

The effect of the RFS Program on ethanol production and consumption in the United States is dynamic, as it is a consumption mandate introduced to a dynamic market that responds to many factors that may change from year to year. The RFS Program is estimated to have had an effect on ethanol production in the years in which it was binding. According to observed D6 RIN prices, the RFS Program may have had an impact on ethanol production for one year over 2008/2009 and each year from 2013 to 2019. The RIN prices in question for ethanol, however, are the conventional biofuel D6 RINs. They are not specific to corn ethanol, but rather for any total renewable fuel under the RFS Program. There are other conventional biofuels, and due to the nested nature of the standard other advanced biofuels that can be used as well. In fact, the result of the higher D6 RIN prices in 2013–2019 was primarily to increase the blending of biodiesel and renewable diesel than to increase ethanol use above the E10 blend wall, as biodiesel and renewable diesel proved to be a lower cost option for refiners to demonstrate compliance with RFS standards. As a result, there is likely little association between RIN prices and ethanol consumption above the blend wall. Furthermore, RIN prices only reflect the effect of the RFS Program on production and consumption from *existing* biorefineries, they do not capture the potential effect of the RFS Program on establishing market certainty and stimulating the construction of new biorefineries. Thus, information from multiple sources must be synthesized to assess the effect of the RFS Program.

For each year from 2006 through 2018, the range of estimated effects varies and includes zero (Figure 6.21). The estimated effect in 2006 from the U.S. EPA (2022) study can likely be disregarded, as explained in section 6.3.5 and 6.2.3 the effect in 2006 was likely due to the sudden increase in the price of ethanol due to the new demand in non-California RFG areas to replace MTBE in summer 2006.

The effect in 2008–2011 was primarily driven by the RFS2 in accelerating market growth, and to a lesser extent by stabilizing demand through the crash in oil prices in 2008/2009. In the BSM, observed D6 RIN prices—without including risk reduction—suggest the effect of the RFS Program on ethanol production and consumption was estimated to be zero in 2008 and 0.02 billion gallons in 2009.[60] Adding

[60] This uses the preferred scenario for this purpose that includes octane value of ethanol and observed D6 RINs.

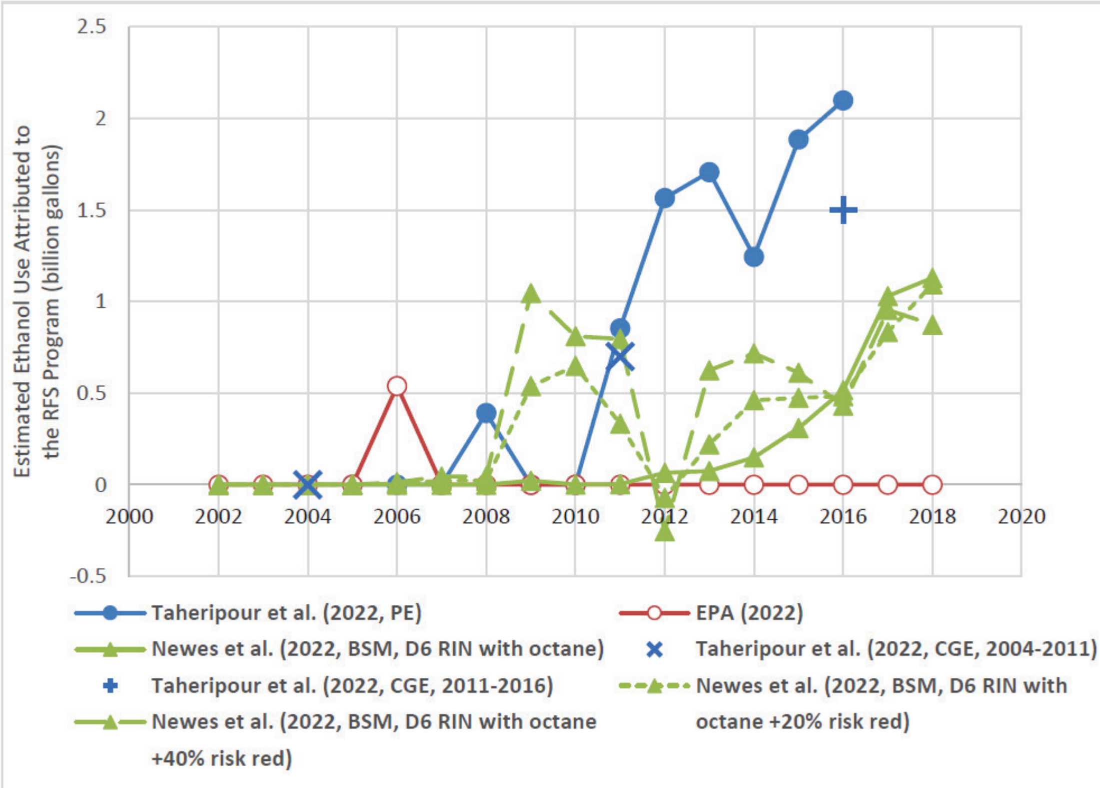

Figure 6.21. Comparison of attribution estimates among studies in section 6.3. Shown are estimates of the effect the RFS Program from Taheripour et al. (2022a) using the PE model (AEPE, blue line, filled circles) and using the CGE model for two periods (i.e., GTAP-BIO; effects from both mandates shown for 2004–2011, blue "x"; and 2011–2016, blue "+"), from Newes et al. (2022) using the BSM (D6 RINs with an octane value, green line, triangles)[61] and from U.S. EPA (2022) (red line). The estimate in 2006 from (U.S. EPA) is driven more by the MTBE phaseout than the RFS Program (see section 6.3.5). The estimates from Newes et al. (2022) show those that include the estimated effect the RFS Program had on increasing market certainty (dashed green lines, two levels of 20% and 40% risk reduction), and those that do not include this effect (solid green line).

the risk reduction effect to this, however, increases the estimate to 0–1 billion gallons. Taheripour et al. (2022a) report an effect of 0.4 billion gallons in 2008 using the PE model and no effect in 2009.[62] The OTAQ economic analysis (U.S. EPA, 2022) found no effect of the ethanol program on ethanol consumption in 2008 or 2009. All three assessments include the effects of oxygen, octane, and MTBE; thus, the authors see no strong reason to prefer one estimate over another on the basis of factors included or excluded. The BSM has more industry detail than the AEPE model, but the AEPE model is a true economic model rather than linear programming model. The OTAQ analysis has more economic detail than either the BSM or AEPE models on the fuel-side but has little detail on the agronomic-side. The

[61] The BSM estimates end after 2018 because there are no other estimates with which to compare to develop a range.
[62] The CGE model reported an effect of 0.7 billion gallons in 2011 compared with 2004. Annualizing the CGE results over this period would suggest an effect of roughly 0.4 billion gallons in 2008, identical to the PE results. However, because of the highly dynamic nature of the estimated effect of the RFS Program, affected by many other dynamic factors, the authors decline to use interpolated estimates from the CGE model here and instead use the point estimates for the year(s) simulated.

AEPE has the least fuel-side detail, but probably the most agronomic detail, and is a true economic model that solves for market clearing conditions across commodities. Each has strengths and weaknesses, and the estimates are similar and small relative to the 10–11 billion gallons of ethanol consumed in 2008 and 2009 (Figure 6.1). Thus, the synthesis of evidence suggests that from 2006 to 2011 the RFS Program—in isolation—accounted for 0–1 billion gallons of ethanol, potentially by encouraging market growth and capital investment from the RFS2 and to a lesser extent by stabilizing demand during the Great Recession of 2008–2009. In other years of this period, the RFS Program is estimated to have had no effect on ethanol production, with other factors having more influence.

The effect from 2012 to the present also varied by year and by study and included zero in the range each year. The PE results from Taheripour et al. (2022a) suggest a range of 1.2–2.1 billion gallons each year from 2012 to 2016, while the CGE results suggest an effect of 1.5 billion gallons in 2016 relative to 2011. Focusing on the results that include an octane value and observed D6 RIN prices, the BSM suggest a range as well, from 0.1 to 1.1 billion gallons, peaking in 2018 (Figures 6.16 and 6.21). The inclusion or exclusion of risk reduction does not influence the range of estimates over the entire interval, as the effect was estimated to dissipate by 2016 and the BSM is intermediate to other estimates. This suggests that the risk reduction effect of the RFS2 stimulated earlier growth of the industry, but not the industry's ultimate size. The OTAQ analysis suggested no effect of the RFS Program in most years, with a very small impact in 2016 (0.1 billion gallons).[63] The estimates from both PE and CGE approaches in Taheripour et al. (2022a) end in 2016, while for the BSM and OTAQ analysis they continue until 2019 and 2018 respectively. Figure 6.21 shows all the estimates together, demonstrating a range in effect from year to year, with smaller ranges and estimates in earlier years and larger ranges and estimates in later years. Only the MathPro analysis (Appendix D) explicitly includes the costs to refineries to revert from producing BOBs back to finished gasoline, which found zero effect of the RFS Program.[64] Thus, it is reasonable to include zero in the estimate of the range—refineries have already made the investments toward the production of BOBs, and with the CAA emission requirements there is little reason to reverse course. As before, the authors see no strong reason to prefer one estimate over another. Thus, the synthesis of evidence suggests a dynamic range of effects from the RFS Program from 2012 to 2019, with the largest effect in 2016 (0–2.1 billion gallons) primarily due to the RFS Program supporting the industry after other factors had either phased out (e.g., VEETC, MTBE) or diminished in effect (e.g., oil prices).

[63] This analysis considers only the degree to which ethanol would be profitable to blend without the RFS Program. It does not consider the ability of the market to replace ethanol with high-octane petroleum blendstocks. As discussed in this report, after accounting for these factors the impact of the RFS Program is very likely much small volume and may even be zero.
[64] The BSM implicitly includes this transition to match blending (Newes et al., 2022), but not explicitly in the costs to refineries.

Over both periods, from 2005 through 2018, the analysis above suggests a cumulative amount 0–9% of ethanol production or consumption is estimated to be attributable to the RFS.

6.3.7 Limitations of the Assessment

The evidence summarized in this chapter is generally based on modeling results and analyses that use historical economic data on significant factors such as the prices of corn and crude oil. There are several limitations in the modeling and analyses that may underestimate the impacts of the RFS Program. Many of these models estimate equilibrium conditions where all markets are cleared. They do not include people in them making decisions, and thus they may underestimate the role the RFS Program played in increasing investor confidence in ethanol production plants by providing a guarantee that there would be a government-mandated demand for ethanol in future years. The RFS Program provided a level of certainty that there would be a domestic market for ethanol even if crude oil prices dropped to pre-2004 levels and ethanol was not economically competitive with gasoline. This effect was attempted to be captured with the risk reduction scenarios in the BSM, but more research in this area is needed. The market certainty provided by the RFS Program is assumed to have accelerated the buildout of the ethanol industry and infrastructure. Much of this appears to have been well underway by 2007 and certainly by 2010 (Figure 6.6) (Denicoff, 2007). The RFS Program may also have encouraged refiners and fuel distributors to make the changes necessary to produce and distribute BOBs (rather than finished gasoline), which further increased the economic competitiveness of ethanol. Had oil prices been lower than those observed since 2005, or had corn prices been higher, the RFS Program would have had a more significant impact on ethanol production and consumption during that time period. All of these factors may be important but are difficult to capture in traditional modeling frameworks and thus are not quantified in the literature. Future research is needed to quantify these market effects. That said, the purpose of the Section 204 Report is to assess the impacts to date of the RFS Program *as it occurred*, not what the impacts of the RFS Program might have been under alternate conditions such as those that existed when the original legislation was drafted and enacted.

Conversely, there are also a number of limitations in the modeling and analysis that may overestimate the potential impacts of the RFS Program. The modeling was not able to take into consideration the temporal nature of the market buildout of production capacity prior to the RFS mandates. The fact that the market buildout preceded the RFS Program could indicate that the RFS Program merely codified what the market was expecting. Whether the RFS Program or the MTBE phaseout—or both—drove these infrastructure changes is a key remaining uncertainty that would increase confidence in the estimates of attribution. In addition, the modeling was not able to account for the significant hurdles associated with reverting back to E0 from E10 during periods when the market might otherwise choose to do so. Only U.S. EPA (2022) explicitly considered the costs to refiners and distributors of gasoline to switch back to producing finished gasoline rather than BOBs. If ethanol were to be removed from gasoline, to maintain production refiners would have to not only replace the lost volume as ethanol but also adjust their refining operations to produce gasoline that meets the minimum octane and emissions requirements of the CAA without ethanol. While refiners could likely produce some quantity of finished gasoline using existing equipment, recent refinery modeling conducted by MathPro on behalf of EPA (Appendix D) concluded that if ethanol was removed from the entire conventional gasoline pool, refiners would have to invest significant capital in some combination of alkylation, isomerization, and reforming units to meet the minimum octane requirements without the addition of ethanol.[65] There would also be costs associated with making the necessary adjustment to the distribution system to accommodate both finished gasoline and BOBs. If refiners anticipated that the lack of cost-competitiveness of ethanol with gasoline in some markets in recent years was likely to be a temporary phenomenon, they may have continued to blend ethanol in these markets even in the absence of the RFS Program to avoid these capital costs. Finally, some of the estimates for the effect of the RFS Program for years after the E10 blend wall was reached (e.g., from Taheripour et al. 2020 after 2012) may be high. These estimates imply that in the absence of the RFS program refiners and blenders would revert to supplying finished gasoline without ethanol (E0) in some parts of the U.S. This seems unlikely due to refiners' reliance on ethanol for blending into their gasoline to meet applicable fuel standards. With the exception of U.S. EPA, 2022, these estimates do not appear to consider the feasibility nor the costs of replacing ethanol in gasoline with something else to meet applicable fuel standards and thus they appear to over-estimate the impact of the RFS program in years after 2012. The RFS Program is likely much more responsible for the portion of ethanol consumed above the E10 blend wall rather than below it (as suggested by U.S. EPA, 2022).

Overall, this analysis suggests the RFS Program had a relatively modest effect during the period of major growth of the industry and a larger effect more recently as other factors have diminished in influence (e.g., oil price, VEETC). The replacement of MTBE with ethanol in RFG areas appears to have been the most likely outcome with or without the RFS Program given the information at the time and maintaining the CAA emissions requirements. California had already transitioned largely by 2003, and the rest of the country rapidly followed suit in 2005–2006. These additional demands in RFG areas are largely in coastal areas, which would have created incentives for infrastructure buildout. These years generally precede the RFS1, which went into effect in 2006, a year where the only two available studies with sufficient industry detail estimate no binding effect (Newes et al., 2022; Taheripour et al., 2022a).

[65] "Analysis of the Effects of Low-Biofuel Use on Gasoline Properties," MathPro Inc., June 7, 2019, prepared for ICF Incorporated, LLC under EPA Contract No. EP-C-16-020. See Appendix D (for summary report) or Appendix C (for all associated information) in Fuel Supply Defaults: Regional Fuels and the Fuel Wizard in MOVES3", March 2021, EPA-420-R-21-006. https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P10119R7.pdf

Based on the discussion above, the replacement of MTBE with ethanol was not due to the RFS Program per se, though that replacement in federal RFG areas was triggered in the same originating EPAct. That said, the existence of the Program was likely an added incentive to blend ethanol, and it created a safe market space for the development of the ethanol industry. Had oil prices remained low, as was expected throughout 2004–2007, the RFS would likely have had a direct and large effect stimulating the growth of the industry. But as events unfolded, this analysis concludes the RFS Program in isolation drove a relatively small fraction of the ethanol produced and consumed in the United States, in contrast to common perception.

6.4 Evidence of the Impact to Date of the RFS Program on Corn Production and Cropland

The effect of the RFS Program on corn production is manifest mainly through the intermediate effect on corn ethanol (addressed in section 6.3) and subsequently on demand for corn, which can have a variety of effects in the corn market. Thus, this section begins with the conclusions of section 6.3—the best available information suggests the RFS Program affected corn ethanol and thus corn production in approximately 2008–2011 (0–1.0 billion gallons) and each year from 2012 to 2018 (with a maximum effect of 0–2.1 billion gallons in 2016). These are volumes of corn ethanol that, based on section 6.3, may not have been produced without the RFS Program. As a shorthand, this section refers to this range of corn starch ethanol volumes attributable to the RFS as "RFS-attributable ethanol." In this section the potential effects of RFS-attributable ethanol on U.S. corn and crop acreage are examined. In other words, how much "additional" corn and crop acreage was there compared to a counterfactual scenario absent RFS-attributable ethanol?

The range of additional corn and cropland resulting from RFS-attributable ethanol includes zero on the low end based on the finding in section 6.3 that the RFS may have been responsible for zero additional gallons of U.S. ethanol production. The authors have found no evidence that RFS-attributable ethanol caused a reduction in U.S. corn or cropland. The upper end of the range is based on the year 2016, as that was the year with the single largest potential volume of RFS-attributable ethanol over the period examined from 2005 to 2018 (see Figure 6.21). The highest estimate for any single year is 2.1 billion gallons of RFS-attributable ethanol based on the PE modeling in Taheripour et al. (2022a). Focusing on this one-year high-water mark is useful for this evaluation of land effects, as the maximum estimate of area of corn and cropland attributable to the RFS Program is a necessary intermediate estimate that flows into the assessment of environmental effects in the chapters that follow.

Translating ethanol production to effects on corn and cropland area is critical for the environmental effects addressed in this report series. However, this step is not straightforward. This is in large part because a share of ethanol is produced from land that was already used to grow corn or in corn rotation; and, any corn used for ethanol results in the coproduction of wet or dry distiller's grains for livestock feed, which offsets some of the corn that otherwise may be needed for livestock feed. Any effects on corn and crop areas are therefore the result of many factors, including but not limited to global and regional crop demands, shifts in crop production (e.g., corn replacing other crops), prices, other policies such as acreages for the Conservation Reserve Program (CRP), crop insurance, and other market-mediated effects (Hendricks et al., 2014; Hertel et al., 2010; U.S. EPA, 2010).

Two general types of studies are discussed in this section to assess the effect of the RFS Program on corn and crop area in the United States: (1) simulation modeling studies (including but not limited to those discussed in section 6.3) and (2) correlational or statistical studies.[66] Simulation models are useful in that they can isolate the estimated effect of the RFS Program and can include direct and indirect effects of a given increase in ethanol demand.[67] However, simulation models often have relatively coarse spatial resolution (e.g., treating the United States as one region or consisting of a few very large regions), rely on assumptions for which supporting data are limited, and have other limitations, as discussed in Chapter 4. These characteristics suggest that simulation models may be well-suited to explore potential land-cover-land-management (LCLM, see Chapter 5) changes from a given policy (direct and indirect), especially into the future where no observational data exists, but estimating *where* these changes took place in the past may be limited using this approach. Furthermore, as with the evaluation of how well simulation models characterize the fuels industry in section 6.3 (e.g., MTBE, match blending), an evaluation of the strengths and weaknesses of models assists with the interpretation of the reliability of the estimates on LCLM for this report. Many models that have good characterization of LCLM and biogeochemical processes, are weak in their characterization of the fuels industry; and, many models that have good characterization of the fuels industry, are weak in their characterization of LCLM.[68] Thus, it may not be optimal to use a single lifecycle modeling framework for the specific purposes in this report (NASEM, 2022).

Statistical studies also have different strengths and weaknesses for the purpose of attributing corn and crop area to the RFS Program. Statistical studies on land use change and biofuels are often derived via statistical/econometric methods that relate a given LCLM response (e.g., re-enrollment in CRP, non-crop to crop conversion, non-corn to corn conversion) to a given treatment (i.e., ethanol plant proximity, ethanol plant capacity, crop price). Compared to simulation modeling, statistical studies rely more heavily

[66] These are not necessarily mutually exclusive since simulation models often use correlational associations in their parameterization. They are merely useful categories for presenting large amounts of literature and information.
[67] See section 5.2 for a more detailed discussion of direct and indirect land use change effects.
[68] See the Model Comparison Exercise (MCE) Technical Document in the Final Set Rule for more discussion on these models (EPA-HQ-OAR-2021-0427).

on observed data and are often at a much smaller spatial resolution (e.g., 30 meters if using the National Land Cover Database or the Cropland Data Layer after 2008). Statistical approaches can be designed to control for many confounding influences on land use change in order to isolate the potential influence of the treatment. Importantly, statistical and spatial studies are often very data intensive (both spatially and topically). Because of this, and because data from other regions or countries of interest are often not available and the same level of detail, most statistical studies on LCLM in the United States only cover the United States. Thus, international effects are often evaluated with simulation models (see section 6.4.1.2 and Chapter 16). As with simulation modeling studies, omission of important variables in a statistical study can bias the results toward finding an association with variables that were included instead of variables that were not included, but may have been causal. However, statistical studies cannot typically estimate the direct effect of a policy on changes in LCLM. Instead, they often use a treatment (e.g., the number/size of ethanol plants, crop prices) as a proxy for the effect of a biofuel policy based on the hypothesis that the treatment has a causal relationship to domestic biofuel policy. Thus, statistical studies often do not disentangle the underlying causes of the ethanol plant production, or crop prices, which are a broad range of factors such as crude oil prices, state-level biofuel mandates, the RFS Program, and other factors discussed in section 6.3. Because of this, many of these studies are useful for estimating the LCLM change effects associated with increased biofuels production broadly—irrespective of cause—but not caused by the RFS Program specifically. That said, together with the evaluation in section 6.3 of the RFS Program on ethanol production, it is the combination of simulation and correlational studies that may be leveraged to estimate the effect of ethanol production on corn and cropland area.

These lines of evidence are discussed in the sections that follow, but starts with an illustrative example to provide some sense of the magnitude of area that may be affected. As an illustration of how much corn it would take to produce the aforementioned volumes of ethanol, Chapters 4 and 5 show that on average 1 billion gallons of ethanol are produced from 0.36 billion bushels of corn. Using the estimated maximum effect in 2016 of 2.1 billion gallons suggests that RFS-attributable ethanol may have consumed 0–756 million bushels of corn in 2016. That represents 0–5.0% of the corn production in 2016. The actual rate of conversion of bushels of corn to gallons of ethanol varies by the technologies employed at specific biorefineries and improve over time as efficiencies improve. The acreage needed to produce these bushels of corn depends on corn yield, which in turn depends on many factors including site fertility, irrigation, tillage, farmer decisions, and other factors discussed in Chapter 3. Generally, the same number of bushels of corn would take less land in a more productive area like Iowa and more land in a less productive area like South Dakota.

Using average yields for the country suggests that 0–756 million bushels of corn could be produced on 0–4.8 million acres of corn, or 0-5% of the planted corn acreage in 2016.[69] These theoretical acreages are merely illustrative. Internal adjustments in other domestic uses of corn can account for the required additional supply without requiring additional planted acres (Oladosu et al., 2011). Production may also be increased through more frequent corn in rotations on existing fields, or new fields established on lands that either grew other crops or were uncultivated. Identifying precisely when and where any additional corn acreages attributable to the RFS Program are located, and what land cover and land management practices were displaced, are critical in determining any associated environmental effects. Setting aside this simple illustrative example, this section turns to available estimates of how much corn and crop land may be attributable to the RFS Program's effects on corn ethanol production.

6.4.1 Simulation Modeling

The simulation modeling studies published to date, like any literature, vary in terms of their utility for the purposes of this report. For instance, different studies report on different time periods and/or magnitudes of ethanol production increase, include different combinations of policy factors either alone or in combination (e.g., VEETC and RFS Program), among other differences. The simulation modeling literature is subdivided here into three groups: (1) retrospective studies that account for many factors known to affect biofuel production and which cover the bulk of the timeframe of the RFS Program, (2) prospective and retrospective studies that isolate the effect of corn ethanol production (as opposed to all biofuels or hypothetical scenarios) on land use change, and (3) the broader literature which varies widely in detail and scope. The first group is the most directly focused on the subject of this chapter (i.e., the actual effects of the RFS Program on LCLM), though individual studies may be few in number. The second and third sets represent less and less specificity for the purposes of this chapter, but likely include more studies. If these three sets generally agree, there may be more confidence in the conclusions.

6.4.1.1. Retrospective Studies that Account for Many Factors Known to Affect Biofuel Production

Only two retrospective studies, to the authors' knowledge, as discussed in section 6.3, include estimates in changes in land use that account for the octane value of ethanol, the MTBE phaseout, crude oil price, and covered the bulk of the timeframe of the RFS Program (Newes et al., 2022; Taheripour et al., 2022a). The CGE modeling in Taheripour et al. (2022a) simulated that about 1 million cropland acres would have gone out of production in the absence of the RFS Program from 2004 to 2011. The same study simulated that approximately 160,000 cropland acres would have dropped out of production from

[69] From 2013 to 2019, an average of 14.3 billion bushels of corn were produced on 91 million acres of corn, for an average yield of 157 bushels per acre. Thus, 0–756 million bushels of corn is 0–4.8 million acres of corn (USDA, NASS). Planted corn acreage in 2016 was 94.0 million acres.

2011 to 2016 in the absence of the RFS Program (i.e., the combined ethanol and biodiesel mandates).[70] Based on the magnitude of the ethanol mandate simulated, this translates to 0.08 and 0.07 million additional cropland acres per billion gallons of ethanol in 2004–2011 and 2011–2016, respectively.[71] Taheripour et al. (2022a) did not report acreages of different crops, but did report changes in production for different crops for 2004–2011 (Table 6.6) and 2011–2016 (Table 6.7). They found that no expansion of corn ethanol had a much larger effect on coarse grains than removal of RFS mandates from 2004 to 2011 (Table 6.6), and that the effects of removing either from the model were smaller and comparable in the latter period (2011 to 2016, Table 6.7). The Taheripour study was focused on the economic effects of the RFS Program, rather than the land use change effects (Taheripour et al., 2022a). The coarse resolution of the model (i.e., smallest spatial scale is several U.S. states) precludes a spatial accounting of effects on land. Annual estimates on land use from the PE modeling were not reported.

Newes et al. (2022) do not include an estimate of total cropland,[72] but they found small or no effects of the RFS Program on corn acreage in the 2006–2012 period (nil including octane, Figure 6.22). Effects varied by year thereafter, from 0.0 to 0.6 million acres in 2012–2016, and a peak value of 2.6 million in 2018 (all values reported include octane) (Figure 6.22). These effects on corn acreage were mirrored by effects on hayland, with decreases in hayland only in the later period (2012–2019, Figure 6.22).

The strength of both modeling approaches (GTAP-BIO and BSM) is in the industry detail for the sector of focus here, biofuels. Unfortunately, these models are not the best tools available to assess changes in LCLM. In particular, common parameterizations of GTAP-BIO (including those used here) preclude conversion of lands in the CRP, despite observations of conversion of these lands (see Chapter 5, Morefield et al. (2016), Bigelow et al. (2020)). As is the case for all models of biofuel-induced LCLM, there are uncertainties and limitations associated with the GTAP-BIO estimates and areas for further research.(Taheripour et al., 2021; Malins et al., 2020) The strength of the BSM is on the fuels side as well, as there is limited represented of cropland in the BSM (Peterson et al., 2019). Thus, this review includes other models and approaches that are better suited for analyses of LCLM.

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[70] Taheripour et al. (2022a) report that from 2004 to 2011, 6.3 million acres of cropland would go out of production without the expansion of biofuels, 16% of which was estimated attributable to the RFS Program; and, from 2011 to 2016, 160,000 acres of cropland would come out of production, all of which was estimated attributable to the RFS Program. This study separated the effect of the corn ethanol implied mandate and the biodiesel mandate, but for the land responses only the combined effect was reported. The CGE modeling in Taheripour did not report corn acreage, only total cropland; and did not report corn production specifically, but rather coarse grains. Coarse grains include many grains like corn, barley, sorghum, and oats, but in the United States are dominated by corn.

[71] From 2004 to 2011 the implied corn ethanol mandate increased from zero to 12.6 billion gallons. Thus, a 1 million acre increase in total cropland is 0.08 million acres per billion gallons. From 2011 to 2016 the implied ethanol mandate increased 2.4 billion gallons (from 12.6 to 15.0 billion gallons). Thus, a 160,000 acre increase in total cropland is 0.07 million acres per billion gallons.

[72] There is no “total cropland” in the BSM, which only models five “crops” (corn, soy, wheat, cotton, other grains).

Table 6.6. Percentage change in crop production under alternative counterfactual experiments for 2004–2011, from removal of (1) the RFS-implied corn ethanol mandate, (2) the RFS-implied corn ethanol and biodiesel mandates, (3) the increase in corn ethanol production, (4) the increase in ethanol and biodiesel production. Source: Taheripour et al. (2022a).

Description	Removing Mandate of Corn Ethanol	Removing Mandates of Corn Ethanol & Biodiesel	No Expansion in Corn Ethanol	No Expansion in Biofuels
Coarse grains	-1.2	-1.4	-20.8	-20.8
Soybeans	0.2	-1.6	3.2	0.1
Wheat	0.1	0.6	2.4	3.0
Rice	0.0	0.2	0.7	1.0
Sorghum	0.0	0.0	0.6	0.6
Rapeseed	0.2	-12.4*	5.6	-11.0
Other oilseeds	0.1	-4.3	1.8	-3.6
Sugar crops	0.0	0.1	0.4	0.4
Other crops	0.1	0.0	1.0	0.8

* The large percentage changes for rapeseed are due to very small quantities in the base year.

Table 6.7. Percentage change in crop production under alternative counterfactual experiments for 2011–2016, from removal of (1) the RFS-implied corn ethanol mandate, (2) the RFS-implied corn ethanol and biodiesel mandates, (3) the increase in corn ethanol production, (4) the increase in ethanol and biodiesel production. Source: Taheripour et al. (2022a).

Description	Removing Mandate of Corn Ethanol	Removing Mandates of Corn Ethanol & Biodiesel	No Expansion in Corn Ethanol	No Expansion in Biofuels
Coarse grains	-2.0	-1.6	-1.6	-1.6
Soybeans	1.1	0.3	0.9	0.5
Wheat	0.8	0.4	0.6	0.3
Rice	1.1	1.2	0.9	1.1
Sorghum	0.0	0.5	0.0	0.4
Rapeseed	1.9	1.5	1.5	1.6
Other oilseeds	1.0	0.5	0.8	0.7
Sugar crops	0.6	0.5	0.6	0.6
Other crops	0.4	0.3	0.3	0.3

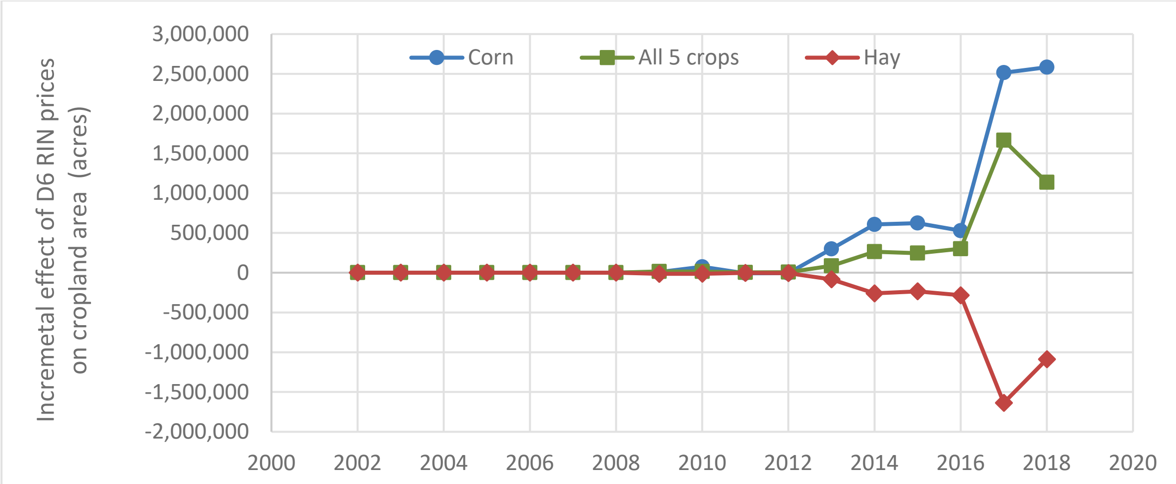


Figure 6.22. Simulated incremental effect of the RFS Program as represented by D6 RINs on acreages of corn, hay, and the sum of all five crops (i.e., corn, soybean, wheat, cotton, other small grains) modeled in the BSM (includes the effect of octane, excludes risk reduction, scenario G-F).

6.4.1.2. Prospective and Retrospective Studies that Isolate the Effect of Corn Ethanol Production

As part of the Agency's analysis of climate change impacts for the Set Rule, EPA compiled biofuel lifecycle greenhouse gas (GHG) estimates from the scientific literature. This included a review of published estimates of the land use change impacts of corn ethanol production, which was published in the Draft Regulatory Impact Assessment (DRIA) for the Set Rule, and portions relevant for the RtC3 are summarized here.[73] The review in the DRIA was focused on GHG analyses, but many of the same models estimate changes in LCLM in the process of the lifecycle GHG analysis, as changes in LCLM are one of the primary drivers of net GHG emissions

As part of the DRIA literature review, the LCLM changes from corn ethanol for various models were reviewed and normalized on a per billion-gallon basis (Figure 6.23). For dynamic models, results are reported for the peak year of the biofuel shock. No other efforts to harmonize these estimates were made. Note that caution is needed when interpreting these figures for many reasons including the divergent scenarios modeled and differences in the definition of cropland between models.

For corn ethanol, there is relatively wide variation across studies in the amount of total cropland expansion estimated per gallon of biofuel production. Across these models that separate U.S. from non-U.S. cropland expansion, estimates show an increase of 0.2–1.0 million acres per billion gallons

[73] Full text of the DRIA is available at https://www.epa.gov/renewable-fuel-standard-program/proposed-renewable-fuel-standards-2023-2024-and-2025 and under docket EPA-HQ-OAR-2021-0427. The most detailed review of land use change estimates was included in the DRIA. The DRIA content remains relevant, although EPA decided this review did not bear repeating in the final RIA. This review is intentionally not included in section 6.3 because these studies were not designed to separate the effects from the RFS Program from the effects of biofuels more generally. Many of these models lack industry detail for such an analysis. Their strengths, rather, are in the integration of effects across sectors and regions associated with biofuel production and consumption.

(Mac/Bgal) domestically and 0.25–0.75 Mac/Bgal outside of the country. Across all the models, the global impact on cropland area range is 0.19-1.3 Mac/Bgal. Across all models, the largest estimated effects domestically come from the Global Change Analysis Model (GCAM) (Plevin et al., 2022) and the smallest from GTAP-BIO (Taheripour et al., 2017). Internationally, the largest effects come from FASOM-FAPRI (RFS2 RIA), and the smallest effects from GCAM (Plevin et al., 2022).[74]

In parallel with the literature review conducted for the DRIA, EPA hosted a virtual public workshop on biofuel GHG modeling February 28–March 1, 2022.[75] The purpose of the workshop was to solicit information on the current scientific understanding of GHG modeling of land-based crop biofuels used in the transportation sector. The associated public workshop included presentations on five models [The Greenhouse Gases, Regulated Emissions, and Energy Use in Technologies Model (GREET), Global Biosphere Management Model (GLOBIOM), Global Change Analysis Model (GCAM), Global Trade Project (GTAP), and Applied Dynamic Analysis of the Global Economy (ADAGE)]. This section provides a brief overview of key similarities and differences among these models (Table 6.8). These models may be used to support either consequential or attributional lifecycle analyses.[76] A more detailed summary of the models is provided in Appendix C and in the DRIA. This selection of models provides a broad cross-section of the most common types of modeling frameworks used to assess biofuels, as discussed in the following paragraph. These models are highlighted based on discussions with USDA and DOE and EPA's experience reviewing scientific literature on the lifecycle GHG emissions of biofuels. Furthermore, this model review was guided by the decision in the 2010 RFS2 rule to include significant indirect effects occurring anywhere in the world (i.e., international impacts) given that potential environmental impacts are global.

There are four general types of models commonly used for biofuel GHG analysis: lifecycle inventory (LCI) models, partial equilibrium (PE) models, computable general equilibrium (CGE) models and integrated assessment models (IAM). LCI models, such as GREET, are designed to estimate in detail the inputs and outputs of a product supply chain, using rule-based methods (i.e., allocation or displacement) to account for co-products. PE models, such as GLOBIOM, equate supply and demand in one or more markets such that prices stabilize at their equilibrium level. PE models offer highly detailed

[74] The potential international effects are discussed in Chapter 16.

[75] https://www.epa.gov/renewable-fuel-standard-program/workshop-biofuel-greenhouse-gas-modeling

[76] The recent NASEM report (2022) summarizes the differences between these approaches: "there are two broad categories of LCA that are relevant to this report and require such fundamentally different approaches that they are important to discuss in greater detail: attributional LCA and consequential LCA. ALCA is defined by "environmentally relevant physical flows to and from a life cycle and its subsystem" (Finnveden et al., 2009). ALCA seeks to attribute a portion of total observed environmental impacts from human activities to the provision of a specific good or service. In contrast, CLCA is defined by its aim to describe "how environmentally relevant flows will change in response to possible decisions" (Finnveden et al., 2009). In other words, CLCA captures the consequences of some change in the provision of goods or services."

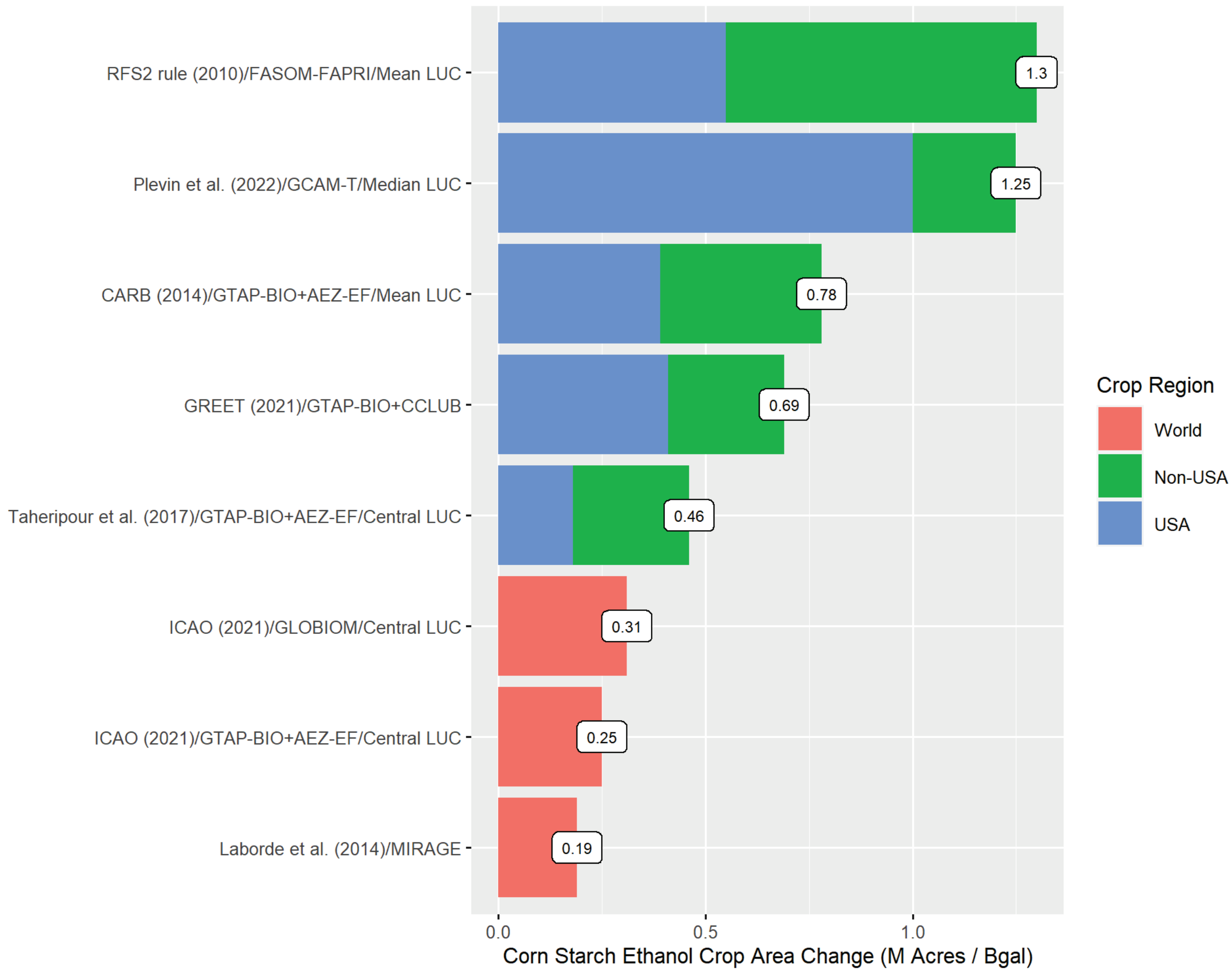


Figure 6.23. Cropland area change estimates per billion gallons of ethanol by study for corn ethanol. The name on the y-axis for each bar/estimate includes multiple descriptors separated by "/". In order, these descriptors are the author or other name (e.g., RFS2 rule), the name of the model used to estimate land use change impacts, and a brief descriptor of the scenario modeled. For studies that did not report disaggregated estimates by U.S. and Non-U.S., only the world total is reported. Scenarios modeled and definitions of cropland differ across studies. ICAO (2021) estimates for corn ethanol to jet fuel were adjusted based on the assumed jet fuel yield. Data sources: CARB (2014), U.S. EPA (2010), ICAO (2021), Laborde et al., Plevin et al. (2022), Taheripour et al. (2017). Reproduced from the DRIA (EPA-HQ-OAR-2021-0427).

representations of one or a few sectors of the economy, such as the agricultural sector, but lack linkages to other sectors of the economy. In contrast, CGE models, such as GTAP and ADAGE, are comprehensive in their representation of the economy, reflecting feedback effects among all economic sectors and factors of production, such as capital and labor. IAMs, such as GCAM, integrate knowledge from several disciplines—for example, biogeochemistry, economics, engineering, and atmospheric science—to evaluate how changes in any of these areas affect the others. While it is hard to state the specific criteria for identifying an IAM, one could distinguish them from PE and CGE models by their deeper integration of human economic systems with Earth (biosphere and atmosphere) systems and GHG emissions into one modeling framework. PE, CGE and IAM models can all be called economic models. LCI models are categorically different from the other three model types as they do not simulate economic behavior or prices. Across the four model types there tends to be a tradeoff between scope and detail, which are summarized in Table 6.8 and discussed in more detail in the Final Set Rule.[77]

The goals of this exercise were to (1) advance the science in the area of analyzing the lifecycle greenhouse gas emissions impacts from increasing use of biofuels; (2) identify and understand differences in scope, coverage, and key assumptions in each model, and, to the extent possible, the impact that those differences have on the appropriateness of using a given model to evaluate the GHG impacts of biofuels; and (3) understand how differences between models and data sources lead to varying results. These outcomes are summarized in the Model Comparison Exercise (MCE) Technical Document.[78] The MCE is scientific in nature and an ongoing activity and does not directly affect implementation of the RFS Program or the estimates of attribution in the RtC3.

Table 6.8. Comparison of Key Characteristics Across Models.

Characteristic	GREET	GLOBIOM	GTAP-BIO	ADAGE	GCAM
Type of Model	Lifecycle inventory (LCI)	Partial equilibrium (PE)	Computable general equilibrium (CGE)	Computable general equilibrium (CGE)	Integrated assessment model (IAM)
Type of LCA analysis typically conducted	Attributional[79]	Consequential	Consequential	Consequential	Consequential
Sectoral Coverage	Fuel supply chains including energy resource and material inputs	Agriculture, forestry, and bioenergy	Economy-wide with 57 sectors	Economy-wide with 36 sectors	Energy (conventional and renewable), agriculture, forestry, water
Temporal Representation	Static	Recursive dynamic (10-year time steps)	Comparative static	Recursive dynamic (5-year time steps)	Recursive dynamic (5-year time steps)
Regional Coverage	Customizable (typically U.S. average)	37 economic regions; 10,000 spatial units (grid cell)	19 economic regions; 18 agro-ecological zones	8 economic and spatial regions	32 economic regions; 235 spatial regions (water basins)
GHG Emissions Coverage	Direct supply-chain emissions + indirect land use change from CCLUB module	Crop production, livestock and land use change	Land use change GHGs calculated with CCLUB or AEZ-EF modules	Economy-wide GHGs including land use change	Global GHGs including land use change
Land Representation (Arable land categories considered in biofuel land use change analysis)	Exogenous (Land use change estimates from GTAP-BIO and CCLUB)	Cropland, other agricultural land, grassland, commercial and non-commercial forest, wetlands, other natural land	Cropland (including cropland-pasture), livestock pasture, “accessible” forestry land	Cropland, pasture, commercial forest, non-commercial forest, natural grassland, other land	Cropland, commercial pasture and forest, non-commercial pasture and forest, shrubland, grassland, “protected” non-commercial land

[77] Additional information on these models are also presented in the Model Comparison Exercise as part of the Final Set Rule (EPA-HQ-OAR-2021-0427).
[78] Available at https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1017P9B.pdf
[79] GREET incorporates induced land use change GHG emissions estimates from a separate module called the Carbon Calculator for Land Use change of Biofuels (CCLUB) which incorporates a consequential analysis of land use changes from the GTAP-BIO model.

6.4.1.3. The Broader Literature

In addition to the literature discussed above and in Chapter 4, Austin et al. (2022) conducted a targeted review of the effect of the RFS Program on land cover and land management in the United States and found that CGE studies tended to have the lowest estimates, empirical the highest, and PE were intermediate (Table 6.9, Figure 6.24). The empirical estimates all clustered close to one another (with one minor outlier), while the PE and CGE modeling approaches both had significant outliers. Across all studies, the range of estimated increases in cropland in the United States was 0.01–0.95 Mac/Bgal (Table 6.9).[80] Using this estimate, along with the estimated effect of the RFS Program on ethanol production (i.e., 0–1 billion gallons in 2008–2011 and 0–2.1 billion from 2012–2019), suggests cropland expansion of 0–0.95 million acres in 2008–2011 and a maximum estimated effect in 2016 of 0–2.0 million acres (Table 6.10).

Table 6.9. Individual estimates from Austin et al. (2022). Shown are the biofuel induced net cropland expansion in the United States by study, the associated simulated increase in biofuel volumes, and the normalized effect (million acres per billion gallons per year) for PE, CGE, and empirical estimates. Studies are ordered in reverse chronology and summarized by type at the bottom.

Study	Model Name and Type[1]	Time Period	Biofuel Induced Net Cropland Expansion in the U.S. (million acres)	Proportion of Net Cropland Expansion from CRP Land (if reported)	Simulated Increase in Biofuel Volumes from the RFS (BGY)	Net Cropland Expansion per Increase in Biofuel Volumes from the RFS (million acres/ BGY)
Lark et al. 2022	Empirical	2008–2016	5.2		5.5	0.95
Chen et al. 2021	BEPAM (PE)	2016–2030	4.7		8.5	0.55
Taheripour, Baumes, and Tyner 2020	GTAP-BIO (CGE) and AEPE (PE)	2004–2011	0.06		0.7	0.09
		2011–2016	0.02		1.5	0.02
		2004–2016	0.09		2.2	0.04
Khanna, Wang and Wang 2020	BEPAM (PE)	2007– 2017	1.2		8.5	0.14
		2017–2027	1.2		0	
Li, Miao, and Khanna 2019	Empirical	2003– 2012	6.9		10.4	0.66
		2003–2014	7		11.5	0.61
		2008–2012	2.1		4.2	0.50
		2008–2014	2.3		5.3	0.43
Chen & Khanna 2018	BEPAM (PE)	2007–2012	3.15	31%	6.7	0.47
Taheripour, Zhao, and Tyner 2017	GTAP-BIO (CGE)	2011– 2015	0.01		1.1	0.01
Wright et al. 2017	Empirical	2008–2012	2.7		4.2	0.64
		2009–2012	0.99	75%	3	0.33

[80] This estimate uses the corrected version of Table 3 in Austin et al. (2022) and reports on the min-max after excluding two notable outliers (Figure 6.24). Table 6.9 and Figure 6.24 also use the corrected version.

Study	Model Name and Type[1]	Time Period	Biofuel Induced Net Cropland Expansion in the U.S. (million acres)	Proportion of Net Cropland Expansion from CRP Land (if reported)	Simulated Increase in Biofuel Volumes from the RFS (BGY)	Net Cropland Expansion per Increase in Biofuel Volumes from the RFS (million acres/ BGY)
Bento, Klotz, and Landry 2015	Multi-market equilibrium model (PE)	2012–2015	1.48	84%	3	0.49
CARB 2014	GTAP-BIO (CGE)	2004–2017	4.45		11.6	0.38
Elliot et al. 2014	PEEL-Co (PE)	2010–2022	7.2		10.1	0.71
Mosnier et al. 2013	GLOBIOM (PE)	2010–2020	7.4		9	0.82
Oladosu & Kline 2013	GTAP-DEPS (CGE)	2001–2030	3.2		11.6	0.28
Cai et al. 2013	ADAGE-Biofuel (CGE)	2010–2022	3.7		18.6	0.2
EPA 2010	FASOM (PE)	2008–2022	8.1	65%	17.1	0.47
Hertel et al. 2010	GTAP-BIO (CGE)	2001–2015	3.95		13.3	0.3
Malcolm, Aillery, and Weinberg 2009	REAP (PE)	2015	4.9	63%	2	2.45
Searchinger et al 2008	FAPRI (PE)	2005–2035	5.44		14.8	0.37
PE median (range)						0.42 (0.02 – 0.82)
CGE median (range)						0.145 (0.01 – 0.38)
Empirical median (range)						0.61 (0.43 – 0.66)
All median (range)						0.43 (0.01 – 0.95)

[1] Partial Equilibrium (PE) and Computable General Equilibrium (CGE)

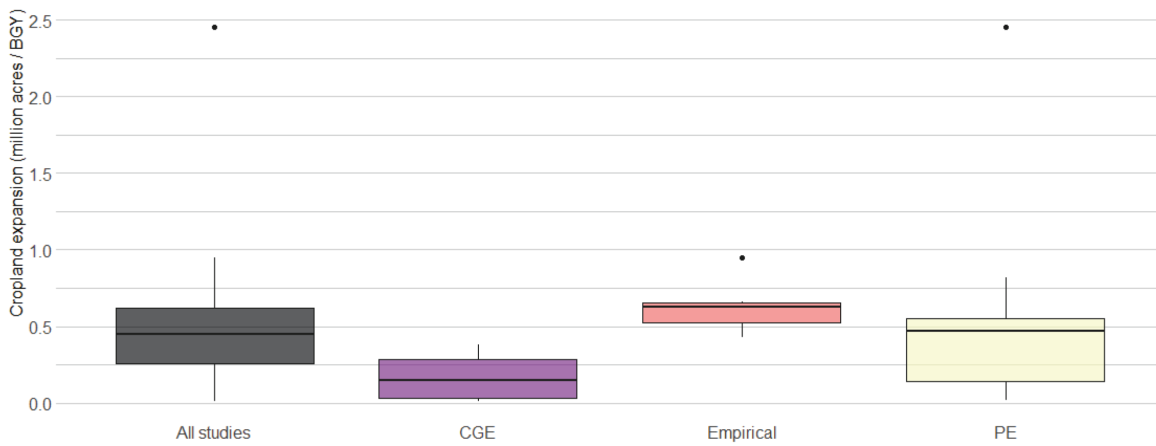


Figure 6.24. Reported net cropland expansion in the United States per billion-gallon increase in biofuel production, by study type [Partial Equilibrium (PE) (n = 10), Computable General Equilibrium (CGE) (n = 6), or national empirical modeling approaches (n = 3)]. This figure includes the results reported by Taheripour et al. (2020), which use a combined CGE and PE model, in the boxplots for both the CGE and PE studies. Source: Austin et al. (2022) (Creative Commons license, https://creativecommons.org/licenses/by/4.0/; no changes made).

Meta-analyses of the literature reviewed in Chapter 4 also provides a useful estimate of land use change for comparison. The Thompson et al. (2016) review summarized in Chapter 4 reviewed over 170 individual studies and reported different characteristics of each study.[81] Thompson et al. (2016) focus on the results that included a corn "supply response," which allows farmers to adjust production to changes in price. A similar focus is appropriate for the purposes of this chapter as well. Based on this review, Thompson et al. (2016) found an average response of approximately 0.7 million acres of additional cropland per billion gallons of corn ethanol production for the 12 studies that included a corn supply response.[82] This estimate suggests that roughly 9.5 million more acres of cropland are in production from the growth in the ethanol industry generally since 2002.[83] Using the maximum effect of RFS-attributable ethanol from section 6.3 in 2016, this suggests of 0–1.5 million acres of additional cropland are estimated attributable to the RFS Program (Table 6.10). From the subset of ten studies that did not assume the RFS was binding, the average response was much smaller—for an additional 0.1 million acres of cropland per billion gallons of ethanol. This subset of the literature suggests that roughly 1.4 million additional acres of cropland from the growth in the ethanol industry generally since 2002, and a maximum effect in 2016 of

[81] As noted in Chapter 4, this FAPRI review, though conducted by experts in the field and reviewed internally at USDA, was not formally peer reviewed. Nevertheless it is the most comprehensive source that isolates different categories of study.
[82] This is the weighted average reported in Table 9 "with supply response" of Thompson et al. (2016). See Chapter 4 for more discussion on this literature review and why this average is highlighted. There were several other subsets of the literature provided in Thompson et al. (2016).
[83] As noted above, the increase in ethanol production from 2002 to 2019 was 13.6 billion gallons (Figure 6.1).

0–0.2 million acres of cropland from RFS-attributable ethanol specifically (Table 6.10). The estimates from Thompson et al. (2016) are comparable to those from the more-recent review in Austin et al. (2022). This is not unexpected since some of the same studies may appear in both sets, but the source literature for the averages in Thompson et al. (2016) were not published, and thus the agreement suggests findings in this chapter are consistent.

Thompson et al. (2016) also reported on changes in corn acreage per billion-gallon increase in ethanol. They reported an estimated 1.0 million additional U.S. corn acres for each billion-gallon increase in corn ethanol production on average across 14 observations from economic simulation studies focused on corn ethanol and allowing for corn supply response (Thompson et al., 2016).[84] Focusing on the subset of ten studies that did not assume that the RFS Program was binding reduced the estimated increase in corn acreage to 0.5 million acres per billion gallons of ethanol. Combining this range (i.e., 0.5–1.0 million acres per billion gallons) with the results of section 6.3 suggests roughly a 6.8–13.6 million acre increase in corn acreage from the growth of the ethanol industry since 2002 from all causes; and, an increase of 0–0-1.0 and 0–2.1 million acres of corn in 2008-2011 and 2016, respectively, from the RFS-attributable ethanol, specifically.

All these estimates from simulation studies, though varying widely in scope, approach, and detail, show similar results (Table 6.10), with increases in U.S. cropland of 0–1.0 million acres in 2008-2011 and 0–2.1 million acres 2016, respectively; and, increases in corn area of 0–1.0 in 2008-2011 and 0–2.6 million acres and 2016. Lower estimates result from the set of studies that do not assume the RFS Program is binding, or that included other factors like MTBE. Differences in the range of estimated effects among these subsets of literature, in terms of absolute acreages, is relatively small, suggesting confidence in the scale of effect estimated.

[84] These estimates are from Table 8 of Thompson et al. (2016), focusing on the weighted averages as before.

Table 6.10. Summary of results from section 6.4.

Source	Detail	Change in U.S. Cropland (Mac)		Change in U.S. Corn Area (Mac)	
		2008-2011	2016	2008-2011	2016
Taheripour et al. (2022a)	CGE (2004–2016)	0–1.0[a]	0–0.16[b]	NA[c]	NA[c]
Newes et al. (2022)	Including octane value of ethanol	NA[d]	NA	0	2.6[e]
Model review for DRIA[f]	Studies isolating the effect of corn ethanol production on land use change	0-1.0	0-2.1	NA	NA
Austin et al. (2022)[g]	CGE, PE, and Empirical	0–0.95	0–2.0	NA	NA
Thompson et al. (2016)[h]	Studies with a supply response	0–0.7	0–1.5	0–1.0	0–2.1
	Studies that did not assume the RFS was binding	0–0.1	0–0.2	0–0.5	0–1.1
Range across all simulation studies		0–1.0	0–2.1	0–1.0	0–2.6
Preferred statistical estimates (Table 6.12)		0–0.9	0–1.9	0–1.6	0–3.5

a The CGE results for 2004–2011 are used for the 2008-2011 estimate.
b The CGE results for 2011–2016 are used for the 2016 estimate.
c The CGE model (GTAP-BIO) simulates coarse grains which are primarily corn in the United States. Coarse grains acreages were not reported for the CGE model. Acreages were not reported for the PE model.
d The BSM does not provide an estimate of total cropland, only five crops are modeled (corn, soy, wheat, cotton, other grains).
e The BSM corn acreage estimates are not elasticities multiplied by an estimated RFS-attributable-ethanol, they are internally generated. So the 2008–2011 estimates (no risk reduction) are presented, and the maximum year-effect which was 2018 (Figure 6.22).
f For the DRIA review, the domestic effect of 0.2-1.0 Mac/Bgal ethanol is multiplied by 0-1.0 billion gallons attributable to the RFS Program in 2008-2011 and 0–2.1 billion gallons in 2016. Changes in corn acreage were not assessed from these studies in the DRIA because not all models reported individual crops.
g For the Austin et al. (2022) review, the 0.01-0.95 Mac/Bgal estimate (all methods range, Table 6.9) is multiplied by 0-1.0 billion gallons for 2008-2011 and by 0-2.1 billion gallons for 2016. Changes in corn acreage were not assessed in Austin et al. (2022).
h For the Thompson et al. (2016) review, for cropland the 0.7 Mac/Bgal (supply response) or 0.1 Mac/Bgal (non-binding) is multiplied by the 0-1.0 billion gallons attributable to the RFS Program in 2008-2011 and 0-2.1 billion gallons in 2016. For corn area, 1.0 Mac/Bgal (supply response) or 0.5 Mac/Bgal (non-binding) is multiplied by the 0-1.0 billion gallons attributable to the RFS Program in 2008-2011 and 0-2.1 billion gallons in 2016.

6.4.1.4. The Model Comparison Exercise Technical Document

In June 2023, EPA published the "Final Renewable Fuels Standards Rule for 2023, 2024, and 2025".[85] The final rulemaking package included a Model Comparison Exercise Technical Document which, among other things, compared land use change impact results from five models for a series of U.S. biofuel consumption scenarios.[86] This Model Comparison Exercise (MCE) was motivated by needs identified at a virtual public workshop on biofuel GHG modeling on February 28 and March 1, 2022.[87] Specifically, this workshop identified that, in support of a better understanding of the potential land use change and other impacts associated with biofuels, it would be helpful to compare available models and identify how and why the model estimates differ.

[85] See 88 FR 44468
[86] See EPA-420-R-23-017
[87] For more information see the Federal Register Notice, "Announcing Upcoming Virtual Meeting on Biofuel Greenhouse Gas Modeling." 86 FR 73756. December 28, 2021. More information is also available on https://www.epa.gov/renewable-fuel-standard-program/workshop-biofuel-greenhouse-gas-modeling.

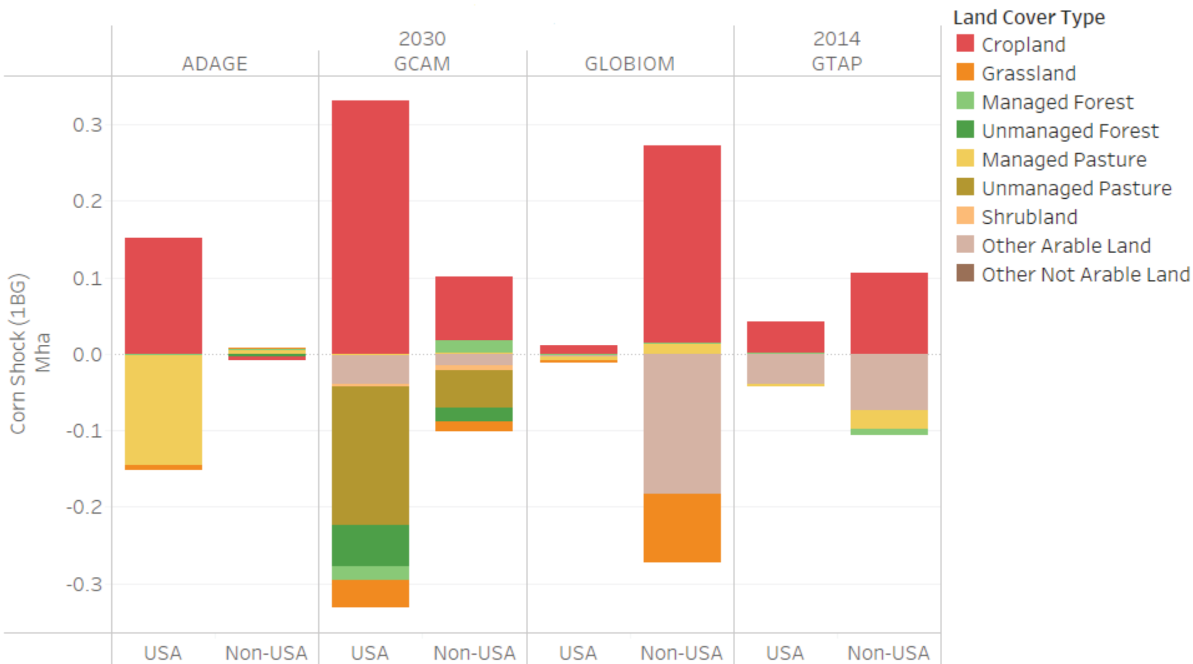
Motivated by this need, EPA conducted the MCE with five models: the Greenhouse Gases, Regulated Emissions, and Energy Use in Technologies Model (GREET), Global Biosphere Management Model (GLOBIOM), Global Change Analysis Model (GCAM), Global Trade Analysis Project (GTAP) model, and Applied Dynamic Analysis of the Global Economy (ADAGE) model. See Table 6.11 for more details on each model. To facilitate appropriate comparisons of these models, EPA ran three common scenarios through each framework: (1) a reference case; (2) a corn ethanol consumption shock with an additional 1 billion gallons (0.076 QBTU[88]) of U.S. corn ethanol consumption in each year, with all other U.S. biofuel consumption volumes set by assumption at the reference case levels (also referred to as the "corn ethanol shock"); and (3) a soybean oil biodiesel consumption shock with an additional 1 billion gallons (0.118 QBTU) of U.S. soybean oil biodiesel consumption in each year, with all other U.S. biofuel consumption volumes set by assumption at the reference case levels (also referred to as the "soybean oil biodiesel shock"). Effects from the corn ethanol shock are discussed here and from the soybean oil biodiesel shock are discussed in section 7.3.4. All these scenarios were hypothetical and designed solely for the purpose of evaluating and comparing the models.

Four of the five models compared in the MCE estimated land use change impacts for the corn ethanol shock (the fifth model, GREET, does not endogenously model land competition and thus did not report these types of results). Figure 6.25 shows the estimated impacts on cropland and other land cover types, resulting from the corn ethanol shock. The figure shows the range in land use change impacts estimated by this suite of models, both domestically and outside the United States. All four models estimate some quantity of new cropland would be cultivated in response to increased consumption of corn ethanol. However, the model estimates show significant variation in the quantity of additional cropland cultivated, the location of that new cropland cultivation (domestically versus internationally), and in the types of land cover which are estimated to be converted to cropland in response to increased consumption of corn ethanol. For example, ADAGE and GCAM estimate most of the increase in cropland area from the corn ethanol shock would occur domestically, while GLOBIOM and GTAP estimate the increase in cropland area would mostly occur abroad. For further discussion of the motivations, methods, and results of this study, and the drivers behind these variations in model estimates, please refer to the MCE Technical Document.[89] EPA has made no firm methodological decisions based on the MCE but continues to gather feedback from stakeholders on the results and findings of this report to support future model developments and decisions. This work is ongoing; EPA will make further announcements regarding progress on these efforts at a later date.

[88] QBTU stands for Quadrillion British thermal units and is a unit of energy.
[89] See EPA-420-R-23-017

Table 6.11. Comparison of key characteristics across models used in the EPA MCE.

Characteristic	ADAGE	GCAM	GLOBIOM	GREET	GTAP
Type of Model	Computable general equilibrium (CGE); consequential LCA	Integrated assessment model (IAM); consequential LCA	Partial equilibrium (PE); consequential LCA	Supply chain LCA	Computable general equilibrium (CGE); consequential LCA
Sectoral Coverage	Economy-wide with 36 sectors	Energy (conventional and renewable), industry, buildings, transportation, agriculture, forestry, water	Agriculture, forestry, and bioenergy	Fuel supply chains including energy resource and material inputs	Economy-wide aggregated into 65 sectors
Temporal Representation	Recursive dynamic (5-year time steps)	Recursive dynamic (5-year time steps)	Recursive dynamic (10-year time steps)	Static (users can select a target year from 1990-2050)	Comparative static
Regional Coverage	8 economic and spatial regions	32 economic regions; 384 land regions (water basins, intersected with economic regions)	37 economic regions; 10,000 spatial units (grid cell)	Customizable (typically U.S. average)	19 economic regions; 18 agro-ecological zones
GHG Emissions Coverage	Economy-wide GHGs including land use change	Global GHGs including land use change	Crop production, livestock, and land use change	Direct supply-chain emissions + indirect land use change from CCLUB module	Economy-wide GHGs, with land use change GHGs calculated with the AEZ-EF model
Land Representation (Arable land categories considered in biofuel land use change analysis)	Cropland, pasture, commercial forest, non-commercial forest, natural grassland, other land	Cropland, commercial pasture and forest, non-commercial pasture and forest, shrubland, grassland, "protected" non-commercial land	Cropland, other agricultural land, grassland, commercial and non-commercial forest, wetlands, other natural land	Exogenous (Land use change estimates from GTAP-BIO and CCLUB)	Cropland (including cropland-pasture and unused cropland), livestock pasture, "accessible" forestry land



Note: this figure appears as Figure 6.6-2 in the original report. Due to differences in model structure and temporal coverage, results are presented for 2030 for ADAGE, GCAM, and GLOBIOM, and for 2014 for GTAP

Figure 6.25. Difference in land use (million hectares) in the corn ethanol shock relative to the reference case in 2014 (GTAP) and 2030 (ADAGE, GCAM, GLOBIOM).

6.4.2 Statistical Studies

As with simulation studies, the statistical studies vary widely in scope and approach, affecting their utility for the purpose of attributing corn and crop area to the RFS Program. The literature review by Austin et al. (2022), identified 14 statistical studies directly relevant to the effects of the RFS Program on U.S. cropland (Tables 6.10 and 6.12). These studies generally found that increases in various corn-ethanol-related drivers (e.g., proximity to a biorefinery, corn price, biorefinery production) were positively associated with changes in LCLM. For purposes here, the focus is on the subset of studies that were spatially explicit and national in scope (3 of 13 studies; (Lark et al., 2019; Li et al., 2019; Fatal and Thurman, 2014), which covered most of the major period of growth in the ethanol industry (2002–2012) and especially the period of large increase in corn acreage from 2006 to 2008 (2 of 13 studies; (Li et al., 2019; Fatal and Thurman, 2014).[90] Fatal and Thurman (2014) found that an additional 1 million gallons of capacity at an ethanol plant leads to an additional 5.21 ± 0.68 acres of planted corn in a given county. Li et al. (2019) found that when ethanol plant capacity increases by 1 million gallons, corn acreage will

[90] Because the land use changes from Lark et al. (2022b) are partly based on the CDL, they begin in 2008 when national coverage was first available. As discussed in sections 6.3.3. and 6.4.3., the Lark et al. (2020) estimates are estimates of land use change from all causes, and the Lark et al. (2022b) estimates are better described as land use change from many causes including the RFS Program. Numerically they most closely align with the effects from MTBE and the RFS Program.

increase by 884 acres (2.2%) and crop acreage by 599 acres (0.65%) in counties within 25 miles of a plant. Ideally, the focus would be on any study that separated the effects of ethanol production from the effect of corn or crop price on land use change (1 of 13 studies; (Li et al., 2019). All three of these characteristics are important for the purposes of this chapter. The last is especially important, as Li et al. (2019) found that the effect on corn acreage from corn price was much stronger than the effect from ethanol capacity, and thus any study that only included the effect from ethanol capacity may inflate the estimated effects from ethanol on land use.

Li et al. (2019) leveraged nationally available, high spatial resolution data to estimate the impact of effective ethanol plant capacity,[91] corn prices, and crop prices on changes in corn and crop acreage nationally at the county level from 2003 to 2014.[92] This study is unique among the statistical studies reviewed, because it modeled each county as a potential supplier of corn to nearby ethanol plants, provides national estimates for both corn and cropland, and controls for changes in the corn and crop prices. Changes in ethanol capacity were assumed to potentially have an effect locally (i.e., within 25 miles [40 km] of an ethanol plant), while changes in price were assumed to have a potential effect nationally. Other statistical studies were either limited to particular geographic regions and/or did not control for changes in crop prices in estimating the effect of ethanol production (Table 6.12). Li et al. (2019) also has the added strength of using an "instrument variable" to statistically isolate the causal effect of ethanol production on changes in corn and crop land.[93]

[91] Effective ethanol capacity in Li et al. (2019) is estimated at the county level as the nameplate capacity of a refinery multiplied by the fraction of a 25 mile radius circle (representing the transportation domain to that refinery) around that refinery that falls within a county, summed across all biorefineries that have a transportation domain that overlaps with that county.

[92] In Li et al. (2019) "crops" are the set of 10 crops that are most common in the Midwest: barley, corn, cotton, oats, peanuts, rice, rye, soybeans, sorghum, and wheat. Together these crops account for 78-80% of the cropland acreage in the United States from 2003-2014 using estimates from USDA NASS.

[93] Instrumental variables (IVs) are a statistical approach for estimating a causal relationship when covariates are also estimated and when randomized controlled experiments are infeasible or not executed adequately (Pokropek, 2016). Studies that implement IVs are often called "quasi-experimental" because the treatment effect is isolated statistically rather than experimentally. They are more common in epidemiology and social sciences, and uncommon for statistical biofuel studies to date.

Table 6.12. Summary of correlational studies.

Study	Influence/ Treatment	Land Use/ Cover Impact	Spatial Extent	Spatial Resolution	Study Period	Land Use Change Attributable to the Influence/Treatment
Barr et al. (2011)	Crop prices	Change in crop acreage	Contiguous United States	Non spatial	2007–2009	A 1% increase in the price of crops results in a 0.007–0.029% increase in cropland acreage.
Brown et al. (2014)	Ethanol plant proximity	Change in crop acreage	Kansas	5 acre grid cells	2007–2009	A 1% decrease in the distance to a refinery corresponds to a 5–15% increase in corn extensification.
Fatal and Thurman (2014)	Ethanol plant production capacity	Change in corn acreage	Contiguous United States	County	2002–2008	An additional 1 million gallons of capacity at an ethanol plant results in 5.21 ± 0.68 additional acres of planted corn in a given county.
Hendricks et al. (2014)	Crop prices	Change in corn acreage	Iowa, Illinois, Indiana	Fields (based on USDA's Common Land Unit boundaries)	2000–2010	A 10% increase in the price of corn results in a 2.9–4.0% increase in corn acreage.
Ifft et al. (2019)	Ethanol plant location and capacity	CRP re-enrollment	Illinois, Indiana, Iowa, Kansas, Minnesota, Missouri, Nebraska, North Dakota, South Dakota, Wisconsin	County	1999–2014	A 100 million gallon increase in ethanol capacity results in 13% less land leaving the CRP.
Krumel et al. (2015)	Ethanol plant proximity and capacity	CRP re-enrollment	North Dakota, South Dakota, Minnesota, Wisconsin, Nebraska, Kansas, Iowa, Illinois, Indiana, Ohio, Oklahoma, Missouri	County	2007–2013	Average increase in ethanol capacity expansion (of 139 million gallons/two years) corresponds to a 0.05–0.06% increase in early exit from the CRP Program.
Langpap and Wu (2011)	Crop prices	Changes in crop acreage	Ohio, Illinois, Indiana, Iowa, Missouri, Michigan, Wisconsin, Minnesota	Fields (based on USDA's Common Land Unit boundaries)	NA	A 1% increase in the price of corn results in a 0.06%–0.14% increase in cropland acreage.
Lark et al. (2019), Lark et al. (2022b)	Crop prices	Change in corn and crop acreage	Contiguous United States	Fields (based on USDA's Common Land Unit boundaries)	2008–2016	A 30% increase in corn price and a 20% increase in soybean price results in 1.8 million hectares of cropland expansion and reduced rates of abandonment by 0.4 million hectares, or a net increase in cropped area of 2.1 million hectares.

Study	Influence/ Treatment	Land Use/ Cover Impact	Spatial Extent	Spatial Resolution	Study Period	Land Use Change Attributable to the Influence/Treatment
Li et al. (2019)	Ethanol plant proximity and capacity, and corn and crop prices	Change in corn acreage and crop acreage	Contiguous United States	County	2003–2014	A 1 dollar increase in corn received prices will increase corn acreage by 2,532 acres (6.3%) and a 1 dollar increase in crop price index will lead to an increase in total crop acreage by 4,484 acres (4.8%). When ethanol plant capacity increases by 1 million gallons, corn acreage will increase by 884 acres (2.2%) and crop acreage by 599 acres (0.65%) in counties within 25 miles of a plant.
Miao (2013)	Ethanol plant location and capacity	Change in corn acreage	Iowa	County	1997–2009	Establishment of a 100-million-gallon ethanol plant increased corn acreage by 8–14%.
Motamed et al. (2016)	Ethanol plant production capacity	Change in corn and crop acreage	Illinois, Indiana, Iowa, Kansas, Minnesota, Missouri, Nebraska, North Dakota, Ohio, Oklahoma, South Dakota, Wisconsin	10 x 10 km grid cells	2006–2010	A 1% increase in refining capacity increases corn acreage by 1.5% and total cropland acreage by 1.7%.
Secchi et al. (2011)	Corn prices	Change in corn acreage and CRP re-enrollment	Iowa	30 x 30 m grid cells	Non spatial	A 27% increase in corn price leads to a 41% reduction in CRP land and a 15% increase in cropland. A 67% increase in corn price leads to a 65% reduction in CRP land and a 15% increase in cropland. A 96% increase in corn price leads to a 71% reduction in CRP land and a 15% increase in cropland.
Stevens (2015)	Ethanol plant proximity	Change in corn acreage	Illinois, Indiana, Iowa, Nebraska	Fields (based on USDA's Common Land Unit boundaries)	2002–2014	300,000 acre increase in corn acreage within 30 miles from refineries.
Wright et al. (2017)	Ethanol plant proximity	Change in corn acreage	Contiguous United States	3.5 x 3.5 mile grid cells	2008–2012	4.2 million acre increase in cropland within 100 miles from refineries, 2.7 million acre increase within 50 miles, and 1.1 million acre increase within 25 miles.

Li et al. (2019) found that with other factors remaining the same, "the increase in ethanol capacity alone led to a modest 3% increase in corn acreage and less than a 1% increase in total crop acreage by 2012 when compared to 2008." Although the study also estimated the effects of corn and crop prices on planted area, they did not estimate the *indirect* effect of corn ethanol on corn and crop prices, which could then affect corn and crop acreages. The effects of corn ethanol on corn and crop prices are available in the peer-reviewed literature. The review in Chapter 4 found that synthesis from multiple studies suggests each billion gallons of corn ethanol increases corn prices by 4% ± 1%. Roberts and Schlenker (2013) estimated that commodity prices more generally increase 20% from a 11.1 billion gallons increase in corn ethanol, which is an increase of roughly 1.8% in crop prices per billion gallons of corn ethanol.

Using these estimates from Roberts and Schlenker (2013) along with the coefficients estimated by Li et al. (2019), the direct and indirect effects of corn ethanol on corn and crop area may be estimated (Table 6.13). Given that the estimated range of additional ethanol production attributable to the RFS includes zero, so does the estimated range of the effect of RFS-attributable ethanol on corn and crop area. As an illustration of the highest end of the estimated range, the combined direct and price-induced effects of 2.1 billion gallons of RFS-attributable ethanol production in 2016 (Figure 6.21) is estimated here. With this approach the estimate for 2016 RFS-attributable ethanol may have added as much as 3.5 ± 1.0 million acres of corn and as much as 1.9 ± 0.9 million acres of cropland in the United States (Table 6.13). Compared with the range of estimates from simulation models (Table 6.10) the range of estimates are similar for increases in cropland and slightly higher for increases in corn acreage.

Table 6.13. Estimated change in U.S. corn and crop areas due to an additional 0-1.00.4 and 0-2.1 billion gallons of corn ethanol production in 2008-2011/09 and in 2016. The 2.1 billion-gallon estimate is from the Taheripour et al. (2022a) PE model, the highest estimate for a single year of the studies reviewed. This chapter's estimated range of RFS-attributable ethanol and associated corn and crop area includes zero. Estimates are based on multiplying corn ethanol production volume by coefficients from Li et al. (2019) and other sources. For convenience, the zero is not repeated in each row and is just shown in the first row and rows j and s.

Element		Units	Calculation	Estimate	
	Direct Effect of Ethanol Production on Corn Area			2008-2011	2016
(a)	Δ Corn Etoh | RFS	Bgal		0–1.0	0–2.10
(b)	Effect Corn Etoh → Corn Area	M acres per Bgal		0.884 ± 0.1449	
(c)	Δ Corn Area | Δ Corn Etoh	M acres	c = a * b	0.88 ± 0.14	1.86 ± 0.3
	Indirect Price Effect of Ethanol Production on Corn Area				
(d)	Effect Corn Etoh → Corn Price	% change per Bgal		4% ± 1%	
(e)	Δ Corn Price | RFS	% change	e = a * d	4% ± 1%	8.4% ± 2.1%
(f)	Elast. of Corn Area to Corn Price	Constant		0.21 ± 0.03	
(g)	Δ Corn Area | Δ Corn Price	% change corn area	g = e * f	0.8% ± 0.3%	1.8% ± 0.7%
(h)	Planted Corn Area	M acres		88.1	94.0
(i)	Δ Corn Area | Δ Corn Price due to RFS	M acres	i = g/100 * (h/(1+g/100))	0.7 ± 0.26	1.66 ± 0.65
(j)	**Total Δ Corn Area | RFS**	**M acres**	**j = c + i**	**0-1.58 ± 0.4**	**0-3.52 ± 0.95**
	Direct Effect of Ethanol Production on Crop Area				
(k)	Effect Corn Etoh → Cropland	M acres per Bgal		0.599 ± 0.205	
(l)	Δ Cropland Area | Δ Etoh Prod.	M acres	l = a * k	0.6 ± 0.21	1.26 ± 0.43
	Indirect Price Effect of Ethanol Production on Crop Area				
(m)	Effect Corn Etoh → Crop Price	% change per Bgal		1.8% ± 0.7%	
(n)	Δ Crop Price | RFS	% change	n = a * m	1.8% ± 0.7%	3.78% ± 1.47%
(o)	Elast. of Crop Area to Crop Price	Constant		0.07 ± 0.02	
(p)	Δ Crop Area | Δ Crop Price	% change crop area	p = n * p	0.13% ± 0.09%	0.26% ± 0.18%
(q)	Planted Crop Area	M acres		250	257
(r)	Δ Crop Area | Δ Crop Price | RFS	M acres	r = p/100 * (q/(1+p/100))	0.32 ± 0.22	0.67 ± 0.46
(s)	**Total Δ Crop Area | RFS**	**M acres**	**s = l + r**	**0-0.92 ± 0.43**	**0-1.93 ± 0.89**

Table Notes:
"|" can be interpreted as "given", "due to" or "attributable to"
"→" can be interpreted as "on" or "effect on"
RFS is short for RFS-attributable ethanol; Elast. is short for elasticity; Corn Etoh is short for corn ethanol production.
(b) Values from Li et al. (2019) Table 2 (Model 2); controls for corn price changes. ± values are the Conley standard errors.
(d) Estimates from Chapter 4. Average from Condon et al. (2015), section 4.3.2.
(f) Li et al. (2019) Table 6 (preferred specification). ± values are the Conley standard errors.
(g) ± = high estimate - low estimate) / 2
(h) Corn area planted averaged for 2008-2016 and in 2016 from USDA NAAS
(k) Values from Li et al. (2019) Table 3 (Model 2); controls for crop price changes. ± values are the Conley standard errors.
(m) Roberts and Schlenker (2013) ("R&S") estimated crop prices increase 20% with ethanol production increase of 11 Bgal, with 95% CI from 14% to 35%. The number after the ± is the approximate standard error (upper end of the 95% CI minus the mean divided by 2). The R&S CI is positively skewed, meaning the low end of the minus standard error would be 0.25% instead of 0.7% as used in this table.

(o) Li et al. (2019) Table 6 (Preferred Specification).
(q) USDA NASS planted area for ten major crops in Li et al. (2018) averaged in 2008-2011 and in 2016 (barley, corn, cotton, oats, peanuts, rice, rye, sorghum, soybeans, wheat) accounting for >85% of cropland area in the United States.
(r) Rows (i) and (p) are the observed corn and crop areas, respectively, inclusive of the RFS price effects. Thus, they are adjusted (e.g., divided by 1 + g) to estimate what the area would have been absent these effects.

6.4.3 Synthesis of Evidence

The range of estimated effects of the RFS Program on corn acreage and total crop acreage based on information from statistical and simulation studies are similar (Table 6.10), suggesting that at the national level the estimates are robust to differences in approach. For effects on corn acreage in 2008-2011, the preferred statistical approach estimates an effect of 0–1.6 million acres, close to though higher than the 0–1.0 million acres estimated from simulation studies. For effects on corn acreage in 2016, the preferred statistical approach estimates an effect of 0–3.5 million acres, again similar but higher than the 0–2.6 million acres estimate from simulation studies. For effects on crop acreage in 2008–2011, the preferred statistical approach estimates an effect of 0–0.9 million acres, as opposed to 0–1.0 million acres from simulation studies. For effects on crop acreage in 2016, the preferred statistical approach yields an estimated effect of 0–1.9 million acres, similar to the 0–2.1 million acres estimate from the simulation studies (Table 6.10).

The recent study by Lark et al. (2022b), based on similar analytical techniques as Pates and Hendricks (2021), provides another useful analysis of the effects from corn and corn ethanol broadly from many causes. They report an increase in corn ethanol production of 5.5 billion gallons each year, corresponding to an increase of total cropland by 5.2 million acres and of corn acreage by 6.1 million acres. These estimates, however, are roughly double the estimates presented here in absolute terms because of several assumptions in the underlying economic model (Carter et al., 2017) that increase the estimated effect of the RFS Program (discussed above, section 6.3.3). However, once the estimates from Lark et al. (2022b) are rescaled to account for attributional differences, the estimates are very similar to those in the RtC3.[94]

Based on the above review of the peer-reviewed literature, the approach summarized above using Li et al. (2019), in combination with other data and literature estimates, provides a robust estimate of the county-level effects of RFS-attributable ethanol on U.S. corn and total crop land that covers the entire period of growth in the ethanol industry. The Li et al. (2019) study is consistent with the other literature, and is based on historical data and is at a much finer spatial scale than either Taheripour et al. (2022a) or

[94] Table 6.13 reports increases of up to 3.52 million corn acres and up to 1.93 cropland acres. The Lark et al. (2022b) study, assuming 5.5 billion gallons of ethanol are attributable to the RFS, reported increases of 6.9 million acres of corn and 5.2 million acres of cropland. Re-adjusting to account for attributional differences discussed above, the Lark et al. (2022b) study imply an increase of 2.6 million acres of corn (i.e., 6.9*2.1/5.5) and an increase of 2.0 million cropland acres (i.e., 5.2*2.1/5.5).

Newes et al. (2022). It is interesting to note that the estimated increase in domestic cropland from the Li et al. (2019) (0.92 Mac/Bgal or 0.37 Mha/Bgal) is slightly higher than the range from the four models evaluated in the MCE (Figure 6-25). This is consistent, however, with the reviews of the broader literature (Austin et al. 2022, Table 6.9), which shows that PE and empirical approaches tend to give comparable estimates, while CGE models tend to estimate lower amounts of cropland expansion. The Li et al. (2019) study cannot be used independently to estimate the effect of the RFS Program on ethanol production because it does not assess the drivers of changes in ethanol. But, with the synthesis in section 6.3, it may be leveraged to translate the RFS Program's estimated effects on ethanol production into effects on LCLM while controlling for coincident effects on price. This leverages the strengths of individual studies to yield a robust estimate. In addition to these strengths, Li et al. (2019) uses instrument variables to attempt to statistically isolate the effect of ethanol production, an improvement that is new to the biofuels literature.

Using this approach, this chapter's estimates suggest that in 2016, RFS-attributable ethanol led to an additional 0 to 3.5 ± 1.0 million acres of corn and 0 to 1.9 ± 0.9 million acres of cropland. Corn acreages increase by more than total cropland because of crop switching on existing croplands from other crops to corn. These results control for changes in corn and crop prices, so to the extent that increased ethanol production (from all causes) increases corn or other crop prices, the effect on corn and crop area would be expected to be larger. Again, the estimated range includes zero on the low end, and on the high end is based on the highest year estimate for RFS-attributable ethanol of 2.1 billion gallons in 2016 (Figure 6.21).

These estimates based on the preferred combination of sources described above are able to be spatially allocated (Figure 6.26, from Li et al. 2018). However, allocation to areas smaller than a county, which is necessary to support detailed environmental modeling discussed in subsequent chapters, is not possible. As techniques and the underpinning science improves, estimated effects may be attributable to the RFS Program at finer scales in future efforts.

To assess whether these changes can be considered to be large or small, their relative magnitude was estimated by comparing the RFS-attributable changes in cropland with estimates of total conversion to cropland from all causes. Lark et al. (2020) estimated a total of 10.09 million acres of non-cultivated land—mostly grasslands like pasture and CRP grasslands—converted to cropland between 2008 and 2016 in the contiguous United States (roughly 1 million acres per year). The USDA's Natural Resources Inventory (NRI) (2020) estimated a net increase of 8.63 million acres in total cropland from 2007 to 2017 (see also Chapter 5). Based on the 0 to 1.9 million acres of new cropland estimated to be attributable to the RFS, or 0 to 19% of the total new cropland from all causes in Lark et al. (2020) and 0 to 22% of the total new cropland from the NRI (2020) (Table 6.14).[95] Given the similarity in the estimates and the inherent uncertainty, an approximate range of 0 to 20% is used for the remainder of the RtC3 as the estimate of cropland expansion from 2008 to 2016 attributable to the RFS Program. For context, the 2.0 million acres represents about 0.5% of total cropland in 2017 or an area slightly larger than Delaware (ca. 1.6 million acres). Although not a large percentage nationally, this upper end of the converted acreage range may have important environmental effects regionally or locally, especially in areas with a higher concentration of converted acres (e.g., southern Iowa and the Dakotas).[96]

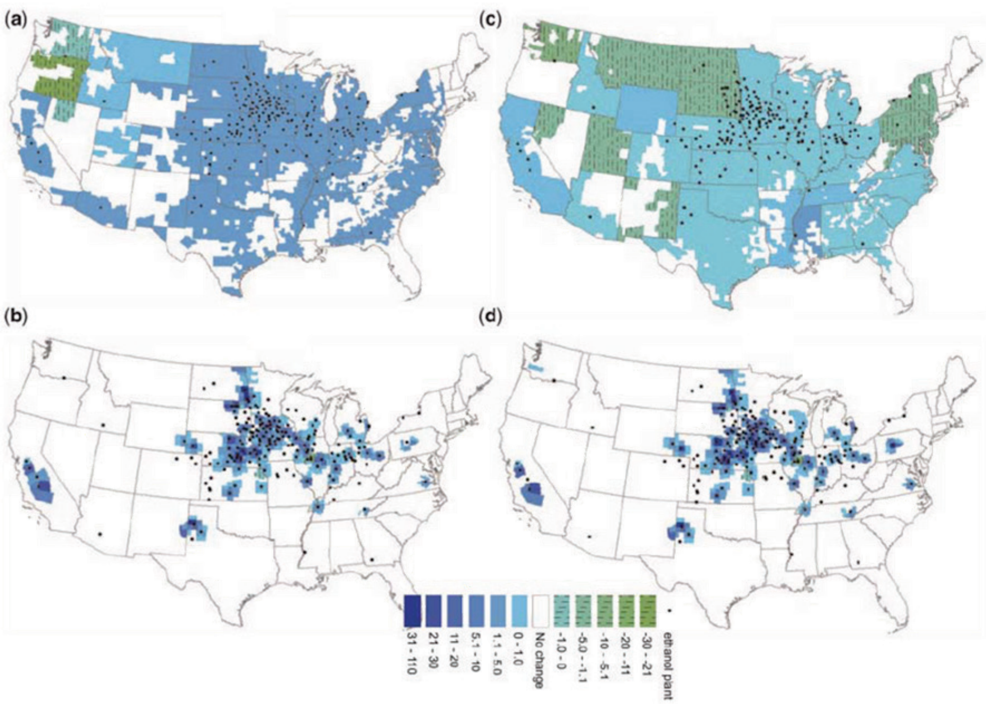


Figure 6.26. Changes in aggregate crop acreage due to crop price or effective ethanol capacity (in 1,000 acres). Maps (a) and (b) represent changes from 2008 to 2012. Map (a) is for changes due to crop price, and map (b) is for changes due to effective ethanol capacity. Maps (c) and (d) represent changes from 2008 to 2014. Map (c) is for changes due to crop price, and map (d) is for changes due to effective ethanol capacity. Source: Li et al. (2019) (used with permission).[97]

[95] Here the 1.0 million acres in 2008-2011 are not added to the 1.9 million acres in 2016, assuming that the cropland converted in 2008-2011 continued to be cultivated after the years that the RFS was no longer estimated to be directly attributable to their cultivation.
[96] The finding that the RFS Program was attributable for additional U.S. cropland does not contradict the "aggregate compliance" approach in the RFS regulations for demonstrating that planted crops and crop residue from the United States complies with the requirements that address lands from which qualifying feedstocks may be harvested. In the February 2020 RFS volume setting rule (85 FR 7016), EPA estimated that U.S. agricultural land reached 379.8 million acres in 2019 and thus did not exceed the 2007 baseline acreage of 402 million acres.
[97] The estimates in Figure 6.22 are from Li et al. (2019) and do not include the attribution estimates from Table 6.13.

Table 6.14. Comparison of estimated changes in cropland with changes in cropland attributable to the RFS Program.

Measure	2008–2016	2007–2017
Total Converted Acreage (millions of acres)	10.09	8.63
Reference	Lark et al. (2020)	NRI (2020)
Total Converted Acreage Estimated to be Attributable to the RFS Program (millions of acres)	0–1.9	0–1.9
Percent of Converted Acreage Estimated to be Attributable to RFS Program	0–19%	0–22%
Acreage Estimated Attributable to the RFS Program as a Percent of Total Cropland in 2017[a]	0–0.5%	0–0.5%

[a] This assumes 367,483,300 acres of total cropland in 2017 from the NRI (Brown-Hruska et al., 2018).

6.4.4 Limitations of the Assessment

A recent National Academies of Science, Engineering, and Medicine (NASEM) report—which coincided with the drafting of this report—reviewed the current methods for lifecycle analyses for low-carbon transportation fuels (NASEM, 2022). NASEM (2022) recommends using a consequential lifecycle analysis (CLCA) for assessing lifecycle effects of a given policy. Reviews of several existing modeling platforms (section 6.4.1.2) found that, in their current form, CLCA platforms do not include certain important industry details that would be useful for assessing the effect of the RFS Program separate from other factors (e.g., inclusion of match blending, MTBE as an oxygenate). Thus, CLCA platforms cannot be used to assess the actual impacts of the RFS Program, because they do not include key factors that influence the effect of the RFS Program. They are, however, well-suited for a more general analysis of the effect of increased biofuels from a variety of causes. Secondly, none of them are at a sufficiently small spatial resolution for this assessment. There is no single CLCA that has all the attributes needed for a retrospective analysis of the effect of the RFS Program. Thus, instead of using individual or an ensemble of models that currently have known weaknesses, this analysis uses the best approach for each step in the analytical chain. The primary feature of a consequential analysis is the comparison of simulations with and without the policy in question—in this case the RFS Program. The analysis in this report uses several partial and general equilibrium models—as recommended in the NASEM report—to estimate these effects on biofuel production (e.g., GTAP-BIO, BSM, the broader literature). EPA recently conducted a comprehensive Model Comparison Exercise (MCE) of major LCA models in the field to better understand their strengths and weaknesses. It may be possible soon to construct such a model that would have sufficient industry detail and spatial granularity to assess the effect of the RFS Program with confidence both historically and into the future.[98]

[98] For more information on the MCE, see the associated Technical Document (https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1017P9B.pdf) and the docket for the Final Set Rule (EPA-HQ-OAR-2021-0427).

There are several specific limitations to the approach in this report that may be improved in the future. Uncertainties relate to limited data, integration of studies with differing temporal scopes and definitions, and the strong reliance on one statistical study given the lack of others that meet the same criteria.

First, data are limited. There is no national accounting system to track corn bushels from the land where they are harvested to their particular end uses. Tracking corn was considered in the Notice of Proposed Rulemaking for the RFS2 in 2009, but it was decided in the Final Rule to be too onerous on farmers and the government to implement.[99] Spatial and census data on land cover and management are also limited in terms of consistency and accuracy, in part due to changing definitions of terms and methods of analysis over time. A major factor impacting corn and total cropland areas are various state and federal subsidies impacting farm operations. No study evaluated above explicitly included all these factors.

Second, as noted above simulation modeling studies have significant limitations and uncertainties. Available studies provide support for the chapter's conclusions as they produce estimates within a similar range. However, these estimates should be approached with caution. Simulation models rely on a large number of assumptions and aggregations. Model validation and sensitivity analyses are inconsistent and limited for the simulation studies reviewed. Thus, the uncertainties associated with these estimates are largely unquantified. Additional sensitivity analyses and model validation exercises, similar to those conducted at EPA through the MCE in the Final Set Rule, may help to reduce this limitation. Furthermore, although there have been many simulation studies to date, very few of these have included sufficient market detail (e.g., RFS Program, oil price, octane, MTBE) to be able to parse out the effect of the RFS Program from other factors. In addition, there are numerous other federal and state policies and programs that affect cropping decisions from year to year, many of which are omitted from these studies and deserve attention. Finally, the spatial resolution of these models is usually coarse (e.g., many states) precluding the ability to estimate where changes in LCLM occurred. These limitations notwithstanding, the close correspondence between simulation and statistical estimates (Table 6.10) at the national level suggests the chapter's estimates are robust.

Third, the estimates here rely strongly on a single statistical estimate. Li et al. (2019) is the only statistical study the authors identified on the effect of ethanol on land that is national in scope (i.e.,

[99] For discussion of the proposed domestic "map and track" system see U.S. EPA (2009). "Regulation of Fuels and Fuel Additives: Changes to Renewable Fuel Standard Program; Notice of Proposed Rulemaking." May 26, 2009. 74 FR 24938 – 24941. For a discussion of the decision to use the aggregate compliance approach domestically see EPA. "Regulation of Fuels and Fuel Additives: Changes to Renewable Fuel Standard Program; Final Rule." March 26, 2010. 75 FR 14699 – 14704. See Section III.B.4.d in the 2009 proposed rule for the RFS2 (Approaches for Domestic Renewable Fuel, https://www.govinfo.gov/content/pkg/FR-2009-05-26/pdf/E9-10978.pdf)

includes all counties of interest), covers the major period of interest, and controls for prices. Lark et al. (2022b) has some of these features as well but makes several assumptions that increase the estimated attributional effect of the RFS Program. However, once adjustments are made to the attributional fraction, the Lark et al. (2022b) estimates are in agreement with Li et al. (2019). Li et al. (2019) also has the additional strength of incorporating instrument variables, a technique that is relatively uncommon in the biofuel and land use change literature. Thus, although the numerical estimates here are from a single study, results from Li et al. (2019) are consistent with the broader literature (Table 6.10).

Fourth, combining information from different efforts may result in some definitional or other inconsistencies that are difficult to resolve. For example, the crop price change from Roberts and Schlenker (2013) used a different definition of crop prices from that in Li et al. (2019). As noted above, total cropland in Li et al. (2019) is an underestimate of total cropland in the United States. Inconsistencies may also be introduced when estimates of RFS-attributable ethanol production are combined with the cropland change estimates from the Li et al. (2019) study. Li et al. (2019) used data from the 2003–2014 time period when total ethanol production increased most dramatically, but the bulk of RFS-attributable ethanol production occurred in the 2013–2019 time frame (Figure 6.21). It is possible that higher crop yields and other differences in later time periods would result in different parameter estimates. Thus, confidence would increase if the Li et al. (2019) study was updated to incorporate more-recent data. However, simply extending the time period from Li et al. (2019) forward may have limited value given that ethanol production levels have been relatively steady since approximately 2014. A single internally consistent CLCA modeling approach is preferred for some reasons (NASEM, 2022), but comes with significant disadvantages discussed above. Given that the regression analyses for this subset were assumed to represent national changes that generate cropland change estimates in each time step, a more complete examination of cropland can be expected to produce different estimates.

Fifth, it is not yet possible with current tools to downscale these estimates to the levels needed for environmental simulation modeling. The finest scale of the Li et al. (2019) study is at the level of the county, and many studies are coarser than that. The exact location of changes in LCLM—whether from the RFS Program or biofuels more generally—is critically important to evaluate environmental effect. There are remote-sensing datasets (e.g., Cropland Data Layer [CDL], National Land Cover Dataset [NLCD], see Chapter 5) and imagery datasets (e.g., National Agricultural Imagery Program, [NAIP]) that can be used at finer scales (Lark et al., 2020; Joshi et al., 2019), but these do not include information on attribution. Furthermore, there remains active and unresolved scientific debate inside and outside the scientific literature on the utility of some of these remotely sensed datasets for these purposes (Lark et al., 2022b; Lark et al., 2022a; Taheripour et al., 2022b; Copenhaver et al., 2021; Reitsma et al., 2016).

Finally, it is inherently difficult to separate the effects from the RFS Program from other factors. Many of these factors co-occurred in time and space, and thus they are highly correlated statistically with one another. This is the main criticism with much of the peer-reviewed literature, a relationship between the RFS Program and ethanol production is observed, and thus the causality is assumed but not tested. This chapter has attempted to overcome these challenges through the use of several independent lines of evidence, but each of these have their own limitations. Simulation models are used to isolate the estimated effect from the RFS Program, but these are limited by the current understanding of the systems that are coded into the models. There is no other “control” in an experimental sense, where ethanol growth or lack thereof in the United States can be observed to experimentally isolate the effect of the RFS Program. Statistical techniques are employed to try and isolate causality such as instrument variables, but even these are limited by the choice of instrument. There is also a wide range of factors that influence ethanol production including engineering components (e.g., MTBE and octane) and economic components (e.g., oil prices and RIN process). Correctly assessing all these factors is challenging. Nonetheless, no single study or approach leads to the conclusions here, but rather it is the confluence of findings from independent approaches and studies that lends confidence to the conclusions.

Thus, even though the estimates here may need to be revisited as additional studies are published, this approach, while not without limitations, provides a credible estimate of the scale of LCLM effects from RFS-attributable ethanol at the county level and nationally.

6.5 Likely Future Effects of the RFS Program

The likely future effects of the RFS Program on biofuel production, consumption, and several economic and environmental factors have been estimated by EPA in the Final Set Rule that covers volumes for 2023–2025.[100] The volumes in the Set Rule are based on a review of the implementation of the RFS Program to date and EPA’s analysis of six groups of factors specified in EISA. The six groups of factors analyzed in the Final Set Rule include many of the environmental end points in Section 204, in addition to other factors like the impact of renewable fuels on infrastructure, the cost to consumers of

[100] On July 26, 2022, the United States District Court for the District of Columbia filed a consent decree, which after stipulations from the parties, required EPA to sign a notice of proposed rulemaking (NPRM) to establish 2023 volumes for the RFS Program on or before November 30, 2022, and to sign a notice of final rulemaking finalized 2023 volumes by June 21, 2023. *Growth Energy v. Regan et al.*, No. 1:22-cv-01191 (D.D.C.), Document Nos. 12, 14, 15. EPA’s proposed RFS volumes for 2023-2025, 87 FR 80582 (proposed and signed on Nov. 30, 2022, and published in the Federal Register on Dec. 30, 2022) available in Docket No. EPA-HQ-OAR-2021-0427 at https://www.regulations.gov, were finalized on June 21, 2023. The estimates of cropland expansion from the RFS Program took considerable time and effort to develop; thus, they are based on the proposed volumes from November 30, 2022, rather than the final volumes from June 21, 2023. Because of that, the acreages discussed here are merely illustrative. However, since the total volumes from crop-based feedstocks changed fairly little between the draft and final rules, they are still relevant here.

transportation fuel job creation, and rural economic development.[101] The biofuels estimated to be consumed in each year in the Final Set Rule are presented in Chapter 2 (Table 2.4). Rather than repeating the extensive information in the Regulatory Impact Analysis (RIA) for the Final Set Rule, the key information relevant for the RtC3 is summarized here and the interested reader is referred to the RIA for more information.

The RIA of the Final Set Rule estimates the biofuel production and consumption that is likely attributable to the RFS for the future years. To estimate attribution, EPA conducted a detailed assessment on each fuel to estimate the future volumes and developed a reasonable counterfactual for a No-RFS Baseline for the years covered in the Final Set Rule (2023–2025).[102] In summary EPA considered the volume of renewable fuels that would be used to meet state biofuel use requirements. EPA then projected the cost for each of the biofuel types, taking into account distribution costs, the blending value of the fuel, and any available non-RFS incentives, and compared this to the cost of the fuel displaced by the renewable fuel. Renewable fuels that were cost-competitive with the fuels they displaced, after accounting for non-RFS incentives, were included in the No-RFS Baseline and are not projected to be directly attributable to the RFS Program. EPA estimated that much of the renewable fuel projected to be used in 2023–2025, including much of the cellulosic biofuel, biodiesel, and renewable diesel, was attributable to the RFS Program. Notably, however, EPA projected that ethanol would continue to be used nationwide in E10 blends in the absence of the RFS Program, so only a small portion of the ethanol projected to be used in 2023–2025, namely the ethanol used in higher level ethanol blends such as E15 and E85, is estimated to be attributable to the RFS Program. Significant imports of ethanol are not anticipated in 2023–2025 absent unexpected events such as widespread drought in the United States.

The volume of those total future volumes from Chapter 2 (Table 2.4) that are estimated attributable to the RFS Program are shown in Table 6.15. Thus, there are an additional 787 million gallons of corn ethanol estimated to be attributable to the RFS Program in 2025. Note that this volume is different from the estimated change in ethanol from 2022-2025 (Table 2.4, -189 million gallons), because the change from 2022 from all causes is different from the volume attributable to the RFS in 2025 (+787 million gallons). Though using different methods, this is within the range of estimates of attribution historically (Figure 6.21).

[101] Section 211(o)(2)(B)(ii) of EISA.
[102] See Chapters 2 and 3 of the RIA (Docket No. EPA-HQ-OAR-2021-0427).

EPA translated these volumes into changes in LCLM for a subset of the biofuels in Table 6.15. Many of the biofuels in Table 6.15 either do not use land in any significant amount (e.g., CNG/LNG or electricity from biogas FOGs, diesel/jet fuel from wood waste/MSW, FOGs), or are a byproduct of existing production and/or are produced in such small amounts that additional changes in LCLM is likely minor (e.g., corn oil). Thus, EPA focused on estimates of changes in LCLM for corn ethanol, soybean oil biodiesel and renewable diesel,[103] and canola oil biodiesel. EPA's assessment of LCLM for the Set Rule was based on the projected volumes attributable to the proposed Set Rule, rather than the Final Rule due to time constraints. These volumes changes are summarized in Table 6.16. While the volumes attributable to the RFS Program in the Final Set Rule differ slightly from the proposed rule, these differences are relatively small in total. For example, although the estimates

Table 6.15. Volume changes for candidate volumes relative to the No-RFS Baseline (million gallons).[a]

Fuel	2023	2024	2025
Cellulosic Biofuel	495	688	932
CNG/LNG from biogas[b]	495	688	932
Ethanol from CKF	0	0	0
Total Biomass-Based Diesel	1,991	1,925	2,107
Biodiesel	1,078	1,036	985
Soybean oil	841	757	755
FOG	-101	-92	-113
Corn oil	46	63	20
Canola oil	292	307	323
Renewable diesel	902	878	1,111
Soybean oil	457	671	729
FOG	99	90	110
Corn oil	130	-64	-20
Canola oil	216	182	291
Jet fuel from FOG	11	11	11
Other Advanced Biofuels	49	49	49
Renewable diesel from FOG	0	0	0
Imported sugarcane ethanol	0	0	0
Domestic ethanol from waste ethanol	0	0	0
Other[c]	49	49	49
Conventional Renewable Fuel	660	731	787
Ethanol from corn	660	731	787
Renewable diesel from palm oil	0	0	0
Total Renewable Fuel	3,195	3,393	3,875

[a] Electricity and CNG/LNG remain in ethanol-equivalent gallons in this table.
[b] Includes 147 million gallons of renewable diesel produced from soybean oil projected to be used to meet the supplemental volume requirement for 2023.
[c] Composed of non-cellulosic biogas, heating oil, and naphtha.

[103] This volume includes soybean biodiesel, soybean renewable diesel, and canola biodiesel. EPA initially intended to project land use impacts assuming canola oil and soybean oil biofuels had the same impacts. The volumes of soybean biodiesel/renewable diesel in Table 6.16 include the biofuels projected to be produced from both soybean oil and canola oil. EPA later decided to separately project the impacts of biofuels produced from canola oil, but did not reduce the projected volume of soybean biodiesel/renewable diesel to reflect this change. Thus, the estimated land use change for soybean biodiesel/renewable diesel is an overestimate. Further, the volumes in Table 6.16 for soybean biodiesel/renewable diesel also include a small volume of jet fuel and other advanced biofuel projected to be produced from soybean oil.

Table 6.16. Potential total U.S. acreage impacts for all crops due to increases in corn ethanol, soybean biodiesel and renewable diesel, and canola biodiesel and renewable diesel that can be attributed to EPA's Set Rule.

	Volume Increase in RFS Set Rule Proposal (million gallons)			Acreage Increase (million acres)		
Fuel	2023	2024	2025	2023	2024	2025
Corn ethanol	706	776	840	0.39	0.44	0.46
Soybean biodiesel/renewable diesel	1,950	1,920	1,890	1.57	1.78	1.93
Canola biodiesel/renewable diesel	240	240	240	0.26*	0.26*	0.26*

*Projected to occur in the North Dakota region

for canola went up, and corn and soy went down between the proposal and the final, the total amounts changed very little (i.e., from 2.8 billion gallons in the proposal to 2.9 billion gallons in the final[104]). Thus, the overall estimated effects on future changes in LCLM due to the RFS Program are likely representative even though based on the proposal. Here the highest year of effect from the proposed Set Rule is used for this summary. The increases in these three biofuels are associated with an increase in total cropland are shown in Table 6.16.[105]

Table 6.16 indicates that the RFS Set Rule could potentially lead to an increase of as much as 2.65 million acres of cropland by 2025. This would constitute approximately 1% of the projected U.S. acreage for major field crops in 2025 (USDA, 2022). This is estimated to occur from an increase of 0.46 million acres due to increased demand for corn ethanol, 1.93 million acres due to demand for biodiesel and renewable diesel produced from soybean oil, and 0.26 million acres due to biodiesel produced from canola oil. The increases in cropland from canola are expected to be mostly in North Dakota, where most of the canola currently is grown. The increases from corn and soybean are expected to be distributed across the Midwest. As discussed in greater detail in the supporting documentation for the Set Rule, the estimates of land use change attributable to the RFS Program are uncertain, and likely overestimate actual land use change attributable to the volumes in the Final Set Rule. EPA was generally unable to target specific areas for these projected future increases in cropland as with the historical estimates for corn ethanol in section 6.4. This is an area for future work. Again, it is important to note the significant assumptions and uncertainty inherent in estimating these nationwide acreage impacts numbers. See the RIA for the Set Rule for further information (docket, EPA-HQ-OAR-2021-0427).

[104] Docket No. EPA-HQ-OAR-2021-0427

[105] For convenience, the acreage increases for all three crop-based biofuels is shown here, rather than only showing corn, and this table is referred to again where discussed in Chapter 7.

6.6 Chapter Synthesis

6.6.1 Chapter Conclusions

- Many factors have impacted ethanol production and consumption in the United States historically, including higher prices of oil and gasoline, the replacement of methyl tert-butyl ether (MTBE) in reformulated gasoline (RFG) areas, the RFS Program, the Volumetric Ethanol Excise Tax Credit (VEETC), the octane value of ethanol, state programs, and air emission standards.
- The period of rapid growth in the ethanol industry was from 2002 to 2010, and nearly 40% of the increase in ethanol consumption had already occurred by 2006 (the first year of the RFS Program, RFS1[106]), and over 90% of the increase had already occurred by 2010 (the first year of the RFS2).
- Because the factors that affect ethanol production and consumption—including the RFS Program—change through time, so too does the estimated effect of the RFS Program on ethanol production and consumption.
- Evidence from simulation models, observed RIN prices, production exceeding consumption from the RFS standards, and other sources suggest that from 2006 to 2011 the RFS Program—in isolation—accounted for 0–1 billion gallons of ethanol per year, primarily by encouraging market growth and capital investment from the Energy Independence and Security Act (EISA) and to a lesser extent by stabilizing demand during the Great Recession of 2008–2009. In other years of this period, the RFS Program is estimated to have had no effect on ethanol production, with other factors having more influence.
- The synthesis of evidence suggests a dynamic range of effects from the RFS Program from 2012 to 2019 as well, with the largest effect in 2016 (0–2.1 billion gallons per year) primarily due to the RFS Program supporting the industry after other factors had either phased out (e.g., VEETC, MTBE) or diminished in effect (e.g., high oil prices).
- In sum over the entire period assessed, the RtC3 concludes that 0–9% of corn ethanol production and consumption is likely attributable to the RFS Program historically. Lower estimated effects of the RFS Program occur if the effect on market certainty is not considered, or if MTBE replacement by ethanol and transitions to match blending are assumed to be independent of, but coincident with, the RFS Program; larger effects occur if

[106] The RFS1 and RFS2 are described further in Chapters 1 and 2 and refer to the different versions of the RFS Program enacted under the Energy Policy Act of 2005 (RFS1) or the Energy Independence and Security Act of 2007 (RFS2).

market certainty is included, or if these other factors are omitted or ascribed to the RFS Program.

- Combining these estimated volumes attributable to the RFS Program with literature reviews and a recent statistical analysis suggests as a best available estimate that the RFS Program may be attributable for cropland expansion of zero to 1.9 ± 0.9 million acres, and additional acres of corn of zero to 1.9 ± 0.9 million acres of cropland, with the largest potential effect estimated in 2016.
- These best available estimates from econometrics of observed trends are consistent with other econometric studies once appropriate adjustments are made and are consistent with estimates from simulation models.
- The likely future effect of the RFS Program was estimated by EPA in the Final Set Rule on June 14, 2023, which estimated 787 million gallons of corn ethanol consumption in 2025 to be due to the RFS Program, potentially inducing up to 0.46 million acres of cropland expansion. These estimates are highly uncertain, due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes.
- Uncertainties in the estimated effect of the RFS Program on ethanol production remain, including the effect of the RFS Program in establishing market certainty before the mandates were in full effect, the cost, ability, and willingness of refiners to switch back to producing finished gasoline without ethanol if blending ethanol were no longer economical, and others. However, these factors are difficult to quantify with current tools available.
- The RFS Program created a guaranteed market demand for biofuels in the United States that certainly could have helped drive much of the increase in ethanol production and consumption in the United States. However, as events played out, non-RFS factors that also affect ethanol production and consumption (e.g., oil prices, octane value, MTBE bans, tax incentives, state programs) were favorable, and appear to sufficiently explain much or all of the increase in ethanol production and consumption historically in the United States.
- A modest effect from the RFS Program on corn ethanol does not preclude a larger effect on other biofuels (see Chapter 7) as the RFS Program affects many biofuels in addition to corn ethanol.

6.6.2 Uncertainties and Limitations

- Very few retrospective studies include factors that are known to influence corn ethanol production in addition to the RFS Program (e.g., oil price, MTBE phaseout, octane value);

thus, the conclusions in the RtC3 are based on a small number of studies that represent the best available information.

- Economic models largely omit behavioral factors (e.g., investor confidence) or other factors that are difficult to quantify; thus, even the most sophisticated models may underestimate the effects of the RFS Program.
- Among the many factors omitted that may be important, none of the evidence examined considered the cost or willingness of refineries to revert from producing BOBs back to finished gasoline. This could influence the effects of the RFS Program after roughly 2010 (after the concentration of ethanol in the gasoline pool reached nearly 10% nationwide and refiners switched to producing BOBs rather than finished gasoline) and into the future. If included, this factor would tend to reduce the impact of the RFS Program on corn ethanol production in years after 2010.
- It remains uncertain the relative contributions from the MTBE phaseout vs. other factors (including the RFS Program) in encouraging the buildout of infrastructure in the 2005–2007 time period.
- Most economic models with good market detail of the biofuels industry (e.g., include oil price, MTBE phaseout, octane value) have less detail for other sectors and coarse spatial resolution (e.g., multi-state areas); thus, using a single modeling framework to link the economic effects with the effects on land cover and land management remains a challenge.
- Inherent uncertainties in global equilibrium (economic) model simulations of agricultural markets are amplified when results are translated to acreage change, a factor exogenous to the model. Furthermore, a model that relies on a defined spatial extent as the basis for change cannot attribute specific changes observed at a finer scale to the economic factors or policies represented in model simulations.
- The fact that other factors are sufficient to explain the increase in ethanol production and consumption in the United States does not necessarily mean that they alone drove the increase in ethanol, future studies with more detail may modify or reverse these conclusions.

6.6.3 Research Recommendations

- Further research to improve CLCA approaches to better enable examination of the attributional effect from the RFS Program on spatially explicit changes in land cover and land management.
- Future studies on the RFS Program should attempt to include to the degree feasible the many federal and state subsidy programs that affect farming operations.

- Additional research is needed on quantifying the role the RFS Program vs. MTBE phase-out had in establishing market certainty and in contributing to the infrastructure buildout; and, the degree to which the conversion to match blending was or was not associated with the RFS Program.
- Additional economic and engineering research is needed on MTBE and octane components of ethanol consumption.
- Additional studies are needed on the effects of the RFS Program on non-ethanol fuels, and on how the RFS Program interacts with other policies, economic factors, and social trends, to influence biofuel production and consumption, including but not limited to market-mediated interactions with livestock markets, land management practices, and dietary change of food/feed type, quantity, and nutritional content.
- Ensemble modeling approaches (e.g., using GTAP-BIO, BSM, and statistical analyses) may be used to assess the future effects of the RFS Program. Using this approach, various assumptions could be considered to yield a probabilistic range of estimates.

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7. Attribution: Biodiesel and Renewable Diesel

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Key Findings

- Some of the same factors that drove ethanol trends in production and consumption in the United States contributed to biodiesel and renewable diesel trends, including federal tax credits, state incentives, and high petroleum prices and low agricultural commodity prices, especially in the early period of growth.
- There is much less information on biodiesel and renewable diesel compared with ethanol, and very few retrospective analyses on the relationship between the RFS Program and biodiesel and renewable diesel production. Therefore, this chapter does not provide a quantitative estimate of the amount of biodiesel and land attributable to the RFS Program in the RtC3 as was done in Chapter 6 for corn ethanol.
- The evidence available suggests that the RFS Program was binding on biodiesel and renewable diesel for the entire period of the RFS2 assessed (2010 to 2019). It does not appear that there was a binding effect prior to this given the lack of an individual biomass-based diesel (BBD) standard from 2006 to 2009 under the RFS1 (2006–2008) or for the first year of the RFS2 (2009), and low RIN prices during these years where data are available (2008–2009).
- Overall, biodiesel and renewable diesel production has been much more strongly dependent on federal and state policies (grants, tax incentives, income tax credits, RIN values, etc.) than has ethanol. The Biodiesel Tax Credit (BTC) and the RFS2 played particularly important roles. A different set of incentives drove production in the early phases compared to more recent years.
- It is not possible in the RtC3 to derive a robust estimate of the volume of soybean biodiesel specifically attributable to the RFS Program. However, economic models suggest that the RFS Program could increase biomass-based diesel consumption 0.6–1.1 billion gallons for every billion gallon increase in the biomass-based diesel volume obligations; and comparison of state and federal mandates suggest that while roughly 0–30% of biodiesel consumption may be due to state programs (mandates and low carbon programs like the Low Carbon Fuel Standard), the remaining 70–100% may be attributable to a combination of other factors, primarily the RFS Program and the BTC. The effects of the RFS Program cannot be isolated at this time for the historical period because most studies do not separate the RFS from other important factors that occurred at the same time such as the BTC and state programs.
- In addition to domestic effects, the RFS Program incentivized the import of foreign biodiesel from different sources in different years (e.g., Argentinian soybean biodiesel, Southeast Asian

palm oil). These direct volumes are small on a relative basis but could have important local effects overseas, and diversion of any vegetable oil toward biofuels could have indirect effects on these markets that are difficult to estimate.

- While this and other chapters have discussed the substitutability of different feedstocks into the food, feed, and fuel industries, the authors of this chapter are not aware of sufficiently rigorous studies that have addressed the impact of increasing demand for qualifying feedstocks (such as fats/oils/greases [FOGs] or soybean oil) for biodiesel and renewable diesel production on commodities that may be used as substitutes in other industries (such as other vegetable oils, including palm oil).
- The likely future effect of the RFS Program was estimated by EPA in the Final Set Rule issued on June 21, 2023, which estimated 1,484 million gallons of soybean biodiesel and renewable diesel consumption in 2025 to be due to the RFS Program. Initial estimates from slightly higher volumes suggest that the RFS Program could potentially lead to an increase of as much as 1.9 million additional acres of cropland. These estimates are highly uncertain due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes.

Chapter Terms (see Glossary): advanced biofuel, biodiesel consumption, biodiesel production, biodiesel, biomass-based diesel, blend wall, cellulosic biofuel, D4 RIN, D5 RIN, D6 RIN, FOG, renewable diesel, Renewable Identification Number (RIN), RIN transaction cost, total renewable fuel.

7.1 Introduction

This chapter discusses the effects of the RFS Program on historical production and consumption of biodiesel and renewable diesel. Just as Chapter 6 analyzed several different factors that could have influenced domestic ethanol production and consumption, this chapter examines the relative importance of the RFS Program on biodiesel and renewable diesel production and consumption compared to other potential drivers. This includes non-RFS federal programs, tax credits and incentives, macroeconomic trends, trade policies, and state mandates such as the California Low Carbon Fuel Standard (CA-LCFS). Although the focus of this chapter is on biodiesel produced from soybean, to understand the potential effects of the RFS Program one must understand broader biodiesel trends. This is different from ethanol, for which corn ethanol dominates the conventional biofuel standard in the United States. Throughout most of the chapter, biodiesel and renewable diesel are discussed in similar contexts.[1] It is difficult to

[1] Renewable diesel and biodiesel are not the same fuel. Renewable diesel is a hydrocarbon produced through various processes such as hydrotreating, gasification, pyrolysis, and other biochemical and thermochemical technologies. It meets ASTM D975 specification for petroleum diesel. Biodiesel is a mono-alkyl ester produced via

isolate the effects of the RFS Program on soybean planting and on the biofuels derived from soybeans, because soybean is grown for many markets (e.g., domestic feed, whole bean exports more recently to China) and commonly in rotation with corn as discussed in Chapters 3 and 5. Thus, as with Chapter 6 various lines of evidence are used to attempt to isolate these effects.

7.2 Historical Trends and Factors Potentially Affecting Biodiesel and Renewable Diesel Production and Consumption in the United States

As with ethanol, there are several potential drivers of the trends in biodiesel and renewable diesel production and consumption that have been suggested in the literature and elsewhere in this report. Biodiesel blending is distinct from corn ethanol blending in several important ways. For example, while the E10 blend wall is a dominant factor limiting ethanol consumption since about 2013, biodiesel and renewable diesel production does not appear to be directly affected by a biodiesel blend wall at current volumes. This is due to differences between ethanol and biodiesel compared with their fossil counterparts in terms of their fuel properties as well as the non-standardized blending levels. Renewable diesel is chemically similar to petroleum-based diesel (Ng et al., 2010) so it can be blended at any proportion; thus, it is not affected by the same engineering and logistical constraints as ethanol. Biodiesel has been approved by EPA for blending up to any level, but in practice most biodiesel in use today has a blend rate of 20% (B20) or less. This is largely due to diesel engine manufacturers setting their warranties based on biodiesel concentrations they feel confident their engines can handle. The current biodiesel standard the engine industry relies on is ASTM D-6751, which was determined based on a maximum of B20.[2] As this is multiple times above the current production volume of biodiesel, there is no practical blend wall for biodiesel at the time that is analogous to that for ethanol. Further obscuring the definition of a biodiesel blend wall, biodiesel and renewable diesel are commonly added to petroleum-based diesel at a wide range of concentrations (see Chapter 3 section 3.6.2). This is driven by a range of local practical, policy, and economic factors, including fuel cloud point limitations[3] in northern states during winter months for biodiesel. Considering these differences, and because the growth in the biodiesel industry was not characterized by as dramatic changes as the ethanol industry in production and consumption with distinct

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transesterification. Biodiesel meets ASTM D6751 and is approved for blending with petroleum diesel (DOE, Alternative Fuels Data Center, https://afdc.energy.gov/fuels/emerging_hydrocarbon.html). Both fuel types can be produced from soybean oil; fats, oils, and greases (FOGs); or any other number of potential feedstocks. These are often both advanced biofuels under the RFS2 and thus are combined here.

[2] The National Biodiesel Accreditation Program (https://bq-9000.org/) works with producers and marketers of biodiesel to ensure that biodiesel achieves the ASTM D-6751 standard (https://www.astm.org/Standards/D6751.htm).

[3] The fuel cloud point is the temperature at which small, solid crystals start to appear in the fuel. This is of concern in colder climates (e.g., in northern states) because fuels with higher blends of biodiesel can gel or freeze at higher temperatures relative to conventional diesel.

time periods, this chapter is divided into the most important factors rather than distinct time periods as was done for ethanol in Chapter 6. These are presented in general chronological order by year of first occurrence. The individual factors assessed here include early federal incentive programs, macroeconomic and external factors, the Biodiesel Tax Credit, state mandates and incentives, the RFS Program, and trade policies.

Biodiesel production and consumption increased in the United States beginning in 2005, decreased during the Great Recession from 2008 to 2009, and then increased until 2016 (Figure 7.1). After 2016, production continued to increase while consumption declined, with the two merging in roughly 2019. In years where consumption was higher or lower than production, there were net imports or exports, respectively.

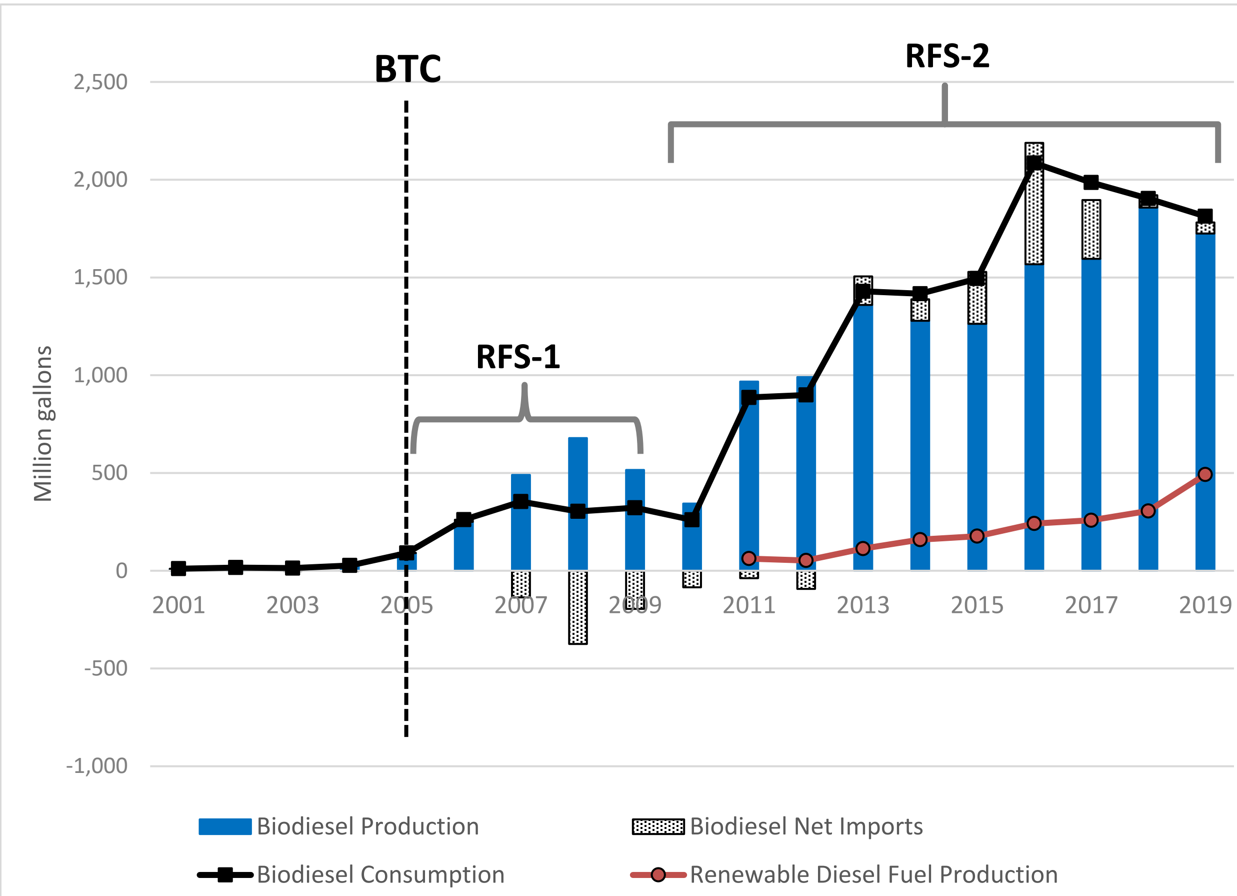

Figure 7.1. Biodiesel production, biodiesel consumption, and biodiesel net imports (imports – exports), and renewable diesel production from 2001-2019. Data from EIA, Monthly Energy Review[4], March 2020. Also shown are the year the Biodiesel Tax Credit (BTC) first went into effect (discussed in section 7.2.2) and the years of the RFS1 and RFS2 (discussed in section 7.2.4). The BTC expired and was renewed many times from 2005 to 2020.

[4] The EIA reports biodiesel production, consumption, and net imports on both monthly and annual scales (https://www.eia.gov/totalenergy/data/monthly/, Table 10.4).

7.2.1 Early Incentives for Biodiesel Production

The first federal program to significantly encourage growth in biodiesel production was the Bioenergy Program, which was started as an executive order (EOP, 1999) and funded through the USDA's Commodity Credit Corporation (Schnepf, 2013). This program started making payments in 2001 to producers of ethanol or biodiesel that showed annual increases in their production. The Bioenergy Program ended in 2006. Building upon and strengthening the Bioenergy Program, the Farm Bill of 2002 established programs that encouraged research, production, and use of biodiesel. During this period, from 2001 to 2004, U.S. domestic biodiesel production increased slowly, with annual increases averaging about 15.3 million gallons per year (Figure 7.1). By 2004 biodiesel production was still relatively low, at 28 million gallons per year.

7.2.2 Biodiesel Tax Credit

In June 2004, the American Jobs Creation Act (P.L. 108-357), created an excise tax and income tax credit (hereafter called Biodiesel Tax Credit, BTC[5]) of $0.50 per gallon for non-agri-biodiesel[6] such as yellow grease and $1.00 per gallon for agri-biodiesel such as soybean oil and animal fats. The Emergency Economic Stabilization Act of 2008 (P.L. 110-343) later granted the $1.00 per gallon credit to both types of biodiesel (Figure 7.1). The BTC, which was the first federal tax incentive for biodiesel (Schnepf, 2013), was set to expire at the end of 2009 but has been repeatedly renewed. Some years it was renewed retroactively, and some proactively, which appears to have affected growth. This intermittent driving effect from the BTC is evident in the year-to-year variation in growth, with peaks during years that the BTC was in effect, and troughs in years in which it was temporarily absent but retroactively applied (Figure 7.2, Table 7.1).

During 2000–2009 the biodiesel industry enjoyed multiple federal (and state) tax incentives in addition to the BTC designed to encourage production and investment in infrastructure (Table 7.2). There was steady growth from 2003 to 2007 (Figure 7.1 and Figure 7.2); however, beginning in 2008 and especially from 2009 to 2010 there was a steep decrease in production. This corresponded to the Great Recession (2008–2009) and then when the BTC had lapsed (2010) and there was uncertainty about if or when it would be reauthorized. After the BTC was reinstated in late 2010 as part of the Tax Relief, Unemployment Insurance Reauthorization, and Job Creation Act of 2010 (P.L. 111-312) production dramatically increased, but then decreased during repeated periods when the BTC was allowed to lapse (Figure 7.2).

[5] For more information about the BTC, refer to U.S. DOE's Alternative Fuels Data Center (https://afdc.energy.gov/laws/396).
[6] Agri-biodiesel is defined as a diesel fuel from virgin oils only (https://afdc.energy.gov/laws/342).

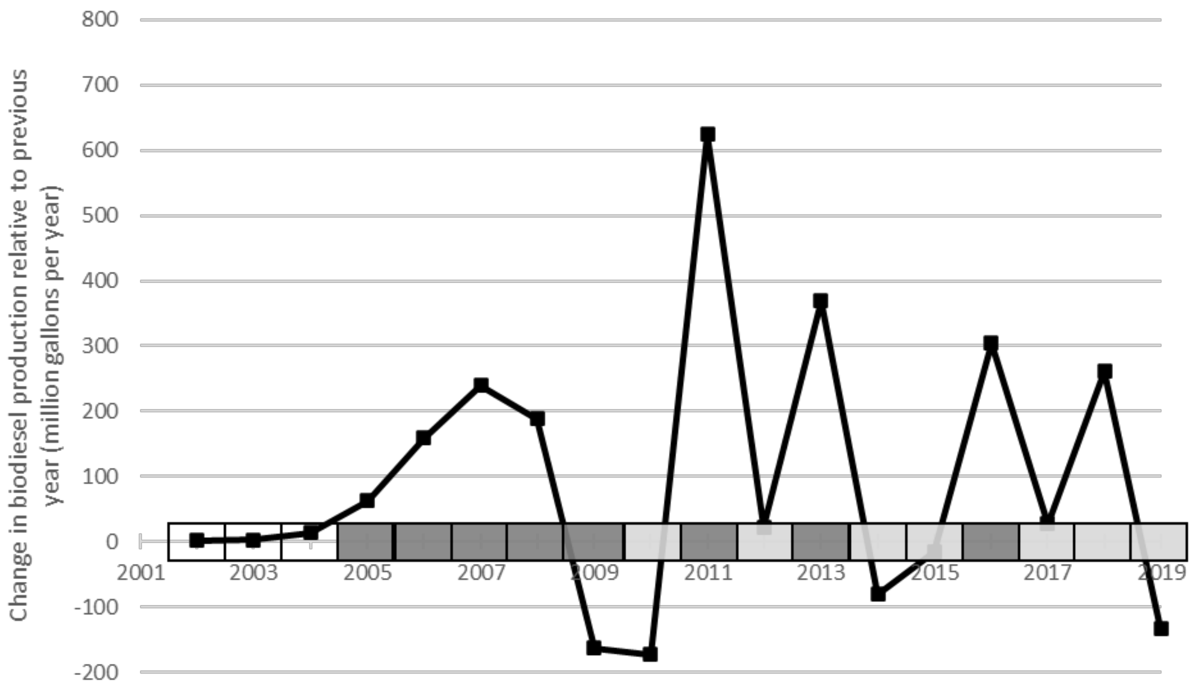

Figure 7.2. Change in biodiesel production relative to previous year. Low points are generally when the BTC was only available retroactively (light gray; i.e., 2010, 2012, 2014–2015, 2017–2019) and high points when it was available prospectively (dark gray; i.e., 2005–2009, 2011, 2013, 2016). Data from EIA, Monthly Energy Review, March 2020, https://www.eia.gov/totalenergy/data/monthly/, Table 10.4.

Table 7.1. Status of the Biodiesel Tax Credit through time. The BTC was prospective when the enactment date precedes the affected years and was retrospective when the enactment date was after the affected years.

Enactment Date	Legislation	Description	Affected Year(s)
June 2004	American Jobs Creation Act (P.L. 108-357)	Biodiesel Tax Credit (BTC) created.	2005–2006
August 2005	Energy Policy Act of 2005 (P.L. 109-58)	BTC extended.	2007–2008
October 2008	Emergency Economic Stabilization Act of 2008 (P.L. 110-343)	BTC extended and amended so that both non-agri-biodiesel and agri-biodiesel are qualified for $1.00 per gallon tax credit.	2009
December 2010	Jobs Creation Act (P.L. 111-312)	BTC reinstated retroactively for 2010 and extended prospectively through 2011.	2010–2011
January 2013	American Taxpayer Relief Act (P.L. 112-240)	BTC reinstated retroactively for 2012 and prospectively through 2013	2012–2013
December 2014	Tax Increase Prevention Act of 2014 (P.L. 113-295)	BTC reinstated retroactively for 2014	2014
December 2015	Consolidated Appropriations Act of 2016 (P.L. 114-113)	BTC reinstated retroactively for 2015 and prospectively through 2016	2015–2016
February 2018	Bipartisan Budget Act of 2018 (P.L. 115-123)	BTC reinstated retroactively for 2017	2017
December 2019	Further Consolidated Appropriations Act of 2020 (P.L. 116-94)	BTC reinstated retroactively for 2018–2019 and prospectively through 2022	2018–2022

Table 7.2. Federal biodiesel programs aside from the BTC (from Alternative Fuels Data Center).[7]

Incentive	Years Active	Description
Advanced Biofuel Feedstock Incentives	Effective October 27, 2010 (final rule) to present	Through the Biomass Crop Assistance Program, qualified producers can be reimbursed for a portion of the cost of establishing a biofuel crop and can receive annual payments. The program also matches payments to the producer for collecting, harvesting, storing, and transporting their crops to advanced biofuel production facilities.
Advanced Biofuel Production Grants and Loan Guarantees	2008 to present[8]	The Biorefinery, Renewable Chemical, and Biobased Product Manufacturing Assistance Program offers loan guarantees to a wide range of potential applicants that need to develop, build, and retrofit commercial-scale biorefineries that will produce advanced biofuels.
Advanced Biofuel Production Payments	2002 to present[9]	The Bioenergy Program for Advanced Biofuels provides eligible producers of advanced biofuels payments to expand production. Funds are limited for large producers.
Biodiesel Mixture Excise Tax Credit	2016 to present[10]	Blenders that produce a mixture of at least 0.1% diesel fuel earn a tax incentive of $1.00 per gallon of biodiesel, agri-biodiesel, or renewable diesel used to create the blend.
Small Agri-Biodiesel Producer Tax Credit	2010 to present[11]	Small (60 million gallon or less production capacity) agri-biodiesel producers may qualify for $0.10 per gallon tax incentive for agri-biodiesel that is sold and used by the purchaser in their trade or business to produce blends, sold and used by the purchaser as a fuel in a trade or business, sold at a retailer for vehicle fuel, used by the producer in a trade or business to produce agri-biodiesel and diesel fuel blends, or used by the producer as a fuel in a trade or business.

7.2.3 Macroeconomic and External Factors

Similar to ethanol, on the macroeconomic scale changes in biodiesel production appear to be impacted by trends in crude oil and diesel prices. The rate of biodiesel production increased after 2004 (Figure 7.1) when the BTC first went into effect and diesel prices climbed past $1.00 per gallon (Figure 7.3), which they had been at or below since at least 1990. As discussed in Chapter 6 (sections 6.2.2 and 6.2.3), oil and petroleum diesel prices generally increased from 2004/2005 until 2008, plummeted for a few years during the Great Recession, then climbed, peaking in 2011. Each of these periods of price increases correspond to periods of biodiesel production increases. They also, however, correspond to years in which the BTC was available prospectively (Figure 7.2). From 2007 (the first year of biodiesel price data from USDA ERS) to present, the price of biodiesel was on average $1.40 (standard deviation $0.40) higher than diesel, making a $1.00 BTC attractive to improve the economics of biodiesel production. The large spike in biodiesel price in 2012 was likely driven by the Midwestern drought

[7] Federal incentives and laws that directly encourage biodiesel production selected from a list (https://afdc.energy.gov/fuels/laws/BIOD?state=US) of all relevant incentives.
[8] The Biorefinery, Renewable Chemical, and Biobased Product Manufacturing Program (BAP) was funded through Section 9003 of the 2008 Farm Bill (https://fas.org/sgp/crs/misc/RL34130.pdf).
[9] The Advanced Biofuel Payment Program was authorized by Section 9005 of the Farm Security and Rural Investment Act of 2002 (https://www.federalregister.gov/documents/2019/12/27/2019-27396/advanced-biofuel-payment-program).
[10] Biodiesel and renewable diesel incentives were extended with the Consolidated Appropriations Act of 2016 (https://www.congress.gov/bill/114th-congress/house-bill/2029/text/pl?overview=closed).
[11] This tax credit was established in 2010 as part of the Tax Relieve, Unemployment Insurance Reauthorization, and Job Creation Act of 2010 (https://www.congress.gov/bill/111th-congress/house-bill/4853/text/pl?overview=closed).

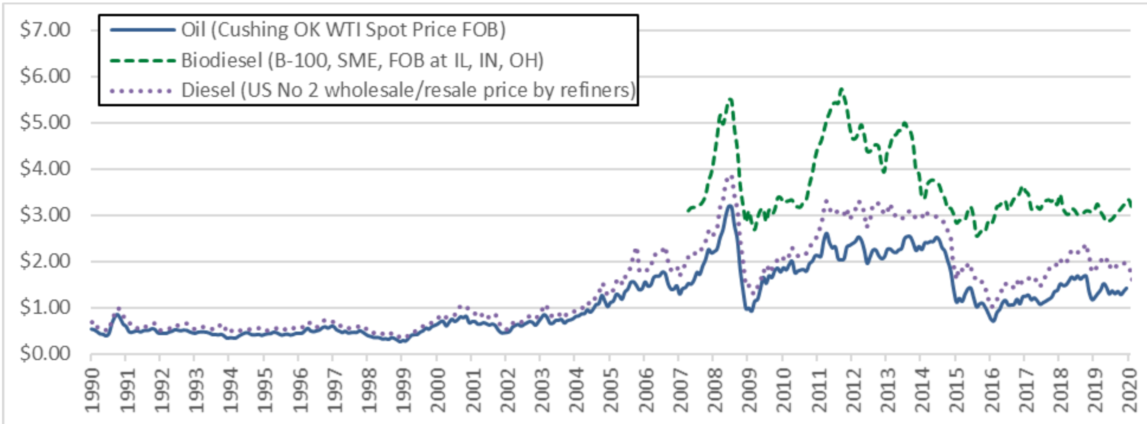

Figure 7.3. Monthly prices of crude oil (blue solid, from EIA), diesel (purple dotted, from EIA), and biodiesel (green dashed, from USDA ERS0).[12]

already mentioned (Riley, 2015), which suppressed harvests across the region, increasing soybean oil prices (Chapter 4, Figure 4.10).

Nested within the overall global trends in oil prices, other macroeconomic factors influenced biodiesel production rates. The price of biodiesel tends to reflect trends in the economics of oil and diesel, especially soybeans. From late-2005 through 2006 increasing petroleum prices and low agricultural commodity prices, (which also were shown to contribute to increased ethanol production, (see Chapter 4 and 6), likely also played a role in the growth in the biodiesel industry (Schnepf, 2013). Soybean and corn markets are influenced by a common set of supply-side variables (land, machinery, and chemical costs, as well as weather) and demand-side factors (competing demands for animal feed and other soy or corn products, see Chapter 4). Therefore, changes in supply-side and demand-side factors may contribute to changing trends in biodiesel consumption.

Within the same time period of rising diesel prices, and especially in 2008 and 2011, the price of soybean oils rose (Figure 7.4). This made it relatively less economical for the biodiesel industry to obtain soybean oil from which biodiesel is made. In addition, the 2008 global financial crisis reduced demand for transportation fuel and lowered diesel prices. Both factors likely contributed to the production declines during 2009 to 2010 (Figure 7.1).

These macroeconomic factors likely contributed some to biodiesel and renewable diesel production trends. However, these macroeconomic factors on their own never reduced the biodiesel price

[12] Crude oil (West Texas Intermediate, spot price in Cushing, OK, which is often used as a reference for the price of crude oil streams) price data from EIA (https://www.eia.gov/dnav/pet/pet_pri_spt_s1_m.htm), diesel (U.S. No. 2 wholesale/resale price by refiners) price data from EIA, and biodiesel prices based on USDA-ERS Agricultural Marketing Service, National Weekly Ag Energy Roundup (https://www.ers.usda.gov/about-ers/partnerships/strengthening-statistics-through-the-icars/biofuels-data-sources/). EIA has biodiesel production data but not price prior to 2007.

below the price of diesel fuel (Figure 7.5). Only after considering the $1.00 BTC was biodiesel cheaper than petroleum diesel for most of the period from 2007 to 2020 (Figure 7.5). This is in contrast with ethanol, which was cost competitive with gasoline absent the VEETC in many years after the year 2010 (Chapter 6, Figure 6.4c). Therefore, while broader macroeconomic factors may have impacted the production of biodiesel and renewable diesel, they may not have been sufficient to drive the production of these fuels absent the incentives provided by the BTC and other programs discussed later, including the RFS Program and other federal and state incentives.

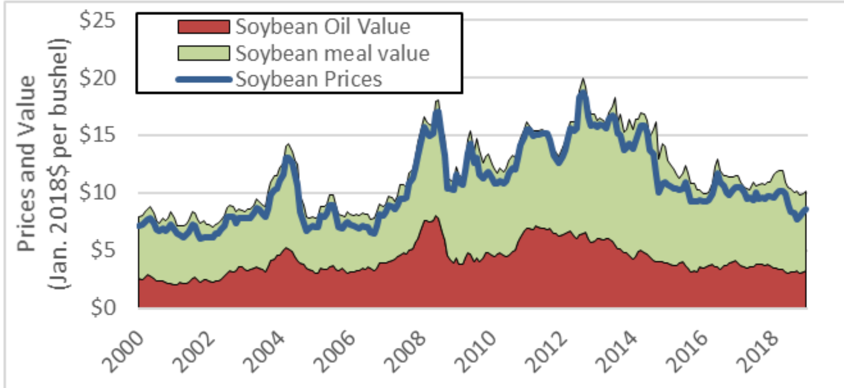

Figure 7.4. Soybeans and related products prices and value (copied from Chapter 4, Figure 4.11). The soybean oil and soybean meal values are stacked to show the total value of the products produced when crushing soybeans.

a

b

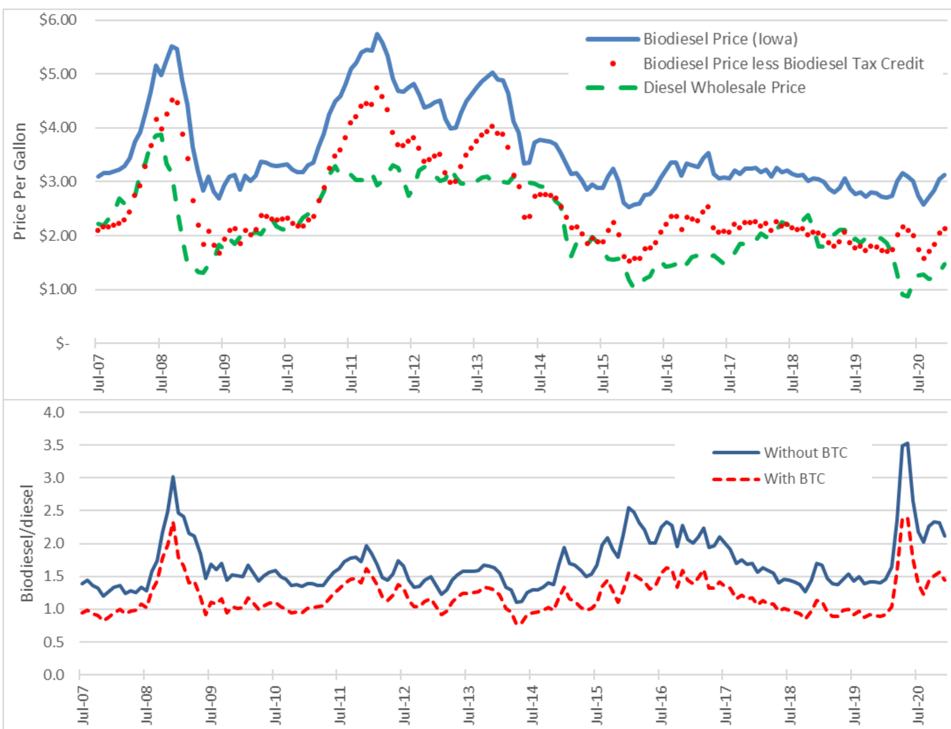

Figure 7.5. Biodiesel and diesel prices through time. In a) Biodiesel spot price in Iowa without BTC (blue solid), biodiesel price with BTC (red dotted), and diesel price (green dashed). In b) Ratio of the price of biodiesel/biodiesel with (red dashed) and without the BTC factored in (blue solid). Source same as Figure 7.2. Price ratios above below 1.0 suggest biodiesel is cost competitive with diesel, all else being equal.[13]

[13] A $1.00 BTC is used here because soybean biodiesel received the $1.00 credit for the entire period and FOGs also received it after 2008.

7.2.4 RFS Program & RIN Markets

As discussed earlier, the RFS Program was established in 2005 as part of EPAct, which set a single total renewable energy standard. Biodiesel could be used to meet this single volume requirement, but in practice nearly the entire volume requirement under the RFS1 was satisfied with corn ethanol (Figure 7.1 and Chapter 2, Table 2.1).[14] In 2007 the Energy Independence and Security Act (EISA) created RFS2, which built upon RFS1. RFS2 included higher mandated biofuel consumption volumes and specified requirements for four different types of renewable fuel: cellulosic biofuel, biomass-based diesel (BBD), advanced biofuel, and total renewable fuel. Mandates for all four categories of renewable fuel were not implemented by EPA until 2010 (see Chapter 1 section 1.1 and Chapter 6 section 6.2 for more information).

Biodiesel and renewable diesel produced from specified feedstocks generally qualifies as biomass-based diesel, though biodiesel and renewable diesel produced at legacy facilities can generate conventional renewable fuel RINs.[15] Since the biomass-based diesel standard is nested within the advanced biofuel[16] and total renewable fuel standards (see Chapter 1, Figure 1.2), biodiesel and renewable diesel can also be used to satisfy either of these obligations. Thus, biodiesel production is potentially influenced by the RFS Program in two ways, through direct biomass-based diesel obligations and the broader advanced biofuel and total renewable fuel obligations.

Biodiesel production increased rapidly in 2011 (Figure 7.1 and Figure 7.2). From 2010 through 2013, rates of biodiesel consumption followed the biomass-based diesel mandates relatively closely. Since approaching the E10 blend wall in roughly 2013 (see Chapter 1 section 1.3.2 and Chapter 6 section 6.2), biodiesel and renewable diesel consumption has exceeded the RFS volume requirement for biomass-

[14] From 2006 through 2010 total domestic biodiesel consumption was 1.5 billion gallons (data from EIA Monthly Energy Review, https://www.eia.gov/totalenergy/data/monthly/). During this same period domestic ethanol consumption was 45.9 billion gallons (data from EIA Monthly Energy Review, https://www.eia.gov/totalenergy/data/monthly/). Even in 2007 and 2008, when biodiesel was increasing above 500 million gallons (Figure 7.1), ethanol consumption represented ~93% of the total biofuel consumed in these years.

[15] Biodiesel and renewable diesel that does not meet the 50% GHG reduction threshold and is produced at legacy facilities can generate conventional renewable fuel RINs. From 2012–2022 annual RIN generation of D6 RINs for biodiesel and renewable diesel ranged from 0 to approximately 450 million RINs per year. However, many of these RINs were retired for reasons other than compliance with the RFS Program, indicating that while RINs were generated for these fuels, they were not used in the United States. Actual use of legacy biodiesel and renewable diesel in the United States peaked at 258 million RINs (161 million gallons) in 2015. No conventional biodiesel or renewable diesel has been used in the United States since 2017.

[16] As defined in the approved fuel pathways (https://www.epa.gov/renewable-fuel-standard-program/overview-renewable-fuel-standard) under the RFS Program, both advanced biofuels and the nested biomass-based diesel must have a 50% life cycle reduction in greenhouse gas emissions compared to a 2005 petroleum baseline and be produced from renewable biomass (see Chapter 1). Biomass-based diesel must be either biodiesel or non-ester renewable diesel and cannot be co-processed with petroleum. Advanced biofuel includes a broader range of renewable fuels, including biodiesel and renewable diesel that is co-processed with petroleum.

based diesel and has approached the advanced biofuel volume (Figure 7.6). The difference in these years may have been made up by imports (discussed in section 7.3.5).

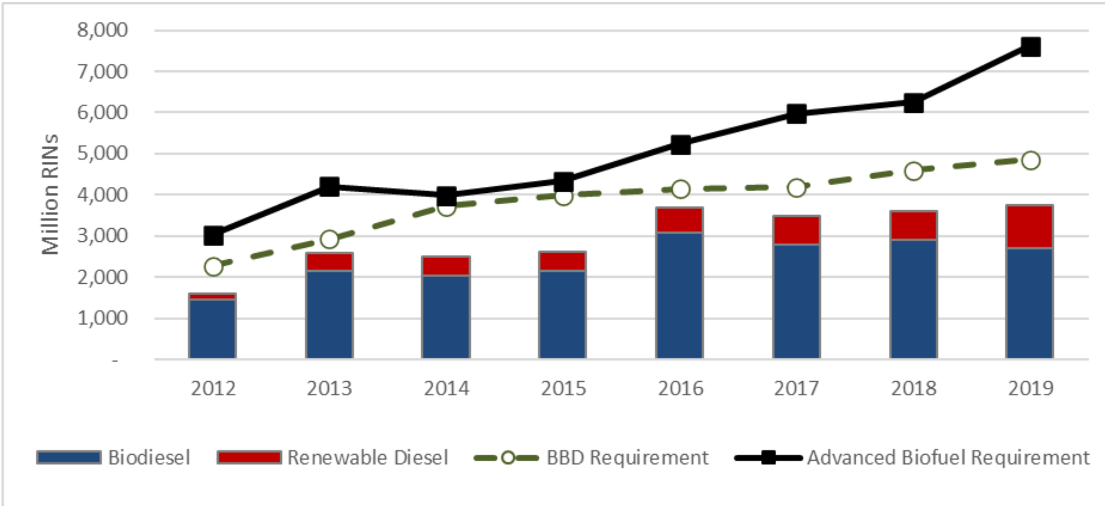

Figure 7.6. Advanced biodiesel and renewable diesel consumption in the United States (stacked bars; from EPA EMTS data) and biomass-based diesel (BBD) and advanced biofuel RFS volume requirements (lines; ex-post values as with Figure 6.10). Note the RFS2 went into full effect part way through 2010, so EPA does not have full year data for biodiesel and renewable diesel use prior to 2012. This figure does not include advanced biofuels other than biodiesel and renewable diesel (such as CNG/LNG derived from biogas) that could be used to meet the advanced biofuel volume requirements, nor does it indicate the additional demand for non-ethanol fuels that results from the shortfall in the supply of conventional ethanol relative to the implied conventional renewable fuel volume requirement.

7.2.5 State Incentives

In addition to federal incentives such as the BTC and the RFS Program, several states have implemented programs to incentivize the production and use of biodiesel and renewable diesel (see Appendix E). These programs include use mandates, state production tax credits and incentives, clean fuels programs, and various investment incentives and tax breaks. Biodiesel blending varies significantly from state to state, and these state-level incentives appear to be a key factor in these differences (see Appendix E).

Several states (Minnesota, New Mexico, Oregon, Pennsylvania, and Washington) have implemented requirements that all diesel sold in the state must contain a minimum quantity of biodiesel or renewable diesel.[17] These mandates range from 2% in Pennsylvania and Washington to 20% in Minnesota.[18] Some of these mandates are structured so that the mandates increase as the volume of biodiesel produced increases. These state mandates create demand for biodiesel and renewable diesel, some of which might exist in the absence of the RFS Program or other federal incentives.

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[17] These state mandates were enacted at various times. The mandates were effective starting in the following years: Minnesota (2005), Oregon (2009), Washington (2009), Pennsylvania (2010), and New Mexico (2012).
[18] The B20 mandate in Minnesota only applies to diesel sold in the summer months (April–September). In the winter months (October–March), Minnesota's minimum biodiesel requirement drops to 5%.

In addition to these mandates, two states have adopted "clean fuels" programs that provide incentives for fuels with low carbon intensity, California and Oregon.[19] Unlike the RFS Program, these clean fuels programs do not specify volume requirements for different types of renewable fuels, but rather specify target carbon intensities for all transportation fuel sold in the state. Fuels with a higher carbon intensity than the target generate debits, while fuels with a lower carbon intensity generate credits that can be sold to other parties. These clean fuels programs have resulted in significant demand for biodiesel and renewable diesel in the states where they exist. The CA-LCFS was enacted legislatively in 2007, but did not go into full effect until 2011, and as of 2019 approximately 830 million gallons of biodiesel and renewable diesel were used in California (Figure 7.7). An additional 37 million gallons of these fuels were used in Oregon in 2019. These programs in California and Oregon likely also create demand for biodiesel and renewable diesel that would exist in the absence of the RFS Program. California's biodiesel and renewable diesel volumes represent significant portions of national consumption levels (see Appendix E).

Perhaps the most common form of state incentives are tax credits or exemptions from state taxes for blends containing biodiesel and renewable diesel. Some of these incentives can also be significant. For example, Illinois exempts all biodiesel blends that contain between 11% and 99% biodiesel or renewable diesel from the state sales and use tax (normally 6.5%).[20] Texas also has a large incentive for biodiesel and renewable diesel blending, exempting the renewable portion of biodiesel blends from the state excise tax (normally $0.20 per gallon). Other states, including Hawaii, Iowa, Kansas, Maine, Montana, North Dakota, Rhode Island, and South Dakota also have state-level incentives for the use of biodiesel and renewable diesel (see Appendix E, Table E.1). Still more states have incentives that apply to only portions of the diesel used in the state, such as heating oil or diesel fuel used in state fleets.

State-level incentives appear to have a significant impact on the consumption of biodiesel and renewable diesel in states where they are active. They may also have a broader effect on regional production of biodiesel and the associated feedstocks. Some of these incentives may be significant enough to drive the production and use of biodiesel and renewable diesel in the absence of the RFS Program. Others may be less significant. Determining the degree to which this is the case would require a year-by-year and state-by-state analysis, which is not feasible for the RtC3.

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[19] In California the program is referred to as the California Low Carbon Fuel Standard (CA-LCFS), and in Oregon it is referred to as the Oregon Clean Fuels Program. More recently, Washington enacted a program that went into effect January 1, 2023 (https://ecology.wa.gov/Air-Climate/Reducing-Greenhouse-Gas-Emissions/Clean-Fuel-Standard).

[20] The Illinois tax credit for biodiesel blends that contain between 11% and 99% biodiesel or renewable diesel began in 2004. According to the EIA SEDS Illinois consumed more biodiesel than any other state from 2004 to 2012.

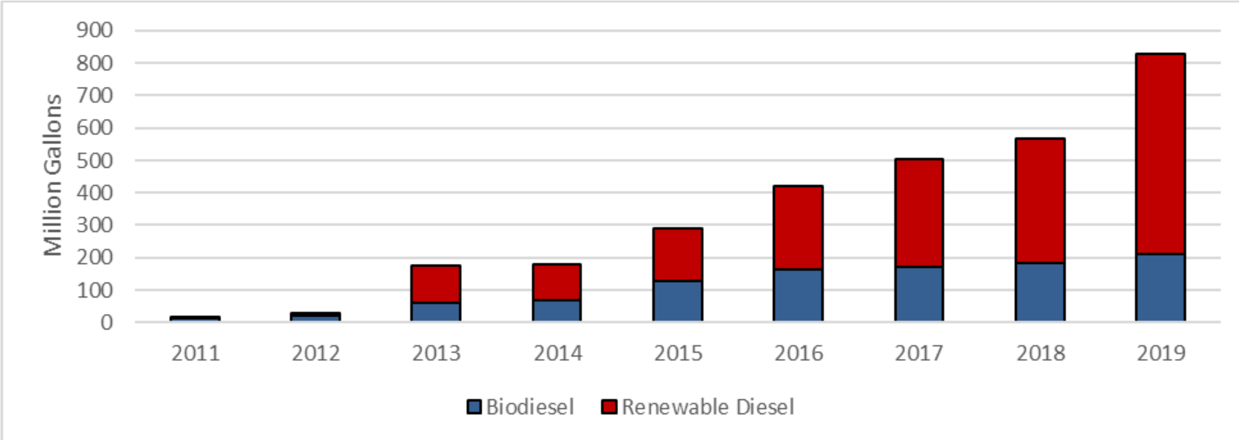

Figure 7.7. Biodiesel and renewable diesel use in California's LCFS program in million gallons. Data and charts from CARB LCFS data dashboard.[21]

7.2.6 Trade Policies

The purpose of the RFS Program is to encourage consumption of renewable fuel in the United States. Previous sections in this chapter as well as Chapter 4 have analyzed factors controlling domestic sources of the biodiesel used to meet RFS Program mandates. The United States has largely been able to meet domestic biodiesel consumption with domestic production (Figure 7.1). However, there have been periods when the United States exported and imported large quantities of biodiesel (Figure 7.8). The dynamics of international biofuel (biodiesel, renewable diesel, and ethanol) trade are discussed for more countries and for more biofuels in Chapter 16, but here the role of the RFS Program is considered, specifically, on biodiesel trade. There are numerous, complicated, and interacting factors connecting U.S. biofuel consumption to international trade. There are two main phases in historical U.S. biodiesel trade discussed below: (1) United States as net exporter (2006–2012), and (2) United States as net importer (2013–2019). Analyzing these separately can help determine the importance of international trade policies on biodiesel and renewable diesel production.

The significant quantities of biodiesel exports from 2006 to 2012 were likely driven by a combination of federal tax policies and international trade policies that were favorable to U.S. exports (Figure 7.8). By this point early incentive programs discussed above had allowed producers to establish domestic production capacity that enabled the U.S. biodiesel industry to respond quickly to the growing demand resulting from early federal and state mandates. The increased domestic production, combined with strong international markets for biodiesel and renewable diesel, enabled increased exports for these fuels through 2012. An important factor of the high trade volumes during many of these years is an

[21] For information, charts, and data about the California LCFS see the California Air Resources Board (https://ww3.arb.ca.gov/fuels/lcfs/dashboard/dashboard.htm).

international trade policy loophole commonly called "splash and dash."[22] From 2007 to 2010 there was a particularly active period of international biodiesel trade, characterized by high volumes of both imports and exports in the same year (Figure 7.8). During this time, U.S. policy had allowed parties to import biodiesel from foreign producers, blend it in the United States with a "splash" of diesel to receive the credit from the BTC, and then "dash" the resulting B99 biodiesel to foreign markets, especially Europe, and take advantage of incentives available to biodiesel in those markets. This period of economically advantageous biodiesel import/export ended in October 2008 with the passing of the Emergency Economic Stabilization Act (P.L. 110-343). In addition to this loophole being closed in 2008, the EU applied duties and tariffs beginning in March 2009 that effectively cut off demand for biodiesel imports. Although the policy loophole and temporary EU demand during the "splash and dash" phase explains most of the increased U.S. exports in the early years of biodiesel growth, exports have remained low but consistent since 2010 (Figure 7.8).

After 2012, the United States switched from being a net exporter to a net importer of biodiesel. During this phase (i.e., 2013–2019, Figure 7.8), domestic and international factors affected U.S. biodiesel production and consumption (see also International Impacts, Chapter 16). Increasing RFS2 mandates, combined with the role that biodiesel can fill to satisfy the total renewable fuel and advanced biofuel categories, have become increasingly important after reaching the E10 blend wall, increasing demand for both domestic and imported biodiesel and renewable diesel. This intersection of the RFS Program and trade is discussed later in the synthesis section (section 7.3.4). An important international factor that facilitated greater imports of biodiesel is the presence of production subsidies and incentives in other countries. After establishing a national biodiesel strategy and other internal policies, Argentina had strong biodiesel exports to the United States from 2013 to 2017, peaking at over 400 million gallons in 2016 (Chapter 16 section 16.4.1). Imports from Argentina to the United States have dropped to zero since 2017, however, due to the United States imposing additional duties on biodiesel imports. Biodiesel and renewable diesel imports were also relatively strong from Southeast Asia from 2013 through 2019. Similar to Argentina, governmental support for exports played an active role, as well as the availability of relatively cheap feedstock in the form of palm oil. Imports from Indonesia have dropped since the end of 2017, when the United States imposed additional duties on imports from that country. However, biodiesel and renewable diesel imports from other parts of Southeast Asia (e.g., Singapore, South Korea) have remained relatively steady, helped by availability of relatively cheap feedstocks, production capacity, and other factors. In recent years, through 2023, imports of Canadian canola and canola-oil-based fuels have

[22] https://www.iisd.org/gsi/news-events/united-states-closes-controversial-splash-and-dash-biofuels-subsidy-loophole

also increased as reflected in the estimates of the likely future (see Chapter 6 section 6.5). See Chapter 16 for additional discussion of trade policies.

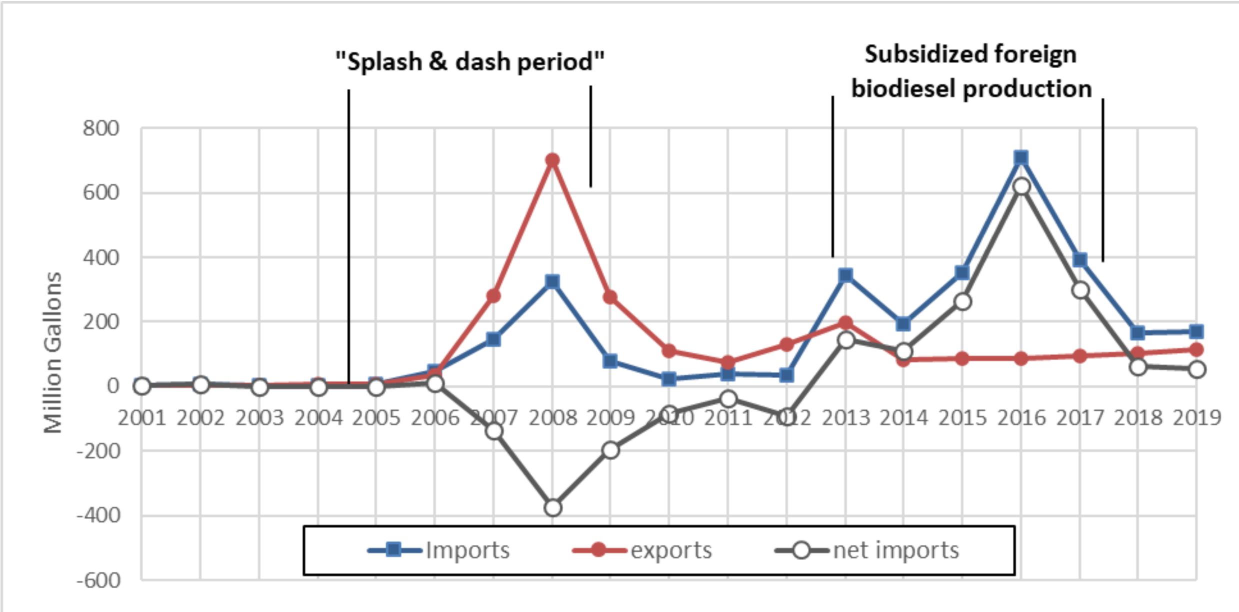

Figure 7.8. Biodiesel imports and exports. Source: EIA, Monthly Energy Review, March 2020, https://www.eia.gov/totalenergy/data/monthly/.[23]

7.3 Evidence of the Impact to Date of the RFS Program on Biodiesel and Soybean Production and Consumption

As was done in Chapter 6 for corn ethanol, several sources of information are used to assess the effect of the RFS Program on biodiesel production and consumption in the United States. These include: (1) comparison of the annual RFS Program mandates with consumption, (2) observation of D4 and D5 RIN prices,[24] and (3) results from the peer-reviewed literature.[25] As discussed in Chapter 4 and section 7.3.3, there is much less information on the effects of the RFS Program on biodiesel and associated feedstocks in contrast with corn ethanol and corn. Thus, the review of effects on the biofuel is combined with effects on the feedstock in this chapter.

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[23] The "splash and dash" period is described in the subsequent text, which began with the American Jobs Creation Act in June 2004 and ended in October 2008 with the Emergency Economic Stabilization Act.
[24] Whereas D6 RINs correspond with conventional biofuel (which is mostly corn ethanol in the United States), the corresponding RINs for soybean biodiesel are D4 (biomass-based diesel) and D5 (other advanced biofuels, see Chapter 1 for more details).
[25] Comparable information that is available for corn ethanol from the Biomass Scenario Model (BSM, see section 6.3.4), OTAQ's analysis of the economics of blending (see section 6.3.5), and other model studies are not yet available for soybean and thus are not included in the RtC3.

7.3.1 Mandate Versus Consumption Levels

As discussed in Chapter 6 (see section 6.3.1), when consumption is higher than the associated mandate, that is evidence that the RFS Program is not binding in that year. When consumption is close to the mandate, the RFS Program may or may not be binding, and more information is needed to determine the potential binding effect (e.g., RINs, section 7.3.2). The RFS Program first contained volume requirements for biodiesel and renewable diesel in 2010 with the promulgation of the RFS2. Because there was no separate biodiesel mandate prior to 2010, most of the total renewable fuel from 2006 to 2009 was made up by corn ethanol, and because the volume of corn ethanol produced and consumed in the United States exceeded the RFS volume obligations during this period, these criteria suggests that the RFS Program was not binding for biodiesel from 2006 to 2009.

From 2010 and up to 2013—prior to reaching the E10 blend wall—total consumption of biodiesel and renewable diesel (both of which generally qualify as biomass-based diesel) in the United States was approximately equal to the biomass-based diesel volume requirement in the RFS2 (Figure 7.6). This suggests a possible binding effect of the RFS Program on biodiesel production in those years. Additional volumes of biodiesel and renewable diesel were generally not economically competitive with ethanol blended as E10, and thus advanced ethanol (generally imported sugarcane ethanol) was generally likely used to meet the remaining advanced volume requirements after the biomass-based diesel volume requirement was satisfied. These dynamics are illustrated by the volume of biodiesel and renewable diesel used in the United States from 2013 to 2020, which exceeded the volume required by the BBD volume obligation (Figure 7.6), and the volume of ethanol imports to the United States, which decreased significantly after 2013 (see Chapter 6 Figure 6.9). After 2013, ethanol began to reach the blend wall, but the BBD, advanced, and total renewable biofuel mandates continued to increase under the RFS2. Thus, biofuel imports switched from ethanol-dominated to biodiesel-dominated after 2013 (compare Figure 6.9 with Figure 7.8). Biodiesel and renewable diesel were generally the lowest cost option for satisfying the additional RFS2 obligations once the E10 blend wall was reached. Thus, comparisons of the mandates with consumption provides evidence that the RFS Program under the RFS2 may have had a significant impact on biodiesel and renewable diesel consumption in the United States under either the BBD or advanced standards for the entire period from 2010 to 2020.

7.3.2 D4 and D5 RIN Prices

RIN markets, which are discussed in detail in Chapter 4 and 6 (for ethanol), offer a strong indication of the influence the RFS Program has on biofuel consumption. When RIN prices are above transactional costs,[26] the RFS Program is assumed to be binding for that biofuel and period. As discussed in Chapter 6 (section 6.3.2), there are no EPA data for RIN prices prior to 2010 and the RFS2; and, because most biofuel under the RFS1 was corn ethanol, it is assumed that the RINs from 2006 to 2009 approximate those for corn ethanol. Prices for biomass-based diesel (D4) RINs have been above the transaction costs since 2010 when separate RIN categories were created (Figure 7.9). This is strong evidence that, unlike ethanol, the RFS Program has been binding since 2010 indicating that at least some volume of biodiesel and renewable diesel has been attributable to the RFS Program.

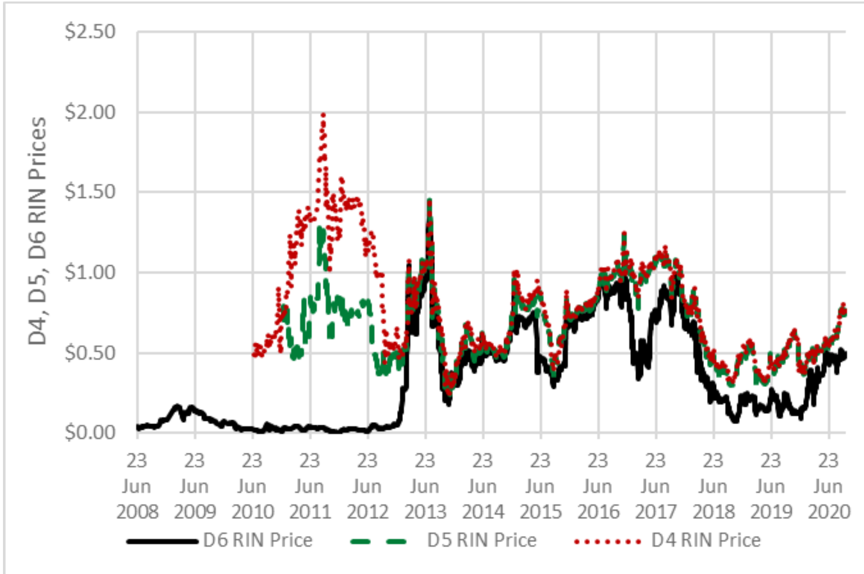

Figure 7.9. Daily RIN prices (June 23, 2008–2019). Source: Argus (copied from Chapter 4, Figure 4.4, y-axis in U.S. dollars). Prior to 2010, all qualifying renewable fuels generated the same type of RIN. The vast majority of renewable fuel produced prior to 2010 was corn ethanol, so pre-2010 prices as D6 RIN prices are shown.

From 2010 through 2012 biomass-based diesel (D4) RINs traded at a higher price than advanced biofuel (D5) RINs. This suggests that D5 advanced biofuels, such as sugarcane ethanol from Brazil and some FOGs and soybean biodiesel, were the marginal advanced RIN[27] during periods when total ethanol consumption was below the E10 blend wall. Since 2013, prices for D4 and D5 RINs have been nearly identical (Figure 7.8). This suggests that since reaching the blend wall in 2013, no more ethanol could be easily blended (whether conventional biofuel from corn ethanol from the United States or advanced biofuel from Brazilian sugarcane), and biodiesel or renewable diesel have been the marginal fuel supplied

[26] Transactional costs are the minimal costs of recording and trading RINs, roughly a few cents per RIN, discussed in more detail in Chapter 6.
[27] Here marginal RINs and biofuels are discussed, which means that due to the nested nature of the RFS Program, the most cost-effective way to meet the RFS obligations may change. The most cost-effective biofuel initially was corn ethanol up to the blend wall. Since reaching the E10 blend wall, biodiesel and renewable diesel have generally been the most cost-effective way to meet RFS obligations above the volume of ethanol that can be blended as E10.

to meet the advanced biofuel volume requirement.[28] At various times since 2013 the conventional renewable fuel (D6) RIN price has risen to the price of D4 and D5 RINs. This suggests that during these time periods biodiesel and renewable diesel production in excess of the biomass-based diesel mandate was the marginal fuel to meet both the advanced biofuel and total renewable fuel volume requirements, as excess biodiesel and renewable diesel became the most cost-effective way for producers to comply with their RFS obligations (Irwin, 2018).

The large variation in D4 and D5 RINs over this period appears to be due in part to the interplay between the RFS Program and whether or not the BTC was in effect (and whether prospective or retrospective), as well as variation in the prices of crude oil and feedstocks used to produce biodiesel and renewable diesel. In years where the BTC expired and was only reinstated retroactively, the RFS Program may have contributed to the bulk of the added incentive over what biodiesel the market would have consumed otherwise, and thus the D4 and D5 RIN prices were higher. In years with a prospective BTC, the credit absorbed a portion of the potential effect from the RFS Program, and thus the D4 and D5 RIN price was lower (see Chapter 6, Box 6.2. What are RINs for more background). In these years the BTC acted like a fuel consumption subsidy. The subsidy encouraged consumption of lower-cost petroleum-based fuels first, but this caused the indirect effect of increasing biofuel demand as well.

7.3.3 Peer-Reviewed Literature

Much of the peer-reviewed literature on the RFS Program has focused on the effects on corn ethanol and corn (see Chapter 4). Thus, studies that examine the effects of the RFS Program on biodiesel production and consumption are lacking. Focusing on the few studies that are available, Chapter 4 found from a subset of six studies that without the RFS Program mandates, production of biodiesel would have been low (0.2–0.4 7 billion gallons) and most of this biodiesel production would have come from FOGs (Moschini et al., 2017; Meyer et al., 2013; Babcock, 2012; Huang et al., 2012; U.S. EPA, 2010; Hayes et al., 2009). These studies estimate that biodiesel production would have increased by 0.6–1.1 billion gallons with a 1 billion gallon mandate for biomass-based diesel (Chapter 4, Table 4.3). Thus, there is nearly a 1:1 correspondence between the mandate and biodiesel production. All but one of these studies either included the cost of oil in their estimates or excluded the BTC, suggesting limited utility for the purposes for this chapter of assessing the effect of the RFS Program specifically.

A combined modeling approach that used a computable general equilibrium model showed the RFS Program was responsible for a 1.6% increase in soybean production during 2004-2011 compared to

[28] This situation may have changed in 2019 with the 1 p.s.i. extension to E15 (later revoked and extended), but that effect appears small to date (EPA Docket # EPA-HQ-OAR-2019-0136, "Estimating the impacts of the 1psi waiver for E15") and this chapter is focused on the retrospective analysis during the major period of growth from 2002 to 2012.

the historical baseline of the same period (Taheripour et al., 2022). The historical baseline was calculated by applying a set of exogenous shocks and using the model to determine production, consumption, and trade levels required to meet observed regional crop production in the absence of the RFS Program (Taheripour et al., 2022).

As part of the calculations of indirect land use change that were used in the development of the CA-LCFS, the California Air Resources Board (CARB) used two models (GTAP-BIO, AEZ-EF) and estimated the impact of the RFS Program on land cover changes (CARB, 2015). Given a soy biodiesel increase "shock" of 0.812 billion gallons, 0.00 to 0.05 million acres (0.00 to 0.02 million hectares) of forest, 0.00 to 0.05 million acres (0.00 to 0.02 million hectares) of pasture, and 0.5 to 0.7 million acres (0.2 to 0.3 million hectares) of cropland pasture was converted to soybean.

As noted in Chapter 6 (section 6.4.1.2), EPA recently compiled biofuel lifecycle GHG estimates from the scientific literature as part of the Agency's analysis of climate change impacts for the Set Rule. This included a review of published estimates of the land use change impacts of corn ethanol production, which was published in the Draft Regulatory Impact Analysis (DRIA) for the Set Rule, and portions relevant for the RtC3 are summarized here.[29] The review in the DRIA focused on GHG analyses, but many of the same models estimate changes in land-cover-land-management (LCLM) in the process of the lifecycle GHG analysis, as changes in LCLM are one of the primary drivers of net GHG emissions. Those results for soybean biodiesel are summarized in this section. For further details on this review see Chapter 6 (section 6.4.1.2) and the docket for the Draft and Final Set Rule (EPA-HQ-OAR-2021-0427).

Figure 7.10 summarizes estimated crop area changes domestically and internationally associated with soybean oil biodiesel production from the Draft Set Rule. The estimates from the 2010 RFS2 rule are excluded from this chart to improve legibility as they project a much larger amount of cropland change (6.6 Mac/Bgal). There is a group of estimates between 1.2 and 1.5 Mac/Bgal, including estimates with GLOBIOM, MIRAGE, and the GTAP-BIO modeling done with CARB for the CA-LCFS. The lowest group of estimates, 0.2 to 0.5 million acres per billion gallons, all come from the GTAP-BIO model. The most recent GTAP-BIO estimates are the lowest among the group, at 0.24–0.25 million acres per billion gallons. Between the GTAP-BIO modeling for CARB and the more recent estimates, a number of updates were made to the GTAP-BIO model that lowered the cropland area estimates, including revising the assumptions that determine crop intensification in response to price changes and multi-cropping

[29] Full text of the DRIA is available online (https://www.epa.gov/renewable-fuel-standard-program/proposed-renewable-fuel-standards-2023-2024-and-2025) and under docket EPA-HQ-OAR-2021-0427. The most detailed review of land use change estimates was included in the DRIA. The DRIA content remains relevant, although EPA decided this review did not bear repeating in the Final RIA. This review is not included in section 6.3 because these studies were not designed to separate the effects from the RFS Program from the effects of biofuels more generally. Furthermore, many of these models lack industry detail for such an analysis. Their strengths, rather, are in the integration of effects across sectors and regions associated with biofuel production and consumption.

(Taheripour et al. 2017). From this review the range of estimates for domestic land use change are 0.1–0.96 million acres per billion gallons (see Chapter 16 for the associated international land use changes). Unlike corn ethanol, empirical estimates of cropland area changes attributable to soybean oil biodiesel production were not identified (e.g., Chapter 6, Table 6.12). Thus, there are no empirical comparisons with the modeled estimates in Figure 7.10.

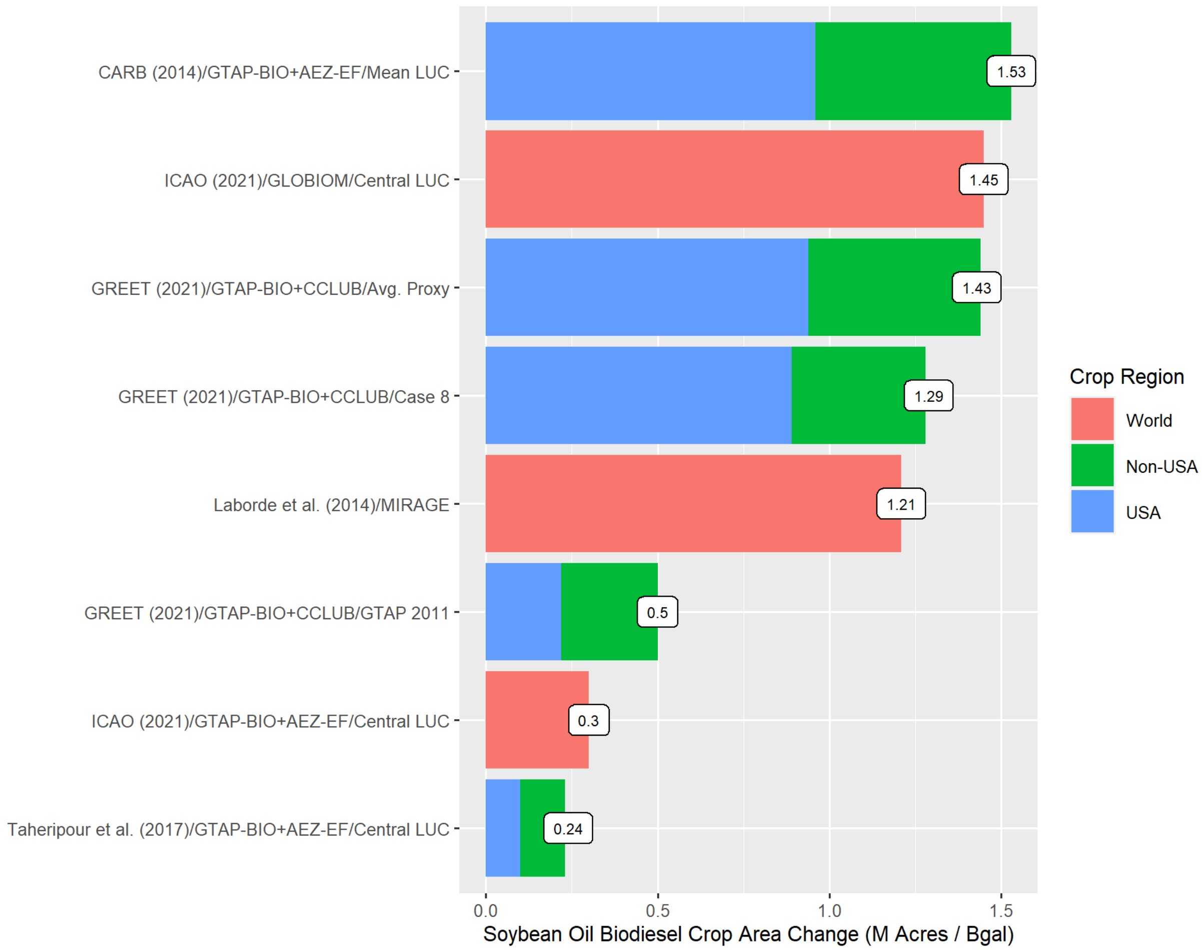

Figure 7.10. Cropland area change estimates per billion gallons of biodiesel by study for soybean biodiesel. The name on the y-axis for each bar/estimate includes multiple descriptors separated by "/". In order, these descriptors are the author or other name (e.g., CARB), the name of the model used to estimate land use change impacts, and in some cases a brief descriptor of the scenario modeled. For studies that did not report disaggregated estimates by USA and non-USA, only the world total is reported. Scenarios modeled and definitions of cropland differ across studies ICAO (2021). Estimates for soybean oil to jet fuel were adjusted based on the assumed jet fuel yield relative to biodiesel. Data sources: CARB (2014), ICAO (2021), Laborde et al. (2014), and Taheripour et al. (2017).

It is important to account for uncertainties associated with modeling studies. In the case of the Taheripour et al. (2022) model, soy biodiesel was not simulated independently of corn ethanol. In the CARB modeling done for the CA-LCFS, the model was simulated with a shock in demand. However, Figure 7.1 shows that the increase in biodiesel was not immediate, and model results based on shocks in demand can miss market responses and other factors that could ameliorate changes in RFS Program mandates (Scher and Koomey, 2011). Regardless, the available peer-reviewed literature agrees with the empirical evidence in 7.3.1 and 7.3.2, and suggests that the RFS Program may have had an effect on increasing biodiesel production in the United States, and the magnitude appears significant relative to the mandate. The effect the RFS Program may have had on establishing market certainty for soybean biodiesel is very difficult to quantify, as it is for corn ethanol. The effect on soybean biodiesel cannot be quantified in the RtC3. This section does not include a conclusive assessment of the impact of the RFS Program on the price of soybeans or soybean plantings. These relationships are complicated by the fact that historically most of the value of a bushel of soybeans has come from soybean meal, which is controlled by meat production, foreign export demand, and other factors beyond the scope of this report.[30] Further complicating attempts at using soybean prices to assess the impact of the RFS Program, the value of soybean oil shifted upward relative to soybean meal starting around 2001. The relationship between biodiesel production, soybean prices, and soybean planting is an area where further research is needed.

7.3.4 Model Comparison Exercise Technical Document

In June 2023, EPA published the "Final Renewable Fuels Standards Rule for 2023, 2024, and 2025."[31] The final rulemaking package included a Model Comparison Exercise Technical Document which, among other things, compared land use change impact results from five models for a series of U.S. biofuel consumption scenarios.[32] This Model Comparison Exercise (MCE) was motivated by needs identified at a virtual public workshop on biofuel GHG modeling on February 28 and March 1, 2022.[33] Specifically, this workshop identified that, in support of a better understanding of the potential land use change and other impacts associated with biofuels, it would be helpful to compare available models and identify how and why the model estimates differ.

Motivated by this need, EPA conducted the MCE with five models: the Greenhouse Gases, Regulated Emissions, and Energy Use in Technologies Model (GREET), Global Biosphere Management Model (GLOBIOM), Global Change Analysis Model (GCAM), Global Trade Project (GTAP) model, and Applied Dynamic Analysis of the Global Economy (ADAGE) model. See Table 6.11 for more details on each model. To facilitate appropriate comparisons of these models, EPA ran three common scenarios through each framework: (1) a reference case; (2) a corn ethanol consumption shock with an additional

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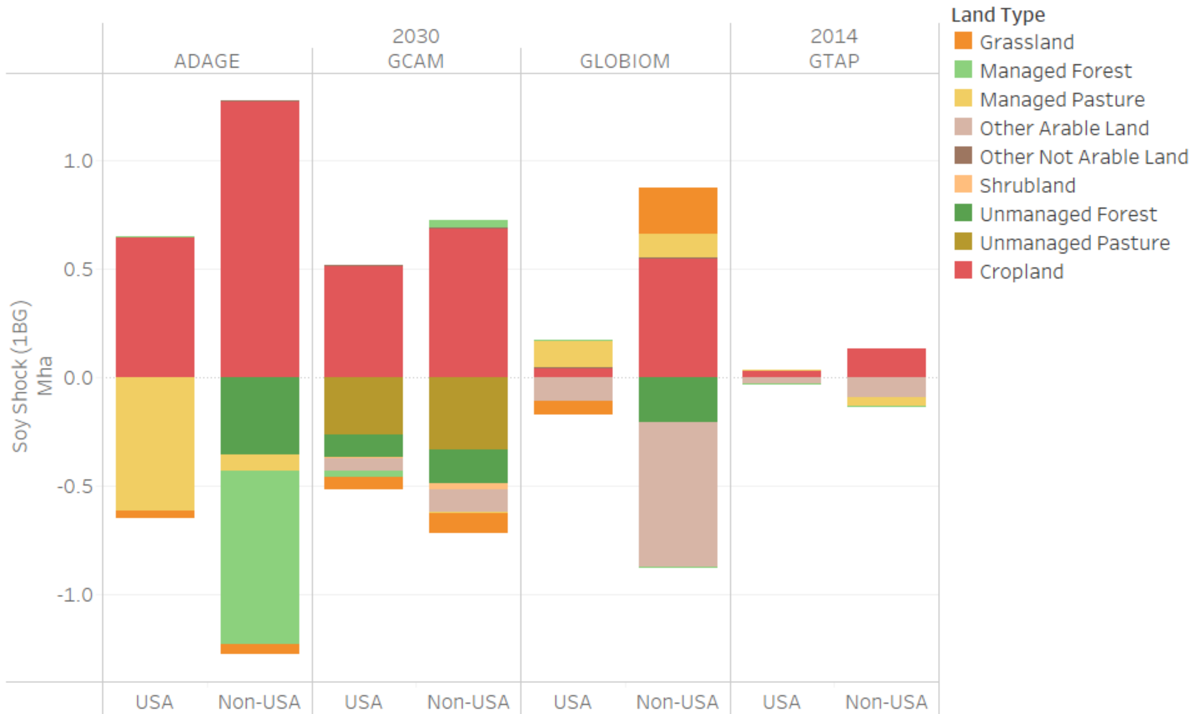
[30] For more information about the interactions between feed, livestock markets, and corn and soybean production see Chapter 4, section 4.5.
[31] See 88 FR 44468
[32] See EPA-420-R-23-017
[33] For more information see the Federal Register Notice, "Announcing Upcoming Virtual Meeting on Biofuel Greenhouse Gas Modeling." 86 FR 73756. December 28, 2021. More information is also available at https://www.epa.gov/renewable-fuel-standard-program/workshop-biofuel-greenhouse-gas-modeling.

one billion gallons (0.076 QBTU[34]) of U.S. corn ethanol consumption in each year, with all other U.S. biofuel consumption volumes set by assumption at the reference case levels (also referred to as the "corn ethanol shock"); and (3) a soybean oil biodiesel consumption shock with an additional one billion gallons (0.118 QBTU) of U.S. soybean oil biodiesel consumption in each year, with all other U.S. biofuel consumption volumes set by assumption at the reference case levels (also referred to as the "soybean oil biodiesel shock"). Effects from the soybean oil biodiesel shock are discussed here and from the corn ethanol shock are discussed in section 6.4.1.4. All of these scenarios were hypothetical and designed solely for the purpose of evaluating and comparing the models.

Four of the five models compared in the MCE provided estimated land use change impacts for the soybean oil biodiesel shock (the fifth model, GREET, does not endogenously model land competition and thus did not report these types of results). Figure 7.11 shows the estimated impacts on cropland and other land cover types, resulting from the soybean oil biodiesel shock. The figure shows the range in land use change impacts estimated by this suite of models, both domestically and internationally outside the United States. All four models estimate some quantity of new cropland would be cultivated in response to increased consumption of soybean oil biodiesel. However, the model estimates show significant variation in the quantity of additional cropland cultivated, the location of that new cropland cultivation (domestically versus internationally), and in the types of land cover which are estimated to be converted to cropland in response to increased consumption of soybean oil biodiesel. For example, while all four models estimate that a significant majority of the increase in cropland will occur outside the United States, the models differ regarding cropland extensification in the United States; ADAGE and GCAM estimate larger increases in U.S. cropland area than do GLOBIOM and GTAP. For further discussion of the motivations, methods, and results of this study, and the drivers behind these variations in model estimates, please refer to the MCE Technical Document.[35] EPA has made no firm methodological decisions based on the MCE but continues to gather feedback from stakeholders on the results and findings of this report to support future model developments and decisions. This work is ongoing; EPA will make further announcements regarding progress on these efforts at a later date.

[34] QBTU stands for Quadrillion British Thermal Units and is a unit of energy.
[35] See EPA-420-R-23-017



Note: this figure appears as Figure 7.6-2 in the original report. Due to differences in model structure and temporal coverage, results are presented for 2030 for ADAGE, GCAM, and GLOBIOM, and for 2014 for GTAP

Figure 7.11. Difference in land use (million hectares) in the soybean oil biodiesel shock relative to the reference case in 2014 (GTAP) and 2030 (ADAGE, GCAM, GLOBIOM).

7.3.5 Synthesis of Evidence for the Effect of the RFS Program on Biodiesel Production and Consumption

This chapter discusses some similarities but also a few key differences in the drivers of the biodiesel and ethanol industries. The differences were especially pronounced below the E10 blend wall when biodiesel and ethanol were largely independent. Above the E10 blend wall, biodiesel and ethanol drivers became more intertwined. Whereas ethanol production was strongly affected by several non-RFS Program factors below the E10 blend wall (e.g., MTBE phaseout, octane), these do not affect biodiesel, and thus biodiesel production appears to have been more dependent on financial incentives and the RFS Program. Both ethanol and biodiesel are affected by the price of oil, but to date crude oil prices have not been high enough for biodiesel to be cost competitive with petroleum diesel without incentives like the BTC (Figure 7.5). The types of incentives and mandates changed over time and came from both non-federal, such as state programs, and federal sources, such as the BTC and the RFS2. Other factors such as macroeconomics and foreign trade policies have also impacted the production, import, and consumption of biodiesel and renewable diesel.

Prior to 2010, the RFS Program did not contain specific volume requirements for biomass-based diesel or advanced biofuel—there only was a total renewable fuel standard. As discussed in Chapters 1 and 2, nearly the entire volume requirement for renewable fuel was satisfied with ethanol from 2006 to 2009, almost all of that from U.S. corn, with small amounts of biodiesel and imports originating from Brazil (see Table 2.1). These early years of biodiesel production were small, and likely more affected by the BTC, which was prospective over this period, rather than the RFS Program, which had no biodiesel mandate in these years. Furthermore, general RIN prices for total renewable fuel remained relatively low (Figure 7.9). Thus, available data suggest that the RFS Program itself was not responsible for a significant portion of the biodiesel and renewable diesel until 2010 when the RFS Program was expanded to the RFS2.

With the expansion of the RFS Program in 2010, which included a specific biodiesel mandate as well as other mandates that could be fulfilled with biodiesel (e.g., advanced biofuels), the available data strongly suggest that the RFS Program has significantly impacted the production, import, and consumption of biodiesel and renewable diesel. Prices for RINs of biomass-based diesel (D4) and advanced biofuel (D5) have never dropped to levels that represent transaction costs (Figure 7.9), which suggests that these volume requirements have been binding in each year. Total production and import of biodiesel and renewable diesel have been similar to the RFS volume requirements for biomass-based diesel (2010–2012) and advanced biofuel (2013–2019), further suggesting the impact of the RFS Program on the production of these fuels (Figure 7.6). The available literature, although sparse, also supports this conclusion.

More recently, the RFS Program may have also contributed to the importation of biodiesel from foreign countries. During the most recent phase when the United States was a net importer (i.e., 2013–2019, Figure 7.8), three major drivers affected U.S. biodiesel production and consumption, two domestic and one foreign (see also International Impacts, Chapter 16).[36] First, high RFS Program mandates above the E10 blend wall, combined with the role that biodiesel can fill to satisfy the advanced biofuel category, have become increasingly important, increasing demand for both domestic and imported biodiesel and renewable diesel. Additional incentives provided by California's LCFS program since 2011 have also likely played a role in the increasing volume of imported biodiesel and renewable diesel. Biodiesel production subsidies in other nations, particularly Argentina, also facilitated U.S. imports from 2013 to 2017. Argentina accelerated soybean production in the late 1990s (Tomei and Upham, 2009) and, with the help of a national strategy established in 2001 (Naylor and Higgins, 2017), developed a modernized biodiesel production system. Argentina's soybean industry was bolstered with incentives and tax policies that created plentiful supplies and was favorable to export (Naylor and Higgins, 2017) The United States

[36] The RFS Program likely did not affect the earlier "splash and dash" period of 2007–2009 since (a) the RFS Program did not have a biodiesel mandate during this time, and (b) the phenomenon coincided with the trade loophole that encouraged import and export in the same year to take advantage of the BTC.

imported 435 and 341 million gallons of soybean biodiesel in 2016 and 2017, respectively. Imports from Argentina dropped to zero after 2017 (Figure 7.8), however, due to a U.S. antidumping complaint and countervailing duties announced by the United States in August 2017 (USDA FAS, 2018).

The RFS Program currently does not contain an approved pathway for biodiesel or renewable diesel produced from palm oil. However, palm oil that meets the renewable biomass definition in the RFS Program can generate D6 RINs if it is produced at a legacy production facility.[37] In some years, particularly when D6 RIN prices were relatively high from 2013 to 2017, EPA data indicates that D6 RINs from foreign legacy biodiesel and renewable diesel facilities were significant (140–300 million gallons from Southeast Asian palm oil, Table 2.1). These volumes are small relative to the total production of palm oil in the region (i.e., 0.1-1.9%, see Chapter 16 section 16.6 for more information), and relative to the total U.S. pool (<2% for all years, see Chapter 2 Table 2.2), but even small effects in sensitive ecosystems may be concerning (see Chapter 16 for greater discussion). This suggests that the RFS Program may have incentivized the import of some biodiesel and renewable diesel produced from legacy palm oil biorefineries in the past and that it may continue to do so in the future, especially in years when D6 RINs are relatively high.

Another, perhaps more important, way that the RFS Program may incentivize the production of palm oil is by enabling the biodiesel and renewable diesel industry to outbid other industries for RFS-qualifying feedstocks such as soybean oil and FOGs. These industries, primarily animal feed and oleochemicals, may then turn to lower-cost palm oil (Figure 7.12), thus increasing global demand for palm oil. This indirect effect may be discernible in information on trade. As the use of soybean oil and FOG to produce biodiesel and renewable diesel has increased, imports of palm oil and palm kernel oil have also increased, from 2.8 billion pounds in 2010 to 4.1 billion pounds in 2018.[38] While other factors, such as the FDA ban in 2015 on partially hydrogenated oils, have also played a significant role in the increasing imports of palm oil and palm kernel oil,[39] there does appear to be an association between the use of soybean oil and FOG to produce biodiesel and renewable diesel and palm oil and palm kernel oil imports.

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[37] Renewable biomass includes planted crops and crop residue harvested from existing land that was cleared or cultivated before December 19, 2007, trees and tree residue from a plantation that was cleared before December 19, 2007, animal waste and byproducts, slash and pre-commercial thinning residue from non-ecologically sensitive forestland, biomass from within 200 feet of buildings in areas of high risk of wildfire, algae, and separated yard waste or food waste (including recycled cooking and trap grease). A legacy facility is one that was in production prior to December 2007. Both terms are defined for the RFS Program in the scoping language of 40 CRF 80.1401 (https://www.law.cornell.edu/cfr/text/40/80.1401).

[38] Palm oil and palm kernel oil import data from USDA oil crops yearbook (https://www.ers.usda.gov/data-products/oil-crops-yearbook).

[39] The FDA released their final conclusions in 2015 that partially hydrogenated oils were not considered generally safe (https://www.fda.gov/food/food-additives-petitions/final-determination-regarding-partially-hydrogenated-oils-removing-trans-fat).

a

b

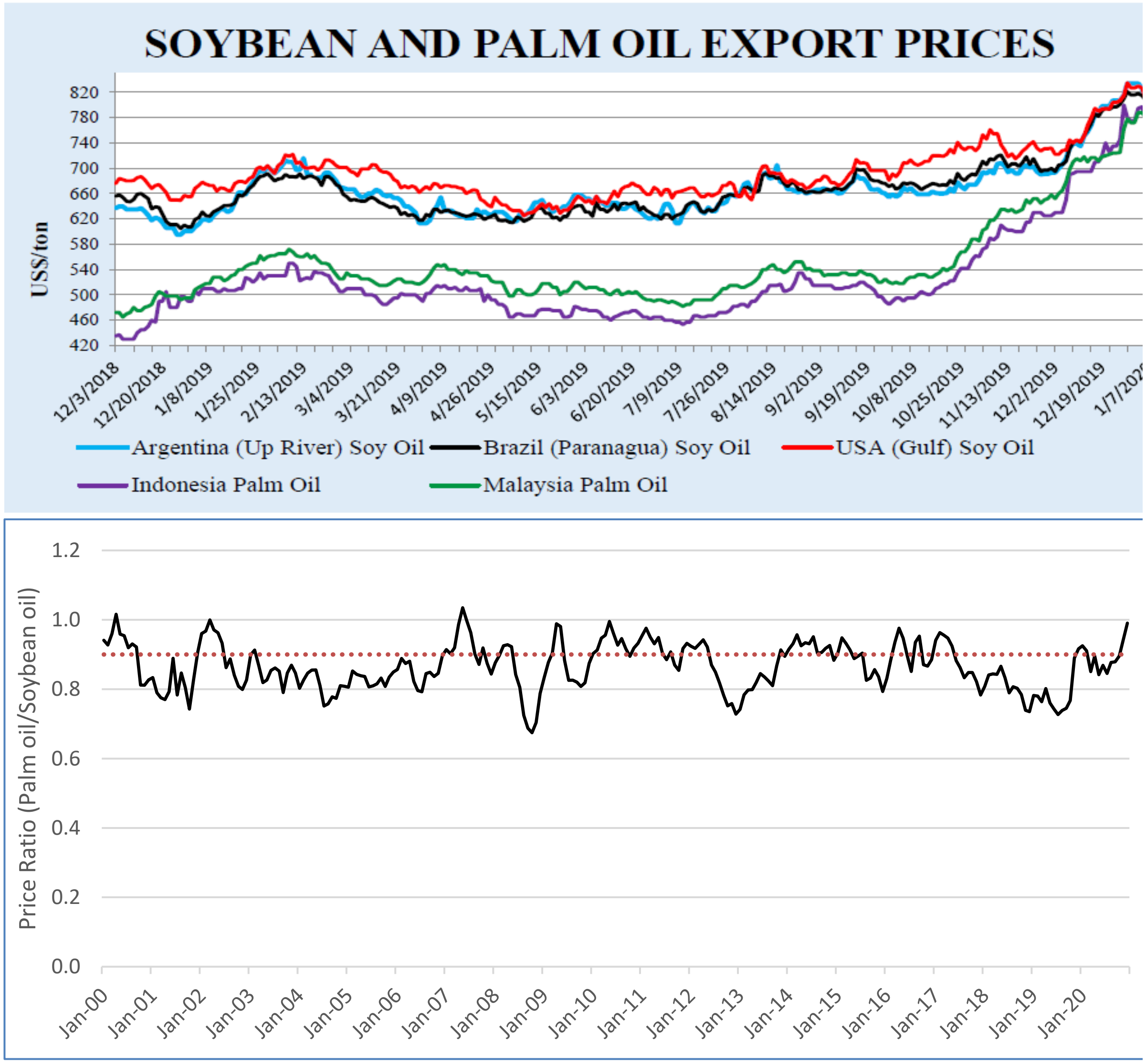

Figure 7.12. Soybean and palm oil export prices. Shown are export prices for soy oil from the United States Gulf of Mexico (red), soy oil from Brazil's Paranagua (black), soy oil from Argentina, up river (blue), palm oil from Malaysia (green), and palm oil from Indonesia (purple): 2018–2019 (a). Source: USDA FAS (2020) Palm oil price relative to soybean oil price. Shown is the price ratio of palm oil to soybean oil (black) and the 1960–2022 average (red dashed): 2000–2020 (b). Source: World Bank (2023) (Creative Commons license, https://creativecommons.org/licenses/by/4.0 /).

One key factor that influences the degree to which an increasing demand for qualifying feedstocks, such as soybean oil and canola oil, increases demand for other non-qualifying vegetable oils such as palm oil is the degree to which these vegetable oils are substitutable. For example, if demand for soybean oil for biofuel production in the United States increases, it is unclear to what degree other markets such as food production continue to demand soybean oil versus seek out other types of vegetable oil. The relative prices of soybean oil, palm oil, and other vegetable oils have generally held constant over the long term, which provides further evidence that these vegetable oils are generally substitutable in the long term. Conversely, soybean oil and palm oil are not substitutable for biofuel production in the United States (and many other parts of the world), since biofuels produced from palm oil are not eligible to generate RINs under the RFS Program unless exempted. Food manufacturers have also indicated that they can change their recipes and formulations to use alternative sources of vegetable oil in the long term, but not in the short term. The substitutability of vegetable oils in the global market, and the impact of the substitutability of vegetable oils on the indirect impacts of biofuel incentives and mandates such as the RFS, is an area where further research is needed.

Thus, overall, it can be concluded that the RFS Program from 2010 to current (i.e., the RFS2) had a direct effect on increasing domestic biodiesel and renewable diesel production, and on increasing the importation of these biofuels from foreign countries. As of writing, the magnitude of that effect historically cannot be confidently estimated, as many of the models and methods used to examine ethanol have not been applied to biodiesel, and many do not have sufficient market-detail (e.g., inclusion of BTC, oil prices, vegetable oil markets) to examine biodiesel confidently. In addition, most agro-economic models that could be used to predict changes in soybean acreage do not incorporate regionally specific agronomic differences such as double cropping and irrigation practices. These regional differences will have implications when attributing crop expansion to the RFS Program and other drivers.

However, as a starting point, the potential maximum impact of the RFS Program on the domestic production of biodiesel and renewable diesel may be estimated by comparing the total volume of these fuels produced domestically to the volume of biodiesel and renewable diesel required to be used by state mandates (i.e., Minnesota, New Mexico, Oregon, Pennsylvania, and Washington) and the volume of these fuels used in states with clean fuels programs or other significant incentives (i.e., California, Illinois, Washington, and Oregon). The estimate of the potential maximum impact of the RFS program presented here assumes that the volume of biodiesel and renewable diesel required under these programs would be used in the absence of the RFS Program (see Appendix E). With the exception of states with biofuel mandates, this estimate is a simplification as the volume of biodiesel and renewable diesel that would be used in states with clean fuels programs or other significant incentives in the absence of the RFS program has not been thoroughly assessed. In the absence of the RFS program it is possible that the mix of renewable fuels used to satisfy the state clean fuels programs would have been different, or that the incentives offered by these state programs may have been insufficient to incentivize the observed consumption of biodiesel and renewable diesel in these states. Nevertheless, this methodology is a reasonable way to estimates the maximum impact of the RFS program in the absence of more precise data.

Based on an assessment of state programs it is clear that the RFS Program incentivized biodiesel and renewable diesel production that exceeded state mandates, but that significant volumes of these fuels would likely have been consumed even in the absence of the RFS Program (Figure 7.13). However, this analysis suggests that the RFS Program and the BTC together accounted for roughly 70–100% of the biodiesel and renewable diesel used in the United States over the entire period assessed. The lower end of the range corresponds to assumptions of primacy to state programs, the upper end of the range corresponds to assumptions of primary from the RFS and the BTC.[40] These estimates do not consider the degree to which other factors discussed in this section (macroeconomic factors, BTC, or trade policy) may have resulted in the production of biodiesel and renewable diesel above the volumes required by the state mandates. As such, these estimates are best understood as the potential impact of all the other factors including the RFS Program on the production of these fuels, with data suggesting that the RFS Program

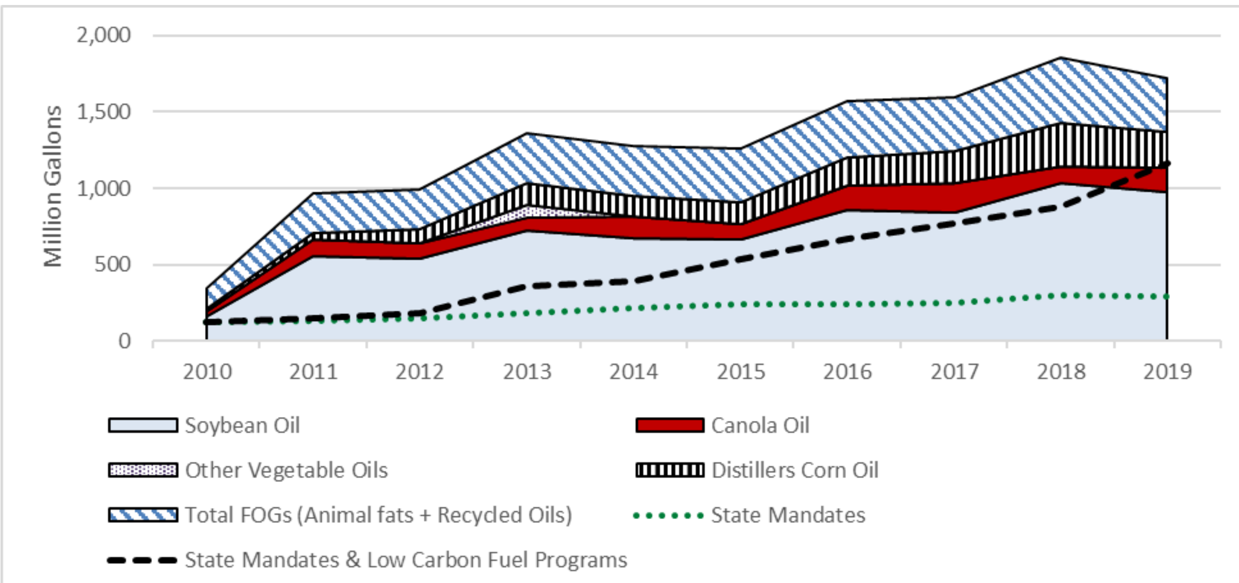

Figure 7.13. Domestic biomass-based diesel (BBD) production volumes compared with state consumption programs. Shown are production domestically from FOG (blue, diagonal lines), distillers corn oil (black, vertical lines), soybeans (light blue, solid), canola oil (dark red, solid), and other/unknown (purple, dots) compared to state-mandated BBD levels (green, dotted line) and state mandates + state low carbon fuels programs (black, dashed line) levels (see Appendix E for more detail). The difference between the black line and the stacked areas is the maximum potential effect of all other factors including the RFS Program.[41]

has had an effect every year since 2010. Further analysis to quantify the impact of programs other than the state mandates on domestic biodiesel and renewable diesel production is needed to refine these

[40] The maximum volume of biodiesel and renewable diesel attributable to the RFS Program from 2010 to 2019 averaged approximately 1,193 million gallons (70%), with a low of 136 million gallons in 2010 (52%) and a high of 1,902 million gallons in 2016 (74%). In total over the years assessed (Appendix E), of the 17.1 billion gallons of biodiesel and renewable diesel used, 5.2 billion gallons (30%) may be attributable to state mandates and state low carbon fuel programs. The remainder may be due to other factors, likely the RFS Program and the BTC primarily.

[41] Although information on total biodiesel dates back to 2001 from EIA (see Figure 7.1), data separated by feedstock does not begin until 2007 from the EIA Monthly Biodiesel Reports. The authors are not aware of any data on biodiesel and renewable diesel production by feedstock prior to 2007. Data before 2011 are from EIA; data from 2011 to 2019 are from EPA EMTS reported data.

estimates. Because at this time developing a robust quantitative estimate is not possible with current information, these estimates are not propagated forward to the environmental and resource conservation effect chapters in Part 3. Future work will aim to fill this knowledge gap.

7.3.6 Limitations of the Assessment

This chapter represents a significant first step in better understanding the role of the RFS Program in driving the increase of biodiesel and renewable diesel in the United States, though it is not without limitations. As discussed above, there are far fewer peer-reviewed studies on biodiesel than there are on ethanol, and almost none include FOGs, the BTC, and potential substitution effects in vegetable oil markets, all of which are likely important for understanding this industry. There is also very little information in the early years of the program (e.g., RINs for 2006–2009) though the relatively small volumes of biodiesel produced and consumed, and the lack of a biodiesel standard, suggest understanding this early period may be less critical to support current decision making.

The important role of the BTC in driving biodiesel and renewable diesel was discussed above. Even though analysis of the timing of production changes compared to the state of the BTC indicates that the BTC was a dominant factor, it is difficult to weigh the exact impact the tax credit had on production decisions in biodiesel and renewable diesel production facilities. If, for example, biodiesel facilities made their year-to-year decisions based on longer-term factors, such as macroeconomics, trade, or others, or if they anticipated the extension of the BTC, then the BTC should receive less weight. It is also unclear how the on-and-off-again nature of the BTC was perceived by biodiesel producers. Research suggests that in general, it is policy uncertainty (i.e., long-term government views towards alternative energy) rather than transient changes within an existing funding source, such as the BTC, that discourages investments in production capacity (Liu et al., 2018).

Another area of uncertainty in the above analysis is the relative importance of state mandates and incentives in determining total national biodiesel and renewable diesel production and consumption. While the state programs created significant demand, it is unclear whether these programs would have existed had there been no RFS Program. It is possible that individual states were encouraged to enact their own biodiesel programs only after the RFS Program mandates were announced. Many of these state policies came after the Energy Policy Act (2005), the RFS1 (2006), and EISA (2007). If state mandate and incentive programs were inspired by earlier national programs, then the analysis could be underestimating the impact of the RFS Program.

An important dynamic to examine in future research efforts is the fungibility of different vegetable oils and how the non-biofuel industries may respond if the biofuel industry shifts to using more soybean or FOG feedstocks. If demand for soy biodiesel increases, the interconnected industries that use soybeans for oil and meal can shift away from using soybeans as their primary feedstock.[42] These shifts lead to indirect effects in how other industries utilize soybeans and can ultimately affect the economics of biodiesel. Partly for this reason, future projections of biodiesel production anticipate small changes to soy biodiesel volumes (Chapter 2, Figure 2.2). However, if the industry shifts towards using more soybean oil in renewable diesel production, then the indirect effects associated with alternate feedstocks, such as distillers corn oil and FOG, could be different.

In parallel with investigating how different feedstocks may be utilized in the future, it is important to consider potential changes to land cover and land management that would result from shifting crop demand. As discussed in Chapter 5, changing land cover and land management is a primary avenue by which environmental effects occur. No quantitative estimates on RFS Program effects on the land were pursued since quantitative estimates on biodiesel were not concluded. In addition to fewer simulation modeling studies, all of the empirical studies of land use change around biorefineries to this chapter's authors' knowledge have examined ethanol biorefineries, thus similar work focused on soybean biorefineries is needed. Furthermore, in contrast with ethanol, the crushing step is not physically part of the biorefinery for many biodiesel facilities and mostly occurs at separate crushing facilities. Public spatial information on where these crushing facilities are located and how they are connected with the farm and biorefinery networks (e.g., train, truck) are needed to parameterize models to examine the soybean markets in greater detail.

This report made significant progress setting up a qualitative framework for attributing combined biodiesel and renewable diesel production generally to the RFS Program. However, further research should be pursued to narrow down more quantitative estimates of how much soybean and renewable diesel are directly attributable to the RFS Program.

7.4 Likely Future Effects of the RFS Program

As discussed in Chapter 6, the likely future effects of the RFS Program on biofuel production, consumption, and several economic and environmental factors have been estimated by EPA in the Final Set Rule that covers volumes for 2023–2025. The volumes in the Set Rule are based on a review of the implementation of the RFS Program to date and EPA's analysis of six groups of factors specified in EISA. The biofuels estimated to be consumed in each year in the Final Set Rule are presented in Chapter 2 (Table 2.4). The quantities of various types of renewable fuel projected to be attributable to the RFS Program in 2023–2025 in the RFS Final Set Rule are summarized in Table 6.12. For soybean oil biodiesel and renewable diesel EPA estimates, an additional 1.4–1.5 billion gallons are attributable to the RFS

[42] Note that canola-based biodiesel is also likely to increase along with soybean biodiesel to meet the RFS mandates.

Program for 2023–2025. As discussed further in Chapter 6.5, across all crop-based biofuels, the RFS Set Rule could potentially lead to an increase of as much as 2.65 million acres of cropland by 2025, relative to a scenario without the RFS renewable fuel volume requirements in place. The majority of this increase is estimated to be due to increased demand for soybean biodiesel and renewable diesel (1.57–1.93 million acres, Table 6.13). As noted in the Set Rule, these projections of increased cropland are uncertain, and the estimates presented likely over-estimate the impact of the RFS Set Rule on cropland due to various reasons discussed in the docket. [43]

While the likely future impact of the RFS Program on biodiesel production is uncertain, as with corn ethanol, factors that are likely to increase or decrease the effect of the RFS Program can be identified. For example, lower crude oil prices, lower diesel consumption, and higher RFS volume requirements are likely to result in higher impacts attributable to the RFS Program in future years, while higher oil prices, higher gasoline consumption, and lower RFS volume requirements are likely to result in lower impacts attributable to the RFS Program. The increasing stringency of the LCFS and emergence of other state-level biofuel standards may also reduce the relative impact of the RFS Program on biodiesel and renewable diesel production.

7.5 Chapter Synthesis

7.5.1 Specific Conclusions

- Some of the same factors that drove ethanol trends in production and consumption in the United States contributed to biodiesel and renewable diesel trends, including federal tax credits, state incentives, and high petroleum prices and low agricultural commodity prices, especially in the early period of growth.
- There is much less information on biodiesel and renewable diesel compared with ethanol, and very few retrospective analyses on the relationship between the RFS Program and biodiesel and renewable diesel production. Therefore, this chapter does not provide a quantitative estimate of the amount of biodiesel and land attributable to the RFS Program in the RtC3 as was done in Chapter 6 for corn ethanol.

[43] On July 26, 2022, the United States District Court for the District of Columbia filed a consent decree, which after stipulations from the parties, required EPA to sign a notice of proposed rulemaking (NPRM) to establish 2023 volumes for the RFS Program on or before November 30, 2022, and to sign a notice of final rulemaking to finalize the same by June 21, 2023. Order, *Growth Energy v. Regan et al.*, No. 1:22-cv-01191, ECF No. 12. EPA's proposed RFS volumes for 2023-2025, 87 FR 80582 (proposed and signed on Nov. 30, 2022, and published in the Federal Register on Dec. 30, 2022) available in Docket No. EPA-HQ-OAR-2021-0427 at https://www.regulations.gov, were finalized on June 21, 2023, and published in the Federal Register on July 12, 2023 (88 FR 44468).

- The evidence available suggests that the RFS Program was binding on biodiesel and renewable diesel for the entire period of the RFS2 assessed (2010 to 2019). It does not appear that there was a binding effect prior to this given the lack of an individual biomass-based diesel (BBD) standard from 2006 to 2009 under the RFS1 (2006–2008) or for the first year of the RFS2 (2009), and low RIN prices during these years where data are available (2008–2009).
- Overall, biodiesel and renewable diesel production has been much more strongly dependent on federal and state policies (grants, tax incentives, income tax credits, RIN values, etc.) than has ethanol. The Biodiesel Tax Credit (BTC) and the RFS2 played particularly important roles. A different set of incentives drove production in the early phases compared to more recent years.
- It is not possible in the RtC3 to derive a robust estimate of the volume of soybean biodiesel specifically attributable to the RFS Program. However, economic models suggest that the RFS Program could increase biomass-based diesel consumption 0.6–1.1 billion gallons for every billion gallon increase in the biomass-based diesel volume obligations; and comparison of state and federal mandates suggest that while roughly 0–30% of biodiesel consumption may be due to state programs (mandates and low carbon programs like the Low Carbon Fuel Standard), the remaining 70–100% may be attributable to a combination of other factors, primarily the RFS Program and the BTC. The effects of the RFS Program cannot be isolated at this time for the historical period because most studies do not separate the RFS from other important factors that occurred at the same time such as the BTC and state programs.
- In addition to domestic effects, the RFS Program incentivized the import of foreign biodiesel from different sources in different years (e.g., Argentinian soybean biodiesel, Southeast Asian palm oil). These direct volumes are small on a relative basis but could have important local effects overseas, and diversion of any vegetable oil toward biofuels could have indirect effects on these markets that are difficult to estimate.
- While this and other chapters have discussed the substitutability of different feedstocks into the food, feed, and fuel industries, the authors of this chapter are not aware of sufficiently rigorous studies that have addressed the impact of increasing demand for qualifying feedstocks (such as fats/oils/greases [FOGs] or soybean oil) for biodiesel and renewable diesel production on commodities that may be used as substitutes in other industries (such as other vegetable oils, including palm oil).
- The likely future effect of the RFS Program was estimated by EPA in the Final Set Rule issued on June 21, 2023, which estimated 1,484 million gallons of soybean biodiesel and

renewable diesel consumption in 2025 to be due to the RFS Program. Initial estimates from slightly higher volumes suggest that the RFS Program could potentially lead to an increase of as much as 1.9 million additional acres of cropland. These estimates are highly uncertain due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes.

7.5.2 Uncertainties and Limitations

- There are not many retrospective analyses on the relationship between the RFS Program and biodiesel production and imports. Therefore, estimates of biodiesel production and import volumes resultant from the RFS Program have substantial uncertainty.
- There are several limitations in the current literature, including the incorporation of the impact of the federal tax credit and state incentives (e.g., LCFS), interactions with oil price, influence of the RFS in establishing market certainty for biodiesel (in addition to corn ethanol), spatial information on crushing facilities, and possible substitution effects in vegetable oil markets. Therefore, the impact of these programs on biodiesel and renewable diesel production has not been quantified in the RtC3.
- More importantly for this report series, there is a shortage of studies that examine the potential effect of the RFS Program on changes to soybean acreage. Soybeans are grown for a variety of markets as well as for soil fertility and pesticidal reasons as a rotational crop. Furthermore, the reasons behind soybean planting decisions are different depending on the region. Hence, conclusions about potential mechanisms connecting soybean-related land cover and land management changes to the RFS Program are not made in the RtC3.
- While this and other chapters have discussed the substitutability of different feedstocks into the food, feed, and fuel industries, the authors are not aware of sufficiently rigorous studies on the fungibility of some of these feedstocks. For example, clean fuel programs could be successfully shifting their feedstock sources from environmentally damaging imported palm oil to domestically produced FOG. However, if other sectors are simultaneously using more palm oil, then the sustainability goals of the RFS Program may be partially negated.

7.5.3 Recommendations

- Agro-economic modeling and other quantitative analyses that investigate mechanisms between the RFS Program and the biodiesel industry are needed to determine the extent to which the RFS Program impacted biodiesel production and imports. To properly weigh the impact of the RFS Program relative to other drivers, and better assess different types of biofuel policies, more model studies that apportion biodiesel volumes between the RFS

Program, the LCFS, the BTC, and other drivers are required. Also critical, future studies should give estimates of soybean acreage so that the Section 204 objective of estimating environmental impacts of the RFS Program can be sufficiently assessed. Ideally, spatial models should have region-level detail because agronomic practices such as double cropping, irrigation, and other biofuel crop-related factors vary by region, and thus their resulting environmental impacts will vary as well.

- Public information on crushing facilities should be collected as with biodiesel and renewable diesel biorefineries so that models and tools can be developed to assess the biodiesel market more thoroughly.
- Improved data collection for FOG supplies, including from used cooking oil sources, which are currently not thoroughly surveyed, would help form a more complete picture of this increasingly important source of renewable diesel. Creating Harmonized System (HS) codes[44] and tracking international FOG trade would further enhance understanding of the role FOGs play in the biodiesel industry and any potential connection to the RFS Program.
- Considering the uncertainties and limitations listed above, more research on the fundamentals of feedstock substitution toward different domestic and international markets are needed.
- FOG feedstocks have played an increasingly important role in the U.S. biodiesel industry. As such, a more thorough examination of potential environmental effects, both positive and negative, and direct and indirect, of the FOG industry is recommended.

[44] The World Customs Organization manages the HS, which supports thorough classification and record keeping for imports and exports. An HS code for FOGs would help categorize this trade product for more thorough tracking of this commodity.

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Part 3

8. Air Quality

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Key Findings

- There is no new evidence that contradicts the fundamental conclusions of previous biofuels Reports to Congress. Those conclusions emphasized that emissions of nitrogen oxides (NO_x), sulfur oxides (SO_x), carbon monoxide (CO), volatile organic compounds (VOCs), ammonia (NH_3), and particulate matter ($PM_{2.5}$) can be impacted at each stage of biofuel production, distribution, and usage.
- Increased corn production results in higher agricultural dust and NH_3 emissions from fertilizer use. Improved nitrogen management practices can decrease these NH_3 emissions, however. Increased corn ethanol production and combustion leads to increased NO_x, VOCs, $PM_{2.5}$, and CO. As the increased ethanol volumes are displacing petroleum and its related emissions in each of these areas, the overall impact on the environment is a complex issue.
- Emissions from production of biodiesel from soybean oil vary depending on the oil extraction method, with mechanical expelling the least efficient with the highest emissions of NO_x, VOCs, CO, and $PM_{2.5}$, followed by hexane extraction and then enzyme-assisted aqueous extraction process (EAEP).
- EPA's "anti-backsliding" study (U.S. EPA, 2020b) examined the impacts on air quality from end-use changes in vehicle and engine emissions resulting from required renewable fuel volumes under the Renewable Fuel Standard (RFS). Compared to the 2016 "pre-RFS" scenario, a 2016 "with-RFS" scenario increased concentrations of ozone (eight-hour maximum average) across the eastern United States and in some areas in the western United States, $PM_{2.5}$ concentrations were relatively unchanged in most areas, while NO_2 concentrations increased in many areas and CO decreased. Furthermore, increases in formaldehyde and acetaldehyde were widespread, while benzene and 1,3-butadiene levels went down. Other recent research addressing air quality impacts of biofuels is limited.
- Using the GREET model (Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation), lifecycle emissions from corn ethanol are generally higher than from gasoline for VOCs, SO_x, $PM_{2.5}$, PM_{10}, and NO_x. However, the location of emissions from biofuel production tends to be in more rural areas where there are fewer people. How this translates to effects on human health is complex, as air pollutants can be transported and transformed downwind, and it depends not only on the number of people, but on their demographics and vulnerability, as well as the dose-response relationship, which is pollutant-specific, among other factors.

- On a per unit energy basis over the period analyzed, biofuels manufacturing has a larger impact than their petroleum counterparts on smog formation, acidification, $PM_{2.5}$ exposure, and ozone depletion potentials, but a smaller potential effect in the total U.S. context due to the smaller size of the biofuels industry. Nonetheless, this conclusion needs to be interpreted in the context of each industry: while petroleum refining is a highly optimized, mature industry, biofuels are still reaching maturity as indicated in their emission profile over the 2002–2017 period. The observed trends seem to indicate that, on a per unit energy basis, the biofuel industry is consistently reducing emissions as it matures.
- The likely future effects of the RFS Program on air quality are highly uncertain; there will be an overall increase in biofuel volume but the volumes of various types of biofuels produced and used will be different compared with past RFS volumes.

Chapter Terms: air quality, anti-backsliding study, CO, criteria air pollutants, fats, oils, and greases (FOGs), gasoline direct injection (GDI), National Ambient Air Quality Standards (NAAQS), NH_3, NO_x, ozone, $PM_{2.5}$, PM_{10}, port fuel injection (PFI), SO_x, VOCs

8.1 Overview

8.1.1 Background

EISA Section 204 requires that the EPA assess the impacts to date and likely future effects from the RFS Program on air quality. Air quality in the United States has seen dramatic improvements in the past 30 years since the passage of the Clean Air Act Amendments of 1990. Nevertheless, some areas still experience poor air quality for part or all of the year. This chapter focuses on outdoor air quality, which is more affected by biofuels in contrast with indoor air quality, and will focus primarily, though not exclusively, on the criteria air pollutants and major precursors, including SO_x, NO_x, VOCs, CO, ozone, PM_{10}, and $PM_{2.5}$.[1]

8.1.2 Drivers of Change

Air quality, as measured by the concentration of air pollutants in the ambient atmosphere, can be directly affected by increased production and use of biofuels through changes in emissions of air pollutants during (1) feedstock production; (2) conversion of feedstocks to biofuels; (3) transport of biofuels and feedstocks; and (4) combustion of biofuels in vehicles. Direct impacts on emissions occur due to changes in biofuel volumes produced and consumed, as well as changes in technologies and practices in each of the previous four processes. Indirect impacts on emissions occur through price-induced impacts associated with increased production and use of biofuels, which result in changes in

[1] As explained in Chapter 2, greenhouse gases are not a part of this report series, but see Chapter 2, Box 2.2.

petroleum fuel consumption and changes in agricultural production and land use; petroleum production displacement from increased use of biofuels; and changes in fuel properties due to the addition of biofuels to petroleum fuels. All of these drivers interact to influence air quality, which will be discussed below.

8.1.3 Relationship with Other Chapters

Air quality also affects many of the other chapters discussed in this report. This occurs through the exposure of nearby communities and natural habitats to air pollutants, and by the potential transport of these and other air pollutants downwind where they may affect other communities and natural habitats either through direct exposure or by atmospheric deposition. These effects are discussed elsewhere in the chapters where they occur (i.e., Chapters 12, 13, and 14).

8.1.4 Roadmap for the Chapter

This chapter first summarizes conclusions on air quality from previous Reports to Congress [i.e.,U.S. EPA (2018, 2011), section 8.2]. The chapter reviews the impacts to date for the primary biofuels, drawing upon published literature and analyses conducted since RtC2 in 2018 (section 8.3). The chapter then summarizes likely future impacts (section 8.4), provides a comparison of potential air quality effects from biofuels and fossil fuels (section 8.5), and ends with a short discussion of other biofuels (section 8.6) and synthesis of the information (section 8.7).

8.2 Conclusions from the 2018 Report to Congress

The second Report to Congress (U.S. EPA (2018) concluded that:

- There was no new evidence that contradicted the conclusions of the 2011 Report concerning air quality. Those conclusions emphasized that lifecycle emissions of NO_x (i.e., the sum of NO and NO_2), SO_x, CO, VOCs, NH_3, and $PM_{2.5}$ can be impacted at each stage of biofuel production, distribution, and usage. These impacts depend on feedstock type, land use change, and land management/cultivation practices and are therefore highly localized. The impacts associated with feedstock and fuel production and distribution are important to consider when evaluating the air quality impacts of biofuel production and use, along with those associated with fuel usage.
- Ethanol from corn grain has higher emissions[2] across the lifecycle than ethanol from other feedstocks.
- Ethanol plants relying on coal have higher air pollutant emissions than plants relying on natural gas and other energy sources.

[2] The focus in the RtC2 and the RtC3 are on emissions of criteria air pollutants.

- The magnitude, timing, and location of all these emissions changes can have complex effects on the atmospheric concentrations of criteria pollutants (e.g., ozone, $PM_{2.5}$) and air toxics, the deposition of these compounds, and subsequent impacts on human and ecosystem health.
- Ethanol increased NO_x emissions from light-duty vehicles certified to Federal Tier 2 Standards, likely occurring during times when the vehicle catalyst is not yet warmed up or air/fuel ratio is not tightly controlled. However, only limited data exist on the impacts of biofuels on the tailpipe and evaporative emissions of light-duty Tier 3 vehicles and light-duty vehicles using advanced gasoline engine technologies to meet greenhouse gas (GHG) emissions standards.[3]
- With the introduction of PM and NO_x catalytic exhaust aftertreatment systems in diesel applications, diesel engine vehicles equipped with exhaust catalysts (2007 and newer heavy-duty applications for PM; and model year 2010 and newer heavy-duty applications for NO_x) were not anticipated to have any significant impact on criteria pollutant emissions due to use of biodiesel fuel blends when compared to petroleum diesel fuel.

8.3 Impacts to Date for Primary Biofuels

The following section discusses implications of recent literature on the understanding of the drivers of air quality impacts of biofuels. It should be noted that most renewable fuel sold is ethanol, primarily produced from corn, and biodiesel, primarily produced from soybean but also other plant- and animal-based oils (see Chapters 2 and 3). There has been very little market penetration of fuels derived from cellulosic and other advanced feedstocks. As a result, research on biofuel impacts on air quality has focused on corn ethanol and soy biodiesel more than on biofuels from other feedstocks. The following discussion focuses on corn ethanol and soy biodiesel research, published since the RtC2. Impacts from fats, oil and grease (FOGs) are discussed in less detail. The limited research on cellulosic ethanol impacts are discussed in section 8.6. Ethanol from Brazilian sugarcane is not addressed since emissions from transport in the United States cannot currently be characterized. However, end-use impacts of ethanol from Brazilian sugarcane are no different than impacts from any other ethanol fuel (see Chapter 16 for more information).

8.3.1 Literature Review: Emission Impacts

The sections below give an overview of some of the key papers that have been published on biofuel emissions since 2018. Studies vary widely in terms of their utility for the purposes of this section (i.e., assessing the impacts for the four primary biofuels), as some include fuels (e.g., E25) or feedstocks

[3] It should be noted that unlike Tier 2 vehicles, Tier 3 vehicles are certified on an E10 test fuel.

(e.g., switchgrass) that are not widely used. Nonetheless, the literature provides a useful overview of the state-of-knowledge to date.

8.3.1.1 Corn Starch Ethanol

As of 2018, 5.6 billion bushels of corn were used for fuel ethanol, which is approximately 38% of the total corn produced in the United States (USDA (2019) and Chapter 3 of this report). A schematic of an idealized biofuel supply chain is shown in Figure 8.1. As discussed in Chapter 3, the supply chain broadly consists of five major components: (1) agricultural feedstock production and storage, (2) feedstock transport to the biorefinery (3) ethanol production at the biorefinery, (4) ethanol distribution, blending and storage, and (5) end use. The terms "upstream" and "downstream" are common in the literature, but they do not have a fixed definition (e.g., everything prior to the biorefinery is upstream). Instead, they are relative terms to a point of reference (e.g., upstream of a biorefinery or upstream of a blending terminal station). Because of this ambiguity, this term can mean many different things in different studies. Thus, for clarity, the steps above are used in this chapter, or the point of reference is listed when using the terms upstream and downstream.[4]

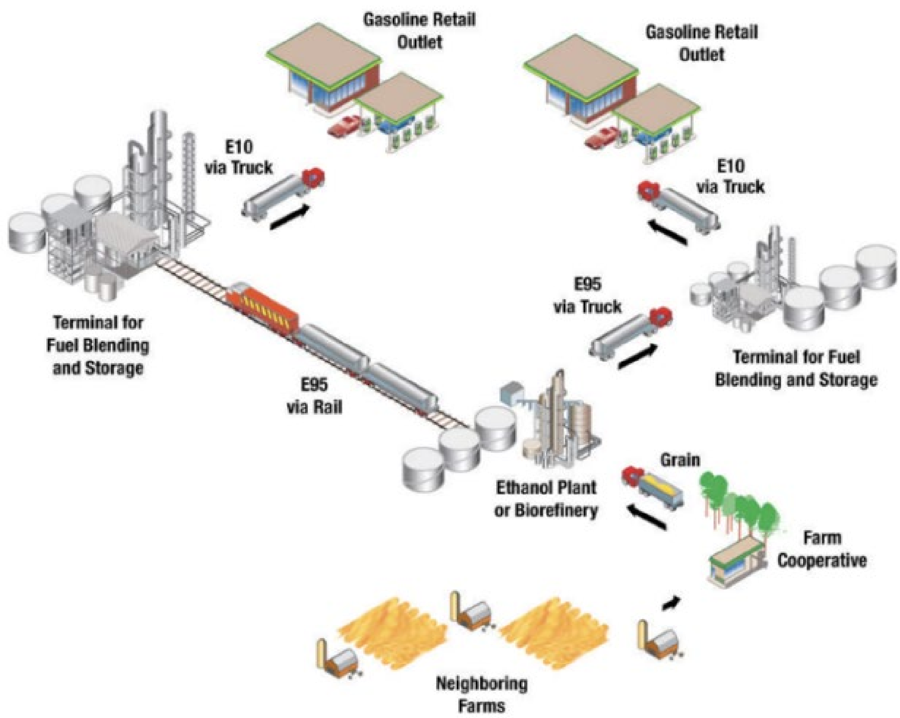


Figure 8.1. Ethanol supply chain components, showing rail and truck-based distribution. Source: National Bioenergy Center, National Renewable Energy Laboratory.

There is little recent literature that addresses cumulative impacts of processes upstream of vehicular emissions. In a literature review, Hoekman et al. (2018) summarized an analysis (Han et al.,

[4] Note that Chapter 3 splits agricultural production and storage from transport to the biorefinery. They are combined in this chapter since many studies on air emissions combine these two steps.

2015) using the Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) model of different ethanol-gasoline blends with corn as feedstock for fuel ethanol. The study found that emissions upstream of vehicular emissions for a 25% blend (i.e., E25) would increase 5%-40% depending on the pollutant, relative to an E10 blend.[5] Among various criteria pollutants, they found that the largest percentage increase in upstream emissions would be for CO and SO_2, with increases of 40% and 38.5%, respectively. In addition, upstream emissions of NO_x increase by 32.8%, and $PM_{2.5}$ increases by 29.2%.

8.3.1.1.1 Agricultural Feedstock Production, Storage, and Transport to the Biorefinery

Recently, Hill et al. (2019) modeled human health effects from air pollution ($PM_{2.5}$, in particular) caused by corn production in the United States. The authors considered health effects from maize produced for human and animal consumption, and for fuel ethanol. They concluded that reduced air quality resulting from maize production is associated with 4,300 premature deaths annually due to $PM_{2.5}$ in the United States. A follow-up study separated the emissions impact by end use of the maize produced, (Domingo et al., 2021). That work found that approximately 1,200 deaths per year are attributable to primary and secondary $PM_{2.5}$ which is associated with corn production for ethanol (see Figure 1 "product" column in (Domingo et al., 2021).

To conduct their study, Hill et al. (2019) developed a spatially explicit emission inventory of primary $PM_{2.5}$ emissions and precursors to $PM_{2.5}$ (including NH_3, SO_x, NO_x, and VOCs). They use a modified version of the GREET model, called GREET-Chemical, Spatial, and Temporal (GREET-CST). GREET-CST tracks emissions by linking processes with those in the EPA National Emissions Inventory (NEI). By running GREET-CST for the top 2000 maize-producing counties, they created an emissions inventory attributing the proportion of emissions of primary $PM_{2.5}$ and its precursors to maize production. This included on-farm emissions, as well as the supply chain emissions upstream of the farm. On-farm emissions included NH_3 emissions from the application of various types of synthetic fertilizers and manure, as well as fugitive dust from agricultural activities. For the 2,000 counties that were studied, 70% of the NH_3 emissions were from synthetic nitrogen fertilizer applications, while the remaining 30% were from manure application. Upstream emissions from production and transport of fertilizers were allocated to counties using the NEI emission factors and shapefiles.

Nitrogen management practices vary over time and region. Figure 8.2 depicts the nitrogen application rate per fertilized acre of corn in corn belt states versus other states for selected years. The amount applied has increased substantially across the United States since 2001, with the highest levels in

[5] While all gasoline engines can use E10, only flex fuel and light duty vehicles (with model year 2001 or later) are approved by the EPA to use E15. Flex fuel vehicles can use higher ethanol-gasoline blends, going up to E85 (E85 may contain 51-83% ethanol).

corn belt states including Illinois, Indiana, Iowa, Kansas, Michigan, Nebraska, North Dakota, Ohio, South Dakota, and Wisconsin.

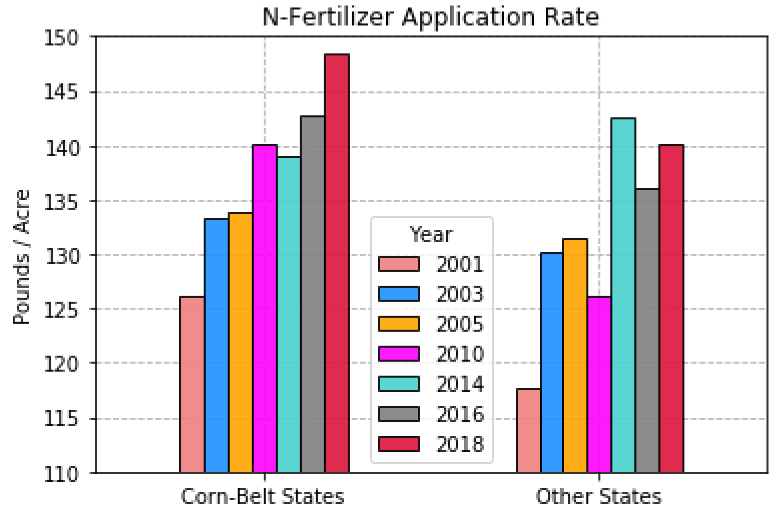


Figure 8.2. Nitrogen application rate per fertilized acre of corn for selected years. Corn belt states include Illinois, Indiana, Iowa, Kansas, Michigan, Nebraska, North Dakota, Ohio, South Dakota, and Wisconsin, as defined in EPA Ecoregion 6. (Source: Table 10 from the USDA ERS Fertilizer Use and Price data series, https://www.ers.usda.gov/data-products/fertilizer-use-and-price.aspx).

Recent studies have indicated that improved nitrogen management practices can increase nitrogen use efficiency (NUE) and therefore decrease ammonia emissions. Sela et al. (2018) compared a dynamic model-based nitrogen management approach with existing static approaches and found that the dynamic approach (a.k.a., "variable rate") substantially reduced the yield-scaled nitrogen losses compared to the static approach. The dynamic approach refers to a real-time fertilizer application recommendation based on a mechanistic model that allows continuous simulation of soil biogeochemical interactions. The dynamic approach results in a 32% reduction in nitrogen application rate without reducing crop yield, which corresponds to a yield-scaled nitrogen loss reduction of 11%. Other studies have also made similar recommendations to improve NUE to reduce environmental pollution from agriculture. For example, Zhang et al. (2015) recommended that NUE for maize should increase to 0.7 to reduce nitrogen loss. Such an increase in NUE can be achieved by use of several strategies—local cropping system, soil type, and weather-based fertilizer application, fertigation (applying fertilizer via irrigation water), slow-release fertilizers, and use of modern technologies for precision agriculture (Zhang et al., 2015). Most of these measures must be implemented at farm scale and can be incorporated in a dynamic nitrogen application approach.

8.3.1.1.2 Ethanol Production at Biorefineries

As of January 1, 2019, there were about 200 ethanol biorefineries in United States, with cumulative nameplate capacity[6] of 16.8 billion gallons per year (EIA, 2020). Figure 8.3 depicts the

[6] Nameplate capacity is the rated maximum output registered with administrative authorities.

location of ethanol and biodiesel refineries in the United States as of 2019. Most of this nameplate capacity (15.5 billion gallons) is in Petroleum Administration for Defense District (PADD) 2, which is the Midwest, where 178 ethanol plants are located. In this section, emissions of selected criteria pollutants or gaseous precursors to ozone and $PM_{2.5}$ from ethanol plants are reported.

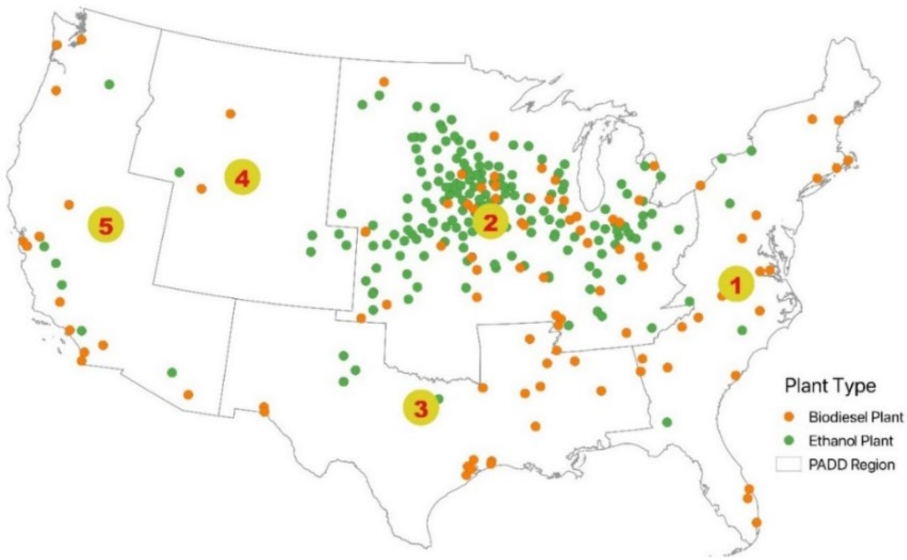


Figure 8.3. Location of biodiesel and corn ethanol plants in the contiguous United States in 2019 by Petroleum Administration for Defense Districts (PADDs). Source: EPA National Emissions Inventory (NEI).

The data presented here are based on analyses of data from EPA's 2016 modeling platform, version 1 (https://www.epa.gov/air-emissions-modeling/2016v1-platform). This platform has been used to support a number of programmatic assessments, including the Clean Air Act Section 211(v)(1) Anti-backsliding Study discussed below. Corn ethanol plants use corn grain as feedstock, which is processed using dry milling or wet milling process, with the former being the dominant process employed in the United States (see Chapter 3, section 3.4.1.1). Dry mill plants produced roughly 12.6 billion gallons of ethanol in 2016, whereas wet mill plants accounted for total production of approximately 1 billion gallons. Once the starch contained in the grain feedstock is broken into component sugars, it is fermented to produce ethanol. Ethanol produced from fermentation is further distilled and purified (see Chapter 3 for more details).

Table 8.1 summarizes emissions of criteria pollutants from biodiesel and corn ethanol plants in 2016. Only 10 of the ethanol plants used coal or coal in combination with other energy sources, although they contributed disproportionately to emissions, especially sulfur dioxide. Figure 8.4 depicts production volumes and ethanol refinery emissions by state. Emissions from corn ethanol plants are dominated by NO_x, VOCs, $PM_{2.5}$ and CO. Most VOCs at ethanol plants are emitted from fermentation scrubbers, with ethanol and acetaldehyde emitted at highest rates (Brady and Pratt, 2007). Moreover, using airborne measurements downwind of a large ethanol biorefinery in Illinois, de Gouw et al. (2015) concluded that emissions of VOCs, particularly those of ethanol, formaldehyde, and acetaldehyde, may be underestimated in the national emission inventory. However, this study focused on a large coal-powered plant that was not representative of the majority of facilities, which are powered by natural gas and thus have lower emissions.

Table 8.1. Pollutant emissions (short tons) from U.S. biodiesel and corn ethanol biorefineries in 2017. (Sources: EPA 2017 NEI and EPA 2016 v1 emissions modeling platform.(U.S. EPA, 2016a)).

Finished Fuel	Number of Facilities	CO	NH_3	NOx	PM_{10}	$PM_{2.5}$	SO_2	VOCs
Corn Ethanol (total)	176	7,362.8	278.7	9,045.5	5,218.7	4,088.5	1,854.4	8,908.7
Coal; Dry Mill	2	75.3	0	55.8	20.7	20.0	n.a.	39.7
Coal; Wet Mill	2	455.9	23.2	603.2	376.5	260.0	547.1	827.9
Natural Gas; Dry Mill	160	6,389.6	246.4	7,880.6	4,533.5	3,647.2	904.4	7,560.3
Natural Gas; Wet Mill	3	251.8	9.0	142.2	184.5	102.7	74.7	270.5
Unknown; Unknown	9	190.1	0.0	363.7	103.4	58.5	327.6	210.3
Biodiesel6	175	960.5	39.7	1,277.0	815.7	556.2	3,384.1	3,987.2
Total	351	8,323.2	318.4	10,322.5	6,034.4	4,644.6	5,238.5	12,895.9

[a] Separate data have not been generated for soy and FOG biodiesel.

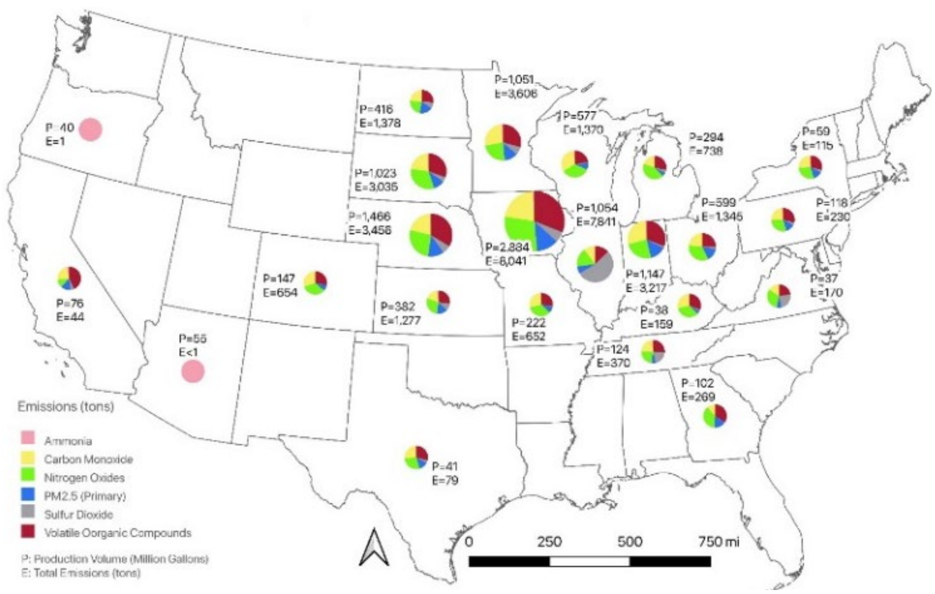


Figure 8.4. Emissions of various pollutants for corn ethanol refineries in the contiguous United States for year 2016 (similar patterns for 2017). Annotated numbers are the production volume (P, in million gallons) and total emissions (E, in tons) from all refineries in respective states. For facilities in AZ and OR, emissions of only ammonia were reported. Size of pie corresponds to the state's fraction of the total national production (not to scale; emissions from EPA 2016 modeling platform, v1; fuel volumes from EPA Moderated Transaction System, https://www.epa.gov/fuels-registration-reporting-and-compliance-help/public-data-renewable-fuel-standard).

8.3.1.1.3 Ethanol Distribution, Blending, and Storage

Once the ethanol is produced at biorefineries, it is transported to terminals for blending and storage (Figure 8.1). At the blending terminal, ethanol is blended with gasoline for various fuel combinations such as E10, E15, or E85. The blended fuel is then sent to retail gasoline outlets where it is sold to the customer. Primary modes of distributing ethanol to the blending terminal and the blended fuel to the retail outlets are rail, road, or barges. As discussed in Chapter 3, it is estimated that 70% of ethanol transportation occurs by rail, 20% by trucks, and the remaining 10% by barges (AAR, 2021; Denicoff, 2007). Rail- and truck-based ethanol distribution occurs in the Midwest region to most marketplaces (East Coast, California, Texas), whereas barges move ethanol around the Great Lakes region (e.g., serving Chicago, IL and Albany, NY terminals) and the Gulf Coast (e.g., serving New Orleans, LA and Houston, TX terminals) (Denicoff, 2007). Emissions during the distribution include both evaporative losses of VOCs during storage and transport, as well as combustion emissions from commercial marine vessels, rail, tanker trucks, and pipeline pumps.

Table 8.2. Emissions from transportation of ethanol by PADD region in tons. Source: EPA 2016 version 1 modeling platform (https://www.epa.gov/air-emissions-modeling/2016v1-platform).

PADD Region	CO	NH_3	NOx	PM_{10}	$PM_{2.5}$	SO_2	VOC
1	1,312	7	6,074	181	160	113	164,363
2	1,121	6	5,072	161	140	98	151,466
3	743	5	3,353	114	95	59	121,058
4	307	3	1,346	61	44	14	92,203
5	744	5	3,427	113	94	56	131,583
Total	4,225	26	19,270	630	533	340	660,674

While most of domestic corn ethanol is produced in the Midwest region of the country (PADD region 2), 74% of ethanol consumption occurs outside this region. East Coast states (PADD 1) consume 36% of total ethanol produced nationally, whereas PADD 3 and 5 account for 17% and 18% of consumption, respectively. Resulting emissions from transportation of ethanol to these demand regions for calendar year 2016, based on data from EPA's 2016 version 1 modeling platform (U.S. EPA, 2016a) are shown in Table 8.2. Emissions come from combustion and evaporation during transport by rail, commercial marine vessel, and truck, as well as storage and transport evaporative losses. The largest emission contribution is for VOC due to evaporation.

8.3.1.1.4 *Ethanol End Use*[7]

After distribution to the retail outlet stations, end use at the vehicle occurs. This step produces combustion emissions during vehicle operation, as well as evaporative losses from refueling, fuel tank venting, and permeation. Light-duty vehicle technology continues to evolve as new emissions standards for both criteria pollutant emissions and GHG emissions are phased in. Some of the standards relevant to recent changes in light-duty powertrain technology include:

1. Federal Tier 2 and Tier 3 light-duty vehicle emission standards regulating NOx, non-methane organic gases (NMOG), CO, $PM_{2.5}$, formaldehyde, fuel sulfur, and evaporative emissions (U.S. EPA, 2014, 2000).
2. The Model Year 2012–2016 and Model Year 2017–2025 (U.S. EPA & NHTSA, 2012, 2010). Federal Light-duty Corporate Average Fuel Economy (CAFE) and GHG Emission Standards regulating fuel economy, CO_2 emissions, methane emissions, and N_2O emissions; and setting standards related to the use of specific hydrofluorocarbons used within automotive air conditioning systems (U.S. EPA, 2018; U.S. EPA & NHTSA, 2012, 2010).
3. The Revised 2023 and Later Model Year Light-Duty Vehicle GHG Emission Standards set more stringent GHG emission standards for passenger cars and light trucks (U.S. EPA, 2021).[8]

Changes to engine technologies and both exhaust and evaporative emissions control systems in response to implementation of these regulations are likely to result in emissions that differ by ethanol blend level when compared to vehicles meeting previous standards running on the same blends. For example, in response to recent CAFE and GHG standards, light-duty vehicles with spark ignition engines have been transitioning from sequential port fuel injection (PFI) to gasoline direct injection (GDI), which impacts PM emissions levels and composition. For the 2018 model year, more than half of all light-duty vehicles used GDI (U.S. EPA, 2019). Many engines in light-duty vehicle applications are also transitioning to boosted induction systems using turbocharging to comply with CAFE and GHG emissions standards. For the 2018 model year, nearly one-third of all light-duty vehicles were turbocharged (U.S. EPA, 2019), and nearly all vehicles with turbocharged engines also used GDI due to synergies between GDI and turbocharging.

At the time of the preparation of the 2018 RtC2, the only comprehensive, multi-vehicle study of the impacts of fuel composition on the exhaust emissions of modern light-duty vehicles complying with

[7] Also see the EPA's "anti-backsliding study" in section 8.3.2.2 for effects from end use. Because that study examined effects from increases in ethanol and biodiesel combined (as opposed to this section on ethanol), and focused on air quality as opposed to emissions, it is discussed later.

[8] As a follow-on to this action, EPA plans to initiate a future rulemaking to establish multipollutant emission standards for model year 2027 and beyond.

Federal Tier 2 emissions standards was the EPA/DOE/CRC EPAct/V2/E-89 Phase 3 Study (U.S. EPA, 2013a, b).[9] This study assessed the effects of five gasoline properties, including ethanol blend level, on exhaust emissions from 15 light-duty vehicles selected to be representative of the Tier-2-compliant light-duty vehicle fleet. This study concluded that ethanol decreased CO emissions but increased NO_x emissions from these vehicles, likely occurring during times when the exhaust catalyst is not fully warmed up or air/fuel ratio is not tightly controlled. The study also assessed the impact of ethanol on several mobile source air toxics (MSATs), with the results being incorporated into the MOVES (MOtor Vehicle Emission Simulator) model (U.S. EPA, 2020a, 2016b). Those results indicated that increasing the ethanol blend level tends to increase emissions of ethanol, acetaldehyde, and formaldehyde, and reduce emissions of benzene and 1,3-butadiene.

The MOVES model also includes effects of ethanol on evaporative and permeation emissions. Evaporative emissions are generally higher for higher-vapor-pressure fuels, and since E10 blends are sold at a higher summertime vapor pressure than E0 in most areas of the country, the overall effect of widespread adoption of E10 has been higher evaporative emissions. Transition from E10 to E15 is not expected to cause further evaporative increases since E15 will be marketed at the same or lower vapor pressure relative to E10. Permeation refers to migration of fuel through tank walls, hoses, and other fuel system components via processes occurring at a molecular level (i.e., different from leaks). Studies done with vehicles meeting earlier emission standards have indicated that ethanol can increase fuel permeation rates. However, this effect has largely been mitigated through use of low-permeation materials in vehicles meeting the latest emission standards (U.S. EPA, 2020d).

No comprehensive, multi-vehicle studies or datasets comparable in scope to the EPAct Phase 3 Study on the impacts of multiple fuel properties on exhaust (or evaporative emissions) were found for vehicles certified to Federal Tier 3 or California LEV III emissions standards within the peer-reviewed literature or among test programs conducted by EPA. However, since the preparation and publication of the 2018 Report, two closely related multi-vehicle studies focused on PM emissions have been published:

- CRC E94-2: This study investigated the impact of match-blended gasoline composition on regulated gaseous exhaust emissions,[10] GHG emissions, PM emissions, and particle number (PN) emissions from a representative fleet of 12 light-duty vehicles equipped with GDI engines (Morgan et al., 2017). It did not include evaporative emissions measurements.
- CRC E94-3: This study used a smaller subset of four GDI vehicles from the CRC E94-2 study to determine the emission effect of adding ethanol to E0 fuels through splash blending,

[9] This study hereafter is called the "EPAct Phase 3 Study." CRC stands for Coordinating Research Council.

[10] This includes hydrocarbons, CO, and NO_x.

and compared those results to what was observed for match-blended E10 fuels from the E94-2 program (Morgan et al., 2018).

8.3.1.1.4.1 Summary of E94-2 Results

The E94-2 study found that changes in particulate matter index (PMI) and ethanol content of gasoline had the strongest impacts on $PM_{2.5}$ emissions. PMI is a predictive index that estimates the tendency of a gasoline formulation to produce particle emissions, based on the vapor pressure and structure of the specific compounds it contains (Aikawa et al., 2010). It indicates that compounds with lower volatility and those containing aromatic rings tend to more readily form soot during combustion. Increasing PMI from low (1.3) to high (2.5) was found to more than double PM emissions. The addition of 9.5% ethanol (E10) increased $PM_{2.5}$ emissions by 12% to 57% relative to the baseline E0 for three of the four fuel pairs with matched anti-knock index (AKI)[11] and PMI (Morgan et al., 2017). The fuel effects on $PM_{2.5}$ emissions from changes in PMI and ethanol were observed for the entire test fleet, including:

- Vehicles subdivided by use of naturally aspirated[12] engines or turbocharged engines, and
- Vehicles subdivided between low, medium, and high levels of PM emissions.

In general, the $PM_{2.5}$ emissions increases associated with high PMI fuels were larger than those observed with increased ethanol levels. However, it should be noted that the impacts of increased ethanol on PM from low $PM_{2.5}$-emitting vehicles were not large enough to be statistically significant in all cases.

Ethanol content up to 9.5% had no statistically significant impacts on either NO_x or total hydrocarbon emissions relative to an E0 fuel. Ethanol content at E10 was also found to decrease CO emissions in a subgroup of 4-cylinder, naturally aspirated vehicles, but not in turbocharged vehicles. The particulate matter and gaseous emissions results for fuel property changes from E94-2 are summarized in Table 8.3.

In summary, the CRC E94-2 study found that total PM increased with higher levels of ethanol (0% to 9.5%), CO decreased in naturally aspirated vehicles, and other emissions were relatively unaffected. However, PMI had a stronger effect than ethanol level.

[11] AKI is a measure of octane and is also known as DON and (R+M)/2.

[12] A naturally aspirated engine refers to internal combustion engine in which air intake depends solely on atmospheric pressure and does not have forced induction through some other means like a turbocharger or supercharger.

8.3.1.1.4.2 Summary of E94-3 Results

The E94-3 study found that the addition of ethanol to E0 fuels through splash blending[13] increased PM mass emissions and solid particle number emissions (SPN), with the impacts primarily observed during the cold-start phase (Phase 1) of the LA92 test cycle (Morgan et al., 2018). The PM results are summarized in Table 8.4.

[13] Splash blending refers to adding ethanol to a market gasoline without any intent to meet target values for other fuel properties in the resulting blend. Most E15 fuels in the current market are made by splashing additional ethanol into an E10 gasoline. Match blending is a scenario where there are specific property targets for the finished fuel (e.g., octane or vapor pressure), which result in specific formulation constraints for the hydrocarbon blendstock prior to ethanol blending. For example, once E10 blends became ubiquitous, refiners began to produce a lower-octane blendstock that meets minimum octane requirements only after ethanol addition.

Table 8.3. Summary of CRC E94-2 particulate matter emissions and composition results. Percentages are changes relative to the lower index in the row (i.e., PMI 1.3, 0% Ethanol, and AKI 87, used with permission). All emissions data were collected over the LA92 dynamometer driving cycle.

Fuel Property Change (Match-blended)	PM Constituents							
	PM*	Phase 1 PM†	SPN‡	EC**	Total HC	CO	NO_X	CO_2
PMI 1.3 to 2.5	+106–142% for all fuels Larger effect in 4-cyl naturally aspirated vehicles	+62–150% for all fuels Smaller effect in 4-cyl naturally aspirated vehicles	+73–117% for all fuels Larger effect in 4-cyl naturally aspirated vehicles	+114–173% for all fuels Similar effects in 4-cyl vehicles y air induction type	+21% (one subgroup, vehicle-specific)	No effect	No effect	No effect
Ethanol 0% to 9.5%	+18–46% for all fuels (except AKI 94 high PMI fuel)	+12–57% for all fuels (except AKI 94 high PMI fuel)	+14–39% for all fuels	+12–57% for all fuels (except AKI 94 high PMI fuel)	No effect	-14% (4-cylinder naturally aspirated vehicles)	No effect	+0.5–0.8%
AKI 87 to 94	No effect	No effect	No effect	No effect	-15% (4-cylinder naturally aspirated vehicles)	No effect	-27% (one subgroup, vehicle-specific)	No Effect

The ranges cited for the percentage changes caused by fuels refer to the lowest and highest percentage effects found for the test fleet overall or in any of the subgroups examined (by air induction type for 4-cylinder engines and by average PM level for all vehicles).
* PM: Particulate matter mass emissions determined gravimetrically
† Phase 1 PM: PM over the initial cold-start phase of the 3-phase LA92 chassis dynamometer test cycle.
‡ SPN: Solid particle number measured according to the particle measurement programme (PMP) protocol.
** EC: Elemental carbon via thermo-gravimetric analysis

Table 8.4. Summary of CRC E94-3 particulate matter emissions and composition results (used with permission). Table notes same as Table 8.3 unless noted.

Fuel Property Change (Splash-blended)	PM	Phase 1 PM	SPN	Phase 1 SPN
E0 to E10	+24% increase for all fuels on average and in the group of low PMI fuels. (p ≤0.01)	+13% increase for all fuels on average and in the group of low PMI fuels. (p ≤0.01)	+17% increase for all fuels on average and in the group of low PMI fuels. (p=0.05)	+12% increase for all fuels on average and in the group of low PMI fuels. (p=0.05)

* Phase 1 SPN: SPN over the initial cold-start phase of the 3-phase LA92 chassis dynamometer test cycle.

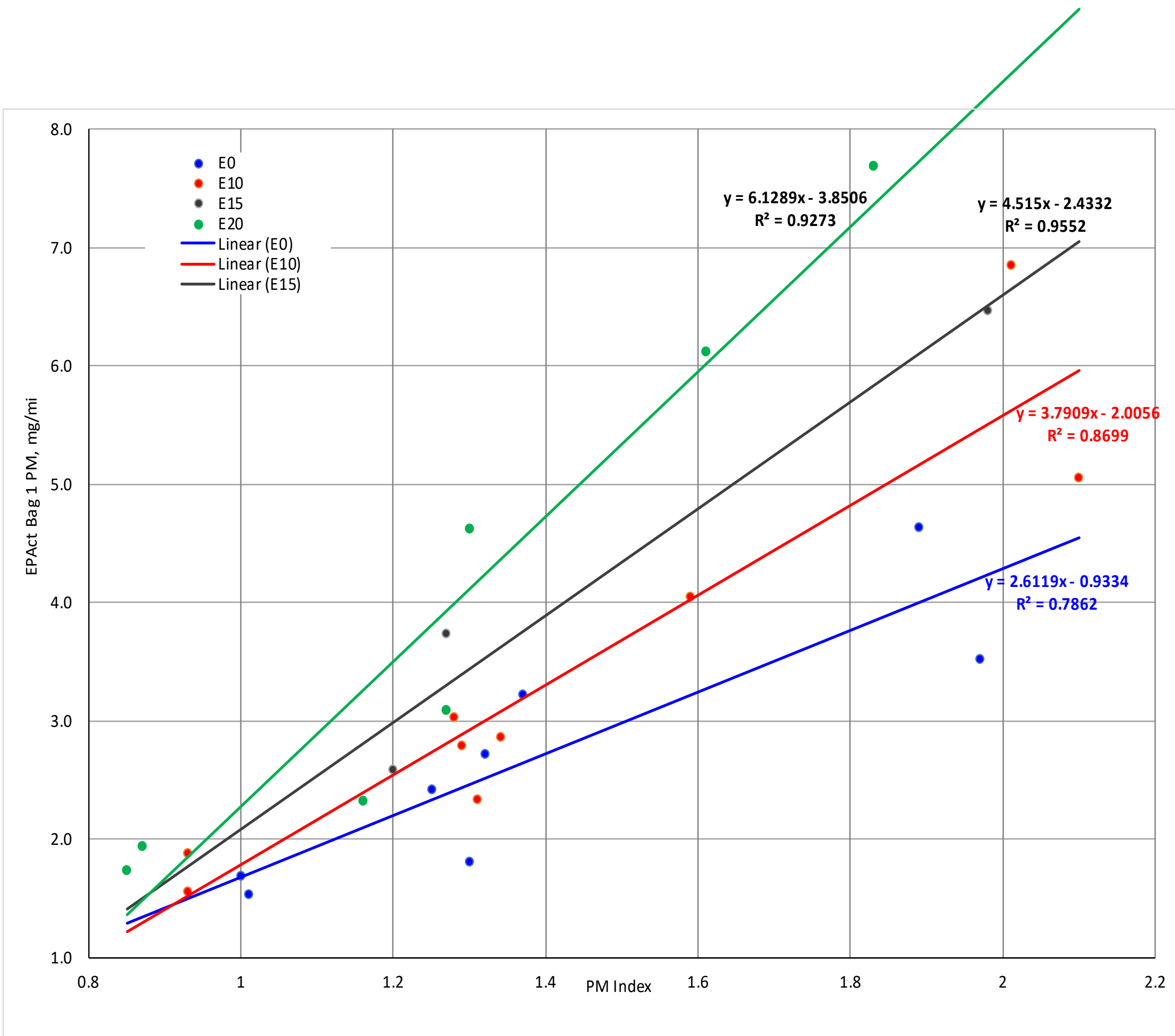
8.3.1.1.4.3 Recent Light-duty Vehicle Analyses

EPA has conducted additional analyses based upon the data sets from the EPAct Phase 3 Study to investigate the impact of ethanol content and other fuel properties on PM emissions and PMI (Butler et al., 2015; Sobotowski et al., 2015). The Butler et al. study clarified the relationship of PM emissions at E0, E10, E15, and E20 blend levels to other fuel properties expressed as PMI (Figure 8.5), which showed a trend of increased PM emissions for increasing ethanol blend levels from 0% to 20% for fuels at a given PMI over the cold-start "bag 1"[14] of the emissions inventory test cycle. In summary, recent research on GDI vehicles has not shown an impact on hydrocarbon and NO_x emissions with increasing ethanol levels. However, $PM_{2.5}$ is impacted by ethanol level and, to a greater extent, PMI.

EPA also recently published work that examined the emissions impact of replacing a small amount of high-boiling aromatics in market-representative E10 gasoline with alternative octane sources (U.S. EPA, 2023), including additional ethanol, which was represented by an E15 test fuel. The results indicate that the largest impact was a reduction in $PM_{2.5}$ emissions, and that no increase in emissions of NOx or NMOG was observed for the test fleet overall.

In addition, a 2022 study prepared for CARB, RFA, and USCAR compared emissions between vehicles using E10 and splash blended E15 (Karavalakis et al., 2022). The study tested emissions from CaRFG E10 and E15 on 20 different Tier 3 vehicles with four different technologies on an FTP cycle. The study found statistically significant decreases in $PM_{2.5}$, total hydrocarbons (THC), and CO emissions for vehicles running on E15 compared to E10, and directional reductions in NO_x emissions, although they were not statistically significant.

[14] Emissions are collected in sample bags. Bag 1 represents the cold start transient phase of the test cycle.



mg/mi = milligrams per mile

Figure 8.5. Data from the EPAct/V2/E-89 Phase 3 study showing the relationship between PM emissions and PM Index for different ethanol blend levels over Bag 1 of the LA92 test procedure. Data from Butler et al. (2015).

8.3.1.1.4.4 E85 Impacts

A detailed analysis of emission differences between E85 and E10 was integrated into MOVES based on the limited data available (U.S. EPA, 2020a, d, e) (U.S. EPA, 2020c, 2016b, c). No significant differences between E85 and E10 were found in emissions of THC, CO, NO_x, and $PM_{2.5}$. However, vehicles fueled with E85 had higher CH_4 emissions, and consequently, lower non-methane hydrocarbon (NMHC) emissions. These vehicles also had higher formaldehyde and acetaldehyde emissions, but lower benzene and 1,3-butadiene emissions (U.S. EPA, 2020a, d, e) (U.S. EPA, 2016b, c). E85 increases permeation emissions relative to E10, with higher emissions of ethanol and lower emissions of other hydrocarbons (Haskew et al., 2006).

8.3.1.2. Biodiesel from Soybean and FOGs

Unlike ethanol, which is predominantly sourced from one source in the United States, biodiesel is sourced from a variety of feedstocks (as discussed in Chapter 3 and Table 2.1). The supply chain for biodiesel thus varies as well. However, although there are many feedstocks currently used in the United States, only domestic soybean and domestic FOGs dominated the national pool from 2005–2020, which together made up nearly 70% of the biodiesel in 2019 (Table 2.1). Thus, this section focuses on domestic soybean and domestic FOGs, with an illustrative supply chain shown in Figure 8.6. Once the feedstock reaches the biorefinery gate, the supply chains for FOG- and soybean-based biodiesel are identical. Prior to that, soybean is an agricultural feedstock often grown in rotation with corn. FOGs are generally considered a waste product of some other activity like animal rendering, thus the emissions for FOGs are often associated with the primary product [but see Chapter 4 Box 4.1. Fats, Oils, and Greases (FOGs)]. Although much less has been published on the air quality effects from biodiesel relative to corn ethanol, the available literature is summarized below.

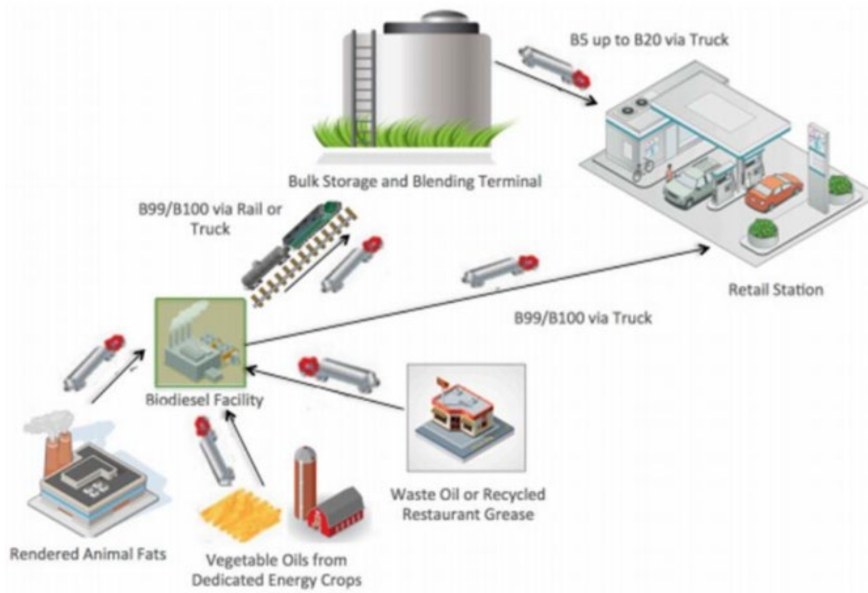


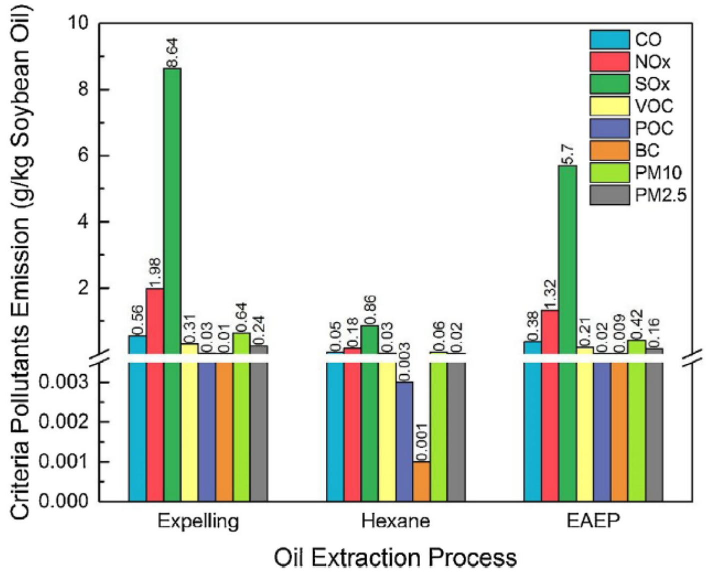
Figure 8.6. Biodiesel supply chain components. Source: Boutwell et al. (2014) (used with permission).[15]

8.3.1.2.1 Agricultural Feedstock Production, Storage, and Transport to the Crush/Biorefinery

Aside from the lifecycle analyses discussed in section 8.3.1.2.5, there are no studies to the authors' knowledge focused on the emission impacts of either soybean biodiesel or FOGs focused on the feedstock production (soybean) or collection (FOGs), storage, and transport stages. A recent analysis using GREET examined the lifecycle effects of three different biodiesels (i.e., soybean, canola, tallow) compared with conventional diesel (Chen et al., 2018). That study, however, was focused on GHGs and thus is out of scope for the RtC3.

[15] Omitted from Figure 8.6 is the soybean crushing facility, which serves as an important intermediary between the farm and the biorefinery for soybean-based biodiesel, receiving an estimated 51% of the soybean harvest. See Chapter 3 section 3.3.2 for more information.

8.3.1.2.2 Biodiesel Production: Crushing Facility and Biorefinery



g = grams; kg = kilograms

Figure 8.7. Select criteria pollutant and precursor emissions for soybean oil extraction processes (POC = precursor organic compounds). Source: Cheng et al. (2018) (used with permission).

As opposed to corn ethanol, where the physical/ chemical processing of the corn to obtain starch occurs at the biorefinery, the processing of soybean to separate the oil from the meal predominantly occurs at the crushing facility. Using Argonne National Laboratory's GREET model, Cheng et al. (2018) evaluated the emissions of soybean biodiesel at the crushing facility comparing three extraction phases. They reported that emissions from different extraction methods have varying effects on the emissions of different criteria pollutants (Figure 8.7). SO_x emissions are highest from mechanical expelling at 8.64 grams per kilogram (g/kg) soybean oil. In comparison, hexane extraction results in an order of magnitude lower emissions. EAEP SO_x emissions are 5.7g/kg soybean oil. Their results show similar trends for other criteria pollutants that were considered (NO_x, VOCs, CO, precursor organic compounds [POC], black carbon [BC], PM_{10}, and $PM_{2.5}$), with mechanical expelling resulting in most emissions. Hexane extraction was most energy efficient and had lowest emissions among the three processes typically used in the industry. EAEP emissions were about 34% lower than the mechanical expelling process.

Figure 8.8 depicts the locations of the roughly 175 biodiesel production facilities in the United States, and Table 8.1 provides the total nationwide emissions using the same 2016 EPA modeling platform presented in section 8.3.1.1.2. Biodiesel production emissions are in general dominated by VOCs, SO_2, and CO. Most VOC emissions from biodiesel facilities are in the form of hexane (a hazardous air pollutant) when vegetable oil is chemically extracted from oilseeds. Chemical extraction is usually more efficient than mechanical extraction, and generally utilized at large biodiesel facilities. Boilers providing steam and energy for process, process flares, and other onsite equipment are sources of SO_2, $PM_{2.5}$, and CO.

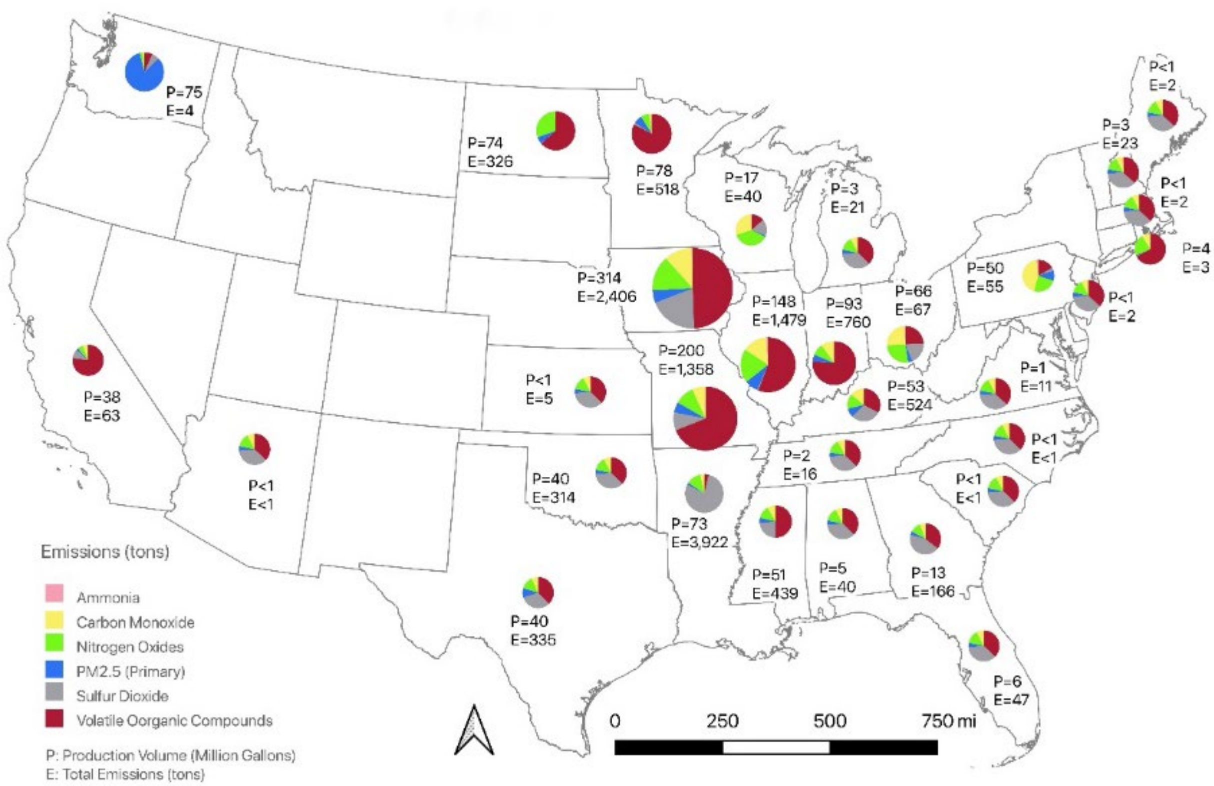


Figure 8.8. Emissions of various pollutants for biodiesel refineries for the contiguous United States, year 2016 (similar patterns for 2017). Annotated numbers are the production volume (P; in million gallons) and total emissions (E; in tons) from all refineries in respective states. Size of pie corresponds to the state's fraction of the total national production (not to scale; emissions from EPA 2016 modeling platform, v1; fuel volumes from EPA Moderated Transaction System, https://www.epa.gov/fuels-registration-reporting-and-compliance-help/public-data-renewable-fuel-standard).

8.3.1.2.3 Biodiesel Distribution and Storage

Table 8.5 provides transport emission estimates for B100 in 2016, from EPA's 2016 version 1 modeling platform (U.S. EPA, 2016a). Evaporative losses during storage and transport of biodiesel fuel are assumed to be negligible due to its low volatility.

Table 8.5. Emissions (tons/yr) from transportation of biodiesel by PADD region. Source: U.S. EPA (2016a)**.**

PADD Region	CO	NH_3	NO_x	PM_{10}	$PM_{2.5}$	SO_2	VOC
1	36	0.1	143	5	5	5	3
2	53	0.2	215	7	7	7	5
3	40	0.1	161	5	5	5	4
4	10	0.0	39	1	1	1	1
5	25	0.1	101	3	3	3	3
Total	164	1	661	22	21	21	16

8.3.1.2.4 Biodiesel End Use

Compression ignition engines using biodiesel or biodiesel blended with petroleum-based fuels must comply with U.S. federal heavy-duty engine, light-duty vehicle, nonroad engine, locomotive, and marine engine emissions standards. Heavy-duty engine and light-duty vehicle emissions standards rely primarily on catalytic exhaust aftertreatment systems (EAS) to reduce NO_x emissions by over 85% relative to non-EASs, using base-metal-exchanged zeolite selective catalytic reduction (SCR) with aqueous urea dosing. These systems also reduce PM emissions by over 95% relative to non-EAS systems using a combination of a diesel oxidation catalyst and a catalyzed diesel particulate filter (CDPF). Similar EASs are also used for compliance with Tier 4 emissions standards for most nonroad and marine diesel applications. As mentioned in the 2018 RtC, when taking into consideration the level of control available from modern heavy-duty diesel and other similar EASs used for emissions control in other applications, significant impact on criteria pollutant emissions is not anticipated from commonly used biodiesel blends (e.g., typically 5% and up to 20%).

Heavy-duty engine applications are anticipated to transition to dual/light-off SCR systems for NO_X control to comply with future NO_X emissions standards that are under development as part of the Cleaner Trucks Initiative (U.S. EPA, 2020b). Light-off SCR uses a second, close coupled zeolite SCR and urea dosing system immediately downstream of the engine's turbocharger, and thus may be more susceptible to chemical poisoning effects than SCR systems that are currently used for compliance with heavy-duty NO_x emissions standards and emissions standards for other diesel applications.

The primary concern with biodiesel, discussed in detail below, is the potential impact of metals in biodiesel blends on emission control system performance. Vegetable oil feedstock (e.g., from soybean or corn) prior to transesterification may contain high concentrations of sodium (Na), potassium (K), calcium (Ca), magnesium (Mg), and phosphorus (P), as well as aluminum (Al), iron (Fe), manganese (Mn), zinc

(Zn), and smaller concentrations of other metals (Chaves et al., 2010).[16] Potential sources of metal contamination include:

1. The potassium and sodium methoxide catalysts that break down triglycerides to methyl esters (NaOH and KOH can also be used) can contribute metals to biodiesel. These metals can form soaps with free fatty acids, and the soaps in both the metal esters and glycerin forms are reacted with acid (hydrochloric acid) to convert the soaps to free fatty acids to simplify their removal. Sodium hydroxide is added to neutralize any acid added to eliminate soaps.
2. Methyl esters are washed, distilled, or filtered to remove the metals added as catalysts. The wash water is recycled, and metal ions can accumulate in the wash water. Hard wash water containing $CaCO_3$, $Mg(OH)_2$, $CaSO_4$ is found in Rocky Mountain states and the Midwest, and these water-soluble compounds can accumulate in the residual water found in biodiesel.
3. The medium used to filter methyl esters could also contribute to metals in the biodiesel. The filter material is typically made up of diatomaceous earth which is primarily silica containing alumina, iron oxide, and calcium oxide. In addition, small amounts of calcium or magnesium can be added to the fuel from the purification process (Alleman, 2013; Alleman and McCormick, 2008).

Across a range of concentrations, metals in biodiesel can be present as ions, abrasive solids, or soluble metallic soaps. Abrasive solids can contribute to wear of fuel system components, pistons, and rings, as well as contribute to engine deposits. Soluble metallic soaps have little impact on wear but may contribute to diesel particulate filter plugging and engine deposits. Metal accumulation in diesel particulate filters can increase pressure drops and result in shorter times between maintenance intervals (Jääskeläinen, 2009; Sappok and Wong, 2007). A level of 1 milligram per kilogram (mg/kg, 1 part per million) of trace metal in the fuel results in an estimated accumulation of approximately 22 g of trace

[16] Biodiesel quality, including metal content, is regulated by ASTM D6751-20a for B100 fuels (ASTM, 2020). ASTM D6751-19 sets a limit of 5 parts per million for combined Na and K (group 1A metals) and a limit of 5 parts per million for combined Ca and Mg (group 2A metals) using the EN14538 inductively coupled plasma optical emission spectroscopy (ICP-OES) measurement method. ASTM D6751-20a also places a 10-parts per million limit on P (group 5 metal) using the ASTM D4951 inductively coupled plasma atomic emission spectroscopy (ICP-AES) measurement method. The limits on metals in ASTM D6751 are meant to be protective when biodiesel is used in blends (e.g., B20, B10). Fuel quality for biodiesel blends in the B6 to B20 range is regulated by ASTM D7467-19. This specification does not contain a metal limit for these biofuel blends because, as the method states, the concentration would likely be too low to measure using the ICP-OES method specified (EN 14538). Similarly, D975 regulates B0 to B5 and does not have a metals specification (just a total ash percent limit of 0.01%). Thus, the basis for control of metals in biodiesel blends is control of the B100 blend stock. The rationale is if the B100 fuel is under the ASTM D6751-19 limit, the combined Na + K and Mg + Ca will be below 1 parts per million respectively for B20 and lower blends. However, the actual metal content of today's fuels can be challenging to quantify when it is lower than the 1 parts per million level specified for B20 and lower blends, because of the detection limit of the current test methods. The detection limit of the EN14538 is 1 parts per million for each metal, and the method includes a statement if the metal is below the limit of detection of the method, then it is not included in the reporting calculation.

metal in diesel particulate filters per 100,000 miles (assuming a fuel economy of 15 miles per gallon and 100% trapping efficiency).

Metallic fuel contaminants can also accumulate on fuel injectors, or be converted to oxides, sulfates, hydroxides, or carbonates in the combustion process, which forms an inorganic ash that can deposit onto the exhaust emission control devices found in modern diesel engines (Williams et al., 2013). Alkali metals are well known poisons for catalysts used in emission control devices, and have been shown to negatively impact the mechanical properties of ceramic substrates (Cavataio et al., 2009; Dou and Balland, 2002). Alkali metal hydroxides such as Na and K are volatilized in the presence of steam and therefore can penetrate the catalyst washcoat or substrate.

During the process of developing the Advance Notice of Proposed Rulemaking (ANPRM) for 2027 and later heavy-duty engine emissions standards (U.S. EPA, 2020b, c), an engine manufacturer raised concerns to EPA that biodiesel was a source of high metal content in highway diesel fuel, that higher biodiesel blends (e.g., B20) were the principal problem, and that the metals content in diesel fuel could pose a challenge to meeting new NOx standards for heavy-duty diesel engines. The engine manufacturer reported higher than normal concentrations of alkali and alkaline earth metals (Na, K, Ca, and Mg) in highway diesel fuel samples, and fouling of the exhaust aftertreatment systems of their engines, which caused an associated increase in emissions. The engine manufacturer sampled the ash that was fouling fuel injectors and aftertreatment systems and determined the ash to be composed of sodium sulfate, sodium carboxylates, and sodium chloride, which they claimed were from biodiesel. The engine manufacturer recommended limiting biodiesel blends to 5% biodiesel (B5). After hearing engine manufacturer concerns about the metal content in biodiesel in early 2019, EPA began to investigate the issue of biodiesel metal content.

As part of the Draft Regulatory Impact Analysis (Draft RIA) for the Cleaner Trucks Initiative (U.S. EPA, 2020b), EPA conducted a literature survey of studies that collected and analyzed emission data from diesel engines operated on biodiesel blended diesel fuel with controlled amounts of metal content. Within the same Draft RIA, EPA also reviewed studies by the DOE's National Renewable Energy Lab (NREL) on the metal content of biodiesel and biodiesel blends conducted between 2007 and 2018 (Alleman, 2020a, b; Alleman et al., 2019; Alleman, 2013; Alleman and McCormick, 2008; Alleman et al., 2007). Analyses of biodiesel metals content within the Draft RIA also included analytical results from an EPA study of 27 B100 fuel samples and results from a separate California Air Resources Board (CARB) study of an additional 355 biodiesel and diesel fuel samples from both #2 diesel-labeled pumps and biodiesel-labeled pumps in California (CARB, 2020).

A review of the NREL, EPA, and CARB datasets indicated that biodiesel fuel is compliant with the ASTM D6751-18 limits for Na, K, Ca, and Mg. While the test results indicate that there is an occasional B100 blend stock that is off specification with respect to the ASTM D6751-18 limits, and occasional B5 to B20 blends that are off specification relative to the pseudo limits,[17] these occurrences are the exception. The EPA, CARB, and recent (2016 and later) NREL data sets all used measurement methods that afford low levels of detection (sub-100 parts per billion), and these datasets further indicate that the Na, K, Ca, and Mg content of biodiesel blends is extremely low in general, on the order of less than 100 parts per billion. While these metals are present in biodiesel blends and testing has shown that exposure to metals can adversely affect emission control system performance, the magnitude of the impact remains a subject of research.

8.3.1.2.5 Full Lifecycle

Hums et al. (2016) performed a lifecycle assessment study for biodiesel, comparing the emissions from soybean biodiesel and grease trap waste (GTW) with low-sulfur diesel. For this well-to-wheel study, Hums et al. (2016) used SimaPro8 (a commercial software package developed by PRé Sustainability commonly used for lifecycle inventory and impact analysis) and the ecoinvent database (Jungbluth et al., 2007) for GTW biodiesel, whereas soybean biodiesel and low-sulfur diesel processes were modeled using GREET-2014 data. Uncertainty in the composition of the FOG has been documented in a previous study (Tu and McDonnell, 2016). Since lipid content in GTW can vary significantly, this analysis included varying lipid content ranging from 2% to 40%. While there is a large range in GTW lipid content, the mean percentage is toward the lower end with a mean of about 4% (Ward, 2012). A comparison of relative change in emissions of select criteria pollutants from soybean diesel, low-sulfur diesel, and GTW-derived diesel is shown in Table 8.6. At low lipid contents, emissions of CO from GTW-derived diesel are much higher than soybean diesel and low-sulfur diesel when the methane produced is flared or when there is cogeneration. When considered without waste management, GTW-derived diesel performs better, with lifecycle CO emissions similar to soybean diesel but much lower than from low-sulfur diesel. At high lipid contents, emissions of most criteria pollutants decrease compared to the soybean diesel, with larger decreases for PM and SO_x.

[17] The limits in ASTM D6751 only apply to B100. In this case, the ASTM D6751 limits were compared to B5–B20 blends. However the finished, blended product is not technically subject to the D6751 limits, only the preblended B100 used for blending is subject to D6751.

Table 8.6. Percent change in emissions of various criteria pollutants per megajoule (MJ) fuel for the 'without GTW waste management' scenario. Emissions are normalized with the soybean diesel emissions. Source: Hums et al. (2016).

Pollutant	Soybean	Low-sulfur diesel	GTW diesel % change in emissions (grams of pollutant/MJ-fuel) for various lipid content levels						
			2%	4%	7%	10%	20%	30%	40%
CO	1	66	13	1	-5	-7	-9	-10	-11
PM	1	5	-7	-22	-29	-31	-34	-35	-36
NO_x	1	-10	13	-7	-16	-20	-24	-25	-26
SO_x	1	-39	-58	-44	-50	-71	-72	-73	-73

MJ = megajoules

8.3.2 Literature Review: Air Quality Impacts

8.3.2.1 Recent Literature

Since the second report, only a handful of research papers were identified that address air quality impacts of biofuels. Hoekman et al. (2018) reviewed research on potential air quality impacts. They noted that ethanol emissions serve as a precursor to two pollutants that participate in the ozone formation process, acetaldehyde and peroxyacetyl nitrate (PAN). They also conclude that because upstream[18] emissions of NO_x, SO_x, $PM_{2.5}$, ammonia, CO, and VOCs are higher for ethanol production from corn than gasoline production, there is a potential for adverse air quality impacts from upstream emissions. Finally, they conclude that E10 provides no ozone benefit compared to E0, and that there is no reason to believe that the performance of E20 would be significantly different from that of E10. Given the lack of more recent studies, conclusions of this review, however, should not be viewed as definitive.

Other investigators have looked at impacts specific to agricultural production or end use. As discussed in section 8.3.1.1.1, Hill et al. (2019) and (Domingo et al., 2021) addressed air quality impacts of corn production, including specifically for biofuel production. Wallington et al. (2016) focused on end-use emissions and concluded that future increases in biofuel content when accomplished in concert with changes in engine design and calibration for new vehicles should not result in problematic increases in emissions impacting urban air quality and may in fact facilitate future required emissions reductions.

8.3.2.2 EPA Anti-backsliding Study

EPA recently released its "anti-backsliding study" (ABS) required under Section 211(v)(1) of the Clean Air Act (U.S. EPA, 2020b, c). The study examined the impacts on air quality from required renewable fuel volumes as a result of changes in vehicle and engine emissions resulting from required renewable fuel volumes under the RFS. Specifically, the study compared two scenarios for calendar year

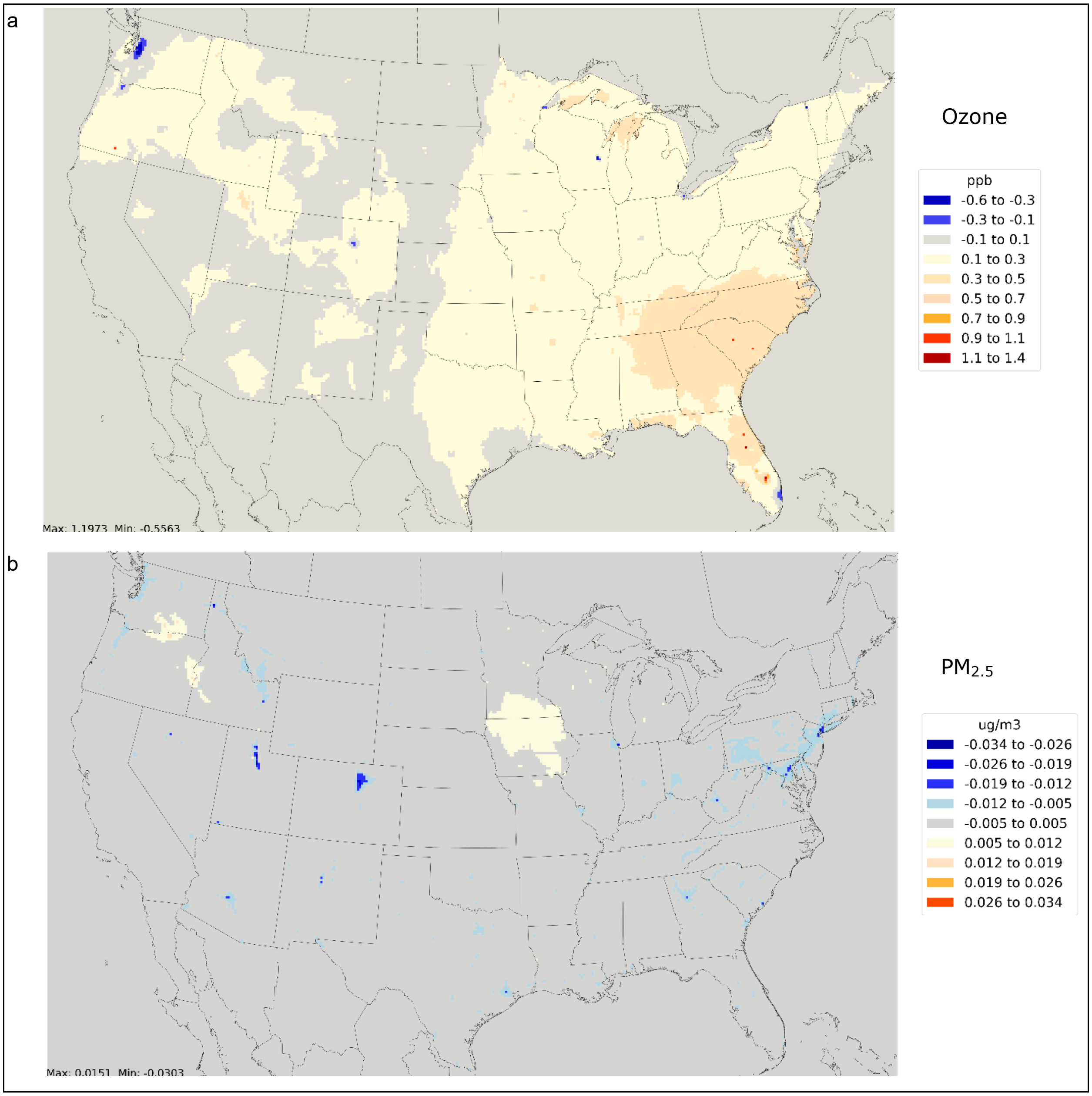
[18] Here upstream means upstream of the vehicle.

2016, one with actual air quality impacts of 2016 ethanol and biodiesel volumes from renewable fuel usage (the "with Renewable Fuel Standard (RFS)" scenario), as compared to another with ethanol and biodiesel air quality that would have resulted in 2016 if renewable fuel usage approximated 2005 levels (the "pre-RFS" scenario).[19]

The "with-RFS" scenario assumed biodiesel at a 5% blend (B5) in all on-road diesel vehicles nationwide, and 10% ethanol (E10) was used nationwide in all on-road and nonroad gasoline-fueled vehicles and engines. This was compared to the "pre-RFS" scenario, which assumed no biodiesel usage (except in California) and E10 usage only in the 2016 reformulated gasoline (RFG) areas and no biodiesel usage (except in California). In California, this scenario assumed that the "pre-RFS" scenario was the same as the "with-RFS" scenario and therefore did not model any emissions changes there. Everything was held constant between the two scenarios except the differing fuel supplies for on-road and nonroad engines; "upstream" emissions from producing, storing, and transporting fuels and feedstocks were also held constant in both scenarios at 2016 levels. EPA's MOVES model was used to estimate on-road inventories and the Community Multiscale Air Quality (CMAQ) model was used for photochemical air quality modeling.[20,21]

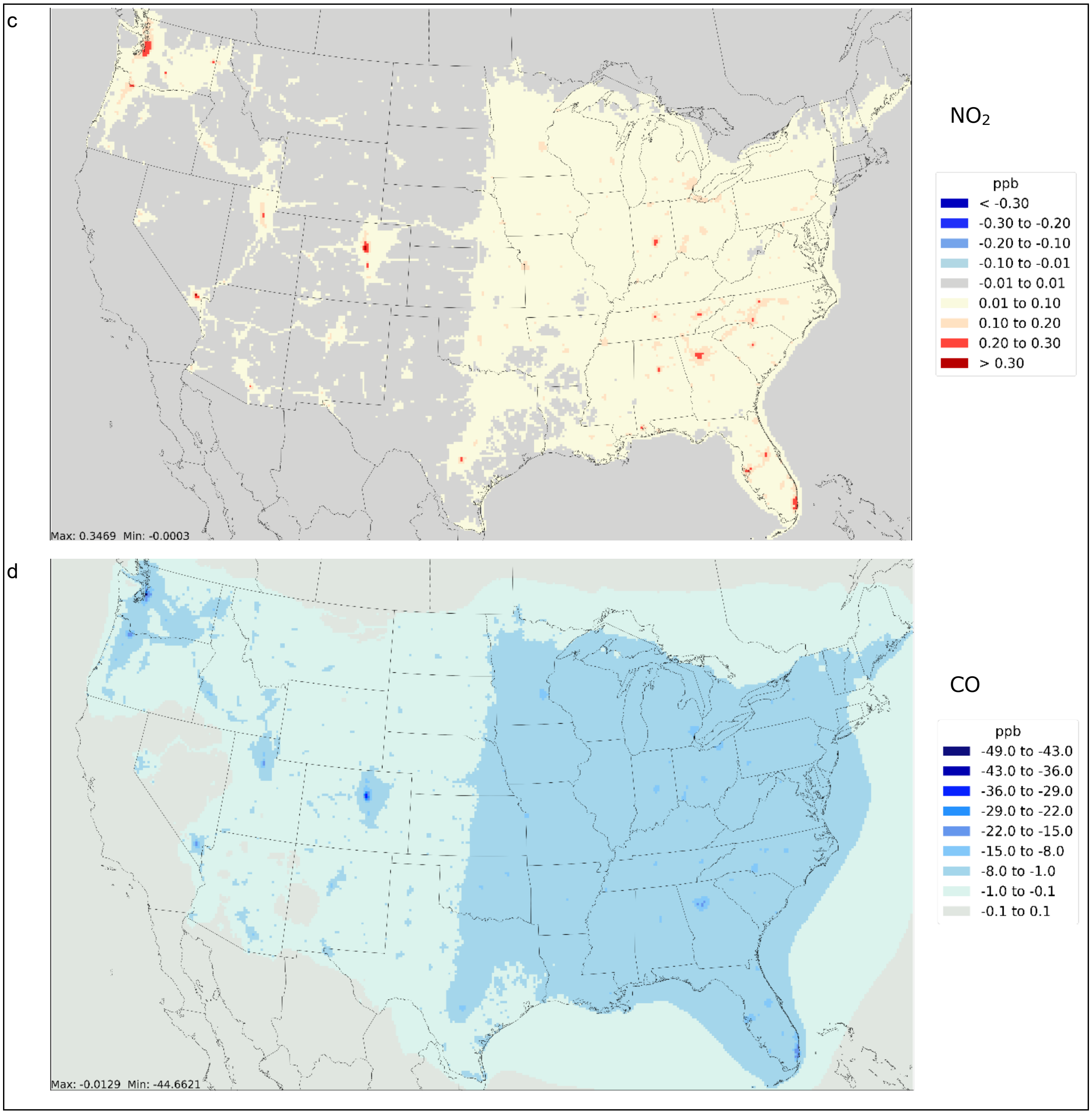
Compared to the "pre-RFS" scenario, the 2016 "with-RFS" scenario increased ozone concentrations (eight-hour maximum average) across the Eastern United States and in some areas in the Western United States, with some decreases in localized areas (Figure 8.9a). In the 2016 "with-RFS" scenario, concentrations of $PM_{2.5}$ were relatively unchanged in most areas, with increases in some areas and decreases in some localized areas (Figure 8.9b). The 2016 "with-RFS" scenario increased concentrations of NO_2 across the Eastern United States and in some areas in the Western United States, with larger increases in some urban areas (Figure 8.9c). The 2016 "with-RFS" scenario decreased concentrations of CO across the Eastern United States and in some areas in the Western United States, with larger decreases in some areas (Figure 8.9d). Compared to the "pre-RFS" scenario, the 2016 "with-RFS" scenario increased concentrations of acetaldehyde across much of the Eastern United States (Figure 8.9e) and some areas in the Western United States, and resulted in increases in formaldehyde concentrations (Figure 8.9f). Compared to the "pre-RFS" scenario, the 2016 "with-RFS" scenario decreased concentrations of benzene (Figure 8.9g, Figure 8.9h) and 1,3-butadiene concentrations were relatively unchanged.

[19] It is important to note that the anti-backsliding study was not a full lifecycle assessment, but rather a detailed assessment of the changes in emissions and air quality at the end use stage of the lifecycle.
[20] https://www.epa.gov/moves
[21] https://www.epa.gov/cmaq



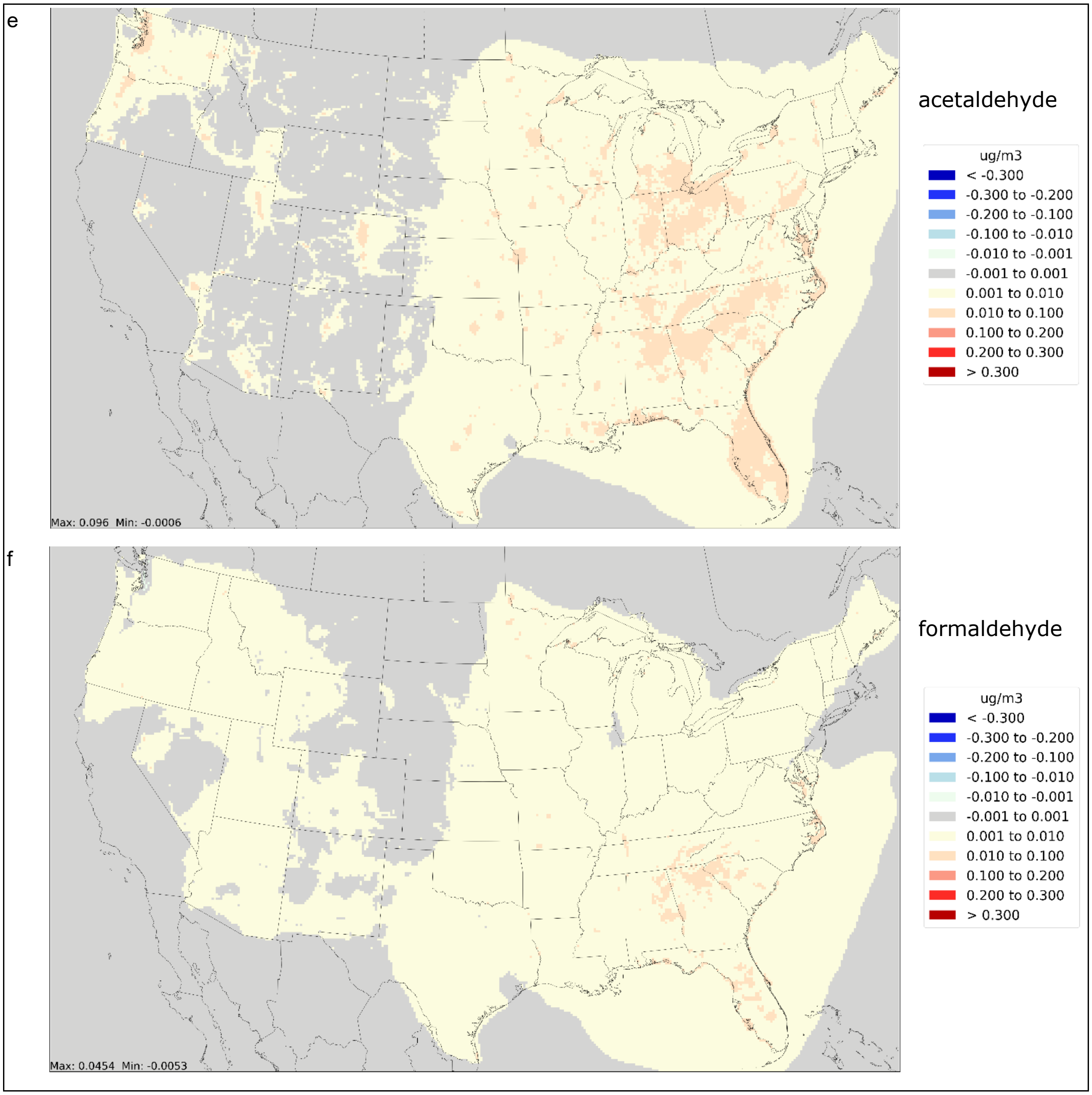
ppb = parts per billion; ug/m^3 = micrograms per cubic meter

Figure 8.9. Absolute change in 2016 between "pre-RFS" and "with-RFS" scenarios for average seasonal concentrations of 8-hour maximum ozone (a), and average annual concentrations of 8-hour maximum $PM_{2.5}$ (b), NO_2 (c), CO (d), acetaldehyde (e), formaldehyde (f) benzene (g) and 1,3-butadiene (h). Results from the EPA Anti-Backsliding Study (U.S. EPA, 2020c). (continued)



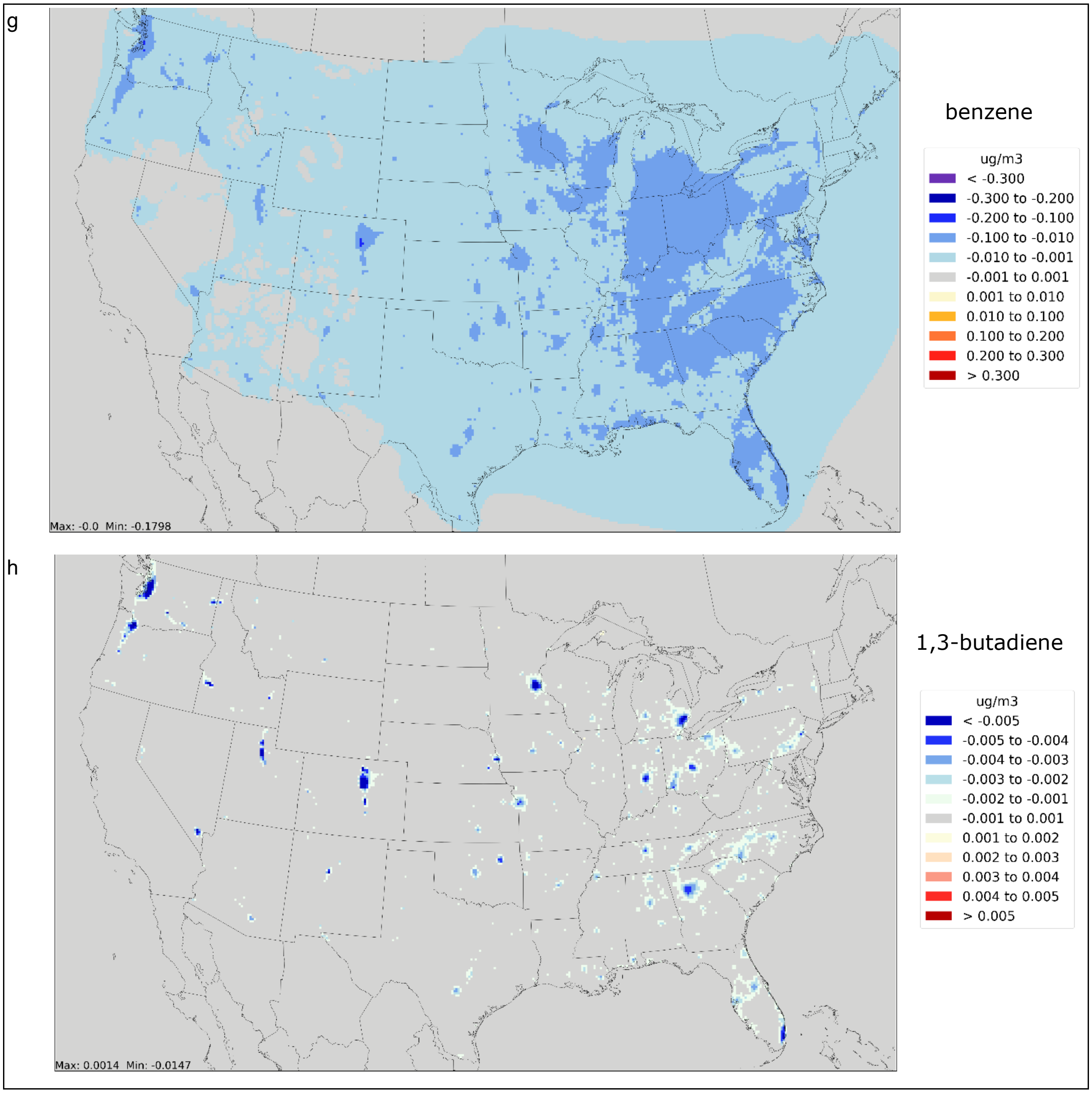
ppb = parts per billion; ug/m^3 = micrograms per cubic meter

Figure 8.9. Absolute change in 2016 between "pre-RFS" and "with-RFS" scenarios for average seasonal concentrations of 8-hour maximum ozone (a), and average annual concentrations of 8-hour maximum $PM_{2.5}$ (b), NO_2 (c), CO (d), acetaldehyde (e), formaldehyde (f) benzene (g) and 1,3-butadiene (h). Results from the EPA Anti-Backsliding Study (U.S. EPA, 2020c). (continued)



ug/m^3 = micrograms per cubic meter

Figure 8.9. Absolute change in 2016 between "pre-RFS" and "with-RFS" scenarios for average seasonal concentrations of 8-hour maximum ozone (a), and average annual concentrations of 8-hour maximum $PM_{2.5}$ (b), NO_2 (c), CO (d), acetaldehyde (e), formaldehyde (f) benzene (g) and 1,3-butadiene (h). Results from the EPA Anti-Backsliding Study (U.S. EPA, 2020c). (continued)



ug/m^3 = micrograms per cubic meter

Figure 8.9. Absolute change in 2016 between "pre-RFS" and "with-RFS" scenarios for average seasonal concentrations of 8-hour maximum ozone (a), and average annual concentrations of 8-hour maximum $PM_{2.5}$ (b), NO_2 (c), CO (d), acetaldehyde (e), formaldehyde (f) benzene (g) and 1,3-butadiene (h). Results from the EPA Anti-Backsliding Study (U.S. EPA, 2020c).

8.3.3 New Analyses

There were no new analyses conducted for the RtC3 under air quality.

8.3.4 Attribution to the RFS Program

Chapter 6 concluded that an estimated 0–1.0 billion gallons of corn ethanol from 2006–2011 and a maximum of 0–2.1 billion gallons in 2016 may be attributable to the RFS Program specifically. A maximum of 0–3.5 million acres of corn and 0–1.9 million acres of cropland in 2016 is estimated attributable to the RFS Program. Chapter 7 concluded that a significant portion of the biodiesel production was likely attributable to the RFS Program but did not derive a quantitative estimate. There are several remaining uncertainties before estimates of the air quality effects of the RFS-attributable biofuel may be conducted. These include the quantitative estimate of biodiesel attributable to the RFS, methods for allocating the RFS-attributable biofuel to the fleet of U.S. biorefineries, and details on local land management for the farms supplying feedstocks to the biorefineries producing RFS-attributable biofuel. Thus, because of these and other uncertainties, the requisite air quality modeling has not been performed to determine the estimated effects on air quality from these estimated amounts of biofuel production and use in the United States due to the RFS Program (e.g., BenMAP).

8.3.5 Opportunities to Offset Negative Effects and Promote Positive Effects

As discussed in section 8.3.1.1.1, improved nitrogen management practices can offset some agricultural impacts. Additionally, greater use of other biofuels (e.g., cellulosic ethanol and renewable diesel) may offset some negative effects (section 8.6). Other opportunities include use of cleaner energy sources for biofuel production, increased use of hexane for extraction in soy biodiesel production (section 8.3.1.2.2), and improvements in emission controls at production facilities and in agricultural equipment. In addition, increasing supply chain efficiency can yield emissions improvements. For example the Billion Ton Study 16 (BTS 16) (DOE, 2017, 2016) evaluated two logistics systems, one conventional and one advanced. The conventional logistics system entails the use of equipment and infrastructure designed for current agricultural commodities. For example, the conventional system for agricultural residues and dedicated herbaceous energy crops utilizes conventional harvest and baling equipment; the biomass is transported in the form of large round bales. The advanced system represented a future scenario that is designed to supply a commoditized feedstock. Results suggested these changes could yield improvements that vary with feedstock.

8.4 Likely Future Impacts

The likely future effects of the RFS Program on biofuel production, consumption, and several economic and environmental factors have been estimated by EPA in the Final Set Rule that covers volumes for 2023–2025.[22] The biofuels estimated to be consumed in each year in the Final Set Rule are presented in Chapter 2 (Table 2.4), and the volume of biofuels estimated to be attributable to the RFS Program each year are presented in Chapter 6 (Table 6.12). Rather than repeating the extensive information elsewhere in this report and in the RIA for the Final Set Rule, only the key information relevant for this chapter is summarized here.

EPA projected that much of the renewable fuel projected to be used in 2023–2025, including much of the cellulosic biofuel, biodiesel, and renewable diesel, was attributable to the RFS Program. Notably, however, EPA projected that ethanol would continue to be used nationwide in E10 blends in the absence of the RFS Program, so only a small portion of the ethanol projected to be used in 2023–2025, namely the ethanol used in higher level ethanol blends such as E15 and E85, is estimated attributable to the RFS Program. Table 6.12 shows that EPA estimates an increase in total renewable fuels from the RFS Program by almost 5 billion gallons by 2025, mostly from increases in biodiesel and renewable diesel (+2.1 billion gallons) from largely soybeans, as well as smaller increases from CNG/LNG from biogas (+0.9 billion gallons) and corn ethanol (+0.8 billion gallons). Note that these volumes (Table 6.12) are different from the estimated change from 2022–2025 (Table 2.4), because the change from 2022 from all causes is different from the volume estimated attributable to the RFS. For the crop-based biofuels, EPA estimated that the RFS Set Rule could potentially lead to an increase of as much as 2.65 million acres of cropland by 2025, mostly due to increased demand for biodiesel and renewable diesel produced from soybean oil. It is important to note the significant assumptions and uncertainty inherent in estimating these nationwide acreage impacts numbers. See the RIA for the Set Rule for further information on potential air quality and other effects (docket EPA-HQ-OAR-2021-0427).

[22] On July 26, 2022, the United States District Court for the District of Columbia entered a consent decree, which after stipulations from the parties, required EPA to sign a notice of proposed rulemaking (NPRM) to establish 2023 volumes for the RFS Program by November 30, 2022, and to sign a notice of final rulemaking finalized 2023 volumes by June 21, 2023. Growth Energy v. Regan et al., No. 1:22-cv-01191 (D.D.C.), Document Nos. 12, 14, 15. EPA's proposed RFS volumes for 2023-2025, 87 FR 80582 (Dec. 30, 2022) available in Docket No. EPA-HQ-OAR-2021-0427 at https://www.regulations.gov, were finalized on June 21, 2023. The estimates of cropland expansion from the RFS Program took considerable time and effort to develop; thus, they are based on the proposed volumes from November 30, 2022, rather than the final volumes from June 21, 2023. Because of that, the acreages discussed here are merely illustrative. However, since the total volumes from crop-based feedstocks changed fairly little between the draft and final rules, they are still relevant here.

8.5 Comparison with Petroleum

The purpose of this section is to compare corn starch ethanol with conventional gasoline and soybean biodiesel with conventional diesel across air quality metrics. Life cycle assessment (LCA) is a rigorous and structured method that allows for a comprehensive comparison incorporating impacts occurring along the full supply chains for production of each fuel. The approaches for LCA have been developed over many years, with concentrated application and standardization of methods beginning in the 1990s. The requirements for LCA are described in the ISO 14000 series of standards, which provide detailed guidance to promote the proper use and interpretation of LCA studies (ISO, 2006). This section attributes biofuels and petroleum across their respective life cycles to *potential* changes in environmental conditions. It does not consider anything about the attribution of the RFS Program (discussed in Chapters 6 and 7). Also, the estimates from these models are *potential* changes in the environment in that they only estimate emissions and are not linked with fate and transport models that include human or natural populations that may be affected downwind.

This section presents results from two LCA models: (1) the GREET model (section 8.5.1) and (2) the Bio-based circular carbon economy Environmentally-extended Input-Output Model (BEIOM, section 8.5.2). GREET is a well-established and detailed process-based LCA model that was originally developed for comparison of transportation fuels and technologies considering the detailed parameters and relationships involved in their production processes. It provides a "bottom up" assessment examining the detailed processes involved in the production and use of a gallon of fuel, and the associated environmental effects. GREET has been extensively developed and used to support formal decision making in several contexts.[23] BEIOM is a newer model developed by NREL that takes an entirely different and novel approach (Lamers et al., 2021). BEIOM is an economy-wide model that uses economic transactions between industries involved in biofuels, together with environmental effect inventories from the EPA's TRACI model (Tool for Reduction and Assessment of Chemicals and Other Environmental Impacts, Bare et al. (2012)) to provide a "top-down" assessment of the environmental effects from the biofuels industry at an economy-wide scale (see Appendix F, Avelino et al. (2020), and

[23] The GREET model has been developed at Argonne National Laboratory with support from the Department of Energy since 1995 with the first publication considering the impact of the renewable oxygenate standard for reformulated gasoline on ethanol demand, energy use, and greenhouse gas emissions (Stork and Singh, 1995) closely followed by the first GREET release (Wang, 1996), which included at that time gasoline, diesel, and ethanol fuel life cycles. Since that time, the GREET model has been regularly updated with expanded and improved datasets for gasoline, diesel, ethanol, and, since 2008, biodiesel (Huo et al., 2008). The GREET model currently has over 40,000 registered users worldwide and has been widely used by industry as well as in connection with pathway analyses for the Renewable Fuels Standard and California's Low Carbon Fuels Standard. It is used across several Department of Energy program offices for technology evaluation and is regularly expanded and updated to incorporate recent data and developments, with new releases annually each October.

Lamers et al. (2021) for details on BEIOM). BEIOM is much newer, and less tested, than GREET, which has been refined over years and used in hundreds of peer-reviewed journal articles. Thus, for estimates of the lifecycle effects of biofuels on the environment more weight is placed on the estimates from GREET. Nonetheless, together these approaches may offer a unique and complementary perspective on the potential environmental effects of biofuels versus fossil fuels across the life cycle of each. Figure 8.10 and Figure 8.11 provide the system descriptions and boundaries for the GREET and BEIOM corn ethanol and soy biodiesel models.

The functional unit for this comparison is the use of one megajoule of fuel. Up to the point of blending, the supply chains for gasoline and ethanol are separate. After blending, they are considered together and then the emissions from transporting the E10 blend, for example, are attributed to the two fuels on the basis of the mass of each component. The analysis includes the full supply chains of production of each fuel. Figure 8.10 and Figure 8.11 illustrate the system and boundaries for the corn ethanol and soybean biodiesel models in GREET and BEIOM, respectively.[24] The scope for gasoline and diesel includes petroleum extraction and refining as well as fuel blending, distribution, fueling, and use, along with the full supply chains of all inputs to each stage of the supply chain and transportation at each stage in the supply chain. In the case of processes that produce multiple products (i.e., coproducts), impacts are allocated to each product on a physical (GREET) or economic basis (BEIOM).

[24] The fuel supply chains for GREET and BEIOM are identical, but BEIOM extends the system boundary to the U.S. economy.

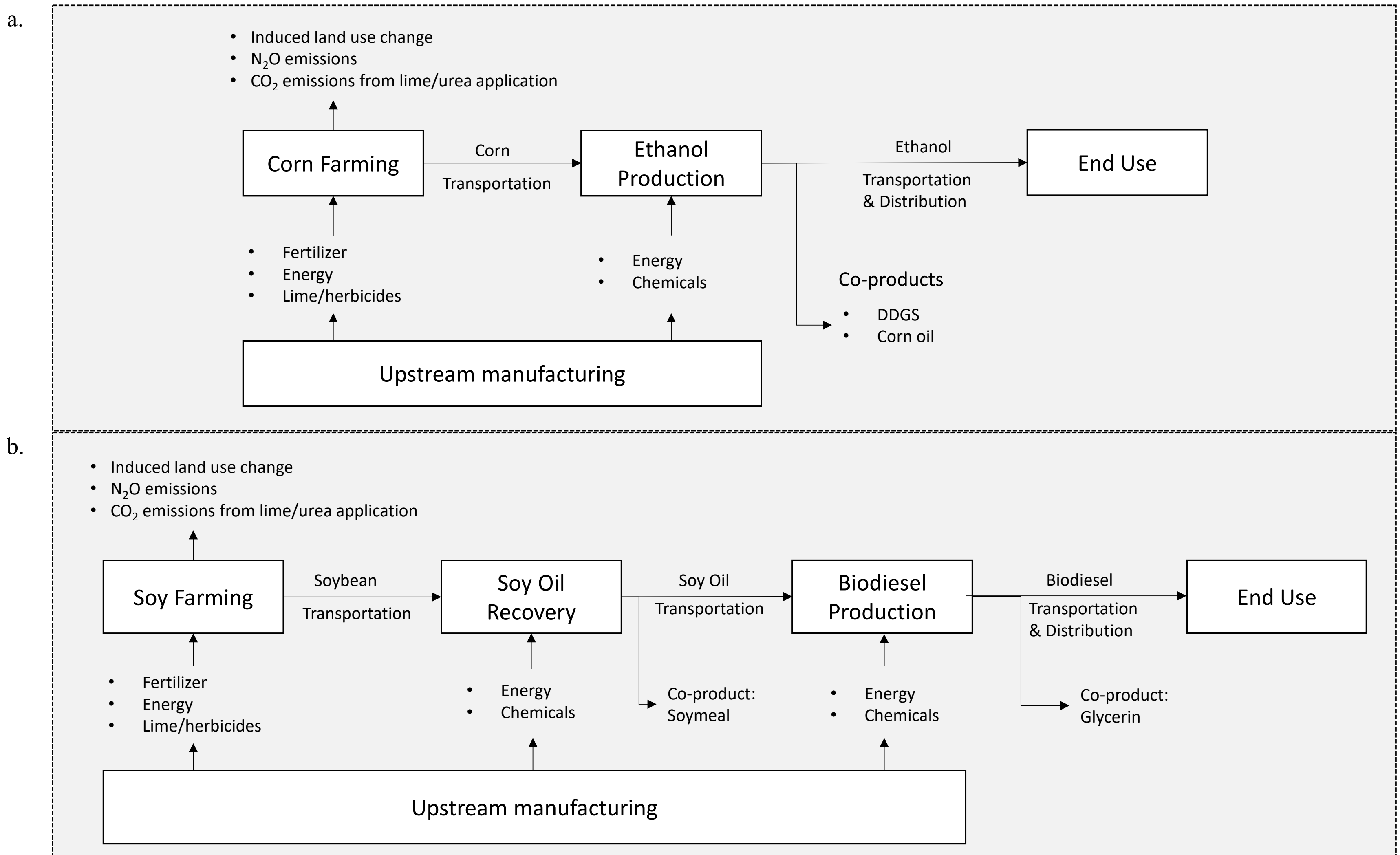


Figure 8.10. System description and boundaries for GREET corn ethanol (a) and soybean biodiesel (b) models.

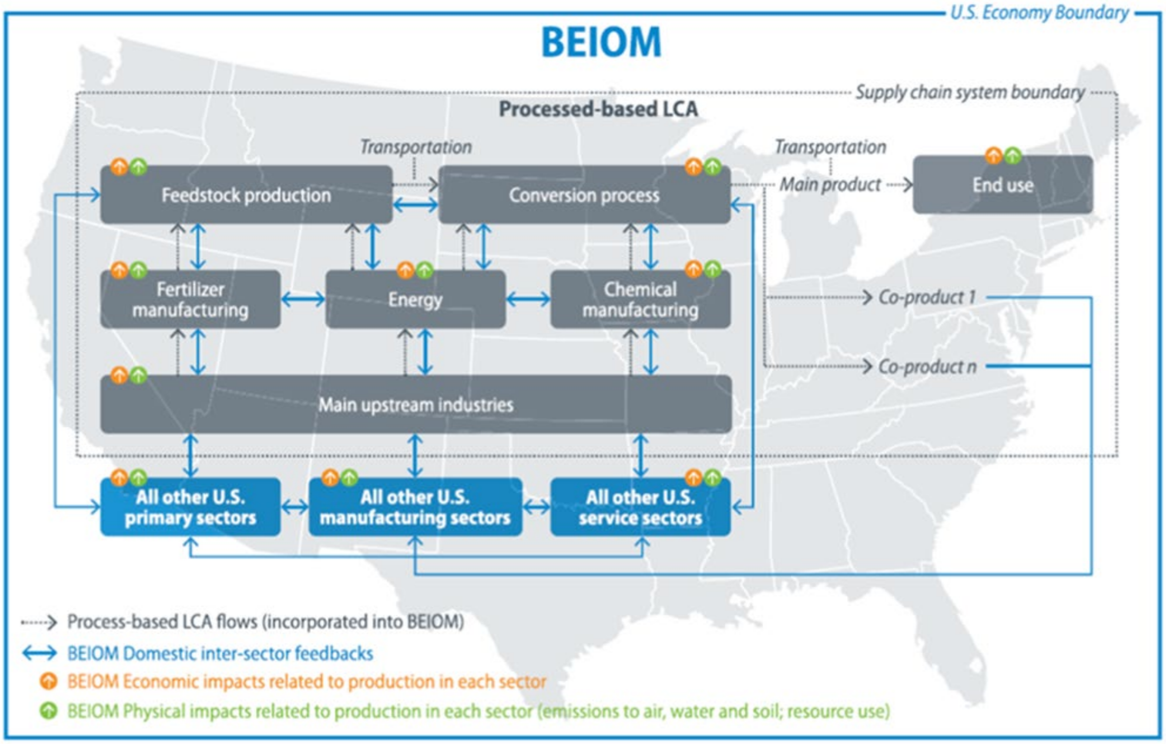


Figure 8.11. System description and boundary for BEIOM corn ethanol and soybean biodiesel models. Source: Lamers et al. (2021).

8.5.1 Life Cycle Analysis of Fuel Pathways with the GREET

GREET 1 2019 was used for this analysis (Wang et al., 2019) with results reported for life cycle emissions of VOC, SO_x, $PM_{2.5}$, PM_{10}, CO, and NO_x.[25] Results for life cycle water consumption are reported in Chapter 11. GREET also tracks life cycle GHGs and energy use, which are out of scope for this report but are reported in the studies cited in connection with the models for each fuel supply chain discussed in this section. Results are reported separately for dry mills with corn oil extraction for ethanol as a fuel, dry mills without corn oil extraction, and wet mills. The distiller's corn oil is primarily used in animal feeds (~50%) and biodiesel production (45%) with the remainder used for other industrial purposes (5%) (Shurson, 2021). In the case of biodiesel, the results reflect U.S. average[26] biodiesel produced from soybean oil. Results for gasoline and diesel reflect U.S. average production, additional detail is provided subsequently in this section. While many of the studies cited in connection with the GREET model development focus on GHG emissions, the models developed for those studies and the

[25] The version of GREET used in this analysis does not track ammonia, which contributes to $PM_{2.5}$. There has been work published, along with a publicly available version of GREET, which includes NH_3 and more finely resolved spatial data from the EPA National Emissions Inventory (e.g., Hill et al. (2019)).
[26] The results reflect average soybean production, weighted across production locations by share of production. Similarly, it considers industry average practices for biodiesel production from soybean oil.

emissions factors in GREET modules include the criteria air pollutant emissions reported here. A full list of GREET publications and summaries of annual updates are provided on the GREET website.[27]

The structure and primary datasets for the corn ethanol model are described by Wang et al. (2012). The results were updated by Wang et al. (2015) to account for corn oil extraction at dry mills and to compare the effect of different coproduct modeling approaches on the results. The "marginal approach" is used, which assumes corn ethanol plants exist primarily to produce corn ethanol, thus the impacts of corn production and conversion are allocated to ethanol except for the energy consumed for corn oil recovery. Distiller's grains are another important coproduct of corn ethanol production (see Chapter 3) and thus their treatment in the GREET model is important for interpreting the results. Here, distiller's grains are assumed to displace other conventional animal feed components, corn, soybean meal, and urea, which would otherwise be produced, in the amounts specified in Table 8.7 (Arora et al., 2010). Table 8.7 also describes the average corn yield, diesel use, and fertilizer use for the corn used for ethanol production and the energy use for ethanol production by technology. The results presented reflect average corn production. Liu et al. (2020) describe the farming model in more detail and provide insight into the performance improvements which could be achieved were agricultural management practices incentivized in biofuels policy.

The structure of the GREET model for soybean biodiesel production is described by Chen et al. (2018) and Huo et al. (2008). The GREET model assumes energy use of 18,433 British thermal units (BTU) per bushel for soybean production plus fertilizer inputs of 48.1, 187, and 299 grams per bushel of nitrogen, P_2O_5, and K_2O respectively (it is assumed no lime is applied) (Table 8.7). The soy oil yield is 52 pounds per bushel of soybean. As explained by Han et al. (2014), a mass-based allocation is used for the oil extraction process to allocate upstream impacts between the soy oil and soybean meal, and a market-based allocation is used for the fuel production process to allocate upstream impacts between the biodiesel and glycerin coproducts based on $0.547 per pound for the biodiesel and $0.250 per pound for the glycerin.

The GREET model for gasoline and diesel supply chains is based on detailed models of petroleum extraction including conventional petroleum extraction as well as oil sands (Cai et al., 2015) and shale oil from the Bakken (Brandt et al., 2015) and Eagle Ford (Ghandi et al., 2015) formations. Refining is modeled at the level of refinery subprocesses using process-specific energy use and yields (Elgowainy et al., 2014) and emissions factors (Sun et al., 2019). The models for all fuels include criteria

[27] Argonne National Laboratory. GREET Publications. Website, updated 2021. Accessed 5/18/2021: https://greet.es.anl.gov/publications

air pollutant emissions associated with the transportation, distribution, dispensing, and use of each of the fuels.[28]

GREET estimates distinguish emissions occurring in urban and non-urban areas to address potential differences in human exposure to the associated air quality effects. Urban shares of emissions are estimated for each process along the supply chains in GREET based on various data sources including the locations of facilities, farms, and mines and their contributions to total production and emissions.

Table 8.7. Key parameters for GREET corn ethanol and soybean biodiesel calculations. Data reflect current conditions subject to data availability (e.g., soybean biodiesel production is based on Chen et al. (2018)).

Parameter	Corn	Soybean
Crop production		
Yield, bushels per acre	166	48
Diesel use, BTU per bushel	3,409	12,985
Gasoline use, BTU per bushel	1,018	2,902
Natural gas use, BTU per bushel	938	933
LPG use, BTU per bushel	1,242	726
Electricity use, BTU per bushel	318	887
Fertilizer, herbicide, and pesticide use, grams per bushel		
Nitrogen	383	48.1
P_2O_5	139	187
K_2O	146	299
$CaCO_3$	1,290	0
Herbicide	5.85	17.9
Insecticide	0.01	0.4

	Dry mills without corn oil extraction	Dry mills with corn oil extraction	Wet mills	Soybean oil extraction	Soybean oil transesterification
Parameter	BTU per gallon ethanol			BTU per pound soybean oil	BTU per gallon biodiesel
Energy use					
Natural gas	23,934	23,480	34,372	372	3,760
Coal	23	191	13,037	183	-
Residual Oil/Diesel Fuel	-	-	-	9	74
Biomass/Other	195	191	-	19	-
Electricity	2,533	2,509	-	80	467

[28] GREET results are also available for FOGs, but these are only available for GHGs (Chen et al., 2018) as the non-GHG results are forthcoming.

	Dry mills without corn oil extraction	Dry mills with corn oil extraction	Wet mills	Soybean oil extraction	Soybean oil transesterification
Parameter	**BTU per gallon ethanol**			**BTU per pound soybean oil**	**BTU per gallon biodiesel**
Amount of animal feed components displaced by DDGS, pounds per gallon of ethanol					
Corn	4.40	4.19	7.149		
Soybean meal	1.73	1.65	0		
Urea	0.128	0.121	0.109		

BTU = British thermal units; DDGS = distiller's dried grains with solubles; LPG = liquified petroleum gas

8.5.1.1 GREET Results for Corn Ethanol and Conventional Gasoline

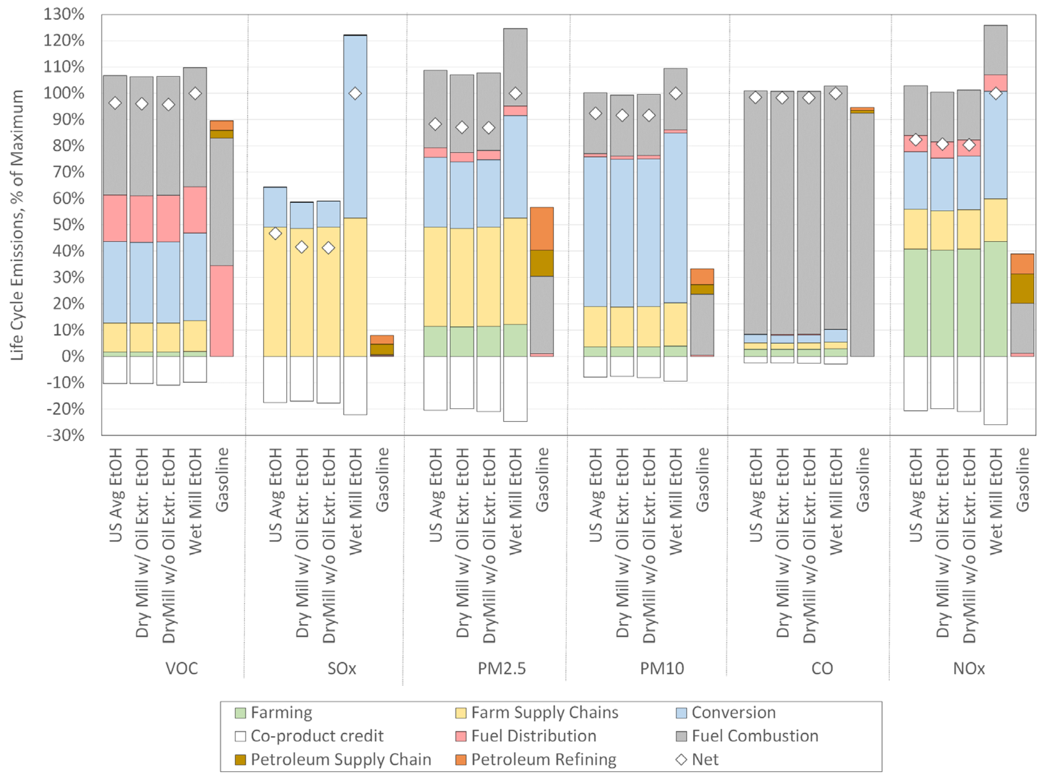
Figure 8.12 provides the comparative results for ethanol and gasoline, and Table 8.8 summarizes the totals. Within Figure 8.12 and Figure 8.13, subpanel (a) details the contributions to net totals from farming, farm supply chains, conversion, coproduct credits (relevant for ethanol only), petroleum supply chains, petroleum refining, fuel distribution (including fuel transportation and distribution as well as VOC emissions from bulk storage refueling stations), and fuel combustion; and subpanel (b) provides the breakdown of emissions occurring in urban and non-urban areas.

Table 8.8. Comparative life cycle criteria air pollutant emissions for corn ethanol, gasoline, soybean oil diesel, and diesel (grams per megajoule, biofuel and fossil fuel separated by a dashed line).

	Corn Ethanol (100%)				Gasoline	Soy Oil Biodiesel	Diesel
Pollutant	U.S. Average	Dry Mill with Corn Oil Extraction	Dry Mill without Corn Oil Extraction	Wet Mill	U.S. Average	U.S. Average	U.S. Average
VOC	0.10	0.10	0.10	0.11	0.097	0.05	0.041
SO_x	0.066	0.058	0.058	0.14	0.011	0.026	0.010
$PM_{2.5}$	0.0062	0.0061	0.0061	0.007	0.004	0.0043	0.0040
PM_{10}	0.021	0.020	0.020	0.022	0.0074	0.0083	0.0079
CO	0.64	0.63	0.63	0.64	0.61	0.76	0.75
NO_x	0.120	0.11	0.11	0.14	0.055	0.069	0.060

Comparing corn ethanol and conventional gasoline, results generally show a trend of increased life cycle emissions for criteria air pollutants for the corn ethanol pathways compared with petroleum-based gasoline. For the U.S. average ethanol production, this trend is stronger for SO_x (+500%), particulate matter ($PM_{2.5}$ [+55%] and PM_{10} [+184%]), and NO_x (+118%). The difference is nearly negligible for VOCs (+3%) and CO (+5%) in the context of the uncertainty/precision of LCA results. Figure 8.12 shows that the contributions to overall results vary between ethanol and gasoline and from one criteria air pollutant to another.

a.



b.

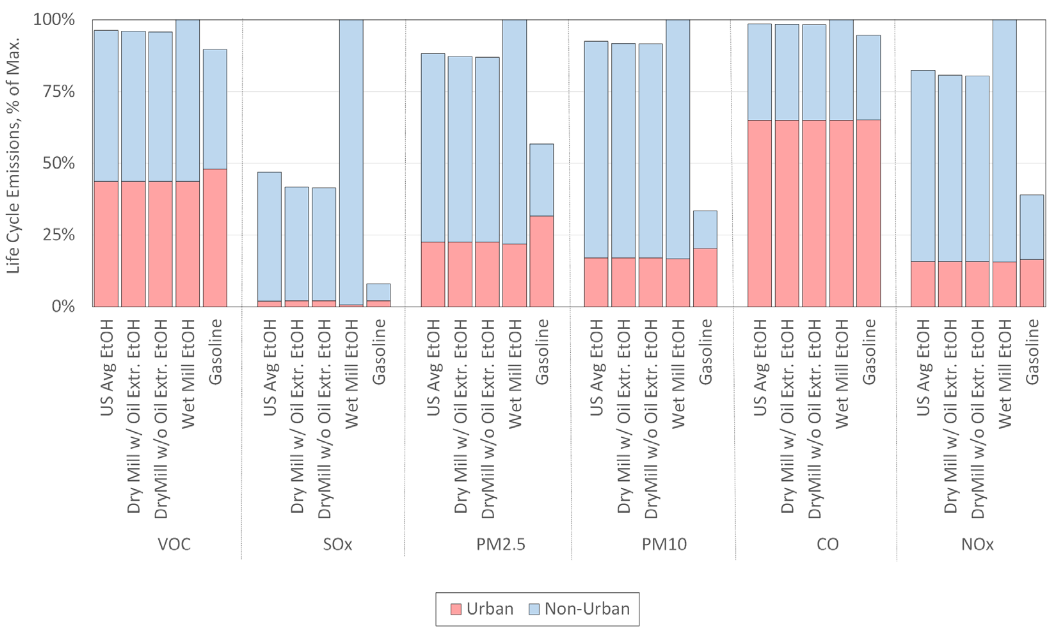


Figure 8.12. Life cycle criteria air pollutant emissions for corn ethanol (100%) and gasoline by life cycle stage (a) and by location of the emissions, urban v. non-urban (b). Source: Wang et al. (2020).[29] Bars are scaled to the maximum result in each category so emissions of all substances can be displayed on the same axes. Negative contributions in (a) reflects the credits associated with the distiller's grain coproduct of ethanol production.

[29] Greenhouse gas emissions are also provided by GREET, but are not in scope for this report series and so are not included here. See Chapter 2 Box 2.2 for additional information.

In the case of VOCs, the most significant contributions for ethanol are fugitive emissions from conversion and during fuel use, with notable contributions also from fuel distribution and farm supply chains. VOCs in the supply chains of gasoline are primarily from fuel distribution and fuel combustion, although it should be noted that Argonne National Lab is currently in the process of estimating additional VOC emissions from petroleum extraction and refining which are not well reported in the National Emissions Inventory (Beath et al., 2020), as described by Allen (2016), which would increase the total for gasoline.

Sulfur oxide emissions from ethanol are primarily associated with farm supply chains, and, to a lesser extent, ethanol production (conversion).[30] Emissions from ethanol production are roughly half due to the electricity used with the other half distributed among the supply chains of other outputs. The share of coal use for dry mills is small, estimated to be 0.8% based on a recent survey of ethanol dry mills by Wu (2019). The current GREET estimate of coal use for wet mills is higher, comprising 27.5% of fuel inputs on an energy basis. Wet mills are estimated to account for 9% of U.S. ethanol production (Chapter 3). Sulfur oxide emissions for gasoline are primarily from petroleum extraction and refining, although these are significantly lower than those for ethanol, owing largely to the economies of scale involved in petroleum extraction and refining as well as process optimization and emissions controls, which have been iteratively improved over the 150+ year history of the U.S. petroleum fuels industry.

Particulate matter emissions ($PM_{2.5}$ and PM_{10}) from ethanol are primarily from conversion, farm supply chains, and combustion, with smaller contributions also coming from corn farming. Particulates from ethanol production are associated with coal combustion as well as fugitive releases from corn grinding, storage, and DDGS. Particulates from corn farming are primarily associated with the use of diesel fuel in farm equipment as GREET does not track dust from fields and tillage in its particulate matter metrics.[31] Particulates from the gasoline life cycle are 35% less than those from the ethanol life cycle in the case of $PM_{2.5}$, and 65% less in the case of PM_{10}. Particulate emissions from gasoline are primarily associated with fuel combustion, although petroleum refining and petroleum supply chains also contribute significantly, in particular to $PM_{2.5}$, where together they comprise nearly half of the total.

Carbon monoxide emissions from both ethanol and gasoline are almost entirely from fuel combustion. As previously noted, there is not a significant difference between the two fuels.

Life cycle emissions of nitrogen oxides from corn ethanol are nearly double those from gasoline with contributions coming from across the entire supply chain/pathway in both cases. The greatest

[30] The fertilizer SO_2 inventory data are compiled from industry reports and literature published before 2010 and may not reflect recent changes in the industry. In addition, they do not account for recent conversion of industrial production from coal to natural gas.
[31] Fugitive dust is incorporated in BEIOM, which partly explains the higher PM estimates there, discussed in section 8.5.2.

contribution for ethanol is from corn farming, which is comprised of contributions from diesel combustion and from field emissions associated with nitrogen fertilizer. Fuel combustion NO_x emissions from ethanol and gasoline are estimated to be roughly the same.

The urban share of life cycle emissions from corn ethanol are uniformly lower than (VOC, $PM_{2.5}$, PM_{10}) or consistent (SO_x, CO, NO_x) with those from gasoline, as shown in part b of Figure 8.12. This is important as the detrimental effects of these pollutants are associated with human exposure to the associated particulates and ozone.

As expected, emissions are concentrated in non-urban areas when they are dominated by farming, farm supply chains, and conversion (for ethanol) (e.g., SOx, PM, NOx), and are concentrated in urban areas when they are dominated by end use (e.g., CO and to a lesser extent VOCs) (Figure 8.12b).

Interpretation of the results should consider the significant variability across operations at various ethanol production facilities, corn farms, oil wells, petroleum refineries, processes along their supply chains, and automobile engines, as described previously in this chapter. While the results presented here reflect a best estimate of the U.S. average operations, emissions for a specific ethanol or gasoline use case would differ. Further, while these estimates provide the best available current accounting for each emissions category, limitations in available data may result in under- or overestimates due to data gaps or measurements not reflective of the most recent operating conditions. Vineyard and Ingwersen (2017) provide a detailed comparison between the life cycle criteria air pollutant emissions for U.S. gasoline based on commonly used LCA models including GREET 2014, ecoinvent 3, National Energy Technology Laboratory's dataset, and the U.S. Life Cycle Inventory. Their comparison found that the results generally vary somewhat widely between the models and the GREET results were always within the range of results reported by the other datasets for the emissions categories reported here. The differences are likely due to differences in the scope and completeness of the datasets. Vineyard and Ingwersen (2017) noted that the GREET model was the most accessible and transparent and its results were better able to satisfy mass and energy balances than the other models. It should be noted that (2017) analysis was conducted prior to the incorporation of significant improvements to the GREET refinery models including "top down bottom up" reconciling of estimates based on emissions factors with facility-specific results reported in the NEI (Sun et al., 2019). These improvements to the GREET model are reflective of current best practices for LCA and were replicated in the widely used Petroleum Refinery Life Cycle Inventory Model (PRELIM) (Young et al., 2019). Nonetheless, the GREET model is in a continual state of updating and improvements including factors and updates as they emerge (e.g., industrial CO_2 as a coproduct of corn ethanol production, improvements in tillage practices).

8.5.1.2 GREET Results for Soybean Biodiesel and Conventional Diesel

Figure 8.13 provides the comparative results for biodiesel and diesel and Table 8.8 summarizes the totals. Comparing soy oil biodiesel and conventional diesel (Table 8.8), results generally show a trend of increased life cycle emissions for the soy oil biodiesel pathways compared with petroleum biodiesel.

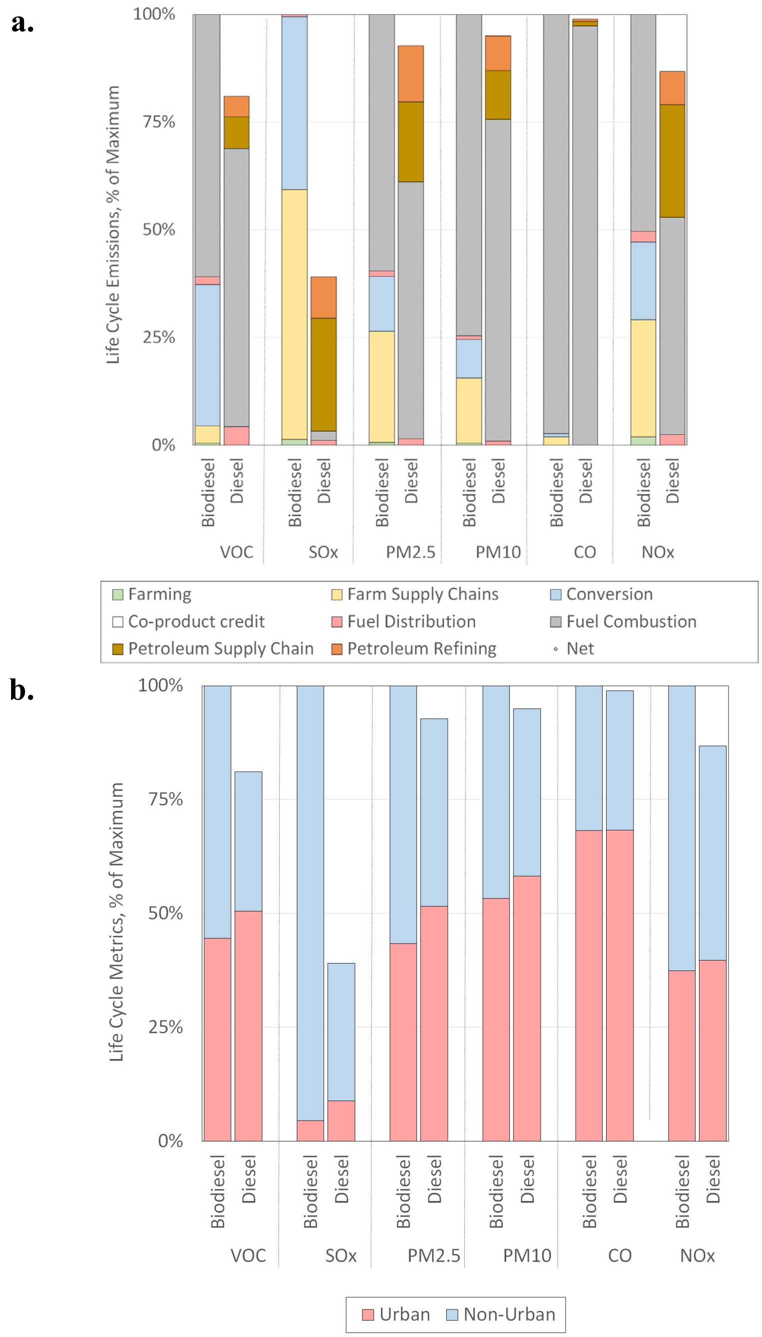


Figure 8.13. Life cycle criteria air pollutant emissions for soy biodiesel and conventional diesel by life cycle stage (a) and by location of the emissions, urban v. non-urban (b) from GREET 2020. Bars are scaled to the maximum result in each category so emissions of all substances can be displayed on the same axes.

This trend is stronger for life cycle emissions of sulfur oxides (+160%) and volatile organic compounds (+22%) and less conclusive for carbon monoxide (+1.3%), particulate matter ($PM_{2.5}$ [+7.5%], PM_{10} [+5.1%]), and nitrogen oxides (+15%). As noted previously for the ethanol and gasoline results, small differences could be considered within the uncertainty bounds of LCA results. Figure 8.13 shows that the contributions to overall results for biodiesel are generally from use (fuel combustion), biodiesel production, and farm supply chains while results for conventional diesel are generally from use (fuel combustion), petroleum supply chains, and petroleum refining.

For VOCs, the results for biodiesel are about 22% greater than those for conventional diesel. The primary source of this difference is increased VOC emissions from biodiesel production compared with petroleum refining, on a per unit energy basis. This is likely due to greater economies of scale for petroleum refining compared with biodiesel production, together with process optimization and more advanced emissions controls associated with the greater maturity of the petroleum refining industry. Thus, biodiesel may "catch up" with conventional diesel in terms of VOC emissions as the industry further develops or if the scale of production increases.

Sulfur oxide shows the most significant difference between biodiesel and conventional diesel with biodiesel exhibiting 160% higher life cycle emissions. Due to sulfur restrictions for on-road fuels, neither biodiesel nor conventional diesel have significant combustion phase emissions compared with other life cycle stages. Sulfur oxide emissions from biodiesel are primarily from farm supply chains and biodiesel production. The emissions from farm supply chains are associated with the production of phosphorus fertilizer, which includes emissions from the production and use of sulfuric acid. GREET results for sulfur oxide emissions from biodiesel production are strongly influenced by the assumption that 28% of the process energy for soy oil extraction is from coal (the balance is primarily natural gas and electricity). Sulfur oxides from the diesel life cycle are primarily from petroleum supply chains, and to a lesser degree, petroleum refining.

Particulate matter emissions are similar for biodiesel and diesel, with the model showing a 7.5% increase in $PM_{2.5}$ and 5.1% increase in PM_{10} for biodiesel. As previously mentioned, this difference between LCA results should be interpreted with caution as it is within the uncertainty bounds of the model and subject to significant variability across supply chains for specific instances of these fuels. Over half of the particulate emissions in both cases are from fuel combustion. Emissions from farm supply chains, conversion, petroleum supply chains, and petroleum refining are primarily associated with combustion of various fuels.

Carbon monoxide results do not show a significant difference between biodiesel and conventional diesel. Fuel combustion is the dominant source of carbon monoxide emissions across the life cycle for both biodiesel and conventional diesel.

Nitrogen oxide emissions are similar to those for PM, with roughly half of the life cycle total from fuel combustion in the case of both biodiesel and conventional diesel. Contributions from other life cycle stages are primarily associated with combustion of fuels at various stages in the fuel supply chains.

Figure 8.13b shows the urban and non-urban emissions of each of the criteria air pollutants across the soy oil biodiesel and conventional diesel supply chains. Biodiesel results show lower overall urban emissions than conventional diesel for all substances except carbon monoxide for which the results are roughly equal. The life cycle urban emissions reduction associated with biodiesel compared with conventional diesel is 12% for VOC, 43% for SO_x, 16% for $PM_{2.5}$ and PM_{10}, and 6% for NO_x. While these differences are modest in most cases, they do suggest biodiesel may have the potential to reduce exposure to these criteria air pollutants along the supply chains of fuel production compared with conventional diesel.

8.5.2 Results from BEIOM

This analysis applied BEIOM v2.0, which harmonized U.S. national-level economic and environmental datasets for 2002–2017 in five-year time-steps (Avelino et al. (2021), Appendix F).[32] The results presented in this chapter are for air-related emissions and their impacts for four potential effects: *smog formation potential (SFP), acidification potential (ACP), PM exposure potential (PEP), and ozone depletion potential (ODP).* Other metrics are reported in BEIOM, and details of the analysis and assumptions are provided in Appendix F and in the peer reviewed literature (Avelino et al. (2021), Lamers et al., 2021). Results are presented in a single graph per biofuel and petroleum substitute (e.g., Figure 8.14 for corn ethanol vs. gasoline) and separately for the four potential effects (i.e., Figure 8.14 a, b, c, and d, for SFP, ACP, PEP, ODP, respectively). For each potential effect, there is a left and right panel. The left panel shows the contribution of the industries (i.e., all their products) to the U.S. national totals[33] for that effect (e.g., SFP) and year (e.g., 2017). The right panel shows the effect per megajoule (MJ) of fuel consumed basis and relativized to be 0–100%. Thus, the right panels are on a relative scale to the largest effect and year, and should not be interpreted as absolute percentages (e.g., an effect of 2 and 4 would be rescaled relative to the highest value of 4 to be 50% and 100%). Specifically, they show how the impacts from producing and consuming 1 MJ of fuel evolved over time accounting for coproduct benefits in the respective year. For the right panels and for comparison purposes, the year with the highest impact is used as the benchmark (100%) and the impacts of the other years are then shown as a relative comparison to that benchmark.

[32] The economic tables are released approximately every five years from the Bureau of Economic Analysis (BEA).
[33] National totals from production not accounting for emissions from households.

Comparing corn ethanol with gasoline, the total potential effects (left panels, Figure 8.14) on SFP, ACP, and ODP were smaller for corn ethanol than for gasoline[34] because there is more gasoline consumed in the United States than ethanol. However, per megajoule (right panels, Figure 8.14), potential effects from corn ethanol were higher than that of gasoline, as was reported in GREET (Figure 8.12). Total potential effects from corn ethanol were increasing through time as the size of the industry grew, although the per megajoule effects from corn ethanol were decreasing as the industry matured and efficiencies increased. The bulk of the emissions from corn ethanol for ACP and PEP were from farming, while the bulk of the emissions from SFP and ODP were from a combination of nonfarming parts of the supply chain.

Comparing soybean biodiesel with diesel, the total potential effects (left panels, Figure 8.15) were much lower from soybean biodiesel than from diesel because (as with corn ethanol) much less soybean biodiesel is consumed than diesel. Per megajoule (right panels, Figure 8.15), potential effects were larger for biodiesel compared with diesel as was reported in GREET; however, as with corn ethanol, per megajoule effects were decreasing through time as the industry matured and efficiencies improved.

[34] The exception is PEP which where the total potential effects from corn ethanol and gasoline were estimated to be comparable (Figure 8.14c). This was highly dependent on the inclusion or exclusion of tillage effects from the estimate (Avelino et al., 2021).

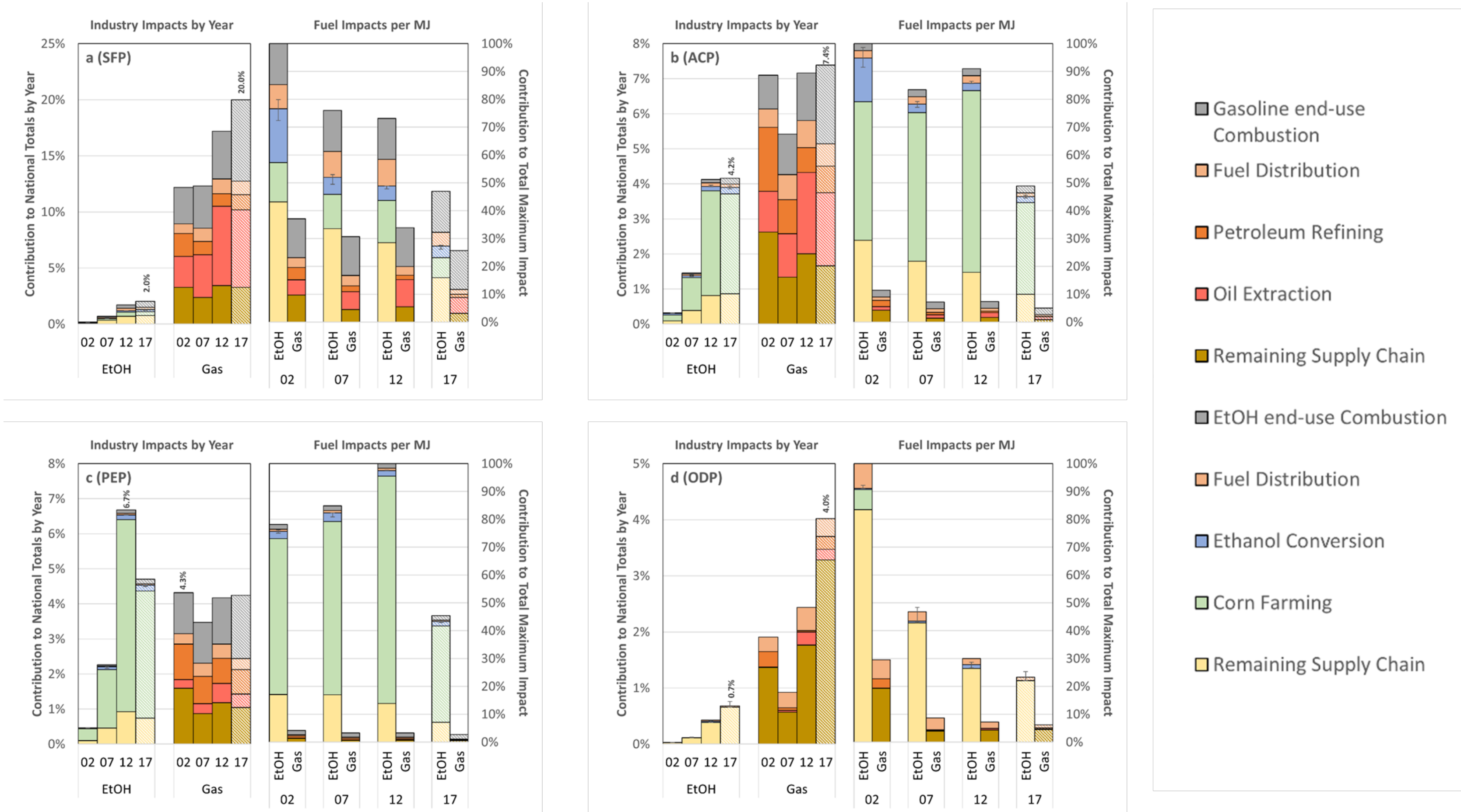


Figure 8.14. Comparisons of corn ethanol (EtOH) vs. gasoline (Gas) for smog formation potential (a, SFP), acidification potential (b, ACP), $PM_{2.5}$ exposure potential (c, PEP), and ozone depletion potential (d, ODP) from BEIOM.[35] Total industry contributions to total U.S. national emission level per year (left panel) and impacts per energy unit of fuel (right panels) for 2002, 2007, 2012, and 2017. Fuel Distribution is listed separately for ethanol and gasoline as both have a fuel distribution step. The results for 2017 are cross-hatched because they are partly based on 2012 data.[36]

[35] Greenhouse gas emissions are also provided by BEIOM, but are not in scope for this report series and so are not included here. See Chapter 2 Box 2.2 and Appendix F for additional information.
[36] Although all the land and emissions data for 2017 are based on 2017, BEA tables for 2017 are not yet available (est. 2022–2023).

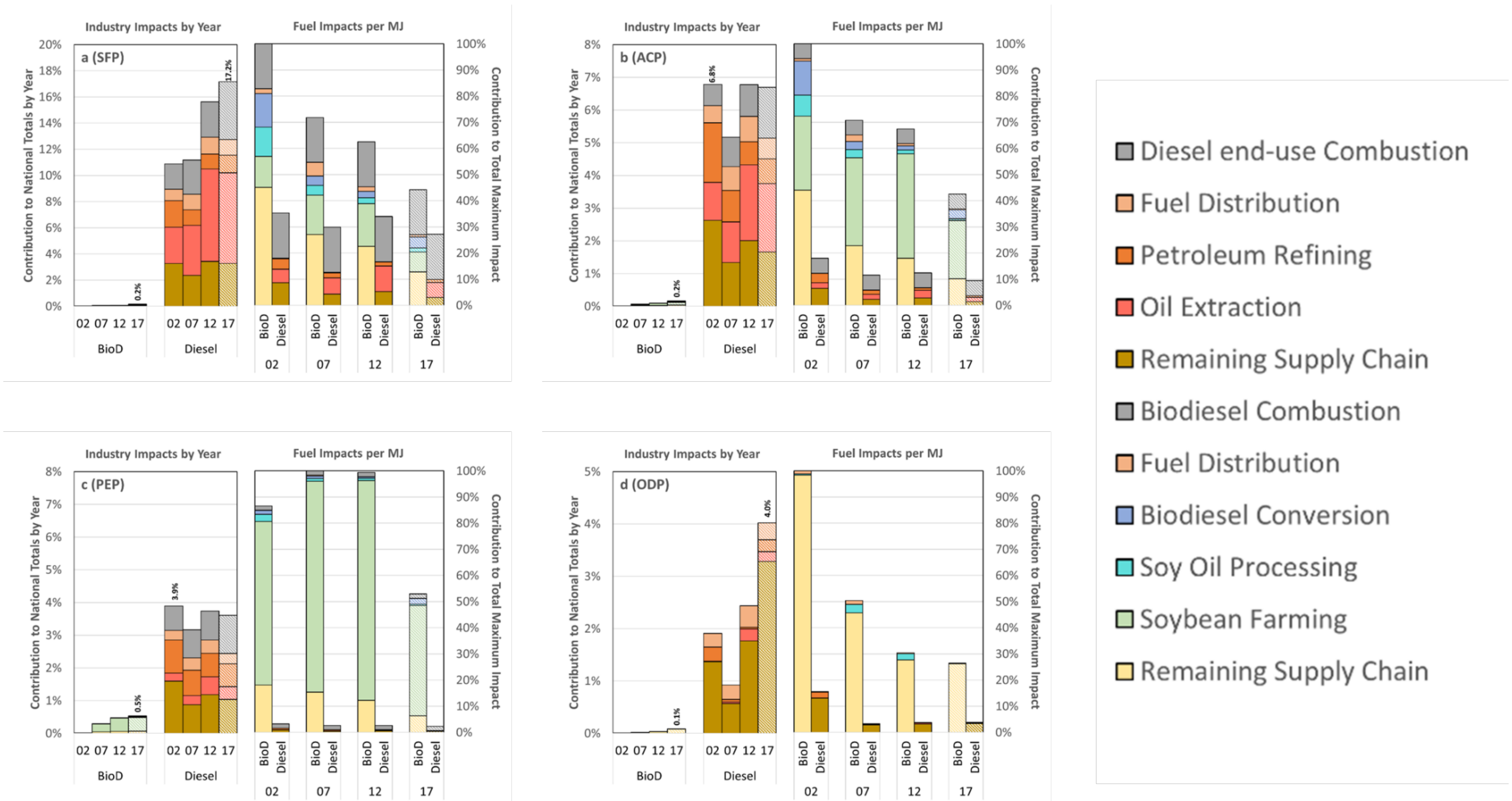


Figure 8.15. Comparisons of soybean biodiesel (BioD) vs. diesel (Diesel) for smog formation potential (a, SFP), acidification potential (b, ACP), $PM_{2.5}$ exposure potential (c, PEP), and ozone depletion potential (d, ODP) from BEIOM. Total industry contributions to total U.S. national emission level per year (left panel) and impacts per energy unit of fuel (right panels) for 2002, 2007, 2012, and 2017.[37] Fuel Distribution is listed separately for ethanol and gasoline as both have a fuel distribution step. The results for 2017 are cross-hatched because they are partly based on 2012 data.

[37] Although all the land and emissions data for 2017 are based on 2017, BEA tables for 2017 are not yet available (est. 2022–2023).

It needs to be acknowledged that both biofuels and fossil fuel counterparts rely on imported inputs, particularly crude oil. The BEIOM version used for this analysis is limited to domestic inter-sectoral linkages and does not incorporate international trade feedbacks.[38] Ongoing model expansions to detail regional effects within the United States and incorporate broader, international effects (through a multi-regional model) were not finalized in time to contribute to the RtC3. The U.S. economic boundary likely affects the results for both domestic corn ethanol and soybean biodiesel as well as their respective fossil substitutes. Performing a proxy estimation of the effects from international trade including foreign environmental releases and resource uses, it was assumed (conservatively) that foreign sectors pollute at the same rate as domestic sectors in 2017 (see Appendix F). While this way of incorporating international effects did not dramatically change the air quality–related impact metrics for biofuels, this modification did reduce the gap between biofuels and their fossil substitutes on a per megajoule basis, especially SFP and ACP. This suggests that an expansion to a multi-regional model can provide additional insights in future analyses. The robustness of such analyses however hinges on the robustness and coherency of international data for environmental releases reflecting the specific conditions in, for instance, crude oil exporting regions (e.g., Niger Delta).

BEIOM's results per megajoule tend to be similar to but slightly higher than those of GREET, even though BEIOM relies on process-level LCA data (from GREET among others) for the two biofuel pathways. This variation is due to the different system boundaries, as outlined in Figure 8.10 and Figure 8.11. In that regard, BEIOM's primary intent is not to provide detailed insights into the effects of specific plant-level supply chain activities. Rather, it aims to analyze fuel production in the context of the entire U.S. economy, providing a holistic estimate of the impacts of a specific industry (or product) including feedback effects from indirect activities occurring in sectors further away from the industry's supply chain in focus.

8.6 Horizon Scanning

Data on potential impacts from cellulosic feedstocks such as switchgrass and corn stover, and from algae, are very limited. The Billion Ton Study 2016 (BT16) was published in two volumes in July 2016 and January 2017, and included a detailed assessment of potential air quality impacts in 2040 assuming conservation practices were applied on a range of biofuels and feedstocks (DOE, 2017, 2016)

[38] The model uses the territorial principle in the environmental datasets and the residence principle in the economic datasets implying that all environmental releases and resource uses are restricted to U.S. territorial boundaries while economic activities encompass transactions from all "residents" (agents whose center of economic interest is the United States (see Horowitz and Planting (2009) for more details). As an example, emissions from a truck owned by a Canadian company transporting freight in the United States are recorded in the environmental data, but economic transactions are considered foreign trade for the national accounts.

(see Chapter 2 Box 2.1: "The 2016 Billion Ton Study"). Dedicated herbaceous energy crop production was estimated to have fewer air pollutant emissions than conventional crop production (e.g., corn grain), but more than an equivalent amount of agricultural residue production (e.g., corn stover, wheat straw). Comparing woody and herbaceous, woody biomass feedstocks generally have the fewest air emissions, with the exception of CO and SO_x. In addition, the U.S. Forest Service, in its 10-Year Strategy to Confront the Wildfire Crisis (USDA Forest Service, 2022), committed to increase forest restoration treatments on tens of millions of acres of national forests and private lands at high risk of wildfire. These treatments are expected to produce millions of tons of renewable waste biomass annually that could be used as feedstock for biofuels production. The BT16, however, was not an assessment of the likely future, but rather an aspirational target for bioenergy in the United States. More details on the potential air quality effects from biofuels from the BT16 can be found in DOE (2017).

Other studies have also examined the potential air quality impacts from biofuels not yet largely in production. Thakrar et al. (2018) concluded that biogenic emissions from switchgrass harvest are potentially large contributors to reduced air quality, and that NH_3 emissions associated with using urea fertilizer may have significant air quality induced health impacts. Chia et al. (2018) argue that production of biofuel from microalgae could reduce emissions of NO_x, SO_x, and metals relative to current market fuels. Ravi et al. (2018) modeled emissions and air quality impacts from a forest residue-based aviation biofuel supply chain in the U.S. Pacific Northwest, and concluded that air quality benefits from reduced slash burning (slash burning is the business-as-usual fate of forest residue) far outweigh any negative impacts from biomass hauling, biorefinery, and finished fuel transport activities. Use of unwanted forestry biomass would have a positive environmental benefit to forests, yet biofuel production from woody feedstocks is currently negligible.

In addition, renewable diesel production may increase in the future. In 2021, domestic production of renewable diesel was 838 million gallons at 16 facilities (Figure 8.16). There are currently significant expansions of renewable diesel underway by a number of major producers. This expansion could take advantage of cellulosic feedstocks, although life cycle emission impacts will depend on technology pathways used in its manufacture. Research on renewable diesel in California concluded that it reduced emissions of PM, NO_x, hydrocarbons, and CO (CalEPA, 2015; Na et al., 2015).

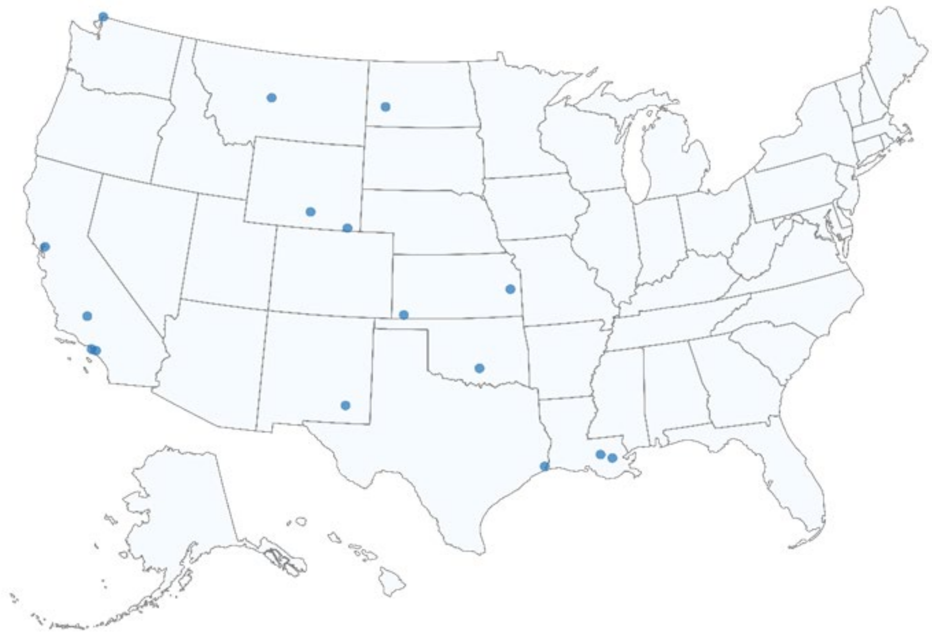


Figure 8.16. Location of Renewable Diesel Production Plants in the U.S., December 2022. Source: Gerveni et al. (2023).

8.7 Synthesis

8.7.1 Chapter Conclusions

- Increased corn production results in higher agricultural dust and NH_3 emissions from fertilizer use; however, improved nitrogen management practices can decrease these NH_3 emissions. Increased corn ethanol production also leads to increased NO_x, VOCs, $PM_{2.5}$, PM_{10}, and CO. Additional pollutant emissions result from evaporative losses of VOCs during storage and transport, as well as combustion emissions from commercial marine vessels, rail, tanker trucks, and pipeline pumps used to transport the ethanol to end use. Finally, the combustion of ethanol in end-use applications causes emissions of NO_x, VOCs, $PM_{2.5}$, and CO as well. As increased ethanol volumes are displacing petroleum and its related emissions in each of these areas, the overall impact on the environment is a complex issue.
- Emissions from production of biodiesel from soybean oil vary depending on the oil extraction method, with mechanical expelling the least efficient with highest emissions of NO_x, VOCs, CO, and $PM_{2.5}$, followed by hexane extraction and then enzyme-assisted aqueous extraction process (EAEP).
- EPA's "anti-backsliding" study examined the impacts on air quality from required renewable fuel volumes due to changes in vehicle and engine emissions resulting from required renewable fuel volumes under the Renewable Fuel Standard (RFS). Compared to the 2016

"pre-RFS" scenario, a 2016 "with-RFS" scenario increased ozone concentrations (eight-hour maximum average) across the Eastern United States and in some areas in the Western United States, with some decreases in localized areas. Concentrations of $PM_{2.5}$ were relatively unchanged in most areas, while NO_2 concentrations increased in many areas and CO decreased. Furthermore, increases in formaldehyde and acetaldehyde were widespread, while benzene and 1,3-butadiene levels went down. Other recent research addressing air quality impacts of biofuels is limited.

- Life cycle pollutant emissions from grease trap waste (GTW, a type of FOG) are dependent on lipid content, which varies considerably, although the mean percentage is toward the lower end, at about 4%. At low lipid contents, CO from GTW-derived diesel is much higher than soybean diesel and low-sulfur diesel when the methane produced is flared or when there is cogeneration. When considered without waste management, GTW-derived diesel performs better, with life cycle CO emissions similar to soybean diesel but much lower than from low-sulfur diesel. At high lipid contents, emissions of most criteria pollutants decrease compared to the soybean diesel, with larger decrease for PM and SO_x.
- A number of metals are present in biodiesel blends that can adversely affect emission control system performance. The magnitude of this impact remains a subject of research.
- Using the GREET model (Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation), lifecycle emissions from corn ethanol are generally higher than from gasoline for VOCs, SOx, $PM_{2.5}$, PM_{10}, and NOx. However, the location of emissions from biofuel production tends to be in more rural areas where there are fewer people. How this translates to effects on human health is complex, as it depends not only on the number of people, but on their demographics and vulnerability, as well as the dose-response relationship, which is pollutant-specific, among other factors.
- On a per unit energy basis over the period analyzed, biofuels manufacturing has a larger impact than their petroleum counterparts on smog formation, acidification, $PM_{2.5}$ exposure, and ozone depletion potentials, but a smaller potential effect in the total United States context due to the smaller size of the biofuels industry. Nonetheless, this conclusion needs to be interpreted in the context of each industry: while petroleum refining is a highly optimized, mature industry, biofuels are still reaching maturity as indicated in their emission profile over the 2002–2017 period. The observed trends seem to indicate that the biofuel industry is consistently reducing emissions as it matures. Significant uncertainties remain with these newer analytical approaches that deserve additional research.

- The likely future effects of the RFS Program are highly uncertain as of the time of writing due to many factors, thus the likely future effects on air quality are also highly uncertain.

8.7.2 Conclusions Compared to Prior Section 204 Reports

There is no new evidence that contradicts the fundamental conclusions of previous reports to Congress. Those conclusions emphasized that emissions of NO_x, SO_x, CO, VOCs, NH_3, $PM_{2.5}$, and PM_{10}, can be impacted at each stage of biofuel production, distribution, and usage. The impacts associated with feedstock and fuel production and distribution are important to consider when evaluating the air quality impacts of biofuel production and use, along with those associated with fuel usage.

8.7.3 Uncertainties and Limitations

- The understanding of the potential health effects of exposure to biofuels and emissions from vehicles using biofuels under real-world conditions, concentrations, and exposures including to susceptible human populations is limited. Recent literature that addresses cumulative impacts of upstream processes is limited. Much of the recent published literature focuses on impacts of individual sectors only.
- Vehicle technology continues to evolve and will likely affect responses to changes in fuel formulation. Only limited data exist on the impacts of biofuels on the tailpipe and evaporative emissions of light-duty Tier 3 vehicles and light-duty vehicles using advanced gasoline engine technologies to meet GHG emissions standards. In addition, only limited data are available for E85 and mid-level ethanol blends.
- Furthermore, understanding how fuel parameters, such as PMI, impacts emissions under different conditions continues to evolve.
- While the lipid content of GTW used to make diesel fuel can vary significantly, and impacts emissions, data on composition are limited.
- There is some recent research on air quality impacts for some cellulosic feedstocks, such as switchgrass and corn stover. However, data on where these potential impacts are likely to occur is very limited. Thus, potential impacts of increased market share are not well understood.

8.7.4 Research Recommendations

- Comprehensive studies of the quantitative impacts of biofuels on the emissions from advanced light-duty vehicle technologies (Tier 3) would improve the understanding of the potential for biofuel-specific pollutants and associated health impacts as new technologies enter the vehicle fleet. These studies should consider engine technologies being phased into use for compliance with current and future light-duty GHG standards, with a focus on vehicles compliant with the Federal Tier 3 or California LEV III criteria pollutant emissions standards currently under implementation. Such technologies would include engine downsizing with addition of turbocharging, gasoline direct injection, and non-traditional thermodynamic cycles such as Miller or Atkinson.

- Additional research and analyses are needed to adequately understand the potential health effects of exposure to biofuels and emissions from vehicles using biofuels under real-world conditions. It would be appropriate to study health effects in populations exposed to biodiesel and ethanol blends in "hotspots," such as fuel production sites, and those exposed to combustion products of biodiesel and ethanol blends, especially at high blend levels. Such studies could include drivers of vehicles utilizing those fuels.
- While population density is likely to differ in close proximity to petroleum versus biofuel production facilities, the magnitude of the difference in ambient levels, affected populations, and dose-response relationships is an issue that warrants additional research.
- More work is also needed to understand the net emissions of harvesting residues versus growing dedicated energy crops.
- Data and research on emissions from the use and production of advanced biofuels, like biogas, would be helpful in assessing the potential impacts of advanced biofuels in the future.

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9. Soil Quality

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Key Findings

- Impacts to date on soil quality from biofuels and the RFS Program are almost exclusively due to corn and soybean production for corn ethanol and soy biodiesel.
- Conversion of grasslands to corn and soybeans causes greater negative impacts to soil quality compared to growing these feedstocks on existing cropland. Simulations using the EPIC (Environmental Policy Integrated Climate) model found estimated grassland conversion to corn/soybeans from all causes generally increased soil erosion (-0.9-7.9%), and losses of soil nitrogen (1.2-3.7%) and soil organic carbon (SOC, 0.8-5.6%) in a 12-state, U.S. Midwestern region between 2008 and 2016. The range in losses depended upon the simulated tillage practices.
- Effects were not uniform across the 12-state region. Hotspots of grassland conversion and subsequent soil quality impacts occurred in locations such as southern Iowa and the Dakotas.
- A range of percentages (0–20%) was applied to the EPIC results to estimate the fraction of soil impacts attributable to grassland conversion estimated to be caused by the RFS Program. According to this estimation, the RFS Program increased erosion, nitrogen loss, and SOC loss from 0-1.6%, 0-0.7%, and 0-1.1%, respectively, across the 12-state region between 2008 and 2016. Notably, these modeling estimates represent a RFS-corn-ethanol effect only, and do not include any additional quantitative effect from the RFS Program on soybean biodiesel and soybean acreage as this effect could not be quantified in Chapter 7, nor do the estimates include any effect from crop switching on existing cropland.
- For context, the magnitude of these changes can be compared to the benefits of conservation programs, like the Conservation Reserve Program (CRP). The RFS-associated increase in nitrogen loss for this 12-state region, for example, represents up to 3.7% of the nitrogen retention benefits of the CRP for the entire United States.
- Additional conservation measures—such as further adoption of conservation tillage and cover crops—would help reduce the impacts of biofuels generally and the RFS Program specifically on soil quality.
- The likely future effects of the RFS Program are highly uncertain. While the projected cropland expansion for 2023–2025 is slightly larger than the estimated historical cropland expansion from the RFS Program, EPA cannot say with reasonable certainty that any particular terrestrial ecosystem or biodiversity will be impacted, due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes.

Chapter terms: Conservation Reserve Program (CRP), conservation tillage, conventional tillage, ecosystem services, no-till, soil health, soil organic matter (SOM), soil quality, tillage

9.1 Overview

9.1.1 Background

The production of biofuel feedstocks affects soil quality, primarily through the feedstock production stage (see Chapter 3 and Figure 1.12). The USDA Natural Resources Conservation Service (NRCS) defines soil quality as: "The capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation. In short, the capacity of the soil to function" (USDA, 2021). The term soil quality is often used interchangeably with soil health.[1] Healthy soils provide a suite of ecosystem services,[2] including carbon (C) sequestration, removing and storing C from the atmosphere, and the retention and infiltration of water, with the potential to reduce downstream flooding. The term soil quality in this chapter is used as a general term—it is used both to describe effects on single soil types and cumulative effects across large areas and multiple soil types. Soil conservation and soil environmental quality, listed separately in Section 204 of the 2007 Energy Independence and Security Act (EISA), are combined under this broader heading of soil quality (see Chapter 2, Table 2.3).

The EPA's 2011 and 2018 Reports to Congress (i.e., RtC1 and RtC2, respectively) focused on soil erosion, soil organic matter (SOM), and soil nutrients as general indicators of soil quality (U.S. EPA, 2018, 2011). Higher soil erosion is negatively related to soil quality since it preferentially removes the finest soil particles at the soil surface, generally higher in organic matter, plant nutrients, and water-holding capacity than the remaining soil. By contrast, higher SOM is a positive indicator of soil quality. It provides plant nutrients and water, promotes soil structure, and reduces erosion, while also sequestering C from the atmosphere and increasing the retention and infiltration of water (Sparks, 2003). Soil nutrients (e.g., nitrogen [N], phosphorus [P]) are necessary for plant growth. Too little of these nutrients can reduce crop yields, yet too much can lead to air quality impacts (e.g., ammonia emissions; see Chapter 8 on Air Quality), and water quality impacts via runoff or leaching (see Chapter 10 on Water Quality). This report (RtC3) also includes a new section on soil biological communities, relating these changes back to soil quality and ecosystem function where possible. As in past reports, it may be advantageous to add other soil quality indicators in the future, depending on the availability of scientific information and the needs of decision makers.

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[1] USDA's NRCS notes the following: "Soil health, also referred to as soil quality, is defined as the continued capacity of soil to function as a vital living ecosystem that sustains plants, animals, and humans" (USDA, 2022).
[2] Ecosystem goods and services, often shortened to ecosystem services, are the benefits that humans receive from nature. These benefits underpin almost every aspect of human well-being, including food and water, security, health, and economy (U.S. EPA, 2020).

As discussed earlier in the RtC3, the dominant biofuel feedstocks currently are corn grain for ethanol and soybeans for biodiesel (see Chapter 2 and 3). Thus, the "impacts to date" in the context of the requirements of this report under EISA are predominantly from these two biofuel feedstocks. Thus, this chapter focuses mostly on domestically produced corn and soybeans. In most cases, the effects of these crops in general and the effects from the amount of production attributable to the RFS Program could not be distinguished. Instead, a separate RFS Program attribution section (section 9.3.3) addresses this topic. Beyond corn and soybeans, two other biofuels are a focus of the RtC3: fats, oils, and greases (FOGs), and Brazilian sugarcane (see Chapter 2). FOGs generally do not affect soil quality and are not addressed in this chapter, and Brazilian sugarcane is addressed in Chapter 16 (International Impacts). The "horizon scanning" subsection later in this chapter focuses on possible future issues (see section 9.6) and briefly addresses the potential impacts of other, minor feedstocks.

9.1.2 Drivers of Change

The types of land converted, as well as production and conservation practices, are the major drivers of soil quality change within the feedstock production stage. The soil quality effects of corn and soybeans are generally more negative when they replace lands under perennial cover, such as grasslands—a process termed agricultural expansion or extensification—compared to cultivation on already existing cropland (U.S. EPA, 2018). The term grassland is used broadly in this chapter to include a spectrum of lands covered in grassy or herbaceous vegetation, from relatively unmanaged to heavily managed, including Conservation Reserve Program (CRP)[3] land in perennial grasses and pasture. Indeed, most of these grasslands were likely at one time cultivated, as agricultural expansion and abandonment has occurred for more than a century in many parts of the United States (Yu and Lu, 2018). In recent decades, a net agricultural expansion has occurred, with land in perennial cover, particularly grasslands, converted to actively managed cropland (see Chapter 5). Biofuel feedstock production and the RFS Program are estimated to be responsible for some of the non-cropland conversion to corn and soybeans (see Chapter 6 for more details). Finally, production and conservation practices alter soil quality, with the potential for both positive and negative outcomes. The effects of these drivers are discussed in greater detail in sections below using the scientific literature. Moreover, an agroecosystem model estimates the cumulative soil quality effects of cropland expansion and production practices.

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[3]The CRP is a program administered by the USDA Farm Service Agency. In exchange for a yearly rental payment, farmers enrolled in the program remove environmentally sensitive land from agricultural production, and plant species to improve environmental health and quality (USDA, 2020). It is a time-limited program (often a 10- or 15-year contract length); after the contract has expired, the land owner is no longer compensated for continued maintenance of the land cover and so the expired CRP acreage often reverts back to agricultural production.

9.1.3 Relationship with Other Chapters

Soils are entwined with all other parts of ecosystems, and so this chapter is also interrelated to other chapters in this report. As noted, land cover and land management (LCLM) change addressed in Chapter 5 is a major driver of soil quality effects. In turn, effects on soil quality can cause changes in other ecosystem components, particularly air and water. For instance, as already mentioned, greater application of fertilizers on soils can increase both ammonia emissions to the atmosphere and runoff or leaching of nutrients to water bodies. Air quality is addressed in Chapter 8 and water quality in Chapter 10. Lastly, this chapter addresses soil biological communities, rather than in the terrestrial ecosystem health and biodiversity chapter (Chapter 12), because the soil biota are often considered a part of soil quality or soil health (USDA, 2022).

9.1.4 Roadmap for the Chapter

Overall, this chapter on soil quality proceeds in the following manner: section 9.2 repeats the soil quality conclusions from the RtC2; section 9.3 updates the scientific literature on the impacts of biofuel feedstocks to date, and presents the modeling results; section 9.4 discusses likely future effects; section 9.5 provides a brief comparison to the soil quality effects of petroleum; section 9.6 considers the soil quality implications of other biofuel feedstocks in a horizon scanning section; and lastly, section 9.7 provides a synthesis of the chapter.

9.2 Conclusions from the 2018 Report to Congress (RtC2)

The following are direct quotes of the major, bulleted conclusions for soil quality in the RtC2 in 2018[4]:

- Corn grain ethanol and soy biodiesel account for most of the biofuel volumes produced to date. As a result, almost all the soil quality impacts from biofuels, thus far, are from the production of the dominant conventional feedstocks.
- Conversion of grasslands to annual cropland typically negatively affects soil quality, with increases in erosion, and the loss of soil nutrients and soil organic matter, including soil carbon. Impacts of this conversion can be partially mitigated—though not entirely—through the adoption of management practices such as conservation tillage.
- The soil quality impacts of converting other crops to corn or soybeans are generally less than those of the conversion of grasslands. The production of corn on existing cropland can provide soil carbon benefits, although these benefits are outweighed on a per area basis by the negative effects of grassland conversion.

[4]Found in sections 3.5.5. and 4.2.6 of the RtC2.

- Overall, these land use trends suggest that negative impacts to soil quality from biofuel feedstocks have increased since 2011, but this has not been quantified and the magnitude of effects depends predominantly on the relative areas of grasslands converted versus existing croplands attributable to biofuels.
- Corn stover is now being harvested at the commercial-scale in Iowa,[5] and the scientific literature indicates this must be done carefully to avoid negatively affecting soil quality and crop yields.

9.3 Impacts to Date for the Primary Biofuels

This section updates the potential soil quality effects of biofuel feedstock production. The section proceeds in four parts: first, it reviews the updated literature on the potential effects of biofuel feedstocks on soil quality (section 9.3.1); second, it presents the results of a new analysis, modeling the soil quality effects of cropland expansion (section 9.3.2; see also section 1.4); third, it discusses effects attributable to the RFS Program (section 9.3.3); and fourth, it reviews conservation practices used to reduce impacts or improve environmental outcomes (section 9.3.4).

9.3.1 Literature Review

The scientific literature was surveyed to determine whether it remained consistent with the above conclusions of the RtC2. This subsection proceeds by specific end point, starting with soil erosion.

9.3.1.1 Soil Erosion

The RtC2 conclusions on soil erosion generally still hold. Corn grain ethanol and soy biodiesel account for most of the biofuel volumes produced, so the impact of increased ethanol production on soil erosion is a function of two questions: (1) if corn or soybean production for ethanol or biodiesel displaced non-cropland or crops, does the corn or soybean production have more or less erosion than the previous LCLM?; and (2) have there been recent improvements in corn or soybean production for biofuels that have decreased soil erosion?

For question 1, there are two different cases based on the prior LCLM type. The first case is extensification where land in perennial cover, such as Conservation Reserve Program (CRP) land, has been converted to corn or soybean production for biofuel feedstocks. While uncertainties in methods used to estimate LCLM change can impact results (Dunn et al., 2017), it is clear that LCLM change has occurred (see Chapter 5). Since soil erosion is generally low for land in perennial cover (Nearing et al.,

[5] Commercial-scale harvesting of corn stover occurred at the time of the writing of the 2018 report, but has been subsequently halted (Bomgardner, 2019).

2017), conversion to corn or soybean will typically result in an increase in soil erosion (Yasarer et al., 2016). The erosion effects of conversion can be reduced if certain conservation measures, such as no-till and/or cover crops, are employed on the newly cultivated land (Lee et al., 1993) (see section 9.3.4).[6,7] This literature finding on conversion and no-till is also consistent with the modeling results presented in section 9.3.2.

The second case around question 1 above is crop switching, where existing cropland not in corn or soybean previously is converted to the production of these biofuel feedstocks. In addition to conversion of land in perennial cover, crop switching has also occurred, most notably with corn and soybean acreage increasing at the expense of wheat and cotton (see Chapter 5). The degree to which crop switching and changes in crop rotations, such as more years of corn in a corn-soybean rotation, impact erosion is a function of crop type, rotation, and the tillage and crop residue management practices used (Clay et al., 2019). In general, crop switching to biofuel feedstock production, such as converting cotton or wheat to corn or soybeans, results in a smaller increase in soil erosion than extensification or can even reduce erosion in the case of conversion to corn. Corn production results in more plant residues than cotton or wheat, and thus the more residue left on fields after harvest promotes erosion control/reduction, assuming the residue is not removed for other uses (Nelson et al., 2015).

Question 2 above relates to soil erosion during corn and soybean production. Erosion for these crops will be greatest when soil is bare and disturbed by tillage, which occurs during fallow times after harvest and before planting, during tillage operations, and after planting before crop maturity and greatest soil cover by crop biomass. Therefore, conservation practices to reduce erosion for corn and soybeans involve minimizing tillage, especially through adoption of conservation tillage practices, including no-till, and maximizing soil cover during fallow periods (Canales et al., 2018; Cassel et al., 1995). Conservation tillage practices were used on approximately 65% and 70% of corn and soybean acres in 2016 and 2012, respectively (see Chapter 3, section 3.2.1.3, (Claassen et al., 2018)). No-till was used at higher rates for soybeans (40%) than in corn (27%) (Claassen et al., 2018). Farmers may adopt these practices continuously or may vary tillage practices between years or between crops in a rotation within a given year (USDA NRCS, 2022a; Claassen et al., 2018). Primarily as a result of adopting conservation tillage

[6] Tillage is the mechanical disturbance of the soil, often in preparation for planting. Types of tillage practices can be defined in multiple ways. Conservation tillage is often defined as any tillage practice leaving at least 30% of the soil surface covered by crop residues; whereas conventional tillage leaves less than 15% of the ground covered by crop residues. No-till, a subset of conservation tillage, disturbs the soil marginally by cutting a narrow planting strip and surface residue is left primarily undisturbed. Mulch tillage and zone tillage, types of conservation tillage, are intermediate between no-till and conventional tillage. Tillage types can also be defined by Soil Tillage Intensity Rating (STIR) values, with higher values reflecting an increase in the amount of soil disturbance (for more information on tillage practices, see (USDA, 2006).

[7] The USDA's 2017 Census of Agriculture defines a cover crop in their survey as: "A crop planted primarily to manage soil fertility, soil quality, water, weeds, pests, diseases, or wildlife" (USDA, 2017).

practices, estimated water-produced erosion[8] decreased on cultivated cropland from 1.9 to 1.7 tons per acre per year between the periods of 2003–2006 and 2013–2016 (USDA NRCS, 2022b). Between these same time periods, total sediment losses from cultivated cropland dropped by 74 million tons or 22% (USDA NRCS, 2022b).

Planting cover crops is another conservation practice protecting the soil from erosion.[9] Cover crops may be planted before or following harvest of the primary crop, or managed by intercropping, when two or more crops are grown simultaneously within the same field (Geertsema et al., 2016). Examples of cover crop types include rye and clover. The prevalence of cover crops in the Midwest has increased from 2010 to 2015, but still only occur on 5% of farmland for all Midwestern regions and years surveyed, with only one exception.[10] These rates are similar to estimates of the national cover crop adoption rate (cover crops were planted on 5.1% of harvested cropland nationally in 2017) (Wallander et al., 2021). Beyond conservation tillage and cover crops, other management practices reducing erosion include terracing, grassed waterways, and prairie strips. More details on conservation practice trends are reported in Chapter 3, and the soil benefits of reduced tillage and cover crops are further discussed in section 9.3.4.

9.3.1.2 Soil Organic Matter

As reported in the RtC2, LCLM change converting perennial systems (e.g., grasslands) into corn or soybeans negatively impacts SOM. Most of the literature measures soil organic carbon (SOC), the largest fraction of SOM (ca. 52–58%). In a recent analysis, (Spawn et al., 2019) estimated that the conversion of perennial systems (i.e., grassland, shrubland, and wetland) to crops in the contiguous United States between 2008 and 2012 substantially decreased soil SOC stocks, releasing approximately 55.0 Mg C per ha to the atmosphere. In their analysis, grasslands were the predominant land cover type converted, and corn, wheat, and soybeans were the dominant crops planted on the new cropland (Spawn et al., 2019). Loss of SOC was the dominant (~90%) source of C lost to the atmosphere in these conversions. In a meta-analysis, (Qin et al., 2016) concluded that conversion of grassland to corn significantly decreased SOC by over 25% relative to the preceding grassland (Figure 9.1). They did not examine the effects of converting to soybeans, but such a conversion would likely reduce SOM even more. This is because soybeans generally result in less soil C accrual compared to corn because of reduced plant biomass, while corn-soybean rotations are typically intermediate (Varvel, 1994).

[8] This is to distinguish between wind- and water-induced erosion, both of which may be important to varying degrees in different parts of the country((USDA NRCS, 2022b).
[9] See (USDA NRCS, 2022b).
[10] These results are from (Baranski et al., 2018). Midwestern regions included the Northern Plains, Corn Belt, Southern Plains, and Lake States. Cover crops were used on approximately 6% of farmland in the Southern Plains USDA region in 2015.

The impacts on SOM from conversion of perennial grass to cropland is also dependent on the type of tillage employed. Tillage practices protecting the soil from erosion (e.g., no-till management) typically will retain more SOM, including C, especially near the surface (West and Post, 2002). In contrast, traditional tillage redistributes SOM to lower portions of the soil, and some studies have found that traditional tillage maintains more total C compared with conservation tillage practices when greater soil depths are included (e.g., Blanco-Canqui and Lal (2008)). Generally, however, the use of conservation tillage practices is thought to at least partially mitigate the effects of converting grasslands to corn or soybeans on SOM, in addition to reducing erosion (Leduc et al., 2017; Gelfand et al., 2011; Follett et al., 2009). For example, (Follett et al., 2009) did not observe a change in total SOC when grasslands (in perennial cover for ca. 12 years)were replaced with corn under no-till management. In combination with no-till, the use of cover crops would also likely reduce the effects of conversion on SOM (see section 9.3.4).

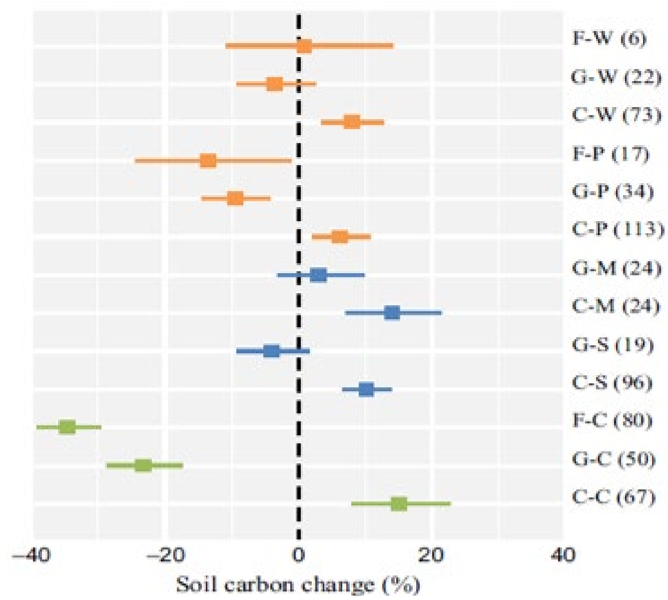

Figure 9.1. Percent soil carbon change in response to land cover changes. The estimates show response ratio (% change of initial control) for land use changes from cropland (C), grassland (G), and forest (F) to corn (C), switchgrass (S), Miscanthus (M), poplar (P), and willow (W), irrespective of soil depth and time horizon. Studies reporting corn residue removals were not included. Number of datasets is shown in parenthesis. Error bars represent 95% confidence intervals. Source: Qin et al. (2016) (used with permission).

In contrast to grassland conversion, switching from other crops to corn or soybeans on current cropland are likely to have more positive effects on SOM. In the same meta-analysis where grassland-to-corn decreased SOC, converting other crops (e.g., wheat, soybeans) on existing cropland to corn significantly increased SOC by approximately 20% relative to the previous cropland between 5 and 10 years post-conversion (Figure 9.1; Qin et al., 2016). Switching to soybeans or corn-soybean is likely to result in less soil C accrual relative to continuous corn unless the soybeans are grown in more complex rotations (Varvel, 1994). Thus, the impacts to date on SOM generally depend on the relative amounts of conversion of grassland to corn and soybeans versus conversion of other crops to corn and soybeans, and the management employed before and after conversion. (Spawn et al., 2019) and the modeling presented below (section 9.3.2) provide estimates of the soil C loss from grasslands to cropland from 2008 to 2012 and 2016, respectively, and estimates the fraction of soil quality effects from this LCLM attributable to the RFS Program (section 9.3.3). A full accounting, however, including cumulative soil quality estimates from the conversion of other crops to corn and soybeans, for biofuels in general and the RFS Program specifically, does not exist yet to the authors' knowledge.

9.3.1.3 Soil Nutrients

In general, the above discussion of soil erosion and SOM also applies to soil nutrient impacts. Converting perennial grasslands to corn or soybeans increases fertilizer inputs to the soil, particularly for corn. Although lesser in magnitude, replacing other crops on existing cropland with corn also generally increases fertilizer usage (see Chapter 3, Figure 3.13). For example, 149 pounds of N per acre were applied to corn in the United States on average in 2018, versus 94, 78, and 17 pounds per acre for cotton, wheat, and soybeans, respectively (USDA, 2019). In contrast to corn, farmers use far less N fertilizer for soybeans since symbiotic bacteria in its roots can fix N from the atmosphere. Phosphorus inputs are more comparable among the crops, with corn receiving an average of 69 pounds of phosphate per acre in 2018, versus 45, 34, and 55 pounds of phosphate per acre for cotton, wheat, and soybean, respectively (USDA, 2019) (see Chapter 3 section 3.2.1.6). Thus, in general, replacing wheat or cotton with corn increases both N and P fertilizer inputs, while replacing wheat or cotton with soybeans decreases N and increases P. Since most soybean is grown in rotation with corn, however, ultimately there is an increase in both N and P inputs with a conversion to a corn-soy rotation.

Greater nutrient inputs can improve crop yield and SOM accrual, but also increase the risk of nutrient loss to the environment through emissions to the atmosphere, soil erosion, runoff, or leaching to groundwater (Yasarer et al., 2016). A major pathway of nutrient loss is soil erosion, so any LCLM change increasing soil erosion, or, conversely, management practices reducing erosion will also generally affect nutrient loss. In general, higher percentages of cultivated land are correlated with greater amounts of nutrient loss (Piske and Peterson, 2020). Within cultivated crops, nutrient losses from the soil are partially a function of nutrient inputs minus biological demand. Past studies have indicated N loss to be higher in corn fields versus soybeans (Powers, 2007)), although a recent analysis concluded that N loss from corn fields can be less than that of soybeans, in part because of greater biomass and N demand of the corn plant (Piske and Peterson, 2020).

9.3.1.4 *Soil Organisms*[11]

The soil is a dynamic ecosystem, full of organisms affecting soil health. These organisms include microorganisms, such as bacteria, and larger organisms, such as earthworms, mites, and a variety of other arthropods including insects and their larvae. Among the three key indicators of soil quality discussed above—soil erosion, SOM, and soil nutrients—the latter two are regulated in part by the community of soil organisms. They both redistribute, modify, and decompose SOM and, in doing so, release nutrients, such as N and P, potentially for plant growth and development.

Recent studies have investigated the response of soil communities to the expansion of agriculture onto perennial grasslands—though these studies typically address responses to agricultural conversion in general and not to corn or soybeans specifically. Comparing different biofuel cropping systems (i.e., corn and perennial energy crops) and native grasses (prairie), greater microbial biomass has been observed in prairie soils and under perennial energy crops (i.e., switchgrass) than under corn, with differences in biomass accompanied by shifts in microbial diversity and structure (Zahorec et al., 2022; Liang et al., 2012). Similarly, densities and community structure of soil arthropods are typically lower under single-crop dominance (e.g., corn) compared to perennial crops or single-crop fields with a greater abundance and cover of weeds or crop residues (Norris et al., 2016; Schrama et al., 2016; Scheunemann et al., 2015). Compared to annual crops, perennial plants generally have deeper rooting depths and higher root densities that support more complex soil food webs and a larger population of beneficial organisms such as arbuscular mycorrhizal fungi (Duchene et al., 2020; Jesus et al., 2016; Boerner, 1992). These fungi enhance storage of soil C (McGowan et al., 2019), while facilitating soil water and nutrient uptake by plant roots (Khalvati et al., 2005). These findings suggest conversion from grassland to corn or soybeans is likely to lower soil community abundance, biomass, and diversity, and consequently negatively affect soil quality.

Crop management practices commonly used for corn and soybean production also significantly affect soil communities. Tillage alters soil community diversity and structure as some organisms are more sensitive to soil disturbance than others (Adams et al., 2017; Coulibaly et al., 2017; Norris et al., 2016; van Groenigen et al., 2010). Tillage elicits a strong response from the soil arthropod community. As tilling frequency is reduced, diversity of soil taxa increases (Coulibaly et al., 2017), including under corn and soybeans (Adams et al., 2017; Norris et al., 2016). Likewise, fertilization can alter the community composition of soil organisms, with microbial communities shifting away from fungal- and toward bacterial-dominance (Jia et al., 2020; Leff et al., 2015; Bradley et al., 2006; Frey et al., 2004). Excessive

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[11]As noted previously, the RtC1 and RtC2 did not address soil organisms as an indicator of soil quality. Because of this, a comparatively longer discussion of the literature for this topic is provided here than for some of the other indicators, like nutrients, addressed in previous reports.

N fertilizer may enhance microbial respiration by eliminating N limitation on microbial growth (Russell et al., 2009), increasing the susceptibility of SOM to microbial decomposition (Singh, 2018). This could result both in reduced SOM and increased CO_2 emitted to the atmosphere. Larger organisms of the soil community have variable responses to fertilizer and those responses depend upon factors such as fertilizer form or the identity of the dominant plants in the system (Coulibaly et al., 2017; Postma-Blaauw et al., 2010; Lindberg and Persson, 2004).

Similarly, the effects of pesticides elicit variable responses. In a global review, (Bünemann et al., 2006) examined responses to the type (herbicide vs. insecticide vs. fungicide) and active chemical of the pesticide. In general, they found that herbicides had effects on microbial enzyme activity; insecticides reduced larger organism density and reproduction rates, with variable effects on microbial abundance and activity; and fungicides had the greatest negative effects on soil organisms—particularly beneficial soil fungi and earthworm populations.

9.3.2 New Analysis

As noted in the RtC2 and the literature review in section 9.3.1, the expansion of corn and soybeans onto grasslands generally causes greater change and negative effects on soil quality than production on existing cropland. The RtC2, however, lacked spatially explicit information on where the conversions may be occurring, estimates of the cumulative soil quality effects from these conversions, and information on the fraction of those changes potentially attributable to the RFS Program. In response, a simulation study was conducted to support the RtC3, using the EPIC (i.e., Environmental Policy Integrated Climate) model, of the effects of grassland conversion to corn and soybeans between 2008 and 2016 across 12 Midwestern states (Figure 9.2). These 12 states account for approximately 80% of U.S. corn production and soybean production (USDA, 2020). The EPIC simulations leveraged recently available estimates of LCLM change occurring from all causes (Lark et al., 2020), and some of the central methods and findings are presented here as well as published in (Zhang et al., 2021). Discussion of the fraction of the changes attributable to the RFS Program are discussed in section 9.3.3.

EPIC is a widely used agroecosystem model (e.g., (Williams et al., 2008; Izaurralde et al., 2006) capable of simulating the effects of corn and soybean expansion onto grasslands. Field-level estimates (at a 30 m resolution) of land converted from predominantly grass cover to cropland between 2008 and 2016 across the 12 Midwestern states were employed as input for the model. These estimates of conversion are from (Lark et al., 2020), a detailed assessment based on the USDA Cropland Datalayer (CDL), using similar methods as employed in (Lark et al., 2015) and (Wright et al., 2017). There is debate over these approaches (Dunn et al., 2017), but they remain the best estimates to date for LCLM change at the fine scale required for EPIC (see (Dunn et al., 2017) and (Lark et al., 2021)) for further discussion of these

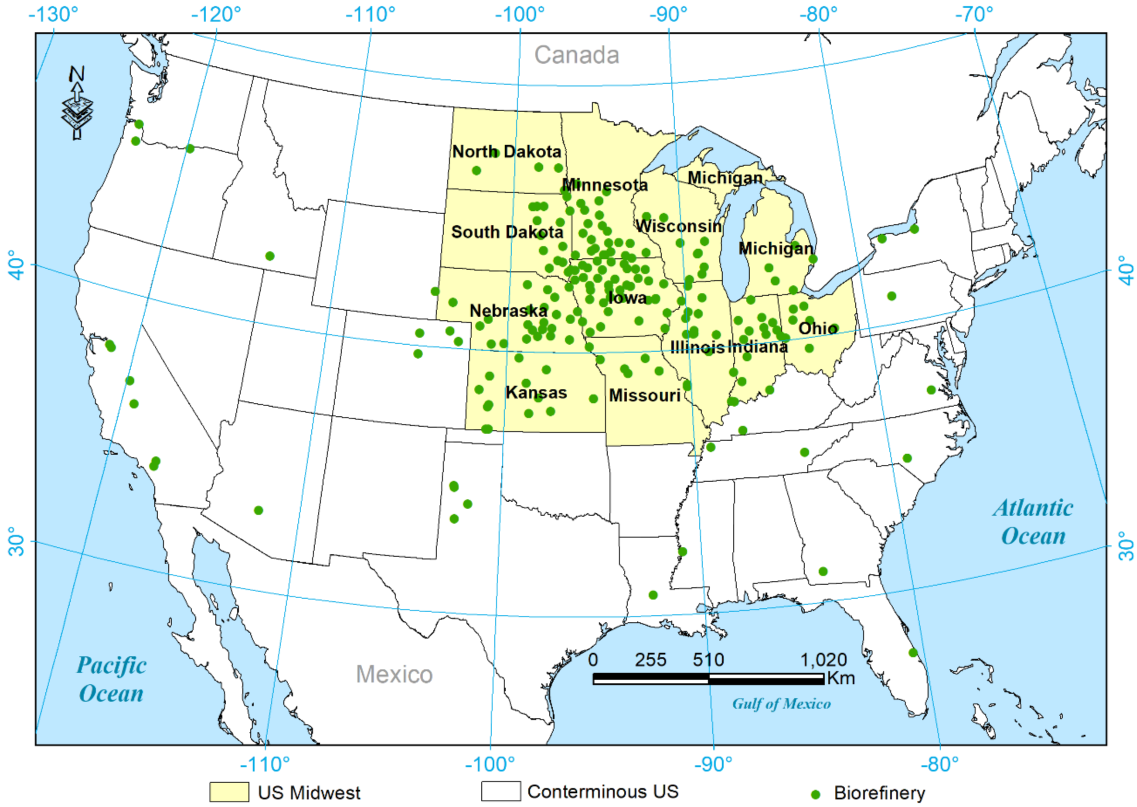
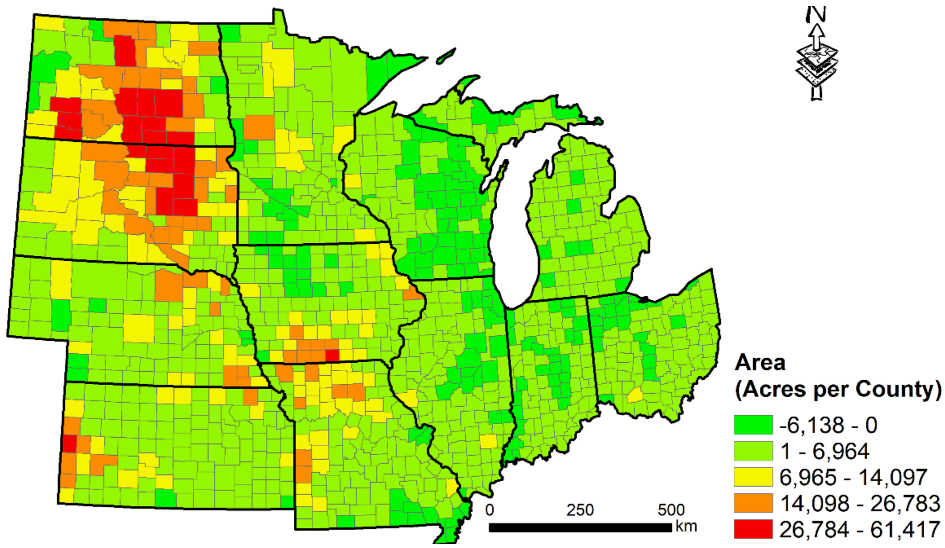


Figure 9.2. Map of the continental United States with 12 Midwestern states outlined. These 12 states constituted the area of modeling for this chapter. Green dots represent locations of U.S. biorefineries (RFA, 2017). Source: (Zhang et al., 2021; Zhang et al., 2015) (used with permission).

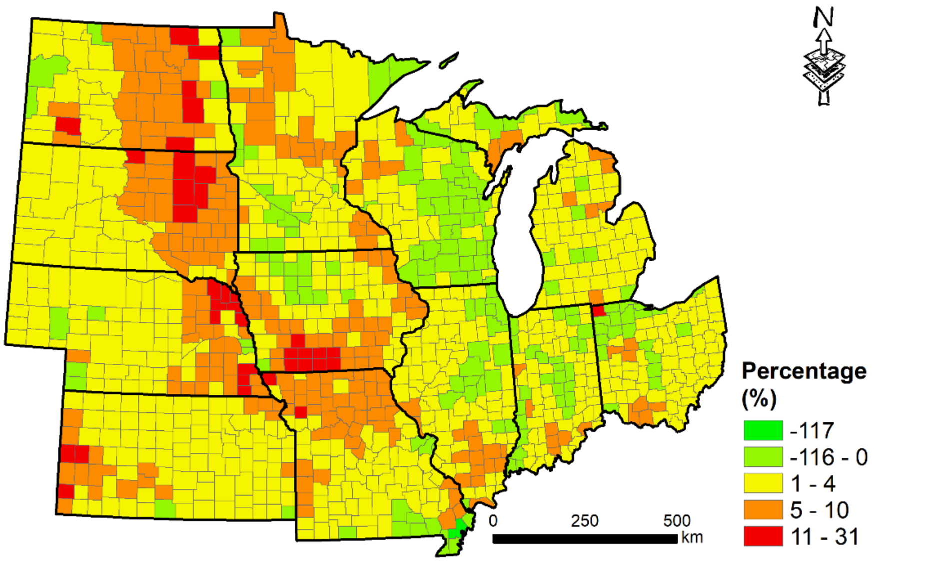
CDL-based estimates). Field-level data on "abandoned" lands, those returning to grass cover from row crops during the same period, were also used. Combined, this provided the estimated net conversion of grassland to cropland with results by county for this 12-state area (Figure 9.3). The field-level estimates were used to simulate the effects of grassland conversion to a corn/soybean rotation on soil erosion, soil N and P loss, and SOC loss.[12] As noted above, SOC is a subset of SOM and often is used an indicator of SOM dynamics. Since tillage has a large impact on soil quality, conversion both under conventional tillage and under no-till management were also simulated. For abandoned lands, the effects of changing from a corn/soybean rotation under conventional tillage to grass cover were simulated. Combined, this yielded an estimate of the net effect of agricultural conversion and abandonment on soil quality (for more details on the methods, see (Zhang et al., 2021)).

Overall, according to data from (Lark et al., 2020), approximately 5 million acres (2 million hectares) of grassland across the 12-state region were converted to crops during this time period (2008 to 2016), and ca. 838,000 acres (339,000 hectares) were abandoned. This is roughly half of the net grassland

[12] Soil N and P loss was estimated as losses from fields through erosion, runoff, and leaching. SOC loss was estimated as losses from fields through emissions to the atmosphere and through runoff, erosion, and leaching.



(a)

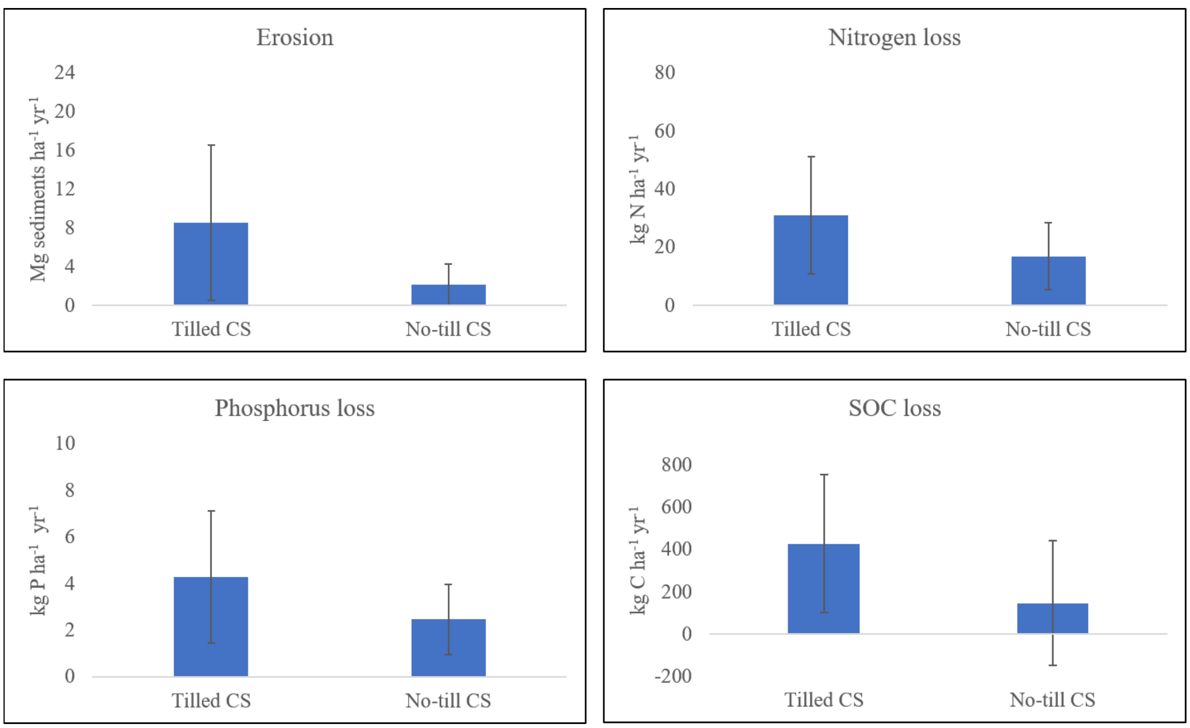


(b)

Figure 9.3. Estimated area (a) and percentage (b) of net conversion of grassland by county in the U.S. Midwest between 2008 and 2016. Net conversion is the sum of grassland conversion to crops minus the abandonment of crops to grassland. Percentage is area of net conversion divided by the total grassland area in that county multiplied by 100. Negative numbers indicate net abandonment of cropland to non-cropland, while positive numbers indicate net conversion of non-cropland to cropland. Data from Lark et al. (2020) and figure from Zhang et al. (2021) (Creative Commons license, https://creativecommons.org/licenses/by/4.0/; figure modified).

conversion estimated across the entire lower 48 from Lark et al. (2020). Thus, in net, approximately 4.2 million acres (1.7 million hectares) (or 2.8%) of the total 151 million acres (61 million hectares) of grassland converted in the region (Figure 9.3a). Annually over the entire period this is a net expansion of roughly 0.5 million acres per year. Expressed as a percentage, the net grassland converted in a county ranged from -117% to 31%, with notable hotspots of conversion in southern Iowa, eastern Dakotas, and western Kansas (Figure 9.3b). Averaging across land capability classes (LCCs) for all converted grasslands yielded an LCC of 3.2.[13] Thus, the grasslands converted were generally not prime farmland (LCC: 1-2), consistent with earlier studies (Lark et al., 2015), and this likely increased environmental effects per unit of land converted.

These simulations found that the soil impacts of the net conversion of grasslands depended greatly upon tillage management assumed, both on a per area basis (Figure 9.4) and in total (Table 9.1). The assumption of no-till in the simulations reduced—but, in most cases did not eliminate—soil quality impacts (Figure 9.4). This is also consistent with the literature discussed in the previous section. In total, the net conversion of grassland to and from conventionally tilled corn-soybeans increased erosion and N, P, and SOC loss (Table 9.1). By contrast, net effects were lower under conversion of grasslands to no-till corn-soybeans (Table 9.1). Overall, the effects of conversion to tilled corn-soybeans likely provide an upper bound of effects, whereas the no-till scenario provides a lower bound, with actual effects likely somewhere in between.



ha = hectares; kg = kilograms; Mg = megagrams; yr = years

Figure 9.4. Simulated soil quality effects of replacing grassland with conventional tillage vs no-till corn-soybean (CS) rotation. Simulations conducted using the EPIC (Environmental Policy Integrated Climate) model. Bars represent mean values across all converted fields within the 12-state region. Whiskers represent ±1 standard deviation from the mean value. Note: negative SOC values reflect soil C accrual. Source: Zhang et al. (2021) (Creative Commons license, https://creativecommons.org/licenses/by/4.0/; modified figure).

[13]The USDA-NRCS classifies land by the capability of it to produce crops, with the higher the number (1–8, unitless) indicating less capable land. Class 1 soils have slight limitations to their use and are considered the best producing lands; Class 2 have moderate limitations; Class 3 have severe limitations; and Class 4 have very severe limitations, with each increase in class requiring greater conservation practices to reduce impacts.

Table 9.1. Simulated soil quality effects of net grassland conversion (conversion minus abandonment) to and from corn-soybeans (CS) under two different tillage scenarios across 12 Midwestern states from 2008 to 2016. Simulations conducted using the EPIC (Environmental Policy Integrated Climate) model. In the first tillage scenario (#1), grasslands converted to *no-till* CS, and *tilled* CS abandoned to grasslands. In the second tillage scenario (#2), grasslands converted to *tilled* CS, and *tilled* CS abandoned to grasslands. Results presented as a range between scenario #1 and #2. Values reflect the simulated impacts summed across all converted and abandoned parcels within the 12-state region.

Soil Quality Metric	Erosion/Sedimentation[c]	Total N Loss	Total P Loss	Total SOC Loss
Total net impact over 12-state area (Tillage Scenario #1-2)	−1.4–11.8 (Tg/yr)	14.9–44.0 (Gg N/yr)	1.1–4.8 (Gg P/yr)	99.8–673.8 (Gg C/yr)
Relative amount compared to U.S. Midwest cropland [a]	−0.9–7.9%	1.2–3.7%	N/A	0.8–5.6%
Relative amount compared to CRP benefits for entire U.S. [b]	−0.8–6.8%	6.3–18.6%	2.4–10.3%	1.1–7.3%

Gg = gigagrams; Tg = teragrams; yr = year

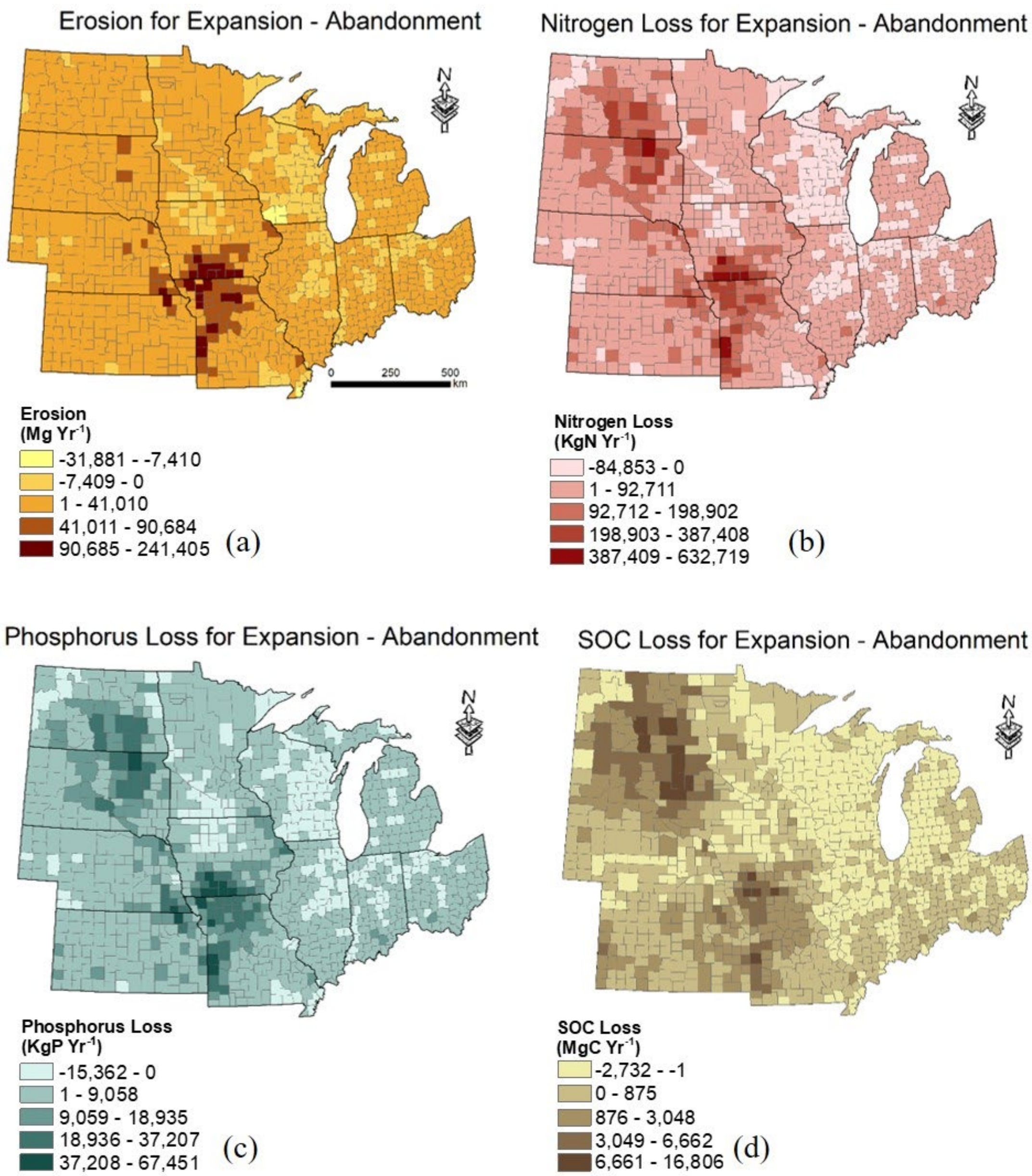
[a] Relative amount is calculated by comparing with the estimate soil erosion (150 Tg sediment/yr) (Zhang et al., 2015), N loss (1,200 Gg N/yr) (Zhang et al., 2015), and SOC loss (12,000 Gg C/yr) (West et al., 2008) from the cultivated cropland in the U.S. Midwest.

[b] Relative to the environmental benefits of Conservation Reserve Program (CRP) for the United States in 2017, estimated in https://www.fsa.usda.gov/Assets/USDA-FSA-Public/usdafiles/EPAS/natural-resouces-analysis/nra-landing-index/2017-files/Environmental_Benefits_of_the_US_CRP_2017_draft.pdf (accessed 6/7/2020). Note: EPIC estimates of erosion and N and P loss are compared to CRP estimates of sediment, N, and P not leaving field or intercepted by buffers. The EPIC estimate of SOC loss is compared to the CRP estimate of CO_2 equivalents sequestered. It was not compared to the C benefits of reduced fuel and fertilizer use, which was not included in the EPIC modeling.

[c] Negative erosion values denote an overall decline in erosion under tillage scenario 1, whereas simulated erosion increased under scenario 2.

Compared to existing cropland in the 12 states, the soil quality impacts of net conversion ranged from a slight improvement in erosion under no-till to an almost 8% increase under conventional tillage, and slightly smaller ranges for total N and total SOC loss (Table 9.1). Thus, net conversion of grassland from all causes over this 12-state area was estimated to increase soil erosion for most scenarios (-0.9 to +7.9%), and to increase nitrogen loss (+1.2 to 3.7%) and SOC loss (+0.8 to 5.6%) for all scenarios (Table 9.1). These effects can also be compared to the benefits of the entire CRP for the United States. This is for context only since the benefits provided by CRP lands and impacts of conversion do not necessarily overlap in time and space. In magnitude, the negative effects of this net grassland conversion under conventional tillage represents an offsetting of approximately 10% to almost 20% of the nutrient (N and P) retention benefits and approximately 7% of the sediment and SOC retention benefits of the entire U.S. CRP (Table 9.1).

Not surprisingly, the spatial pattern of soil quality effects followed the spatial pattern of grassland conversions in the 12 states (Figure 9.5a-d). The net effect on soil erosion (Figure 9.5a) was highest in southern Iowa, likely because of combination of higher rates of conversion and soils on steeper slopes. In comparison, the eastern Dakotas experienced less erosion despite similar acreages of conversion, likely because of the flatter terrain. Nutrient losses (both N and P; Figure 9.5b, c, respectively) from runoff and leaching also followed the pattern of net grassland conversion, as did changes in SOC (Figure 9.5d).



Kg = kilograms; Mg = megagrams; Yr = years

Figure 9.5a-d. Simulated erosion (a), nitrogen (b), phosphorus (c), and soil organic carbon (SOC) (d) loss from net grassland conversion (conversion minus abandonment) to and from corn-soybean rotations with conventional tillage across the 12 Midwestern states. Simulations conducted using the EPIC (Environmental Policy Integrated Climate) model. Results aggregated by county. Note: negative SOC values reflect soil C accrual. Source: Zhang et al. (2021) (Creative Commons license, https://creativecommons.org/licenses/by/4.0/; no changes made).

Several points should be considered when assessing these modeling results. First, the converted and abandoned lands were not specifically related to biofuel feedstock production, but rather agriculture in general. Hence, the results should be viewed as the soil quality effects of general agricultural expansion across the Upper Midwest from all causes from 2008–2016, while the effects attributable to the RFS Program specifically are a proportion of the total shown (see section 9.3.3). Second, crop and tillage types for each specific parcel converted could not be modeled due to data and computational limitations. Therefore, a general scenario of conversion to and abandonment from corn and soybean rotations was used. The focus on corn and soybeans is because they are the dominant biofuel feedstocks currently, the dominant crop rotation in the region ((Sahajpal et al., 2014)), and were also the most prevalent crops planted on converted grasslands between 2008 and 2016. Almost 60% of acres converted nationally were planted with corn (29.3%) and soybeans (26.7%) (Lark et al., 2020), and this percentage was over 70% for the area modeled. Likewise, an exact tillage type could not be applied to each field since this information at this level is not currently available. Moreover, as noted above, producers sometimes change between tillage practices over multiple years or change tillage practices between crops in rotation (USDA NRCS, 2022b; Claassen et al., 2018). In this current study, the spectrum of tillage is bracketed by assuming no-till or conventional tillage in all converted lands, representing the two extremes in tillage practices, with the actual effects in this region likely in between these end points. Third, the grassland parcels represented a spectrum of grassland and management types, including pasture, lands managed for hay, and CRP grasslands, not necessarily solely undisturbed or unmanaged grasslands prior to conversion. Fourth, EPIC simulates "edge-of-field" results, meaning in this case, it only simulates gains or losses on the converted or abandoned parcels. The model does not have a landscape routing function to stream or river networks. Soil, N, P, and SOC lost from the agricultural fields or parcels may or may not end up in waterways. They instead may be retained, at least temporarily, in other locations in the landscape (e.g., by buffer strips or forested riparian zones). Thus, soil quality effects are clear, but comparable water quality impacts cannot be directly assumed. Rather, the water quality effects of grassland conversions are presented in the water quality chapter of this report using a different model (i.e., Soil & Water Assessment Tool [SWAT], see Chapter 10).

In a last consideration, a recent study concluded that updating plant trait parameters in models to better represent recent changes, such as improvements in crop yields, can increase modeling performance in simulating environmental impacts of agroecosystems (Ren et al., 2022). In Zhang et al. (2021), the parameters currently available in EPIC were used, but a review and potential update of plant trait parameters, including crop yields, may be warranted in the future (see also section 9.7.4). Thus overall, these simulations are not intended to represent exactly "what happened" across the Midwest from 2008 to

2016. Rather, they provide the directionality of effects (whether negative, positive, or no effect), and a range of estimated effects, with the actual effect likely in between these estimates.

9.3.3 Attribution to the RFS

The chapter material above addressed the soil quality effects of corn and soybean production in general, but not impacts from the RFS Program specifically. For instance, in the review of the literature (section 9.3.1), studies generally did not examine how corn or soybean production attributable to the RFS Program affected soil quality, but instead focused on the effects of corn and soybeans in general. Likewise, in the soil modeling analysis above (section 9.3.2), the effects of grassland conversion to corn and soybeans were simulated regardless of end use. This section addresses potential effects of the RFS Program on soil quality to the extent possible, building from the information presented above and in Chapter 6.[14]

A recent study by (Lark et al., 2022) attributed increases in erosion and soil nutrient loss to the RFS Program between 2008 and 2016. Using a modeling approach, they concluded the RFS increased soil erosion by 4.7% above a non-RFS scenario, and that nitrate leaching and P runoff increased by 5.3% and 3.2%, respectively, above the same non-RFS scenario. Most of the effects centered in the U.S. Midwest. These effects were driven by an increase of total cropland by 5.2 million acres and an increase of corn acreage by 6.1 million acres estimated attributable to the RFS, according to their analysis. These estimates are approximately double the estimates attributable to the RFS made in this report for the same period (0 to 3.5 million acres of additional corn and 0 to 1.9 million acres of additional cropland) because of several underlying assumptions made by (Lark et al., 2022) which increased the estimated effect of the RFS Program (see Chapter 6, section 6.3.3). However, once the estimates from Lark et al. (2022) are rescaled to account for attributional differences, the estimates to the land are very similar to ours (see Chapter 6, section 6.4.3).

The attributional estimates presented in this report can be combined with the soil modeling results above (in section 9.3.2) to estimate the soil quality impacts of the RFS Program. As noted in previous sections, the production of corn and soybeans can affect soil quality through the expansion of these crops onto former grasslands; the switching of other crops to corn and soybeans on current cropland; and the mix of production and conservation practices on corn and soybean acreage. Regarding expansion onto grasslands, the analysis in Chapter 6 estimated that 0 to 1.9 million acres of additional cropland were associated with corn ethanol attributable to the RFS Program between 2008 and 2016, or approximately 0 to 20% of the observed net increase in U.S. crop area over this period (see Chapter 6, Table 6.11). These

[14] Because a quantitative estimate of the soybean production attributable to the RFS in this report could not be reached (see Chapter 7), the focus is on the results from Chapter 6 on the fraction of corn and cropland acreage change attributable to the RFS Program.

percentages (0–20%) can be applied to the overall soil quality effects from net conversion estimated in section 9.3.2 (see Table 9.1). For instance, cropland associated with RFS-attributable corn ethanol may have increased soil erosion from agriculture by up to 1.6% in the Midwest and total N and SOC loss by 0.7% and 1.1%, respectively. Compared to CRP benefits for context, the magnitude of cropland increases associated with RFS-attributable corn ethanol represents up to 3.7% of the N retention benefits for the entire United States (Table 9.2).

Some considerations should be noted regarding these estimates. First, the estimates represent an effect from the RFS Program only on corn ethanol and corn (Chapter 6), and would likely be larger if the effects of the RFS Program on soybean biodiesel or other biofuels were added (see Chapter 7). Second, these estimates were derived as a fraction of the effects of grassland conversion to a tilled corn-soybean rotation. Values from tilled corn-soybeans were used to estimate the upper bound of effects, while the effects from conversion to no-till corn-soybeans would likely fall in between the range calculated in Table 9.2. Third, the upper bound estimates of an RFS Program effect would have been higher than shown in Table 9.2 if the tilled continuous corn results shown in (Zhang et al., 2021) were applied. Corn is most

Table 9.2. Estimated range of soil effects associated with RFS corn ethanol production. Calculated by applying 0–20% RFS attribution estimate to the simulated soil quality effects of net grassland conversion (conversion minus abandonment) to and from tilled corn-soybeans in 12 Midwestern states from 2008 to 2016 (see Table 9.1; simulations conducted using the EPIC model).

Soil Quality Metric	Erosion/ Sedimentation	Total N Loss	Total P Loss	Total SOC Loss
Range of net impacts over 12-state area due to the RFS Program	0–2.4 (Tg/yr)	0–8.8 (Gg N/yr)	0–1.0 (Gg P/yr)	0–134.8 (Gg C/yr)
Percent range due to the RFS Program compared to U.S. Midwest cropland [a]	0–1.6%	0–0.7%	N/A	0–1.1%
Percent range due to the RFS Program compared to CRP benefits for entire U.S. [b]	0–1.4%	0–3.7%	0–2.1%	0–1.5%

Gg = gigagrams; Tg = teragrams; yr = years

[a] Relative amount is calculated by comparing with the estimate soil erosion (150 Tg sediment/yr) (Zhang et al., 2015), N loss (1,200 Gg N/yr) (Zhang et al., 2015), and SOC loss (12,000 Gg C/yr) (West et al., 2008) from the cultivated cropland in the U.S. Midwest.

[b] Relative to the environmental benefits of Conservation Reserve Program (CRP) for the United States in 2017, estimated in https://www.fsa.usda.gov/Assets/USDA-FSA-Public/usdafiles/EPAS/natural-resouces-analysis/nra-landing-index/2017-files/Environmental_Benefits_of_the_US_CRP_2017_draft.pdf (Accessed 6/7/2020). Note: EPIC estimates of erosion and N and P loss are compared to CRP estimates of sediment, N, and P not leaving field or intercepted by buffers. The EPIC estimate of SOC loss is compared to the CRP estimate of CO_2 equivalents sequestered. It was not compared to the C benefits of reduced fuel and fertilizer use, which was not included in the EPIC modeling.

often grown in rotation with soybeans, however, and therefore the continuous corn results for these calculations were not the preferred estimates for the impact from the RFS Program. Fourth, the estimates of soil quality impacts assume new cropland acres due to RFS corn ethanol came from grasslands. This is likely a valid assumption since these were the predominant land type converted (almost 90% nationally according to Lark et al. (2020), and simulation models often report that the land use change from a simulated biofuel policy comes from grasslands (Chen and Khanna, 2018; Hellwinckel et al., 2016).

Furthermore, other land use conversions (e.g., conversions of forests to croplands) are not eligible for credits under the RFS. Fifth, and finally, this initial estimate assumes that the parcels estimated as attributable to the RFS Program are a random subset of the Midwestern parcels that converted in Lark et al. (2020) and the effects are uniform across this large area of 12 Midwestern states. In reality, parcels attributable to the RFS Program may not be uniformly distributed, and attributable effects may be more pronounced in certain areas—for example, they may be greater closer to biorefineries or in areas with higher soil erosion rates or in areas with higher amounts of converted acres (e.g., southern Iowa, the Dakotas)—and conversely smaller in others. Resolving the location of grassland conversion attributable to the RFS Program is an important research need (section 9.7.4).

The switching of other crops to corn and soybeans and the mixture of production versus conservation practices on corn and soybean acreages are the other mechanisms identified in this chapter that could affect soil resources. Chapter 6 includes estimates that corn ethanol production attributable to the RFS Program caused an estimated increase of between 0 and 3.5 million acres of corn from 2008 to 2018. Nearly 2.0 million acres may have overlapped with those of expanding cropland, leaving only approximately 0 to 1.5 million acres of corn due to crop switching on existing cropland. Similar estimates for RFS-attributable soy biodiesel production are not available. Further analyses are needed to quantitatively estimate the soil quality impacts of crop switching and production versus conservation practices on RFS-associated corn and soybean acreages.

9.3.4 Conservation Practices

Conservation practices have the potential to improve soil health and reduce many of the impacts from corn and soybeans in general and the RFS Program specifically. As previously mentioned in the chapter, two practices are notable in particular: (1) conservation tillage, including no-till; and (2) cover crops. Conservation tillage practices retain crop residues on the soil surface, reducing soil erosion and minimizing the breakdown of stable soil aggregates protecting SOM from fast microbial decomposition (Paustian et al., 2019). Use of conservation tillage practices is also important when grasslands are converted to corn or soybeans since it can at least partially mitigate C loss due to such LCLM conversion (Leduc et al., 2017; Gelfand et al., 2011; Follett et al., 2009). Conservation tillage is a widely adopted practice for both corn and soybeans as noted previously (Baranski et al., 2018; Claassen et al., 2018), and its benefits are widespread, yet some producers still use conventional tillage or do not use conservation tillage practices continuously (USDA NRCS, 2022b; Claassen et al., 2018).The carbon benefits of no-till may be diminished if tillage is used occasionally or intermittently (Conant et al., 2007). This likely depends upon the frequency of tillage. No-till interrupted by tillage once every 5 to 10 years has been shown to maintain stocks of C (Blanco-Canqui and Wortmann, 2020). Thus, further increases in the adoption of conservation tillage employed continuously would likely be beneficial for soil quality.

Beyond conservation tillage, the use of cover crops has been on the rise, yet were still only planted on approximately 5-6% of harvested cropland nationally in 2017 (Wallander et al., 2021). Cover crops may be planted following harvest of the commodity crop, or managed by intercropping, when two or more crops are grown simultaneously within the same field (Geertsema et al., 2016). Both approaches increase in-field plant species richness of the cropping system, and soil is protected for a greater portion of the year. Cover crops reduce soil erosion, especially when coupled with conservation tillage (Dabney et al., 2001; Langdale et al., 1991). Although cover crops may increase dissolved phosphorus loss from fields (Carver et al., 2022), cover crops are generally associated with neutral or positive shifts in SOM, soil nutrients, and soil quality (Adetunji et al., 2020; Sharma et al., 2018; Austin et al., 2017) without reducing subsequent crop yields if properly managed (Marcillo and Miguez, 2017). Cover crops have also been found to broadly increase microbial biomass and activity and alter community structure (Zahorec et al., 2022). The magnitude of microbial response to cover crops appears to depend upon the co-occurring use of no-till and crop rotation practices, as well as depending on species identity of the cover crop used (USDA, 2022; Blanco-Canqui et al., 2015; McDaniel et al., 2014; Treonisa et al., 2010; Six et al., 2006). Similarly, larger soil organisms respond positively when plant cover is higher in fields or when a crop has more continuous cover or greater overall biomass (Adams et al., 2017; Norris et al., 2016; Wardle et al., 1999; Wardle et al., 1995). Despite this, the use of cover crops remains low in the United States, limiting its benefits currently.

9.4 Likely Future Effects

As noted previously, corn ethanol and soy biodiesel will likely remain the dominant biofuels out to 2025, the end date of consideration for the RtC3 (see Chapter 2). Furthermore, FOGs have no known effect on soil quality aside from isolated effects in landfills (see Chapter 10), and the soil quality effects from Brazilian sugarcane cultivation occur in Brazil and were relatively small and temporary in the early years of the RFS Program and the growth of the industry (see Chapter 16). Therefore, the soil quality effects in the near term will remain predominantly from the production of corn and soybean. Whether grasslands continue to be converted to corn or soybeans will in large part determine the magnitude of future effects, since the largest impact on soil quality generally occurs from this LCLM shift. Lark et al. (2020) reported a slowdown nationally in cropland expansion since 2011 and especially since 2015 (Figure 2 in Lark et al. (2020)), in agreement with other sources from Chapter 5 and with reaching the E10 blend wall in 2013. When the Set Rule was finalized on June 21, 2023, EPA published with those biofuel volumes estimates of cropland expansion attributable to the RFS (see Chapter 6, section 6.5).[15] The EPA estimated the biofuel volumes in the Final Set Rule could potentially lead to an increase of as much as 2.65 million acres of cropland in the Midwest by 2025, mostly from soybean expansion in the Midwest, but to a lesser extend from corn expansion in the Midwest and canola expansion in North Dakota. Thus, the effects on soil quality from an increase in cropland by 2.65 million acres may be expected to be comparable to or slightly larger than the effects estimated in section 9.3.2–9.3.3 from a cropland increase of 1.9 million acres. More information—already discussed above—on where these changes are anticipated and what agricultural practices are employed is needed to better constrain estimates of the likely future effects to soil quality. Additional information may be found in the associated docket for the Set Rule (EPA-HQ-OAR-2021-0427).

9.5 Comparison with Petroleum

The soil quality impacts described in this chapter do not occur in isolation, but rather represent one side of a tradeoff with petroleum, the alternative to biofuels. Unfortunately, a detailed, quantitative soil quality comparison between the two industries is not available, but it can be helpful context to consider qualitatively how biofuels and petroleum differentially impact soil quality. When comparing the two, both the spatial extent of the effects (e.g., the acreage or volume of soil impacted) and the time or effort to recover from any effects should be considered.

Trainor et al. (2016) estimated land requirements of a variety of energy sources, including biofuels and petroleum, in the United States. They estimated that biofuels required more than two-thirds of the land used for all energy sources domestically between 2007 and 2011. Projecting into the future, biofuels and petroleum production become similar in their land requirements if the spacing requirements between oil wells are included. However, the soil quality effects of the petroleum industry may be less

[15] On July 26, 2022, the United States District Court for the District of Columbia entered a consent decree, which after stipulations from the parties, required EPA to sign a notice of proposed rulemaking (NPRM) to establish 2023 volumes for the RFS Program by November 30, 2022, and to sign a notice of final rulemaking finalized 2023 volumes by June 21, 2023. Growth Energy v. Regan et al., No. 1:22-cv-01191 (D.D.C.), Document Nos. 12, 14, 15. EPA's proposed RFS volumes for 2023-2025, 87 FR 80582 (Dec. 30, 2022) available in Docket No. EPA-HQ-OAR-2021-0427 at https://www.regulations.gov, were finalized on June 21, 2023. The estimates of cropland expansion from the RFS Program took considerable time and effort to develop; thus, they are based on the proposed volumes from November 30, 2022, rather than the final volumes from June 21, 2023. Because of that, the acreages discussed here are merely illustrative. However, since the total volumes from crop-based feedstocks changed fairly little between the draft and final rules, they are still relevant here (see Chapter 6, section 6.5).

than its footprint since much of the infrastructure of petroleum wells is underground far below the soil layer.

Beyond a spatial comparison, the time and effort required to recover from any soil quality effects need to be considered. The soil quality effects of petroleum production may be longer lasting and harder to mitigate than those of biofuel feedstocks (Parish et al., 2013). Brine or oil spills onto soil can require substantial remediation to clean up, potentially including the expensive process of excavating and replacing the impacted soil. Ultimately, however, a detailed quantitative comparison on soil quality is lacking between petroleum and biofuels and therefore, there is high uncertainty regarding their comparative effects.

9.6 Horizon Scanning

Corn ethanol and soy biodiesel are likely to remain the dominant biofuels in the near term, yet it is possible that one or more alternative feedstocks may increase in importance in later years. Alternative feedstocks include cellulosic feedstocks, such as corn stover, perennial grasses, and short-rotation woody species, non-cellulosic feedstocks, such as algae (DOE, 2016), canola oil, and waste feedstocks for biogas (Kougias and Angelidaki, 2018). Except for algae and waste feedstocks, their impacts on soil quality will often depend upon on prior LCLM (Robertson et al., 2017) and the balance between production and conservation practices employed.

Leftover residues from annual crops are a potential type of cellulosic feedstock, with corn stover as an example. Corn stover consists of the leaves, stalks, and other parts of the corn plant after the grain is harvested. Utilizing corn stover offers an opportunity to intensify biofuel production without needing to expand the acreage of land in production, because both grain and stover could be harvested from the same parcel of land. The amount of stover harvested, however, needs to be constrained so adequate amounts remain to provide soil cover for erosion control and biomass to sustain SOM stocks (Xu et al., 2019; Wilhelm et al., 2007). Furthermore, nutrients are removed by stover harvests, so soil nutrients need to be monitored to prevent reduced crop yields in subsequent years (Karlen et al., 2014). Whether corn stover can be harvested sustainably, and at what removal rate, depends on many site-specific factors, including yields, topography, soil characteristics, climate, and tillage practices(Karlen et al., 2014). Pairing stover removal with no-till practices and cover crops can reduce effects—for example, (Lehman et al., 2014) reported this combination had only limited impacts on soil microbial communities.

Other cellulosic feedstocks, such as short-rotation woody perennials (e.g., hybrid poplar) and perennial grasses (e.g., switchgrass), could have positive impacts on soil quality depending upon the preceding LCLM type. If these feedstocks replace relatively unmanaged grasslands, then the soil quality effects could be negative. However, the effects are likely to be positive if they replace annual row crops, abandoned agricultural land, or where soil quality has been degraded. Since they can be harvested as biofuel feedstocks for multiple years without needing to disturb the soil for planting, these perennial feedstocks can reduce erosion and subsequent SOM or soil nutrient loss (Robertson et al., 2017). Repeated harvesting of perennial plants for biofuel production may require fertilizer inputs at some point (Johnson and Barbour, 2016), raising off-site water and air quality concerns. The inputs could be offset, at least in part, by including N-fixing plants in rotation with the perennial species. Perennial grasses or short-rotation woody species can also provide greater quantities of leaf and root matter to the soil food web than annual crops (Duchene et al., 2020). These effects suggest cellulosic feedstocks may help preserve soil biological communities and the ecosystem processes they mediate.

Finally, algae are a potential feedstock for bioenergy, as well as waste products for biogas. Following oil extraction, algal residues could be used as a soil amendment to enhance soil C and SOM (Rothlisberger-Lewis et al., 2016). Moreover, some algae can grow in the soil, contributing to soil C and enhancing soil N status and cycling (Renuka et al., 2018). Biogas produced from waste feedstocks yield both gas and digestate, remaining after the process. As is done for compost, this digestate can be applied to soils to enhance nutrient and organic matter content (Glowacka et al., 2020). Further research is needed to resolve the utility and effects of algae as a soil amendment.

9.7 Synthesis

9.7.1 Chapter Conclusions

- Impacts to date on soil quality from biofuels and the RFS Program are almost exclusively due to corn and soybean production for corn ethanol and soy biodiesel.
- Conversion of grasslands to corn and soybeans causes greater negative impacts to soil quality compared to growing these feedstocks on existing cropland. Simulations using the EPIC (Environmental Policy Integrated Climate) model found estimated grassland conversion to corn/soybeans from all causes generally increased soil erosion (-0.9-7.9%), and losses of soil nitrogen (1.2-3.7%) and soil organic carbon (SOC, 0.8-5.6%) in a 12-state, U.S. Midwestern region between 2008 and 2016. The range in losses depended upon the simulated tillage practices.
- Effects were not uniform across the 12-state region. Hotspots of grassland conversion and subsequent soil quality impacts occurred in locations such as southern Iowa and the Dakotas.
- A range of percentages (0–20%) was applied to the EPIC results to estimate the fraction of soil impacts attributable to grassland conversion estimated to be caused by the RFS Program. According to this estimation, the RFS Program increased erosion, nitrogen loss, and SOC loss from 0-1.6%, 0-0.7%, and 0-1.1%, respectively, across the 12-state region between 2008 and

2016. Notably, these modeling estimates represent a RFS-corn-ethanol effect only, and do not include any additional quantitative effect from the RFS Program on soybean acreage as this effect was not quantified in Chapter 7, nor do the estimates include any effect from crop switching on existing cropland.

- For context, the magnitude of these changes can be compared to the benefits of conservation programs, like the Conservation Reserve Program (CRP). The RFS-associated increase in N loss for this 12-state region, for example, represents up to 3.7% of the N retention benefits of the CRP for the entire United States.
- Additional conservation measures—such as further adoption of conservation tillage and cover crops—would help reduce the impacts of biofuels generally and the RFS Program specifically on soil quality.
- The likely future effects of the RFS Program are highly uncertain. While the projected cropland expansion for 2023–2025 is slightly larger than the estimated historical cropland expansion from the RFS Program, EPA cannot say with reasonable certainty that any particular terrestrial ecosystem or biodiversity will be impacted, due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes.

9.7.2 Conclusions Compared with the RtC2

The findings from this chapter strengthen and extend the conclusions of the 2018 Report to Congress (i.e., the RtC2). The RtC2 emphasized the potential for negative soil quality effects of grassland conversion to biofuel feedstocks. This report does the same, yet also presents estimates of the soil quality effects of grassland conversion to agriculture, and of the subset that may be attributable to the RFS Program. Although relatively small percentages regionally, the soil quality impacts of the RFS Program may be meaningful at the local scale in areas with higher rates of conversion and/or soils more susceptible to impacts because of factors such as topography (e.g., in local watersheds in southern Iowa). Additional conservation practices could be needed to offset effects, particularly in these locations. The modeling and literature both conclude that conservation practices, particularly conservation tillage—including no-till—and cover crops, can improve the soil quality outcomes of feedstock production.

9.7.3 Uncertainties and Limitations

- Foremost, estimates are broadly lacking on the spatial location of grasslands converted to corn and soybeans due to biofuels generally and the RFS Program specifically. This chapter employs an estimate of the amount and location of grassland conversion to crops from all causes between 2008 and 2016 (Lark et al., 2020) to simulate the soil quality effects of this

LCLM change. Percentages (0–20%) were applied to estimate the fraction attributable to RFS-associated corn ethanol. This is a large step forward, yet soil effects can vary by site-specific factors, such as soil type, topography, and climate. Thus, estimates of the locations of grasslands converted due to biofuels and the RFS Program would greatly improve the quantification of soil effects, even if other uncertainties and limitations are not resolved.

- The amount of land, location, and crop type switching to corn and soybeans due to biofuel demand and the RFS Program remains uncertain. Having this information would allow an estimate of the soil quality effects of crop switching. Further uncertainty exists regarding the relative mix of production and conservation practices implemented on lands used to grow feedstocks because of the RFS Program.
- Information on management practices are generally only available at large multi-state scales, inadequate to support detailed soil quality modeling. Spatially resolved and time-series data on management practices (e.g., tillage, tiling) by crop, at the Crop Reporting District, county, or smaller scale are needed.

9.7.4 Research Recommendations

- Resolving some of the fundamental uncertainties listed above should be the next steps for research, particularly the location of grassland conversion attributable to biofuels generally and the RFS Program specifically. Location-specific estimates are needed, if not at the field scale, then by Crop Reporting District, county, or local-scale watersheds.
- Future research efforts are also needed to examine potential soil impacts beyond the 12-state region focused on in this chapter.
- Research is needed to derive information on the management practices (e.g., tillage) by crop at spatial and temporal resolutions detailed enough to support soil quality modeling without compromising privacy.
- A review and update of plant trait parameters for EPIC and other agroecosystem models is an additional research need, further improving the simulation of current and future effects of biofuels and the RFS Program.
- Research is needed on the socioeconomic barriers to greater use of cover crops as a conservation practice, and the policies that may stimulate greater use.
- Research is needed to estimate the soil quality effects of crop switching to corn and soybeans from an array of crops (e.g., cotton, wheat) due to the RFS Program.
- Further research is needed to quantify the utility and effects of algae and biogas digestate as soil amendments.

- The Inflation Reduction Act invests billions in climate-smart agriculture, and research is needed to follow up on these investments to quantify their environmental and agronomic benefits.

9.8 References

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10. Water Quality

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Key Findings

- Water quality impacts to date from biofuel production are almost exclusively due to corn and soybean production for corn ethanol and soy biodiesel. Conversion of grasslands to corn and soybeans causes greater negative impacts to water quality compared to growing these crops on existing cropland.
- A Missouri River Basin (MORB) Soil and Water Assessment Tool (SWAT) model was applied to a 30-year period (1987 to 2016) to assess the effects of recent estimated cropland expansion on water quality, where the highest rate of grassland to cropland conversion have occurred (1.18% of the total land area was converted from 2008 to 2016 basin wide). Conversion to cropland resulted in little change in streamflow basin wide. For total nitrogen (TN) and total phosphorus (TP), grassland conversion to continuous corn resulted in the greatest increase in TN and TP loads (6.4% and 8.7% increase, respectively); followed by conversion to corn/soybean (TN increased 6.0% and TP increased 6.5%); and then conversion to corn/wheat (TN increased 2.5% and TP increased 3.9%). These increases are relatively small on an absolute basis, which earlier chapters suggest approximately 0–20% may be due to the RFS Program (Chapter 6), but these increases may aggravate conditions in watersheds already impacted by nutrients.
- Groundwater and drinking water nitrate concentrations may increase with increasing acreage of corn. Switching from corn or other crops to dedicated biofuel crops (e.g., switchgrass) may lead to reductions in nitrogen losses to water bodies and thereby reduce future drinking water nitrate levels in both groundwater and surface water.
- Pesticides in drinking water could be impacted by increasing acreage of corn or soybean for biofuels or other uses. Certain pesticides, such as atrazine, are more widely used than others on these crops, and have also been frequently detected in surface and ground water. Pesticides whose usage on corn or soybeans has changed in recent years would presumably see commensurate changes in their detection likelihood in water, including in drinking water supplies. Fewer pesticides may need to be applied to dedicated biofuel crops than corn and soybean crops.
- Lifecycle potential eutrophication effects for both corn ethanol and soybean biodiesel are higher than their fossil fuel counterparts per megajoule and overall in most cases. This is driven primarily by fertilizer application to corn and soybean crops and by the resulting nutrient runoff and leaching.
- Continued implementation of conservation practices has been shown to reduce soil erosion, nitrate loss, and phosphorus release. Integrating landscape design and conservation practices (reduced tillage, riparian buffer, saturated buffer, cover crops) in current corn/soybean land and

cropland converted to perennial grass at field tests has been shown to decrease nutrient loss to surface water while maintaining corn/soy productivity. Conservation practices, such as reduced tillage and the use of cover crops, can reduce the negative impacts of corn and soybean feedstock production and improve soil health.

- The likely future effects of the RFS Program on water quality are highly uncertain. While the projected cropland expansion for 2023–2025 is slightly larger than the estimated historical cropland expansion from the RFS Program, it cannot be said with reasonable certainty that water quality will be impacted due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes.

Chapter Terms: Disinfection By-Products (DBP), dissolved organic carbon (DOC), drinking water, eutrophication, groundwater, lifecycle assessment, National Water-Quality Assessment (NAWQA), nitrate, nitrogen, nutrients, sediments, total organic carbon (TOC), underground storage tanks, water quality, watershed

10.1 Overview

10.1.1 Background

Changes in nutrient, pesticide, and sediment transport associated with agriculture can impact water quality (Capel et al., 2018). In many cases, biofuel feedstock production contributes to these impacts, depending on the situation. Water quality can be adversely affected by the production of biofuel feedstocks, primarily due to the sediment, nutrients, pesticides, and pathogens directly or indirectly released during primarily the feedstock production phase (Demissie et al., 2017, 2012; Secchi et al., 2011; Costello et al., 2009; Thomas et al., 2009; USDA, 2004). These releases vary depending on the biofuel feedstock source, the feedstock production site's management practices, and direct or indirect land use changes associated with feedstock production. Water quality impacts of these changes, in the context of this report, are examined for groundwater, freshwater (rivers, lakes and streams), drinking water, and coastal waters. Chemical (e.g., nitrogen [N], phosphorus [P], pesticide) and sediment loadings to surface water and groundwater are the most significant effects related to feedstock production (Welch et al., 2010). The authors also briefly consider effects on temperature and organic matter. Hypoxia and harmful algal blooms are significant downstream water quality impacts that can be related to increased nutrient and sediment transport, which can be found both in coastal and non-coastal waters (discussed in Chapter 13). Generally this chapter will focus on water quality effects from growth of corn for production of corn ethanol and soybeans for production of biodiesel (and renewable diesel), and thus will focus on water quality effects in regions of the country where these crops are grown.

This RtC3 on biofuels adds several components of the impacts on water quality. The 2011 and 2018 reports did not closely examine the impacts of pesticides or biofuel storage on drinking water quality. Earlier reports focused on surface water streams and lakes, which are also major receiving bodies of nutrients. Movement to water bodies of pesticides associated with biofuel feedstocks is also explored here. Finally, the storage of biofuel products in underground storage tank systems (UST) or from aboveground fuel infrastructure such as tanks or dispensers sometimes results in release of these products into the environment where they can contaminate surface and groundwater. Releases from USTs may occur when the UST equipment is incompatible with the fuel, or from various other causes of releases, such as the overfilling of a UST. These impacts are reviewed in this chapter, with varying levels of detail dependent upon data availability and linkages to biofuel-associated drivers.

This chapter summarizes the results of long-term data collection efforts and from modeling efforts on water quality from biofuels. Because of the scale of the midwestern U.S. region involved, the number of empirical studies on water quality effects directly due to biofuels, especially those conducted over time at a meaningful local scale, are limited. Long-term studies are the most valuable for the purpose of this chapter, but they are hardest to carry out given the short-term character of most scientific funding and the scale and heterogeneity of the region studied. For identifying the impacts of biofuels on water quality, several modeling efforts were reviewed as well, and these may be the most directly applicable to distinguishing the effects of biofuels when viewed with the appropriate caveats related to the uncertainties.

10.1.2 Drivers of Change

The drivers discussed in earlier chapters (e.g., biofuel volumes, land use, conversion technologies, agricultural practices) are inherently connected to water quality. Water moves through the agricultural landscape, infiltrates the soil, percolates to groundwater, and also runs off directly to surface water. Where surface and subsurface drainage structures exist in the agricultural landscape, infiltrating water may bypass deeper soil and groundwater, feeding more directly into streams and rivers. In addition, storage of biofuel products in USTs can contribute to environmental releases if the equipment is not designed to use the biofuel blend. All of these releases may alter water quality in surface freshwaters, groundwater, and estuarine/coastal systems.

The typical drivers of enhanced nitrate in groundwater or surface waters are fertilizer (Robertson and Saad, 2021; Howarth et al., 2002), atmospheric deposition (Du et al., 2014), animal waste (Sobota et al., 2013), and crop N fixation (Sabo et al., 2019; Sobota et al., 2013). Agricultural practices that can influence the amount of surplus N left on the land (after accounting for inputs and losses) are important factors in determining N concentration in surface and groundwaters (Sabo et al., 2019; McLellan et al., 2018). Similarly, use of row crops compared to perennial crops has resulted in greater N losses to groundwater and surface water (Randall and Mulla, 2001). Agricultural practices can also reduce or mitigate the impacts from fertilizer application using several approaches including buffer strips, changes in tillage practices, and cover crops, among others (Duriancik et al., 2008).

Four biofuels are the main focus of this report: corn ethanol, soy biodiesel, FOGs, and Brazilian sugarcane. Biofuel feedstocks from agriculture, like other agricultural crops, require fertilization and chemical inputs at varying levels. These can influence nutrient and chemical levels in water bodies, including nitrate in drinking water supplies. Some biofuel crops may have less of an impact on N inputs than other row crops (Smith et al., 2013; Love and Nejadhashemi, 2011). Currently, most biofuels are derived from corn and soybean cultivation (see Chapters 1 and 2), and thus these are the focus of this chapter.

Estimates from the USDA Natural Resources Inventory (NRI) suggest that between 2007 and 2017, roughly 10 million acres of land were converted to crop production from many drivers (including the RFS Program), with net conversion being concentrated in the Dakotas, Iowa, Kansas, Kentucky, and North Carolina (Chapter 5). These estimates agree broadly with other sources such as the USDA Cropland Data Layer for 2008–2016 despite differences in methodology (Lark et al. (2020)), also discussed in Chapter 5. Despite varying nutrient application and runoff characteristics of these different crop production areas, direct connections between increased feedstock production and water quality impacts are only beginning to be assessed. Research to evaluate the impacts of increased biofuel production and use on water quality has largely been based on modeling rather than observed changes. Models enable evaluation of the change in water quality attributable to biofuel feedstock production, which is a challenging problem to broadly examine by field measurements. In section 10.3.2 below, modeling results for previous land use changes from all causes including the RFS Program in the Missouri River Basin (MORB) are presented.

Based on the conclusions from earlier chapters, corn production has intensified on land already under cultivation, and corn and soybeans have expanded on other cropland and to land that was previously uncultivated. Correlational inference suggests that biofuel production contributes to these changes (Wright et al., 2017; Lark et al., 2015; Brown et al., 2014), and previous chapters in the report quantify how much change is attributable to the RFS Program (see Chapter 6 and 7 for corn ethanol and soybean biodiesel, respectively). Of this total converted acreage, approximately 0–20% may be due to the RFS-effect on corn ethanol, with the largest estimated effect in 2016 as other factors that affect ethanol production diminished in effect (see Chapter 6, Table 6.11). EPA was not able to quantify the effect of the RFS Program on soybean biodiesel (see Chapter 7). There is also an unknown amount of net conversion to corn from other crops at the national level, as well as changes in crop rotations to more continuous corn (see Chapter 6). Regional studies suggest these unknowns could be significant (Ren et al., 2016; Plourde et al., 2013). This expansion of cropland has important implications for N and P fertilizer use across the landscape, which could result in increased leaching and runoff of nutrients to groundwater and surface waters.

Many factors affect the fraction of N, or any other nutrient or chemical applied to land, that might reach water bodies. Higher crop yields (bushels per acre) entail higher nutrient uptake, and conservation measures such as no-till production can reduce loss of nutrients or chemicals that run off into water bodies (Wade et al., 2015). Conservation practices can mitigate nutrient release to surface water and groundwater. Activities through the USDA's Conservation Reserve Program (CRP) in 2017 were estimated to prevent the loss of over 192 million metric tons of sediment, 521 million pounds of N, and 103 million pounds of P compared to land that is cropped [(USDA, 2017); see also section 10.3.4]. Between 2010 and 2013, approximately 30% of expiring CRP lands were converted back to agriculture (Morefield et al., 2016). Over time, there has been a reduction in the cumulative amount of lands enrolled by approximately 16.3 million acres, declining from a high of 36.8 million acres in 2007 to 20.5 million acres in 2021 (Figure 5.11). These changes in CRP acreage are set by the Farm Bills and are independent of policies set in the RFS Program. How these lands are managed after exiting the CRP Program, however, may be attributed to biofuels generally or the RFS Program specifically. Although leveling off, the CRP is currently at its lowest acreage since 1988, though the USDA Long Term Agricultural Projections Report estimates those levels may increase as the cap increases from 24 to 27 million acres (IAPC, 2021). Therefore, the watershed-level dynamics of cropland and conservation practices, as well as other pollutant sources, are important in evaluating the net impacts of biofuel production on water quality.

10.1.3 Relationship With Other Chapters

This chapter on water quality draws upon important baseline information about pesticide and fertilizer use (Chapter 3), trends in land use changes (Chapter 5), and from the attribution chapter on corn ethanol (Chapter 6). The water quality chapter also connects with the Missouri River Basin Soil and Water Assessment Tool (SWAT) modeling efforts in the Soil Quality chapter (Chapter 9), and lifecycle assessment modeling efforts in the Water Quantity Chapter (Chapter 11). While this work identifies impacts of biofuels on chemical concentrations and loads in water bodies, the implications of the water quality changes for aquatic end points and wetlands are explored in the Aquatic Ecosystems (Chapter 13) and Wetlands chapters (Chapter 14). For example, this chapter might identify an impact on loads and concentrations of nitrate or pesticides, while the Aquatic Ecosystems chapter will focus on their implications for aquatic life.

10.1.4 Roadmap for the Chapter

This chapter on water quality begins by presenting the previous RtC findings (section 10.2), then conducts a literature review of new information about the effects of release of nutrients, pesticides, carbon, and other issues related to biofuel production from work published and produced since the 2018 report (section 10.3.1). Then new modeling results (section 10.3.2), attribution to the RFS Program (section 10.3.3), and connections to conservation practices are shown (section 10.3.4). Likely future impacts (including impacts of underground storage, section 10.4), comparison with petroleum using a lifecycle assessment approach (section 10.5), and horizon scanning of next generation biofuels and other potential issues are then discussed (section 10.6). Conclusions, uncertainties, and research recommendations complete the chapter (section 10.7).

10.2 Conclusions from the 2018 Report to Congress (RtC2)

The following are the major, bulleted conclusions from the RtC2 related to water quality:

- Demand for biofuel feedstocks may contribute to harmful algal blooms, as recently observed in western Lake Erie, and to hypoxia, as observed in the northern Gulf of Mexico. Changes to future nitrogen and phosphorus loadings will depend on feedstock mix and crop management practices.
- The 2011 Report found that corn production intensification was associated with higher levels of erosion, chemical loadings to surface waters, and eutrophication.
- Empirical studies documenting cropland extensification and crop switching to more corn suggest water quality impacts, but the magnitude of these changes is variable across the landscape and so may be detectable only in some regions.
- Implementation of conservation practices has been observed to result in a decrease of nitrogen, phosphorus, and soil erosion.
- Changes to future nitrogen and phosphorus loadings will depend on feedstock mix and crop management practices. Decreases in nitrogen and phosphorus loadings are possible should perennial feedstocks become dominant.
- Specific biofuel production scenarios expected to improve water quality may help decrease the water quality impact of predicted future extreme weather events.

10.3 Impacts to Date for the Primary Biofuels

The following sections examine the water quality impacts in freshwater (rivers, lakes and streams), groundwater, drinking water, and coastal waters. The four primary biofuels examined in the

RtC3 (see Chapter 2 section 2.3.1) are corn ethanol, soybean biodiesel, Brazilian sugarcane, and domestic FOGs. Refer to Chapter 16 for information on Brazilian sugarcane.

10.3.1 Literature Review

10.3.1.1 Nutrient and Sediments Release Effects on Surface Freshwater Quality

Across the United States, the primary sources of N and P originate from agricultural land use (farming and livestock), and other sources like atmospheric deposition, residential fertilizers, and human waste are a less dominant input [Figure 10.1a,b; (Sabo et al., 2021; Sabo et al., 2019)]. Between 2002 and 2012, there were widespread increases in surplus N and P in the midwestern United States.[1] Part of this increase may be due to the increases in corn acreage observed over this interval (Chapter 5), and part of this may be due to the anomalous drought of 2012, which meant less N and P were removed through crop harvest and more was leftover as surplus. Although the precise contribution or share of these increases in N and P surplus that are due to changes in biofuel feedstocks are not known, the increased surplus may have implications for nutrient loads and concentrations in streams.

The U.S. Geological Survey (USGS) online mapper for the National Water-Quality Assessment (NAWQA) project allows examination of long-term trends in surface water quality data, providing results from a long-running assessment of water quality changes in the United States from 1972 to 2012[2] (Stets et al., 2020; Oelsner and Stets, 2019). NAWQA illustrates and provides data for surface water chemistry trends (i.e., nutrients, pesticides, sediment, carbon, salinity) and aquatic ecology from 1972 to 2012. An example is shown in Figure 10.2 and Figure 10.3, which present trends in several water quality parameters from 2002 to 2012. This resource unfortunately has limited data from many of the hotspots of land use change identified in Chapter 5 (e.g., South Dakota, North Dakota). However, it does show in the central agricultural areas that total nitrogen (TN) concentrations appear to be declining in Iowa and increasing in Oklahoma from 2002 to 2012. Total phosphorus (TP) concentrations appear to be decreasing in Iowa and increasing in Kansas, Oklahoma, and parts of western South Dakota. It also shows the responses within larger rivers, which indicate potential downstream impacts.

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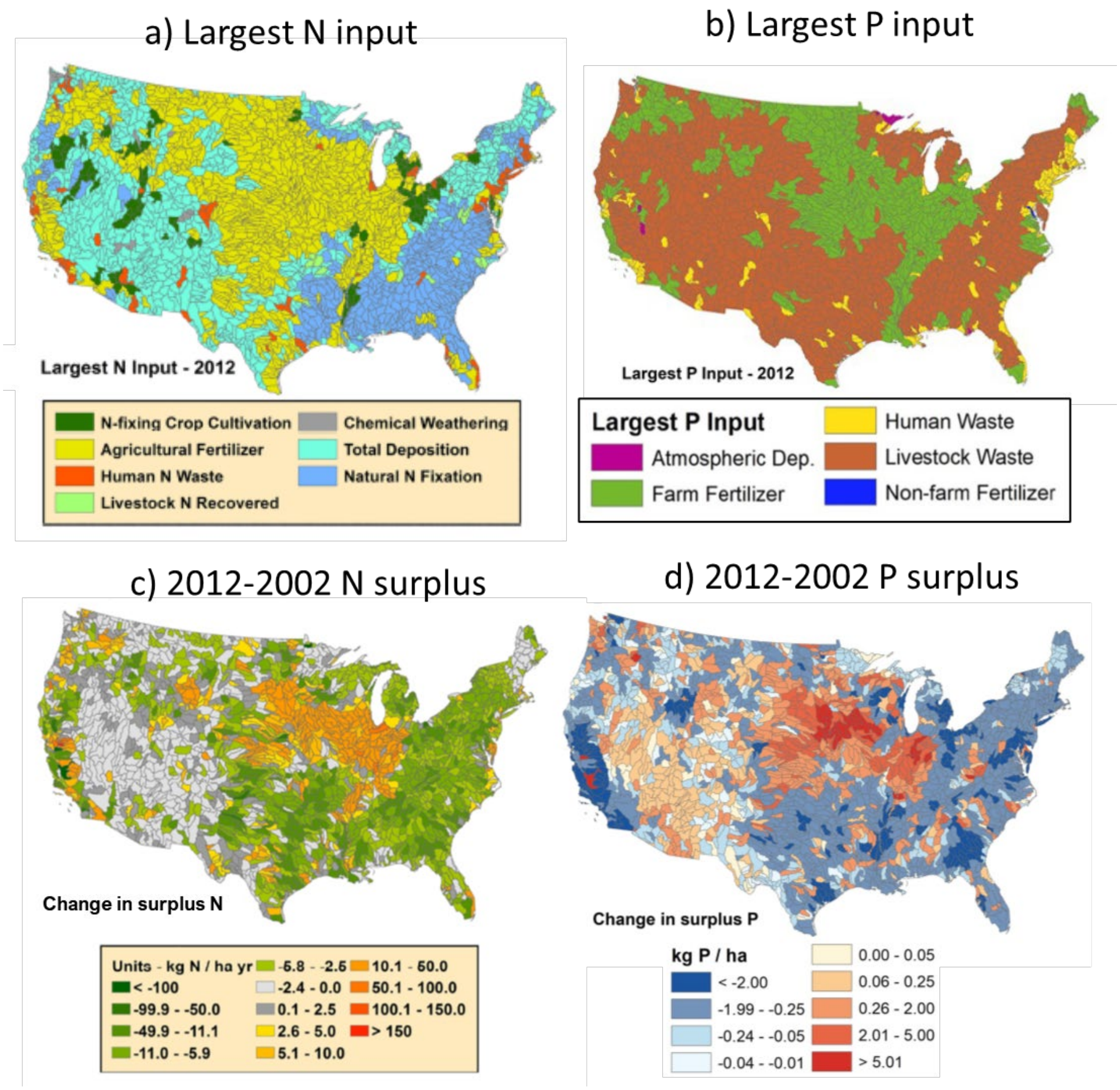
[1] Surplus nitrogen and phosphorus in Sabo et al. (2019) and Sabo et al. (2021) refers to the difference between inputs (I; e.g. fertilizer, atmospheric deposition) and non-hydrologic outputs (O; e.g. crop removal, livestock removal), or I minus O. It represents the fluxes of nitrogen and phosphorus that are more difficult to account for empirically and that may be stored in the soil or lost via leaching.

[2] https://nawqatrends.wim.usgs.gov/swtrends/



ha = hectare; kg = kilogram; yr = year

Figure 10.1. Largest nitrogen (N) and phosphorus (P) inputs to the conterminous U.S. landscape in 2012 (a, b) and change in agricultural N and P surplus in 2012 minus 2002 (c,d). Agricultural surplus is all inputs minus crop harvest N or P. Data from Sabo et al. (2021); Sabo et al. (2019).

Recent analysis of the 1992–2012 data further explores these trends by examining them by dominant land use within the watersheds (Stets et al., 2020). There is substantial variation, but Stets et al. (2020) found that there has been little change in TN concentrations and a slight increase in TP concentrations at agricultural sites across the United States. Future analyses using more recent data, as available (i.e., 2012–2020), would be useful for understanding whether changes that occurred during the growth in the biofuels industry (i.e., 2002–2012, see Chapter 6) have continued or not.

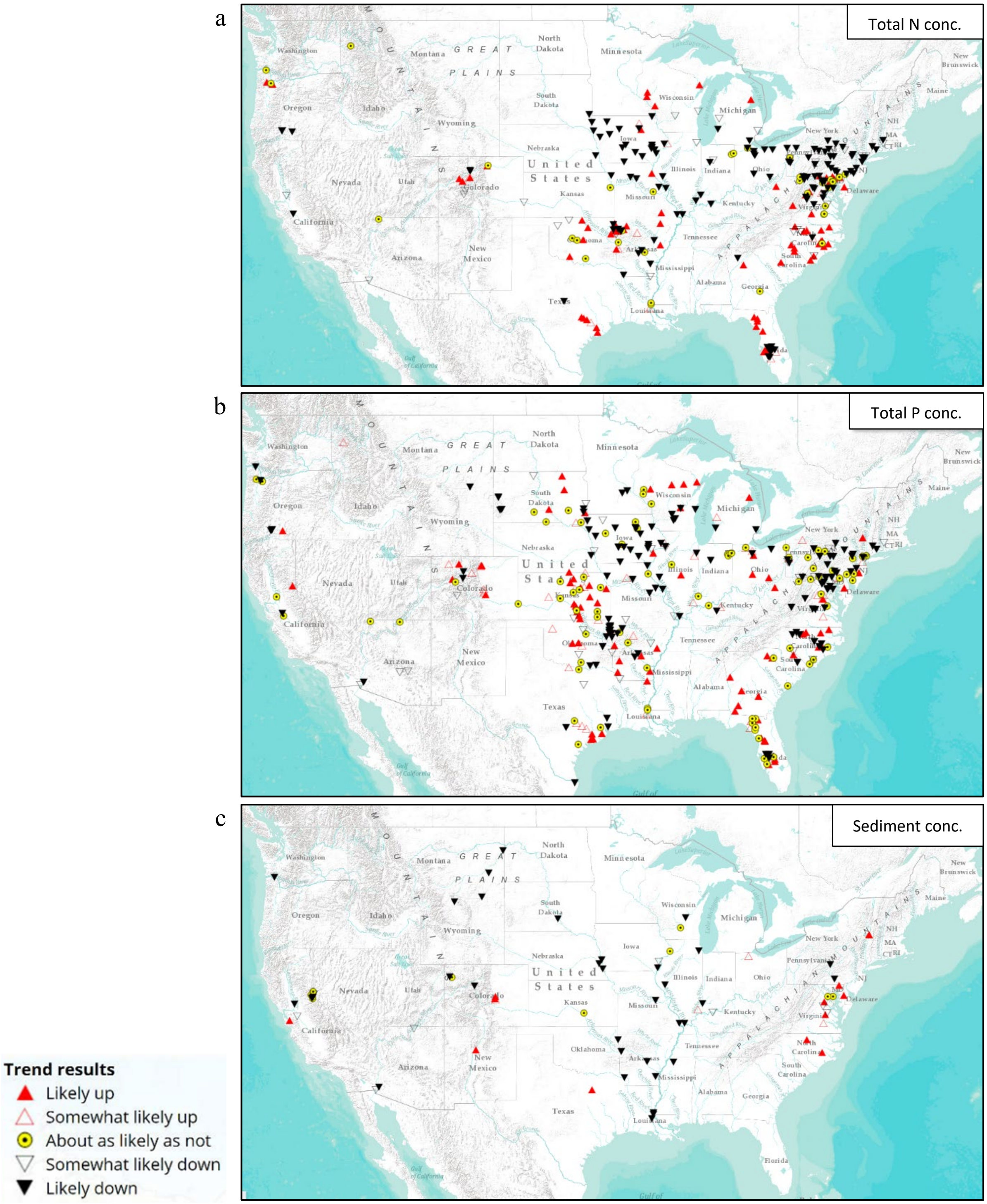


Figure 10.2a-c. Time trends in concentrations of total nitrogen (N) (a), total phosphorus (P) (b), and sediment (c) from 2002 to 2012. Source: USGS NAWQA.[3]

[3] U.S. Geological Survey, Water-Quality Changes in the Nation's Streams and Rivers, https://nawqatrends.wim.usgs.gov/swtrends/.

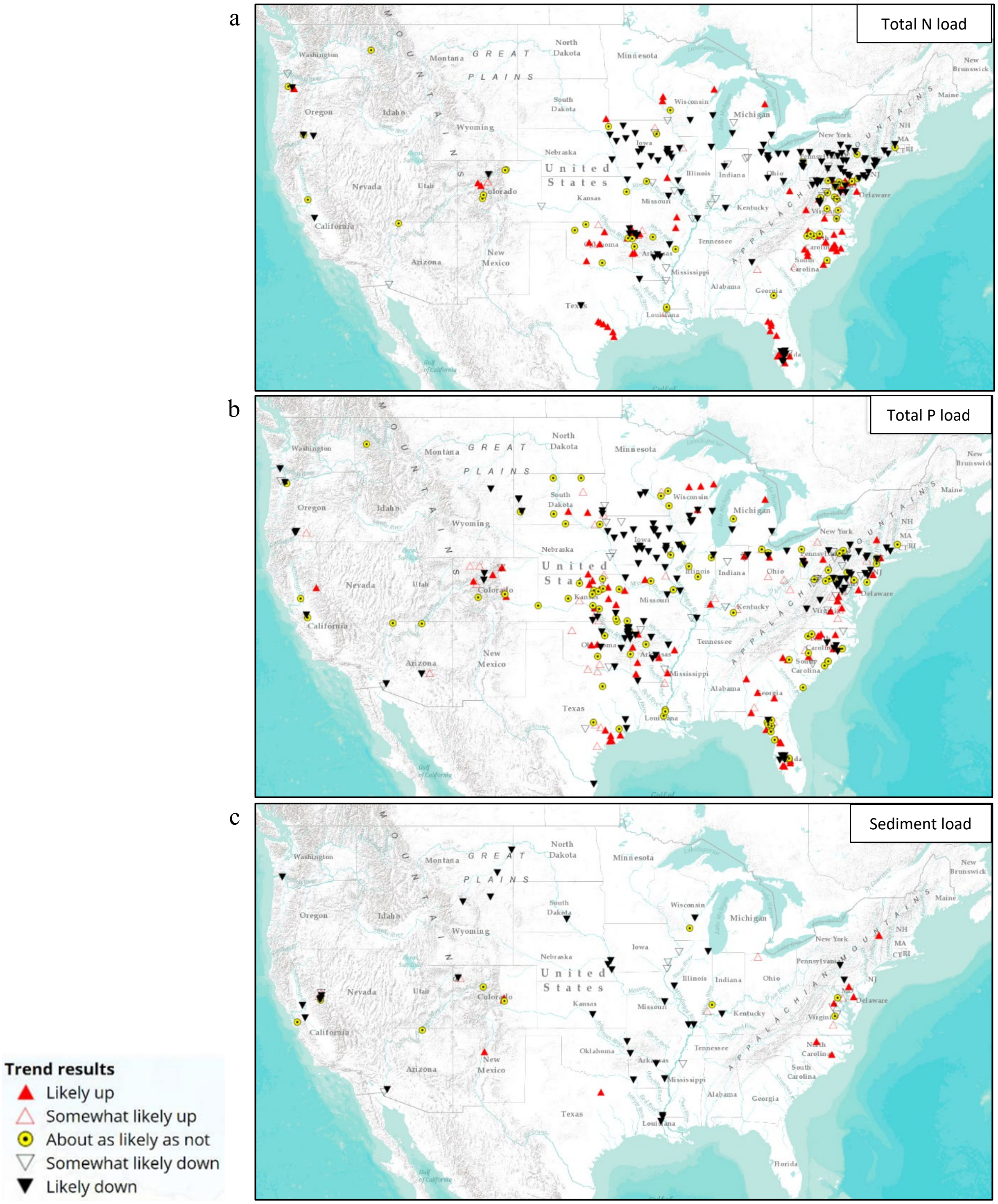


Figure 10.3a-c Time trends in loads of total nitrogen (N) (a), total phosphorus (P) (b), and sediment (c) from 2002 to 2012. Source: USGS NAWQA.[3]

The U.S. EPA's National Aquatic Resource Surveys (NARS) assess the condition of the nation's freshwater and coastal ecosystems. The first national survey by NARS was the Wadeable Streams Assessment (WSA) in 2004 (U.S. EPA, 2006) with subsequent data collected in the National Rivers and Streams Assessment (NRSA) in 2008–2009 and 2013–2014, which collectively provide information about the condition of the nation's freshwater streams prior to the RFS Program and growth in the biofuels industry. The condition classes (poor, fair, and good based on nutrient concentrations) were determined from data and observations from the "best" remaining (i.e., reference) stream sites in each ecoregion and the continuous gradient of observed values across the population of streams and rivers in the United States [(Van Sickle and Paulsen, 2008; Stoddard et al., 2006); see Table 10.1 for concentration categories by ecoregion]. NARS data and additional datasets, such as those from the USGS mapper results shown in Figure 10.2 and Figure 10.3, were used to elucidate trends in water quality over time, and the potential effects from biofuels and the RFS Program.

According to data from the WSA 2004 and the NRSA 2013–2014, and consistent with USGS mapper, the TN condition of wadeable streams in the conterminous United States has not changed between surveys (Figure 10.4a), except in the Upper Midwest ecoregion where the percentage of stream miles in good condition have decreased and stream miles in poor condition have increased (Figure 10.4d). Along with the Upper Midwest, the Temperate Plains and Northern Plains ecoregions roughly coincide with areas of feedstock production but change in TN condition was not observed beyond the margins of error. There was, however, a much greater change in condition of the nation's wadeable streams for TP, with clear decreases in percentage of stream miles in good condition and increases in stream miles in poor condition (Figure 10.5a). The same trend occurred at the ecoregional scale, including ecoregions in corn- and soy-producing areas (i.e., Figure 10.5b, d, h) as well those outside traditional corn/soy-production areas (Figure 10.5e–j). The increase in the nation's streams with poor TP condition is also seen in rivers and lakes, especially for minimally disturbed streams, but the causes are not well established at this time (Stoddard et al., 2016).

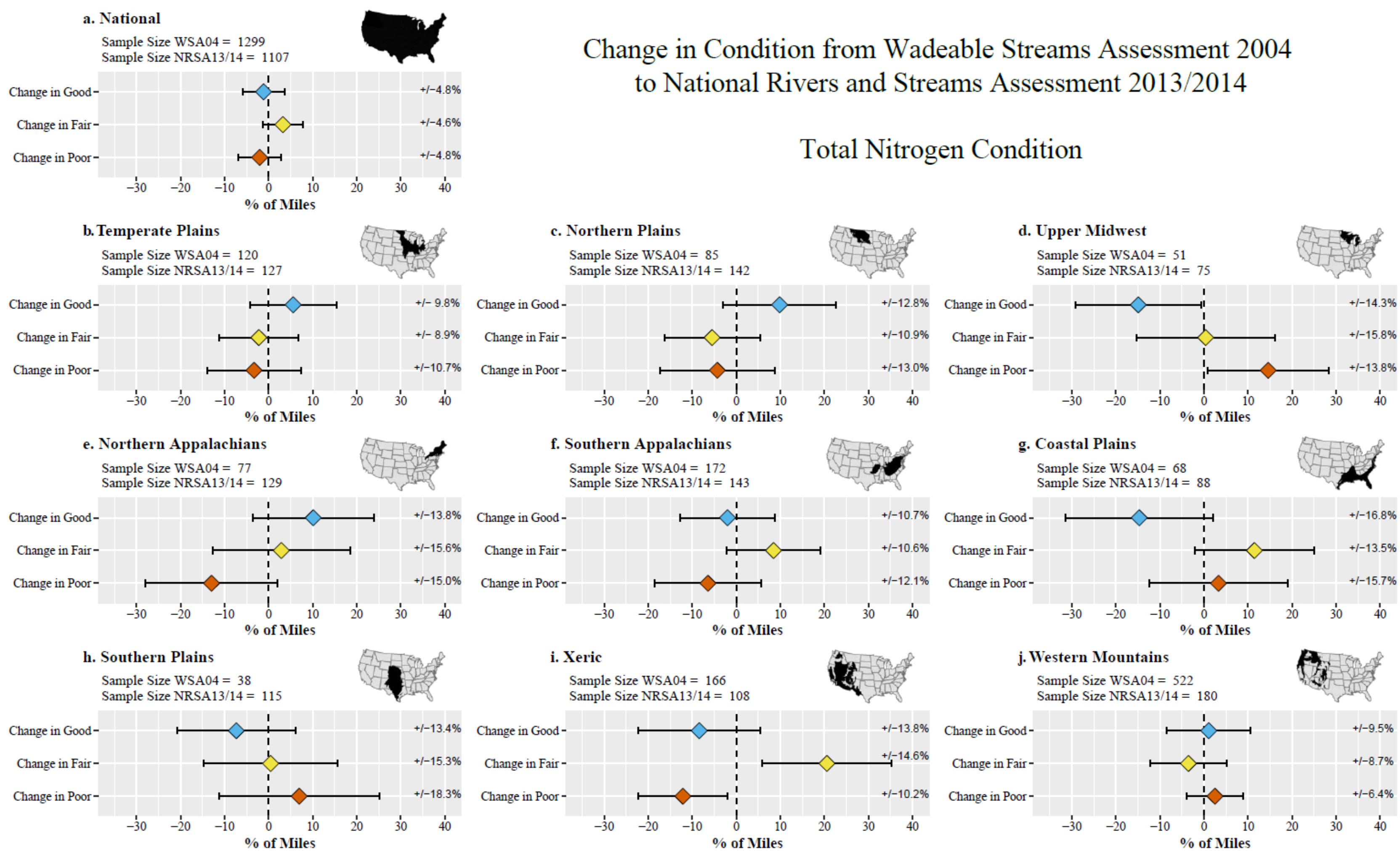


Figure 10.4. Change in total nitrogen condition in wadeable streams across the conterminous United States (a) and ecoregions (b-j) from the 2004 Wadeable Streams Assessment (WSA04) to the National Rivers and Streams Assessment 2013/2014 (NRSA13-14). The % of Miles refers to the total wadeable stream miles surveyed by U.S. EPA. The condition categories (Good, Fair, and Poor) are relative to the least-disturbed streams (see Table 10.1 for more information on the categories). Error bars are margins of error calculated from standard error × 1.96 (when error bars overlap with zero there is no significant change). Data from USEPA (https://www.epa.gov/national-aquatic-resource-surveys/data-national-aquatic-resource-surveys).

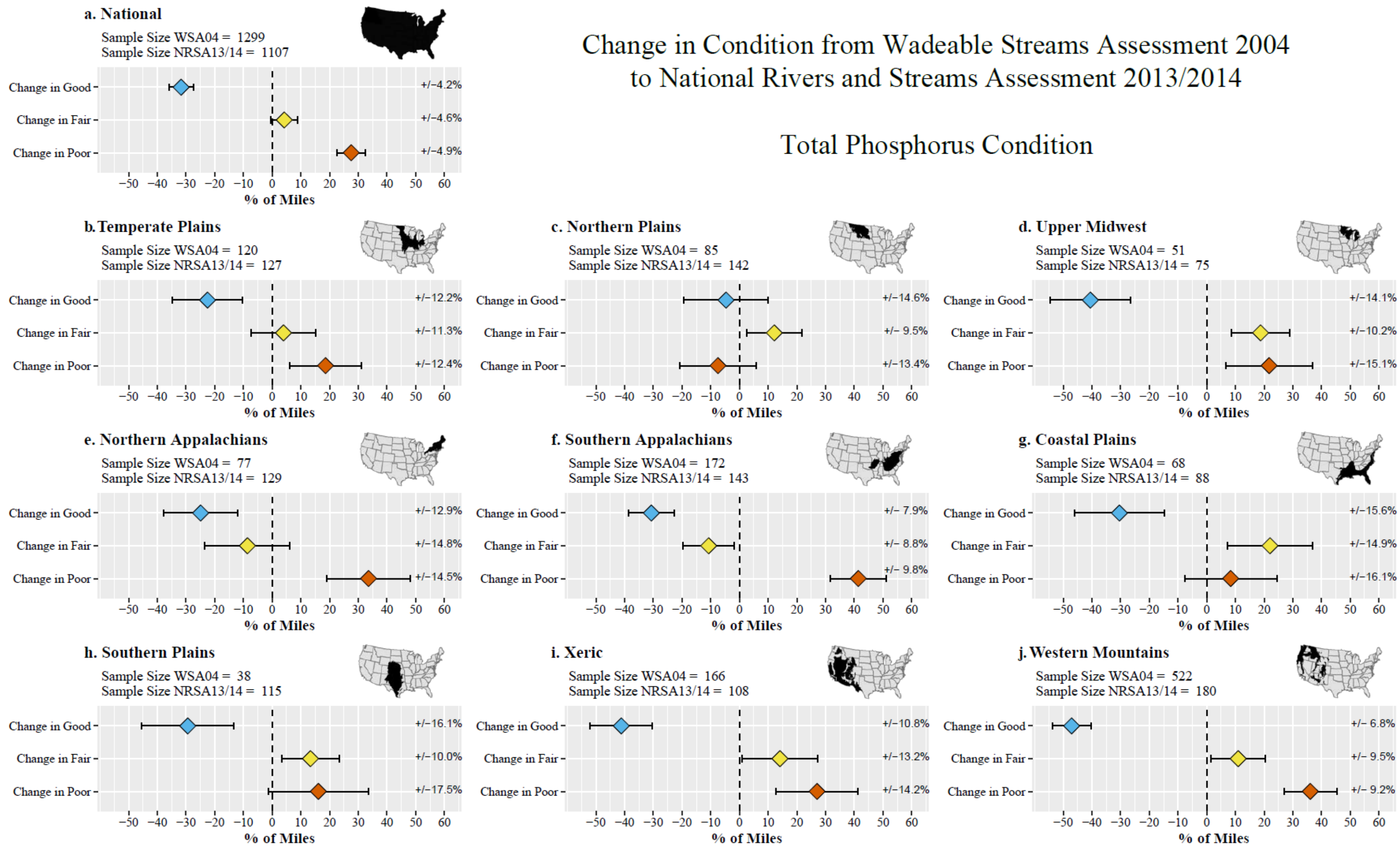


Figure 10.5. Change in total phosphorus condition in wadeable streams across the conterminous United States (a) and ecoregions (b-j) from the 2004 Wadeable Streams Assessment (WSA04) to the National Rivers and Streams Assessment 2013/2014 (NRSA13/14). The % of Miles refers to the total wadeable stream miles surveyed by U.S. EPA. The condition categories (Good, Fair, and Poor) are relative to the least-disturbed streams (see Table 10.1 for more information on the categories). Error bars are margins of error calculated from standard error × 1.96 (when error bars overlap with zero there is no significant change). Data from USEPA (https://www.epa.gov/national-aquatic-resource-surveys/data-national-aquatic-resource-surveys).

Table 10.1. Nutrient condition class benchmarks from NRSA. Different concentration thresholds (total nitrogen [TN] and total phosphorus [TP]) are used to characterize least-disturbed ("Good"), moderately disturbed ("Fair"), and most-disturbed ("Poor") sample reaches in ecoregions surveyed as part of the EPA's 2013–2014 National Rivers and Streams Assessment, part of the National Aquatic Resources Survey (NARS). Modified from table 6.1 in U.S. EPA (2019).

EPA NARS Aggregate Ecoregions	TP (mg/L)			TN (mg/L)		
	Good	Fair	Poor	Good	Fair	Poor
Central Plains	<0.06	0.06–0.10	>0.10	<0.62	0.62–1.08	>1.08
Northern Appalachians	<0.02	0.02–0.03	>0.03	<0.35	0.35–0.48	>0.48
Northern Plains	<0.06	0.06–0.11	>0.11	<0.58	0.58–0.94	>0.94
Southern Appalachians	<0.01	0.01–0.02	>0.02	<0.24	0.24–0.46	>0.46
Southern Plains	<0.06	0.06–0.13	>0.13	<0.58	0.58–1.07	>1.07
Temperate Plains	0.09	0.09–0.14	>0.14	<0.70	0.70–1.27	>1.27
Upper Midwest	0.04	0.04–0.05	>0.05	<0.58	0.58–1.02	>1.02
Western Mountains	0.02	0.02–0.04	>0.04	<0.14	0.14–0.25	>0.25
Xeric	0.05	0.05–0.10	>0.10	<0.29	0.29–0.53	>0.53

L = liters; mg = milligrams

Commercial-scale biofuel production increased steadily in recent years and reached roughly 16 billion gallons per year for ethanol and 1.8 billion gallons for biodiesel by 2018. A number of studies have evaluated the impacts of such growth on water quality based on 20-year climate, land use, and water quality measurements. As part of the Department of Energy's 2016 Billion Ton Study, which looked at aspirational targets of biofuel production levels and methods,[4] Demissie et al. (2017, 2012) simulated water quality impacts for the Upper Mississippi River Basin (UMRB) and Ohio River Basin (ORB), based on projected national feedstock production characteristics through 2022, which included changes in acreages for corn, soybean, and wheat, increased idle land, decreased pasture-hay land, increased no-till and decreased conventional tillage, increased continuous corn, and harvest a portion of corn stover as feedstock. While it is not possible to comprehensively evaluate the long-term dynamics of these projected characteristics based on the empirical record, short-term trends (e.g., 2008–2012, see land use change discussion in Chapter 5) suggest that these assumptions are mostly consistent with observations, although soybean production may be increasing more than assumed in UMRB and ORB. Demissie et al. (2012) concluded that projected feedstock production has mixed effects on water quality, projecting a 12% increase in annual suspended sediment and a 45% increase in TP loadings, but a 3% decrease in TN loading for UMRB. Findings from the ORB

[4] See Chapter 2, Box: The 2016 Billion Ton Study (DOE, 2017).

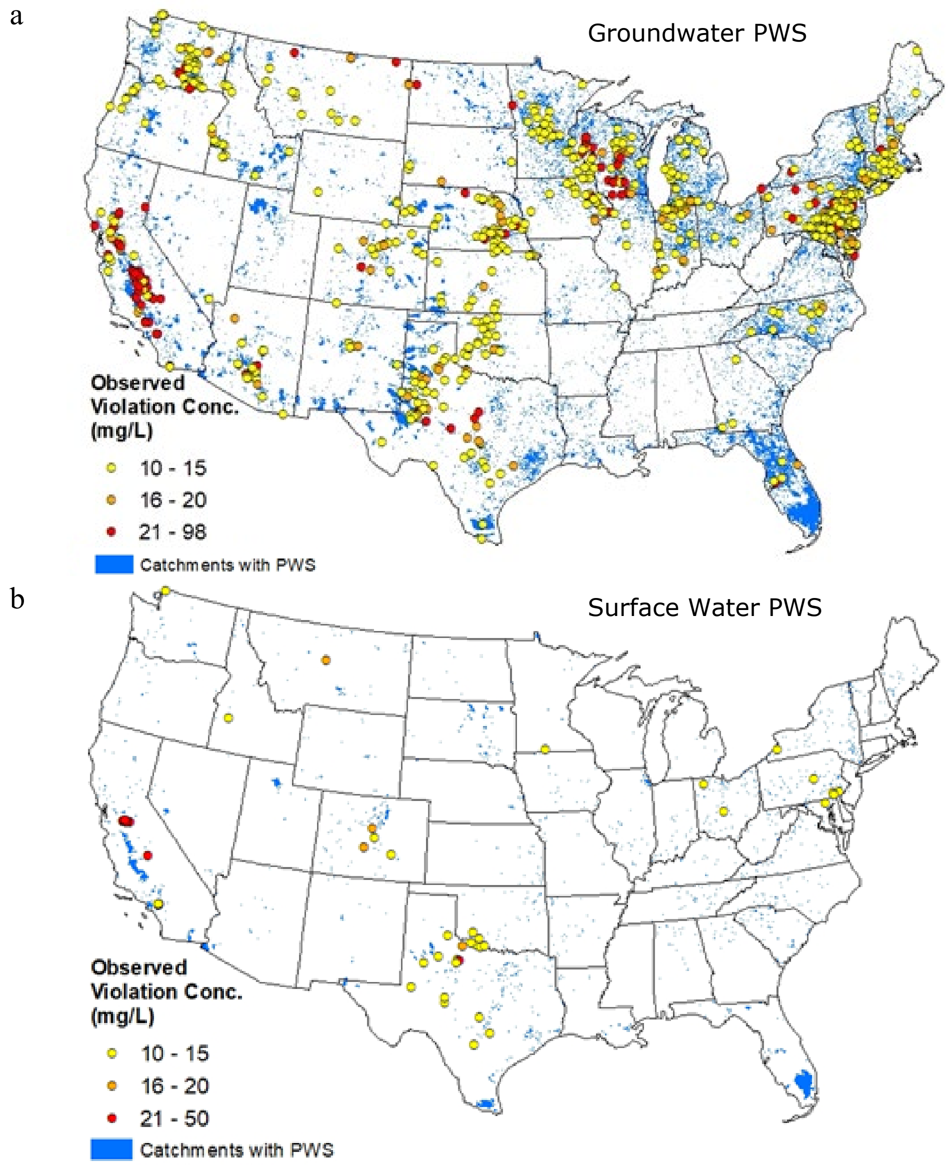
study (Demissie et al., 2017) suggested that the overall impact on water quality is much stronger than the impact on hydrology. The scenario modeling showed an increase in annual evapotranspiration of 6%, a 10% decrease in runoff, and no change in soil moisture, while the sediment and P loading increased by 40–90%. N loading was variable: it would decrease 10% when corn stover is harvested and continuous corn is in place, but would increase up to 45% in some regions when production land increased in ORB. Field-level analysis revealed substantial variability in water quality impacts in the region. Garcia et al. (2017) simulated groundwater nitrate contamination responses associated with N fertilizer application and increased corn production at a national level from 2002 to 2022, with an emphasis on agricultural areas throughout the United States. They concluded that projected increases in corn production between 2002 and 2022 could result in approximately a 56% to 79% increase in nitrate-N groundwater concentrations in areas vulnerable to high nitrate (>5 milligrams per liter [mg/L]).

10.3.1.2 Nitrate in Drinking Water

Nitrate in drinking water is a known human health concern (Ward et al., 2005). Public water suppliers, from both surface and groundwater sources, are required to report whether nitrate exceeds its 10 mg N/L maximum contaminant level (MCL). Since 1994, violations of the nitrate MCL are most commonly found in California's central valley, southwestern Washington, western Texas, Oklahoma, and Nebraska, parts of the Upper Midwest, Delaware, and southeastern Pennsylvania [(Pennino et al., 2017); Figure 10.5 – 1c or 1d from Pennino et al. (2020)]. Although the temporal connections to specific drivers such as crop types or practices are unclear, drinking water nitrate violations were increasing from 1994 until 2009 and then started decreasing after this (Pennino et al., 2017).

The typical environmental drivers of nitrate in groundwater or surface waters are fertilizer (Howarth et al., 2002), atmospheric deposition (Du et al., 2014), animal waste (Sobota et al., 2013), and crop N fixation (Sabo et al., 2019; Sobota et al., 2013). It has also been found that the specific agricultural practices, which can influence the amount of surplus N[5] left on or in the soil, after accounting for inputs and losses, is an important factor in determining N in surface and groundwaters [(Pennino et al., 2020; Sabo et al., 2019); Figure 10.6]. Similarly, use of row crops compared to perennial crops has resulted in greater N losses to groundwater and surface water (Randall and Mulla, 2001). Biofuel feedstocks (i.e., corn and soybean in this report), like other agricultural crops require fertilization and can influence nitrate levels in drinking water sources (Garcia et al., 2017; Ruan et al., 2016; Sobota et al., 2013). The full impact of biofuels on drinking water nitrate has not been estimated to date, and largely depends on the

[5] The terrestrial N surplus is defined as the difference between total inputs and non-hydrologic outputs.



L = liters; mg = milligrams

Figure 10.6. Map of the conterminous United States showing 88,083 catchments with groundwater public water systems (PWS) (blue area) and 748 catchments with groundwater PWS nitrate violations (non-blue circles) (a), and 6,934 catchments with surface water PWS (blue area) and 50 catchments with surface water PWS nitrate violations (non-blue circles) (b). Source: Pennino et al. (2020) (used with permission).

amount and type of fertilizer (Ruan et al., 2016). While there is no explicit connection in the literature between crop types and drinking water nitrate violations, it is well known that corn results in more leaching of N than soybean crops, and this provides evidence to suggest soybean crops would likely be correlated with fewer nitrate violations than corn crops. N fixed from biological fixation of soybean may also be less labile in the soil due to high carbon-to-nitrogen residues when compared with synthetic N fertilizer (Drinkwater et al., 1998). However, because these are often grown on the same land in rotation, isolating the fractional effects from one or the other is difficult. It is also assumed that there are no differences in management between corn grown for biofuels versus corn grown for feed on drinking water nitrate levels, such that they have similar nitrate leaching rates. Finally, drinking water violations are often the result of many years of accumulated legacy fertilizer N in the soil, thus parsing out the quantity from each crop would require detailed information on crop rotation and tillage practices through time. Further research could help elucidate the specific impacts of specific biofuels on drinking water nitrate levels.

10.3.1.3 Pesticides in Surface Water and Groundwater

Numerous long-term sampling studies have collected data showing a variety of pesticides in surface and groundwaters, particularly in high agricultural areas (and demonstrating the likelihood of pesticide residue presence in drinking water supplies).[6] As part of the Midwest Stream Quality Assessment (MSQA)—a collaborative effort between the USGS National Water Quality Assessment Program (NAWQA) and EPA's NRSA—water column samplers were deployed for five weeks during 2013 in 97 streams across a midwestern area dominated by corn and soybean agriculture. Results showed residues of 141 pesticide compounds at one or more sampled sites, with a median of 62 compounds detected per site [(Van Metre et al., 2017); Figure 10.7]. At a majority (81%) of sampled sites, concentrations of at least one pesticide exceeded one or more aquatic-life benchmarks established jointly by EPA's Office of Water and Office of Pesticide Programs, especially those for the protection of nonvascular plants and benthic invertebrates (Nowell et al., 2018). Of the identified compounds, the neonicotinoid imidacloprid was the most widely detected, being found at 98% of sites. Other widely detected compounds included atrazine, methoxyfenozide, and metolochlor, as well as the herbicides dimethenamid, prometon, and propazine, and the fungicides azoxystrobin, metalaxyl, and propiconazole. An analysis of stream bed sediment contaminants also conducted as part of the MSQA study (Moran et al., 2017) documented the presence of 16 additional pesticides.

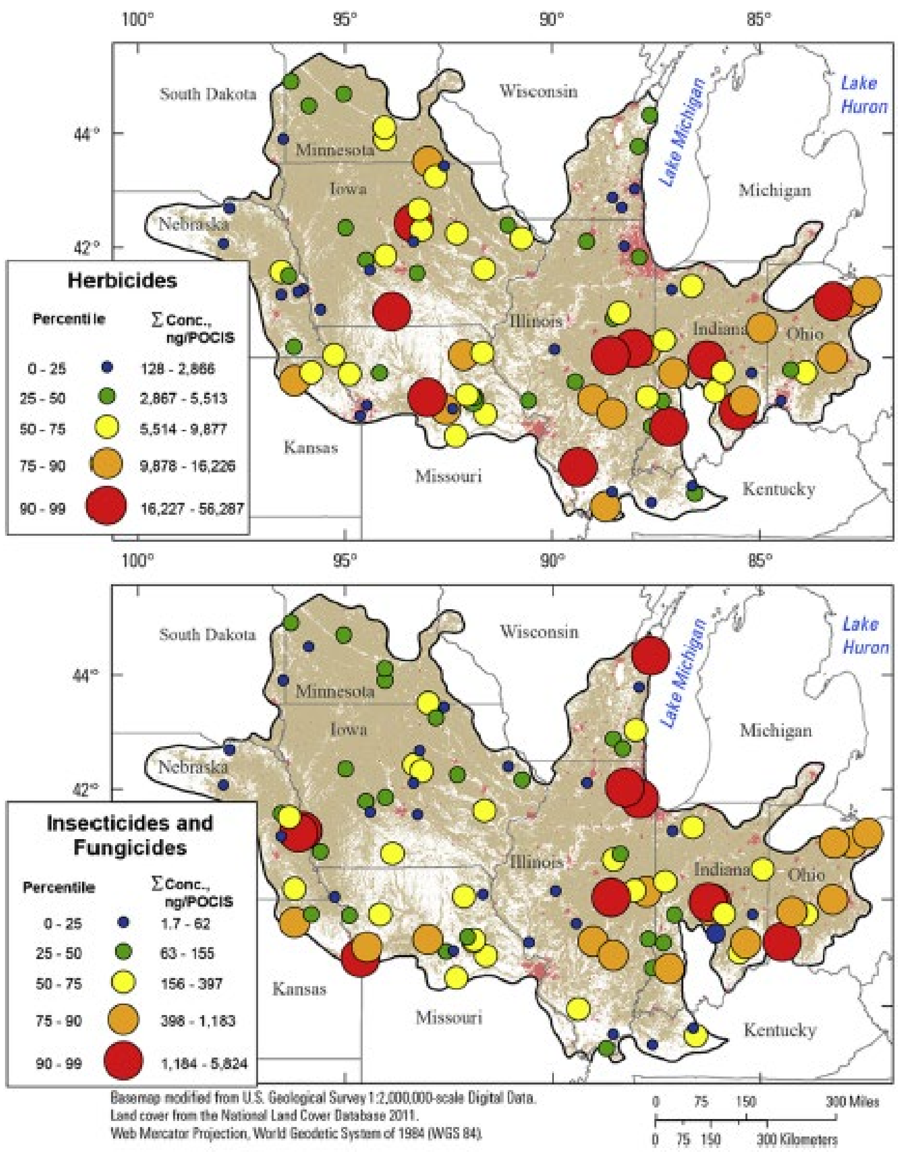
A newly published USGS analysis (Stackpoole et al., 2021) reported the results of pesticide sampling at river monitoring sites located throughout the conterminous United States. At least one pesticide was detected at 71 of the 74 sites, which were sampled biweekly to monthly from 2013 to 2017. On average 17 unique pesticides were detected at every site, and 105 of 221 studied pesticides were detected at least once. The most frequently detected herbicides were atrazine, metolachlor, and 2,4-D, and the most highly detected insecticides were acephate, imidacloprid, and carbaryl. All of these pesticides

[6] Application rates for pesticides on biofuel crops and the crops they often replace is discussed in Chapter 3 section 3.2.1.5.

are used on corn and/or soybeans as well as other crops. More pesticides were detected in rivers of the Midwest than in rivers of other regions.

Another study of nine streams in an area with intense corn and soybean production in Iowa found neonicotinoid residues at all sites, with a 75% detection frequency at up to a maximum concentration of 257 nanograms per liter (ng/L; ng = nanograms), thiamethoxam with a 47% detection frequency at up to 185 ng/L, and imidacloprid with a 23% detection frequency at up to 42.7 ng/L (Hladik et al., 2014). Although neonicotinoid usage in the corn belt is low compared with other pesticides (because they are

a



b

Figure 10.7. Locations of 97 MSQA sites where POCIS samplers were successfully deployed and summations of herbicides (a) and insecticides plus fungicides (b). Summations include degradates for the use group. Light brown shade is cropland, pink shade is urban land use. Source: Van Metre et al. (2017) (used with permission).

primarily used as seed coatings rather than as broadcast sprays or granular applications), they are relatively highly toxic to aquatic invertebrates, and thus of concern for aquatic ecological resources (see Chapter 15 for more details). In the Hladik et al. (2014) study, temporal concentration patterns revealed pulses associated with "rainfall events during crop planting." A 2012–2013 study on prairie wetlands in Saskatchewan found clothianidin and thiamethoxam in a majority of samples, at maximum concentrations of 3.1 and 1.5 micrograms per liter (µg/L), respectively (Main et al., 2014). Another study on floodplain wetlands in Missouri found neonicotinoid residues in a majority (63%) of sediment samples, at up to 17.99 micrograms per kilogram (µg/kg), and in water at up to 0.97 µg/L (Kuechle et al., 2019).

Concentrations were normalized to the mean deployment interval of 37 days. Heavily used corn and soybean herbicides are often detected in streams of the Midwest. For example, Fairbairn et al. (2016) monitored 26 hydrophilic and "moderately hydrophobic" (log octanol-water partition coefficient [KOW] <4) contaminants of emerging concern in 68 water samples collected in 2011 and 2012 in the Zumbro River watershed of Minnesota. Atrazine and metolochlor were detected in more than 70% of the samples, at maxima of 0.16 and 0.44 µg/L, respectively, while acetochlor was detected in more than 30% of samples, at a maximum concentration of 0.15 µg/L. Mahler et al. (2017) investigated temporal patterns in glyphosate and atrazine concentrations in Midwestern streams sampled under MSQA. Their analysis found that glyphosate was detected in 44% of samples (at up to 27.8 µg/L), and atrazine in 54% (at up to 120 µg/L). Atrazine's peaks were of longer duration than glyphosate's, though transport of both compounds "appeared to be controlled by spring flush." Summarizing the results of over 3,700 water and sediment samples collected in 38 states between 2001 and 2010, Battaglin et al. (2014) found that glyphosate and its degradate aminomethylphosphonic acid (AMPA) were usually detected together in water, though at concentrations "below levels of concern for humans or wildlife." More recently, in the "broadest survey of glyphosate in streams and rivers in the US to date," Medalie et al. (2020) found glyphosate and AMPA in 74% and 90% respectively, of 70 U.S. streams and rivers sampled between 2015 and 2017.

Figure 10.8 shows the USGS mapper trends in five of the seven most common pesticides in pesticides used on corn, for the time period from 2002 to 2012. Data for glyphosate and 2,4-D were not available in the mapper. The five pesticides were the only commonly used pesticides for corn (from MSQA above) that were available on the mapper. Atrazine shows a downward trend across much of the Midwest. In contrast, metalochlor and acetochlor concentrations are increasing in many areas.

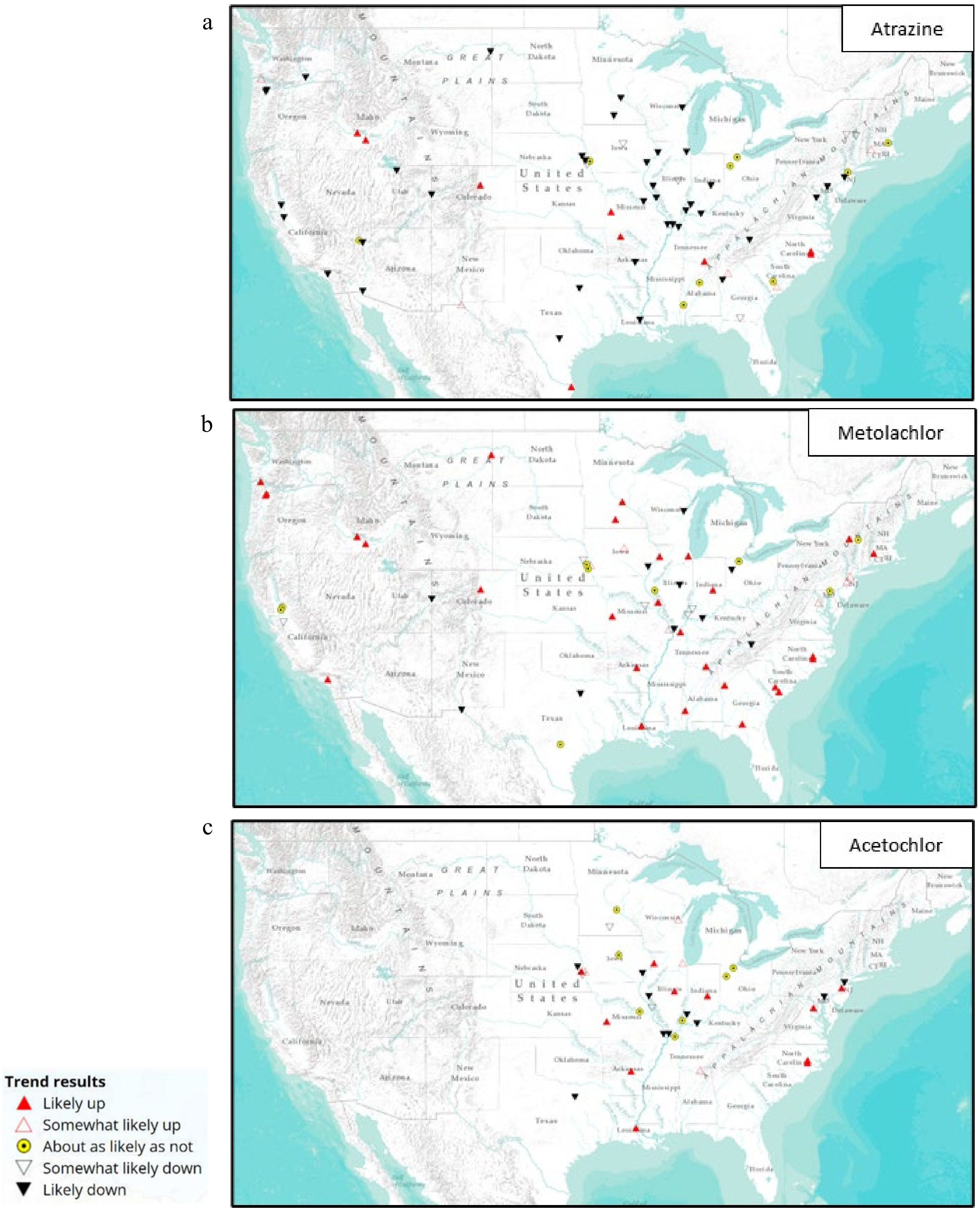


Figure 10.8. USGS mapper tool showing pesticide concentration trends between 2002 and 2012 for five pesticides commonly used on corn. U.S. Geological Survey, Water-Quality Changes in the Nation's Streams and Rivers, https://nawqatrends.wim.usgs.gov/swtrends/. (continued)

d

e

Figure 10.8 (continued). USGS mapper tool showing pesticide concentration trends between 2002 and 2012 for five pesticides commonly used on corn. U.S. Geological Survey, Water-Quality Changes in the Nation's Streams and Rivers, https://nawqatrends.wim.usgs.gov/swtrends/.

10.3.1.4 Pesticides in Drinking Water

The conversion of forest lands, grasslands, or other non-agricultural lands to biofuels could increase pesticide transport to water bodies (Arshad, 2018; Toccalino et al., 2014; Searchinger and Heimlich, 2009) and potentially impact ambient water quality and drinking water supplies (Sjerps et al., 2019; Noori et al., 2018; Klarich et al., 2017). Of the top 60 corn belt pesticides by usage identified by NAWQA, 23 have had MCLs set by the EPA under the Safe Drinking Water Act (SDWA) (U.S. EPA, 2022), as shown in Table 10.2 (five of these contaminants were banned from use sometime between 1985 and 2004). However, the conversion of row crops to perennial biofuels or changing crop type could also result in lower pesticide loads to the environment and water resources (Correa et al., 2019; Shah and Wu, 2019; Hoekman et al., 2018; Hossard et al., 2016; Dominguez-Faus et al., 2009; Paine et al., 1996). Also, the relative toxicities and modes of action of pesticides applied to different crops vary by crop type (Xue et al., 2015; Brown et al., 2007). Even though the biofuel industry has been around for multiple decades, it is challenging to distinguish the impact that can be attributed to biofuels and related specific land management decisions specifically on drinking water contamination (Thomas et al., 2014; Thomas et al., 2009).

Table 10.2. List of pesticides regulated under the SDWA.

Pesticides	**Names** (Applied to corn/soy; Y=yes; N=no)
Herbicides (11)	2,4-D (Y/Y); 2,4,5-TP (Silvex, Y/Y); Alachlor (aka LASSO, Y/Y); Atrazine (Y/Y); Dalapon (Y/Y); Dinoseb (Y/Y); Diquat (Y/Y); Endothall (N/N); Glyphosate (Y/Y); Picloram (N/N); Simazine (Y/N)
Insecticides (5)	Endrin (Y/N); Lindane (aka BHC-GAMMA, N/N); *Methoxychlor (2004, Y/Y); Oxamyl (Vydate, N/Y); *Toxaphene (1990, Y/Y)
Other Pesticides (6)	*1,2-Dibromo-3-Chloropropane (1985, N/Y); Carbofuran (Y/Y); *Chlordane (1988, Y/N); Pentachlorophenol (N/N); *Heptachlor (1974, N/N); Heptachlor epoxide (Y/Y)
No longer regulated, but within SDWIS (1)	Aldicarb sulfone (aka Aldoxycarb or Sulfocarb, Y/Y)

Source: U.S. EPA (2022)
*Indicates use of the contaminant is has been cancelled since sometime in the year listed in parentheses.

10.3.1.5 Potential Effects on Surface Water Temperatures

Corn management to supply biofuel feedstocks, for example leaving corn stover in place or removing it, may alter soil temperature, which can affect the temperature of surface water, negatively impacting water quality. Blanco-Canqui and Lal (2007) showed that removing 50% of corn stover can substantially increase soil temperature. For example, they observed that 75% stover removal resulted in an increase of soil temperature from 25.2 to 34.0°C at the depth of 2 inches of a silt loam soil. Sindelar et al. (2013) also found that corn stover removal/tillage increased soil temperature by as much as 4°C. Recent experiments (Haruna et al., 2017) further confirmed that perennial biofuel crops like switchgrass and cover crops could alter soil thermal properties, thereby stabilizing soil temperature and avoiding extreme fluctuations in soil thermal conditions. However, that change in soil temperature will likely affect temperature of terrestrial water flows that enter water bodies. Up to now, there is a lack of understanding and quantification of the impact that bioenergy crop management has on downstream water temperature, which is not only an important water quality indicator, but also directly influence other water quality parameters (e.g., dissolved oxygen, rate of chemical and biological reactions). Specific biofuel production management practices (e.g., retaining corn stover) could mitigate the effects on water temperature and thus water quality. This is further explored in the section below on “Conservation Practices”.

10.3.1.6 Potential Effects of Organic Carbon Leaching on Water Quality

Natural dissolved organic carbon (DOC) in drinking water is of concern because it can interact with other constituents to influence water quality. For example, DOC may interact with disinfectants to form toxic Disinfection By-Products (DBP) in drinking water supplies (U.S. EPA, 2005). This concern generally applies to surface waters because the presence of naturally occurring organic matter is much lower in groundwater. Different land use types (such as cropland, grassland, wetland, and forest) and areas with differences in soil organic matter content and sorption may yield varying levels of DOC leaching from soils into surface water or groundwater. Grassland had higher levels of DOC leaching (5.3±2.0 grams per square meter per year [$g/m^2/yr$]) than cropland (4.1±1.3 $g/m^2/yr$) (Kindler et al., 2011). Therefore, the choices between perennial grasses (e.g., switchgrass) or corn/soy as bioenergy feedstocks can influence the inputs of DOC into surface and groundwater used as drinking water sources. Furthermore, presence of dissolved organic matter could influence toxicity of herbicides (Coquillé et al., 2018), concentrations in sediments (Hung et al., 2007), bioconcentration of organic chemicals in aquatic organisms (Haitzer et al., 1998), and environmental fate of metals (Aiken et al., 2011). The linkages between leaching of DOC associated with biofuel production and surface water, groundwater, and drinking water quality are not well studied, but there is potential for important interactions to occur.

Organics can combine with disinfectants (e.g., chlorine, chloramines) when mixed at water treatment plants and distribution systems to form organic DBPs, including trihalomethanes (THM) and haloacetic acids (HAA) (Carpenter et al., 2013; Sham et al., 2013; Edzwald, 2011). Increased sediment and DOC reaching a treatment plant can result in the public water supply needing increased use of chlorine and other disinfectants to maintain treatment efficiency, exacerbating the formation of DBPs (Hohner et al., 2019; Richardson et al., 2007; Boorman, 1999; Singer, 1994). A number of studies also found a positive relationship between total organic carbon (TOC) and THM and HAA in treated drinking water (Chow et al., 2019; Evans et al., 2019; Hohner et al., 2019; Hohner et al., 2016). Like DOC and TOC, increased dissolved organic nitrogen (DON) contributes to both regulated and non-regulated DBP formation (Emelko et al., 2011). Increased total suspended sediment (TSS) and dissolved organic matter (e.g., DOC, TOC, DON) may reduce the coagulation ability of treatment plants, which could increase the need for disinfectants, resulting in greater DBP formation (Hohner et al., 2019).

10.3.1.7 Underground Storage Tanks Systems

Releases from underground storage tank systems (USTs) can threaten human health and the environment, contaminating both soil and groundwater. From the beginning of the UST program in 1988 to September 2019, 555,384 UST releases have been confirmed across the United States (U.S. EPA, 2020a). Of these, 490,624 have reached cleanup completed status, leaving a backlog of 64,760 sites that have not yet reached cleanup completed status. Since the mid-2000s most releases of regulated substances reported to the EPA Office of Underground Storage Tanks, which regularly exceed 5,000 per year, contain petroleum/biofuel blends, since those fuels are ubiquitous across the country (U.S. EPA, 2020a).

Many of those historical releases contain gasoline/ethanol blends since E10 is commonly used across the country; similarly, many diesel releases are likely diesel/biodiesel blends, since diesel may contain biodiesel up to 5% by volume. No release data exists to determine the percentage of historical releases or active cleanup sites that contain biofuels as a portion of the fuel release. However, as discussed in Chapter 6, given that nearly all biofuel after roughly 2013 was E10, and fairly little outside of the Midwest and California prior to 2005 was E10, one can assume that most gasoline releases since that time contain ethanol, and many diesel releases since that time contain small amounts of biodiesel.

10.3.2 New Analysis

The Missouri River Basin (MORB) has experienced an increase of grassland conversion to crop production in recent years (Lark et al., 2020; Wright et al., 2017; Lark et al., 2015), due in part to increased production of corn and soybeans in the vicinity of biorefineries. Increased crop production can adversely impact water quality and ecosystem services relative to grasslands. In support of the RtC3, to estimate the water quality changes resulting from these recent land use changes (Chen et al., 2021) applied the SWAT to the MORB, where the greatest cropland increase has been observed.[7] This SWAT model run was constructed using data collected from various sources including weather, soil, and land use. Eight-digit Hydrologic Unit Codes (HUC8s) were used as pre-defined sub-watersheds. The USDA Cropland Data Layer (CDL) for 2008 and 2009 (Figure 10.9) was used as the initial baseline. The model was then calibrated and validated using USGS monitoring data.

After model calibration and validation, the model was used to simulate three crop production scenarios representing conversion of grassland from Lark et al. (2020) (S1) to either: continuous corn (S2); corn/soybean rotation (S3); or corn/wheat rotation (S4). Conversion was simulated only in the locations of observed land use changes from Lark et al. (2020) over two periods, from 2008 to 2012 and from 2008 to 2016 (Figure 10.10). The SWAT model was used to estimate stream flow and riverine sediment and nutrient loads throughout the MORB after model calibration and validation (Chen et al., 2021).

[7] This SWAT analysis compliments the EPIC analysis described in Chapter 9 section 9.3.2, which used the same land use change estimates from Lark et al. (2020). Details of the SWAT and EPIC analyses, respectively, are available in Chen et al. (2021), and Zhang et al. (2021).

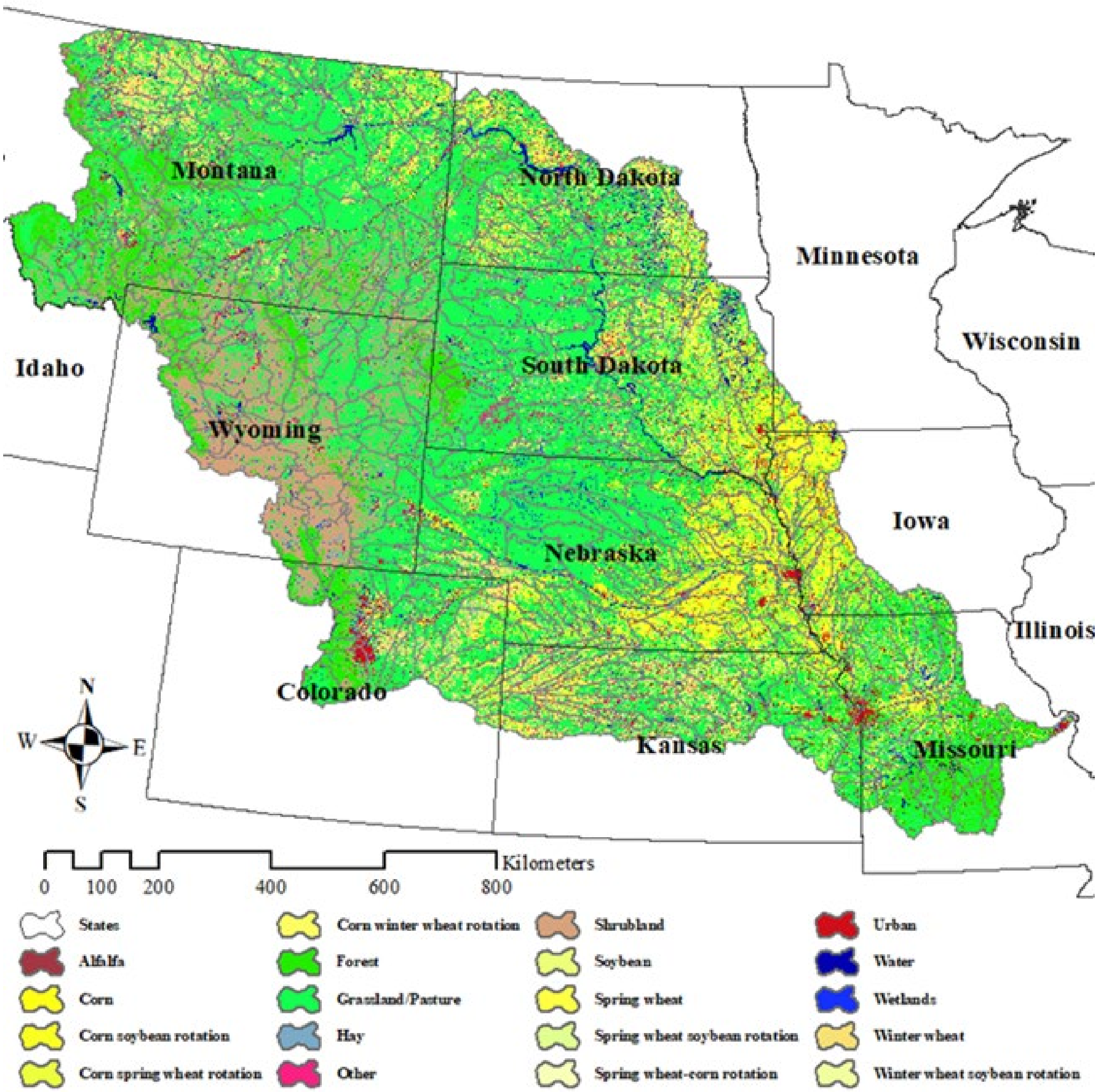


Figure 10.9. Missouri River Basin and its 2008/2009 land use/land cover based on Cropland Data Layer. Source: Chen et al. (2021). (Creative Commons Attribution License - CC BY).

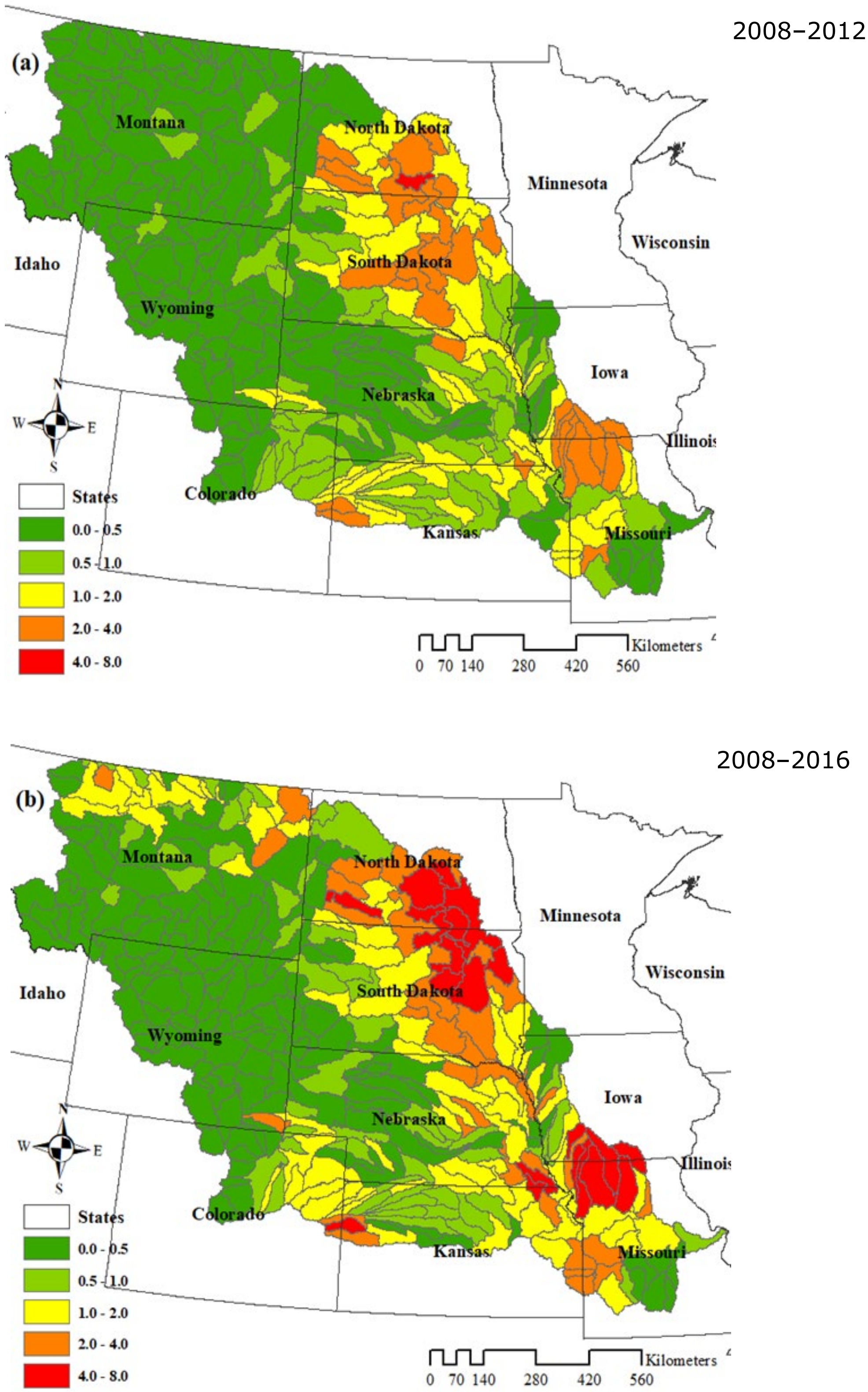
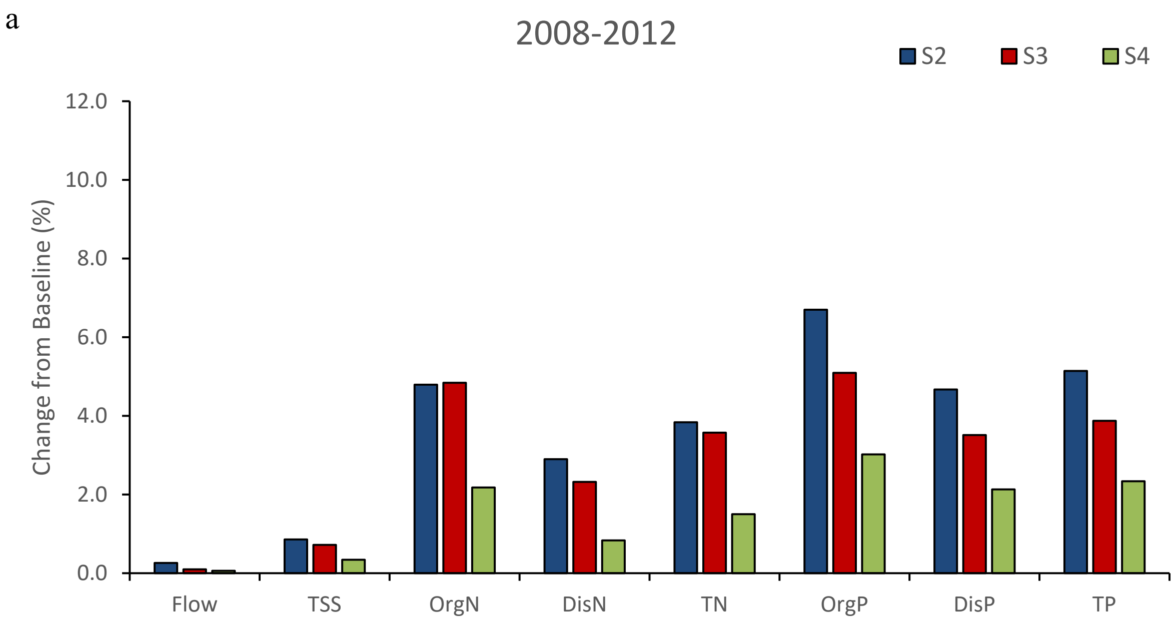


Figure 10.10. Percentage of area converted from non-crop land to crop land in each eight-digit Hydrological Unit Code 8 (HUC8) during 2008–2012 (a) and 2008–2016 (b). Source: Chen et al. (2021). (Creative Commons Attribution License - CC BY).

The differences between baseline and different conversion scenarios on streamflow and sediment are trivial at the watershed outlet, but nutrient export from the watershed increased for all crop conversion scenarios (Figure 10.11). Comparing water quality between 2008 to 2012 and 2008 to 2016, changes from 2008 to 2016 are similar to that of 2008 to 2012 with larger magnitudes (Figure 10.11). This is because cropland expansion was estimated to continue from 2012 to 2016 in Lark et al. (2020). The non-cropland to cropland conversion was 0.77% for the period of 2008–2012, and was 1.18% for the period of 2008–2016 (Figure 10.10). Therefore, adverse impacts on water quality were estimated to continue to increase due to the ongoing cropland expansion between 2012 and 2016. The water quality changes from 2008 to 2016 are about 1.5 times those observed from 2008 to 2012, consistent with the magnitude of increased cropland conversion (Figure 10.11).

For the time period from 2008 to 2016, the SWAT model results showed that at the MORB outlet: grassland (S1) conversion to continuous corn (S2) resulted in the greatest increase in TN and TP loads (6.4% and 8.7% increase, respectively); followed by conversion to corn/soybean (S3) (TN increased 6.0% and TP increased 6.5%); and then conversion to corn/wheat (S4) (TN increased 2.5% and TP increased 3.9%). Across the watersheds in the MORB, the greatest percentage increases of TN and TP occurred in North Dakota and South Dakota, coinciding with the highest amount of grassland conversion (Chapter 9, Figure 9.3). However, these areas still contributed relatively low absolute amounts of TN and TP to the total basin loads due to a relatively low percentage of cropland in these areas (compare Figure 10.12 and Figure 10.13). Rather than homogeneous effects, specific watersheds appear to be "hotspots" of change—predominantly in Iowa, Missouri, Nebraska, and Kansas—and contributed the greatest amounts of TN and TP to basin-wide loads (Figure 10.12), driven by a combination of grassland conversion, precipitation, and loading from pre-existing cropland. The spatial pattern of unit area changes (Figure 10.12) and percentages changes (Figure 10.13) between two periods are also similar. How these fluxes are converted to stream concentrations, and how they relate to different thresholds for ecological effects, are discussed in Chapter 13 section 13.3.2.1.



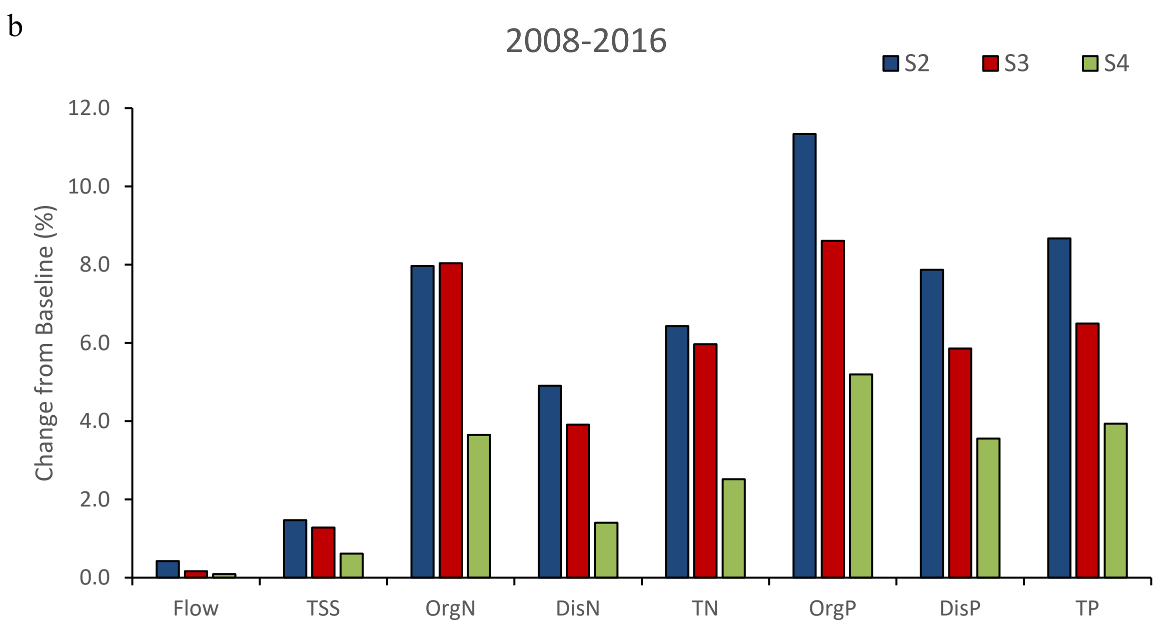
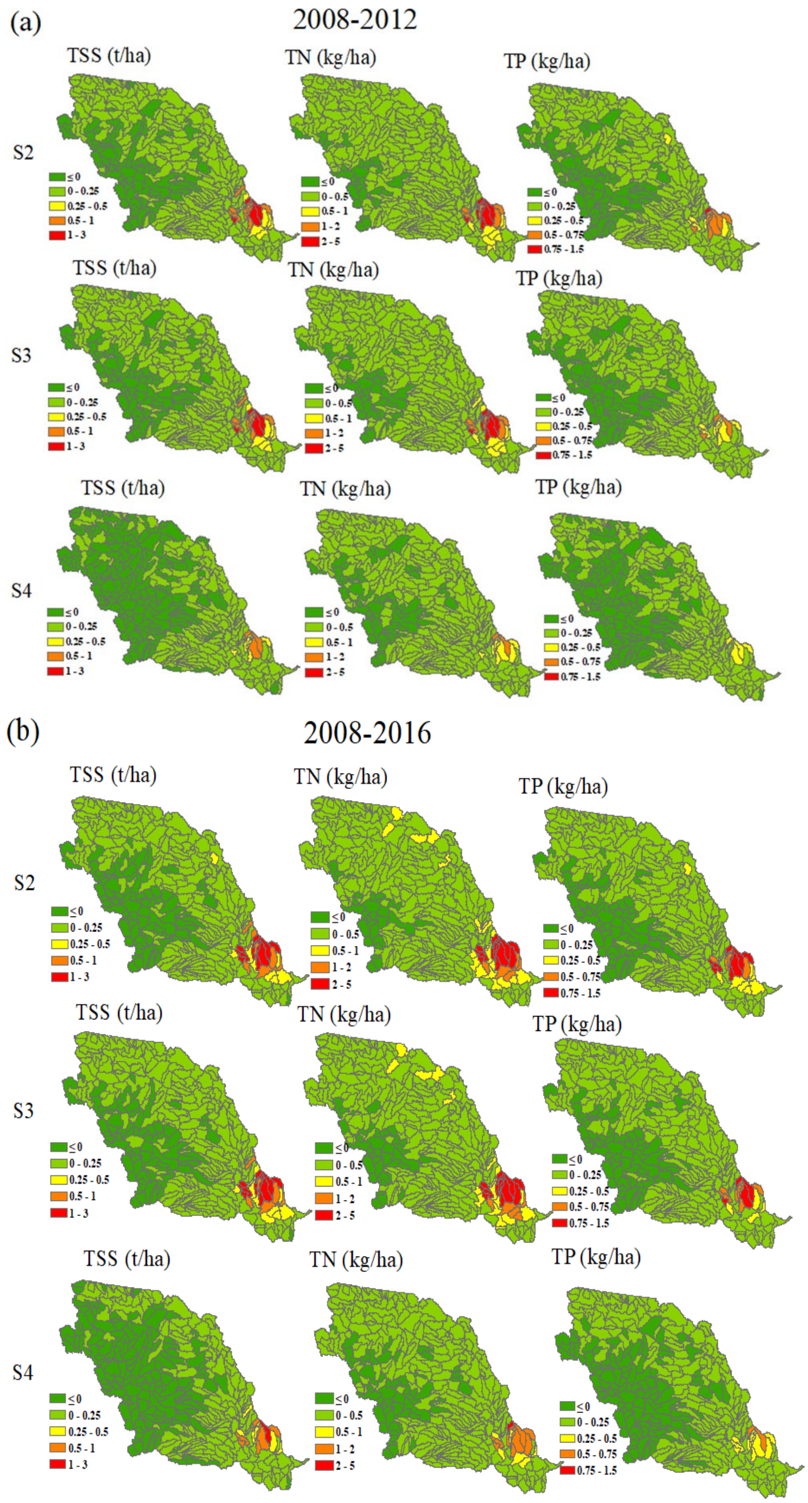


Figure 10.11. Summary of results at the MORB outlet. Shown are the mean annual changes in flow, total suspended sediment (TSS), organic nitrogen (OrgN, including organic and ammonium nitrogen), dissolved nitrogen (DisN, including nitrate and nitrite), total nitrogen (TN), organic phosphorus (OrgP), dissolved phosphorus (DisP, referring to mineral phosphorus), and total phosphorus (TP) loads between the baseline scenario and different biofuel scenarios (S2, S3, S4) during 2008–2012 (a) and 2008–2016 (b). Source: Chen et al. (2021). (Creative Commons Attribution License - CC BY).



ha = hectare; kg = kilogram; t = metric tonnes

Figure 10.12. Differences in per unit area (refer to per hectare of watershed) of total suspended sediment (TSS), total nitrogen (TN), and total phosphorus (TP) at S2 (baseline vs. continuous corn), S3 (baseline vs. corn/soybean), and S4 (baseline vs. corn/wheat) during 2008–2012 (a) and 2008–2016 (b) in the southeastern portion of the Missouri River Basin. Source: Chen et al. (2021). (Creative Commons Attribution License - CC BY).

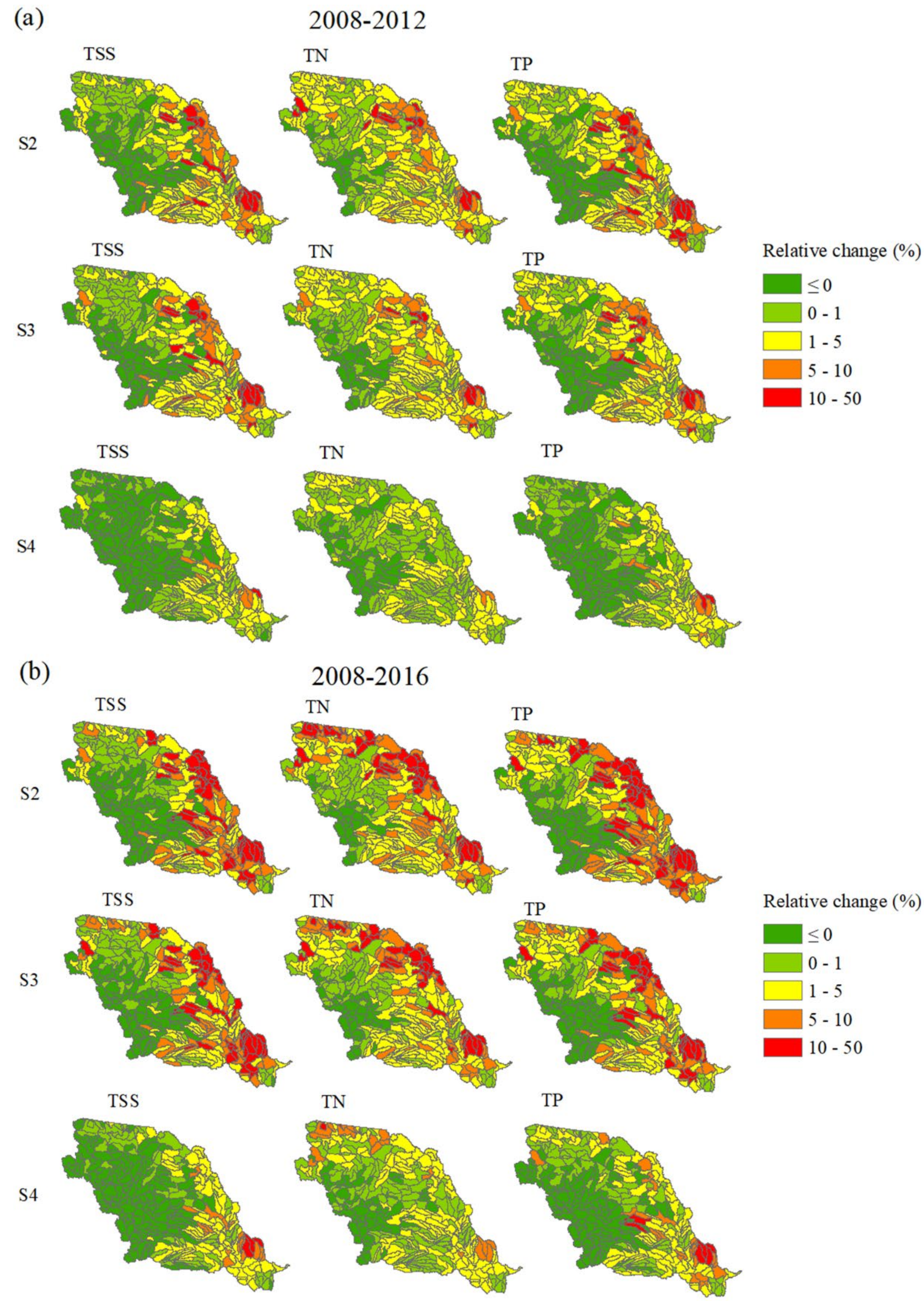


Figure 10.13. Percent differences relative to baseline for total suspended sediment (TSS), total nitrogen (TN), and total phosphorus (TP) for S2 (baseline vs. continuous corn), S3 (baseline vs. corn/soybean), and S4 (baseline vs. corn/wheat) during 2008–2012 (a) and 2008–2016 (b) in the southeastern portion of the Missouri River Basin. Source: Chen et al. (2021). (Creative Commons Attribution License - CC BY).

As with the EPIC modeling discussed in Chapter 9, several points should be considered when assessing the SWAT results above. First, the converted and abandoned lands were not specific to biofuel feedstock production, but rather agriculture in general. Hence, the results should be viewed as the water quality effects of general agricultural expansion in the MORB, while the effects attributable to the RFS Program specifically are a proportion of the total shown (see Chapter 6 and Chapter 10, section 10.3.3). Second, the crop and tillage types for each specific parcel converted could not be computationally modeled. Therefore, three general scenarios of conversion for the entire study area were examined. All lands in the MORB did not convert to the same agricultural practice, but what actually occurred is likely less than the S2 scenario and is some combination of the three. Almost 60% of acres converted nationally were planted with corn (29.3%) and soybeans (26.7%) (Lark et al., 2020), and this percentage was over 70% for the area modeled. Third, the grassland parcels likely represented a spectrum of grassland and management types, including pasture, lands managed for hay, and CRP grasslands, not necessarily solely undisturbed or unmanaged grasslands prior to conversion. Fourth, some of the SWAT model parameterization is based on data from the early 2000s, which could underestimate conservation practice adoption. For example, EPIC and SWAT models are often parametrized with older crop growth and crop characteristics that may not represent current crop performance well (Ren et al., 2022). Thus, overall, these simulations are not intended to represent the precise changes across the Midwest from 2008 to 2016. Rather they provide the directionality of effects (whether negative, positive, or no effect), and a range of estimated effects, with the actual effect likely in between the ranges shown in these simulations.

Analysis of the NAWQA data trends from 2002–2012 found that a number of stations increased in TN and TP, especially in the Southern Plains ecoregion (Figure 10.3 above). The EPA NARS data indicate that the proportion of poor TN condition streams increased by approximately 6% in the Southern Plains, while stream TN condition improved in the Northern and Temperate Plains (Figure 10.4). Therefore the reconstructive modeling results are comparable to the small observed ±10% changes in TN and TP concentrations and loads between 2000 and 2014.

10.3.3 Attribution to the RFS Program

Chapter 5 presented general trends in land use change in the United States irrespective of cause, and Chapter 6 quantified the subset of that estimated to be attributable to corn ethanol and corn associated with the RFS Program (see Chapter 6, section 6.4.3). Lark et al. (2020) report that corn was the predominant crop planted on these lands newly converted to cropland between 2008 and 2016, and results from Chapter 6 suggest that approximately 0–20% of the cropland expansion is estimated to be due to the RFS Program. Thus, the initial estimate of the effects from the RFS Program on water quality *from the* *expansion of cropland* alone is approximately 0–20% of the results presented in section 10.3.2.[8] As noted in section 10.3.2, the actual crops grown on newly converted lands are likely some mixture of the three scenarios examined, with the actual effect from cropland expansion constrained by 0–20% of the S2 scenario (conversion to continuous corn; Figure 10.11–Figure 10.13), and likely lower than that. On the other hand, the effects of biofuel expansion and the RFS Program may each be larger than the results above for at least two reasons: (1) biofuel expansion and the RFS Program may not only have affected cropland expansion, but also crop switching from other crops to corn (e.g., Ren et al., 2016; Plourde et al., 2013) which are not included in the Lark et al. (2020) rasters and often lead to increased levels of fertilization (see Chapter 3, section 3.2.1.6); and (2) biofuel expansion and the RFS Program may have also induced increases in soybean biodiesel which would affect soybean acreage and additional cropland expansion. The latter effects in Chapter 7 could not be quantitatively estimated.

10.3.4 Conservation Practices

The U.S. Department of Agriculture (USDA) National Resources Conservation Service (NRCS) has promoted the adoption of conservation practices on cultivated cropland for mitigating agricultural nonpoint source nutrient pollution since 1985. Conservation practices for corn and soybean production land include cover crop, crop rotation, reduced tillage, riparian forest buffer, saturated buffer, grassed waterway, nutrient management, drainage management, bioreactors, constructed wetland, and others. They are designed to conserve water and reduce the amount of nutrients and other pollutants entering water resources while maintaining or even enhancing agricultural production.

10.3.4.1 Conservation Effects Assessment Project

The Conservation Effects Assessment Project (CEAP) is a multiagency effort led by the NRCS to quantify the environmental effects of conservation practices and develop the science to manage the agricultural landscape for environmental quality. Initiated in 2003, CEAP is built from five components to achieve its goals: cropland, watersheds, wetlands, grazing lands, and wildlife.

The cropland component assesses the impacts of voluntary conservation on edge-of-field losses of nutrients, sediment, and pesticides and other physical processes such as soil carbon sequestration at national and regional scales. This assessment is built from a nationally distributed farmer survey and a field-scale modeling effort. Statistically sampled from the NRI framework, the farmer survey has been conducted twice, in 2003–2006 (CEAP I) and 2013–2016 (CEAP II), covering 12,000–18,000 cropland

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[8] The finding from Chapter 6 that the 20% estimate is the largest effect in a single year (i.e., 1.9 million acres in 2016, Tables 6.10 and 6.11) does not affect the results here. SWAT estimates differences between scenarios over equilibrium conditions. The 1.9 million acres of new cropland estimated attributable to the RFS in 2016 are assumed to continue to be cultivated after 2016, such that the largest potential effects on the environment are estimated from the effect in 2016.

fields each. It is a comprehensive interview regarding all operations, conservation practices, irrigation, and nutrient and pesticide applications, regardless of whether NRCS provided conservation assistance.

The farmer survey data are used in the Agricultural Policy Extender (APEX) model to estimate edge-of-field losses. APEX is a field-scale, process-based model that simulates interactions between weather, farming operations, crop growth and yield, and the movement of water, soil, carbon, nutrients, sediment, and pesticides. CEAP I results showed that nationally the conservation on the ground from 2003 to 2006 decreased sediment losses from water erosion by 53% (278.1 million tons per year), N surface losses by 41% (1.7 billion pounds per year), N subsurface losses by 31% (2.1 billion pounds per year), and P losses by 44% (584.1 million pounds per year). Estimated N losses came primarily from two pathways, subsurface flow (44%) and volatilization (19%), and from a minority of acres, with just 8% of acres showing total losses greater than 70 pounds per acre per year and 10% (29 million acres) showing surface water runoff losses of greater than 15 pounds per acre per year. Often, these acres needing the most treatment to prevent losses are interspersed throughout the landscape, requiring precision management within an individual field for best conservation results. While 94% of acres had at least one nitrogen fertilizer management practice on them, only 28% of acres met full N management criteria considering rate, timing, and method of application. These findings suggest there is room for improvement in application of these voluntary conservation practices and that targeting to the land that needs the practices most may realize lower edge-of-field N loss in the future.

CEAP II data allow comparison of changes in both practice adoption and estimated edge-of-field losses over time (USDA NRCS, 2022). There were numerous benefits in terms of reducing surface nutrient losses, for example N and P losses through surface hydrologic pathways declined by 3% and 6%, respectively. However, changes in crops and tillage systems outpaced the capacity to retain nutrients efficiently, most notably in the northern and southern plains where corn and soybean production replaced wheat and other crops that had lower average nutrient needs and fallow periods. Application rates of N and P in fertilizer increased by 7% and 15% for N and P respectively, and corn yields increased by 14% between the survey periods. While sediment management practices resulted in substantial declines in sediment load (22%), subsurface losses of N and soluble P increased by 13% and 11%, respectively. Subsurface losses include natural lateral drainage, deep drainage, and tile and ditch drainage. The expansion of crops, such as corn, with higher nutrient demand and conservation tillage systems, appear to have promoted infiltration and subsurface flow of soluble nutrients. Conservation tillage systems reduced the risk of N loss through surface pathways and increased infiltration for subsurface flow, while the increase in surface application of fertilizer promoted surface conversion to soluble nitrogen and movement through the soil profile. Overall, there were some improvements between CEAP I (2003–

2006) and CEAP II (2013–2016) in terms of the acreages exceeding resource thresholds for erosion, sediment and surface losses, and deterioration for subsurface losses (Table 10.3).

USDA-NRCS recognizes that the conservation needs vary within and among fields, and considers addressing soil health and nutrient management as a system critical to achieving the full benefits of advanced technology, tillage efficiency, and conservation measures. For example, in each CEAP survey period, a minority of acres accounted for most nutrient and sediment losses. In CEAP II, 28% of acres were responsible for 73% of the subsurface N losses, with similar findings for P. Challenges in optimizing both erosion control and nutrient management to reduce subsurface nutrient losses point to the need for precision technologies such as variable rate applications and enhanced efficiency fertilizers.

The other four CEAP components provide additional data to support conservation decision making on and off croplands. The watersheds component gathers on-the-ground and SWAT-modeled estimates of the effects of conservation on watershed-level water quality. This work has helped validate the APEX cropland modeling and has been used to fill knowledge gaps and assist with data needs of major watersheds of concern such as the Chesapeake Bay or Western Lake Erie Basin. The wetlands component collects field and remotely sensed data to help determine the benefits provided by natural and restored wetlands, and to help guide decisions on where these wetlands may be best placed to maximize

Table 10.3. Cultivated cropland exceeding resource thresholds by survey. Source: USDA NRCS (2022).

	CEAP I		CEAP II		CEAP II minus CEAP I	
Resource Concern (Loss Threshold)	Acres (1,000s)	Percent of Acres	Acres (1,000s)	Percent of Acres	Acres (1,000s)	Percent of Acres Relative to CEAP I
Sheet & Rill Erosion (>T)	35,519	11	31,171	10	-4,348	-12
Wind Erosion (>T)	38,634	12	30,994	10	-7,640	-20
Sediment (>2 t/a/y)	38,113	12	29,335	9	-8,778	-23
Surface Nitrogen (>15 lbs/a/y)	35,084	11	33,946	11	-1,138	-3
Sediment-Transported Phosphorus (>3 lbs/a/y)	35,211	11	33,630	11	-1,581	-4
Subsurface Nitrogen (>25 lbs/a/y)	74,779	24	88,914	28	14,135	19
Soluble Phosphorus (>0.5 lbs/a/y)	72,909	23	84,361	27	11,452	16
Soil Carbon (Maintaining/Losing)	49,703	16	48,511	15	-1,192	-2

lbs/a/y = pounds per acre per year; t/a/y = tons per acre per year; T = threshold ecosystem services and water quality benefits. The grazing lands and wildlife components conduct studies estimating the impacts of management practices and valuing the ecosystem services provided by these critical habitat areas. Additional efforts to analyze and assess conservation practices are discussed in the Horizon Scanning section (section 10.6).

10.3.4.2 Conservation Modeling Scenarios

Many studies have used modeling to examine different potential portfolios of conservation that may improve watershed conditions. Modeling scenarios using SWAT (https://swat.tamu.edu/) suggest that conservation practices (e.g., filter strips, cover crops, riparian buffers) can help achieve environmental goals. For example, TP targets can be met with conservation practices, whereas dissolved reactive P is much more responsive to reductions of P application to fields. Modeling also suggests that conversion to perennial grasses such as switchgrass and Miscanthus, even with manure application, would significantly reduce P runoff into water bodies (Muenich et al., 2016). Conservation tillage (no-till and reduced till) has demonstrated positive effects on reducing soil erosion. To date, reduced tillage has become dominant in corn and soybean farms across Corn Belt regions [see Chapter 3 and Baranski et al. (2018)]. Although a transition from conventional till to no-till reduces P loss, the effect on nitrate is estimated to be limited (Demissie et al., 2017). Simulated winter cover crops after corn harvest led to reductions of 20–30% for N, and 20–40% for P and suspended sediments (SS), compared to historical baseline conditions, which is consistent for several Midwest watershed studies (Ha et al., 2020; Gassman et al., 2017). In 2005, the USDA developed a Denitrifying Bioreactor conservation practice standard. Wood chip bioreactors have been shown to achieve 33% annual nitrate load removal in tile drain applications (Christianson et al., 2012), with N removal rates averaging 4.7 gallons of N removed per bioreactor (Addy et al., 2016). With a lifespan of 7–15 years (Christianson et al., 2012), denitrifying bioreactors have shown potential to help with significant water quality challenges.

Multipurpose vegetative buffers, especially riparian buffers, have been demonstrated as effective in trapping nutrients, reducing soil loss, and increasing soil organic carbon to restore ecosystem services (Christianson et al., 2018; Ha and Wu, 2017; Kalcic et al., 2015; Moore et al., 2014; Fageria et al., 2005). In a corn-soybean dominated watershed in Iowa—South Fork of the Iowa River (Figure 10.14)—SWAT predictions for the riparian buffer were for reductions of approximately 131 metric tonnes (MT) (5%) of N, 7.5 MT (30%) of P, and 24,030 MT (62%) of SS, when a 33 yard riparian buffer is installed in the stream network (Wu and Ha, 2017). When the buffer is extended to 90 meter, up to 17%, 37%, and 70% for N, P, and SS, respectively can be reduced (Ha et al., 2020). More recently, a multistakeholder effort compared three types of buffers: riparian buffer (RB), riparian buffer/saturated buffers (RBSB), and grassed waterways (GRSW) (Ha et al., 2020). In response to the buffers, nutrients and sediment loadings can decrease by up to 1.14 metric tons per hectare (MT/ha) of SS, 5.43 kg/ha nitrate, 7.23 kg/ha TN, and 2.07 kg/ha across the watershed (Figure 10.5). RBSB was the most effective in reducing TN (7.23 kg/ha) and nitrate-N loadings (5.43 kg/ha), followed by RB. N reductions by GRSW were limited. The three practices had a similar effect on sediment loadings. P changes among the three buffers were similar to those for SS. Results demonstrated those practices can be effective in reducing the direct entry of sediments and nutrients in these watersheds, as reported for numerous other watersheds.

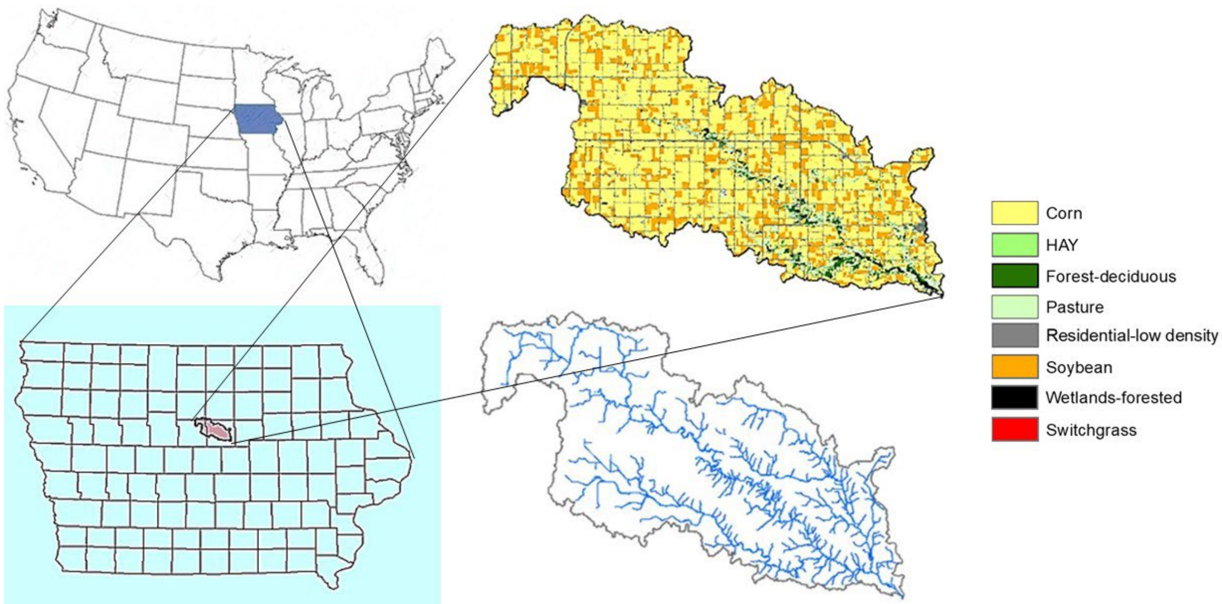
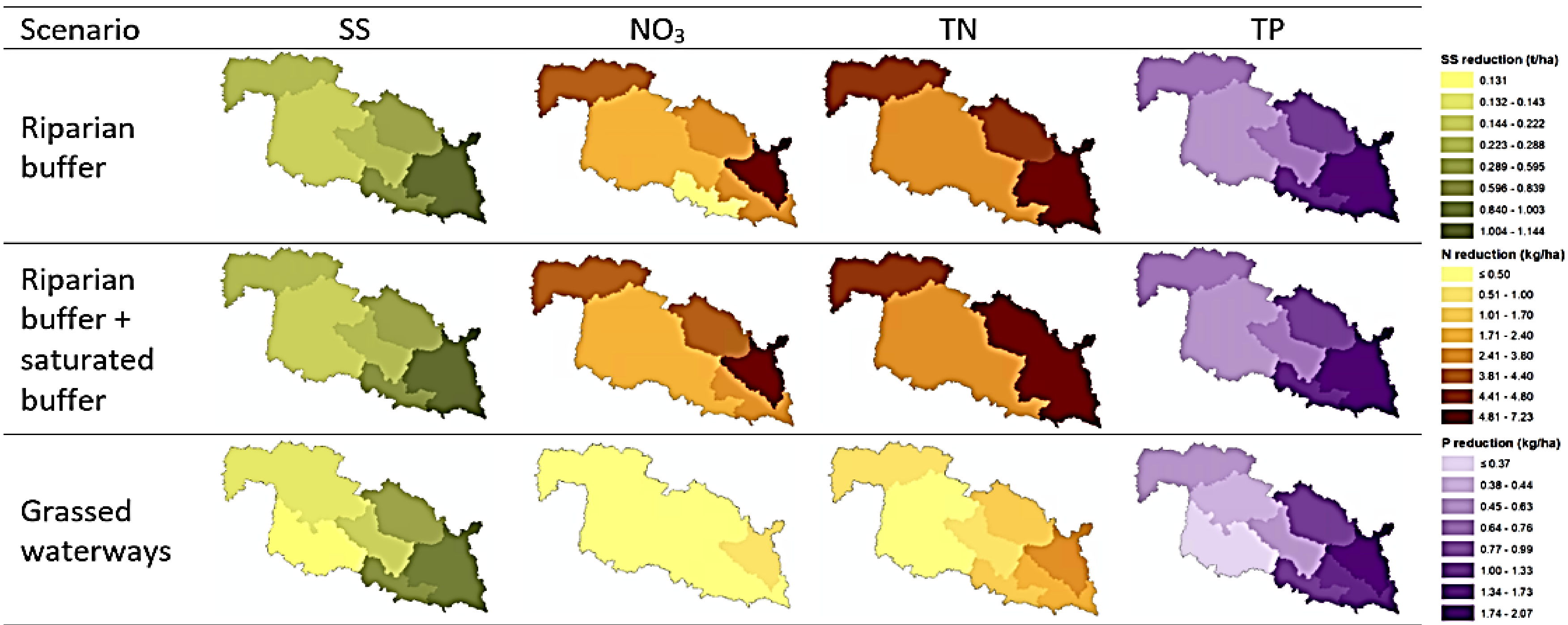


Figure 10.14. Location of and land use within in the South Fork of Iowa River watershed, Iowa. Corn and soybean are the predominant land use by far. Source: Wu and Ha (2017) (used with permission).



ha = hectare; kg = kilogram; t = metric tonnes

Figure 10.15. Spatial distribution of suspended sediments (–SS - t/ha), nitrate (NO_3 – N kg/ha), total nitrogen –TN - kg/ha), and total phosphorus –TP - kg/ha) loading reductions after conservation practices riparian buffer (RB), saturated buffer (SB), and grassed waterway (GRSW) were applied for the South Fork of Iowa River. Source: Ha et al. (2020). (Creative Commons Attribution - NonCommercial-NoDerivs License; no changes made).

At the large basin scale, a study by Ha et al. (2018) for the Lower Mississippi River Basin (LMRB) concluded that implementing a riparian buffer in the agricultural region within the LMRB could reduce N, P, and SS loadings by up to 65%, 35%, and 39%, respectively [Figure 10.16; (Xu et al., 2019)]. Implementation of this approach can potentially improve the water quality of the discharge from the LMRB into the Gulf of Mexico. The value of nutrient abatement by using trapped nutrients as fertilizer to grow riparian buffer was quantified by Xu et al. (2019). The value of trapped nutrients is considerable (mean = $69/ha/year) but far less than the cost of implementing a switchgrass buffer (mean = $163/ha/year) (Figure 10.17). Factors of future feedstock price, fertilizer prices, and forgone income could all impact the outcome. The economics of reducing nutrient loss from cropland by implementing switchgrass and riparian buffers and harvest as feedstock would be highly dependent on the cellulosic biomass market. Future work could improve upon these estimates since the value of nutrients trapped or stored in soil and biomass of riparian buffers may or may not be well-approximated by the value of those nutrients in commercial fertilizers (e.g., on an atomic mass equivalence basis) because those nutrients are not in a production area (crop fields adjacent to riparian buffers).

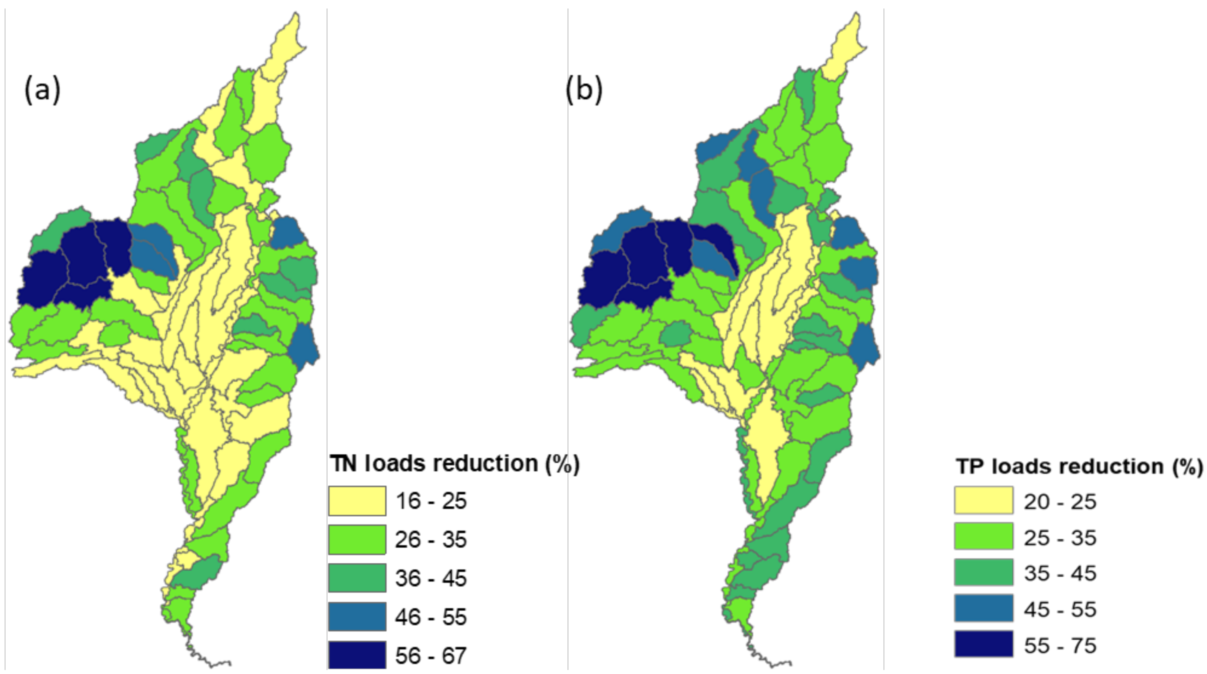
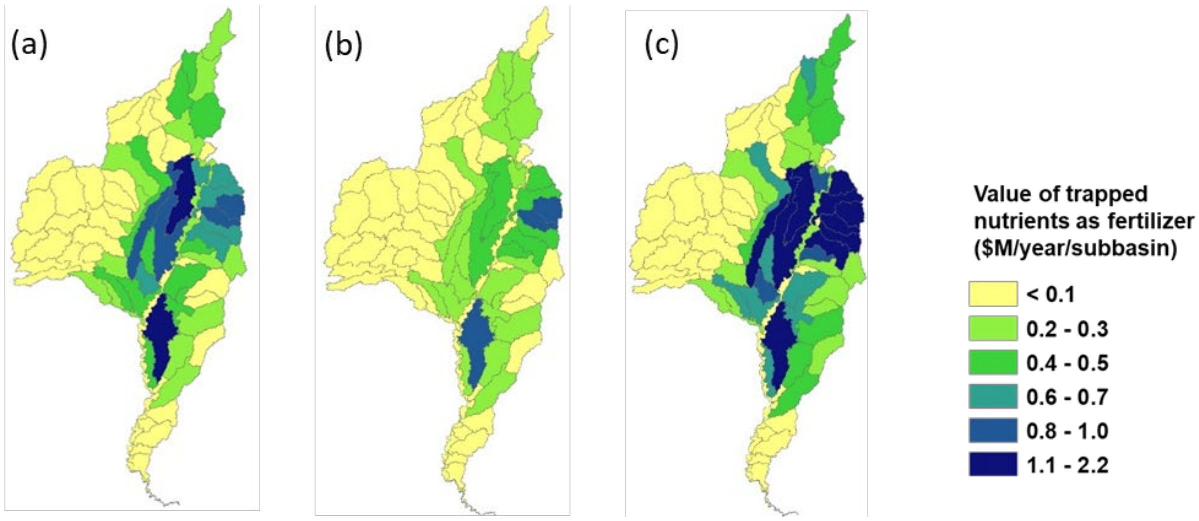


Figure 10.16. Spatial distribution of reductions in annual total nutrient loads discharged from cropland after riparian buffers were installed in Lower Mississippi River Basin. Panels (a) and (b) show percentage reductions in annual total nitrogen (TN) and total phosphorus (TP) loads at the subbasin level. Source: Xu et al. (2019). (Creative Commons Attribution License - CC BY).



M = million

Figure 10.17. Economic value of total nitrogen (TN) (a), total phosphorus (TP) (b), and TN and TP (c) stored in riparian buffer zone at subbasin level. Nutrient value refers to the value of TN and TP stored in the riparian buffer zone, estimated using nitrogen and phosphorus fertilizer prices. Twenty-one-year (1990–2010) average mean annual reductions in TN and TP after RB implementations were simulated using the SWAT model. Source: Xu et al. (2019). (Creative Commons Attribution License - CC BY).

10.4 Likely Future Impacts

As noted previously, corn ethanol and soy biodiesel will likely be the dominant biofuels out to 2025, the end date of consideration for this report (see Chapter 2). FOGs have no known effect on water quality aside from potentially beneficial effects from diverting FOGs from wastewater streams where they can clog infrastructure and contribute to overflows. Thus, as FOGs increase in the volume produced and consumed domestically the effects on water quality from all biofuels in total may decrease. The water quality effects from Brazilian sugarcane occur in Brazil (see Chapter 16) and were relatively small and temporary in the early years of the RFS Program and the growth of the industry. Therefore, the water quality effects in the near term will be predominantly from changes in the cultivation of corn and soybean.

Chapter 3 reported that cover crops are increasing in the United States although the adoption rates remain low [(Baranski et al., 2018); generally <5%]. Tillage practices, on the other hand, are improving in many areas especially for corn, with increasing no-till corn in the Northern Plains. Tillage practices were comparably stable for soybean (Baranski et al., 2018). As discussed in section 10.5, for these and other reasons the estimated potential water quality effects per megajoule (MJ) of biofuel appear to be decreasing for corn ethanol as practices improve, even though the estimated total potential effects have been increasing as the industry grows. For soybean biodiesel, the estimated potential water quality effects per MJ appear relatively constant, although the total potential effects are estimated fairly small due to the small size of the industry and because soybean receives much less fertilizer than other crops (see section 10.5).

Further along the supply chain, fuel releases from underground storage tank systems continue to occur regularly with 2019 data showing 5,375 releases reported (U.S. EPA, 2020a). That approximate release rate will likely continue for the next several years. Recent observations of corrosion in UST systems that may contain biofuel blends have become common, but no data exist to correlate those trends with releases (U.S. EPA, 2021). Gasoline containing 10% ethanol (E10) is ubiquitous in the United States. UST systems storing gasoline and ethanol blended fuels like E10 often show accelerated corrosion of some metal components, potentially leading to shortened service lives. UST systems storing diesel fuel, which may contain biodiesel up to 5%, also commonly experience corrosion (U.S. EPA, 2021). EPA is aware of numerous anecdotes of fuel releases caused by corrosion, although attributing cause is difficult for numerous reasons. These issues are discussed further in section 10.5.2.

Regardless, when the Set Rule was finalized on June 21, 2023, EPA published estimates of cropland expansion along with those biofuel volumes (see Chapter 6, section 6.5).[9] EPA estimated the biofuel volumes in the Final Set Rule could potentially lead to an increase of as much as 2.65 million acres of cropland by 2025, mostly from soybean expansion in the Midwest, but to a lesser extent from corn expansion in the Midwest and canola expansion in North Dakota. Still, the likely future effects of the RFS Program are highly uncertain. While the projected cropland expansion for 2023–2025 is slightly larger than the estimated historical cropland expansion from the RFS Program, it cannot be said with reasonable certainty that water quality will be impacted due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes. More information on where these changes are anticipated, and what agricultural practices are employed, are needed to better constrain estimates of the likely future effects to water quality. Additional information may be found in the associated docket for the Set Rule (EPA-HQ-OAR-2021-0427).

[9] On July 26, 2022, the United States District Court for the District of Columbia filed a consent decree, which after stipulations from the parties, required EPA to sign a notice of proposed rulemaking (NPRM) to establish 2023 volumes for the RFS Program on or before November 30, 2022, and to sign a notice of final rulemaking finalizing 2023 volumes by June 21, 2023. Growth Energy v. Regan et al., No. 1:22-cv-01191, Document Nos. 12, 14, 15. EPA's proposed RFS volumes for 2023-2025, 87 FR 80582 (proposed and signed on Nov. 30, 2022 and published in the Federal Register on Dec. 30, 2022) available in Docket No. EPA-HQ-OAR-2021-0427 at https://www.regulations.gov, were finalized on June 21, 2023, and published in the Federal Register on July 12, 2023 (88 FR 44468). The estimates of cropland expansion from the RFS Program took considerable time and effort to develop; thus, they are based on the proposed volumes from November 30, 2022, rather than the final volumes from June 21, 2023. Because of that, the acreages discussed here are merely illustrative. However, since the total volumes from crop-based feedstocks changed fairly little between the draft and final rules, they are still relevant here (see Chapter 6, section 6.5).

10.5 Comparison with Petroleum

10.5.1 Lifecycle Analyses with BEIOM

Lifecycle analyses focused on water quality are relatively rare in the literature, in contrast to air quality and water quantity (see sections 8.5 and 11.5, respectively).[10] A notable exception is the recent "whole economy" lifecycle analysis conducted by the DOE's National Renewable Energy Lab (NREL) to compare corn ethanol with petroleum, and soy biodiesel with diesel, across 15 different environmental and economic metrics (Avelino et al., 2021; Lamers et al., 2021). This approach uses lifecycle assessment (LCA) combined with an environmentally-extended input-output (EEIO) analysis to estimate the effects across the economy and fuel lifecycle.[11] This model is called the Bioeconomy Economic Input Output Model (BEIOM) [(Avelino et al., 2021); Appendix F]. The results presented here are for water-related releases to the environment and their impacts measured in eutrophication potential and freshwater ecotoxicity potential. Eutrophication potentials account for releases to air and water, with water being the dominant environmental medium. Freshwater ecotoxicity potentials account for releases to air, soil, and water, with water representing the dominant medium. Other end points are found in other chapters, and details of the analysis and assumptions are provided in Appendix F and in the peer-reviewed literature (Avelino et al., 2021; Lamers et al., 2021).

Results are presented in a single graph per biofuel and petroleum substitute, consisting of two panels each (Figure 10.18 for eutrophication potential and Figure 10.19 for freshwater ecotoxicity potential). The left panels (subpanels a and c) show the percentage contribution of the biofuel industries relative to the U.S. national total from all industries for the years evaluated. These results reflect total direct and indirect potential effects[12] due to the production of the respective fuel and their related co-products across the years and their impacts from fuel combustion. The right panels (subpanels b and d) show how the impacts from producing one energy unit (i.e., 1 MJ) of biofuel or fossil fuel changes over time. For ease of comparison, the year with the highest impact per metric is used as the benchmark (100%) and the per MJ impacts of the other years are then shown as a relative comparison to that benchmark. The impacts are broken down into supply chain steps (stacked bars), including upstream supply chain activities, corn/soybean farming, oil processing, ethanol/biodiesel conversion, fuel distribution, and fuel combustion.

[10] Current versions of GREET include lifecycle estimates of effects on water consumption but not on water quality.
[11] A comparable analysis on water quality using the more-established Greenhouse Gases, Regulated Emissions, and Energy Use in Technologies (GREET) Model by Argonne National Laboratory is not currently available.
[12] BEIOM estimates potential effects that combine lifecycle emissions with generalized effect responses from EPA's Tool for the Reduction and Assessment of Chemical and other environmental Impacts (TRACI). BEIOM is not spatial and does not account for actual exposure, fate, and transport to receiving areas. Thus, it represents potential effects from emissions and discharges to the environment.

The 2017 results are plotted in a shaded/non-solid pattern to stress their hybrid data (2012 economic and 2017 environmental accounts, see Appendix F).

At the industry level, both biofuels show an overall increase in water-related impact potentials relative to the total impacts of the U.S. economy (Figure 10.18a, c; Figure 10.19a, c). These increases were driven by the overall growth of both sectors, which increased at a faster pace than the rest of the U.S. economy across 2002–2017 (Avelino et al., 2021; Lamers et al., 2021). Moreover, the observed increases were mainly due to environmental releases attributable to feedstocks (corn and soybean farming). The peak in contribution observed for both metrics in 2012 is directly related to lower yields due to the drought in that year (18% reduction in corn yields as compared to 2007). Less biofuel was produced in that year but the same amount of inputs (e.g., fertilizer) were used, thus increasing the effects per MJ. While soybean yields also declined in 2012, the drop was smaller than it was for corn (4% reduction). Compared with their fossil fuel counterparts, the total effects from ethanol were larger than gasoline after the industry had increased (i.e., 2007 and later) for both eutrophication potential (Figure 10.18a) and freshwater ecotoxicity potential (Figure 10.19a). Because soybean receives much less N fertilizer and the industry is fairly small, the total effects from soybean biodiesel were smaller than diesel for eutrophication potential (Figure 10.18c). The total effects for soybean biodiesel were larger for freshwater ecotoxicity potential because of high rates of pesticide usage (Figure 10.19c). However, the freshwater ecotoxicity potentials in BEIOM may not accurately characterize the effects from oil spills, which in small spills have localized but significant effects and in larger spills can have regionally significant effects.

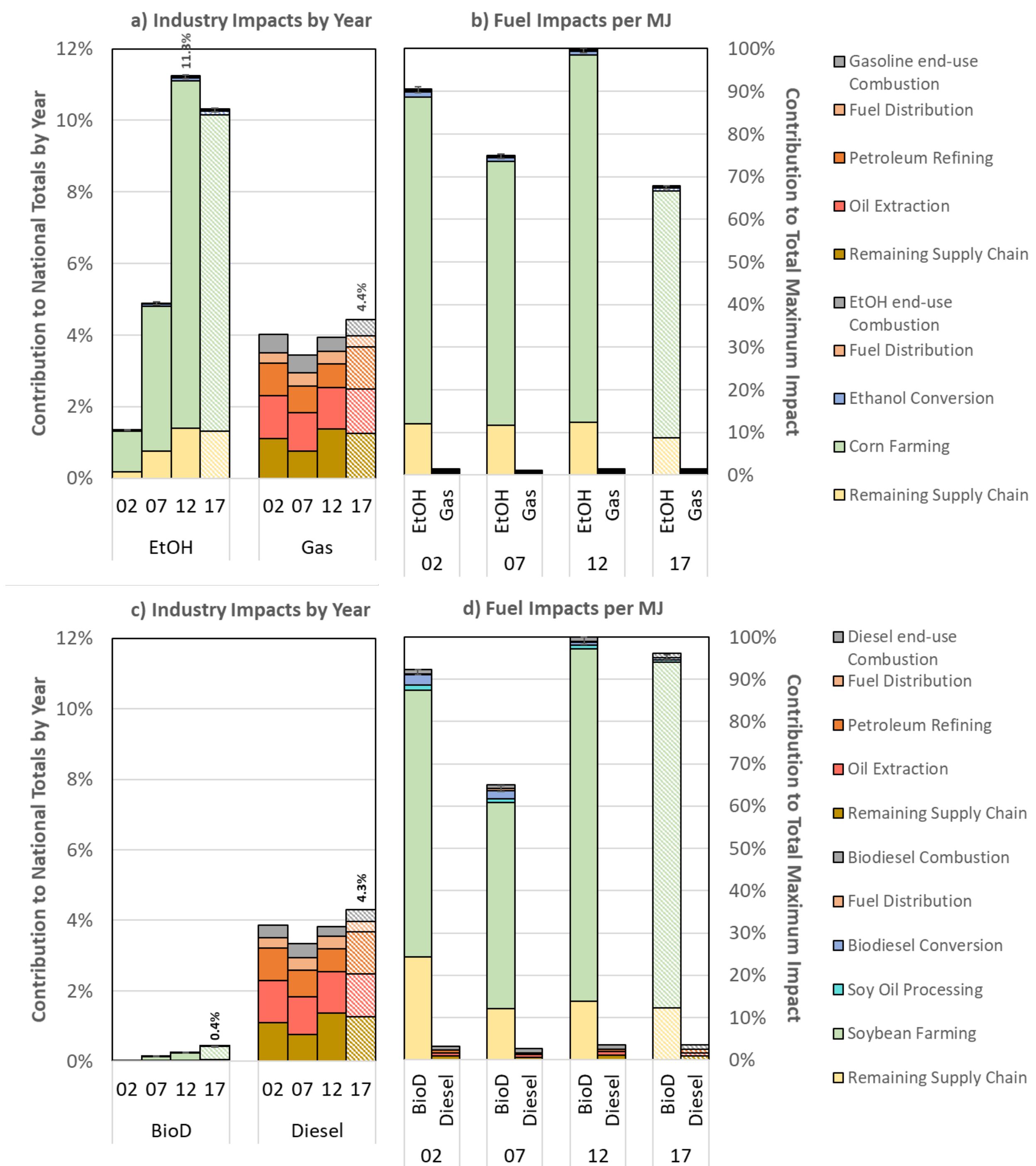
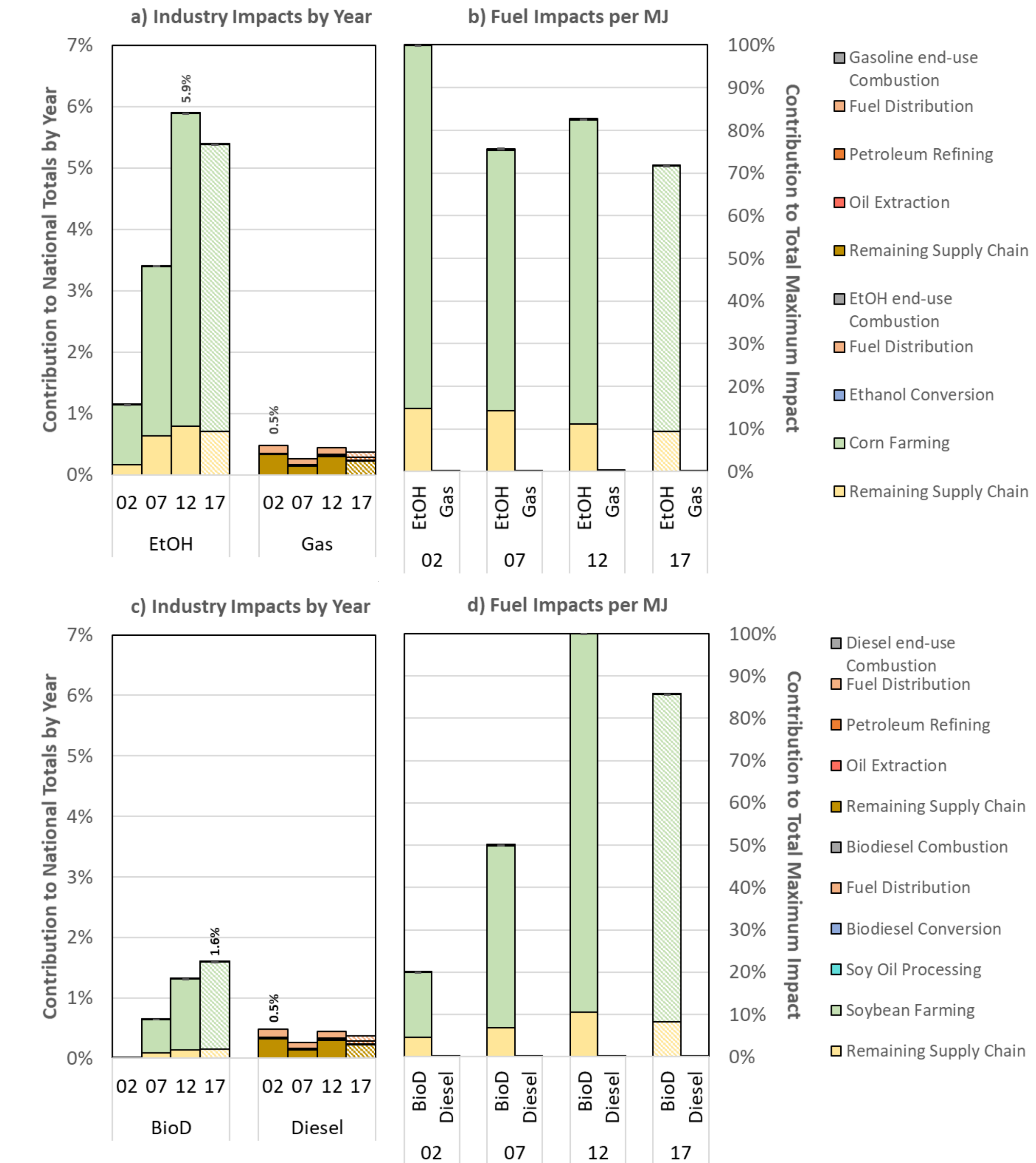


Figure 10.18. Eutrophication potential for corn ethanol (EtOH) vs. gasoline (Gas) (a, b) and soybean biodiesel (BioD) vs. diesel (Diesel) (c, d). Biofuel industry contributions to total U.S. national emission level per year (a, c) and impacts per energy unit of fuel (b, d). Source: Avelino et al. (2021) and Lamers et al. (2021). (Creative Commons CC-BY-NC-ND and Creative Commons license https://creativecommons.org/licenses/by-nc-nd/4.0/). Fuel Distribution is listed separately for ethanol and gasoline as both have a fuel distribution step.



MJ = megajoules

Figure 10.19. Freshwater ecotoxicity potential for corn ethanol (EtOH) vs. gasoline (Gas) (a, b) and soybean biodiesel (BioD) vs. diesel (Diesel) (c, d). Total industry contributions to total U.S. national emission level per year (a, c) and impacts per energy unit of fuel (b, d). Source: Avelino et al. (2021) and Lamers et al. (2021). (Creative Commons CC-BY-NC-ND and Creative Commons license https://creativecommons.org/licenses/by-nc-nd/4.0/). Fuel Distribution is listed separately for ethanol and gasoline as both have a fuel distribution step.

Trends through time on a per megajoule (MJ) basis show that corn ethanol improved over time (2002-2017) while the opposite is true for soybean biodiesel (Figure 10.18b, d and Figure 10.19b, d). Pesticide use for both fuels grew over the period, particularly the application of glyphosate herbicides (Osteen and Fernandez-Cornejo, 2016). For corn, the growing adoption of glyphosate in lieu of traditional herbicides such as atrazine, acetochlor, and s-metolachlor has *reduced* the freshwater aquatic ecotoxicity potential due to its lower characterization factor compared with other herbicides; a finding that is in line with those of Yang and Suh (2015). Contributing to the reduction in freshwater aquatic ecotoxicity potential for corn ethanol was a decline in corn acreage treated with insecticides (particularly chlorpyrifos and tefluthrin) from 24% in 2002 to 13% in 2018 (USDA, 2019). For soybean production, despite a widespread adoption of herbicide-resistant soybean species and the resulting substitution of traditional herbicides with glyphosate compounds, the freshwater aquatic ecotoxicity potential has increased over time due to an increasing use of insecticides (particularly lambda-cyhalothrin and cyfluthrin). The pest management choice may have been in response to the invasion of soybean aphid that appeared in Wisconsin in 2000 and rapidly spread throughout the Midwest (Yang and Suh, 2015).

The eutrophication potential for both biofuels is mainly driven by fertilizer applications in corn and soybean farming and the resulting N and P runoff and leaching. The trends follow the amounts of planted acres treated respectively. The eutrophication potential decline per energy unit of corn ethanol was due to increasing ethanol yields, while the almost V-shaped evolution for biodiesel was primarily due to an increase in the number of soybean fields treated with fertilizers. Soybean fertilizer applications were generally lower, 8 times lower for N and 1.3 times lower for P, than for corn farming (USDA, 2019). Also, fertilizer applications averaged 97% of planted acres for corn and only 79% for soybeans.

10.5.2 Underground Storage Tank Considerations

Fuel releases from infrastructure must be cleaned up. But the physical, chemical, and biological properties of biofuels differ from those of conventional petroleum fuels. This impacts their environmental behavior following a release from a UST or from aboveground infrastructure. If releases occur from UST systems storing biofuels blended with petroleum fuels, understanding the biodegradation behavior of biofuel blends can help to inform the site assessment, sampling, and remediation strategies for releases (U.S. EPA, 2015; ITRC, 2011).

While microbes in the subsurface will typically biodegrade petroleum releases, both aerobically (in the presence of oxygen) and anaerobically (without oxygen), biofuel blends have a different biodegradation profile. As a result, biofuel releases can have several complicating factors:

1. Biofuels containing higher percentages of ethanol have the potential to produce significant amounts of methane during aerobic biodegradation. The methane produced can reach

explosive levels if it travels through the subsurface and collects in confined spaces such as storm drains or basements. However, at most sites, the accumulation of methane near the surface is unlikely, as methane is highly biodegradable in normally well-oxygenated soil.

2. Generation of methane from biofuel releases can induce pressure gradients that may allow for the advective migration of methane and other gases toward potential receptors.
3. Microbes in the subsurface will rapidly aerobically biodegrade the ethanol in biofuels before other petroleum hydrocarbons and some or all the available oxygen in the subsurface may be consumed. This added oxygen demand can also reduce natural attenuation rates of petroleum hydrocarbons, which can potentially allow petroleum vapors to migrate further and increase the risk of petroleum vapor intrusion into nearby buildings and structures.

Fuel can contaminate water a variety of ways, including spills or overfilling of an underground or aboveground fuel tank. USTs may also release fuel to the environment due to corrosion of metals or incompatibility of the tank or other components of a storage tank system with the fuel being stored. Examples of observed incompatibility between fuels stored and UST materials include equipment or components such as piping or gaskets and seals on ancillary equipment that have become brittle, elongated, thinner, or swollen when compared with their as-installed conditions (U.S. EPA, 2020b).

Most older and even some newer existing UST systems (which includes but is not limited to the tanks, pumps, ancillary equipment, lines, gaskets, and sealants in the system) are not fully compatible with E15 and E85 and require modification before storing them. For example, the actual tank is often compatible with E15, but some of the connectors and pump components may not be (U.S. EPA, 2020b). This situation can lead to leaks. Dispensers are not part of the UST system, by definition, but face the same compatibility concerns and are a critical part of the fueling system (U.S. EPA, 2020b).

Since 1988, EPA's UST regulations require fuel to be stored in systems that are compatible with the type of fuel being stored. Limited use of ethanol started in some parts of the United States in the late 1970s, and in response decades ago some organizations, such as Underwriters Laboratories (UL), first designed or tested some UST system components—such as tanks and piping—for compatible use with E10. Today, most tanks and piping are now only available in 100% ethanol-compatible options. But most other UST equipment today remains available in multiple versions with different levels of compatibility with ethanol, often with the standard choices still compatible only up to E10. Increasing the amount of ethanol from 10% to 15% in fuel can make a significant difference in materials' compatibility with many UST system components over the life of the UST system. Most existing UST systems will not be able to meet the compatibility demonstration requirement in the UST regulation to store higher blends of ethanol or biodiesel without replacing some equipment (U.S. EPA, 2020b).

Ensuring UST systems are compatible with the substances they store is essential because USTs contain many components made of different materials. In certain percentages, petroleum-biofuel blends are more aggressive toward certain materials used in UST system construction than conventional fuel without biofuels (U.S. EPA, 2020b). The whole UST system—including the tank, piping, containment sumps, pumping equipment, release detection equipment, spill prevention equipment, and overfill prevention equipment—needs to be compatible with the fuel stored to prevent releases to the environment. Compatibility with the substance stored is required for all UST systems under EPA regulations, and storing certain biofuels requires additional actions of UST owners and operators.

Some higher blends of biofuels could also potentially affect the proper functionality of some types of UST release detection equipment, which means the facility owner or operator may not know that they have a leak (U.S. EPA, 2020b). Functionality is different from compatibility. Owners and operators should ensure that their release detection equipment is both compatible with the biofuel stored and meets EPA's release detection performance standards for use with the biofuel.

It is probable that most owners and operators of existing UST systems wishing to store higher blends of biofuels will find, after evaluating their systems and documentation, they are not able to demonstrate compatibility for their entire UST system as required by the 2015 UST regulation. These owners can upgrade their existing UST systems to be compatible, or they may choose not to store the substance (U.S. EPA, 2020b). Owners and operators storing only 10% ethanol blends or lower or 20% biodiesel blends or lower do not need to demonstrate compatibility of their UST system under the federal regulation (although they may have to do so by their state or local implementing agency) and most do not need to change equipment, but still must ensure compatibility and functionality of their system to prevent releases caused by various types of degradation possible when biofuel blended fuels are stored.

10.6 Horizon Scanning

Next generation biofuel feedstocks, such as cellulosic-based biofuels from either corn stover or dedicated energy crops, may increase in the future, potentially affecting water quality. Studies have shown that switchgrass, as a perennial native plant, offers several advantages, including drought and flood tolerance; high yield capacity with little to no fertilizer application; the ability to stabilize soils and sequester carbon with long root systems; and the potential to improve water quality (Dale et al., 2014; Tolbert et al., 2002; McLaughlin and Walsh, 1998). Wu et al. (2020) and Wu and Zhang (2015) developed future scenarios of biofuel feedstock production to assess potential water quality and quantity changes associated with an increase in converting land to switchgrass production in UMRB and MORB. These studies found that the water quality improved significantly with regard to N and P in the areas that grow switchgrass. In MORB, where nitrate runoff is a major concern, incorporating switchgrass into 2.5 million acres of land in the Kansas River watershed was estimated to significantly reduce the nutrient loss and sediment loss across all of the chemical compounds evaluated. The loss of N could be reduced by up to 220 million pounds (Wu and Zhang, 2015). These studies were based on projected impacts; future work with a focus on observable and attributable water quality impacts resulting from biofuels is needed to evaluate the accuracy of those projections.

Changing precipitation patterns associated with climate change may influence current water quality (Ballard et al., 2019). Loecke et al. (2017) statistically associated drought-to-flood transitions (termed "weather whiplash") to increases in riverine N loads and concentrations, and pointed out that these whiplash events are projected to increase in the future. Given that recent studies have connected cellulosic biofuel feedstock production to relatively lower N loadings in surface waters, there is potential to decrease the water quality impact of weather whiplash events under specific biofuel feedstock production scenarios.

EPA has not estimated what mix of fuels will make up the liquid fuel market after 2025 but anticipates modest increases in E15 (see Chapter 2, section 2.3.2) and biodiesel blends up to B20 in the next several years. Other biofuel blends may enter the market, but will likely have less distribution than the more established E15 and biodiesel blends less than 20% concentration. Regardless of the exact mix of fuels in the market after 2025, EPA can anticipate that historical trends in the UST program regarding corrosion, material incompatibility, releases, and cleanups will likely continue (U.S. EPA, 2020a). Challenges with corrosion will likely continue, but industry is developing new technologies and treatments to address these challenges. Current regulations about UST system compatibility are such that material incompatibility of fuel systems may be a limiting factor to widespread national use of fuels containing more than 10% ethanol or more than 20% biodiesel unless more UST infrastructure compatible with ethanol blends over 10% or biodiesel blends over 20% is installed. UST systems stay in the ground for decades and most older systems are not fully compatible with today's fuels (U.S. EPA, 2020b). Some releases are caused by other challenges other than corrosion or material incompatibility—such as overfilling a fuel tank during refilling—and it is likely those will continue to be a risk of release for all fuels, regardless of any infrastructure developments that could reduce the risk of infrastructure challenges associated with petroleum-biofuel blends.

10.7 Synthesis

10.7.1 Chapter Conclusions

- Water quality impacts to date from biofuel production, whether from the RFS Program or other factors, are almost exclusively due to corn and soybean production for corn ethanol and soy biodiesel. Conversion of grasslands to corn and soybeans for biofuels are expected to cause greater negative impacts to quality compared to growing these crops on existing cropland.

- A Missouri River Basin (MORB) SWAT model was applied to a 30-year period (1987–2016) to assess the general effects of cropland expansion from all causes on water quality over 2008–2016 from conversion of grassland to either continuous corn, corn-soy rotation, or corn-wheat rotation (Chen et al., 2021). Actual transitions are likely in between the ranges of these three scenarios. Chen et al. (2021) found that flow was relatively unaffected (0.1–0.4% across three scenarios), total suspended sediments increased (0.6–1.5%), organic nitrogen increased (3.6–8%), dissolved nitrogen increased (1–4.9%), total nitrogen increased (2.5–6.4%), organic phosphorus increased (5.2–11.3%), dissolved phosphorus increased (3.6–7.9%), and total phosphorus increased (3.9–8.7%). In general these expected changes are relatively small (<15%). There was much spatial variation in the response, with many watersheds showing little change, and a few watersheds in Iowa, Missouri, Nebraska and Kansas continuing to be "hotspots" due to high pre-existing cropland, precipitation, and grassland conversion rates. Only a portion of this (approximately 0–20%, see Chapter 6) is estimated to be attributable to the RFS Program.
- Lifecycle eutrophication impacts for both corn ethanol and soybean biodiesel are driven primarily by fertilizer application to corn and soybean crops and by the resulting nutrient runoff and leaching. Lifecycle analyses suggest that, on a per megajoule basis, the potential for eutrophication and freshwater ecotoxicity is higher for corn ethanol than gasoline and higher for soybean biodiesel than diesel. These estimates, however, are averages across industries and do not fully account for large-scale events from either the petroleum industry (e.g., spills) or from biofuels (e.g., lost harvests).
- Groundwater and drinking water nitrate concentrations may increase with increasing acreage of corn for biofuels. Switching from corn or other conventional biofuel crops to dedicated biofuel crops may lead to reductions in nitrogen to water bodies and thereby reduce future drinking water nitrate levels in both groundwater and surface water.
- Pesticides in drinking water could be impacted by increasing acreage of corn biofuels. Certain pesticides are more prevalent than others, such as atrazine, while other pesticides are no longer used are also no longer found in drinking water. Fewer pesticides may need to be applied to dedicated biofuel crops than corn and soybean crops.
- Continued implementation of conservation practices has been shown to reduce soil erosion, nitrate loss and phosphorus release. Integrating landscape design and conservation practices (reduced tillage, riparian buffer, saturated buffer, cover crops) in current corn/soybean land and cropland converted to perennial grass at field tests have shown a decrease in nutrient loss to surface water while maintaining corn/soy productivity. Conservation practices, such as reduced tillage and the

use of cover crops, can reduce the negative impacts of corn and soybean feedstock production and improve soil health.

- A decrease of nitrogen and/or phosphorus loadings is possible should perennial feedstocks that replace corn become dominant. Although not yet in use at the commercial scale, perennial grasses and woody species could improve soil quality, contingent on the type of land used to grow these crops.

10.7.2 Conclusions Compared with the RtC2

A number of studies have attempted to evaluate the changes of water quantity and quality in response to different future scenarios of land use change driven by biofuel development. However, the environmental impacts, particularly water quality impacts, that have already resulted from historical land use changes have not yet been studied. The MORB-SWAT assessment presented in the report is the first attempt to assess water quality impacts resulted from historical land use changes from all causes, and the fraction of those that may be attributable to the RFS Program.

10.7.3 Uncertainties and Limitations

- Not all biofuel feedstocks impact nitrate leaching equally. There are few comprehensive comparisons of the impacts of different biofuel feedstocks on nitrate in surface waters and drinking waters.
- The relative toxicities and modes of action of pesticides applied to different crops varies by crop type. Much is unknown about the impact of corn and soybean and related specific land management decisions specifically on drinking water contamination related to pesticides. Pest pressure is always changing, new pesticides will be used in the future and the current suite of pesticides may change. Genetic modification of corn may alter pesticide requirements and dynamics.
- As with other chapters (e.g., see Chapters 9 and 12), the largest source of uncertainty in the impacts to date from the RFS Program stems from the range of estimated additional cropland potentially due to the RFS Program, and a lack of understanding of the exact location of these converted lands attributable to the RFS.
- Conversion of lands/expansion of different biofuel crops may occur in the future but the temporal and spatial patterns are uncertain, including shifting production to new areas within and outside the United States.

10.7.4 *Research Recommendations*

- Further research could help elucidate the impacts of different biofuel feedstocks on leaching to groundwater and impacts on drinking water nitrate and pesticides specifically. Corn grown as a biofuel feedstock can lead to nitrate leaching, and in some cases violation of drinking water standards, as described in the chapter. However, since corn is grown for multiple purposes (e.g., animal feed, biofuels, other products), it is difficult to distinguish the impacts of biofuels from other end uses, and thus future work could focus on this differentiation.
- Further research is needed on pesticides whose toxicity and movement to nearby waters may harm aquatic organisms in U.S. streams and rivers.
- The availability, quality, and frequency of water monitoring data that watershed modeling relies on varies extensively. Although USDA ARS has established and implemented water monitoring programs in several watersheds in the Midwest, water quality monitoring data and research in many small agricultural dominant watersheds are lacking.
- Water quality responses to a change in cropland management at the watershed scale may take several years to observe and verify, due to the hydrological cycle and legacy impacts of nutrient inputs retained within soils and groundwater over time. Appropriate modeling and accounting for these legacies is important for understanding and managing water quality.
- Climate has an impact on the watershed hydrology and water quality. Current watershed modeling uses the past 10–20 years climate baseline. A shift of this baseline could affect water quality/nutrient results. Researchers should consider taking into account this shifting baseline in future studies.
- Further analysis is needed of longer-term monitoring data of changes in the landscape/fertilizer use examining the monitoring data over time. Researchers generally have to rely on modeling, while there are only scattered studies by USDA of, for example, the South Fork of the Iowa River, Raccoon River watershed scale and another watershed in Kansas with additional examples being added to USDA watershed studies over time.
- Currently the USGS SW mapper effort does not include a causal analysis component, and future efforts should link the changes in land use and other nutrient input-related factors to determine the drivers of change.
- Long-term surface water quality monitoring for corn/soybean farmland at the watershed scale would benefit from increased geospatial resolution.
- Studies are needed on integrated landscape design and conservation practices for feedstock production based on soil characteristics, productivity, water quality, farmer's income, and ecosystem services. In addition, the Inflation Reduction Act will invest billions on climate smart

conservation practices that have the potential to have positive impacts on soil health and water quality, thus there is a need for work in examining the effects of these investments.

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11. Water Use and Availability

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Key Findings

- Water use and water availability impacts of biofuels are primarily related to irrigation needs (the feedstock production stage), while water use in biorefineries (the conversion stage) represents a small and declining percentage of lifecycle water use.
- For corn-based ethanol, when accounting for ground and surface water ("blue water") used for irrigation, 88% of total lifecycle biofuel water use is for irrigation for feedstock production (on a gallon per megajoule [MJ] basis). For soybean-based biodiesel, feedstock irrigation is 98% of total lifecycle biofuel water use.
- The overall irrigated area of corn, according to USDA surveys, increased from between 9.3 and 9.7 million acres before the 2005 Energy Act to between 12 and 13 million acres reported in the 2008 and 2013 surveys, before declining to 11.6 million reported in the 2018 survey (representing 14% of total corn acres in 2018).
- The majority of total irrigation withdrawals (81%) and irrigated lands (74%) in 2015 occurred in the 17 conterminous western states located west of and including the Dakotas, Nebraska, Kansas, Oklahoma, and Texas overlying the High Plains Aquifer (HPA). Some satellite-based studies show irrigated croplands (all crops, all uses) over the HPA increased from approximately 14 million acres to 15 million acres (all crops/uses) between 2000 and 2017.
- Continued irrigation at present rates over the Southern HPA is not sustainable where the extraction rate exceeds recharge, most notably in eastern Colorado, western Kansas, the Texas Panhandle, and eastern New Mexico. However, for the Northern HPA, climate change is expected to increase precipitation, and the projections show that the irrigated area of the "MonDak" region (eastern Montana and western North Dakota) could expand, while irrigation at present rates is considered sustainable in much of eastern Nebraska.
- Water requirements for producing a gallon of corn ethanol (including total irrigation and refinery water) ranges from 8.7 to 160 gal/gal (i.e., gallons of water per gallon fuel) of ethanol (average 76 gal/gal), compared to petroleum-based gasoline, which ranges from 1.4 to 8.6 gal/gal of gasoline (average 5.7 gal/gal). The major factors determining the range are the regional variation in irrigation requirements for these corn-producing regions.
- Though a small fraction of the lifecycle water use, the water intensity of ethanol production in biorefineries decreased by 12% between 2011 and 2017 and by 54% between 1998 and 2017. These reductions have resulted from the adoption of energy-efficient and water-efficient technologies, water reuse and recycling, increased system integration in retrofitting existing plants, and diversification of water sources.

- Combining the GREET (Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation) model with WATER (Water Analysis Tool for Energy Resources) showed that, on a per megajoule basis, corn ethanol requires 0.084 –1.103 gallons (Corn Belt and Northern Plains states, respectively), with a U.S. weighted average of 0.377 gallons per megajoule. In comparison, gasoline averages 0.082 gallons per megajoule. Lifecycle water consumption for soybean biodiesel is slightly higher, from 0.102 to 1.697 gallons per megajoule, compared with 0.057 for diesel.

Chapter terms: blue water, dryland production, extraction and production (E&P), flood irrigation, furrow irrigation, green water, High Plains Aquifer (HPA, a.k.a., Ogallala Aquifer), irrigation water applied, rainfed, Soil and Water Assessment Tool (SWAT), sprinkler irrigation, water availability, water consumption or consumptive use, water footprint, water use, water withdrawal.

11.1 Overview

11.1.1 Background

The scope of this chapter encompasses the impacts of the production and use of biofuels and biofuel feedstocks on the use and availability of water in the United States. Water is used across the full biofuel supply chain. Biofuel feedstocks, such as corn and soybean, require either rainwater or irrigation water for their production. The irrigation of crops that are used for biofuels production is the predominant driver of biofuel-related water use. Estimates of irrigation water use for biofuel feedstock production are often orders of magnitude larger (on a gallon of water per gallon of fuel basis) than the water use in the biorefinery, where feedstocks are converted into biofuels (U.S. EPA, 2018). Most of the chapter does not distinguish between the effects of biofuels production in general compared to the RFS Program specifically. This chapter discusses broader irrigation trends for crops for all uses, not just biofuels. Water use attribution to the RFS Program is discussed, but attribution to the RFS more broadly is discussed in depth in Chapter 6 and in section 11.3.3.

In the United States, the U.S. Geological Survey (USGS) compiles and estimates national (states, District of Columbia, and territories) water use information in cooperation with state, federal (e.g., USDA), and local agencies to track water use trends through time, including water withdrawals and water consumption. Water withdrawal is the water removed from the ground or diverted from a surface water source for use, while water consumption represents the part of water withdrawn that is evaporated, transpired, incorporated into products or crops, consumed by humans or livestock, or otherwise not available for immediate use.[1] For 2015, irrigation withdrawals, which were all freshwater and account for all types of crop and non-crop uses, were 118 billion gallons per day (averaged over the full year) and

[1] https://www.usgs.gov/mission-areas/water-resources/science/water-use-terminology (USGS, 2019)

irrigated about 63.5 million acres. Irrigation accounted for 37% of total water withdrawals (when including all uses of freshwater and saline/brackish water, see Figure 11.1) and 42% of total freshwater withdrawals for all uses (Dieter et al., 2018b). Irrigation water use draws relatively similar amounts of water from surface water (60.9 billion gallons per day) and groundwater sources (57.2 billion gallons per day) (Dieter et al., 2018b). Consumption from irrigation was about 73.2 billion gallons per day and accounted for 62% of total irrigation withdrawals. In the United States, 55% of acreage uses sprinkler systems, 35% use furrow or flood methods, and 10% use drip or microirrigation systems (USDA, 2019).[2]

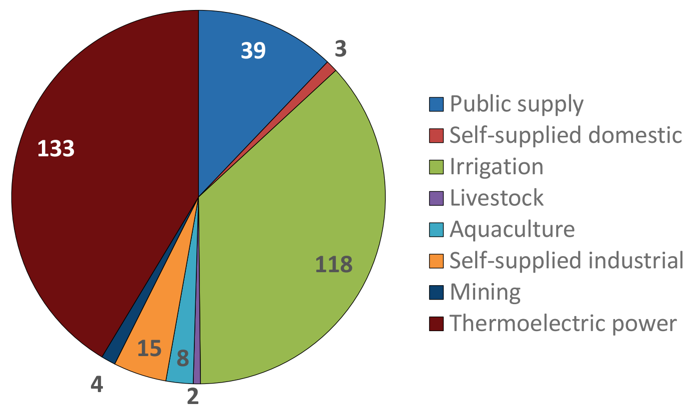


Figure 11.1. Total water withdrawals (billion gallons per day of freshwater and saline water) for all major uses based on Dieter et al. (2018b) data for 2015. Note that irrigation water withdrawals include nonagricultural uses, including golf courses, parks, nurseries, turf farms, cemeteries, and other self-supplied landscape-watering.

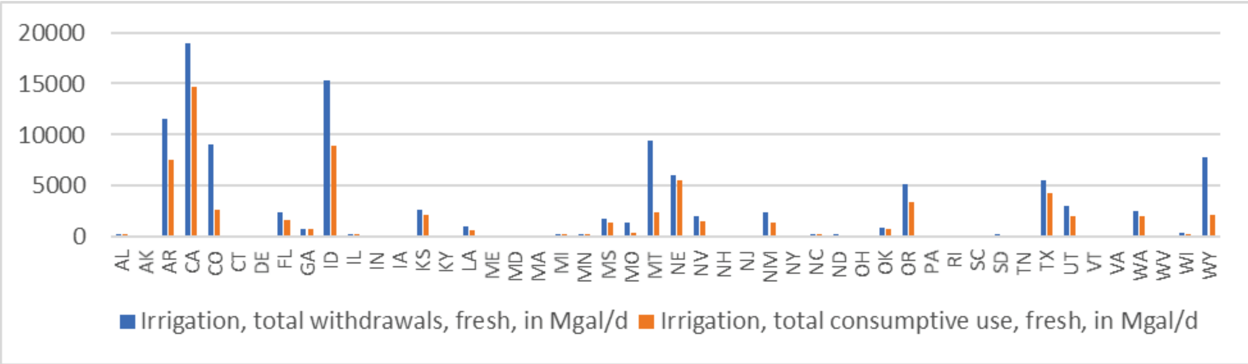
These water withdrawals and consumption have implications for water availability, whether that is for groundwater (including critical aquifers such as the High Plains Aquifer [HPA], also referred to as the Ogallala Aquifer)[3] or surface water. The majority of total irrigation withdrawals (81%) and irrigated lands (74%) in 2015 occurred in 17 conterminous western states located west of and including the Dakotas, Nebraska, Kansas, Oklahoma, and Texas (Dieter et al., 2018a) (see irrigation withdrawals and consumptive use by state in Figure 11.2). Of the 84.7 million acres of corn grown in the United States in 2017, just 14% or 11.6 million acres were irrigated, almost all in the western portion of the corn belt (USDA, 2019). Because 86% of corn acreage in the United States is rainfed or dryland,[4] corn yield and price are strongly dependent on both intra-annual and interannual variations in weather. Corn is an internationally traded commodity and so corn prices are also strongly affected by world market forces. The same is true for soybeans. Total soybean acreage is 90.1 million acres and 10.4% or 9.35 million

[2] Sprinkler/spray irrigation applies water to a controlled manner that is similar to rainfall. Water is distributed and applied through pumps, valves, pipes, and sprinklers. Furrow or flood methods are where farmers flow water through small trenches running through their crops. Drip or microirrigation systems run water through pipes (with holes in them) that are either buried or lying slightly above the ground next to the crops. Water slowly drips onto the crop roots and stems (USGS, 2021).

[3] The rest of the chapter primarily uses the term High Plains Aquifer (HPA), except when directly quoting reports or studies.

[4] Dryland farming is a system of producing crops in semiarid regions without the use of irrigation. Frequently, part of the land will lie fallow in alternate years to conserve moisture.

acres of soybeans were irrigated in 2017. When looking at irrigated crops and their relative share of the 55.1 million acres of irrigated lands, corn grown for grain accounted the most irrigated land in the United States at 22.4% but was followed closely by forage crops[5] (18.6%) and soybean (17.0%). Just nine crops account for 91% of U.S. irrigated lands. (Figure 11.3). Thus, corn and soybeans, the primary biofuel feedstock crops emphasized in this report, are planted to 40% of total irrigated acres in the United States. However, only a fraction of the total corn and soybean crops are for biofuel production (see Chapter 5 and 6 on land use change and attribution).



Mgal/d = million gallons per day

Figure 11.2. Total irrigation water withdrawals and consumptive use (freshwater only) based on Dieter et al. (2018b) data for 2015 for all 50 states. Note that irrigation water withdrawals and consumptive use include nonagricultural uses.

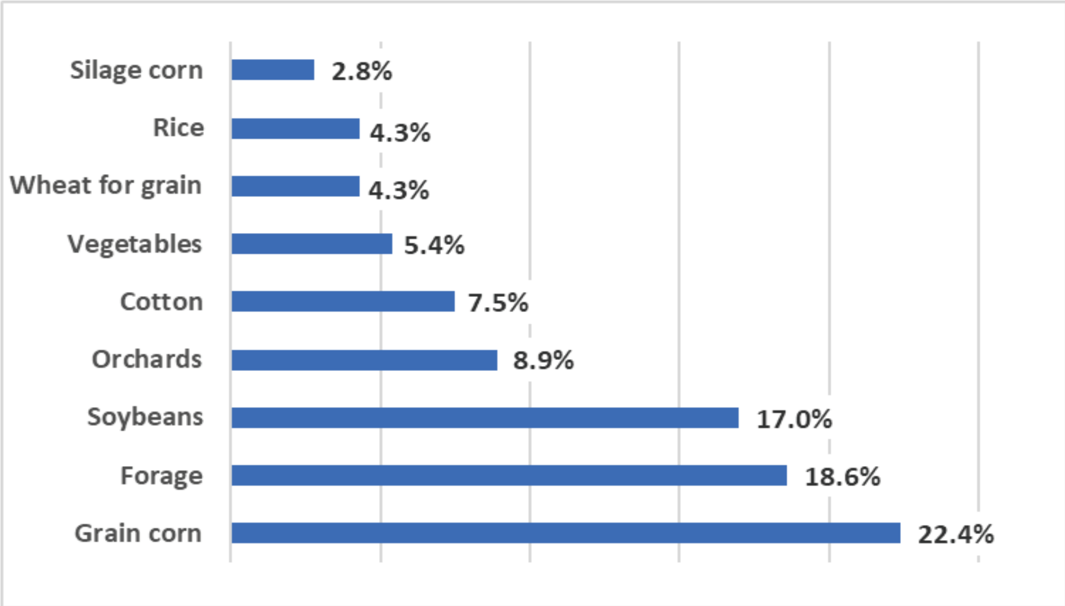


Figure 11.3 Percentages of the 55.1 million acres of U.S. irrigated land area planted with the top nine U.S. irrigated crops occupying or 91% of total irrigated lands. Corn leads in irrigated acreage but is followed closely by forage and soybeans. Data from USDA (2019).

[5] Forage includes crops such as hay and alfalfa.

Most of the focus of this chapter is on feedstock production and biofuel production. Other parts of the supply chains, such as transport of feedstocks or fuel, have no significant impacts on water use (RtC2) (U.S. EPA, 2018). Impacts from Brazilian ethanol derived from sugarcane are discussed in Chapter 16, but also have water use impacts. While Brazilian sugarcane is often in rainfed areas, full or supplementary irrigation may be required for sugarcane grown in the more semiarid regions of northeastern Brazil (da Silva et al., 2013). Water is also used in processing of sugarcane to ethanol (Gonçales Filho et al., 2018). Other feedstocks such as fats, oils, and greases (FOGs) have negligible direct water use. A recent study by Caldeira et al. (2018) compared the water footprint (WF) profile for four biodiesel feedstocks, including waste cooking oil. Their approach considered FOGs, such as cooking oils, to be wastes from the production of the primary commodity, and therefore any water use that did occur in the upstream part of the lifecycle would be allocated to the original product. As described in a critical review on FOGs by Abomohra et al. (2020), studies have concluded that waste cooking oil for biodiesel production showed the lowest impact on WF as there is no water consumption for generating FOGs.

11.1.2 Drivers of Change

The drivers of changes in feedstock water use are closely linked to changes in land use and land management. Corn and soybean production varies from primarily rainfed production in Iowa, Minnesota, Illinois and elsewhere, to irrigated production in the western parts of Nebraska and Kansas. Therefore, land use changes (as discussed in Chapter 5) associated with biofuels can affect both the water footprint of those biofuels and water availability in the regions where they are produced.

Irrigation water use is arguably the most critical factor in understanding the potential water availability impacts associated with biofuels production and use, but also must be understood in the context of broader agricultural production. Corn produced for biofuel production is identical to corn produced for other uses—what determines the effects on water availability is not what is grown, but where the crop is grown and under what management conditions. Irrigation water use is critical in supporting agricultural production for food, fuel and feed, enhancing yields, reducing risk, and buffering against changes in precipitation. As will be discussed, there are multiple interacting drivers that affect irrigated area and irrigation intensity, including climatic conditions, changes in irrigation efficiency and crop water productivity, precipitation-related events such as drought or flooding, water rights conflicts, and prices and demands for different crops that may drive land use change and management.

Biofuel refineries also require water for converting corn grain to ethanol (in wet- or dry-mill ethanol plants[6]) or soybean and other oil crops to biodiesel. Biorefineries vary in their size, feedstock,

[6] Most milling operations (i.e., ~90%) are currently dry mill, see Chapter 3 for details. In these cases, the mash is dewatered and the water is returned to the receiving water body or reused. Water is also used for steam generation and cooling towers and is evaporated to the atmosphere.

technology, and use of water conservation measures, and this will affect the total amount of water used by these facilities. The sources of water used for the biorefining process will also vary, and this chapter will highlight some of those differences: well water, city water, reclaimed water, or other sources. In the previous Report to Congress (U.S. EPA, 2018), there was little data on biorefinery water use. New survey data from Argonne National Laboratory provides more detailed insights on facilities and their water uses and sources (Wu (2019), discussed in section 11.3.1.5). Other steps in the biofuel supply chain, including the collection and transport of feedstocks and fuel transportation, distribution, blending, and end use, have minimal water needs, if any.

11.1.3 Relationship with Other Chapters

As stated above, the scope of this chapter encompasses the impacts of the production and use of biofuels and biofuel feedstocks on the use and availability of water in the United States, and more specifically, on the use and availability of freshwater resources, including both surface and groundwater. This chapter will also review the current state of knowledge on the potential impact this water use may have on water resource availability. The statutory language from Section 204 identified multiple end points related to water resources (see Table 2.3 in Chapter 2). Some of these end points are addressed in Chapter 10: Water Quality. However, under the language of Section 204, *environmental issues* that were identified included the "acreage and function of waters," while *resource conservation* issues included "water availability." To assess these issues, this chapter will assess both water use and availability. "Function of waters" is covered more directly in the chapter on water quality (Chapter 10), aquatic ecosystems (Chapter 13), and wetlands (Chapter 14) (also see Chapter 2, Table 2.3).

It is important to note that these effects involve attribution of those impacts to the end-use fuel or to the RFS Program. In simple terms, one can ask how much of the irrigation water used for a bushel of corn is attributed to the gallon of biofuel that fuels a vehicle. However, the information needed to quantify that attribution is complex and depends on factors ranging from the location of feedstock sources and the associated irrigation practices, to the allocation of water use to a range of biofuel co-products. Chapters 6 and 7 address attribution in depth. This chapter highlights any additional attribution considerations that are unique to water intensity, use, or availability.

11.1.4 Roadmap for the Chapter

The rest of the chapter reviews the impacts to date and the likely future impacts from biofuels, beginning with the conclusions from the RtC2 (U.S. EPA, 2018). It also briefly compares the water use and availability impacts of biofuels relative to petroleum. A brief section on scientific understanding and next steps for research is discussed, including other drivers of changes, such as climatic changes and variation in precipitation patterns, that may affect the longer-term impacts of biofuels on water use.

11.2 Conclusions from the 2018 Report to Congress

The overall conclusions from the 2018 Report to Congress on Biofuels and the Environment were as follows:

- As discussed in the 2011 Report, the irrigation of corn and soybeans grown for biofuels is the predominant water quantity impact. Water use for feedstock production is significantly greater than water use in the biofuel conversion process.
- There are indications of increased water consumption in irrigated areas for corn between 2007 and 2012 and elevated rates of land use change to corn production in more arid Western states including the Ogallala region. Adverse water availability impacts will most likely arise in already stressed aquifers and surface watersheds.
- Irrigation practices are dependent on a number of economic and agronomic factors that drive land management practices making attribution of increased irrigation and water quantity to biofuels difficult.

11.3 Impacts to Date for the Primary Biofuels

11.3.1 Literature Review

Estimates of lifecycle water use of biofuels are driven by the irrigation water use, which varies due to climate (wet years, dry years, drought periods) and across regions. Of note in the RtC2 was that many of the assessments were working toward a more refined analysis of regional variability in water demands under different production scenarios. In the previous Report to Congress (RtC2), a large share of the literature reviewed focused on assessment of the lifecycle water use or water footprint of biofuels (Wu et al., 2014; Dominguez-Faus et al., 2013; Chiu et al., 2009). Recent estimates, such as Wu et al. (2018) and Wu (2019), show biorefinery water use as 2.65 gallons of water per gallon of denatured ethanol, while total consumptive water use ranges from 8.7 gal/gal (USDA Region 5) to 160 gal/gal (USDA Region 7) due to regional variation in irrigation.[7] Using a weighted regional average of 76 gal/gal for total consumptive water use, suggests that 3% is biorefinery water use, while 97% of total lifecycle consumptive water use is for irrigation (Wu et al., 2018).[8,9] More recent analyses using 15-year averages

[7] The regions in this study are based on the USDA Farm Production Regions. Region 5 includes Iowa, Indiana, Illinois, Ohio, and Missouri. Region 7 includes North Dakota, South Dakota, Nebraska, and Kansas.

[8] These figures are based on a mass-based co-product allocation, meaning some of the water use is allocated to other products derived from the corn ethanol production process, such as dried distillers' grains soluble (DDGS). Without allocation of water use to co-products such as DDGS and CO_2, lifecycle water use is approximately 50% higher.

[9] Consumptive water use as defined in Wu et al. (2018) is the sum total of water input less water output that is recycled and reused for the process. The estimate applies to surface and groundwater sources for irrigation but does not include precipitation.

and lifecycle water use on a gallon per megajoule basis suggest that irrigation may be approximately 88% of total lifecycle water use (see section 11.6 and Figure 11.22 and Figure 11.23 for lifecycle comparisons with petroleum and updated estimates from the GREET and WATER models).

Hoekman et al. (2018) in a two-part review of the environmental implications of ethanol production, also reviewed literature on the water footprint and lifecycle water requirements for ethanol. Biodiesel and other biofuels were not discussed in the review. Several of the themes emerging from that review are similar to the conclusions of the RtC2 (U.S. EPA, 2018). The literature reviewed for the RtC2 placed water usage for ethanol production plants (2–3 gallons of water per gallon of ethanol) as "modest" compared to the water use for irrigation of feedstocks, which can be 8–10 times higher (U.S. EPA, 2018). In addition, the review found general consistency in the results that lifecycle water impacts or water footprint are much higher for ethanol than for gasoline. Finally, the authors noted that more location-specific analysis is required to understand the water footprint, and expansion of corn into regions requiring irrigation can exacerbate water shortages in some areas (Hoekman et al., 2018).

This review includes recent literature on trends in water use and water availability impacts of biofuels. This includes published literature in journals, but also draws heavily from reports and data that were not available for the RtC2, including updated data from the USDA survey on irrigation and water management (USDA, 2018), as well as an extensive survey undertaken by Argonne National Laboratory of all biofuel facilities currently operating in the United States (Wu, 2019). This report also delves into more recent literature drawing on satellite-based data.

Most of the discussion focuses on irrigation, given that approximately 90–98% of total lifecycle water use is attributed to the feedstock production stage for corn- and soybean-based fuels. Three primary methods or sources of data are described in the sections on irrigation water use. The survey-based data for irrigation water use are reviewed in section 11.3.1.1, focusing on the USDA survey data on crop irrigation and water management. These data, collected on a five-year cycle, are critical inputs to other analyses such as lifecycle estimates of water use. Emerging areas of research utilizing satellite-based data to provide greater spatial and temporal resolution to irrigation trends over time are reviewed in section 11.3.1.2. Model-based studies that attempt to estimate both historic and future changes in water use and hydrologic impacts of crop production scenarios are briefly covered in section 11.3.1.3. Finally, current and historical water stress and its relationship to changes in irrigation for all crops are discussed in section 11.3.1.4.

11.3.1.1 Changes in Water Use for Feedstock Production: USDA Survey-Based Data

Water use data (including withdrawals, deliveries, returns and consumptive use as well as ancillary data such as irrigated acres by system type), including trends, for the nation have been reported by USGS at various scales (statewide, county, HUC, aquifer) for all major categories of use (see Figure 11.1) on 5-year intervals since 1950 through 2015.[10] The 2015 compilation (Dieter et al., 2018a) by USGS represents a continuous 65-year timespan of water use accounting for major categories,[11] which have changed over time. Irrigation-specific data are regularly collected by USDA on a five-year basis through the Census of Agriculture and Farm and Ranch Irrigation Survey. Crop-specific irrigated acres (not irrigation withdrawals) were compiled for the USDA-NASS Census of Agriculture reports (USDA-NASS, 2019, 2014, 2007, 2002; U.S. Census Bureau, 1997; USDA, 1994) and state-level irrigation water use quantities are reported from the Irrigation and Water Management Survey (formerly called the Farm and Ranch Irrigation Survey) reports (USDA, 2019, 2014, 2010, 2004, 1998, 1994).[12] Thus, while individual datasets do not provide all the necessary information (e.g., USGS irrigation withdrawals and consumptive use data do not distinguish between crop types, USDA Census and Irrigation Surveys do not capture withdrawals), the synthesis of these datasets inform this report. This section also reviews the trends, with a focus on corn production, that can be determined from the USDA-NASS survey data. It is important to note that the years of compilation between USGS and USDA do not coincide. USGS compilations consult USDA datasets and adjust as necessary to account for the different agricultural activities between years.

The RtC2 covered USDA irrigation survey data up to 2012; this report examines more recent studies and analysis up to 2018 to provide a more complete picture of changes in irrigation as well as the drivers of those changes. As discussed in Chapter 6, although the bulk of growth in the industry was from 2002 to 2012, and thus earlier data is sufficient to assess general effects of industry growth, most of the quantifiable effects from the RFS Program, if any, were after 2013, in which case the more recent information is informative. The effect of increased ethanol production on the quantity of water used to irrigate grain corn in the United States is difficult to discern due to multiple factors that affect irrigated area, where and under what climatic conditions corn is irrigated, changes in irrigation efficiency and crop water productivity, drought, flood, climate change, price and the global factors influencing demand. In addition, neither the USDA-NASS nor the USGS data distinguish between end uses of crops, so there are no data in those reports to substantiate how much irrigated corn or soybean were used for biofuel production.

Producers choose to grow more or less irrigated corn and soybean due largely to price, constrained by water availability, production costs (including land prices), risk and prices of other crops.

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[10] https://www.usgs.gov/mission-areas/water-resources/science/water-use-united-states?qt-science_center_objects=0#qt-science_center_objects (USGS, 2015) (Dieter et al., 2018a)
[11] The most recent report available is 2015: https://www.usgs.gov/mission-areas/water-resources/science/changes-water-use-categories?qt-science_center_objects=0#qt-science_center_objects (USGS, 2018)
[12] In 2018, the Farm and Ranch Irrigation Survey was renamed to the Irrigation and Water Management Survey.

Overall, irrigated area in corn increased from between 9.3 and 9.7 million acres before the 2005 Energy Act to between 12 and 13 million acres reported in the 2007 and 2012 censuses, before declining to 11.6 million reported in 2018,[13] about a 21% increase from the period prior to 2005, or roughly 1.6% per year (Figure 11.4a). This percentage increase in irrigated corn (21%) was substantially larger than the percentage increase in all corn acres (12.5%). However, irrigated area for corn production in the nation was increasing steadily at 0.5% per year prior to 2005, so the increase in irrigation to meet the higher corn ethanol production volumes since then may be limited to a maximum of roughly 1.1% per year (see Chapter 6 for more attribution information). In contrast, all acreage devoted to corn production also rose and fell after 2005 in much the same way but with larger dynamic changes for irrigated acreage (Figure 11.4b). That said, the absolute change in irrigated corn acreage was relatively small compared with the absolute change in unirrigated corn acreage, largely occurring between 2002 and 2007 survey years (Figure 11.4b). The change in irrigation is coincident with the major market forces discussed in Chapter 6.

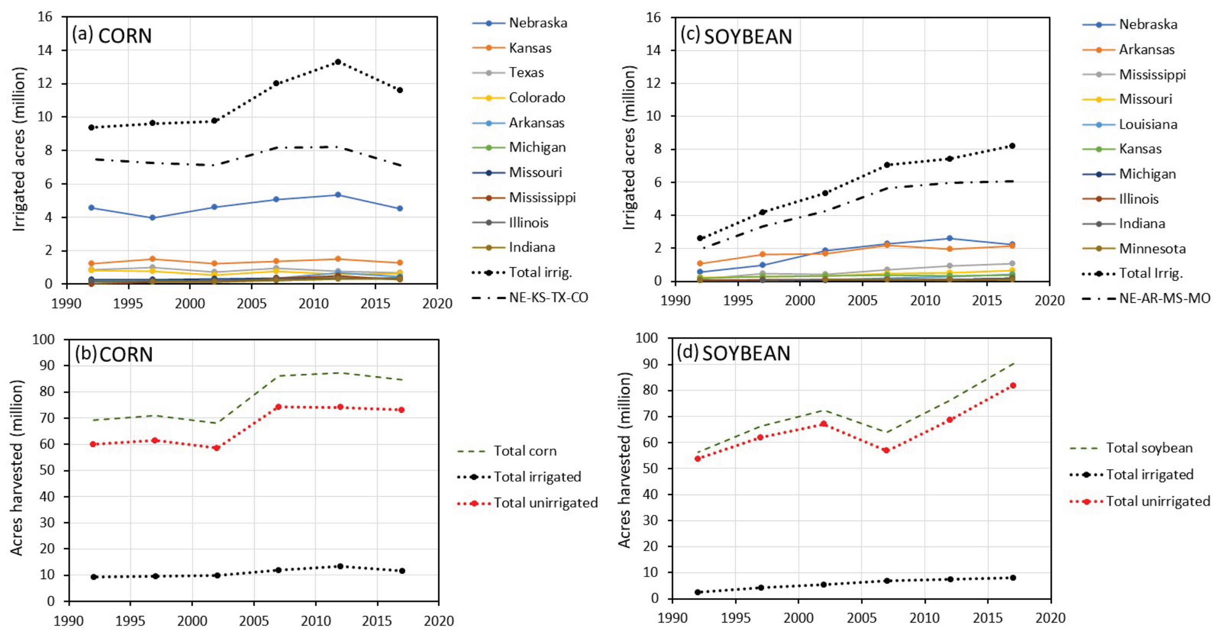


Figure 11.4. Acreage devoted to irrigated grain corn production (a) and irrigated dry soybean (c) production in the United States in the 10 states historically hosting the greatest irrigated acreage for each crop from 1992 through 2017 (5-year increments based on data from the USDA NASS Census of Agriculture, i.e., 1992, 1997, 2002, 2007, 2012, 2017). Comparison of irrigated corn acreage to unirrigated corn acreage and total acreage in grain corn (b), and comparison of irrigated to unirrigated soybean acreage and total acreage in dry soybean (d). Note the change in states included in the legends for (a) and (c).

[13] Data in Figures 11.3 through 11.9 are from USDA NASS Census of Agriculture reports for 1992, 1997, 2002, 2007, 2012, and 2017, and from Farm and Ranch Irrigation Survey reports for 1994, 1998, 2002, 2008, and 2013, and from the 2018 Irrigation and Water Management Survey.

Irrigated soybean acreage steadily increased at a rate of approximately 300,000 acres per year from 2.6 million acres in 1992 to 7.0 million acres in 2007 before the rate of increase slowed between 2007 and 2017 when 8.2 million acres of irrigated dry soybean were harvested (Figure 11.4c). Total soybean acreage (irrigated and unirrigated) actually declined by 8.5 million acres between 2002 and 2007 while irrigated acreage increased by 1.7 million acres over the same period. Unlike corn, irrigated soybean acreage was less synchronized with the dynamics of overall soybean production (Figure 11.4d) and variations in irrigated soybean acreage were much smaller than variations in total soybean acreage.

Figure 11.5 and Figure 11.6 show the geographic distribution of irrigated and total corn acreage for 2017, as well as a comparison of irrigated acreage change between survey years 2007 and 2017. Despite the total increase in both irrigated and unirrigated corn acreage over the past decades (Figure 11.4b), the share of irrigated acreage has steadily declined in Nebraska, Kansas, Texas and Colorado (NE-KS-TX-CO region) (Figure 11.7). These four states alone planted 80% of irrigated corn acreage in 1994, but in 2018 planted only 60% of total irrigated corn acreage. That relative decline is related more to the increase in irrigated corn acreage in more eastern states (Arkansas, Michigan, Missouri, Mississippi, Illinois, and Indiana) than it is to any decline in irrigated corn acreage in the NE-KS-TX-CO region. Within the NE-KS-TX-CO region, total irrigated acres have overall remained fairly steady as a percentage of that region's total acreage (12 to 14%) over the past quarter century. The total irrigated acreage in this region has risen and fallen more or less in concert with total irrigated acreage in the nation but with less variation over the years, and overall it has decreased slightly from 7.5 million acres in 1994 to 7.1 million acres in 2018, mostly due to decreases in irrigated corn acreage in Colorado and Texas (Figure 11.4a).

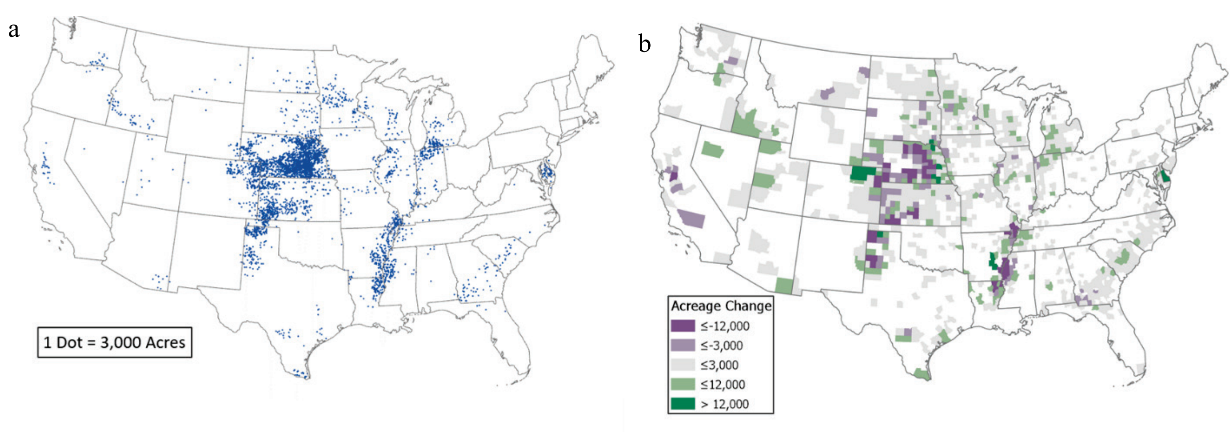


Figure 11.5. Irrigated corn for grain in 2017, harvested acres (1 dot = 3,000 acres) (a), and Irrigated corn acreage change from 2007 to 2017, by county (b). Source: USDA NASS – Census of Agriculture.

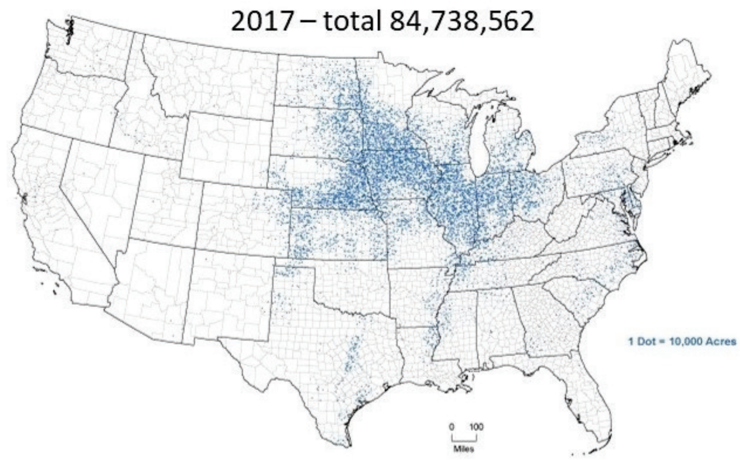


Figure 11.6. All corn for grain in 2017, harvested acres (1 dot = 10,000 acres). For comparison, this was about 2% lower than total acreage (86.2 million) in 2007 Source: USDA NASS– Census of Agriculture.

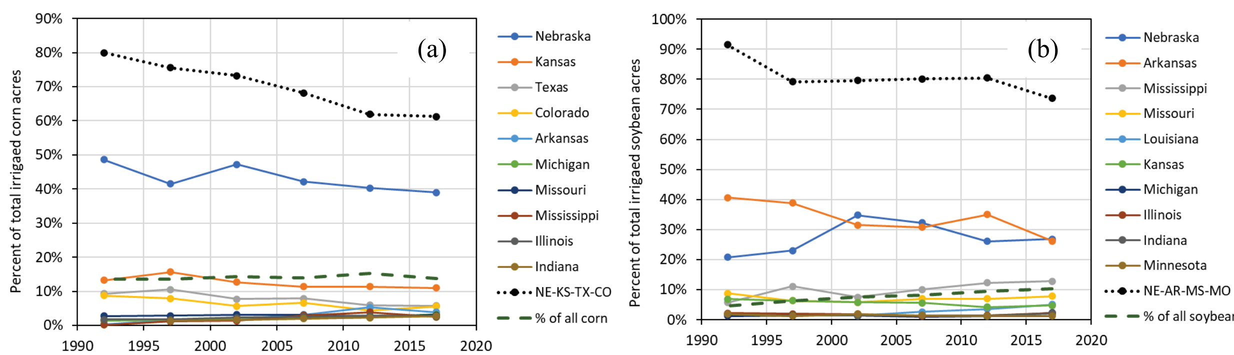


Figure 11.7. Percent of total irrigated corn (a) and soybean (b) acreage for the ten states with the most irrigated acreage historically for the period from 1992 to 2017 (5-year increments based on data from the USDA NASS Census of Agriculture, i.e., 1992, 1997, 2002, 2007, 2012, 2017). For corn, the share of region including Nebraska, Kansas, Texas, and Colorado (NE-KS-TX-CO) is also shown; for soybean, Nebraska, Arkansas, Mississippi, and Missouri (NE-AR-MS-MO) shares are shown. Data from USDA NASS (USDA, 2020, 2014, 2010, 2004, 1998, 1994).

While changes in irrigation acreage is one aspect of total water use, another factor is water applied per acre. The efficiency of irrigation systems (percentage of water applied to a field that is available to be used in evapotranspiration by the crop), and crop water productivity (yield per unit of water used in evapotranspiration) both affect the water applied per acre. Application rates also vary regionally. Overall, total irrigation water applied in a growing season has declined since 1992, more so in the subhumid to semiarid regions of Nebraska, Colorado, Kansas and Texas where more than 60% of irrigated corn is grown (USDA, 2020, 2014, 2010, 2004, 1998, 1994) (Figure 11.8a). Irrigation depth applied is greatest in Texas and Colorado, which feature the driest climates and greatest evaporative demand, followed by Kansas (Figure 11.8a). States like Nebraska and Arkansas have semiarid (western NE) to subhumid and humid climates and reduced evaporative demand, and irrigation depth applied varies with drought and precipitation levels. Drought periods between 2010 to 2014 caused some of the largest increases in depth of water applied.

Other than weather, change in depth of water applied was mostly tied to changes in irrigation application methods from gravity flow, which served approximately 58% of irrigated acres in 1994 in the four states where most corn is irrigated (NE-KS-TX-CO), to pressurized systems, which now serve more than 90% of irrigated acres in those states (Figure 11.8b). The change from gravity flow to pressurized systems has increased irrigation efficiency, reducing the depth of water applied, and increased crop water productivity due to more uniform irrigation applications (Evett et al., 2020a).

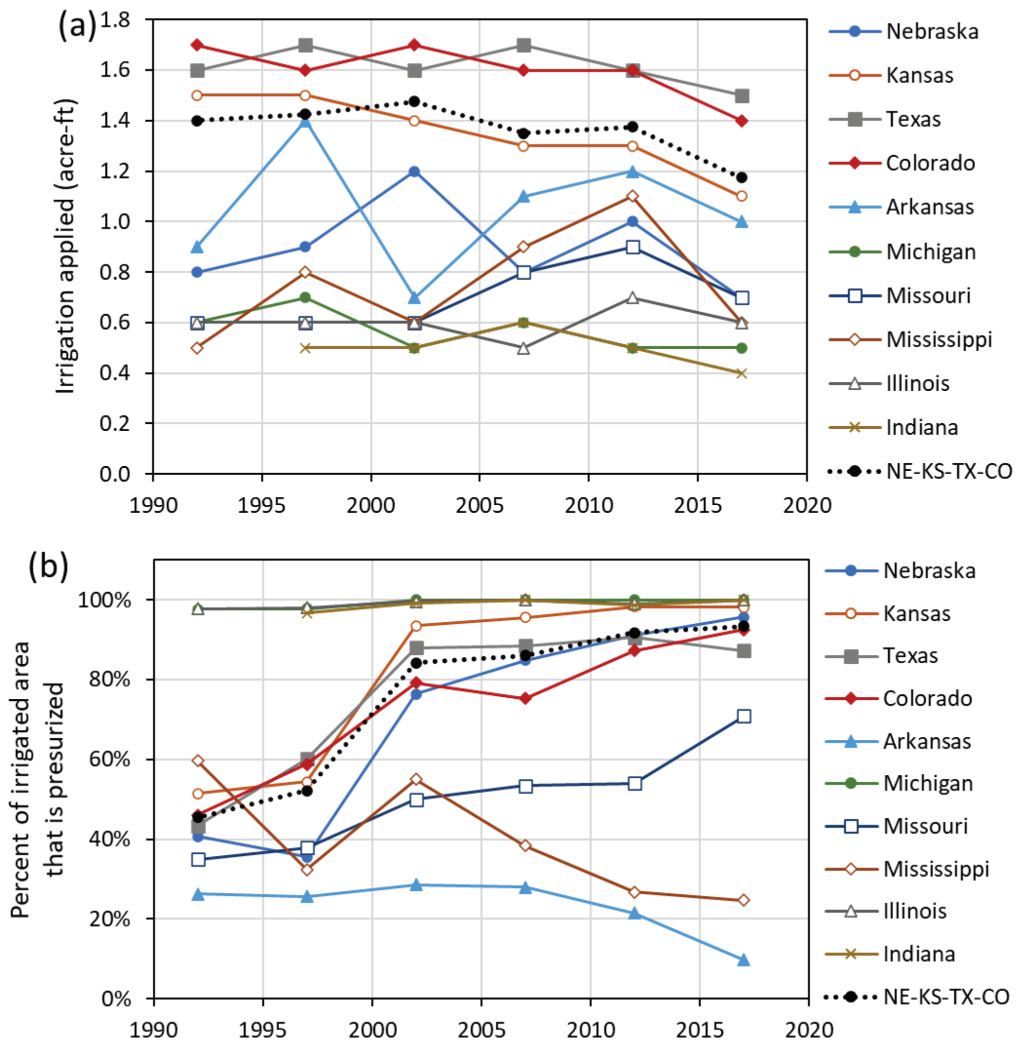


Figure 11.8. Water applied (acre-feet [ft]) per acre of irrigated corn from 1992 to 2017 for the 10 states where irrigated corn acreage is historically greatest (5-year increments based on data from the USDA NASS Census of Agriculture, i.e., 1992, 1997, 2002, 2007, 2012, 2017) (a). Also shown is the average of water applied in the four states with the greatest irrigated corn acreage, Nebraska, Kansas, Texas and Colorado. **Percent of irrigated area that is pressurized (mainly center pivot and subsurface drip irrigation systems) for the same states over the same period (b).** Pressurized irrigation serves 93.4% of irrigated area in Nebraska, Kansas, Texas and Colorado. Data from USDA NASS (USDA, 2020, 2014, 2010, 2004, 1998, 1994). For reference, 1 acre-ft = 325,851 gallons.

Similar trends were seen at the national level for all cropland acres in the most recent Conservation Effects Assessment Project (CEAP) report, released in March 2022 (USDA NRCS, 2022). Based on natural resource data and farmer surveys, CEAP I surveys were conducted in 2003–06, and CEAP II surveys were conducted in 2013–16. Between CEAP I and II, irrigated cropland increased by 36% in the North Central and Midwest (primarily in eastern Nebraska) and 8% in the Southern and Central Plains (which covers much of the irrigated lands of Texas, Oklahoma, Colorado, western Kansas, and western Nebraska). Comparison of these surveys and data also allows estimation of conservation adoption and effects between the CEAP survey periods, including irrigation and water management. It was found that irrigators (nationally, all crops) were using more efficient pressure-based systems by CEAP II, and improved water management strategies had decreased per-acre water application rates by 3.6% from 19.2 inches per acre (1.6 feet per acre) in CEAP I to 15.6 inches per acre (1.3 feet per acre) in CEAP II. The decrease was slightly less pronounced in the Southern and Central Plains (which covers much of the irrigated acreage of Texas, Oklahoma, Kansas, Colorado, and Nebraska) where there were decreases of 1.7% (from 1.3 to 1.2 feet per acre). This is consistent with trends seen for irrigated corn acreage in Figure 11.8a-b.

Yields have changed as well, with irrigated corn yields steadily increasing since 1992 (Figure 11.9a-b). Some of these improvements can be attributed to improvements in genetics, planting methods, fertilization, and other agronomic practices (as discussed in Chapter 3, section 3.2.1). For irrigated corn, yield improvements were also a result of greater uniformity of irrigation with the replacement of gravity flow systems by center pivot systems. Unirrigated corn yields are typically lower and more variable due to their greater dependency on weather (Figure 11.9b). For example, the increase in total corn production reported in 2007 was due to both the increase in harvested acres (Figure 11.4b) and the increase in yield per acre (Figure 11.9a-b) while weather was relatively good. In contrast, the large decrease in total yield and yield per acre reported in 2012 was almost entirely related to unirrigated corn, which was greatly impacted by continuing drought. The yield gap (Figure 11.9b) between irrigated and unirrigated crops is twice as large for corn as it is for any other major crop, meaning that the risk of not irrigating is proportionally larger for corn than for other crops such as sorghum, soybean or wheat (Kukal and Irmak, 2019). In other words, irrigation is an effective drought-mitigation strategy for reducing the impacts of climate change on crop yields and production (Kukal and Irmak, 2018), particularly for corn. Climate change is bringing greater variability in growing-season precipitation to the Midwest, which is one reason why irrigation is increasingly employed there.

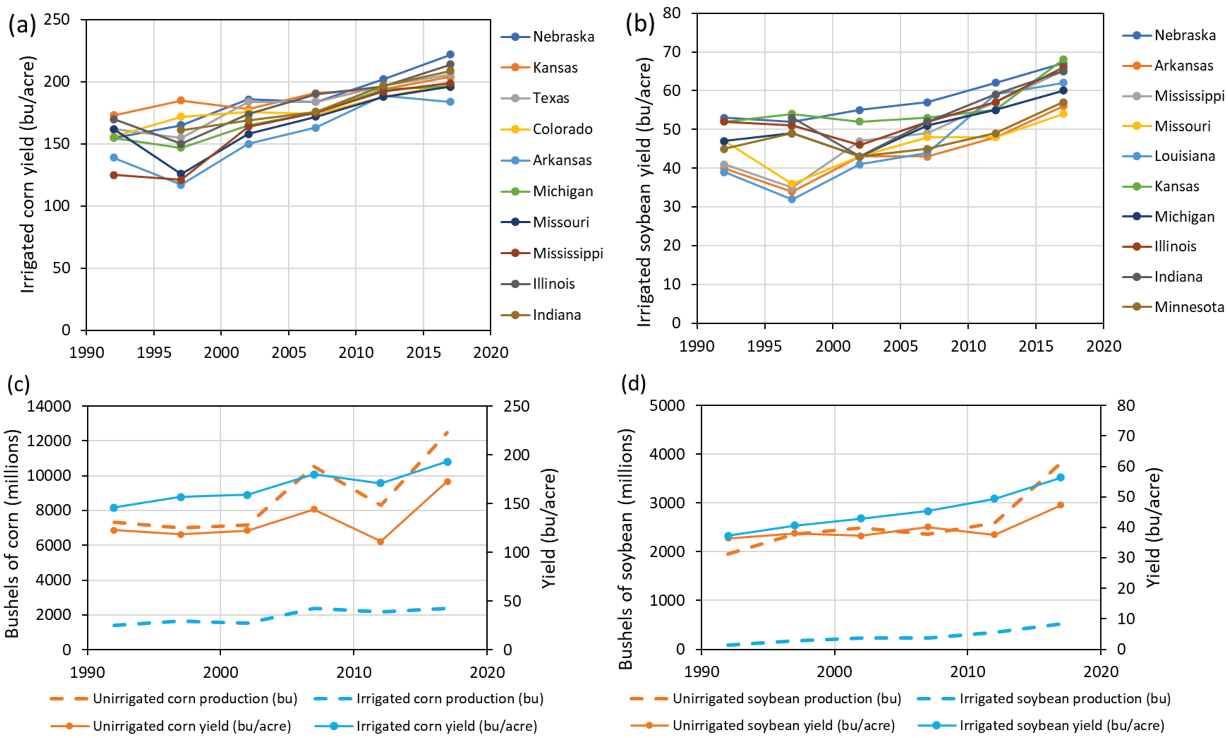


Figure 11.9. Yield of irrigated corn (bushels [bu]/acre) from 1992 to 2017 in the 10 states with historically the most irrigated corn acreage (5-year increments based on data from the USDA NASS Census of Agriculture, i.e., 1992, 1997, 2002, 2007, 2012, 2017) (a). Yield of irrigated soybeans (bushels [bu]/acre) for the same time period for 10 most irrigated soybean states (b). For comparison, average irrigated and unirrigated yields are shown for corn (c) and soybean (d) (right axes). Total corn and soybean production (c and d) is also shown for both irrigated and unirrigated production in millions of bushels (left axis). Data from USDA NASS (USDA, 2020, 2014, 2010, 2004, 1998, 1994).

11.3.1.2 Changes in Water Use for Feedstock Production: Satellite and Other Remote Data

One of the challenges in using survey-based data in understanding irrigation dynamics is the lack of information on temporal and spatial variation. The survey-based datasets provide a detailed and comprehensive snapshot in time of irrigation and water management practices across all crops for the entire United States. However, inter- or intra-annual dynamics are not captured, spatial resolution is limited, and survey-based data may be prone to any potential reporting biases or changes in survey methodology. There is a recognized need for moving toward annual mapping of irrigated lands to track irrigation over time (Brown and Pervez, 2014). But, the vast amounts of data involved in developing these satellite-derived maps, as well as the lack of reference data to "ground truth" satellite observations, have been a barrier to this work. However, new computational approaches and data analysis techniques are now being developed and tested to overcome these barriers. Recently, researchers have been leveraging datasets with higher spatial resolution (moving toward 100 feet) from satellite data, such as Landsat imagery, to develop approaches to map irrigation dynamics over time. While there are multiple efforts to apply these approaches across different countries/regions of the world, the focus of this report is on studies in the United States, and in particular, the primary corn and soybean producing regions of the country.

As early as the 1970s the University of Nebraska Remote Sensing Center used remote sensing to count center pivot irrigation systems in Nebraska (Stoddard, 1977). Remote sensing technologies provide opportunities to quantify crop canopy stress, crop water use and yield. USDA-ARS and NASA researchers developed a system to provide daily evapotranspiration with resolutions as fine as 100 feet over large portions of California (Anderson et al., 2018; Anderson et al., 2012) and the technology has been used in several other states. Work in Colorado in the 1980s correlated crop coefficients used for irrigation scheduling to vegetation indices from proximal sensing (Bausch and Neale, 1987) and satellite imagery (Christopher et al., 1990). The relationship was updated by Kamble et al. (2013) and Campos et al. (2017) for the High Plains. Kipka et al. (2016) and Mcmaster et al. (2014); McMaster et al. (2013) integrated fine-resolution remote sensing data into programs to simulate crop development (phenology) and to follow regional crop rotations over large Great Plains areas. Campos et al. (2018) and others have led ongoing research in Nebraska using this remote sensing-based approach to estimate crop evapotranspiration, yield and crop water productivity.

Deines et al. (2017) combined satellite images from Landsat with climate and soil covariables in Google Earth Engine to provide high resolution (100 feet) annual maps for irrigation for all crops from 1999 to 2016 for the Northern High Plains. The area covered was the greater Republican River Basin region—in the corner of NE Colorado, NW Kansas, and SE Nebraska (Figure 11.10). They then used these annual maps to explore the changes in irrigation from year to year, finding that total area and individual locations of irrigated fields changed substantially as farmers may reduce the number of

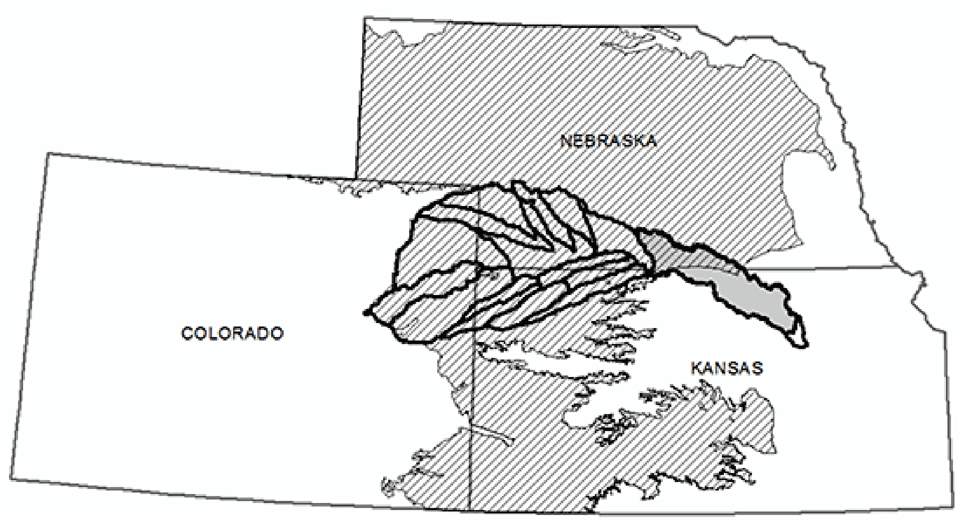
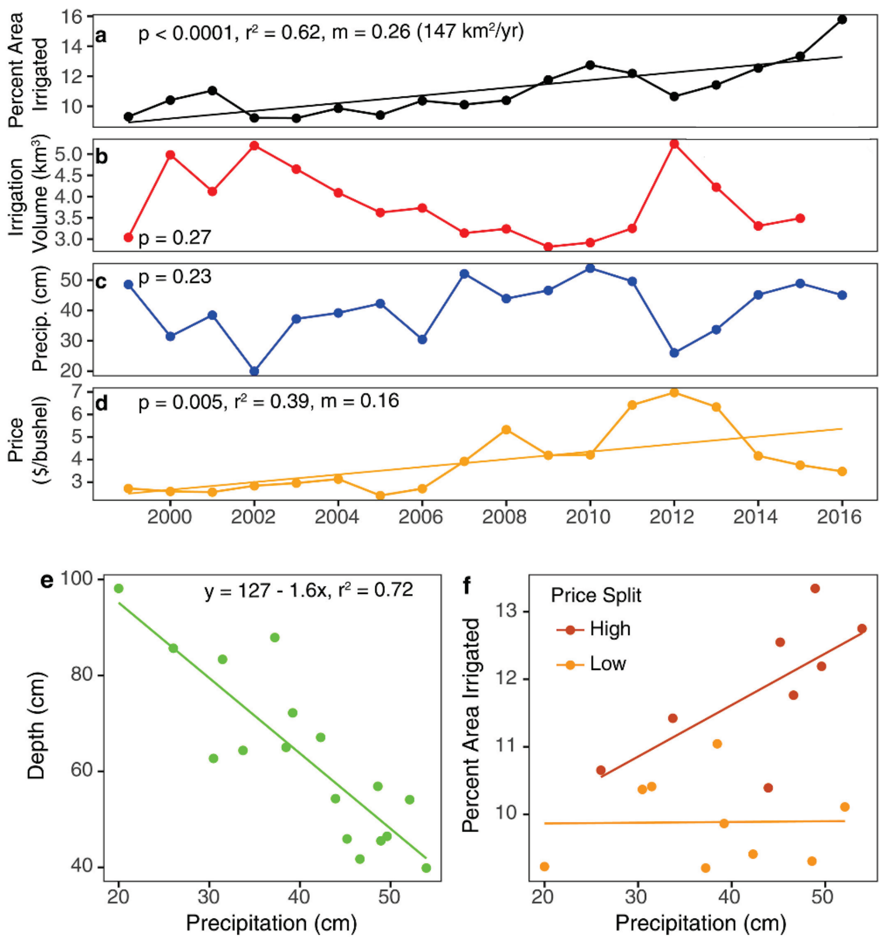


Figure 11.10. The Republican River Basin in Colorado, Nebraska, and Kansas overlain on a map of the High Plains aquifer. Heavy black lines outline the basin and its subbasins. Hatching indicates the extent of the aquifer within the three states. Source: Brookfield and Wilson (2015).

irrigated fields and irrigate those more heavily. They also used the maps to explore the factors that influenced irrigation extent (Figure 11.11a) and volumes (Figure 11.11b) and applied statistical modeling to see how natural drivers (precipitation, Figure 11.11c) and economic drivers (commodity prices, Figure 11.11d) influenced irrigated areas or irrigation intensity over time. In the Republican River Basin in Colorado, Nebraska, and Kansas from 1999 to 2016, statistical modeling showed that in dry years (lower precipitation), irrigated area decreased, but irrigation intensity actually increased as farmers irrigated more heavily over each field (Deines et al., 2017). The data also showed that irrigated acreage generally increased over the time period (average of 57 square miles per year) (Figure 11.11a). There was no statistically significant trend in irrigation water volume over the full time period (Figure 11.11b), although the increase in volume during the drought season in 2012 is observed when precipitation fell sharply



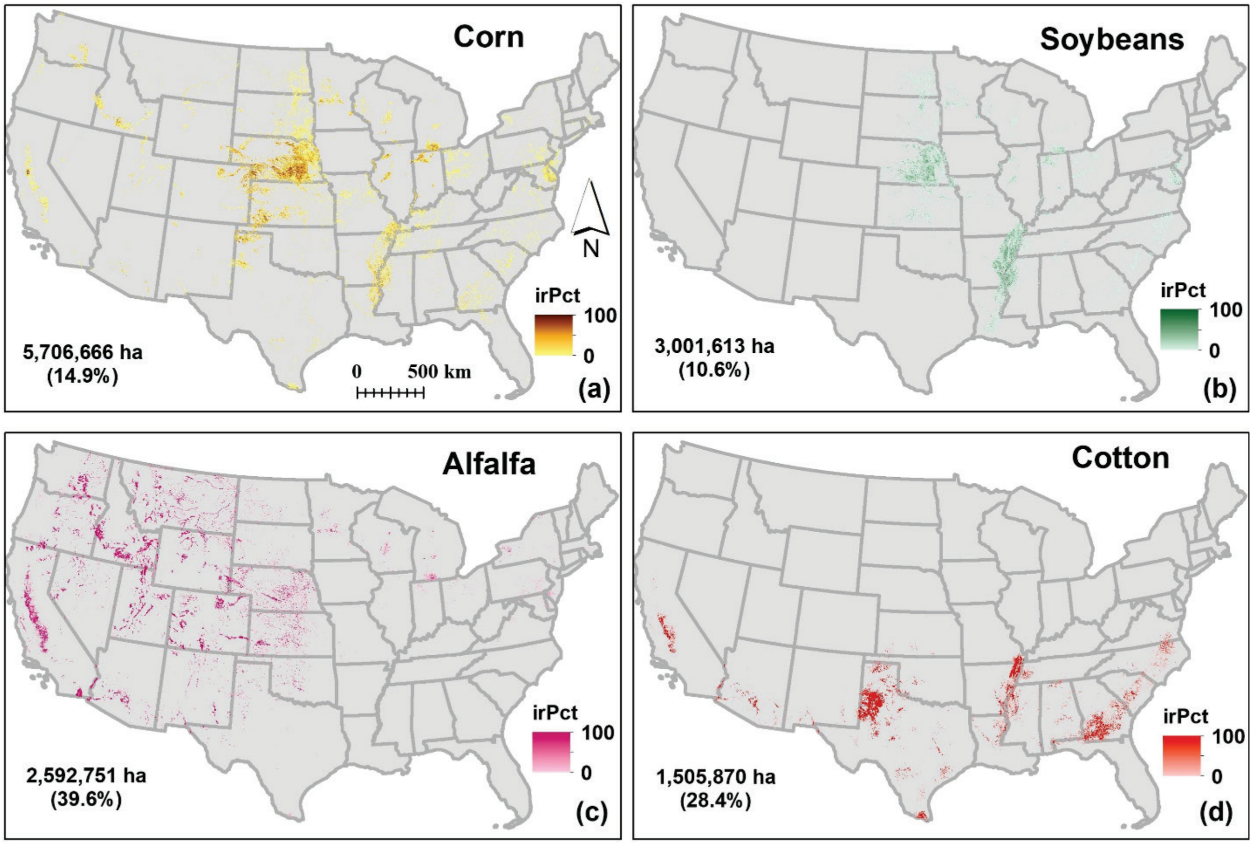
cm = centimeters; km^3 = cubic kilometers; yr = years

Figure 11.11. Irrigated area over time and associated drivers. For the portion of the Republican River Basin overlying the High Plains Aquifer: Percent irrigated area from the Annual Irrigation Maps-Republican River Basin (AIM-RRB) dataset. Rate of change (meters) is given in percent and actual area (a). Irrigation water volume (b). Precipitation from December 1 to August 31 (c). Corn price in 2016 dollars (d). Linear regression of irrigation application depth (volume/area) versus precipitation (e). Trends in irrigated area versus precipitation for years with high and low prices (f). Source: Deines et al. (2017) (used with permission).

(Figure 11.11c). When looking for these price-induced behavioral responses, the authors modeled the impact of previous years' commodity prices and separated the years into low and high prices to assess the response in terms of both the extent of irrigation and intensity (Figure 11.11d-e). They suggest that "when price was low, irrigated area was low regardless of precipitation" but that "high prices incentivized irrigation expansion, but was modulated by annual precipitation" (Deines et al., 2017). However, while price and weather are important factors, irrigation practices are also determined by legal constraints in regions such as the Republican River Basin, where water rights have been highly contested.

The research group built on this same set of maps and methodology to further explore other factors. Specifically, Deines et al. (2019) and Deines et al. (2021) examined the efficacy of the state of Kansas' Local Enhancement Management Area (LEMA) program for groundwater conservation. This is discussed further below with factors that can mitigate impacts on water demands and availability (see Box 11.1). Utilizing annualized maps at high resolution to assess the drivers of irrigation changes will provide valuable information. The factors determining irrigation practices are complex and may be due largely to price, but are also constrained by water availability, production costs (including land prices), risk and prices of other crops.

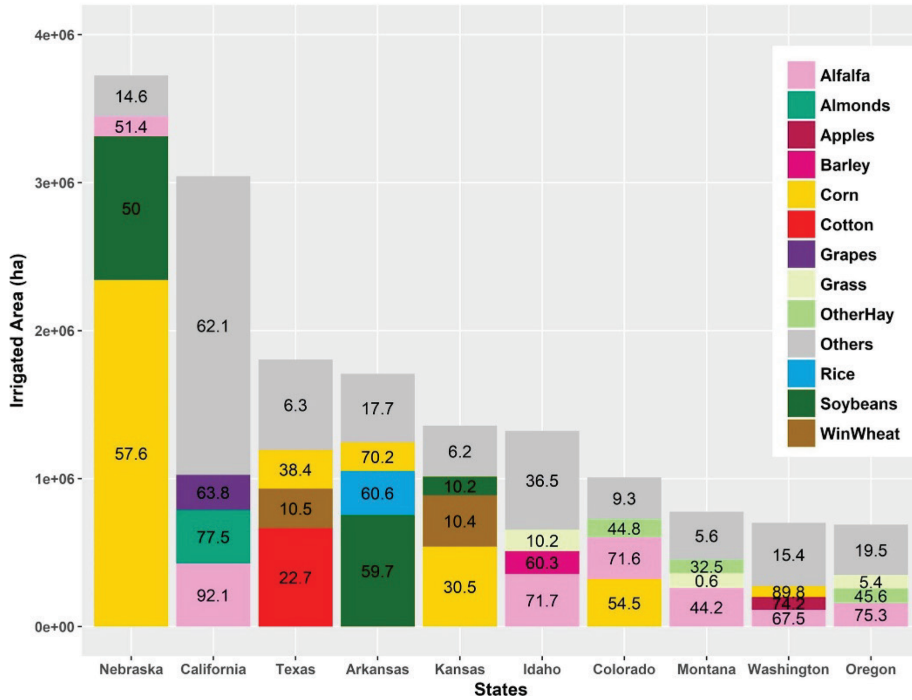
Xie et al. (2019a) have also used Landsat-derived data to develop an approach to map irrigated croplands across the entire conterminous United States (CONUS), therefore encompassing a much broader geographical area at the same 100-foot resolution. Their mapping approach enables identification of crop-specific irrigated areas (see Figure 11.12 for distributions of irrigated areas, aggregated from 100-foot to 0.6-mile resolution, and Figure 11.13 for a summary of most irrigated crops for the top 10 irrigated states). With the caveat that they represent different years, for comparison, the Landsat-derived mappings (for 2012) shown in Figure 11.12 provide a very similar picture of the distribution of irrigated corn acres with what was shown earlier in the survey-based data (for 2017) from the USDA Census of Agriculture (Figure 11.5, left side).



ha = hectares; irPct = percent of crop area irrigated per square kilometer

Figure 11.12. LANID (Landsat-based Irrigation Dataset)-derived and CDL (Cropland Data Layer)-derived distribution of irrigated corn, soybeans, alfalfa, and cotton in 2012. Maps were aggregated to 0.6-mile (1-kilometer) resolution for visualization purposes). Numbers in the bottom left of panels are the total irrigated area of the crop and the percent of crop total area that is irrigated nationally. Source: Xie et al. (2019a) (used with permission).

Figure 11.13 shows the large areas of irrigated acres for Nebraska (primarily for corn followed by soybean) as well as Kansas (corn followed by winter wheat) and Texas (corn and cotton). Because of the large geographic scope, and therefore large amounts of data, their work differed from Deines et al. (2017), because they had not yet developed multiple annual maps to examine trends over time. Xie et al. (2019a) applied their methodology to capture circa 2012 for the CONUS, but noted that they had set up the computational approach and platform to allow extension to other years to produce annual maps.

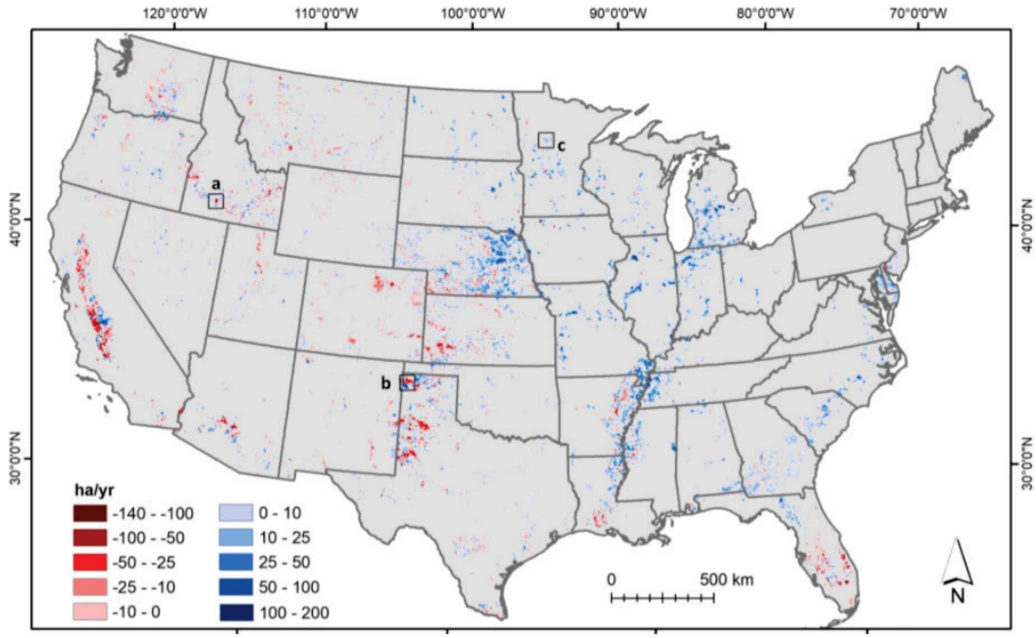


ha = hectares

Figure 11.13. Top three most irrigated crops (by area) for the top 10 irrigated states. The numbers show the crop-specific irrigation percentage within each state. Note that crop area used to calculate irrigation percentage of "Others" refers to all other crops. Source: Xie et al. (2019a) (used with permission).

The Xie et al. (2019a) paper focused primarily on the methodology for a single year. More recent work by Xie et al. (2019b)[14] and updated in Xie and Lark (2021) extended the methodology to generate annual 100-foot resolution datasets of irrigated agriculture across the CONUS for all years between 1997 and 2017. This approach promises to be a bridge between the USDA survey data and satellite-derived data. Their results showed they could generate datasets (Landsat-based Irrigation Dataset for the US, LANID-US) that could reconstruct the USDA-NASS data at the county and state level (for census years 1997, 2002, 2007, 2012, and 2017) and provide annual estimates for the periods between census years. Annual changes of irrigation intensification for the 1997–2017 time period show that most increases were mainly located in the eastern United States (Figure 11.14) (Xie et al., 2019a). In terms of the irrigation dynamics specifically across the HPA, increases were seen in parts of Nebraska, while reductions were seen in the southern HPA, for example, Texas, where lands growing cotton saw reduced irrigation. Irrigation expansion is tied to water availability. Xie and Lark (2021) suggest that groundwater depletion and recharge differences may explain some of the subregional variations of irrigation dynamics, noting the irrigation gains in parts of the northern HPA and irrigation declines in the central and southern HPA.

[14] This work was presented as a poster at the American Geophysical Union (AGU) 2019 Fall Meeting. Xie and Lark (2021) is a more recent, published version of this analysis.



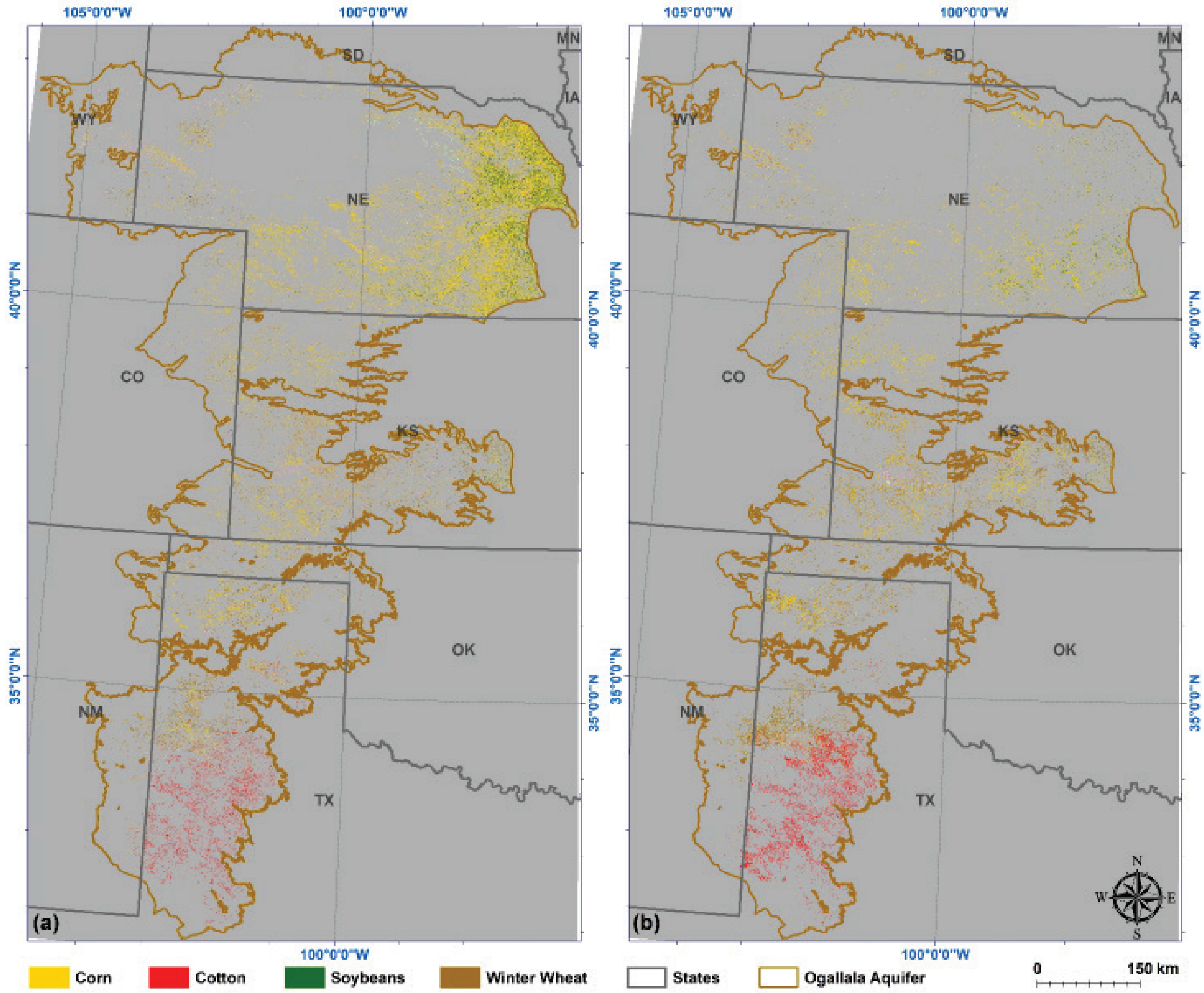
ha = hectares; km = kilometers; yr = years

Figure 11.14. LANID-derived spatially explicit irrigation trends during 1997–2017 at pixel scale. Rates of change (hectares per year [ha/yr]) are modeled using linear regression and calculated for each 3.7 mile x 3.7 mile grid. Changes are shown only for grids with significant trends (p value of linear model <0.05) or areas with an irrigated area > 5%. The rest is shown in gray. Source: Xie and Lark (2021) (used with permission).

For the HPA in particular, Xie and Lark (2021); Xie et al. (2019b) found overall increasing area of irrigation at the county and state level from 1997 to 2017, while noting that localized trends and patterns (both increases and reductions in irrigation) exist both within the HPA, but also more broadly across the country. In a related research brief (Xie et al., 2019c), the authors provided additional detail regarding the changes over the HPA. They found that irrigated croplands in the HPA increased from approximately 14 million acres to 15 million acres between 2000 and 2017, an annual average rate of expansion of 186,000 acres per year, most notably in Nebraska with increases of 170,000 acres per year.

These studies (Xie and Lark, 2021; Xie et al., 2019a) show annual changes in irrigation for all crops. For irrigation changes by crop type, Xie et al. (2019c) provide a preliminary analysis using the LANID dataset.[15] They found that corn and soybeans were the most common crops grown in areas with new irrigation (see Figure 11.15). These increases are most notable in eastern Nebraska for both corn and soybean. Figure 11.15 shows the irrigation intensification between the periods 2000–2008 and 2009–2017 in eastern Nebraska with a very high level of resolution (0.6 mi resolution), compared to the acreage changes that are shown at the county scale based on the Census of Agriculture data in Figure 11.5 (shown above in section 11.3.1.1). At this much coarser level of resolution, Figure 11.5 shows some counties with increases in irrigated acreage and other counties with decreases between 2007 and 2017.

[15] Note that this research briefing has not yet been published in the peer-reviewed literature (Xie et al., 2019c).



km = kilometers

Figure 11.15. Crop-specific changes in irrigation: irrigation intensification (a) and irrigation reduction (b) between the periods 2000–2008 and 2009–2017. Only four major crops are shown. Source: Xie et al. (2019c) (used with permission).

As noted by the authors, this "approach holds promise for characterizing broad-scale trends in irrigation while also capturing critical fine-scale details in spatial and temporal dynamics" (Xie et al., 2019b). Looking ahead, these satellite-derived maps at a national scale can capture changes in irrigation across the continental United States on an annual basis and can help provide additional information about changes in the proportion and intensity of irrigation of local feedstocks around biorefineries. This is discussed in section 11.7.4.

Irrigation dynamics, such as those shown in Figure 11.14, may have differing trends based on surface water versus groundwater availability. Past expansion of irrigation in Nebraska was largely related to availability of the HPA, while expansion of irrigation in the MonDak region, for example, is tied to availability of surface water. In the southern Great Plains (Kansas, Texas and New Mexico) irrigated area has declined due to decreasing availability of water from the HPA (Evett et al., 2020a). Currently, expansion of irrigation in Nebraska is being curtailed as the water resource has become overallocated, while expansion may continue in the MonDak region for some time (Evett et al., 2020a), because the region has potential for further irrigation expansion of more than 500,000 acres (>200,000 hectares) (USDA-ARS, 2019). Earlier work by Brown and Pervez (2014) also noted irrigation increases over the HPA and mapped pre-EISA period changes comparing 2002 and 2007 data.

11.3.1.3 Changes in Water Use for Feedstock Production: Modeling-Based Studies

As discussed in Section 11.4, a common modeling framework for assessing both hydrologic and water quality impacts is the Soil and Water Assessment Tool (SWAT) Model (also see Chapter 10). This model has been used broadly for assessing impacts of biofuel scenarios (very often for cellulosic feedstock production of corn and soybean) for specific watersheds, as will be discussed in a later section. However, issues with the SWAT irrigation algorithms limit the usefulness of SWAT for analysis of irrigated acres, particularly for deficit irrigation.

A number of studies have attempted to evaluate changes of water quantity and quality in response to different scenarios of changes in Land Cover Land Management (LCLM) driven by biofuel production (Panagopoulos et al., 2017; Deb et al., 2015; Gu et al., 2015; Lin et al., 2015). However, none of those studies emphasized irrigation water use due to the difficulties in capturing irrigation schedule (amount and timing) accurately at large scales. Results from those studies show that the impact of LCLM change on streamflow is not significant (less than 1%). It is noteworthy that in addition to input uncertainty of irrigation schedule, it is also very challenging to capture the spatial and temporal variation of weather at large scales, and spatial data on where tiling occurs is generally lacking. Furthermore, irrigated corn and soybean are a small portion of corn and soybean production in general (see section 11.3.1.1), and is likely a negligible amount for large scale SWAT analyses of biofuel impacts of corn and soybean production.

Lin et al. (2015) applied the SWAT watershed model for the Red River of the North Basin (along the border of North Dakota and Minnesota) in order to assess land use change impacts on hydrology and water quality, with a specific focus on the pre- and post-impacts of the 2007 EISA. The study watershed is not a traditionally irrigated area, but one that observed increases in corn and soybean production following EISA. For this primarily rainfed area, SWAT results show that the magnitude of peak flows resulting from spring snowmelt did not change from pre-EISA to post-EISA, although the variation of downstream streamflow was estimated to be greater under post-EISA than under pre-EISA. This indicates that it is more challenging to estimate spring snowmelt floods under the post-EISA land use scenario.

More relevant to water stress and water availability issues due to biofuel production emphasized in this chapter, no SWAT-based studies have focused on croplands that are traditionally irrigated, including areas over the HPA. Issues with the SWAT irrigation algorithms and modeling of leaf area index reduce confidence in the model for simulations under irrigated conditions (Chen et al., 2019; Chen et al., 2017; Marek et al., 2016a; Marek et al., 2016b). Simulations under deficit irrigation are more severely affected (Marek et al., 2015). Work to improve the SWAT irrigation algorithms has provided improvement but requires further testing and development under limited (deficit) irrigation conditions (Chen et al., 2018) before SWAT can be considered a reliable tool for studies of crop water use and its impacts on runoff and recharge under irrigation.

11.3.1.4 Changes in Water Availability

While the previous sections discuss irrigation water consumption, another key question is how changes in irrigation trends may affect water availability in the region, particularly for critical groundwater aquifers. This section will generally discuss the HPA and trends over time for irrigation of all crops, not irrigation specifically for biofuels or their feedstocks such as corn or soybean.

The HPA is "the most intensively used aquifer in the United States" (Maupin and Barber, 2005). Extensive irrigation in the region has led to declines in groundwater levels across large sections for many decades (as shown by changes in groundwater levels from predevelopment to 2015 in Figure 11.16). Declines over this period are larger in the southern section, where the aquifer is thin and irrigation demands are greater (Haacker et al., 2016; Smidt et al., 2016). Some areas of the southern HPA show declines in water levels of greater than 150 feet (Figure 11.16). In contrast, certain areas of the northern HPA have seen increases in water levels from predevelopment to 2015. Between 2013 and 2015 (Figure 11.16, right panel), the rate of declines are more uniform across the HPA, although some increases are still visible along the Platte River and northeastern Nebraska.[16]

With the strong caveat that these represent very different timescales, if one were to overlay the growth in irrigated acres between 2000–2008 and 2009–2017 as shown in Figure 11.14 and Figure 11.15 with the areas of largest historic declines or rises in water levels in Figure 11.16, the picture would be highly varied. For example, any increase in irrigation demands over the western Kansas and Texas panhandle would be contributing to long-term declines in water levels. The impact of the expansion of irrigated acres in eastern Nebraska, as shown in Figure 11.15a, would vary locally, as some parts of that region north of the Platte River show rises in water levels, other areas show historic declines of approximately the same magnitude (5–10 feet or 10–25 feet from predevelopment to 2015).

[16] In the Northern HPA, aquifer recharge has been due to primarily to precipitation (as well as some seepage from canals) continuing to exceed outflows due to irrigation, base stream flows, and evapotranspiration.

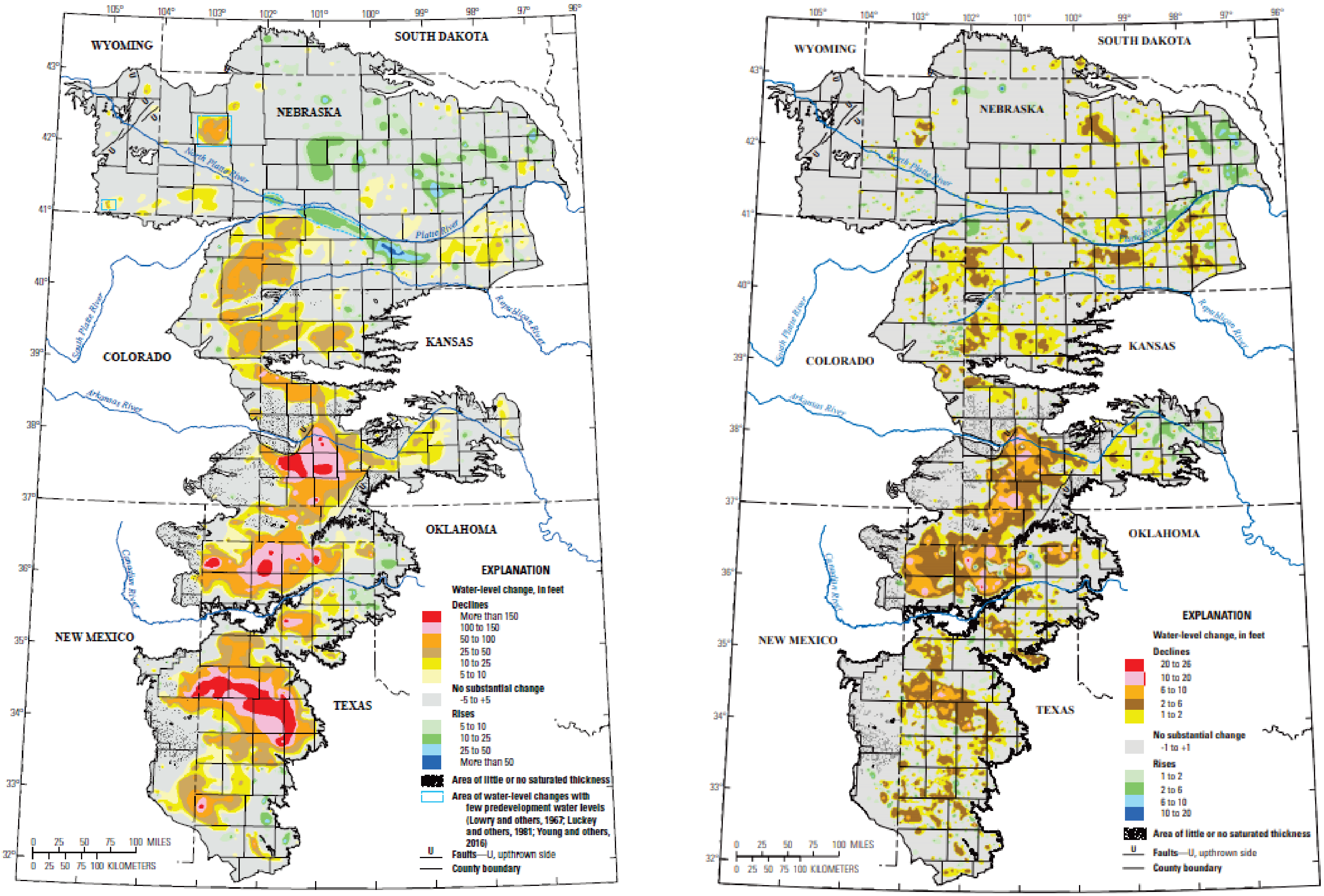


Figure 11.16. Changes in groundwater levels in the HPA Aquifer from predevelopment (around 1950) to 2015 (left panel) and 2013–2015 (right panel). Source: McGuire (2017).

A review by Smidt et al. (2016) discussed the various drivers, including both direct drivers and how indirect policies may have helped to protect HPA groundwater or not. They suggest that the "biofuel mandate generated a profitability incentive to farmers, ultimately increasing the planting of water-intensive biofuel crops (e.g., corn)." This question of attribution of the impacts of the RFS Program on prices and production of irrigated crops is covered in Chapters 4 and 6. What is clear is that continued trends of irrigation over the southern part of the HPA are not sustainable, as long as extraction rate greatly exceeds recharge. Indeed, both overall irrigated area and water applied per acre are decreasing in the Texas panhandle and western Kansas in concert with decreased water availability from the HPA. As discussed in section 11.7, climate change is also a critical factor in both the extraction rate (through irrigation demands) and recharge levels (through precipitation). According to the Fourth National Climate Assessment, "current extraction for irrigation far exceeds recharge in [the High Plains] aquifer, and climate change places additional pressure on this critical water resource" (Hayhoe et al., 2018).

11.3.1.5 Changes in Water Use for Biofuel Facilities

Water use in biorefineries represents a point resource demand that affects local and regional freshwater availability. In terms of total lifecycle water use, the conversion of corn to ethanol, or soybean oil to biodiesel is a small percentage of the overall water demand. Recent estimates of biorefinery water use are 8.7% for corn to ethanol and 1.1% for soybean biodiesel (see Figure 11.22 and Figure 11.23 for more details and comparison to petroleum-based fuels on a gallon per megajoule basis).[17] Water is used for pretreatment and processing of corn grain to ethanol as well as for cooling towers, which accounts for the major share of water use. Decreased water resources would interrupt existing operations and constrain new project development. In particular, the effect of limited water resources could be detrimental in drought-prone regions. The progress of technology development for conventional and advanced biofuel production processes in the United States has been reviewed by several groups over the last two decades (Warner et al., 2017; Mueller and Kwik, 2013; Wu et al., 2009; Wu, 2008; Shapouri and Gallagher, 2005). A recent study from Argonne National Laboratory, provides a comprehensive report on water resources, water use, and water and wastewater management for the biofuel industry, representing the most up-to-date analysis of commercial-scale dry mills that were available in the United States as reflected in 2017 plant operation data (Wu, 2019).

The biofuel industry has made a concerted effort to reduce both water and energy consumption, diversify energy sources, increase and maintain efficiency, and recycle and reuse water (Wu, 2019). According to the report, more than half of the dry-mill ethanol plants source water from wells

[17] The biorefinery water use is relatively consistent and has been declining over the years. However, the percentage of biorefinery water use as a share of total lifecycle water use is affected by the year-to-year variations in weather that drive the water consumption demands for feedstocks.

(groundwater). Less than 40% use a city water supply. Surface streams are used by 7% of facilities (Figure 11.17). Groundwater use represents 56% of total water volume (Figure 11.17), making it the main water source for ethanol plants in the last few decades. Because of the concern of groundwater water level decline in aquifers where the rate of withdrawal exceeded rate of recharge and in response to local regulation, some biorefineries switched to surface water resources. As a result, groundwater-dependent biofuel plants are decreasing in number. Another trend of water resource selection is using alternative water resources in water-stressed regions. At present, wastewater from power plant cooling towers and reclaimed municipal wastewater have been used by 3% of production facilities (Figure 11.17). With the projected increase of water stress in certain regions, production facilities in those regions are more likely to switch away from groundwater sources to surface water (which does not reduce local freshwater stress) or alternative water sources (which can reduce stress on freshwater resources).

Within a biorefinery, it takes 2.65 gallons of freshwater to produce a gallon of denatured ethanol on average (Figure 11.18). Compared to previous surveys, water intensity (water consumption per gallon of fuel) has decreased by 12% since 2011 and by 54% in the 19 years between 1998 and 2017. While water consumption decreased, ethanol production yield increased. In 2017, 2.88 gallons of denatured ethanol were produced from a bushel of corn, an increase from 2.82 gallons per bushel in 2012 (Mueller and Kwik, 2013). The report found that most ethanol plants use a natural-gas-fired dryer or an electric dryer, instead of using a steam drying process. Switching from steam to natural gas or electricity in the drying process reduces the need for both fresh water supply and potential associated water treatments.[18] Thus, replacing steam with natural gas or electricity represents a reduction in not only the water footprint, but also the cost associated with water acquisition and treatment. Newer plants with improved energy and

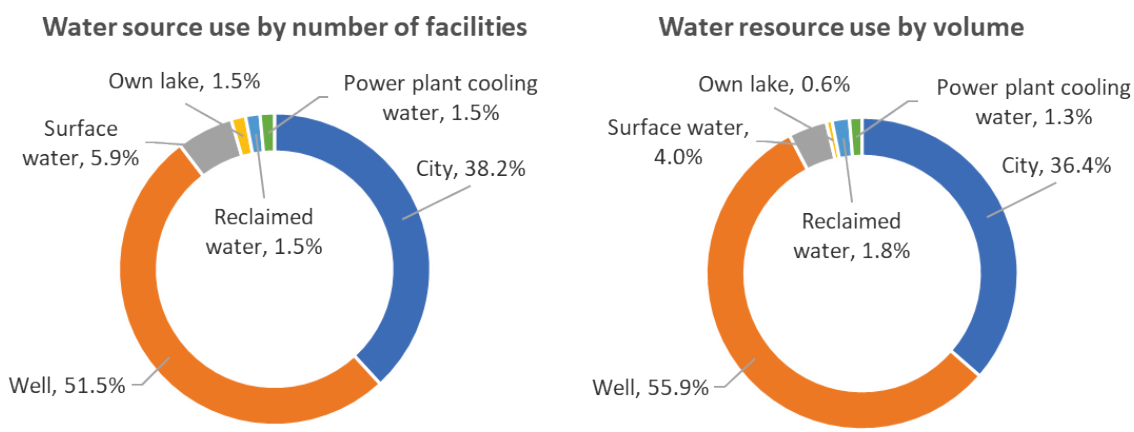


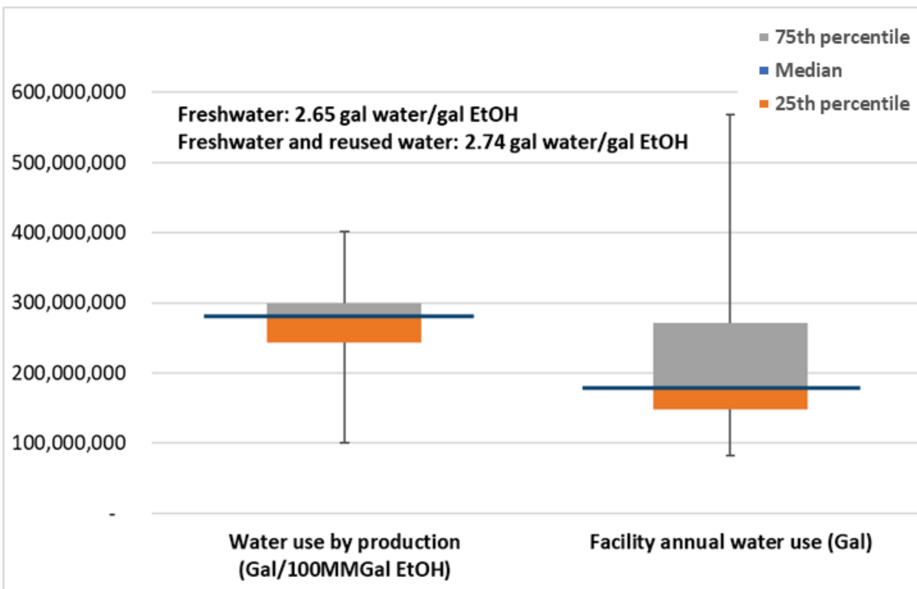
Figure 11.17. Types of water resources used in biofuel production, by number of facilities (left) and by production volume (right). Source: Wu (2019).

[18] Depending on source water quality, the water for steam can require various degrees of treatment prior to use.

steam integration dominate biofuel plants, which illustrate continued water efficiency improvements over time.

Ethanol plants have also conserved water and energy by increasing production of wet distillers' grain and modified wet distillers' grain, reducing the demand for natural gas and electricity for drying. The water content in these co-products is reused as a part of animals' diets in feedlots. As noted in Chapter 3 and elsewhere, however, wet distillers' grains are less commonly produced as a co-product (~9% of biorefineries), so this likely would only have modest and local effects on water balance.

The study also found that 5% of plants have implemented on-site electricity generation to replace grid electricity, and several plants have become net electricity exporters. Such change represents a decrease in water use associated with electricity generation. Analysis of the survey results demonstrated that the production practices of the biofuel industry increasingly address water conservation, moving toward biofuel production that is energy-efficient, water-efficient, and environmentally sustainable.



gal = gallons; MMGal = million gallons

Figure 11.18. Water intensity (fresh and reused water consumption per gallon of ethanol produced): maximum, 75th percentile, median, 25th percentile, and minimum value of water consumption per 100 million gallons of ethanol produced, and annual facility total water consumption. The centerline inside the boxes represents the median value. The top of the gray box displays the 75th percentile, and the bottom of the orange box displays the 25th percentile. The maximum and minimum values are displayed with vertical lines ("whiskers") connecting the points to the center boxes. Source: Wu (2019).

11.3.2 New Analysis

A new analysis performed for this report is the SWAT application to the Missouri River Basin (MORB), which is discussed in Chapter 10 (see section 10.3.2, also Chen et al. (2021)). The SWAT model was applied to assess recent cropland expansion on water resources for the Missouri River Basin, where the highest rate of grassland to cropland conversion (est. 1.18 % of the total land area was converted from 2008 to 2016 basin wide) have occurred. Simulations of three crop production scenarios represent conversion of grassland to: (1) continuous corn; (2) corn/soybean rotation; and (3) corn/wheat rotation at the locations of observed land use changes over two periods. Chen et al. (2021) reported conversion to cropland resulted in little change in streamflow basin wide (est. 0.4% increase in streamflow by converting to continuous corn; 0.1 increase in streamflow by converting to corn soybean rotation; 0.0 increase in streamflow by converting to corn wheat rotation, see section 10.3.2 for more details).

As noted above in section 11.3.1.3, no SWAT-based studies have focused on croplands that are traditionally irrigated, including areas over the HPA. This indicates the future needs for modeling studies focusing on watersheds with irrigated corn and soybean production.

11.3.3 Attribution to the RFS

The chapter material above focused on the effects of corn and soybean production and biofuels in general, but, for the most part, did not address the effect of the RFS Program specifically. For instance, in the review of the literature (section 11.3.1), studies generally did not directly examine how corn or soybean production attributable to the RFS Program affected water demands and availability. The review instead focused on the broader trends in irrigation patterns (acres and amount applied) for corn and soybeans, which by extension drive the demand for water. Some studies examined the effects of biofuels in general, estimating corn and soybean feedstock demands, and then the associated irrigation water demands for those feedstocks. These studies also highlight the strong regional variation in water demands. In this section potential effects of the corn ethanol volumes used to fulfill a portion of the RFS Program requirements specifically on water demand and availability are addressed to the extent possible. References to land use conversion and attribution estimates in Chapter 6 are included, but additional considerations based on studies in previous sections are noted.

As described throughout this chapter, there are two major mechanisms by which the biofuels production supply chain utilized water resources and can therefore affect water availability. The predominant mechanism is water use for the irrigation of corn and soybeans, the dominant biofuel feedstocks to date. The second mechanism is water use for the biofuel conversion process, much smaller in scale on a regional or national scale, but with potential for local impacts on water availability. Chapter 6 represents the best estimate of attribution of converted acres from grassland to cropland. In that chapter, an estimated 0 to 1.9 million acres in 2016 of additional cropland is estimated to be associated with corn ethanol production attributable to the RFS Program between 2008 and 2016, or approximately 0 to 20% of the observed net increase in United States crop area over this period (see Chapter 6, Table 6.10, 6.11).

That study, however, did not indicate where these RFS-attributable lands were specifically nor how much of any new acreage might have been irrigated.

In Lark et al. (2020), a higher concentration of converted acres was shown to be located in southern Iowa and the eastern halves of the Dakotas. Conversion of acres in rainfed areas such as southern Iowa would be highly unlikely to significantly increase water demands, as these are generally not irrigated acres. In the Dakotas, however, irrigation may be required. Additional work could expand on where the grassland to cropland conversion (as described in Chapter 6) that may be attributable to the RFS Program occurred, and estimate the specific irrigation demands with those converted acres. However, at this point, only a bounding estimate is possible. Some portion of the 0 to 1.9 million acres converted from grassland to new cropland were likely to be new irrigated acres. Of those newly irrigated acres, irrigation trends from Xie and Lark (2021) indicate that these would likely be concentrated in the northwestern states of the corn belt. Further research is needed.

However, changes in irrigation and water demands can and do occur without any changes in land use or conversion from grassland to cropland. Therefore, in addition to land use change, section 11.3.1.1 addressed national changes in irrigated acres and water applied. As noted in the previous section, overall irrigated area in corn, according to USDA survey data, increased from between 9.3 and 9.7 million acres before the 2005 Energy Act to between 12 and 13 million acres reported in the 2008 and 2012 censuses, before declining to 11.6 million reported in 2018. However, these are overarching trends that have not been attributed specifically to the RFS Program or even biofuels generally, given that the survey data do not specify end uses of crops and given that corn acreage was increasing steadily before initiation of the RFS Program. Biofuel feedstocks are often locally sourced. One study used field-level data for the Kansas portion of the HPA to estimate the effect of ethanol plant location and capacity on local irrigation water demand from 2003 to 2017 (Al-Sudani et al., 2020). Looking at irrigation decisions for fields within approximately 30 miles of ethanol plants, they found that a 10% expansion of ethanol capacity increased annual water use by 0.22% per field (4.8 acre-inches per field).

More regional-level studies (e.g., Xie et al. (2019c)) show irrigated croplands (all crops, not only corn) over the HPA increased from approximately 14 million acres to approximately 16 million acres (for all crops) between 2000 and 2017, an annual average rate of expansion of 186,000 acres/year, almost all in Nebraska with increases of 170,000 acres/year. However, that reflects changes in irrigation due to all drivers for all crops. The share of acreage attributable to the RFS mandates is unknown.

11.3.4 *Conservation Practices*

> **Box 11.1. Stakeholder-Driven Groundwater Management**
>
> Programs such as the Local Enhanced Management Area (LEMA) program in Kansas are another mechanism for management of limited groundwater resources. In 2012, Kansas established a framework for irrigators to work with local and state officials to create enforceable management plans for groundwater conservation. The first LEMA starting in 2013 in northwest Kansas included restrictions to reduce total groundwater pumping by 20% compared to 2002–2012 levels, as well as a 5-year allocation of 55 acre-inches per irrigated acre, with flexibility to roll over unused water to the next LEMA.
>
> Based on well records and satellite-based modeling of the LEMA and a business as usual (BAU) scenario, Deines et al. (2021) estimated that groundwater extraction volumes decreased by approximately 25% due to reductions in irrigation application depths and frequency. Estimated cost savings from reduced pumping were about 4.5 times greater than income lost from minor yield penalties. Based on this, Deines et al. find that LEMA promote both economic and water sustainability, while also suggesting that more stringent water targets may be needed to stabilize groundwater levels. See Deines et al. (2021); Deines et al. (2019).

There are a range of opportunities for reducing water use for both feedstock production as well as fuel production processes (see Box 11.1 for an example).[19] There have been a number of improvements to date in irrigation technologies and management practices. Evett et al. (2020a) provided an overview of irrigation practice changes in the Great Plains from the 1940s to present, largely in relation to the HPA. In general, irrigation water conservation strategies can include (1) reducing irrigated area, (2) improving irrigation efficiency, (3) improving crop water productivity (e.g., through deficit irrigation[20]), (4) moving irrigated production to more humid or higher latitude regions where crop water requirements are less and precipitation supplies a greater portion of water needs, and (5) switching to less water-intensive crops.

The overall increase in pressurized irrigation has resulted in greatly improved irrigation efficiency (e.g., 90 to 95% for pressurized systems) from the lesser efficiency associated with most gravity irrigation systems (on the order of 60 to 65%), while also improving crop water productivity due to the fact that in general irrigation uniformity in pressurized systems is much greater than that in gravity flow systems (Evett et al., 2020a). More uniform irrigation application leaves a smaller part of each field underirrigated or waterlogged and thus comes closer to optimum return in crop yield for irrigation applied while reducing losses to deep percolation and runoff. Irrigation management technology is also steadily improving. Up to 30% of irrigators now use some kind of scientific irrigation scheduling, and that percentage is steadily increasing and bringing with it improvements in water conservation. Evett et al. (2020b) reported on precision irrigation advances in the United States to present. All modern center pivot irrigation systems allow for some degree of site-specific or variable rate irrigation, and these occupy 55% of United States irrigated lands. While precision irrigation has been stymied by lack of decision support

[19] Results from the CEAP II report as it pertains to conservation practices related to irrigation were discussed in section 11.3.1.1 (USDA NRCS, 2022).
[20] Deficit irrigation is defined as irrigated at less than the amount required to produce maximum yield, and can be used under limited irrigation water supplies or as proposed means to improve water use efficiency of crops.

systems, recent advances have provided the unattended wireless sensor systems needed to automatically provide plant and soil feedback data to decision support systems (Evett et al., 2020a).

There are also opportunities for improvements in water reuse for biofuel facilities. Water and wastewater management is progressing toward zero liquid discharge (ZLD) through in-facility water reuse and recycling. In this approach, wastewater generated from one production unit such as a cooling tower or boiler is used in another unit such as the fermentation process. More than a third of facilities achieved ZLD by recycling cooling and boiler wastewater blowdown. Eighteen percent of the facilities sent the wastewater offsite to a local publicly owned treatment works (POTW), and less than a half (45.9%) treated it on-site through various means such as evaporation ponds, settling ponds, and other chemical treatment to meet regulated discharge limits. Although significant progress has been made, there are still 2/3 of the process wastewater not being recycled and reused (Figure 11.19). Still, the current level of water reuse in biorefinery is substantial and ranked at the top across industries in the energy sector.

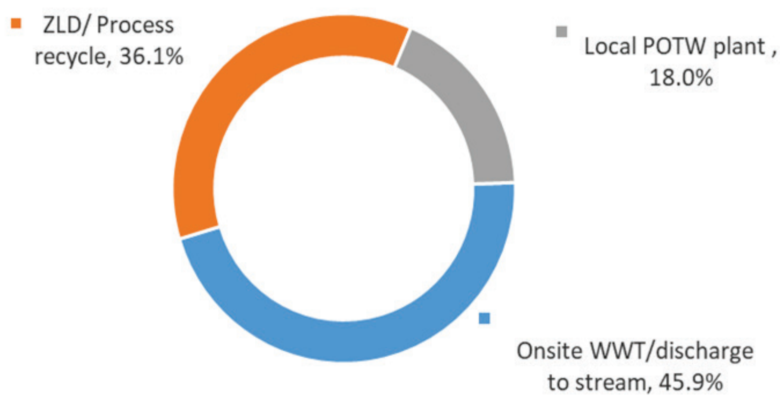


Figure 11.19. Fate of wastewater from biofuel production facilities. Source: Wu (2019).

11.4 Likely Future Impacts

When the Set Rule was finalized on June 21, 2023, EPA published estimates of cropland expansion attributable to the RFS with those biofuel volumes (see Chapter 6, section 6.5).[21] EPA

[21] On July 26, 2022, the United States District Court for the District of Columbia filed a consent decree, which after stipulations from the parties, required EPA to sign a notice of proposed rulemaking (NPRM) to establish 2023 volumes for the RFS Program on or before November 30, 2022, and to sign a notice of final rulemaking finalizing 2023 volumes by June 21, 2023. Growth Energy v. Regan et al., No. 1:22-cv-01191, Document Nos. 12, 14, 15. EPA's proposed RFS volumes for 2023-2025, 87 FR 80582 (proposed and signed on Nov. 30, 2022 and published in the Federal Register on Dec. 30, 2022) available in Docket No. EPA-HQ-OAR-2021-0427 at https://www.regulations.gov, were finalized on June 21, 2023 and published in the Federal Register on July 12, 2023 (88 FR 44468).The estimates of cropland expansion from the RFS Program took considerable time and effort to develop; thus, they are based on the proposed volumes from November 30, 2022, rather than the final volumes from June 21, 2023. Because of that, the acreages discussed here are merely illustrative. However, since the total volumes from crop-based feedstocks changed fairly little between the draft and final rules, they are still relevant here (see Chapter 6, section 6.5).

estimated the biofuel volumes in the Final Set Rule could potentially lead to an increase of as much as 2.65 million acres of cropland by 2025, mostly from soybean expansion in the Midwest, but to a lesser extent from corn expansion in the Midwest and canola expansion in North Dakota. EPA did not include spatially resolved estimates of where that cropland expansion may take place, nor the fraction of those crops that may or may not be irrigated. Given that any increases in these three crops are likely to occur in the Midwest, impacts on water availability may be limited if expansion were to occur in primarily rainfed areas. However, if irrigated acres were to increase in areas over the High Plains Aquifer ranging from western Kansas, eastern Colorado, to the Oklahoma and Texas panhandles, water availability could be affected as the aquifer in these regions is recharged at rates less than current rates of withdrawal. Expansion of irrigation in Nebraska could be limited by overallocation, as discussed earlier (section 11.3.1), while expansion of corn and soybean acreage in rainfed states, such as Iowa, would have little effect on water availability.[22] Expansion of irrigated crops in states such as North Dakota could lead to additional irrigation demands, but as discussed in section 11.3.1, some expansion of irrigated acreage may continue in the MonDak region without impacts on water resources due to availability of surface water.

11.5 Comparisons with Petroleum

Water use in petroleum-based fuel production has been changing as the production technologies advanced in the last decade. Several in-depth studies on water consumption in the various life stages of oil production have been published over the past decade. Water use estimated by Goodwin et al. (2012) and Mangmeechai et al. (2014) investigated site-specific horizontal and vertical oil wells in Colorado. Veil Environmental (2015) developed a produced-water[23] management report documenting state-level water to oil ratios and produced-water reinjection practices based on 2012 data. Ali and Kumar (2017) compared five major onshore and offshore oil-producing sites across the United States, Canada, and Mexico. These studies yielded valuable information about the state of technology and water management in the production processes. Based on a synthesis of the available data, a recent analysis (Wu et al., 2018) reports an updated water use in petroleum fuel production at geospatial resolution and provides a

[22] Numerous studies on potential future effects are watershed specific (e.g., Demissie et al. (2012) for the Upper Mississippi River Basin). These are valuable contributions to the literature and several of these studies were discussed and summarized in the RtC2. However, they are not focused on here due to their focus on specific river basins and watersheds. More importantly, because these studies have biofuel volumes that differ significantly from the likely future as described in Chapter 6, they do not provide insights into the likely future impacts. Most modeled scenarios project high volumes of cellulosic-based biofuel and use full RFS2 volumetric targets or other national clean fuel or low carbon standards to estimate feedstock demands.

[23] Occurring naturally in the formation itself or due to water injection, produced water is the water portion of an oil-water mixture with a high concentration of dissolved solids that is pumped to the surface.

comparison to earlier estimates for the United States. The study was conducted for each Petroleum Administration Defense District (PADD)[24] before being aggregated and weighted to national level.

The report concludes that water consumption in oil extraction and production (E&P) is highly sensitive to the age of the oil well, the recovery technology employed, and the degree of produced-water recycling and reuse. Primary oil recovery[25] requires only 0.2 gallons of water per gallon of crude oil produced. However, U.S. onshore oil production relies heavily on secondary recovery, which extends an oil field's productive life by injecting water or gas to displace oil and drive it to the well bore. In 2014, 42% of U.S. crude oil production used water flooding, a decrease of 8% from 2006. This secondary recovery technology requires an average of 15.7 gallons of water per gal of crude oil recovered and, as a result, accounts for 94% of the water injected into onshore wells for oil recovery. Use of water flooding technology has been in decline over the last decade; it decreased 25% from 2006 to 2014. In most regions, produced-water supplies much of this injection water. It was estimated that on average 46% of produced water is reinjected to oil wells for production nationally.

Nationally, on average it takes a net 4.5 gallons of water to produce 1 gallon of crude oil from U.S. onshore wells on average, with a range of 0–7.6 gallons for the five oil production regions (PADD I, II, III, IV, and V). Note that there are significant variations from oil field to oil field. Produced water is especially low in parts of West Texas, necessitating significant use of saline groundwater for injection. This compares to 2.1–5.4 gal/gal estimated based on data available prior to 2009 (Wu et al., 2009). PADDs III and V are both at the national average.

Figure 11.20 presents the net water consumption rate and total crude oil production from onshore wells in the PADD regions. PADD III has shares of onshore crude production that are similar to that of net water use, and so does PADD V. PADD IV consumes negligible amounts of injection water, and its oil production shares are small (<10%). In contrast, PADD II accounts for 32% of total water consumption to produce 24% of total crude in the United States. PADDs II and III together account for 76% of U.S. onshore crude oil production. Reducing injection water consumption in these regions could have a much greater national impact than other regions.

Results from the study show that the type of recovery technology and the share of production contributed by that technology are important factors in water consumption for oil recovery. Produced-water yield and the degree of produced-water reinjection for oil recovery also have significant effects on water consumption. Increase of the total crude production, decrease of produced-water yield, and the decrease in produced-water reinjection for oil recovery led to an increase of net water consumption rate.

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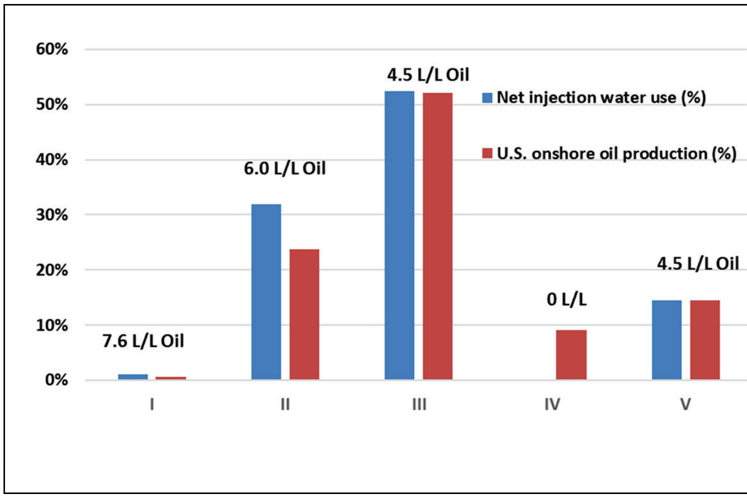
[24] See Figure 11.20 for PADD definitions.

[25] The crude oil or natural gas recovered by any method that may be employed to produce them where the fluid enters the well bore by the action of natural reservoir pressure (energy or gravity).

On the other hand, wells with large amounts of produced water can have low net water use if there is extensive produced-water reinjection (as in PADD IV).



PADD	States
I East Coast	ME, VT, NH, MA, MA, CT, RI, NY, PA, NJ, DE, MD, WV, VA, NC, SC, GA, FL
II Midwest	OH, MI, KY, TN, IN, IL, WI, MN, IA, MO, ND, SD, NE, KS, OK
III Gulf Coast	AL, MS, AR, LA, TX, NM
IV Rocky Mountain	MT, WY, CO, ID, UT
V West Coast	WA, OR, NV, AZ, CA, AK, HI

L = liters

Figure 11.20. Onshore oil production and water consumption for major U.S. oil-producing regions (PADD). Note that water consumption for injection in PADD IV is negligible. Source: Wu et al. (2018).

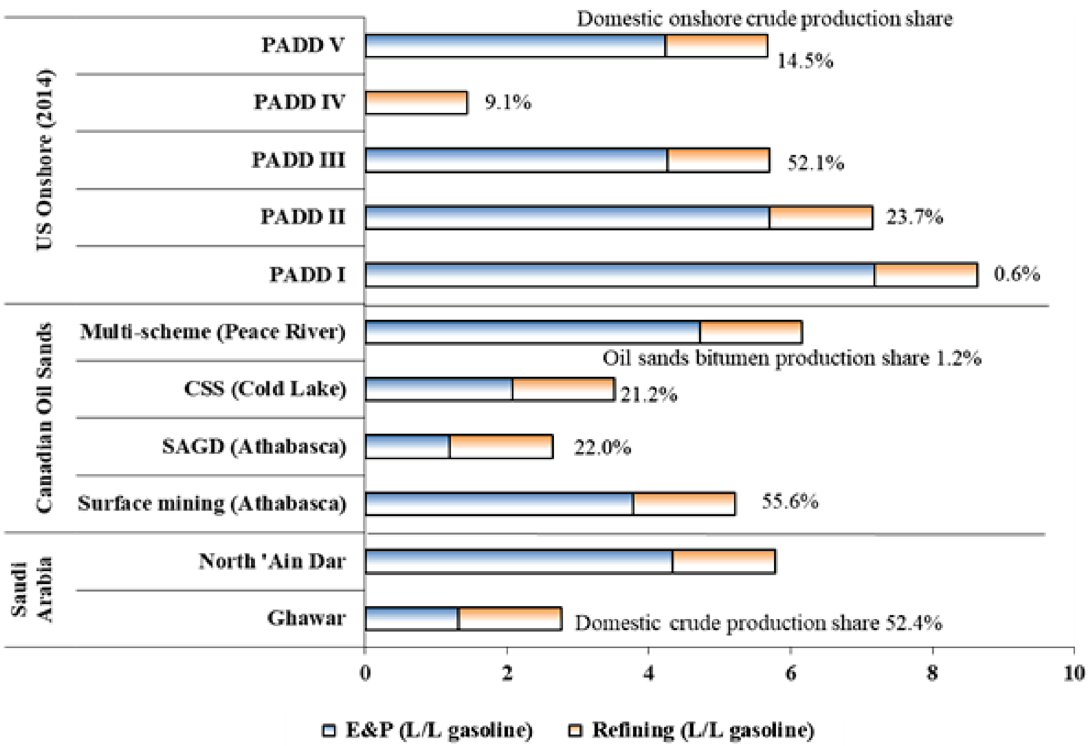
In the oil E&P stage, the report found that although enhanced oil recovery (EOR), via technologies like steam injection and CO_2 flooding, is less prevalent than water flooding, it accounts for an increasing share of onshore production—up to 9%. As of 2014, water inputs for steam injection and CO_2 flooding represented nearly 5.3% of total water injection in domestic onshore wells, which is a significant decrease from the previous estimate of 17% for 2006. CO_2 flooding is dominant in PADD III, whereas steam injection is prevalent in PADD V. Alternative water sources for oil recovery have been explored to displace fresh groundwater. Brackish water was also used as injection water in PADD III.

In contrast to E&P, oil refining consumes a relatively small amount of water—an average of 1.5 gallons per gallon of crude oil processed. Combining oil E&P and refining, producing 1 gallon of gasoline from conventional crude in Saudi Arabia or in the United States can consume as little as 1.4 gallons or as much as 8.6 gallons of water (Figure 11.21). The regional weighted average water intensity in the United States is an estimated 5.6 gallons of water per gallons of gasoline, which remains similar to the value a decade ago.

While a majority of water consumption in biofuel production is feedstock water use, the water intensity varies with types of feedstock and regions it was grown. For current biofuels based on corn and soybean, irrigation water intensity varies substantially across crop production regions. According to Wu et al. (2018), blue water[26] use intensity of irrigation and fuel production for corn ethanol can be 8–10 fold

[26] Blue water is water supplied from surface or groundwater.

higher than petroleum gasoline in some regions. It takes at minimum 8.7 gallons of blue water to produce a gallon of ethanol, if the corn is grown in regions that receive abundant rainfall, such as Iowa. At the



L = liters

Figure 11.21. Net water use for gasoline production from conventional (United States and Saudi Arabia) and nonconventional crude (oil sands) by lifecycle stage, location, and recovery method. Lifecycle stages are extraction and production (E&P) in blue and refining in orange. Source: Wu et al. (2018).

higher end, it takes 160 gallons of blue water per gallon of ethanol. Comparatively, the blue water use intensity in petroleum gasoline production averages 5.6 gallons of blue water per gallon of gasoline (1.4–8.6 gal/gal) (Table 11.1).[27] While regionally specific analysis is critical in comparing water intensity of biofuel and petroleum fuels, even the highest water intensity petroleum gasoline production is lower than the most water-efficient corn-based ethanol.

Several tools have been developed to compare biofuels with petroleum from a lifecycle perspective. In the lifecycle assessment (LCA), in addition to direct blue water consumption in biofuel lifecycle stages—irrigation and conversion—indirect water consumption across the supply chain for the production of fertilizer, enzymes, and other agriculture chemicals, cooling, and production of electricity and fuels used

[27] This comparison focuses on blue water because both bio-based and petroleum fuels use blue water, which allows direct comparison of fuels as well as regional variation. However, green water (i.e., precipitation) in crop evapotranspiration reduces the amount of precipitation that is available for wetlands, streams, lakes, and rivers. Even in rainfed acres, there is green water use associated with feedstock production whether for corn, soybean, or cellulosic crops.

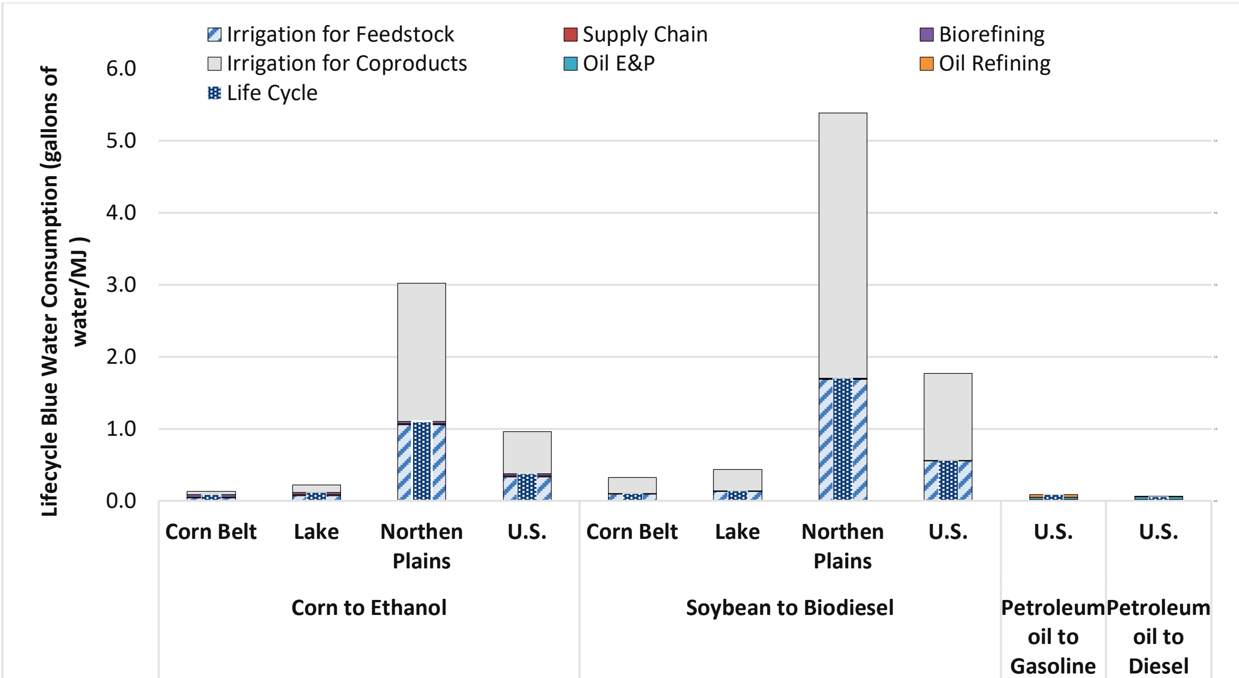
Table 11.1. Water consumption for ethanol and petroleum gasoline production. Source: Wu et al. (2018).

Fuel (Feedstock)	Net Water Consumed [a]	Major Factors Affecting Water Use
Corn ethanol	8.7–160 gal/gal ethanol [b]	Regional variation caused by irrigation requirements due to climate and soil types
Gasoline (U.S. onshore conventional crude)[c]	1.4–8.6 gal/gal gasoline	Age of oil well, production technology, and degree of produced water recycle

[a] In gallons of water per gallon of fuel specified.
[b] Water use for processing ethanol co-product is allocated using mass-based method. Data cover water consumption for corn in USDA regions 5, 6, and 7.
[c] PADD I, II, III, IV, and V combined.

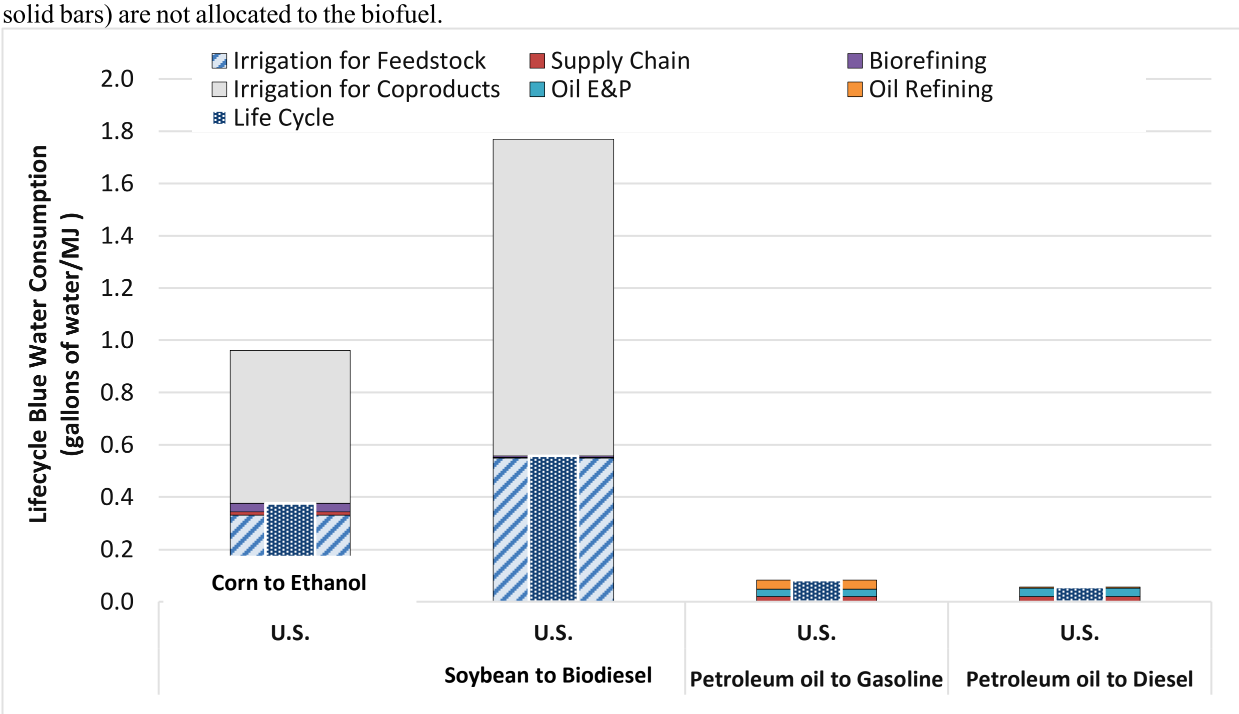
in farming and biorefinery operation are accounted. The methodology also allocated water use attributable to co-products generated from the production process for other uses. Based on a well-established LCA tool GREET (Greenhouse gases, Regulated Emissions, and Energy use in Technologies) (Argonne National Laboratory, 2019), a regional-based water footprint tool WATER (Water Analysis Tool for Energy Resources),[28] and the most recent biorefinery survey (Wu, 2019) and petroleum study (Wu et al., 2018), researchers at Argonne National Laboratory compared ethanol with gasoline, and soy biodiesel with diesel. The study relies on historical climate, land use, water footprint data for biofuels produced in major production regions in the United States. The production regions—Corn Belt, Lake, and Northern Plains[29]—together account for 85% of corn production and more than 90% of ethanol production in the United States. The study found significant variations in lifecycle blue water consumption for corn ethanol and soybean biodiesel across the United States (Figure 11.22 and Figure 11.23). While producing 1 megajoule of corn ethanol can consume as low as 0.084 gallons of blue water in Corn Belt states, which is comparable to that of gasoline (0.053 gallons), it can take as much as 1.103 gallons in Northern Plains states, with a U.S. weighted average of 0.377 gallons.[30] Lifecycle water consumption for soybean biodiesel is slightly higher, from 0.102 to 1.697 gallons per megajoule.

[28] https://water.es.anl.gov (Anl, 2023)
[29] The Corn Belt region (Region 5) includes Iowa, Indiana, Illinois, Ohio, and Missouri; the Lake region (Region 6) includes Minnesota, Wisconsin, and Michigan; and the Northern Plains Region (Region 7) includes North Dakota, South Dakota, Nebraska, and Kansas.
[30] Note that these lifecycle numbers are in units of gallons of water per megajoule of fuel, compared to the figures in Table 11.1 that show gallons of water per gallons of fuel. The allows for more direct comparison across fuels on a per energy basis.



MJ = megajoules

Figure 11.22. Lifecycle water consumption for corn ethanol and soybean biodiesel in major producing regions, and petroleum fuels. The dark blue dotted bar shows net lifecycle value. Water consumption for the co-product (gray solid bars) are not allocated to the biofuel.

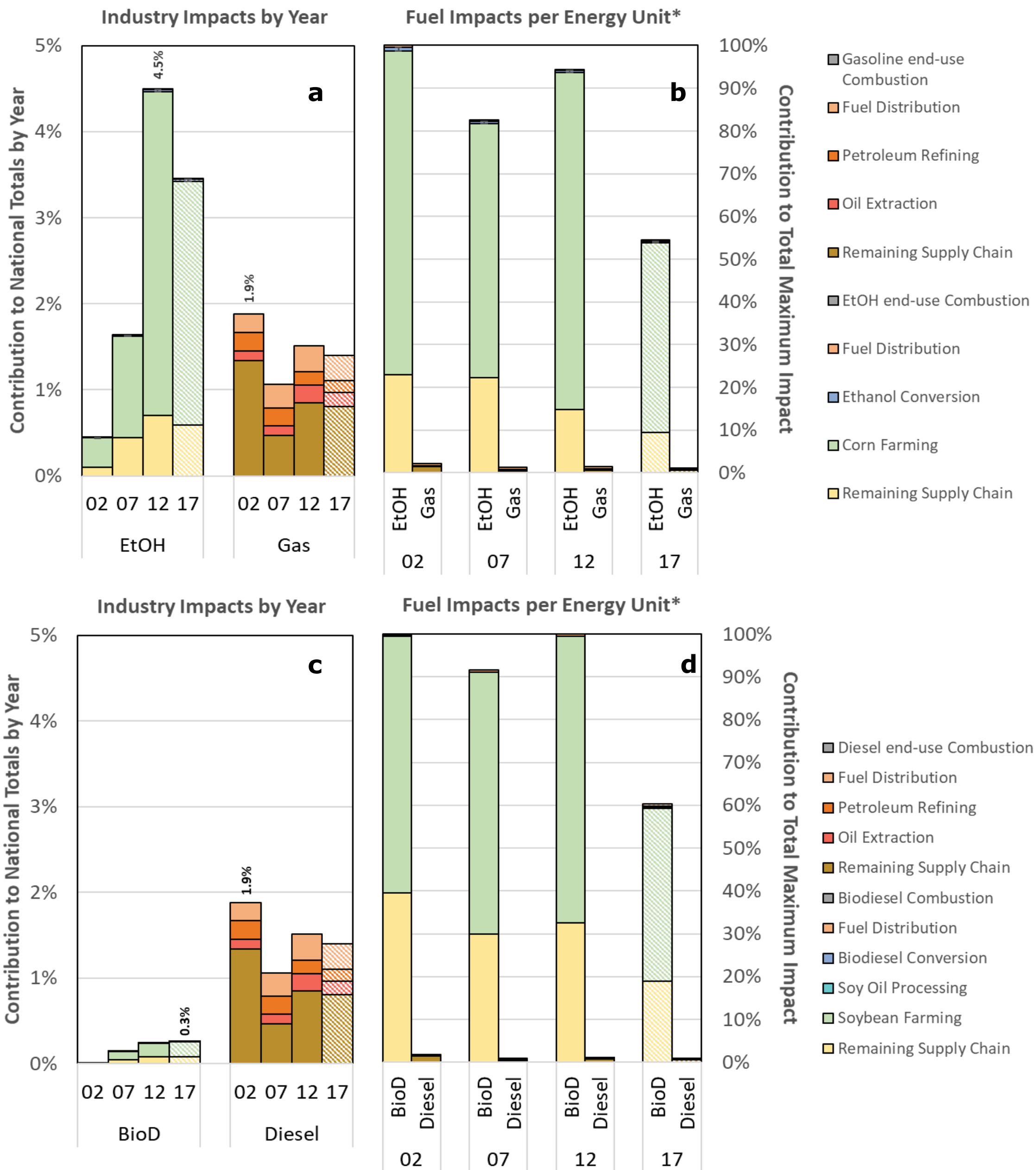


MJ = megajoules

Figure 11.23. Lifecycle water consumption for corn ethanol, soybean biodiesel, and petroleum fuels—U.S. average only. Dark blue dotted bar shows net lifecycle value. Water consumption for the co-product (gray solid bars) are not allocated to the biofuel.

Another method was developed recently to compare total freshwater withdrawals for corn ethanol with petroleum, and soy biodiesel with diesel. Developed at the National Renewable Energy Lab (NREL), this approach uses LCA combined with an environmentally-extended input-output (EEIO) analysis to estimate the effects across 15 different environmental and economic metrics (Avelino et al., 2021; Lamers et al., 2021). Details of the analysis and assumptions are provided in Appendix F. Results are presented in a single graph per biofuel and petroleum substitute, consisting of two panels each (Figure 11.24). Total industry-level freshwater withdrawals for both ethanol (Figure 11.24a) and biodiesel production (Figure 11.24c) increased in the period due to the exponential growth of both sectors, which increased at a faster pace than the rest of the U.S. economy on average across 2002–2017. In relation to gasoline, despite the smaller size of the ethanol industry in 2017, it contributed more than double to the water footprint of the nation compared with gasoline (Figure 11.24a).

While the ethanol conversion process cut water use by almost 2 gallons of water per gallon of ethanol over the period, this supply chain step represents less than 1% of total water requirements for corn ethanol. Freshwater withdrawals for both biofuels are primarily driven by irrigation and therefore directly influenced by weather and irrigation efficiencies. Throughout the period, water withdrawal intensity in both corn and soybean crops has decreased (Figure 11.24b, d), partially due to a wide adoption of sprinklers and low-flow irrigation systems instead of traditional gravity irrigation system (Dieter et al., 2018a). These improvements are illustrated by the 2002 highpoint per energy unit in both cases. Note that for comparison purposes, the year/fuel with the highest impact per metric is used as the benchmark (100%) and the impacts of the other years and fuels are then shown as a relative comparison to that benchmark. However, the effect of the 2012 drought is clearly visible for both conversion pathways as well (Figure 11.24b, d). It led to an 18% reduction in corn yields as compared to 2007, due to increased irrigation and lower corn yields per acre planted. The water withdrawal reductions for petroleum products are due to reduced indirect (external) effects and not related to extraction and refining industry improvements. Refining and oil extraction account for roughly 2 gallons of water withdrawal per gallon of gasoline equivalent or gallon of diesel equivalent respectively across all years.



*Megajoule or GGE/GDE (gallons of gas equivalent, gallons of diesel equivalent)

Figure 11.24. Total freshwater withdrawals for corn ethanol (EtOH) vs. gasoline (Gas) (a, b) and soybean biodiesel (BioD) vs. diesel (Diesel) (c, d). Total industry contribution to total national U.S. emissions (a, c) and impacts per energy unit (b, d). The left panel shows the relative contribution of the biofuel industries to the U.S. national totals for the years evaluated. These results reflect total direct and indirect impacts due to the production of the respective fuel and their related co-products across the years and their impacts from fuel combustion. The right panel shows how the impacts from producing one energy unit of fuel evolved over time by dividing the total effects from producing the fuels (not considering other co-products) from each year by the total U.S. production in the respective year. For comparison purposes, the year with the highest impact per metric is used as the benchmark (100%) and the impacts of the other years are then shown as a relative comparison to that benchmark. The impacts are broken down into supply chain steps (stacked bars), including upstream supply chain activities, corn/soybean farming, oil processing, ethanol/biodiesel conversion, fuel distribution, and fuel combustion. The 2017 results are plotted in a shaded/non-solid pattern to stress their hybrid data (2012 economic and 2017 environmental accounts) (Avelino et al., 2021; Lamers et al., 2021). (Creative Commons CC-BY-NC-ND and Creative Commons license

https://creativecommons.org/licenses/by-nc-nd/4.0/). Fuel Distribution is listed separately for ethanol and gasoline as both have a fuel distribution step.

11.6 Horizon Scanning

From a horizon scanning standpoint, one critical question is the development of cellulosic feedstock markets. In order to look at the water impacts of a much greater expansion of cellulosic feedstocks, Xu et al. (2019) used the *2016 U.S. Billion-Ton Report* (see Table 11.2) scenarios for 2008, 2017, and 2040. They developed county-level estimates of renewable water available for bioenergy feedstock production in the United States. The feedstock scenarios for the quantity of corn produced (in million tons) for 2017 and both 2040 future scenarios are relatively similar. Where the differences emerge are with the yields and therefore associated acreage, as well as the assumptions regarding crop residues (which are significant in 2017, which is a modeled year, not observed data) as well as herbaceous and woody bioenergy crops. Soybean amounts are also roughly similar and vary little from the 2008 production levels.

The methodology for this work used the *BTS16* scenarios and linked to the WATER model described in Chiu and Wu (2012). The authors examined six different blue water footprint (BWF) estimation methods, which showed the importance of taking preseason soil moisture carryover into account to avoid overestimating irrigation water consumption. The results for the total blue water footprint are shown in Figure 11.25, and show that the majority of the water footprint is located in the Northern Plains (including Kansas, Nebraska, North and South Dakota), where there is also wide variability in the range of outcomes across the scenarios. For the Northern Plains, the water use between 2008 and 2017 (a modeled year) increased substantially. The 2040 water footprints are smaller than 2017, but still represent an increase over the 2008 levels for the Northern Plains.

Table 11.2. Feedstock production in historical (2008) and proposed future production scenarios for 2017 and 2040, based off the 2016 Billion-Ton (BT16) report. Source: Xu et al. (2019).

	Corn		Soybean		Crop residues	Herbaceous	Woody crops
Scenario	Million tons	Million acres	Million tons	Million acres	Million tons	Million tons	Million tons
2008	92.0	23.5	9.6	8.9	N/A	N/A	N/A
BC1 2017	129.5	29.8	11.8	9.2	104	N/A	N/A
BC1 2040	132.1	24.9	11.7	7.8	176	340	71
HH3 2040	131.1	19.3	10.5	7.1	200	594	142

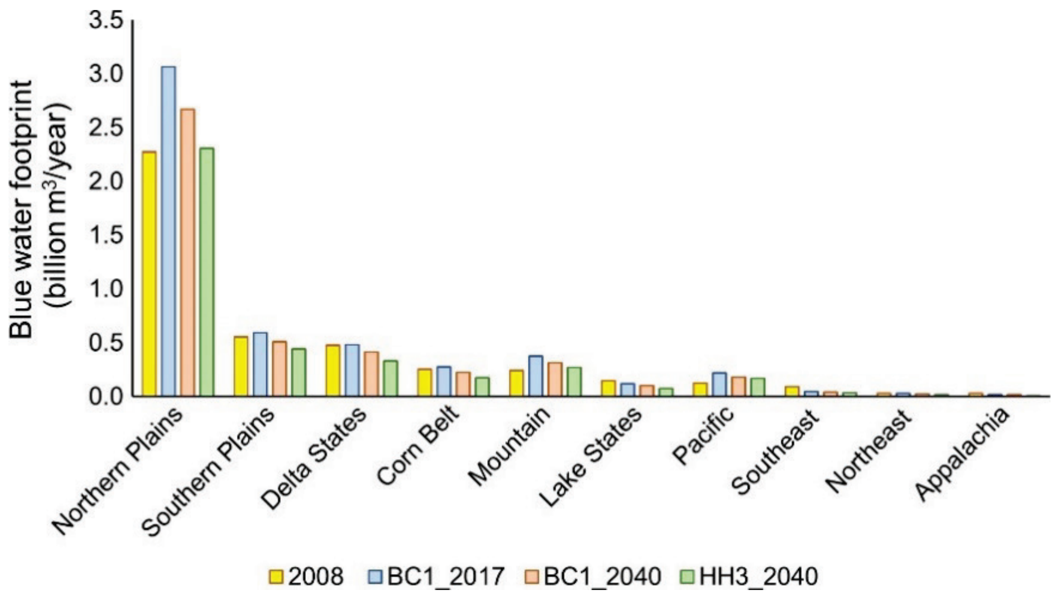


Figure 11.25. Comparison of feedstock blue water footprint (billion cubic meters [m^3]/year) under historical (2008) and proposed future production scenarios. Source: Xu et al. (2019). (Creative Commons license https://creativecommons.org/licenses/by-nc-nd/3.0/legalcode)

Xu et al. (2019) also advanced the methodology and understanding of the impacts on water availability, by developing two indexes: stream availability index and percolation flow availability index. These indices "measure how irrigated bioenergy feedstock production may reduce renewable surface water and groundwater availability." The largest impacts of these scenarios were on renewable groundwater availability. The resolution is at the county level, and as highlighted by the authors, "for both the 2008 and BT16 scenarios, there are about 88 to 99 counties, depending on the scenario, where groundwater irrigation demand exceeds annual percolation flow." Most of these counties are located in western Kansas, southern Nebraska, eastern Colorado, and northern Texas. In these cases, where percolation flow alone is not sufficient, demands are met by stored groundwater in aquifers.

Beyond cellulosic feedstocks, from a water use standpoint, there is a considerable literature on algae-based biodiesel. While algal-based fuels can be grown on areas that are ill suited to agriculture, algae require significant amounts of water for production, processing, and extraction of the fuels and co-products (Brennan and Owende, 2010). An earlier assessment of freshwater needs to produce algae-based fuel estimated 1000 gallons/gallon bio-oil (Wigmosta et al., 2011). However, as shown by Chiu and Wu (2013), freshwater can be replaced by municipal wastewater under the same land use.

Another critical factor affecting irrigation trends into the future is climate change. Climate change over the quarter century since 1994 has certainly influenced the amount of irrigation water applied and will continue to influence irrigation due to changes in air temperatures, precipitation, heatwave duration, and length of the growing season (Evett et al., 2020a; Kukal and Irmak, 2019). As was shown in Chapter 6, periods of drought coincided with the larger drops in the growth of ethanol production (see Figure 6.1) over the period from 1982 to 2019. The 2018 *National Climate Assessment* discussed implications for irrigation.

> "Expanded irrigation is often proposed as a strategy to deal with increasing crop water demand due to higher trending temperatures coupled with decreasing growing-season precipitation. However, under long-term climate change, irrigated acreage is expected to decrease, due to a combination of declining water resources and a diminishing relative profitability of irrigated production. Continuing or expanding existing levels of irrigation will be limited by the availability of water in many areas. Surface water supplies are particularly vulnerable to shifts in precipitation and demand from nonagricultural sectors. Groundwater supplies are also in decline across major irrigated regions of the United States." (Hayhoe et al., 2018).

The impact of climate change on irrigation was also discussed in Evett et al. (2020a), where they noted that "depending on crop and latitude, irrigation water requirements will either increase or remain relatively static, but in large areas irrigation water requirements are expected to increase." These considerations are key to understanding where future production may occur. In the Southern High Plains (SHP), precipitation is expected to decrease while temperatures increase. The resulting increased crop water requirement will mean additional irrigation to grow crops at current levels; and the decline of the High Plains Aquifer is expected to hasten. Thus, irrigated area in the SHP is expected to decrease over time. However, climate change is expected to increase precipitation in the more northern Great Plains states of Nebraska, North and South Dakota, Wyoming, and Montana, which include the western corn belt. Present water supplies and suitable land in the MonDak region are considered to allow for up to 500,000 more acres of irrigation to be developed (USDA, 2020); with increased precipitation, the projected MonDak irrigated area could further expand. In the eastern corn belt, irrigated area is already expanding due to short-term summer droughts (flash droughts) that are increasing in frequency and severity with climate change even as future precipitation totals are projected to increase in that region. Figure 11.26 illustrates that the percentage of total U.S. irrigated area that is in the eastern states is now more than 31% and increasing.

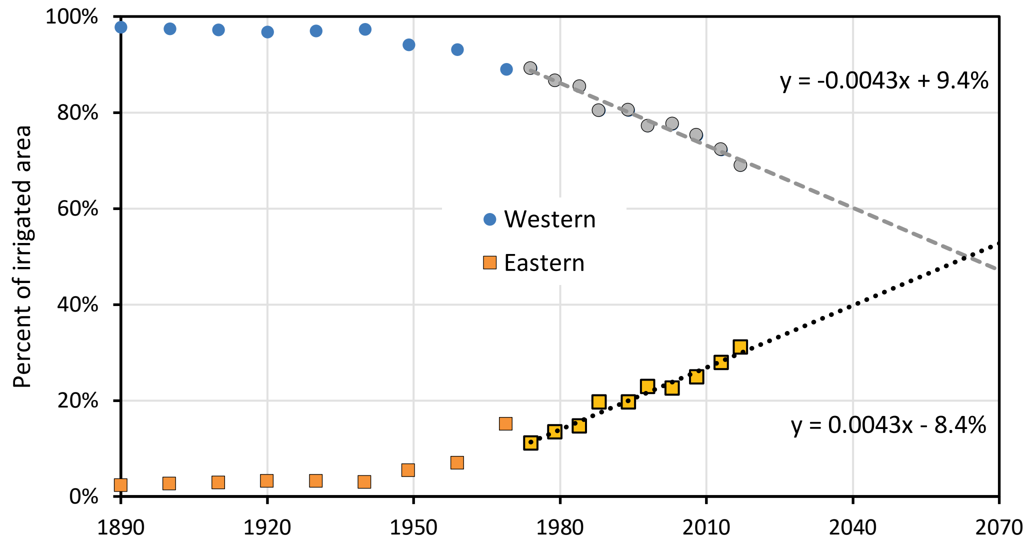


Figure 11.26. Decline of irrigated area as a percentage of total U.S. irrigated area in the 17 western states compared with increase in percentage of total U.S. irrigated area occurring in the eastern U.S. Gray circles are those data for the Western states that were used to fit the regression line showing the rate of decrease of percentage of total U.S. irrigated land area that was in those states. Yellow squares are those data for the Eastern states that were used to fit the regression line showing the rate of increase of percentage of the total irrigated area that was in those states. Prior to 1978, data were primarily on a 10-year basis. After 1978, data were mostly on a 5-year basis. Data from (USDA, 2014, 2010, 2004, 1998, 1994; USDA-NASS, 2019; U.S. Department of Commerce, 1990, 1986, 1982, 1973, 1965, 1941a, b).

11.7 Synthesis

11.7.1 Chapter Conclusions

Types of Water Use

- Water use and water availability for biofuels are primarily due to irrigation during the feedstock production stage, while water use in biorefineries (the conversion stage) represents a small and declining percentage (approximately 1–9%) of lifecycle water use. Water use in other parts of the biofuel supply chain is minimal.
- For corn-based ethanol, when accounting for ground and surface water (“blue water”) used for irrigation, 88% of total lifecycle biofuel water use is for irrigation for feedstock production. For soybean-based biodiesel, feedstock irrigation is 98% of total lifecycle biofuel water use.
- However, even feedstocks that do not require irrigation, including corn and other feedstock crops in rainfed production areas, may still have “green water” use (rainwater) due to high evapotranspiration, and therefore can affect stream flows by reducing the amount of precipitation that is available for other pathways such as infiltration, runoff, and deep percolation to groundwater.

Irrigation Trends and Changes

- The overall irrigated area of corn, according to USDA survey data, increased from between 9.3 and 9.7 million acres (between 1992 and 2002) before the 2005 Energy Act to between 12 and 13 million acres reported in the 2008 and 2013 irrigation surveys, before declining to 11.6 million reported in the 2018 survey (representing 14% of total corn areas in 2018). This includes corn grown for all purposes, not just biofuels.
- The change in irrigated corn acreage was smaller than the change in unirrigated corn acreage on an absolute basis, but still represented a roughly 30% increase in irrigated acres, while the total volume of irrigation water applied has been decreasing in recent years. However, the variation cannot be easily attributed to biofuel production or the RFS Program.
- Yields have changed as well, with irrigated corn yields steadily increasing since 1992, which can be attributed to improvements in genetics, fertilization, and uniformity of irrigation. Unirrigated corn yields are typically lower and more variable due to their greater dependency on weather.
- Depth of water applied varies according to weather, irrigation application methods, management, and other factors. Notably, droughts that occurred between 2010 and 2014 caused an increase in depth of water applied.
- Other than weather, changes (decreases) in depth of water applied were mostly tied to a long-term shift in irrigation application methods from gravity flow to more efficient pressurized systems that occurred starting in the 1990s.
- The majority of total irrigation withdrawals (81%) and irrigated lands (74%) in 2015 occurred in the 17 conterminous western states located west of and including the Dakotas, Nebraska, Kansas, Oklahoma, and Texas overlying the HPA. Satellite-based studies, which have developed annual maps with greater spatial and temporal detail to track irrigation changes over the HPA, show that irrigated croplands (all crops, all uses) over the HPA increased from approximately 14 million acres to 15 million acres (for all crops/uses) between 2000 and 2017, an annual average rate of expansion of 186,000 acres/year, most notably in Nebraska with increases of 170,000 acres/year.

Groundwater Supply and Aquifers

- Continued irrigation at present rates over the Southern HPA is not sustainable, where the extraction rate greatly exceeds recharge, most notably in eastern Colorado, western Kansas, the Texas Panhandle, and eastern New Mexico. However, for the Northern HPA, climate change is expected to increase precipitation, and projections show that the MonDak irrigated area could expand, while irrigation at present rates is considered sustainable in much of eastern Nebraska.

- In regions where water supply is available, irrigated acreage is expected to increase because it greatly improves the overall productivity from approximately 50% of potential yield to 80% of potential yield, and thus greatly increases the yield per unit of water used – the crop water productivity.

Biorefinery Water Use

- Though a small fraction of biofuel lifecycle water use, the water intensity of ethanol production in biorefineries decreased by 12% between 2011 and 2017 and by 54% in the 19 years between 1998 and 2017.
- These reductions have resulted from the adoption of energy-efficient and water-efficient technologies, reuse, and recycle, increased system integration in retrofitting existing plants, as well as diversification of water sources.

Comparison With Petroleum

- Producing a gallon of corn ethanol (including total irrigation and refinery water) requires 8.7–160 gallons water per gallon ethanol (average 76 gal/gal), compared to petroleum-based gasoline which ranges from 1.4-8.6 gal/gal gasoline (average 5.7 gal/gal). The major factors determining the range are the corn-producing regions and associated variation in irrigation requirements.
- Another approach combines the GREET model with WATER, and finds that on a per megajoule (MJ) basis, corn ethanol requires 0.084–1.103 gallons of blue water (Corn Belt and Northern Plains states, respectively) with a U.S. weighted average of 0.377 gallons per megajoule. In comparison, gasoline averages 0.053 gallons per megajoule, Lifecycle water consumption for soybean biodiesel is slightly higher, from 0.102 to 1.697 gallons per megajoule, compared with 0.0–0.057 for diesel.
- Using an approach combining LCA with an environmentally-extended input-output (EEIO) analysis, researchers also account for industry size. They found that in relation to gasoline, despite the smaller size of the ethanol industry in 2017, it still contributed more than double to the water footprint of the nation than the former.

11.7.2 Conclusions Compared with the RtC2

As noted in the 2018 RtC2s, the irrigation of corn and soybeans grown for biofuels is the predominant water quantity impact, and is significantly greater than the biofuel conversion process. This has now been better quantified and remains consistent with RtC2. The RtC2 noted some increases in irrigated areas for corn between 2007 and 2012 and elevated rates of land use change to corn production in more arid Western states including the HPA region. USDA survey data shows that was the case, but 2018 data shows that the total area of irrigated corn has been relatively stable since 2012, with declines in the southern HPA region. New satellite-based studies are expanding the understanding of annual changes with greater spatial resolution and show growth in irrigated acres over the HPA, with most growth in Nebraska. Attribution to biofuels and the RFS Program was not a focus of the RtC2, and this continues to be a challenge especially as it pertains to irrigated acreages. There are also a range of additional factors that affect both the extent and location of irrigated areas as well as depth of irrigation. These factors add a layer of complexity to the attribution of water availability to biofuels and the RFS Program. Irrigation practices are dependent on a number of economic and agronomic factors that drive land management practices making attribution of increased irrigation and water quantity to biofuels difficult. Looking ahead, climate change has affected and will continue to affect irrigation trends. Recent literature and this report have refined the understanding of the adverse water availability impacts that will most likely arise in already stressed aquifers and surface watersheds. In particular it highlights that the Southern HPA is the area of highest concern due to continued rates of withdrawal exceeding recharge. The Northern HPA has greater variability in its level of depletion.

11.7.3 Uncertainties and Limitations

- While there is growing information and data to better understand historic irrigation trends and patterns there are still gaps in data: end uses of irrigated crops, relative impact of factors driving changes in irrigation, spatial and temporal changes in irrigation, impacts on aquifers, and attribution of specific areas of change to the RFS Program.
- Neither the USDA-NASS nor the USGS data distinguish between end uses of crops spatially, so there are no data in USDA or USGS reports to substantiate how much irrigated corn or soybean were used specifically for biofuel production.
- The question continues to be to what extent changes in irrigated acres and irrigation depth for corn and soybean crops can be attributed to biofuels and the RFS Program. A number of factors affect the conversion of acres to corn or soybean, and additional factors drive irrigation. Prices and weather are important factors, but irrigation practices are also determined by the availability of new technology and the varying legal constraints for water rights and allocation in different regions.
- A major uncertainty for understanding impacts to date and future impacts is the role of climate change. Depending on the crop and latitude, irrigation water requirements will either

increase, decrease, or remain relatively static, but in large areas of the southern HPA, irrigation water requirements are expected to increase.

- Satellite data are advancing the understanding of temporal and spatial changes in irrigation, and next steps would be understanding whether these irrigation changes can be associated with crops used for biofuel production.

11.7.4 Research Recommendations

- Continued development of satellite-based maps with higher temporal and spatial resolution can provide additional insights into changes in irrigated crops, irrigated area, and irrigation needs (evapotranspiration), including mapping of changes in irrigation of local feedstocks around ethanol production facilities. Future research on land cover and land management change estimates (see Chapter 6) can be used to estimate any fractional effect of the RFS Program on water availability.
- Satellite-based systems for determining evapotranspiration at 100-foot resolution have been demonstrated over large parts of the United States and are poised to become widely available for greater use in irrigation management, irrigation and underground water conservation district management, and to develop better understanding of multi-scale crop water use for future analyses and assessments.
- Conducting a nation-wide biorefinery survey every 2–3 years can capture the changes in water use and management and the advancement of technologies. These surveys could be expanded to also characterize biodiesel facilities, as well as assess a wider range of impacts relevant to not only water use and management, but also water effluents, air emissions, and other waste streams.

11.8 References

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12. Terrestrial Ecosystem Health and Biodiversity

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Key Findings

- Impacts to date from biofuels on domestic terrestrial biodiversity, as an indicator of ecosystem health, are primarily due to corn and soybean feedstock production for ethanol and soy biodiesel. Shifts in perennial plant cover to corn and soybeans, and corn and soybean production practices are the two main drivers of effects.
- Of land in perennial cover shifting to annual crops, the vast majority was from grasslands, ranging from relatively unmanaged to highly managed grasslands (e.g., hay, pasture). The loss of grassland cover to annual crops, such as corn and soybeans, negatively impacts terrestrial biodiversity, including grassland species of birds, bats, pollinators and other beneficial organisms (e.g., insects that provide pest control), and plants.
- Between 2008 and 2016, shifts from land in perennial cover to corn and soybeans due to all causes, including potentially biofuels, occurred in areas adjacent to or within critical habitat of 27 terrestrial threatened and endangered (T&E) species across the contiguous United States, according to an analysis processing data from the USDA Cropland Data Layer (CDL). The CDL (whether processed or unprocessed) is relatively accurate at large spatial scales (e.g., states) but can be more uncertain at local scales. Thus, it may require verification with imagery or direct visitation to confirm these results.
- Beyond change in land cover, crop production practices for corn and soybeans can also negatively affect terrestrial biodiversity, particularly through pesticides.
- The range of possible impacts from the RFS Program likely spanned from no effect to a negative effect on terrestrial biodiversity historically (2008 to 2016). Further refinement of the acreage estimates attributable to the RFS Program are needed to reduce this range of possibilities. These findings do not necessarily apply for years beyond 2016, when the effects of the RFS Program on corn ethanol and soy biodiesel production may have changed.
- Further evaluation would be needed to quantify the magnitude of any historical impacts of the RFS Program on biodiversity. Any effects may be relatively small compared to those of total U.S. cropland, but may be more important regionally or locally. Finally, whether T&E species were impacted by the RFS Program during this period (2008 to 2016) is also possible, but unknown, and requires further evaluation.
- Conservation practices can reduce negative impacts to terrestrial biodiversity. These practices include protecting environmentally sensitive lands, increasing habitat heterogeneity, and decreasing the use of pesticides.

- The likely future effects of the RFS Program are highly uncertain. While the projected cropland expansion for 2023–2025 is slightly larger than the estimated historical cropland expansion from the RFS Program, EPA cannot say with reasonable certainty that any particular terrestrial ecosystem or biodiversity will be impacted due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes.[1]

Chapter Terms: biodiversity, Conservation Reserve Program (CRP), Endangered Species Act (ESA), grassland, landscape simplification, Threatened and Endangered (T&E) species

12.1 Overview

12.1.1 Background

The Energy Independence and Security Act (EISA) requires EPA to assess the effects of the Renewable Fuel Standard (RFS) Program on "ecosystem health and biodiversity, including impacts on forests, grasslands, and wetlands." Here, ecosystem health is the condition of ecological systems, including their physical, chemical, and biological characteristics, and the processes and interactions connecting them.[2] Because the physical and chemical components of ecosystems are addressed elsewhere in this third report to Congress (RtC3, see section 12.1.3), this chapter addresses biological characteristics and processes.

The focus of this chapter is principally on terrestrial biodiversity since it both serves as an indicator of ecosystem health and is specifically mentioned in EISA. Biodiversity is the variety and variability among living organisms and the ecological complexes in which they occur (Heywood, 1995). Biodiversity can be measured in different organizational units, including genes, individuals, species, habitat types, and up to whole ecosystems. This chapter addresses biodiversity by discussing both changes in the number of species within individual taxonomic groups (e.g., birds, insects) and abundance within species or types of species.

Among ecosystem types listed in EISA (forests, grasslands, and wetlands), the focus here is on impacts to grasslands and grassland species in this chapter. As addressed in Chapter 5, land cover and land management (LCLM) change from biofuel feedstock crops has impacted greater areas of grasslands than any other terrestrial land cover type. Of land in perennial cover shifting to annual crops between

[1] On August 3, 2023, EPA completed its Endangered Species Act informal consultation on the Renewable Fuel Standard (RFS) Program: Standards for 2023-2025 and Other Changes rulemaking (also known as the RFS "Set Rule"). With the Biological Evaluation that EPA submitted to the National Marine Fisheries Service (NMFS) and Fish and Wildlife Service (FWS) on May 19, 2023, EPA determined that the RFS Set Rule is not likely to adversely affect listed species and their designated critical habitats. EPA received letters of concurrence with this determination from NMFS on July 27, 2023, and from FWS on August 3, 2023, thereby concluding the consultation.
[2] For more discussion of ecological conditions, see https://www.epa.gov/report-environment/ecological-condition.

2008 and 2016, 88% was estimated from grasslands, while 3% and 2% were estimated from wetlands and forests, respectively, according to estimates from Lark et al. (2020). Wetlands are addressed in Chapter 14. Under EISA, annual crops are not eligible as renewable biomass if produced on forested land cleared after 2007; however, the potential for using woody feedstocks directly from nonfederal, managed forests is addressed in a horizon scanning section later in this chapter (section 12.6).[3] Other ecosystem types, such as deserts or alpine areas, are not addressed because they are not used to grow feedstocks for biofuels.

Lastly, this chapter focuses on the direct domestic effects of feedstock production, predominantly on the biodiversity effects of the dominant biofuel feedstocks produced to date (i.e., corn grain for ethanol and soybeans for biodiesel; see Chapter 2 for scope). Overall, using the scientific literature, the effects of producing corn and soybeans in general could not be distinguished from any potential effects attributable to the RFS Program. As a result, this topic is addressed in a separate RFS attribution section (section 12.3.3). Beyond corn and soybeans, fats, oils, and grease (FOG) biofuels are a byproduct of other activities and thus generally do not affect terrestrial habitats independently of the main product (e.g., beef). Potential effects from Brazilian sugarcane on terrestrial ecosystems are addressed in Chapter 16, as are any other potential international effects. The potential impacts of other, minor feedstocks are addressed later in this chapter in the horizon scanning subsection focused on possible future issues (section 12.6).

12.1.2 Drivers of Change

Most potential biofuel-related impacts to terrestrial ecosystem health and biodiversity occur in the feedstock production stage. This is because feedstock production affects LCLM, which in turn directly impacts biodiversity. There are also effects on terrestrial ecosystems from the biofuel conversion and end-use stage, as emissions from those practices can travel downwind and deposit on ecosystems, contributing to nutrient deposition and shifting the competitive balance between species, for example. These emissions, however, are controlled via air quality standards that apply to emissions sources, and likely have a small effect in comparison with the direct effects of land use change at the feedstock production stage.

Within the feedstock production stage, the major drivers of impacts are land cover changes to corn and soybeans, and land management practices. As noted above, most land converted to annual crops,

[3]Under EISA and the RFS, renewable biomass may include slash and precommercial thinnings from nonfederal forestlands, and planted trees and tree residue from actively managed tree plantations on nonfederal land. Biomass from forests on federal lands is eligible only if it is harvested from the immediate vicinity of buildings and other areas regularly occupied by people, or of public infrastructure, at risk of wildfire (CAA Amendments, 1990). CAA section 211(o)(1)(I)(v)

such as corn and soybeans, has been from grasslands. This shift from grasslands to crops reduces total habitat for many species, while also reducing overall landscape complexity, impacting biodiversity and biodiversity-mediated ecosystem services (Landis, 2017; Meehan et al., 2011). Notably, the term grassland in this chapter is used broadly to include a spectrum of grasslands from relatively unmanaged to heavily managed, including Conservation Reserve Program (CRP)[4] land in perennial grasses and pasture. Hence, the definition used in this chapter is based on cover type, not use. Additionally, the effects of land management practices, including production and conservation practices, on agricultural lands also drive biological effects. For instance, fertilizer and pesticide usage can negatively impact ecosystem health and biodiversity; whereas, conservation measures, such as maintaining pollinator habitat in the margins of fields, can promote positive outcomes. The effects of land cover and land management drivers are discussed in section 12.3.

12.1.3 Relationship with Other Chapters

This chapter addresses terrestrial health and biodiversity, a subject interwoven with those of many other chapters. The second triennial report to Congress (i.e., the RtC2) addressed both aquatic and terrestrial ecosystems together in one chapter, which also included wetlands. The bulleted conclusions from that chapter are listed in section 12.2 and reflect its combined focus. In this RtC3, there are three separate chapters on ecosystem health and biodiversity: terrestrial ecosystems (this chapter), aquatic ecosystems (Chapter 13), and wetlands (Chapter 14). These systems are interconnected, and changes in one often produce effects in the others. However, having three separate chapters helps address the complexity of the effects on each ecosystem type more completely. Amphibians are addressed in the wetlands chapter and not here, even though they often utilize terrestrial habitats. Similarly, waterfowl and migratory waterbirds are addressed in the wetlands chapter. Finally, as noted above, physical and chemical characteristics and processes of ecosystems are addressed elsewhere in this RtC3, primarily in the air, soil, and water quality chapters (Chapters 8, 9, and 10, respectively).

12.1.4 Roadmap for the Chapter

Overall, this chapter proceeds in the following manner: section 12.2 provides the ecosystem health and biodiversity conclusions from the RtC2; section 12.3 reviews the literature on the impacts to date on specific groups of terrestrial organisms (i.e., birds, bats, pollinators and other beneficial insects,

[4]The Conservation Reserve Program (CRP) is a program administered by the USDA Farm Service Agency. In exchange for a yearly rental payment, farmers enrolled in the program remove environmentally sensitive land from agricultural production and plant species to improve environmental health and quality (USDA, 2020b). It is a time-limited program (often a 10- or 15-year contract length); after the contract has expired the land owner is no longer compensated for continued maintenance of the land cover and so the expired CRP acreage often reverts back to agricultural production.

plants, and threatened and endangered [T&E] species[5]); section 12.4 discusses likely future effects; section 12.5 compares the effects of biofuels to petroleum; section 12.6 considers other biofuel feedstocks in a horizon scanning exercise; and lastly section 12.7 provides a synthesis, with chapter conclusions, uncertainties, and next steps for research.

12.2 Conclusions from the Second Triennial Report to Congress

The RtC2 made major bulleted conclusions at the end of the combined ecosystem health and biodiversity chapter. Notably, these conclusions were not specific to the RFS Program, but rather on the impacts of agriculture and biofuels broadly. They are as follows:

- Loss of grasslands and wetlands is occurring in ecologically sensitive areas, including the Prairie Pothole Region.
- Loss of habitat and landscape simplification are associated with negative impacts to pollinators, birds, soil-dwelling organisms, and other ecosystem services in both terrestrial and aquatic habitats.
- Increased fertilizer applications of nitrogen and phosphorus can have negative effects on aquatic biodiversity.

12.3 Impacts to Date for the Primary Biofuels

12.3.1 Literature Review

This section updates and reviews the scientific literature on the effects of biofuel feedstock production on terrestrial ecosystems by taxonomic category. Each of these categories is addressed by the effects of land cover change and land management practices (the broader trends on conversion of grasslands—and other habitat types—are described in Chapter 5). The scientific literature was often not specific to the effects of corn and soybeans grown for biofuels, and instead addressed the general effects of agriculture and corn and soybeans. The summary below reflects the assumption that land management for corn and soybeans and their effects are generally the same regardless of end use, whether for food, feed, or biofuel feedstock. Furthermore, the studies focus on different time periods for different purposes, yet can still generally address how LCLM change, and the production of corn and soybeans specifically, affect terrestrial biodiversity.

[5]Threatened and endangered (T&E) species are classified by the U.S. Fish and Wildlife Service under the 1973 Endangered Species Act. "Threatened" means a species is likely to become endangered within the foreseeable future. “Endangered” are organisms in danger of extinction throughout all or a significant portion of its range (https://www.fws.gov/endangered/laws-policies/).

Finally, this section discusses the effects of pesticides, as one type of land management practice, using the scientific literature. More information on the toxicity of pesticides and other chemicals by organism type can be derived from the EPA ECOTOX Knowledge Database. For example, ECOTOX includes 81, 326, 48, 28, and 1488 records for the herbicides glyphosate, atrazine, acetochlor, metolachlor, and 2,4-D, respectively, for birds, along with 366, 227, and 17 records for the neonicotinoid insecticides imidacloprid, clothianidin, and thiamethoxam, respectively. It is beyond the scope of the RtC3 to summarize the testing results contained in ECOTOX, but interested readers are encouraged to consult the database and/or the original citations (see Supplemental Table 12.1 for more information).

12.3.1.1 Birds

Shifts in LCLM from grasses to annual crops, such as corn and soybean, can affect bird populations, depending upon the groups and species of birds examined. Based on the North American Breeding Bird Survey (BBS), nearly three-quarters of species of grassland- or farmland-dependent birds declined between 1960 and 2013 (Stanton et al., 2018). The largest decreases in avian taxa were aerial insectivores (-39.5%), followed by grassland (-20.9) and shrubland (-16.5) species (Stanton et al., 2018). By contrast, some waterbirds have shown increases in response to increased crop acreage (see Chapter 14 on wetlands). Another recent study based on the BBS did not detect greater overall avian declines in crop-intensive areas, yet species varied in their response (Belden et al., 2018). More grassland-dependent species exhibited a clearer pattern of decline with increasing cropland.

Both direct and indirect threats from agricultural intensification have contributed to some of these declines (Figure 12.1) (With et al., 2008; Haig et al., 2005). Hill et al. (2014) found that grassland habitat loss was the main cause of declines in grassland birds, followed by pesticide use. Grassland loss to agriculture not only reduces total habitat, but also typically decreases overall landscape complexity. A negative effect of grassland loss on birds is evident when comparing bird populations near annual row crops with CRP grasslands. Studies have shown that annual row crops tend to support lower densities of birds than CRP grasslands. According to Best et al. (1997), row crops (corn, soy, and sorghum) hosted fewer (7.4%) nests than CRP grasslands, but nest survival was close to that in CRP. Avian diversity was also lower: row crops hosted one-third the number of nesting species found in CRP (Best et al., 1997). Habitat arrangement has also been shown to have an effect. For instance, clustered grasslands supported bird communities better than isolated fields within agricultural landscapes in modeling scenarios (Blank et al., 2016).

Beyond habitat loss, pesticide usage has also been implicated in avian population declines. In a systematic review of 122 studies of avian abundances between 1963 to 2003, pesticides (42% of studies) and habitat alteration (27%) were the most frequently cited causes of avian declines, with pesticides and

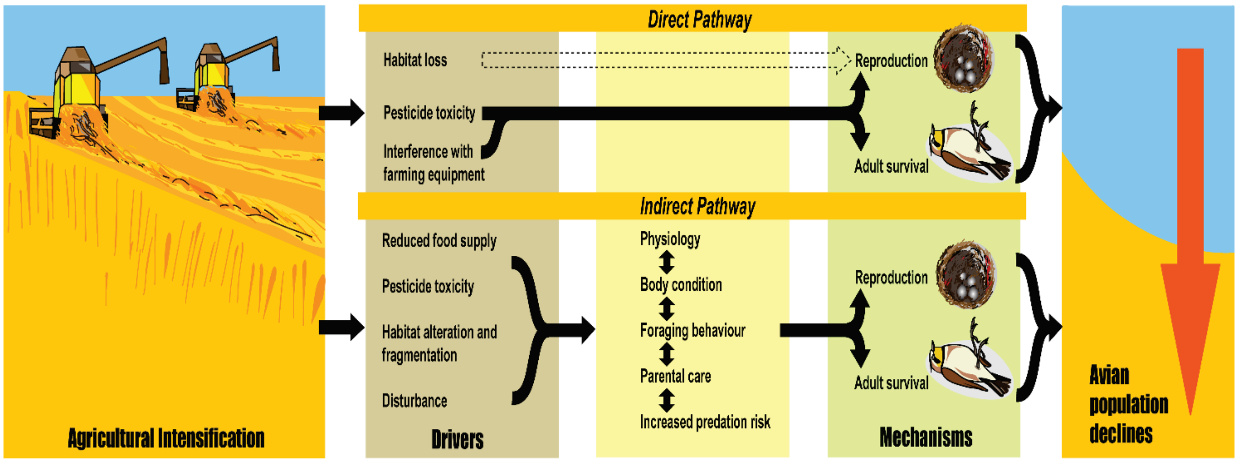

Figure 12.1. Potential direct and indirect effect pathways of agricultural intensification on avian population declines. Reproduction includes mortality or impairment at egg and nestling/juvenile stages, whereas survival represents adult survival. Although shown separately, direct and indirect effects on reproduction and survival could act simultaneously. Source: Stanton et al. (2018) (used with permission).

harvesting/mowing consistently identified as causes of negative effects from agriculture (Figure 12.1) (Stanton et al., 2018). Evaluation of declines in grassland bird abundances across the United States prior to 2003 pointed to acute toxicity of insecticides (Mineau and Whiteside, 2013).

Relative to studies on the effects of agriculture in general, biofuel feedstock-specific studies are much less common. Meehan et al. (2010) used a regression model to estimate that expanded annual bioenergy crop production (e.g., corn and soy) in the Upper Midwest would reduce avian species richness.[6] The authors simulated changes in bird species richness associated with planting marginal lands (representing 20% of the area) either with high-input, low-diversity annual biofuel crops (corn, soy) or with low-input, high-diversity crops (such as hay, alfalfa) (Meehan et al., 2010). The quadratic model estimated that expanded production of corn and soy in the Upper Midwest would reduce avian richness over a wide range of landscape configurations (when crops represented only a small amount of the landscape, corn and soy had a positive effect). However, other studies differentiate between the effects of corn and soy. For example, a field study of spring migrants in the Northern Great Plains found higher densities of granivorous birds feeding in harvested corn and sunflower fields (with post-harvest vertical structure) than in small-grain and soybean fields (Galle et al., 2009).

One study attempted to directly relate ethanol production to bird diversity. Based on the BBS data for 2006 to 2012, Evans and Potts (2015) concluded that total cropland acreage had low elasticity to the price of ethanol and that avian responses to modest changes in land use were heterogeneous across the 22 avian grassland species included. The changes in simulated abundances attributed to corn ethanol expansion (−0.17 to 0.15%) were small compared to the overall trends in species population sizes.

[6]Richness is the number of species or other biological organization units in a particular unit of area.

Species elasticities were sensitive to model assumptions. However, four species had significant responses regardless of model; two species with significant negative effects were avian grassland species of conservation concern, bobolink (*Dolichonyx oryzivorus*) and sedge wren (*Cistothorus platensis*), whereas positive effects were significant for horned lark (*Eremophila alpestris*) and sharp-tailed grouse (*Tympanuchus phasianellus*). The approach used by Evans and Potts (2015) could potentially be used to quantitatively explore the effects of ethanol prices on other species as well.

12.3.1.2 Bats

In addition to birds, shifts in LCLM from grasslands to annual crops can impact bats. Insectivorous bats are important consumers of crop pests in agricultural ecosystems; their prey species include many destructive pests like corn earworm (*Helicoverpa zea*), cabbage looper (*Trichoplusia ni*), fall armyworm (*Spodoptera frugiperda*), and tobacco budworm (*Heliothis virescens*), among others (Krauel et al., 2018; Maine and Boyles, 2015). It has been estimated that bats provide around $22.9 billion per year (or about $74 per acre of cropland) in ecosystem services through reduction in insect damage of crops and reduction of pesticide use in the continental United States (Boyles et al., 2011). Because bats are generalist insect predators (McCracken et al., 2012), they may also help to control the development of *Bacillus thuringiensis* (Bt)-resistant insect pests (Federico et al., 2008). In addition, bat control of corn earworm can also significantly decrease the spread of two crop pathogens, *Aspergillus flavus* and *Fusarium graminearum* (Maine and Boyles, 2015), suggesting ecosystem services provided by insectivorous bats may be more important than previously assumed. Though bats provide these important ecosystem services, they are generalist insectivores and so do not rely solely on agricultural pests for food sources.

Land use conversion to agriculture can potentially impact bats through loss of suitable roost sites or loss or degradation of foraging areas. Bats generally prefer heterogeneous habitats over simplified agricultural fields, requiring access to roosting sites, foraging habitat, and fresh water. Roosting requirements (whether caves, trees, old buildings, etc.) depend on the bat species, and roosting needs change for many species throughout the year. Foraging preferences also vary by species, with some bats preferring open habitat, while others prefer woodlands or edge habitats. Heterogeneous habitats are richer in both roosting and foraging sites. Several studies have found bats avoid intensive agricultural habitats, instead favoring native woodland or remnants of seminatural habitat within agricultural landscapes (Fuentes-Montemayor et al., 2013; Womack et al., 2013; Henderson and Broders, 2008). Thus, significant decreases in bat species richness and activity have been observed with increasing agriculture (Put et al., 2019; Monck-Whipp et al., 2018; Put et al., 2018). The critical habitat of one endangered bat species (Indiana bat, *Myotis sodalis*) and one threatened bat species (Northern long-eared bat, *Myotis* *septentrionalis*) coincide with regions of high corn and soybean production; and grassland loss to corn and soybeans may have occurred within the critical habitat of the Indiana bat (for more details, see section 12.3.2 and Supplemental Table 12.1).

Pesticide application associated with increased corn and soybean production may also adversely impact bats. Bats' high metabolic rate and insectivorous diet increase their likelihood of exposure to bioaccumulating chemicals in the environment. Additionally, these contaminants may be mobilized into the brain and other tissues since their seasonal lifecycles require significant fat deposition followed by extreme fat depletion during hibernation or migration (O'Shea and Clark, 2002). Studies have shown that insecticides applied to soy and corn can accumulate in bats, including chlorpyrifos (Eidels et al., 2007) and pyrethroids (Eidels et al., 2012). Chlorpyrifos is associated with impaired flight, impaired movement, and tremors in bats (Eidels et al., 2016). Mason et al. (2013) hypothesized that neonicotinoids may suppress the immune system of bats making them more prone to infectious disease and other stressors. Some of these pesticides may no longer be commonly used but were used during the period covered in this report (see Chapter 3, section 3.2.1.5). Other studies have found neonicotinoid exposure impairs echolocation—interfering with vocal, auditory, orientation, and spatial memory processing in bats (Wu et al., 2020; Hsiao et al., 2016). More information on mammals and pesticides general to agriculture (not specifically to the RFS Program) or other uses are available in the ECOTOX Knowledge Database (see Supplemental Table 12.1).

12.3.1.3 Pollinators and Other Beneficial Organisms

LCLM changes are also primary drivers of insect decline (Raven and Wagner, 2021; Sánchez-Bayo and Wyckhuys, 2019; Goulson et al., 2015). The composition of agricultural landscapes in the United States has shifted to greater dominance of a few annual crops, less perennial cover, and loss of noncropland habitat (Socolar et al., 2021; Meehan et al., 2011). This landscape simplification is associated with declines in biodiversity-mediated ecosystem services, including pest predation and pollination (Dainese et al., 2019; Meehan et al., 2011; Landis et al., 2008).

Insects are responsible for pollinating 85% of all flowering plants globally, including one-third of agricultural crops worldwide (Ollerton et al., 2011; Kremen et al., 2007). Bees in particular play a critical role in supporting agriculture, terrestrial food webs, and ecosystem function. In the United States, bees and other insects are responsible for pollinating \$15.1 billion worth of food crops per year (Calderone, 2012). The United States lacks a national monitoring program for native bee populations, but emerging evidence shows both native bees and honey bees are in decline due to habitat loss, pesticide exposure, pathogens, and other factors (Goulson et al., 2015). Native bee populations in the United States declined by an estimated 23% from 2008 to 2013 (Koh et al., 2016). Native bumble bee species, once plentiful across the United States, have undergone significant range contractions, particularly in the Midwest (Cameron et al., 2011). Managed honeybees are also experiencing die-off rates not observed in the past. Beekeepers in the United States experienced a 30–40% loss of their honey bee colonies over the past decade (Kulhanek et al., 2017). These losses have economic implications for migratory beekeepers who transport their bees across the country to produce honey and fulfill pollination contracts.

Recent LCLM changes, driven in part by expanding biofuel crop production, have reduced forage and nesting habitat for pollinators (Otto et al., 2018; Hellerstein et al., 2017; Koh et al., 2016; Otto et al., 2016). Bees require nectar- and pollen-producing flowers, blooming throughout the growing season, to complete their lifecycle. In agricultural areas of the United States, flowers are most abundant on grassland patches. Beekeepers actively seek out these grassland patches to keep honey bees (Otto et al., 2016), and native bee diversity is highest on grasslands such as pasture and CRP lands (Evans et al., 2018). By contrast, corn and soybeans provide little forage value for native bees and, consequently, native bee diversity is limited in these areas (Evans et al., 2018). Native bee abundance can be 2 to 3 times lower in corn and soybeans than in grasslands (Gardiner et al., 2010). The abundance of specialized bees, those bees that forage on only a small subset of the flowering plant community, is often greater in fields surrounded by more grasslands relative to row crops (Bennett et al., 2014). Thus, grasslands provide important refuge for native bees occurring in agricultural landscapes. In turn, increased abundance of native pollinators has a positive impact on yield of some annual crops (Mallinger and Prasifka, 2017; Bennett et al., 2014). Predictive models developed by Bennett et al. (2014) for fruit and vegetable producing regions in Michigan indicate the expansion of biofuel row crops into 1.5 million acres (600,000 ha) of marginal land could reduce native bee abundance by 24%.

In addition to loss of habitat, corn and soy are often treated with pesticides, another driver of bee declines (Goulson et al., 2015). In the United States, neonicotinoids are used to reduce insect pest pressure on crops (see Chapter 3). Between 79 and 100% of all corn and between 34 and 44% of all soybeans were treated with neonicotinoids in 2011 (Douglas and Tooker, 2015). This class of chemical is extremely toxic to bees and causes both lethal and sublethal effects (Baron et al., 2017; Stanley et al., 2015; Henry et al., 2012). While these chemicals are applied to target crop pests on agricultural fields, they can inadvertently end up in adjacent wildflower patches, flower patches within crop fields, and wetlands serving as forage sites for bees (Wood et al., 2019; Mogren and Lundgren, 2016; Main et al., 2014).

Besides neonicotinoids, the use of glyphosate on corn and soybean fields has increased 15-fold since 1974 (Benbrook, 2016) (see Chapter 3, section 3.2.1.5). Development of genetically engineered corn and soybean resistance to glyphosate has allowed for the increased use of this chemical on these crops. While the prophylactic use of glyphosate on agricultural fields provides an effective tool for controlling weeds on cropland, it also eliminates forage plants for pollinators occurring in agricultural areas. For example, the use of glyphosate has been implicated in the decline of monarch butterflies (*Danaus plexippus*) due to the elimination of milkweed (*Asclepias* spp.) from agricultural fields (Box 12.1: The Monarch Butterfly).

Beneficial arthropods such as spiders, ladybeetles, and lacewings aid in the control of crop pests, thereby increasing crop yield and reducing reliance on agrochemical inputs (see Chapter 9, section 9.3.1.4

Box 12.1. The Monarch Butterfly

Although once relatively common, monarchs were proposed for listing under the Endangered Species Act (ESA) in 2014 due to significant population declines and extinction risk. The eastern population of monarch butterflies declined 84% from 1996 to 2014 (Thogmartin et al., 2020; Thogmartin et al., 2017; Semmens et al., 2016) while the western population declined more than 99% since the 1980s (Pelton et al., 2019). In December 2020, the U.S. Fish and Wildlife Service (USFWS) announced that listing the monarch butterfly as threatened or endangered under the Endangered Species Act is warranted, but precluded by higher priority listing actions (https://www.fws.gov/savethemonarch/ssa.html). With this finding, the monarch butterfly becomes a candidate for listing; USFWS will review its status each year until they are able to begin developing a proposal to list the monarch.

Monarch females lay eggs on milkweed (*Asclepias* spp.) and larval monarchs forage exclusively on milkweed leaves during their development. Thus, milkweed is the essential host plant of monarch larva—without it, monarchs cannot survive. The migratory pathway of the eastern population falls within the Midwestern region of the United States; an area where a significant proportion of corn and soybeans are grown and a historic hotspot of milkweed growth and monarch production. From 1999 to 2010, there was a 58% decline in milkweed from the Midwest landscape (Pleasants and Oberhauser, 2013). Lark (2020) estimate that approximately 220 million common milkweed stems were lost due to conversion of grasslands, wetlands, and shrublands to corn, soybeans, and other crops across the Midwest from 2008 to 2016. This decrease in milkweed coincided with increased use of genetically modified, herbicide-resistant corn and soybeans. Because of decreased milkweed, there was an 81% decline in monarch production from the Midwest over this same period (Pleasants and Oberhauser, 2013).

Figure B.12.1. Adult monarch butterfly. The monarch butterfly (*Danaus plexippus*) is the only butterfly sub-species known to make a two-way migration; traversing the North American continent from Canada to Mexico in the spring and fall. Source: Lissy McCulloch, U.S. Geological Survey, Northern Prairie Wildlife Research Center.

Use of herbicide-resistant crops (and thus increased use of glyphosate for removing plants like milkweeds from cropland) has become more prevalent in the core summer breeding range of monarchs, likely contributing to a landscape-level reduction in milkweed (Stenoien et al., 2016; Pleasants and Oberhauser, 2013). As of 2019, 92% of corn and 94% of soybeans grown in the United States were genetically engineered for insect-resistance and herbicide tolerance [(USDA, 2020a); see also Chapter 3]. This allows producers to prophylactically eliminate non-cropped plants from corn and soy fields. The increased production of genetically modified corn and soybeans has contributed to reduction of milkweeds in farmland, and adjacent habitats, when glyphosate drifts from non-target cropland (Olaya-Arenas and Kaplan, 2019). While several environmental factors such as climate, habitat loss, and disease have been proposed as threats to monarchs, agricultural intensification and reduction in milkweed is a principal threat to monarch populations (Lark et al., 2020; Thogmartin et al., 2017).

for details on soil organisms). Research has shown that heightened insect biodiversity on and adjacent to croplands can serve as an effective biological control mechanism for reducing crop pests on agricultural fields (Lundgren and Fausti, 2015; Meehan et al., 2011; Landis et al., 2008). The estimated pest control provided by insect predators and parasitoids to U.S. agricultural producers is valued at $4.5 billion, annually (Losey and Vaughan, 2006). Perennial cover and greater plant species diversity within and adjacent to crop fields, as well as greater heterogeneity of the landscape matrix results in greater diversity of beneficial insects (Helms et al., 2020; Caballero-Lopez et al., 2016; Norris et al., 2016; Werling et al., 2014; Meehan et al., 2012; Werling et al., 2011b; Gardiner et al., 2009). Thus, landscape simplification may require additional agrochemical inputs to compensate for the loss in natural biological control services, perpetuating the use and associated cost of agrochemicals for treating pests (Meehan et al., 2011).

12.3.1.4 Plants

Plant biodiversity is also directly affected when perennial cover is converted to annual crops. The conversion of natural habitats to cropland results in loss of diversity of plant species because crops are often planted in monoculture, meaning only the crop plant species is present. A recent global assessment of biodiversity found that land use change is the largest driver of biodiversity loss, with agricultural expansion the most widespread form of land use change (IPBES, 2019). An estimated 2.8 million acres of new cropland (28% of the roughly 10 million acre total) in the United States from 2008 to 2016 were from relatively long-term habitat, defined as locations estimated without cultivation for at least 25 years (Lark et al., 2020). This is of particular concern because these areas often contain disproportionately high numbers of native plant species (Lark et al., 2020). Plant diversity loss can be calculated different ways. Recently, Chaudhary et al. (2018) calculated that taxon affinity—defined as the ratio of species richness in a particular land use type to that in a natural undisturbed area—of vascular plants relative to the ecoregion total is inversely related to cropland intensity (there were three levels of cropland intensity in the modeling study: minimal use, light use, and intense use).

Another aspect of crop management potentially affecting wild plant diversity is the application of pesticides (e.g., insecticide, herbicide, fungicide). These chemicals can stray (by leaching, runoff, volatilization, and/or spray drift) from the point of application into adjacent habitats where they can decrease the survival, flowering, seed production, and seedbank replenishment of non-target species. This may ultimately cause declining species richness, abundance, and diversity of non-target species. Olszyk et al. (2017) found that usage of the herbicide dicamba, commonly associated with corn and soy production (Chapter 3, Table 3.4 and 3.5), caused decreased seed production in perennial grassland species in Oregon. Other studies on the effects of dicamba drift show decreased flower production by plants in nontarget field margins resulting in fewer pollinator visits (Bohnenblust et al., 2016), and changes in herbivore arthropod communities (Egan et al., 2014), some of which were directly related to changes in plant chlorophyl content of non-target plants (Johnson and Baucom, 2022). Feber et al. (1996) found glyphosate decreased wildflower abundance in uncropped field edges. The timing of herbicide application relative to the lifecycle of wild plants is important as reproductive phases tend to be more susceptible than vegetative stages (Boutin et al., 2014). In a comprehensive evaluation of the herbicide atrazine, both spray drift and runoff resulting from application of the chemical to target-crop fields caused exposure to plants living in non-target (off-field) areas. Atrazine has been found to be toxic to seedlings of a wide range of terrestrial plants, with likely effects on terrestrial plant biodiversity and communities (U.S. EPA, 2016).

Finally, crop management also often includes the application of fertilizers such as nitrogen (N)-based fertilizers (e.g., urea and ammonium nitrate) to stimulate crop growth. Ammonia (NH_3), however, can volatize from the soil to contribute to atmospheric N concentration and potentially alter wild plant biodiversity. Direct exposure to NH_3 can alter lichen and plant physiology starting at low concentrations (Sutton et al., 2009). NH_3 volatilized from fields also contributes to total N deposition from the atmosphere. The effects of total N deposition in the United States were recently reviewed (U.S. EPA, 2020) and can lead to reductions in plant biodiversity and changes in plant nutrient status, among other effects (Clark et al., 2019; Carter et al., 2017; Pardo et al., 2011). Indeed, while historically atmospheric deposition of N was dominated by oxides from fossil fuel combustion, N deposition is increasingly dominated by chemically reduced forms linked to agriculture (Li et al., 2016). This does not occur to the same extent for phosphorus-based or other fertilizers because they do not have a gaseous loss pathway and are only released to the atmosphere via dust. Though thought to be limited in magnitude compared to N deposition, phosphorus from dust is hypothesized to contribute to eutrophication in remote water bodies (Stoddard et al., 2016).

12.3.1.5 Threatened and Endangered (T&E) Species

Under the Endangered Species Act (ESA), the federal government has the responsibility to protect threatened species (those likely to become endangered in the near future) and endangered species (those currently likely to become extinct throughout all or a large portion of their range). The ESA also requires the establishment and protection of critical habitat, areas which provide vital resources essential to the survival, reproduction, and population stability of T&E species. Many T&E species are sensitive to LCLM changes from grassland to annual crops, such as corn and soybeans. Several illustrative species are discussed below and in Lark (2023), and a full list of T&E species occurring in 12 Midwestern states—containing greater than 80% of the corn and soybean acres planted in the United States [(USDA, 2020b) Figure 12.2]—is provided in Supplemental Tables 12.2 and 12.3.

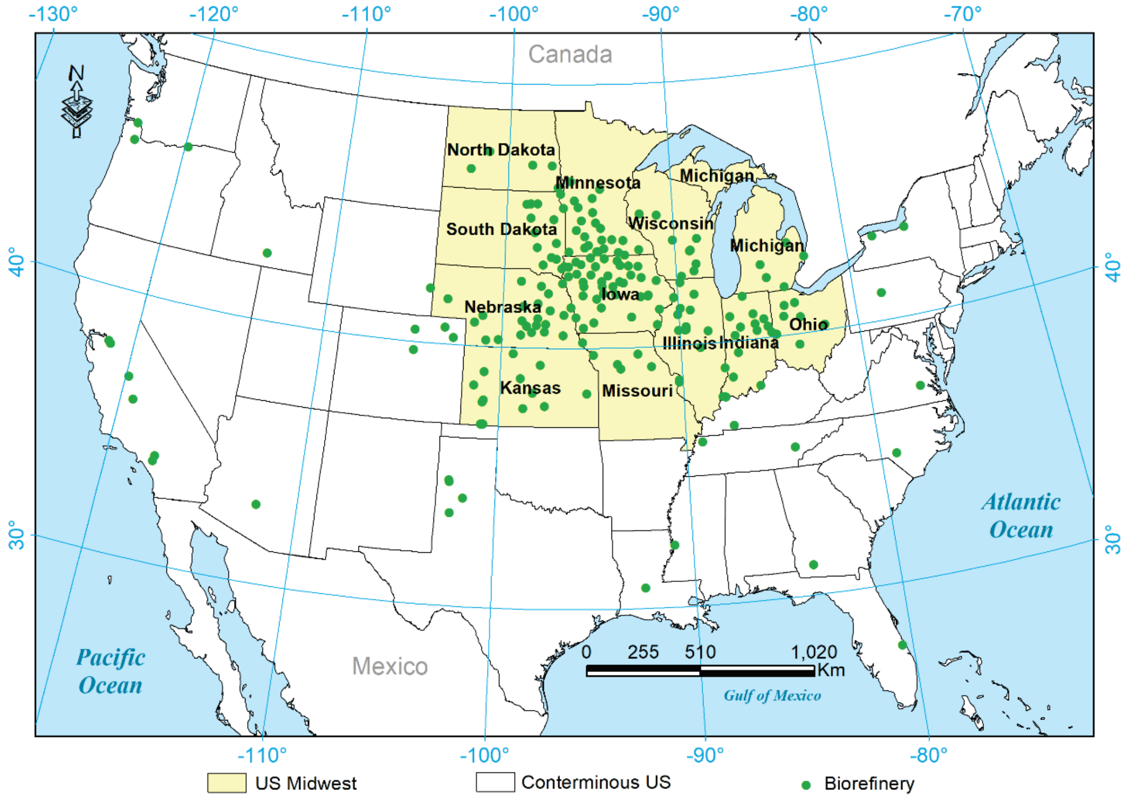

Figure 12.2. Map of the contiguous United States with 12 Midwestern states outlined (Zhang et al., 2021; Zhang et al., 2015), containing over 80% of planted corn and soybean acres in the country (USDA, 2020b). Dots represent locations of U.S. biorefineries (RFA, 2017).

One species affected by habitat loss to agriculture is the endangered whooping crane (*Grus americana*). The main migration routes of whooping cranes pass through the Great Plains of North America (Armbruster, 1990). The historical expansion of agricultural lands replaced wetland and grassland habitats in the region the whooping cranes depends on, and by the 1940s whooping cranes were extirpated from much of their historic range (Allen, 1952). Although the population has increased in more recent years, habitat loss continues within the well-defined and relatively narrow migration corridor for the only self-sustaining population of whooping cranes (Pearse et al., 2018). Although some individual cranes will feed on corn in agricultural fields, others avoid these areas (Barzen and Ballinger, 2017). Continued loss of habitat to agriculture in general, including from biofuel feedstock production, could further negatively impact whooping crane population recovery and survival.

In another example, an endangered butterfly, the Powesheik skipperling (*Oarisma poweshiek*), is dependent on a number of graminoids and forbs, native to tallgrass prairie ecosystems, for egg laying, larval food sources, and adult nectar sources (Belitz et al., 2019; Pogue et al., 2016; Swengel and Swengel, 2014). Hence, this butterfly has been particularly impacted by the loss of habitat from land conversion and the subsequent loss of plant biodiversity. Fragmentation of native prairie habitat limits distribution of the species, as they are only able to fly for short periods at a time and cannot travel long distances between prairie remnants (Pogue et al., 2016). Another iconic butterfly species, the monarch butterfly (*Danaus plexippus*) is currently being monitored by the U.S. Fish and Wildlife Service for potential protection under the ESA (see Box 12.1: The Monarch Butterfly). The rusty patched bumble bee (*Bombus affinis*) and Mitchell's satyr (*Neonympha mitchellii*) are among other imperiled pollinators of the Midwest United States likely impacted by loss of native habitat.

Several protected plant species also illustrate the impacts of habitat loss to agriculture. Four of these plants grow in remnants of the once-vast prairie ecosystem and now are among the region's rarest flora—the eastern prairie fringed orchid (*Platanthera leucophaea*), western prairie fringed orchid (*Platanthera praeclara*), prairie bush clover (*Lespedeza leptostachya*), and Mead's milkweed (*Asclepias meadii*). Ranges of these rare prairie species lie within the areas of intensive land conversion to agriculture (Lark et al., 2015; USFWS, 1989, 1988, 1987). Habitat loss due to land conversion is the main reason for the rarity of prairie plants, and so continued land conversion and agriculture extensification could have severe negative consequences on these species' persistence and future recovery. As noted above, Lark et al. (2020) estimated shifts in land cover from relatively long-term grassland habitat, which they define as areas not cultivated for cropland or pasture for at least 25 years, to annual crops. Of the 10 million acres of new cropland in total, 2.8 million acres came from this category, with most of that from unimproved grasslands (2.3 million acres), potentially home to many native plant species. Although this study did not specifically address T&E plants, it is evidence of the loss of native plant habitat, which likely includes rare species.

Although habitat loss is likely the main stressor to T&E species, certain agricultural production practices can have deleterious effects. Pesticide exposure likely negatively impacts the Powesheik skipperling butterfly, both through impacts on nectar plants and because their larvae overwinter on their host plants (USFWS, 2014). As discussed in the Monarch Butterfly box earlier, the loss of milkweed species due to pesticides has been cited as one of the reasons for the decline of the monarch butterfly. Pesticide impacts on bats in general agricultural settings were also discussed previously (see section 12.3.1.2). Furthermore, T&E plant species can be affected by fertilizer volatilization and atmospheric N deposition just like other plant species (see section 12.3.1.4), and indeed rare species and native species of higher conservation value have been found to be more vulnerable to N-induced losses (Clark et al., 2019; Clark and Tilman, 2008; Suding et al., 2005).

12.3.2 New Analysis

To better understand potential impacts to T&E species, a new analysis was conducted for this chapter of the RtC3. T&E critical habitat was compared to a shift in perennial land cover to corn and soybeans. Specifically, the U.S. Fish and Wildlife Service Critical Habitat linear and polygon features (USFWS, 2020) and 2008 to 2016 cropland conversion data from Lark et al. (2020)—a 30 m resolution raster of land in perennial cover to crop conversion from processing the CDL data (see Chapter 9)—were used. In brief, the area of perennial cover conversion to corn and soybeans overlapping with critical habitat and a 1-mile buffer were calculated. Each T&E species was classified as "terrestrial," "aquatic," or "both" based on knowledge of the species, and those species with 10 acres or more of conversion to corn or soybeans were identified. Although the habitats for the aquatic species were water bodies, they were included if 10 acres or more were converted to corn or soybeans within the surrounding 1-mile buffer of the associated water body.

Across the contiguous United States, 27 terrestrial T&E species had an estimated 10 acres or more of conversion of land in perennial cover to corn or soybeans within 1 mile of its critical habitat (Table 12.1; Figure 12.3a). Of those, six T&E species had estimated conversion within their critical habitats. For example, the Indiana bat, discussed in section 12.3.1.2, had conversion of perennial cover to both corn and soybeans within its critical habitat (Figure 12.3b). Supplemental Tables 12.2 and 12.3 contain the full list of species potentially affected in the 12 U.S. Midwestern states.

Table 12.1. Habitat types and numbers of threatened and endangered (T&E) species with 10 acres or more of perennial cover converted to corn or soybeans within their critical habitat plus 1-mile buffer between 2008 and 2016 for the contiguous United States. Values in parentheses are numbers of species with 10 acres or more converted land within critical habitat only, not including 1-mile buffer. Values calculated by comparing the critical habitat assigned by the USFWS (2020) with cropland conversion data from Lark et al. (2020).

Species habitat type[a]	Number of T&E species with ≥10 acres converted to…		
	…corn within T&E critical habitat + 1-mile buffer (and within critical habitat only)	…soybeans within T&E critical habitat + 1-mile buffer (and within critical habitat only)	…corn or soybeans within T&E critical habitat + 1-mile buffer (and within critical habitat only)[d]
Terrestrial	25 (6)	13 (3)	27 (6)
Aquatic[b]	76	63	78
Both terrestrial and aquatic[c]	6 (4)	4 (2)	6 (4)

[a] Habitat type of species represents the predominant, but not necessarily the sole, habitat type of species, as assigned by best professional judgment of authors.
[b] Aquatic species were included if 10 acres or more of corn or soybeans were planted within the surrounding 1-mile buffer of their aquatic habitat. No values are listed in parentheses since corn and soybeans are not planted directly in aquatic critical habitat (i.e., water).
[c] These are species that routinely use both terrestrial and aquatic habitats, or wetland species.
[d] This column represents the number of species with either 10 acres or more converted to corn or 10 acres or more converted to soybeans—corn and soybean acres were not summed. Because many species are potentially affected by land conversion to both corn and soybeans, these numbers are also not the sum of the two preceding columns.

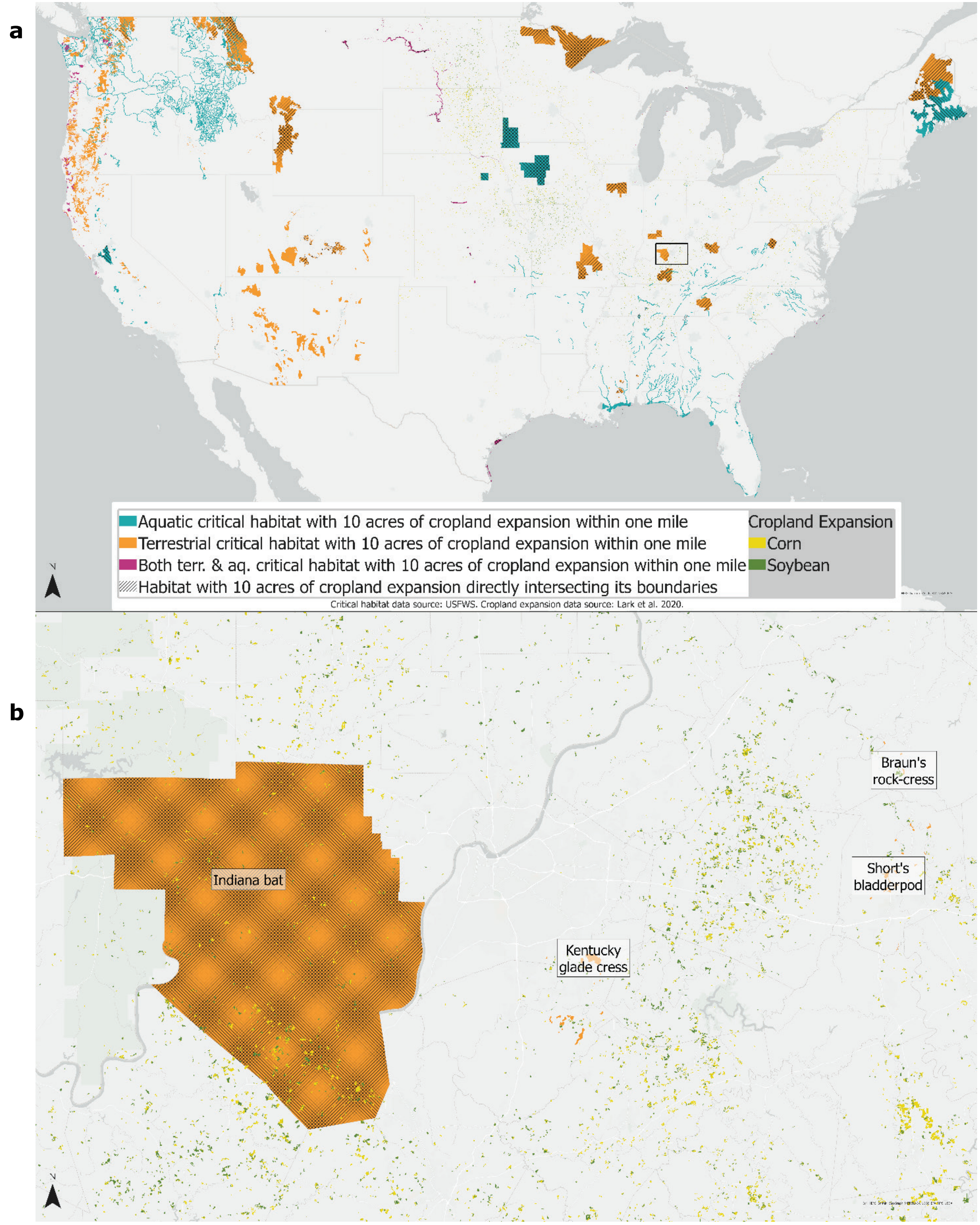

Figure 12.3. Agricultural expansion in and around critical habitat for threatened and endangered (T&E) species. Shown are critical habitat of aquatic and terrestrial T&E species within the conterminous United States (a) and within the Lower Ohio River Valley (b), with ≥10 acres of corn or soybean expansion onto land previously under perennial cover within 1-mile or intersecting its boundaries. Critical habitat data were from USFWS (2020) and data on shifts from perennial cover to corn and soybean were from Lark et al. (2020). Land had been under perennial cover for at least 6–10 years prior to conversion, according to analysis of the USDA's Crop Data Layer by Lark et al. (2020).

12.3.3 Attribution to the RFS

Up to this point, this chapter has largely focused on the effects of corn and soybean production in general, with the rare study further apportioning the effects of these crops for biofuels. The scientific literature generally did not address the effect of the RFS Program on terrestrial biodiversity, with the exception of a recent publication [(Lark et al., 2022); see discussion below]. This section addresses this topic, building in part upon the attribution information in Chapter 6.

As detailed in previous sections, there are two major mechanisms by which the production of corn and soybeans—the dominant feedstocks to date—negatively impact terrestrial biodiversity: (1) the expansion of these crops onto former perennial cover; and (2) the agricultural practices for corn and soybeans. Regarding the first mechanism, the scientific literature overwhelmingly shows the negative impacts of a shift in LCLM from mixed-perennial grasslands to annual crops on grassland birds, bats, beneficial insects and pollinators, and native plants (see section 12.3.1). Regarding the second mechanism, the literature also shows that corn and soybean production practices can have negative impacts on terrestrial biodiversity, particularly through pesticide usage, but these effects from agricultural practices largely depend on whether the land was previously cultivated in a different crop or not. Thus, the effects of the RFS Program on terrestrial biodiversity depend upon whether it induced shifts from land in perennial cover, such as grasslands, to corn or soybeans, or increased corn or soybean production on existing cropland.

In a recent publication, Lark et al. (2022) used a modeling approach to estimate the effects of the RFS Program from 2008 to 2016 compared to a non-RFS scenario. They concluded that the RFS increased total cropland by 5.2 million acres (2.1 million hectares) and corn acreage by 6.9 million acres (2.8 million hectares)[7], with most of the changes in LCLM occurring in the areas of the U.S. Midwest. These estimates are larger than the maximum cropland and corn acreage estimates attributable to the RFS made in this report (0-1.9 and 0-3.5 million acres, respectively; see Chapter 6, section 6.4.2) because of several underlying assumptions in Lark et al. (2022) that increased the estimated effect of the RFS Program (see Chapter 6, section 6.3.3). Nevertheless, once these attributional differences are aligned, the estimates of LCLM change are similar. Thus, while the estimated acreages in Lark et al. (2022) attributable to the RFS Program are higher than the estimates in this report, their analysis similarly finds the RFS Program increased cropland and corn acreage in part from land in perennial cover.

The attributional analysis in Chapter 6 estimated that 0 to 1.9 million acres of additional cropland were associated with RFS Program corn ethanol production between 2008 and 2016. This range represents approximately 0 to 20% of the observed net increase in U.S. crop area over this period (see

[7] The estimated increase in corn acreage exceeded the increase in total cropland because corn acreage also came at the expense of other crops, planted on existing cropland.

Chapter 6, section 6.4.3). These estimates are relatively small compared to total U.S. cropland (0–0.5%, Table 6.11), but may be important regionally or locally, especially in areas with a higher concentration of converted acres. Further, according to Chapter 6 estimates, corn ethanol production attributable to the RFS Program caused an increase of between 0 and 3.5 million acres of corn. Up to 1.9 million acres could have overlapped with those of expanding cropland, leaving any remaining acres from corn due to crop switching on existing cropland. Notably, these estimates represent only the effects of the RFS Program from corn ethanol, and may be larger if an RFS-induced effect of soy biodiesel on land use were quantified (see Chapter 7).

The estimated range of 0 to 20% can be applied to potential impacts on terrestrial biodiversity, suggesting impacts from RFS-associated cropland expansion could range anywhere from no-effect in certain years to negative effects in other years during the time period assessed (roughly 2005–2016, see Chapter 6 for details). As noted above, this range of estimates does not include any acreage associated with RFS soy biodiesel volumes, since these acreage estimates were not estimated in this report (see Chapter 7 for details), nor does it include the likely negative effects associated with any potential increase in corn and soybean production on existing corn and soybean acreage associated with the RFS Program.

On T&E species, the analysis in section 12.3.2 found that conversion from land in perennial cover to corn or soybeans overlapped in some areas with the critical habitat of T&E species between 2008 and 2016. As noted previously, additional analyses are needed to confirm these findings due to noted uncertainties in the CDL. Moreover, conversion can occur for multiple reasons, including, but not limited to, the RFS Program. If up to 20% of the additional cropland was due to the RFS Program, RFS-attributable conversion may or may not have occurred within the critical habitat of T&E species. Further analysis is needed to determine whether the RFS Program negatively affected T&E species through the conversion of grasslands to cropland, and if so, the magnitude of any such impact. These major uncertainties also hold for determining any future effects of the RFS Program, as evidenced in the Final Set Rule for the years 2023–2025 (Docket # EPA-HQ-OAR-2021-0427), where EPA found the RFS Set Rule may affect, but is not likely to adversely affect, T&E species.

Overall, the range of effects of the RFS Program on terrestrial biodiversity likely spanned from no effect to a negative effect historically through the conversion of land in perennial cover, such as grasslands, to corn or soybeans, or corn and soybean production practices. Further refinement of the acreage estimates attributable to the RFS Program, and allocation of those acreages to the landscape, are needed to reduce this range of possibilities. This period is limited to 2008 through 2016 and does not address effects before or after, since the effects of the RFS Program on corn ethanol and soy biodiesel production may have changed. Moreover, the magnitude of any effects on biodiversity is unknown and requires further evaluation. The magnitude of the RFS Program on biodiversity may have been relatively small across the entire United States, but could have been more important in localized areas. Finally, whether T&E species were impacted by the RFS Program is also consistent with the information above, but remains unknown, and requires further evaluation.

12.3.4 Conservation Practices

Agricultural landscapes can serve in a dual role as production areas for food, feed, fuel, and fiber, as well as spaces for conservation of biodiversity and generation of the ecosystem services required by society. As a mirror image of the land cover and management discussion above, these conservation practices can be viewed in two main categories: (1) the preservation of habitat or land set-aside programs, and (2) measures taken to reduce impacts from currently planted cropland. Nationally, Farm Bill conservation programs are the most significant source of public funds for private lands conservation (McGranahan et al., 2013). Since its launch in 1985, the CRP has become the primary policy instrument for promoting grassland habitat on private lands nationwide (Hellerstein et al., 2017; Stubbs, 2013), and enrollment of roughly 20–35 million acres in perennial cover through time (Figure 5.11) has generated an array of environmental benefits (Johnson et al., 2016; Belden et al., 2012; Wiens et al., 2011), including provision of habitat for a diversity of grassland species (Heard et al., 2000).

Within an agricultural landscape, greater heterogeneity or complexity benefits wildlife, which often provides ecosystem services as well. For instance, annual croplands (e.g., corn) with an abundance of perennial grasslands in the landscape supported larger populations of generalist predator insects, providing a reduction in crop pests (Werling et al., 2011b; Werling et al., 2011a). Interspersion of different row crops did not benefit avian taxa in the Midwest, but interspersion of woody crops, wide field margins, and/or riparian buffers was found to be beneficial (Wilson et al., 2017; Conover et al., 2007). Higher farmland heterogeneity may also benefit bat communities by increasing length of field boundaries, particularly fields with hedgerows, and reducing distances between foraging and roosting habitats (Monck-Whipp et al., 2018).

Additionally, best management practices (BMPs) have been developed to protect terrestrial ecosystem health and biodiversity on planted croplands. Implemented BMPs include conservation tillage, cover crops, and vegetative buffers. Whether implemented in-field (e.g., conservation tillage, contour strips, grassed waterways), at field margins (e.g., vegetative buffers, pollinator habitat), or along waterways (e.g., riparian buffers), installation of BMPs have reduced soil erosion (Dosskey et al., 2012) and established habitat in the agricultural landscape (Lemke et al., 2011). Even small areas of grassland have been associated with increasing trends in grassland birds (Veech, 2006). Prairie strips were added as a cost-share practice under the CRP in the 2018 Farm Bill. Installation of prairie strips within production fields can positively affect grassland bird and invertebrate communities, in addition to reducing water runoff and soil and nutrient losses (Schulte et al., 2017) (Liebman et al., 2013). When strategically placed, prairie strips generate disproportionally greater environmental benefits than would be expected from their area alone (Helmers et al., 2012), with placement of prairie strips near croplands more than doubling avian diversity (Schulte et al., 2017).

Similarly, integration of perennials within the cropping rotation has been shown to generate environmental benefits. These benefits include improved soil health (Ryan et al., 2018; Crews and Rumsey, 2017), disruption in pest cycles, habitat for beneficial insects (Power, 2010), and increased annual crop yields following the perennial segment of the cropping rotation (Duiker and Williamson, 2018). Development of perennial grain crops is of increasing interest, with the grain from intermediate wheat grass as the most prominent example (Ryan et al., 2018). Markets developed for perennial grains present new opportunities for farmers to align their economic and environmental goals.

Other conservation options that protect birds and other wildlife on planted croplands include protective harvest equipment and timing, reduced pesticide use, and safer pesticide application methods. Farm operations, such as planting and harvesting, can destroy a significant number of nests (20–40%) if they occur during the nesting season (VanBeek et al., 2014; Stallman and Best, 1996). Timing of planting and harvest operations to avoid the late-spring nesting season can reduce mortality. Pesticides are also an important factor contributing to the loss of biodiversity. Using alternative management practices to lower pesticide usage would likely reduce bird mortality (Stanton et al., 2018). Similarly, in one study, bat species richness, total activity, and activity levels were significantly higher in organic versus non-organic fields for five out of seven bat species examined, and relationships were in the same direction for the other two species (Put et al., 2018).

Finally, precision agriculture, which comprises the technologies and data used to adapt management practices to site and in-field variability (Berry et al., 2003), has been identified as an approach for improving environmental outcomes. When used effectively, precision agriculture reduces input costs and losses of nutrients and pesticides to the environment (Sela et al., 2017; Schieffer and Dillon, 2015; Bongiovanni and Lowenberg-Deboer, 2004).

12.4 Likely Future Impacts

As noted in Chapter 2, corn ethanol and soy biodiesel will likely remain the dominant biofuels in the near future considered in this report (out to 2025). Whether grasslands continue to be converted to corn or soybeans will in large part determine the magnitude of future effects, since the largest impact on terrestrial ecosystems generally occurs from this LCLM shift. Lark et al. (2020) reported a slowdown nationally in cropland expansion since 2011 and especially since 2015 [Figure 2 in Lark et al. (2020)], in agreement with other sources from Chapter 5 and with reaching the E10 blend wall in 2013. When the Set

Rule was finalized on June 21, 2023, EPA published estimates of cropland expansion with those biofuel volumes (see Chapter 6, section 6.5).[8] EPA estimated the biofuel volumes in the Final Set Rule could potentially lead to an increase of as much as 2.65 million acres of cropland by 2025, mostly from soybean expansion in the Midwest, but to a lesser extend from corn expansion in the Midwest and canola expansion in North Dakota. Still, the likely future effects of the RFS Program are highly uncertain. While the projected cropland expansion for 2023–2025 is slightly larger than the estimated historical cropland expansion from the RFS Program, EPA cannot say with reasonable certainty that any particular terrestrial ecosystem or biodiversity will be impacted, due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes. More information on where these changes are anticipated, and what agricultural practices are employed, are needed to better constrain estimates of the likely future effects to terrestrial ecosystems. Moreover, the potential biodiversity effects of canola expansion require investigation. Additional information may be found in the associated docket for the Set Rule (EPA-HQ-OAR-2021-0427).

12.5 Comparison with Petroleum

All energy sources have environmental effects, and so it can be useful to compare the effects of biofuels relative to petroleum, the dominant fuel it displaces. As noted in earlier chapters, the greenhouse gas emissions of biofuels relative to petroleum are outside the scope of this report, but both fuel types can have other environmental effects. Biofuels and petroleum often affect terrestrial biodiversity through habitat loss of the land area required for production (i.e., the land footprint of the industry). Studies have compared the area required by the two industries [e.g. (Dale et al., 2015; Parish et al., 2013)]. According to Dale et al. (2015), the petroleum industry out to 2030 will require more than double the area of biofuels globally, including areas of the ocean and remote locations in the Arctic. In total, these areas overlap with a higher number of threatened species than that of projected biofuel production over the same period (Dale et al., 2015). In a limitation, this analysis did not consider the amount of energy produced by the two industries. Conversely, Elshout et al. (2019) did compare the two industries on a per unit energy

[8] On July 26, 2022, the United States District Court for the District of Columbia filed a consent decree, which after stipulations from the parties, required EPA to sign a notice of proposed rulemaking (NPRM) to establish 2023 volumes for the RFS Program on or before November 30, 2022, and to sign a notice of final rulemaking finalizing 2023 volumes by June 21, 2023. *Growth Energy v. Regan et al.*, No. 1:22-cv-01191, Document Nos. 12, 14, 15. EPA's proposed RFS volumes for 2023-2025, 87 FR 80582 (proposed and signed on Nov. 30, 2022 and published in the Federal Register on Dec. 30, 2022 available in Docket No. EPA-HQ-OAR-2021-0427 at https://www.regulations.gov, were finalized on June 21, 2023 and published in the Federal Register on July 12, 2023 (88 FR 44468). The estimates of cropland expansion from the RFS Program took considerable time and effort to develop; thus, they are based on the proposed volumes from November 30, 2022, rather than the final volumes from June 21, 2023. Because of that, the acreages discussed here are merely illustrative. However, since the total volumes from crop-based feedstocks changed fairly little between the draft and final rules, they are still relevant here (see Chapter 6, section 6.5).

basis, and concluded the production of biofuels negatively affected biodiversity more on a per unit energy basis than gasoline and diesel fuel production in most locations considered in a global analysis. This study, however, appears to assume all biofuel feedstock production leads to habitat loss, yet this is not the case in the United States, where feedstocks are grown on existing cropland as well as on newly converted lands.

In a study specific to the United States, Trainor et al. (2016) estimated land requirements of a variety of energy sources, including biofuels and petroleum. They estimated biofuels required more than two-thirds of the land used for all energy sources in the United States between 2007 and 2011, while only producing 6% of the country's total energy production. Like Elshout et al. (2019), however, this analysis focuses solely on the land requirements and does not differentiate between biofuels grown on existing cropland versus newly converted cropland. Projecting into the future, Trainor et al. (2016) estimated biofuels and petroleum production become similar in their land requirements if the spacing requirements between oil wells are considered.

In addition to land required, the time or effort to recover from any adverse impacts should be included when comparing the two industries. In a qualitative weighing of the effects along the supply chain, Parish et al. (2013) concluded that the maximum recovery time for petroleum environmental effects would exceed that from biofuels. They primarily point to the extraction step along the supply chain, where effects of petroleum exploration and extraction may take geologic timescales to recover from, while biofuel feedstock production, conducted on arable land, is reversible on the scale of years to decades.

12.6 Horizon Scanning

Although the substantial majority of biofuel volumes consists of corn starch–based ethanol, other feedstocks such as algae, municipal waste (for biogas), and cellulosic feedstocks—like perennial grasses, woody residues, and corn stover—have been deployed in smaller quantities or may be deployed in the near future (see Chapter 2 for details). It is unclear currently whether there are any terrestrial biodiversity effects of algal feedstocks or municipal waste for biogas, other than they could provide benefits to biodiversity by requiring less additional land than many other feedstocks [e.g., see Correa et al. (2020); see also Chapter 13 for discussion of the potential effects of algae on aquatic biodiversity and Chapter 9 for the effects of biogas digestate on soil quality]; however, the use of cellulosic feedstocks is discussed here briefly.

The scientific literature suggests perennial grasses as cellulosic feedstocks would increase biodiversity relative to row crops. Mixing perennial grasses or woody crops into a landscape can improve biological diversity through the landscape heterogeneity created and the benefits of the habitat provided by the perennial species (Robertson et al., 2011). For instance, in a modeling study, Meehan et al. (2010) estimated that increases in bird species richness throughout the Upper Midwest could be achieved by replacing corn and soybeans on marginal lands with mixed perennials. Although perennial feedstocks generally produce less energy per area than annual feedstocks, they often create better biodiversity outcomes (Núñez-Regueiro et al., 2021). The advantages to wildlife of growing perennial grasses include delayed harvesting times for biomass (after the nesting season) and not replanting or tilling after establishment (Best et al., 1997). Herbicide is generally applied only during establishment and insecticides are rarely required (Meehan et al., 2011). Moreover, perennial biomass has the potential to provide high-quality habitat for bees if the perennial cover includes a wildflower or forb component. Researchers estimated replacing annual energy crops (i.e., corn and soybeans) with perennial grass energy crops along Wisconsin waterways would increase pollinator abundance by 11% (Meehan et al., 2013).

As with all biofuel feedstocks, the effects of perennial grasses depend highly upon the prior land use or the baseline comparison. The effects discussed in the literature are often compared to row crops, and not compared to unmanaged, or lightly managed, grasslands. If highly managed, perennial grass-based bioenergy crops replace lightly managed grasslands, the effects could be less positive or even negative (Núñez-Regueiro et al., 2021). In the Department of Energy's Billion Ton study, for instance, positive changes in grassland bird richness were dominated by grid cells that were planted in cotton or corn in 2014 that transitioned to switchgrass (*Panicum virgatum*) in 2040 in the model (Jager et al., 2017). By contrast, negative changes in grassland bird richness were dominated by grid cells planted in pasture or hay in 2014 that changed to switchgrass in 2040.

Under EISA, renewable biomass may include slash and precommercial thinnings from nonfederal forestlands, and planted trees and tree residue from actively managed tree plantations on nonfederal land. The biodiversity effects of harvesting woody residues in forests are generally compared to a non-removal baseline. Studies have generally shown negative to no effects of residue removal on biodiversity, except for a few studies showing a positive effect on understory plant species (Ranius et al., 2018). Among biological groups, amphibians and reptiles may be particularly sensitive to changes in moisture and temperatures in the soil and forest floor caused by woody residue removal (Semlitsch et al., 2009; Todd and Andrews, 2008). Beyond these biodiversity effects, there can be ecosystem health benefits of harvesting woody residue in targeted circumstances, such as the potential to reduce large, severe fires in certain fire-prone ecosystems. For instance, woody biomass removal from forest ecosystems that historically experienced frequent, low intensity fires, such as ponderosa pine systems, can return these forests to a more open structure and function, while reducing fuel loads and the potential for severe fires (Moritz et al., 2014). In contrast, woody biomass removal has fewer ecological benefits in other fire-prone forests already predicated on severe, stand-destroying fires for renewal (e.g., lodgepole pine)

(Moritz et al., 2014). In general, matching woody biomass removal with the natural disturbance ecology of the ecosystem is likely to provide the greatest ecosystem benefits, while minimizing potential negative impacts.

Under its Wildfire Crisis Strategy, the USFS is removing hazardous fuels from 50 million acres over 10 years, which will generate millions of tons of non-marketable biomass each year. This material presents a disposal and management challenge. Biofuels and the RFS Program may offer a potential to reduce that challenge by financially incentivizing utilization of this material for higher value end uses. To date, production of renewable fuel from forestry feedstock has been very limited in the RFS Program, but there are organizations looking into its commercial viability.

Finally, direct studies of corn stover removal on biodiversity are generally lacking, yet inferences can be drawn from studies of tillage practices and the resulting residue left on the surface of agricultural fields. In general, conservation tillage practices, including no-till, increase biodiversity, including of birds and small mammals, by leaving greater residue on site compared to conventional tillage (Brady, 2007). The remaining crop residue under no-till systems can provide greater bird nesting habitat (Basore et al., 1986), as well as increased forage for some birds (Rodenhouse and Best, 1994). Additionally, while corn stover research has focused primarily on soil health effects (see Chapter 9), any changes in erosion and nutrient cycling, for example, can have bottom-up effects on biodiversity. This suggests removal of corn stover may tend to have negative effects on biodiversity at the scale of individual fields. At the larger landscape level, stover removal could either increase or decrease biodiversity depending upon whether it reduces or increases the amount of land used to grow corn for biofuels (Fargione et al., 2009). Stover theoretically allows more ethanol to be produced per unit of land and so it might improve landscape-level biodiversity if it enables other land to remain in natural habitat. The opposite could happen, however, if it incentivizes more conversion, rather than less (Fargione et al., 2009).

Beyond biofuel feedstocks, future biodiversity outcomes will be also affected by other stressors, in particular climate change. A thorough discussion of the effects of climate change on terrestrial biodiversity is beyond the scope of this chapter [for more information see (Lipton et al., 2018)]. In some cases (e.g., cover crops), the conservation practices discussed in section 12.3.4 can both promote terrestrial biodiversity in agricultural-biofuel systems and resiliency to climate change effects.

12.7 Synthesis

12.7.1 Chapter Conclusions

- Impacts to date from biofuels on domestic terrestrial biodiversity, as an indicator of ecosystem health, are primarily due to corn and soybean feedstock production for ethanol and soy biodiesel. Shifts in perennial plant cover to corn and soybeans, and corn and soybean production practices are the two main drivers of effects.

- Of land in perennial cover shifting to annual crops, the vast majority was from grasslands, ranging from relatively unmanaged to highly managed grasslands (e.g., hay, pasture). The loss of grassland cover to annual crops, such as corn and soybeans, negatively impacts terrestrial biodiversity, including grassland species of birds, bats, pollinators and other beneficial organisms, and plants.
- Between 2008 and 2016, shifts from land in perennial cover to corn and soybeans due to all causes, including potentially biofuels, occurred in areas adjacent to or within critical habitat of 27 terrestrial threatened and endangered (T&E) species across the contiguous United States, according to an analysis using the USDA Cropland Data Layer (CDL). The CDL is relatively accurate at large spatial scales (e.g., states) but can be more uncertain at local scales. Thus, it may require verification with imagery or direct visitation to confirm these results.
- Beyond change in land cover, crop production practices for corn and soybeans can also negatively affect terrestrial biodiversity, particularly through pesticides.
- The range of possible impacts from the RFS Program likely spanned from no effect to a negative effect on terrestrial biodiversity historically (2008 to 2016). Further refinement of the acreage estimates attributable to the RFS Program are needed to reduce this range of possibilities. These findings do not necessarily apply for years beyond 2016, when the effects of the RFS Program on corn ethanol and soy biodiesel production may have changed.
- Further evaluation would be needed to quantify the magnitude of any historical impacts of the RFS Program on biodiversity. Any effects may be relatively small compared to those of total U.S. cropland, but may be more important regionally or locally. Finally, whether T&E species were impacted by the RFS Program during this period (2008 to 2016) is also possible, but unknown, and requires further evaluation.
- Conservation practices can reduce negative impacts to terrestrial biodiversity. These practices include protecting environmentally sensitive lands, increasing habitat heterogeneity, and decreasing the use of pesticides.
- The likely future effects of the RFS Program are highly uncertain. As part of the analysis for 2023–2025, EPA determined that the RFS Set Rule is not likely to adversely affect listed species and their designated critical habitats. While the projected cropland expansion for 2023–2025 is slightly larger than the estimated historical cropland expansion from the RFS Program, EPA cannot say with reasonable certainty that any particular terrestrial ecosystem

or biodiversity will be impacted, due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes.

12.7.2 Conclusions Compared with the RtC2

The findings from this chapter strengthen and build upon the conclusions in the RtC2 (2018). The net change from grassland cover to crop production, including to corn and soybeans, has persisted. Moreover, the scientific literature continues to emphasize the negative effects of habitat loss on terrestrial biodiversity. On T&E species, this chapter advances the fundamental understanding beyond that of the RtC2. According to the analysis derived from the CDL (Lark et al., 2020), loss of land in perennial cover to corn and soybeans has occurred adjacent to (within 1 mile) or inside the critical habitat of 27 terrestrial T&E species. How much of this is due to biofuels generally or the RFS Program specifically remains an unanswered question. Beyond habitat loss, the literature also continues to emphasize that land management practices on working agricultural lands affect terrestrial biodiversity. Recent studies highlight the negative effects of pesticide use on taxa such as grassland birds, bats, and pollinators. Conservation practices can reduce the negative effects of crop production on terrestrial biodiversity; examples include setting aside sensitive land, avoiding planting and harvesting during nesting season, and decreasing pesticide use or finding safer alternatives.

12.7.3 Uncertainties and Limitations

- The largest source of uncertainty on historical impacts to terrestrial ecosystems stems from the range of estimated additional cropland potentially due to the RFS Program and a lack of understanding of the exact location of these converted lands. Chapter 6 includes estimates of 0 to 1.9 million acres of additional cropland associated with corn ethanol production attributable to the RFS Program since 2008. Much of this additional cropland likely came from grasslands, but this range includes zero, and did not include any cropland associated with soy biodiesel production attributable to the RFS Program.
- Biodiversity effects can vary depending upon the location of the grasslands shifting to corn and soybeans. Estimates exist of the amount and location of grassland conversion to crops, but these are not specific to biofuels or the RFS Program.
- The amount of crop switching to corn and soybean due to biofuels and the RFS Program and the subsequent biodiversity effects remain uncertain as well.
- Further uncertainty exists regarding the relative mix of production and conservation practices implemented on lands used to grow biofuel feedstocks generally and as a result of the RFS Program.

12.7.4 Research Recommendations

- As noted above, an estimated 0 to 1.9 million acres of additional cropland was associated with RFS Program corn ethanol production historically. Further spatial analysis could improve estimates of the location of these lands in perennial cover that potentially shifted to corn or other biofuel crops. This would be a critical step toward quantifying potential RFS Program effects on terrestrial biodiversity historically. Comparing the location of these grassland conversions to T&E species ranges would also contribute to an understanding of any past effects of the Program on these species.
- Additional research is needed to understand how shifting cropping patterns, such as from small grains to corn and soybeans, may impact biodiversity.
- More research is needed on quantifying the biodiversity effect of production practices, including pesticide usage, on croplands potentially attributable to the RFS Program.
- Research focused on potential effects of production practices on the biodiversity of T&E species would be particularly beneficial.

12.8 References

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12.S Supplemental Tables

Supplemental Table 12.1. The number of records arranged by pesticide and organism group contained in the EPA ECOTOX Knowledge Database. This public database compiles information on the toxicity of pesticides, derived predominantly from the peer-reviewed literature. Example pesticides listed are some of the more commonly used in corn and/or soybean production. Glyphosate, atrazine, metolachlor, acetochlor, and 2,4-D are herbicides, and imidacloprid, clothianidin, and thiamethoxam are neonicotinoid insecticides. See https://cfpub.epa.gov/ecotox/ for more information (accessed April 2021).

	Terrestrial Species Group							Aquatic Species Group						
Pesticide	Birds	Mammals	Insects & Spiders	Worms	Reptiles	Other Terrestrial Animals[a]	Plants & Fungi[b]	Fish	Amphibians	Crustaceans	Insects & Spiders	Worms	Other Aquatic Animals[c]	Plants & Fungi[d]
Glyphosate (1071-83-6)	81	395	765	330	97	154	7007	4291	1915	1165	203	189	935	2074
Atrazine (1912-24-9)	326	1679	222	157	90	70	4334	2822	1235	927	237	52	780	5039
Acetochlor (34256-82-1)	48	0	6	46	0	1	257	248	74	57	0	0	4	42
Metolachlor (51218-45-2)	28	27	47	43	0	1	2855	103	56	88	25	2	12	377
2,4-D (94-75-7)	1488	4913	2251	3	613	268	20950	10726	1820	3499	1230	172	1703	10162
Imidacloprid (138261-41-3)	366	285	6151	470	125	404	1269	1230	97	1068	1582	125	577	272
Clothianidin (210880-92-5)	227	252	1609	282	248	227	200	69	38	69	206	6	0	2
Thiamethoxam (153719-19-23)	17	0	2282	83	100	53	766	533	31	196	991	229	119	661

[a] Other terrestrial animals include other invertebrates, mollusks, and miscellaneous species.
[b] Terrestrial plants include three subsets: (1) moss and hornworts, (2) fungi, and (3) flowers, trees, shrubs, and ferns.
[c] Other aquatic animals include other invertebrates and mollusks.
[d] Aquatic plants include four subsets: (1) algae, (2) moss and hornworts, (3) fungi, and (4) flowers, trees, shrubs, and ferns.

Supplemental Table 12.2. Listed threatened and endangered animal species occurring within 12 U.S. Midwestern states accounting for 80% or more of planted corn and soybean acres (see Figure 12.2). Species are listed by species group and then by status. Species with populations with different statuses are listed twice. Additional species may occur within these regions that are protected by the Endangered Species Act wherever they are found. **Bolded** (^) species had at least 10 acres of perennial cover converted to corn or soy within 1 mile of—or intersecting—its federally designated critical habitat (calculated using a spatial overlay of conversion data from (2020) with USFWS critical habitat data). Data from USFWS (2019).

Scientific Name	Common Name	Species Group	Status	Estimated States of Occurrence
Cryptobranchus alleganiensis bishopi	Ozark hellbender	Amphibians	Endangered	MO
Charadrius melodus^	**piping plover***	**Birds**	**Endangered**	**IL, IN, MI, MN, OH, WI**
Grus americana^	**whooping crane***	**Birds**	**Endangered**	**KS, NE, ND, SD**
Numenius borealis	Eskimo curlew	Birds	Endangered	NE
Picoides borealis	red-cockaded woodpecker	Birds	Endangered	MO
Sterna antillarum	least tern*	Birds	Endangered	IA, IL, IN, KS, ND, NE, SD
Calidris canutus rufa	red knot*	Birds	Threatened	IN, KS, MI, MN, MO, ND, NE, OH, SD, WI
Charadrius melodus^	**piping plover***	**Birds**	**Threatened**	**IA, KS, ND, NE, SD**
Cumberlandia monodonta	spectaclecase (mussel)	Clams	Endangered	IA, IL, KS, MN, MO, WI
Cyprogenia stegaria	fanshell	Clams	Endangered	IL, IN, OH
Epioblasma florentina curtisii	Curtis' pearly mussel	Clams	Endangered	MO
Epioblasma obliquata	purple cat's paw pearly mussel	Clams	Endangered	OH
Epioblasma obliquata perobliqua	white cat's paw pearly mussel	Clams	Endangered	IN, OH
Epioblasma torulosa rangiana	northern riffleshell	Clams	Endangered	IL, IN, MI, OH
Epioblasma triquetra	snuffbox (mussel)	Clams	Endangered	IL, IN, MI, MN, MO, OH WI
Lampsilis abrupta	pink mucket (pearly mussel)	Clams	Endangered	IL, MO, OH
Lampsilis higginsii	Higgins' eye (pearly mussel)	Clams	Endangered	IA, IL, MN, MO, SD, WI
Lampsilis rafinesqueana^	**Neosho mucket**	**Clams**	**Endangered**	**KS, MO**
Leptodea leptodon	scaleshell (mussel)	Clams	Endangered	IL, MO, NE, SD
Plethobasus cooperianus	orangefoot pimpleback (pearly mussel)	Clams	Endangered	IL
Plethobasus cyphyus	sheepnose (mussel)	Clams	Endangered	IA, IL, IN, MN, MO, OH, WI
Pleurobema clava	clubshell	Clams	Endangered	IL, IN, MI, OH
Pleurobema plenum	rough pigtoe	Clams	Endangered	IN
Potamilus capax	fat pocketbook	Clams	Endangered	IL, IN, MO
Quadrula fragosa	winged mapleleaf	Clams	Endangered	MN, MO, WI
Villosa fabalis	rayed bean	Clams	Endangered	IN, MI, OH
Quadrula cylindrica cylindrica^	**rabbitsfoot**	**Clams**	**Threatened**	**IL, IN, KS, MO, OH**

Scientific Name	Common Name	Species Group	Status	Estimated States of Occurrence
Cambarus aculabrum	Benton County cave crayfish	Crustaceans	Endangered	MO
Gammarus acherondytes	Illinois cave amphipod	Crustaceans	Endangered	IL
Cottus specus	grotto sculpin	Fishes	Endangered	MO
***Notropis topeka*^**	**Topeka shiner**	**Fishes**	**Endangered**	**IA, KS, MN, MO, NE, SD**
Noturus trautmani	Scioto madtom	Fishes	Endangered	OH
Scaphirhynchus albus	pallid sturgeon	Fishes	Endangered	IA, IL, KS, MO, ND, NE, SD
Amblyopsis rosae	Ozark cavefish	Fishes	Threatened	MO
***Etheostoma nianguae*^**	**Niangua darter**	**Fishes**	**Threatened**	**MO**
***Notropis girardi*^**	**Arkansas River shiner***	**Fishes**	**Threatened**	**KS**
Noturus placidus	Neosho madtom	Fishes	Threatened	KS, MO
Bombus affinis	rusty-patched bumble bee*	Insects	Endangered	IA, IL, IN, MN, OH, WI
Brychius hungerfordi	Hungerford's crawling water beetle*	Insects	Endangered	MI
***Cicindela nevadica lincolniana*^**	**Salt Creek tiger beetle***	**Insects**	**Endangered**	**NE**
Lycaeides melissa samuelis	Karner blue butterfly	Insects	Endangered	IL, IN, MI, MN, OH, WI
Neonympha mitchellii	Mitchell's satyr butterfly*	Insects	Endangered	IN, MI, OH
Nicrophorus americanus	American burying beetle	Insects	Endangered	KS, NE, OH, SD
***Oarisma poweshiek*^**	**Poweshiek skipperling***	**Insects**	**Endangered**	**IA, MI, MN, ND, SD, WI**
***Somatochlora hineana*^**	**Hine's emerald dragonfly***	**Insects**	**Endangered**	**IL, MI, MO**
***Hesperia dacotae*^**	**Dakota skipper**	**Insects**	**Threatened**	**IA, MN, ND, SD**
***Canis lupus*^**	**gray wolf**	**Mammals**	**Endangered**	**MI, WI**
Corynorhinus townsendii ingens	Ozark big-eared bat	Mammals	Endangered	MO
Mustela nigripes	black-footed ferret	Mammals	Endangered	KS, SD
Myotis grisescens	gray bat	Mammals	Endangered	IL, IN, KS, MO
***Myotis sodalis*^**	**Indiana bat**	**Mammals**	**Endangered**	**IA, IL, IN, MI, MO, OH**
***Canis lupus*^**	**gray wolf**	**Mammals**	**Threatened**	**MN**
***Lynx canadensis*^**	**Canada lynx**	**Mammals**	**Threatened**	**MI, MN, WI**
Myotis septentrionalis	northern long-eared bat*	Mammals	Threatened	IA, IL, IN, KS, MI, MN, MO, ND, NE, OH, SD, WI
Nerodia erythrogaster neglecta	copperbelly water snake*	Reptiles	Threatened	IN, MI, OH
Sistrurus catenatus	eastern massasauga*	Reptiles	Threatened	IA, IL, IN, MI, OH, WI
Antrobia culveri	Tumbling Creek cavesnail	Snails	Endangered	MO
Discus macclintocki	Iowa pleistocene snail	Snails	Endangered	IA, IL, WI

*Species requires wetland habitats to complete at least part of its lifecycle, or uses wetlands for foraging, refuge, migrations, or alternative breeding/rearing habitat.

Supplemental Table 12.3. Listed threatened and endangered plant species occurring within 12 U.S. Midwestern states accounting for 80% or more of planted corn and soybean acres (see Figure 12.2). Species are listed by family and then by status. Additional species may occur within these regions that are protected by the Endangered Species Act wherever they are found. **Bolded** (^) species had at least 10 acres of perennial cover converted to corn or soy within 1 mile of—or intersecting—its federally designated critical habitat (calculated using a spatial overlay of conversion data from (2020) with USFWS 2020 critical habitat data). Data from USFWS (2019).

Scientific Name	Common Name	Family	Status	Estimated States of Occurrence	Habitat Type and Region
Asclepias meadii	Mead's milkweed	Asclepiadaceae	Threatened	IA, IL, IN, KS, MO, WI	Prairie
Asplenium scolopendrium var. americanum	American hart's-tongue fern	Aspleniaceae	Threatened	MI	Ravines in mixed hardwood forests
Solidago shortii	Short's goldenrod	Asteraceae	Endangered	IN	Grasslands
Boltonia decurrens	decurrent false aster*	Asteraceae	Threatened	IL, MO	River floodplains
Cirsium pitcheri	Pitcher's thistle	Asteraceae	Threatened	IL, IN, MI, WI	Sand dune shorelines in the Upper Great Lakes
Helenium virginicum	Virginia sneezeweed*	Asteraceae	Threatened	MO	Wet meadows in mountain highlands
Hymenoxys herbacea	lakeside daisy	Asteraceae	Threatened	IL, MI, OH	Limestone seeps in grasslands along the great lakes
Solidago houghtonii	Houghton's goldenrod	Asteraceae	Threatened	MI	Calcareous shores of the great lakes
***Physaria globosa*^**	**globe bladderpod**	**Brassicaceae**	**Endangered**	**IN**	**Limestone barrens**
Physaria filiformis	Missouri bladderpod	Brassicaceae	Threatened	MO	Limestone glades
Geocarpon minimum	tinytim	Caryophyllaceae	Threatened	MO	Sandstone slicks in grasslands
Rhodiola integrifolia ssp. leedyi	Leedy's roseroot	Crassulaceae	Threatened	MN, SD	Dolomite cliffs
Dalea foliosa	leafy prairie-clover*	Fabaceae	Endangered	IL	Glades and prairies with limestone
Trifolium stoloniferum	running buffalo clover	Fabaceae	Endangered	IN, MO, OH	Forested streambanks
Apios priceana	Price's potato-bean*	Fabaceae	Threatened	IL	Stream bottoms in mixed hardwoods
Lespedeza leptostachya	prairie bush-clover	Fabaceae	Threatened	IA, IL, MN, WI	Prairie
Oxytropis campestris var. chartacea	Fassett's locoweed	Fabaceae	Threatened	WI	Sandy lakeshores
Iris lacustris	dwarf lake iris*	Iridaceae	Threatened	MI, WI	Calcareous shores of the great lakes
Lindera melissifolia	pondberry*	Lauraceae	Endangered	MO	Bottomland hardwood wetlands
Erythronium propullans	Minnesota dwarf trout lily*	Liliaceae	Endangered	MN	Forested hardwood slopes and floodplains

Scientific Name	Common Name	Family	Status	Estimated States of Occurrence	Habitat Type and Region
Gaura neomexicana var. coloradensis	Colorado butterfly plant*	Onagraceae	Threatened	NE	Wet grasslands on the high plains
Isotria medeoloides	small whorled pogonia*	Orchidaceae	Threatened	IL, MI, MO, OH	Forested streambanks
Platanthera leucophaea	eastern prairie fringed orchid*	Orchidaceae	Threatened	IA, IL, IN, MI, MO, OH, WI	Prairie
Platanthera praeclara	western prairie fringed orchid*	Orchidaceae	Threatened	IA, KS, MN, MO, ND, SD	Prairie
Spiranthes diluvialis	Ute ladies'-tresses*	Orchidaceae	Threatened	NE	Wet meadows
Aconitum noveboracense	northern wild monkshood	Ranunculaceae	Threatened	IA, OH, WI	Shaded cliffs and streamsides in the Driftless Area
Spiraea virginiana	Virginia spiraea	Rosaceae	Threatened	OH	Forested streambanks
Mimulus michiganensis	Michigan monkey-flower*	Scrophulariaceae	Endangered	MI	Calcareous shores of the great lakes
Penstemon haydenii	blowout penstemon	Scrophulariaceae	Endangered	NE	Sand dunes

*Obligate or facultative wetland species

13. Aquatic Ecosystem Health and Biodiversity

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Key Findings

- Many watersheds in the Midwest and other U.S. regions have historically been impacted by agriculture generally and by crops used for biofuels specifically, but the incremental effect from recent (2008–2016) agricultural expansion from all causes, including any potential impact from the RFS Program specifically, appears to be minor in comparison.
- Water demand for feedstock production may change stream flow and alter flow patterns that are important for supporting fish diversity.
- Pesticides used in feedstock production including atrazine, glyphosate, and neonicotinoids, have direct toxicity to some nontarget organisms as well as a variety of sublethal, indirect environmental effects on aquatic ecosystem health and biodiversity. Based on overlap of species ranges and critical habitat with atrazine usage, EPA judged atrazine was likely to adversely affect 180 out of 207 federally listed (i.e., threatened and endangered) aquatic invertebrate species assessed, including mussels, snails, shrimp, amphipods, water beetles, and crayfish.
- Based on data from nationally representative surveys of the nation's wadeable stream miles in 2004 and about 10 years later in 2013–2014, biological and nutrient conditions worsened in the ecoregions roughly coinciding with areas of corn and soybean production compared to the rest of the continental United States. National surveys found that benthic macroinvertebrates were nearly twice as likely to be in poor condition in waterbodies with high nutrient concentrations and/or excess sediments.
- For the scenarios examined in a recent modeling study on agricultural expansion due to all causes from 2008–2016, the flow-weighted nutrient concentrations increased by less than 5% on average across the Missouri River Basin (MORB). For the scenario of conversion from grassland to corn/soy rotation, only 0.11% of watersheds in the MORB had increases in nutrient concentrations that were more than 10% of the baseline scenario. Given the RFS Program may have impacted corn planting by 3.5 million acres or less in 2016 (refer to Chapter 6), increases in nutrient concentrations that may be attributable to the RFS Program are unlikely to result in *new exceedances* of current state numeric nutrient criteria in agricultural regions of the United States, such as the MORB. Total effects may be larger or smaller because this study only included effects from agricultural expansion (expected to be the largest source) and not agricultural intensification or recent improvements in tillage practices.

- Demand for biofuel feedstocks may contribute to increased frequency and magnitude of harmful algal blooms and hypoxia. Altered food webs and changes in nutrient cycling can trigger feedback loops that make it difficult to prevent or mitigate the effects of harmful algal blooms and hypoxia on aquatic ecosystems.
- Adoption and expansion of sustainable conservation practices and technologies remain critically important to reducing impacts on aquatic ecosystems by restoring flow and decreasing loads of nutrients, sediment, and pesticides to levels that are less harmful to aquatic organisms.
- The likely future effects of the RFS Program are highly uncertain. While the projected cropland expansion for 2023–2025 is slightly larger than the estimated historical cropland expansion from the RFS Program, the authors cannot say with reasonable certainty that any particular aquatic ecosystem will be impacted due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes.[2]

Chapter terms: algae, benthic invertebrates, cyanobacteria, harmful algal blooms (HABs), hypoxia, macroinvertebrates, sediment, sedimentation, zooplankton

13.1 Overview

13.1.1 Background

The second triennial Report to Congress on biofuels (i.e., the "RtC2") did not include a standalone chapter on ecosystem health and biodiversity specifically about aquatic organisms. Aquatic organisms are important because they contribute to ecosystem services and indicate whether waterbodies can support designated uses such as recreation and aquatic life. This chapter focuses on organisms that live at least some part of their lifecycle in aquatic ecosystems. Many aquatic ecosystems, both small and large, exist in watersheds that are also used for agriculture. Smaller headwater streams have an intimate connection with the land and are direct recipients of pollutants originating from their watersheds. Downstream rivers, larger lakes, and coastal systems more typically receive pollutant loads from multiple tributaries draining upstream lands. In addition to water quality impacts, the practices of corn and soy production change the flow and quantity of water delivered to downstream systems.

[2] On August 3, 2023, EPA completed its Endangered Species Act informal consultation on the Renewable Fuel Standard (RFS) Program: Standards for 2023-2025 and Other Changes rulemaking (also known as the RFS "Set Rule"). With the Biological Evaluation that EPA submitted to the National Marine Fisheries Service (NMFS) and Fish and Wildlife Service (FWS) on May 19, 2023, EPA determined that the RFS Set Rule is not likely to adversely affect listed species and their designated critical habitats. EPA received letters of concurrence with this determination from NMFS on July 27, 2023, and from FWS on August 3, 2023, thereby concluding the consultation.

Production of the primary domestic biofuel feedstocks, corn and soy, contributes to direct and indirect effects on water quantity and water quality, with both local and downstream potential impacts on aquatic life. There are no known negative effects on aquatic habitats in the United States from the other two biofuels that are the focus of the RtC3 (i.e., fats, oils, and greases [FOGs] and Brazilian sugarcane). Diversion of FOGs for biofuels may improve aquatic habitats in that they decrease the potential for clogging in water infrastructure, which contributes to combined sewage overflows (CSOs)[3]. Effects in Brazil from the cultivation of sugarcane for biofuels are discussed in Chapter 16. The remainder of this chapter focuses on effects on aquatic ecosystems from the production and use of corn for ethanol and soybean for biodiesel.

Section 304(a)(1) of the Clean Water Act requires EPA to develop and publish nationally recommended criteria for water quality based on scientific knowledge of the environmental effects on a water body's designated uses, such as aquatic life support. Although the Clean Water Act does not define "aquatic life," it states that water quality should provide for the protection and propagation of "fish, shellfish and wildlife." EPA recommends deriving aquatic life use criteria from data on toxicity to several categories of organisms typically present in waterbodies, including fish and other aquatic vertebrates, invertebrates such as crustaceans, insects, and mollusks, and at least one alga or vascular plant species (U.S. EPA, 1985). This chapter includes discussion of these categories of aquatic organisms used to assess the biological integrity of waterbodies for EPA to meet the requirements of the Clean Water Act. This chapter also addresses harmful algal blooms (HABs) and hypoxia because they are significant environmental effects related to biofuel feedstock production.

13.1.2 Drivers of Change

The drivers of change discussed in this chapter are related to corn and soybean, the major biofuel feedstocks produced in the United States (see Chapter 2, Tables 2.1 and 2.2). The production of corn and soy contributes to environmental effects in aquatic ecosystems, primarily by direct or indirect release of pesticides, nutrients, and sediments during different biofuel production phases (e.g., upstream feedstock production, biofuel production, transportation) (U.S. EPA, 2003), as well as alterations to stream flow (McCarthy and Johnson, 2009). EPA's National Aquatic Resource Surveys (NARS) assess the condition of the nation's freshwater and coastal ecosystems. The first national survey by NARS was the Wadeable Streams Assessment (WSA) in 2000–2004 (U.S. EPA, 2006), which included information about the

[3] CSOs occur when runoff, domestic sewage, and industrial wastewater exceed the capacity of a sewer system or wastewater treatment plant (e.g., during heavy rainfall events or snowmelt). When CSOs occur, the untreated stormwater and wastewater transports waste, debris, and potentially toxic materials directly into nearby streams, rivers, and other waterbodies. CSOs are among the major sources responsible for beach closings, shellfishing restrictions, and other water body impairments for nearly 860 municipalities across the United States.

biological condition of macroinvertebrates and physical condition of fish habitat in the nation's freshwater streams prior to the RFS Program and growth in the biofuels industry. Biological condition of fish was added to the national survey program for the National Rivers and Streams Assessment (NRSA) in 2008–2009 (U.S. EPA, 2016c). The latest findings from NARS are available from NRSA 2013–2014 (U.S. EPA, 2019a), the National Lakes Assessment (NLA) 2012 (U.S. EPA, 2016b), and the National Coastal Condition Assessment (NCCA) 2010 (U.S. EPA, 2016a). Findings from NARS include the condition of waterbodies related to excess nutrients (refer to Chapter 10), biological condition of fish and benthic macroinvertebrates, condition of instream fish habitat,[4] fish tissue contaminants, and sediment contaminants and toxicity. These studies are not designed to estimate the causes of these changes in condition and aquatic habitat except generally and in the aggregate (e.g., correlation with agriculture and other human impacts). This chapter uses NARS data and additional datasets, such as those from the U.S. Geological Survey and the scientific literature, to elucidate trends in aquatic ecosystem health and biodiversity over time, and the potential correlations between observed changes and changes in agriculture due to increases in biofuel volumes.

13.1.3 Relationship with Other Chapters

Aquatic ecosystems in this chapter include streams, rivers, lakes, and coastal zones. Wetlands are addressed separately in the report (Chapter 14). The terms ecosystem health and biodiversity are introduced in the Terrestrial Ecosystem Health and Biodiversity chapter (Chapter 12) and not repeated here. This chapter applies the results of new analyses described in Chapter 10 (section 10.3.2) to discuss potential effects on aquatic organisms of shifting crop cultivation practices.

13.1.4 Roadmap for the Chapter

Overall, the organization of this chapter is similar to that of Chapter 12 on terrestrial ecosystems with the following differences. Conclusions from the RtC2 presented in section 13.2 are select statements related to aquatic ecosystem health and biodiversity. Section 13.3 is about impacts to date based on the available literature and data. The subsections of section 13.3 provide background information about the major stressors (flow, pesticides, nutrients, and sediment) associated with biofuel production and agricultural land use in general, the environmental effects of these stressors on biota (fish, invertebrates, aquatic plants, algae, and other organisms), specific sections dedicated to HABs and hypoxia, attribution of the environmental effects to the RFS Program, and opportunities for offsetting negative effects and promoting positive effects of biofuel production. The remainder of the subsections are similar to other chapters, including a discussion of likely future effects (section 13.4), comparisons with petroleum

[4] Instream fish habitat refers to the areas fish need for concealment and feeding. These areas include large wood within the stream banks, boulders, undercut banks, and tree roots.

(section 13.5), and horizon scanning for potential future issues and effects from biofuels (section 13.6). Section 13.7 provides a synthesis, recommendations, and conclusions.

13.2 Conclusions from the 2018 Report to Congress

The RtC2's conclusions about ecosystem health and biodiversity were about biofuels in general. The overall conclusions relevant to aquatic organisms from the 2018 report were:

- Increased fertilizer applications of nitrogen (N) for corn and phosphorus (P) for corn and soy have known negative effects on aquatic biodiversity.
- Continued adoption and expansion of sustainable conservation practices and technologies are expected to decrease nutrient loadings and associated adverse impacts.
- Loss of habitat and landscape simplification are associated with negative impacts to ecosystem services in aquatic habitats.
- Changes in hydrologic and sediment generation dynamics through land use change—mainly conversion to row crops—may extirpate native mussel populations.
- Aquatic invertebrates were correlated with the greatest risk from imidacloprid, a pesticide used in corn and soy cultivation.
- The pesticide atrazine, 80% of which is used in corn cultivation, is moderately toxic to freshwater and estuarine/marine fish, highly toxic to freshwater aquatic invertebrates, and even more toxic to estuarine/marine aquatic invertebrates.
- Demand for biofuel feedstocks may contribute to HABs, as recently observed in western Lake Erie, and to hypoxia, as observed in the northern Gulf of Mexico.

13.3 Impacts to Date for the Primary Biofuels

13.3.1 Literature Review

This in-depth literature review is intended to provide information specific to aquatic ecosystems that was not detailed in previous reports. This section reviews the scientific literature on the effects of biofuel feedstock production on aquatic ecosystems by taxonomic category. Specifically, it discusses effects on fish, invertebrates, aquatic plants, algae, and other organisms (sections 13.3.1.1 through 13.3.1.3). It also discusses HABs and hypoxia (sections 13.3.1.4 through 13.3.1.6). Each taxonomic category is addressed by the major stressors (flow, nutrients, sediment, and pesticides) associated with agricultural land use in general. For changes in these stressors, see Chapter 10 (for nutrients, sediment, and pesticides) and 11 (for flow). Disturbances in water quantity (e.g., stream flow) and water quality (e.g., nutrients, sediment, pesticides) can have deleterious effects on aquatic ecosystems, including impacts on the designated uses of waterbodies because of biodiversity loss or alterations in aquatic species composition (Figure 13.1). Notably, the literature was not specific enough in most cases to address the effects of corn and soybeans grown for biofuels, let alone any potential changes from the RFS Program, and instead addresses the general effects of corn and soybean cultivation (but see sections 13.3.2 and 13.3.3). The summary below reflects the assumption that land management of corn and soybeans is the same regardless of end use, whether for food, feed, or biofuel feedstock.

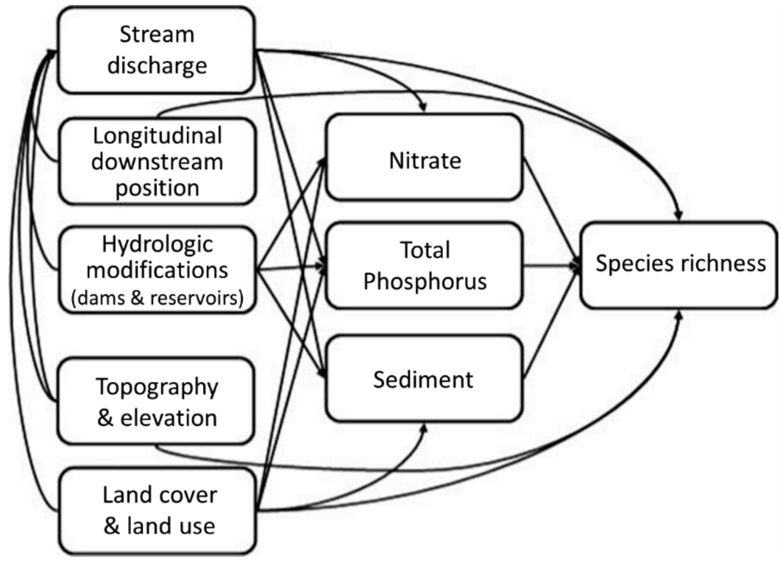


Figure 13.1. Conceptual diagram from Schweizer and Jager (2011) (used with permission). The diagram shows the combined influences of hydrology, land cover, and water quality on native fish species richness.

River and stream flows—including flow magnitude, timing, frequency, duration, and rate of change (Poff and Zimmerman, 2009)—are vitally important for maintaining habitat conditions that support sensitive aquatic plants and animals. For example, the Ozark hellbender (*Cryptobranchus alleganiensis bishopi*) is a federally endangered amphibian species that requires well-oxygenated, running waters because it breathes through its skin and requires large river rocks for shelter (Fobes, 1995). Hydrologic changes can make habitat unsuitable for such species. The ecological value of running waters is thus vulnerable to anthropogenic activities that disturb flow, such as water withdrawal for irrigation (Xenopoulos and Lodge, 2006; Vörösmarty and Sahagian, 2000) or the use of subsurface tile drainage (Schilling and Libra, 2003). The way water flows through riverine networks determines much of the large-scale biodiversity patterns seen in aquatic communities (Schweizer and Jager, 2011; Muneepeerakul et al., 2008; Oberdorff et al., 1995). Agricultural practices can vary in terms of their impact on flow regime and corresponding impact on freshwater ecosystems, depending on the extent of associated practices like water extraction, dam storage, diversions, and dredging and clearing of riparian vegetation (Rideout et al., 2022). While increased flow (e.g., from landscape modification, tile drainage) may result in excess streambank erosion, sediment and pollutant transport (Blann et al., 2009), decreases in flow may intensify the impacts of excess nutrients and other anthropogenic pollutants on aquatic biodiversity (Acharyya et al., 2012). Grassland conversion to continuous corn production modeled using SWAT in the Missouri River Basin (MORB) resulted in a small (<1%) increase in mean annual change in flow (Chen et al., 2021); a watershed-scale study using similar modeling methods found up to 12% increases in stream flow in some months compared to baseline crop production, potentially associated with genetic and agronomic improvements in crop traits (Ren et al., 2022). Changes in flow can reduce biodiversity, alter lifecycles, and cause mortality in aquatic organisms (Poff and Zimmerman, 2009; Bunn and Arthington, 2002).

Deleterious effects on aquatic communities, including the loss of biodiversity (Hillebrand and Sommer, 2000; Carpenter et al., 1998), have been linked to the large amounts of plant nutrients (i.e., N and P) applied to land as fertilizer that reach aquatic ecosystems through leaching and/or tile drainage from agricultural areas to downstream waterbodies. Feedstock production areas roughly coincide with the ecoregions of the Temperate Plains/Northern Plains/Upper Midwest (TPL/NPL/UMW, Figure 13.2). The input of excess nutrients (both N and P) into lakes, reservoirs, and impoundments can result in algal blooms (Paerl et al., 2016), some of which can produce toxins or accumulate excessive biomass resulting in HABs. As the algal biomass decomposes, the water becomes depleted of oxygen and the hypoxic conditions make the habitat unsuitable for fish and other aquatic organisms.

Another important category of agriculture-derived water contamination is excess sediment. Each year in the United States, cropland produces about 6 metric tons or more of eroded soil per hectare (Nearing et al., 2017), of which about 60% is estimated to reach streams and rivers (Pimentel, 2006); however, it is important to note that these estimates vary widely year to year because of many factors, including weather conditions and tillage. Areas with higher crop production modeled within the Missouri River Basin produced more sediment (Chen et al., 2021). Suspended sediment reduces light penetration, reducing photosynthesis by primary producers. Nutrients and toxins sorbed to sediment from terrestrial ecosystems can enter waterbodies in runoff. Clay and organic material from sediment often form associations with bacteria, and such associations are typically high in N and P (Weisse, 2003; Rothhaupt, 1992). Sediments can also transport organic carbon and fuel heterotrophic microbial growth, leading to changes in aquatic species composition

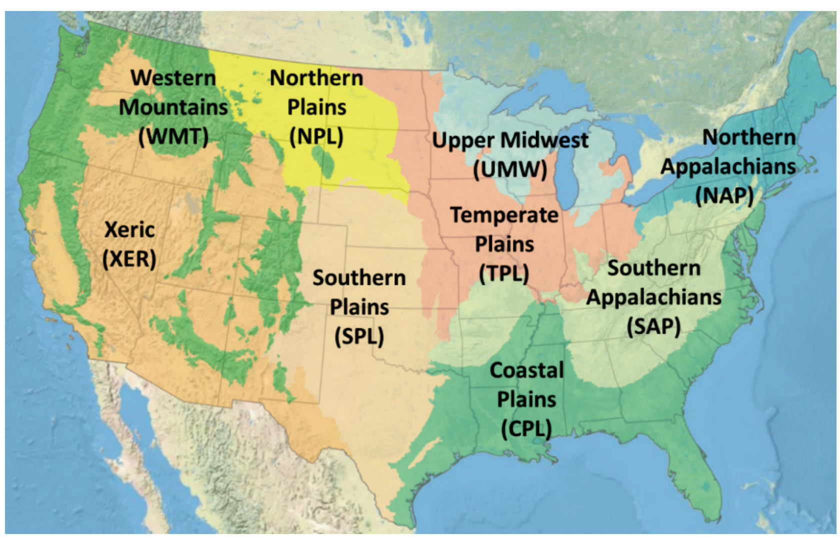


Figure 13.2. Ecoregions and their abbreviations. Modified from U.S. EPA (2016c).

[e.g., (Lind et al., 1997; Cuker and Hudson, 1992)]. In addition to chemical and biological effects, the physical effect of sediments filling interstitial spaces in streambeds can disturb important habitat for some organisms. Sedimentation generally leads to loss of biodiversity because of decreased habitat complexity (Balata et al., 2007).

A third important category of anthropogenic pollutants is agricultural pesticides (e.g., herbicides, insecticides, fungicides). As noted in Chapter 3 (section 3.2.1.5), there is a wide range of pesticides used currently or historically on corn and soybean since the beginning of the RFS Program (Tables 3.4 and 3.5), including glyphosate, atrazine, mesotrione, metolachlor, dicamba, and others. These various compounds fall into a number of chemical classes, possess a range of environmental fate properties (i.e., mobility, persistence), and operate via a wide range of modes-of-action against target and nontarget organisms in various classes (e.g., fish, invertebrates, plants). A full review of all these chemicals relevant for biofuel feedstocks is beyond the scope of this report. Furthermore, generalizations concerning their collective impacts on aquatic ecosystems are difficult to draw, but broad patterns may be observed. The usage of agricultural pesticides leads to deleterious effects on aquatic organisms and aquatic ecosystems, as pesticide residues are transported from the point of application to nearby waterbodies via runoff, leaching/tile drainage, spray drift, and other transport mechanisms. Studies show detections in surface water of agricultural pesticides at concentrations above those that are toxic to aquatic organisms in laboratory studies (see Chapter 10, section 10.3.1.4). In addition to direct toxic impacts, herbicides like glyphosate can contribute to nutrient pollution because more than 18% of glyphosate acid by mass is P. The salt form of glyphosate commonly used in herbicide formulations rapidly degrades in water and can release simpler P compounds easily used by organisms for growth (Hébert et al., 2019). P loading from glyphosate and other pesticides, though small relative to P from fertilizers [i.e., <2%, (Sabo et al., 2021)], has now reached levels in aquatic systems near levels of P derived from detergents prior to legislation banning these products, in part because of negative impacts of excess P on aquatic life (Hébert et al., 2019).

Among the noteworthy insecticides widely employed in corn and soybean production are fipronil and three neonicotinoids: imidacloprid, thiamethoxam, and clothianidin. Both fipronil and neonicotinoids are considered systemic insecticides because they have broad-spectrum toxicity (Gibbons et al., 2015). Another noteworthy class of insecticides used in corn and soybean production is the pyrethrins and pyrethroids. In contrast with neonicotinoids, pyrethrins and pyrethroids are highly hydrophobic compounds that tend to be more resistant to flushing by streams and rivers because they partition to organic matter in sediments (U.S. EPA, 2016e). Pyrethrins are natural insecticides derived from chrysanthemum flowers, and pyrethroids are synthetic compounds that are similar in structure to pyrethrins. Compounds in this class are neurotoxins that act by interfering with voltage-gated ion channels in the neurons of vertebrates and invertebrates. In the aquatic environment, fish may be particularly susceptible to this class of compounds (section 13.3.1.1.4).

A new study found that an EPA chronic aquatic-life benchmark was exceeded at least once at more than half of the stream sites sampled in every region of the United States—Midwest, South, Northeast, West, and Pacific—from 2013 to 2017 (Stackpoole et al., 2021). Benchmark exceedances indicate the potential for harmful effects to aquatic life such as fish, algae, and invertebrates like aquatic insects. However, an EPA human-health benchmark was exceeded only four times (1.1% of samples). Of the 221 pesticides measured, just 17 were responsible for the aquatic-life benchmark exceedances. Many of these 17 were herbicides, which frequently occurred at relatively high concentrations that exceeded benchmarks for fish, invertebrates, and plants. Others were insecticides, which occurred at lower concentrations, but are much more toxic to aquatic invertebrates than herbicides. One of the insecticides, the neonicotinoid imidacloprid posed the greatest potential threat to aquatic life with a total of 245 benchmark exceedances at 60 of the 74 sites (Stackpoole et al., 2021).

13.3.1.1 Fish

13.3.1.1.1 Environmental Effects of Flow Alterations

Fish species in flowing waters are vulnerable to anthropogenic disturbances including water withdrawal for irrigation (Xenopoulos and Lodge, 2006; Vörösmarty and Sahagian, 2000). The structure of river networks, and the manner in which water flows through these networks, can explain much of the large-scale biodiversity patterns in freshwater fishes despite complex local and basin-scale habitat heterogeneity (Muneepeerakul et al., 2008). For instance, flow was the dominant driver of fish species richness in the Arkansas-White-Red River Basin (Schweizer and Jager, 2011). Fish communities have been predicted to be impacted when maximum flows fall below 40% of expected natural flow magnitudes (Carlisle et al., 2011). Fish and other aquatic organisms that have life history strategies in response to natural flow regimes and the habitat conditions created by the flow patterns are vulnerable to flow alterations and habitat homogenization associated with agricultural drainage (Blann et al., 2009). Changes in stream flow magnitude and patterns interact with other impacts of agriculture, such as erosion, which could negatively impact fish with reproduction strategies that depend on well-oxygenated spaces in the bottom of streams if those spaces become clogged with sediment after heavy rain events as seen in some cases of erosion-prone corn production in Europe(Mueller et al., 2020).

13.3.1.1.2 Environmental Effects of Nutrients

As of 2014, fish in the TPL ecoregion were in good biological condition in 28% of river and stream miles and poor biological condition in 34% (U.S. EPA, 2019a). Between an earlier (2008–2009) and later (2013–2014) time period, river and stream miles across the continental United States rated good for fish condition decreased by approximately 10% in the NRSA study; in the TPL/NPL/UMW ecoregions, there was a more dramatic decrease of about 17% [Figure 13.3, (U.S. EPA, 2019a)]. Fish multi-metric index (MMI) condition was not assessed by EPA before 2008, so the latest fish MMI condition cannot be compared to the WSA 2000–2004 coinciding with the period before the RFS and major growth in the biofuels industry.[5] However, WSA 2000–2004 did assess the physical characteristics of streams related to fish habitat condition, which often becomes degraded by sedimentation (see section 13.3.1.1.3). A different dataset from NatureServe showed that by 2014, the greatest reduction since before 1970 in number of fish species occurred in portions of the Midwest and the Great Lakes, where several watersheds have lost more than 20 species previously known to occur in those locations (U.S. EPA, 2019b). The reduction in fish species may be causally related to high levels of agricultural land use in the Midwest. However, the effects of land use on fish may be indirect and not necessarily explained by nutrient pollution from fertilizer applications. Schweizer and Jager (2011) showed when models accounted for river discharge, fish species richness in the Arkansas-White-Red River Basin had a positive correlation with mean annual total phosphorus (TP) concentrations. By contrast with TP, concentrations of nitrate nitrogen exhibited negative correlation with fish species richness (Schweizer and Jager, 2011).

[5] The fish MMI from NRSA includes variables on water quality (e.g. TN, TP), physical habitat (e.g. riparian cover), and substrate type (e.g., % sand) (U.S. EPA, 2019a).

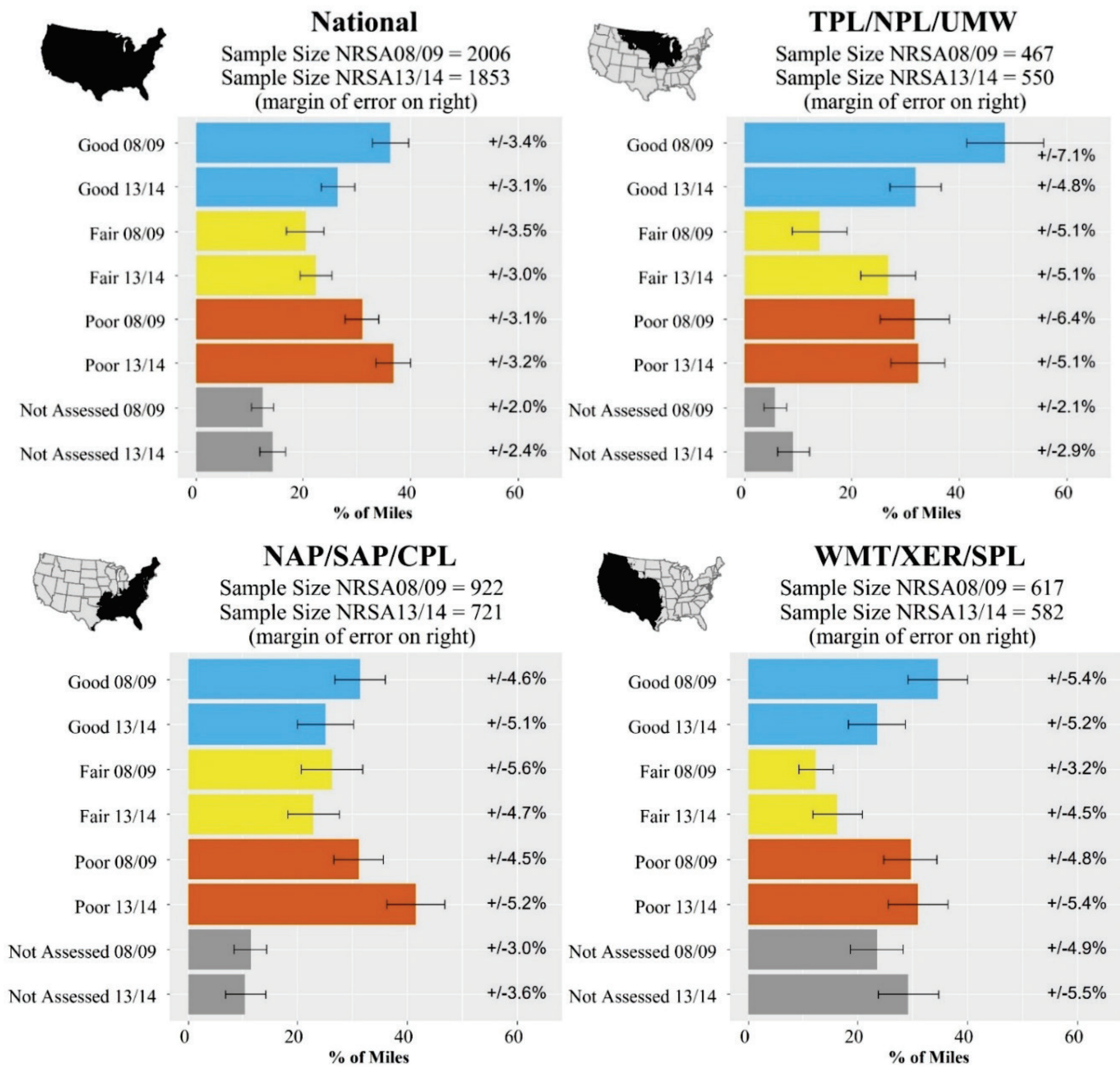


Figure 13.3. Fish Multi-Metric Index (MMI) condition in rivers across the conterminous United States (a) and select ecoregions (b–d). The % of Miles refers to the total river and stream miles surveyed by EPA. The condition categories (Good, Fair, and Poor) are relative to the least disturbed rivers and streams. The numbers "08/09" and "13/14" refer to the surveys completed in 2008–2009 and 2013–2014. Data from U.S. EPA (2019a, 2016c). Refer to Figure 13.2 for ecoregion abbreviations. Fish MMI condition was not assessed in WSA 2000–2004.

13.3.1.1.3 Environmental Effects of Sediments

Different trophic classes or life stages of fish have different responses to increases in sediment, but negative effects on fish occur when levels of sediment exceed normal ranges (Utne-Palm, 2002). High sediment loads lead to clogged gills (Bruton, 1985), reduced habitat for spawning (Chapman, 1988), modified fish migration patterns (Alabaster and Lloyd, 1982), reduced availability of food (Doeg and Koehn, 1994; Gregory and Northcote, 1993; Bruton, 1985), and decreased foraging efficiency particularly for fish that rely on visual cues (Ryan, 1991; Bruton, 1985). Conversely, turbid waters may provide refugia from predators for some species of fish (De Robertis et al., 2003; Gregory and Northcote, 1993). The abundance and assemblage composition of fish can change in response to degradation of fish habitat because of increased sedimentation from erosion of agricultural land (Berkman and Rabeni, 1987). Between 2000–2004 (from the WSA) and 2013–2014 (from NRSA), the physical characteristics of streams related to fish habitat condition slightly deteriorated in the United States with strong regional variation (Figure 13.4). The percent of miles in good fish habitat decreased nationally by 4.7% (±4.8%) while poor fish habitat increased by 4.7% (±3.2%). The percent of miles in good fish habitat improved in the Upper Midwest (Figure 13.4d) and deteriorated in the Coastal Plains (Figure 13.4g) and showed no significant trend at the ecoregion level elsewhere. These trends in physical conditions for fish were not as pronounced as those in biological condition of fish associated with excess nutrients as discussed in section 13.3.1.1.2.

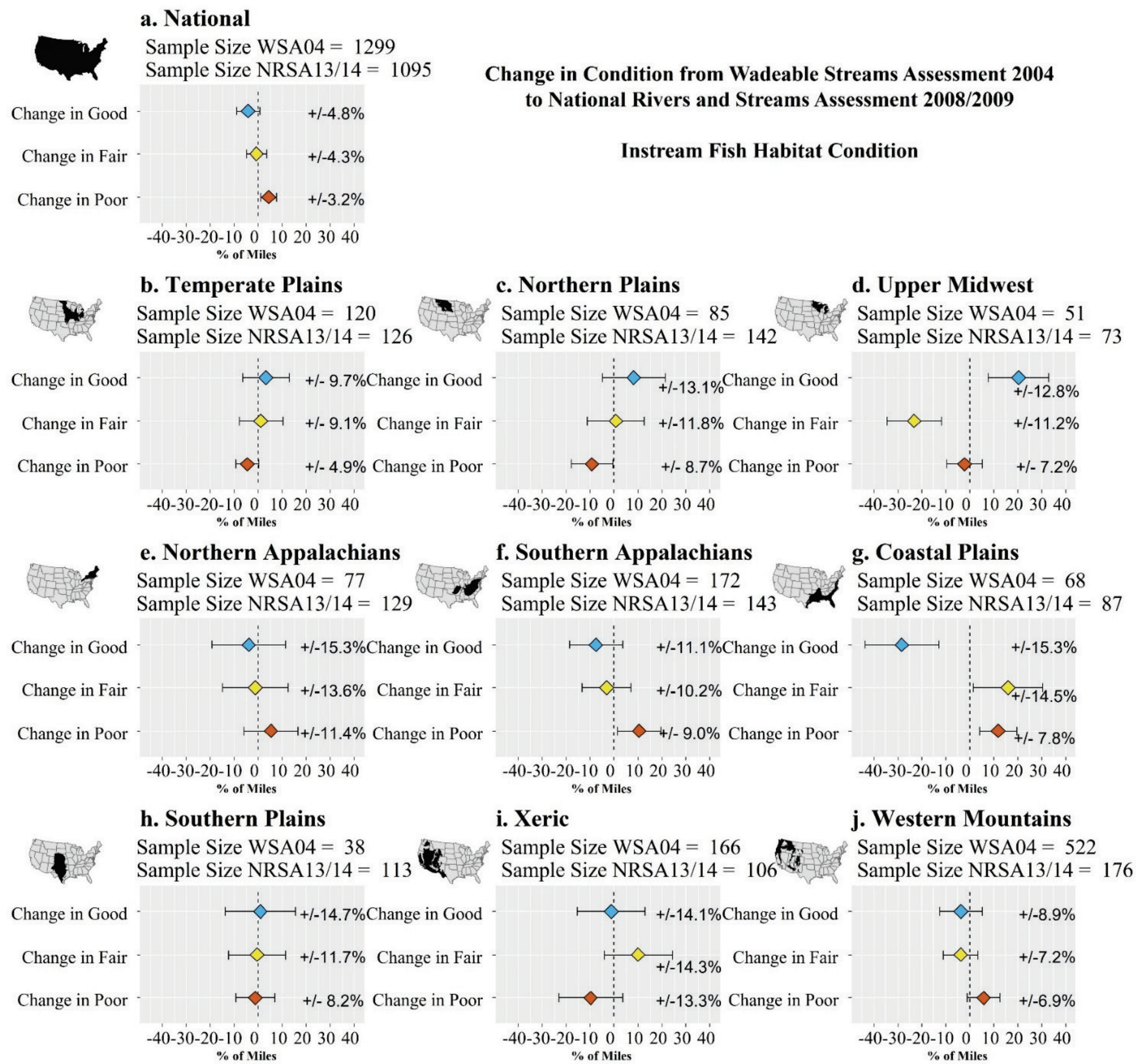


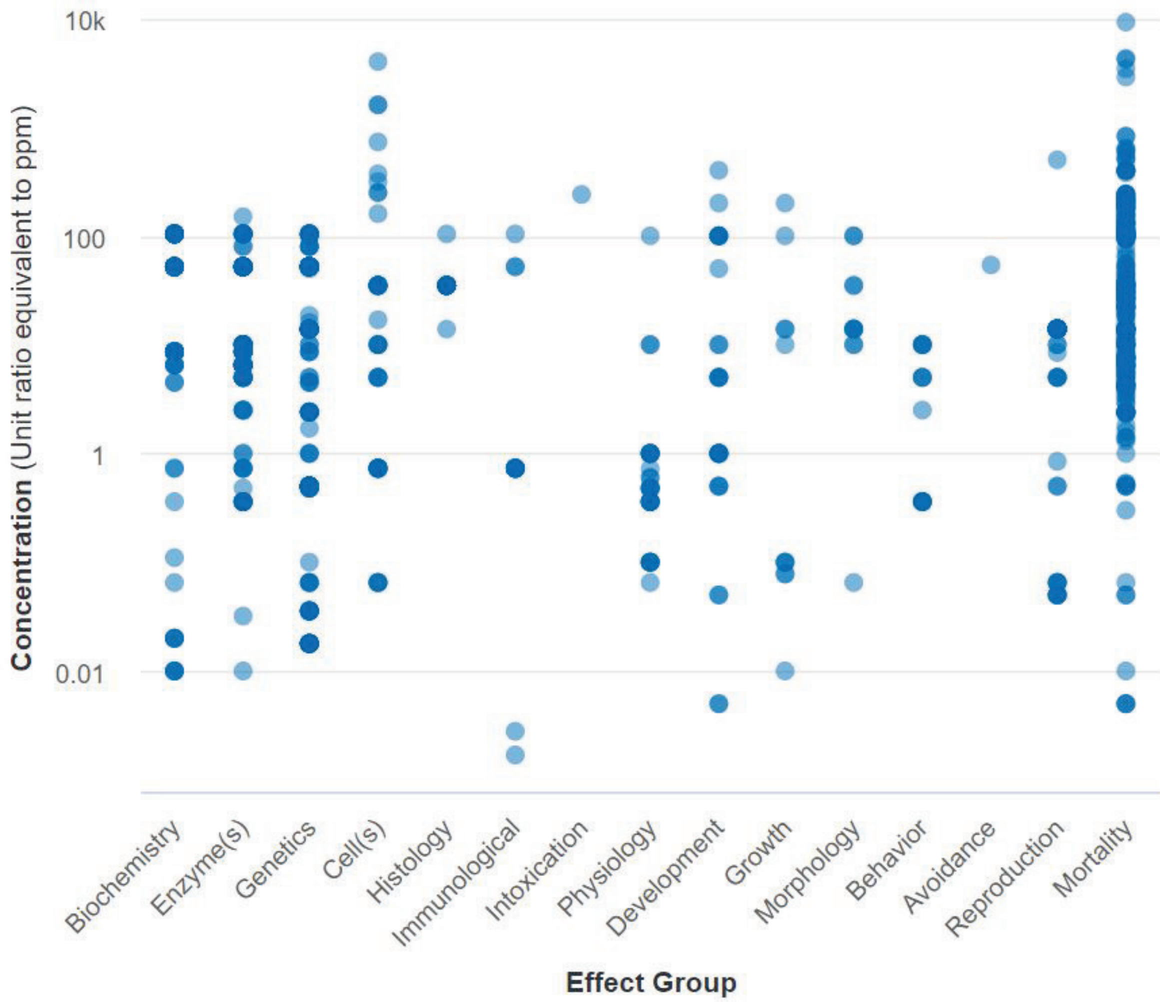
Figure 13.4. Instream fish habitat condition related to the physical characteristics of wadeable streams across the conterminous United States (a) and ecoregions (b–j). The % of Miles refers to the total river and stream miles surveyed by EPA. The condition categories (Good, Fair, and Poor) are relative to the least disturbed streams in each ecoregion. "WSA04" refers to the Wadeable Streams Assessment conducted in 2000–2004 and "NRSA13/14" refers to the National Rivers and Streams Assessment conducted during 2013–2014. Data from (data from U.S. EPA, 2019a; U.S. EPA, 2006). Note that fish MMI is not available in the WSA which focused on physical habitat conditions that are less sensitive to agricultural effects. The error bars and percentages on the right of each plot are margins of error.

13.3.1.1.4 Environmental Effects of Pesticides

Laboratory studies on fish exposed to glyphosate residues have documented mainly sublethal effects, such as DNA damage in organ tissues after exposure to 116 micrograms per liter (μg/L) and altered muscle and brain function with exposure to 10 milligrams per liter (mg/L) of glyphosate-based herbicide (Guilherme et al., 2012; Modesto and Martinez, 2010). Based on hundreds of toxicity studies, over 20 years of monitoring data, and aquatic exposure models, EPA concluded that in areas where atrazine use is heaviest, there is potential chronic risk to fish, and fish exposed for several weeks to 5 µg/L average concentration of atrazine are predicted to suffer reproductive impacts (U.S. EPA, 2016b). Based on overlap of federally listed (i.e., threatened and endangered) species ranges and critical habitat with areas affected by atrazine usage, runoff, and spray drift, atrazine is likely to adversely affect 170 out of 190 fish assessed, including 90 fish species with strong evidence of a likelihood to adversely affect (U.S. EPA, 2021). Among aquatic organisms, fish may be particularly susceptible to pyrethrins and pyrethroids. For example, the pyrethroid lambda-cyhalothrin is classified as very highly toxic to freshwater fish, with a 96-hour median lethal concentration (LC50) of just 0.078 µg/L in golden orfes (*Leuciscus idus*) (U.S. EPA, 2010). Pyrethrins and pyrethroids are relatively hydrophobic compounds that tend to partition to stream sediments and may bioconcentrate in the tissues of fish, although bioconcentration factors in fish are not as high as would be expected based on their octanol-water partition coefficients (U.S. EPA, 2016e).

Through ecological risk assessments (ERAs), EPA has established thresholds of effect from top corn and soybean pesticides on various ecological end points, including fish and aquatic invertebrate acute and chronic end points (Supplemental Table 13.1), various other aquatic-life benchmarks (Supplemental Table 13.2), and summarized key biophysical properties (Supplemental Table 13.3). Related to this, the EPA ECOTOX Knowledge Database (https://cfpub.epa.gov/ecotox/help.cfm) compiles information on the toxicity of pesticides and other chemicals to terrestrial and aquatic organisms in various taxonomic categories, derived predominantly from the peer-reviewed literature. For fish, ECOTOX includes 4,291, 2,822, 248, 103, and 10,726 records for the common pesticides used on biofuel feedstocks (corn and soybean), including the herbicides glyphosate, atrazine, acetochlor, metolachlor, 2,4-D, respectively, along with 97, 38, and 31 records for the neonicotinoid insecticides imidacloprid, clothianidin, and thiamethoxam, respectively (refer to Chapter 12, Supplemental Table 12.1). An example of the range of effects from glyphosate on fish is shown in Figure 13.5. It is beyond the scope of this report to summarize the ecological risks for all these chemicals used in corn and soybean cultivation for fish or other taxonomic groups (e.g., sections 13.3.1.1.4, 13.3.1.2.4, 13.3.1.3.4), which are officially addressed in ERAs conducted by the EPA's Office of Chemical Safety and Pollution Prevention (OCSPP) under the authority of the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), the Food Quality Protection Act (FQAPA), and the Pesticide Registration Improvement Extension Act (PRIA 4).[6]

[6] For a general overview see: https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/factsheet-ecological-risk-assessment-pesticides. For a more technical description see: https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/ecological-risk-assessment-pesticides-technical.



ppm = parts per million

Figure 13.5. Overview of the concentration of glyphosate that affects 15 different effect groups for fish. Data from the EPA ECOTOX database (https://cfpub.epa.gov/ecotox/help.cfm).

13.3.1.2 Aquatic Invertebrates

13.3.1.2.1 Environmental Effects of Flow Alterations

Human-induced alteration of the natural flow regime can degrade a stream's physical and chemical properties, leading to loss of aquatic life and reduced aquatic biodiversity, including for aquatic invertebrates (Novak et al., 2016). Water extractions for human use reduces stream flow, which can reduce dissolved oxygen needed to support aquatic life (McKay and King, 2006). Human-caused decreased flows were associated with a twofold increase in the likelihood of biologically impaired macroinvertebrate communities (Carlisle et al., 2011). While there are some macroinvertebrate species adapted to high flows and some adapted to lower flows, artificial creation or extension of low-flow periods adversely affects macroinvertebrates that are no longer able to meet their physiological, nutritional, and habitat requirements (Dewson et al., 2007).

13.3.1.2.2 Environmental Effects of Nutrients

As of 2014, nearly half of the rivers and stream miles in the continental United States were in poor biological condition based on a MMI of pollution-tolerant and pollution-sensitive benthic macroinvertebrate taxa, compared to 30% river and stream miles that were in good biological condition (U.S. EPA, 2019a). From 2004 to 2014, there was in increase in streams with poor macroinvertebrate condition nationally (Figure 13.6a), in the TPL (Figure 13.6b), and in the Coastal Plains (Figure 13.6g). Moreover, there was a decrease in streams with good macroinvertebrate condition nationally, in the TPL, the UMW (Figure 13.6d), the Coastal Plains, and the Xeric ecoregion (Figure 13.6i). In these ecoregions with worsening macroinvertebrate condition, there was concurrent worsening of excessive P (U.S. EPA, 2019a). In both rivers and lakes with excess P (poor rating based on regional least-disturbed reference sites), macroinvertebrates were almost twice as likely to be rated poor biological condition (U.S. EPA, 2019a, 2017a). As of 2012, over 30% of the nation's lakes had poor biological condition and over 35% had excess nutrient concentrations (U.S. EPA, 2016b). For coastal and Great Lakes nearshore waters, P was again a widespread problem (poor rating in 21% of sites) and biological condition was poorest along the Northeast coast (poor rating in 27% of sites), followed by the Great Lakes nearshore waters (poor biological condition in 18% of sites) (U.S. EPA, 2016a).

13.3.1.2.3 Environmental Effects of Sediments

Benthic macroinvertebrate condition was almost twice as likely to be rated poor when sediment levels were rated poor by a national survey of streams and rivers from 2013 to 2014 (U.S. EPA, 2019a). Benthic invertebrates are particularly susceptible to direct and indirect effects of sediments in their habitat (Donohue and Irvine, 2004; Höss et al., 1999; Wood and Armitage, 1997). For example, sedimentation may decrease the ability of some macroinvertebrates to stay attached to their habitat because of deposited sediment or unstable substrate (Donohue and Garcia Molinos, 2009). With increasing fine sediment in the streambed, there was a decrease in relative species richness of invertebrates that cling to interstitial spaces (Pollard and Yuan, 2010). High suspended sediment interferes with macroinvertebrate filter feeding (Aldridge et al., 1987), decreases respiratory rates because of direct silt contact, and decreases oxygen because of deposited silt (Donohue and Garcia Molinos, 2009). In lakes, filter-feeding zooplankton taxa such as *Cladocera* may be particularly vulnerable to the effects of sedimentation, as they cannot discriminate between phytoplankton food resources and sediment particles (Kirk and Gilbert, 1990; Koenings et al., 1990). Suspended sediment has been shown to reduce the abundance and biomass of zooplankton in lakes, as well as alter community composition (Donohue and Garcia Molinos, 2009). Deposited and suspended sediments negatively affect the survival of freshwater mussels that are not adapted to high levels of sediment and are unable to avoid intake of sediments from the water column (Henley et al., 2000).

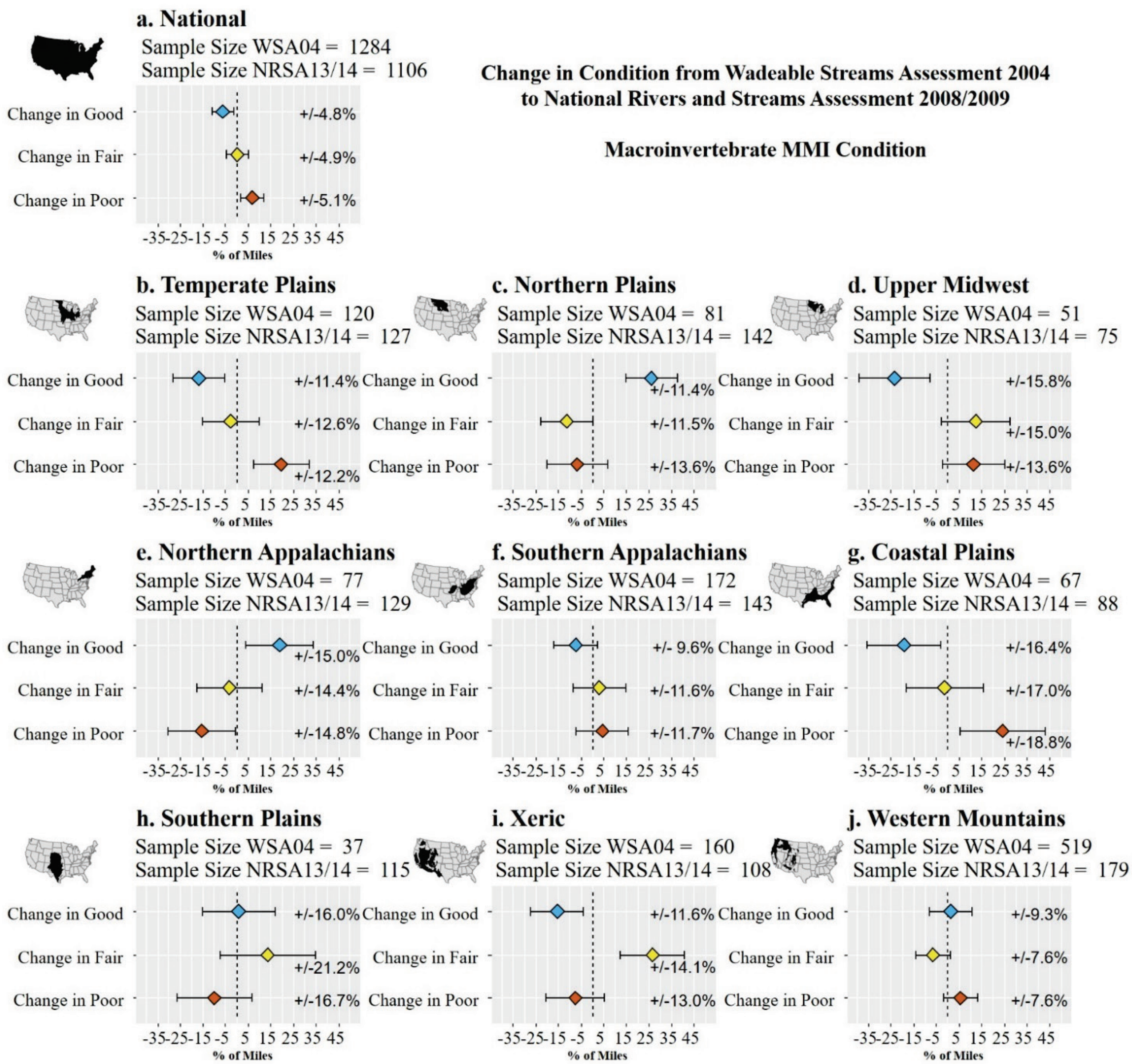
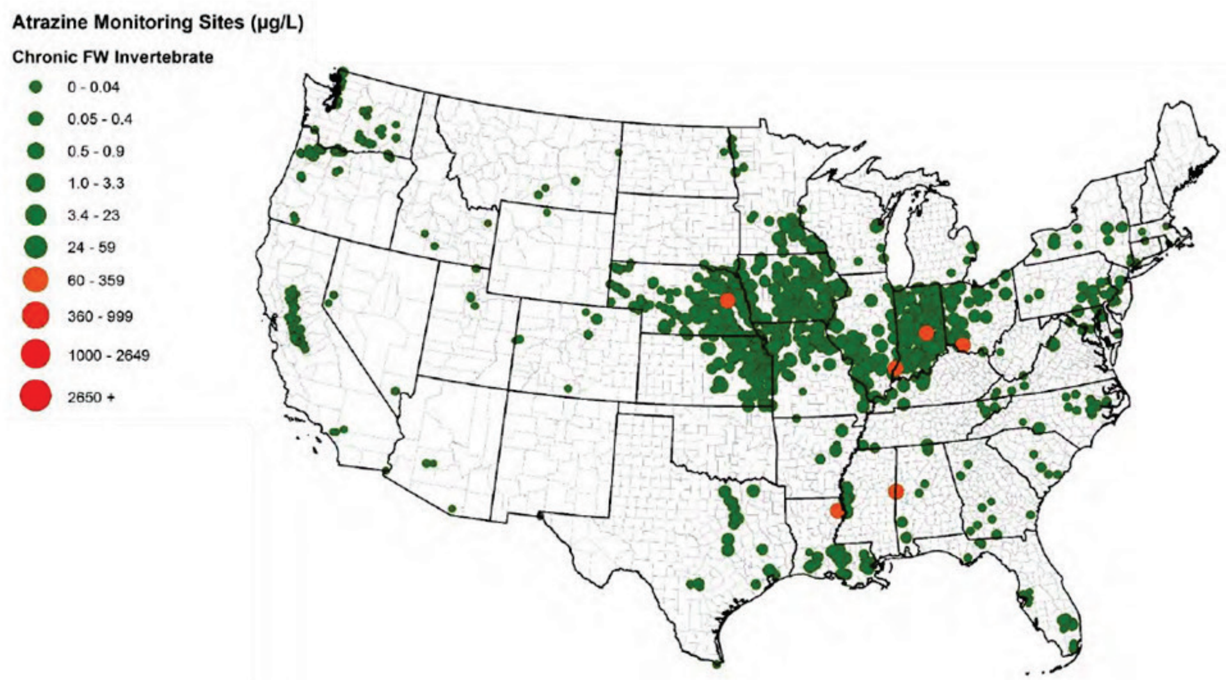


Figure 13.6. Change in macroinvertebrate Multi-Metric Index condition in wadeable streams across the conterminous United States (a) and ecoregions (b–j). The % of Miles refers to the total river and stream miles surveyed by EPA. The condition categories (Good, Fair, and Poor) are relative to the least disturbed streams in each ecoregion. "WSA04" refers to the Wadeable Streams Assessment conducted in 2000–2004 and "NRSA13/14" refers to the National Rivers and Streams Assessment conducted during 2013–2014. (data from U.S. EPA, 2019a; U.S. EPA, 2006). The error bars and percentages on the right of each plot are margins of error.

13.3.1.2.4 Environmental Effects of Pesticides

Besides insecticides, other kinds of pesticides (e.g., herbicides) may also cause harm to aquatic life. In 2016, EPA concluded that in areas where the herbicide atrazine's use is heaviest (Figure 13.7), there is potential chronic risk to aquatic invertebrates (U.S. EPA, 2016c). The conclusion was based on hundreds of toxicity studies on the effects of atrazine on plants and animals, over 20 years of surface water monitoring data, and aquatic exposure modeling. Based on overlap of species ranges and critical habitat



L = liters; µg = micrograms

Figure 13.7. Geographic distribution of atrazine monitoring sites. Shows sites with 21-day maximum average concentrations exceeding the chronic freshwater invertebrate level of concern (60 µg/L). Source: U.S. EPA (2016f).

with atrazine usage, atrazine was judged likely to adversely affect 180 out of 207 federally listed (i.e., threatened and endangered) aquatic invertebrate species assessed, including mussels, snails, shrimp, amphipods, water beetles, and crayfish (U.S. EPA, 2021).

Although the usage of neonicotinoid insecticides in the corn belt is low compared with other pesticides (e.g., none are on the list of 40 most-used chemicals on a total mass-applied basis from 2000 to 2016), their relatively high toxicity to aquatic invertebrates renders them of potential concern to aquatic resources in areas where these chemicals are used. This concern is exacerbated by the fact that usage of these chemicals as seed coatings has increased dramatically over the past decade (see Chapter 3), in part as a replacement for organophosphate and carbamate insecticides (Chrétien et al., 2017; Hladik et al., 2014). In invertebrates, neonicotinoids act by causing continuous nervous system stimulation (i.e., neonicotinoids are nicotinergic acetylcholine receptor agonists). Aquatic invertebrates have relatively high exposure potential because neonicotinoids are highly water soluble, hydrolytically stable compounds with long soil and water half-lives (Bonmatin et al., 2015; Morrissey et al., 2015). For example, chlothianidin has a half-life in soil from 148 to 6931 days, or up to almost 19 years (Thompson et al., 2020). Chlothianidin may also be generated by the degradation of another neonicotinoid, thiamethoxam, in soil (U.S. EPA, 2017c), and can subsequently enter waterbodies via runoff. Although reportedly less acutely toxic to mammals, fish, and birds than other classes of insecticides, neonicotinoids are often more toxic to aquatic invertebrates, including pollution-sensitive benthic macroinvertebrates, at widely varying concentrations, for example, with LC50s that range over six orders of magnitude.

Neonicotinoids can affect nontarget insects, such as aquatic macroinvertebrates with similar physiology as target insects. Among macroinvertebrate species whose susceptibility to neonicotinoids has been measured in the lab, those in the orders Ephemeroptera, Plecoptera, and Trichoptera (mayflies, stoneflies, and caddisflies, respectively) are among the most sensitive (e.g., with short term lethal effects at <1 μg/L for all three chemicals), while the commonly tested macroinvertebrate species *Daphnia magna* is relatively insensitive (Morrissey et al., 2015). Sampling in 2013 from 97 streams in a multistate midwestern region dominated by corn and soybeans detected residues of hundreds of pesticides, with imidacloprid found at 98% of sites (Van Metre et al., 2017). Another study from a corn and soybean growing area in Iowa found neonicotinoid residues in all sampled streams (nine in total), including clothianidin at up to 257 nanograms per liter (ng/L), thiamethoxam at up to 185 ng/L, and imidacloprid at up to 42.7 ng/L (Hladik et al., 2014). In preliminary risk assessments of these three chemicals, EPA (2017b, c, 2016d) noted that the detected concentrations of one or more of them (especially imidacloprid) in streams, rivers, lakes, and other water bodies regularly exceed chronic and/or acute toxicity thresholds reported in the literature and submitted to EPA by chemical registrants for freshwater invertebrates, especially insects (Supplemental Tables 13.1, 13.2, and 13.3). EPA's assessments of all three neonicotinoids identify corn and soybeans as their predominant uses, particularly in the form of coatings applied to the seeds of both crops (see Chapter 3, Supply Chain).

For fipronil, a relatively hydrophobic insecticide with broad-spectrum toxicity, a similar concern is that in waters where it has been detected, the compound has been at concentrations likely to be toxic to sensitive aquatic invertebrates. In experimental tests at higher concentrations, fipronil has been found to decrease diversity, alter the timing by which juvenile aquatic insects emerge as flying adults, and disrupt food webs (Miller et al., 2020).

Among the also relatively hydrophobic pyrethroids and pyrethrin insecticides, residues of bifenthrin in sediments have been found to exert significant influence on invertebrate community composition. The influence of insecticide residues on invertebrate communities is great enough that models that included pyrethroid degradates have been found to perform better than models based only on landscape variables at predicting macroinvertebrate metrics at impacted sites (Waite and Van Metre, 2017).

The EPA ECOTOX Knowledge Database (https://cfpub.epa.gov/ecotox/help.cfm) compiles information on the toxicity of pesticides and other chemicals to terrestrial and aquatic organisms in various taxonomic categories, derived predominantly from the peer-reviewed literature. For aquatic animals including crustaceans, insects, spiders, worms, mollusks, and other invertebrates, ECOTOX includes 2,492, 1,996, 61, 127, and 6,604 records for the common pesticides used on biofuel feedstocks (corn and soybean), including the herbicides glyphosate, atrazine, acetochlor, metolachlor, 2,4-D, respectively, along with 3,352, 281, and 1,535 records for the neonicotinoid insecticides imidacloprid, clothianidin, and thiamethoxam, respectively (refer to Supplemental Table 12.1). As in section 13.3.1.1.4, it is beyond the scope of this report to summarize the testing results for these chemicals.

13.3.1.3 Aquatic Plants, Algae, and Other Aquatic Organisms

13.3.1.3.1 Environmental Effects of Flow Alterations

Assemblage structure and diversity of aquatic plants are strongly related to stream and river flow rates (Bunn and Arthington, 2002). Reduction of flow variability has been linked to excessive growth of submerged aquatic macrophytes, usually prolific growths dominated by a single species such as the curly-leaf pondweed (Ochs et al., 2018). Reduced volume of discharge (e.g., from water displacement for irrigation) can also intensify the impacts of agricultural pollutants, eutrophication, and other environmental effects on aquatic biodiversity. For example, when nutrient pollution from land drainage and agricultural fields are not flushed by sufficient river discharge, downstream estuaries can experience intense phytoplankton blooms (Acharyya et al., 2012).

13.3.1.3.2 Environmental Effects of Nutrients

The addition of nutrients can substantially shift algal community composition (Liess et al., 2009; Lavoie et al., 2008; Passy, 2007; Hillebrand and Sommer, 2000). Algal growth is often limited by the availability of N and/or P and increases in concentrations of these nutrients generally stimulate growth of algal biomass in aquatic ecosystems (Elser et al., 2007; Hillebrand, 2002). In waterbodies that are nitrogen-limited, often due to excess P, toxic cyanobacteria (discussed in section 13.3.1.4) may have an advantage over other algae because some species have the ability to fix their own N from the atmosphere (Conley et al., 2009). Nutrient-fueled algal blooms or mats can also degrade aquatic ecosystems by decreasing light penetration into the water column, producing toxins, or reducing dissolved oxygen when large blooms or mats decompose (Carpenter et al., 1998).

13.3.1.3.3 Environmental Effects of Sediments

Sedimentation may negatively affect primary producers including both algae and macrophytes. Suspended sediment attenuates light that is required for primary production (Van Nieuwenhuyse and LaPerriere, 1986; Tilzer, 1983), leading to light-limiting conditions (Hoetzel and Croome, 1994) that reduce growth rates of algae and submerged macrophytes. Sediment may also damage primary producer cells via abrasion (Steinman and McIntire, 1990). Additionally, sediment can cover larger substrates, thereby increasing substrate instability for attached primary producers including macrophytes and benthic algae (Brookes, 1986). Ultimately, sediment has been shown to reduce macrophyte biomass, growth, and diversity (Lloyd et al., 1987), and lakes with higher turbidity tend to have low levels of submerged vegetation (Kimmel et al., 1990; Baxter, 1977). Sediment-induced decreases in primary production deplete important food resources for consumers like zooplankton, insects, mollusks, and fish (Henley et al., 2000). While sedimentation can lead to mortality at all trophic levels, decreases in food availability also lead to decreased consumer growth rates, reproduction rates, and recruitment (Henley et al., 2000).

13.3.1.3.4 Environmental Effects of Pesticides

In a 2012 survey, EPA detected atrazine in 30% of randomly sampled U.S. lakes, though concentrations reached or exceeded EPA's level of concern for plants in freshwater (4 μg/L) in less than 1% of them (U.S. EPA, 2016c). The 2012 lake assessment was based on samples from the summer months and the sampling design did not specifically target pesticide usage areas. Based on hundreds of toxicity studies, over 20 years of monitoring data, and aquatic exposure models, EPA concluded that in areas where atrazine use is heaviest (mainly in the TPL ecoregion), there is a high probability of changes to aquatic plant assemblage structure, function, and primary production at or above a 60-day average concentration of 3.4 μg/L atrazine (U.S. EPA, 2016f). Changes to aquatic plant assemblage structure, function, or productivity can affect other parts of the food web because they result in reduced food and altered habitat for fish, invertebrates, and birds. Besides such direct and indirect toxicity effects, pesticides can also affect ecosystems in less obvious ways. For example, some bacteria can use glyphosate for growth, enhancing microbial proliferation. There are also cyanobacteria with natural tolerance to glyphosate and certain concentrations of glyphosate can stimulate photosynthesis in a common bloom-forming cyanobacterial species, *Microcystis aeruginosa* (Harris and Smith, 2016; Hove-Jensen et al., 2014; Qiu et al., 2013).

The EPA ECOTOX Knowledge Database (https://cfpub.epa.gov/ecotox/help.cfm) compiles information on the toxicity of pesticides and other chemicals to terrestrial and aquatic organisms in various taxonomic categories, derived predominantly from the peer-reviewed literature. For aquatic plants, ECOTOX includes 2,074, 5,039, 42, 377, and 10,162 records for the common pesticides used on biofuel feedstocks (corn and soybean), including herbicides glyphosate, atrazine, acetochlor, metolachlor, 2,4-D, respectively, along with 272, 2, and 661 records for the neonicotinoid insecticides imidacloprid, clothianidin, and thiamethoxam, respectively (refer to Chapter 12, Supplemental Table 12.1). As for other taxonomic groups, it is beyond the scope of this report to summarize these results.

13.3.1.4 Harmful Algal Blooms (HABs)

HABs are increasing in frequency and occurrence worldwide in response to global-scale changes, including the intensification and extensification of agriculture, and are considered one of the major threats to aquatic biodiversity (Reid et al., 2019). HABs affect U.S. lakes, reservoirs and coastal zones, and damages estimated to be over $4 billion in losses are occurring annually in the United States (Kudela et al., 2015). Excess nutrients (both N and P) in waterbodies can result in algal blooms (Paerl et al., 2016), some of which can produce toxins or accumulate excessive biomass resulting in HABs.

Lakes and reservoirs with excess nutrient concentrations are susceptible to recurring algal blooms, such as western Lake Erie, which receives nutrients loads from a drainage area dominated by agricultural land use. A bloom observed in western Lake Erie in 2011 was attributed to unusual weather patterns coupled with long-term trends in agricultural practices that increased runoff of dissolved reactive phosphorus (Michalak et al., 2013). The main driver of HABs in western Lake Erie is P, particularly from the Maumee River watershed. While P loadings determine the physical volume of a HAB, N loading appears to play a critical role in determining bloom composition. The cyanobacterium *Microcystis*, which produces the hepatotoxin microcystin, lacks the N-fixing capability of other cyanobacteria and therefore is favored by the presence of excess N. The detection of microcystin in source water led to a temporary shutdown of the Toledo, Ohio drinking water supply during a Lake Erie HAB in 2014 (Levy, 2017).

Analyses by Taranu et al. (2017) and Yuan et al. (2014) confirm that total nitrogen (TN) concentration in lake water is a much stronger predictor than TP of the probability of detecting *Microcystis* in U.S. lakes; the percent of land cover that was agriculture within the ecoregion of a given lake was also a strong predictor (Taranu et al., 2017). A modeling study by Michalak et al. (2013) concluded that, if corn acreages continued to be at recent high levels, along with projected future increases in spring precipitation, similar events could be more likely in the future, and in fact have continued to occur. Therefore, it appears likely that demand for biofuel feedstocks could lead to increases in agriculture-related nutrient loadings to surface waters, and in turn increased risk of HABs.

Studies have shown that corn and soybean production could contribute to increased P loadings to surface waters (Labeau et al., 2014) and aquatic systems (Jarvie et al., 2015). Modeling scenarios using the Soil and Water Assessment Tool watershed model (SWAT; https://swat.tamu.edu/) suggest that conservation practices (e.g., filter strips, cover crops, riparian buffers) can help achieve TP targets, whereas dissolved reactive P is much more responsive to reductions of P application to fields (especially inorganic P). Modeling also suggested that conversion to perennial grasses such as switchgrass (*Panicum* spp.) and *Miscanthus* spp., even with manure application, would significantly reduce P runoff into waterbodies (Muenich et al., 2016).

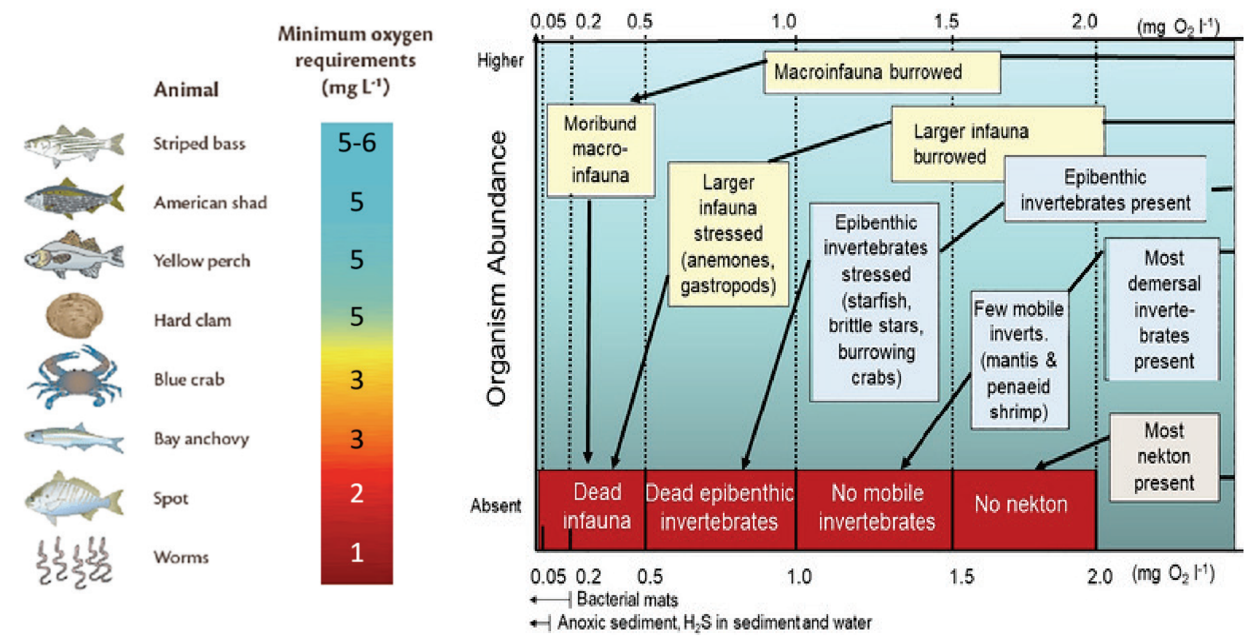
While fertilizer use by current agricultural practices contributes to much of the nutrient loading that stimulates algal responses in many waterbodies, the total nutrient budgets of some waterbodies also include internal sources of nutrients released from the bottom sediments (Chen et al., 2018). Nutrients released from bottom sediments may originate in part from legacy inputs from historical agricultural land use in the watershed. Feedback loops between HABs and hypoxia exacerbate problems from nutrient enrichment (see section 13.3.1.6). Along lake shorelines, blooms of filamentous green algae such as *Cladophora* harbor potentially pathogenic bacteria and foul recreational beaches when the algae proliferate and decay (Ibsen et al., 2017). The risk of HABs is not limited to lakes (Fetscher et al., 2015). Streams and rivers can develop toxin-producing algal proliferations in the benthic zone, potentially when nutrient inputs coincide with lower flow (Mcallister et al., 2018), as well as in the water column (Otten et al., 2015). Rivers can also act as conduits of HABs and associated toxins discharged from inland lakes to estuaries and oceans, where marine organisms such as sea otters can become exposed and sickened (Miller et al., 2010).

The species composition, toxins, and anthropogenic disturbances associated with HABs are diverse and there is ongoing research on the complexities of bloom mechanisms, toxin production, and effects on aquatic organisms. What unifies the diverse forms of HABs is that they cause harm (Ramsdell et al., 2005), by competing with co-occurring organisms for resources (e.g., nutrients, light), altering food web dynamics, and/or producing toxins and other deleterious compounds (Ibelings et al., 2008). Exposure to cyanobacterial toxins can lead aquatic organisms to exhibit disturbances in behavior, physiology, growth, reproduction, and other factors, depending on the toxin's mode of action (Bownik and Pawlik-Skowrońska, 2019; Wiegand and Pflugmacher, 2005). When toxins enter the food web via ingestion, the toxins can transfer and bioaccumulate from herbivorous zooplankton to predatory zooplankton (Laurén-Määttä et al., 1995) and fish (Sotton et al., 2014). In addition to toxins, cyanobacteria may exude compounds that inhibit the growth of co-occurring organisms (Wang et al., 2017; Valdor and Aboal, 2007) or result in abnormal physical development in animals (Yeung et al., 2020).

13.3.1.5 Hypoxia

In both freshwater and coastal marine systems, algal blooms senesce as microbes decompose algal cells and deplete oxygen from the water column, creating hypoxic zones. Hypoxic zones are inhospitable to many aquatic organisms and may result in direct mortality when oxygen drops below tolerable levels (Figure 13.8a,b). Increases in hypoxia have led to increased frequencies and magnitudes of fish kills (Thronson and Quigg, 2008). According to the Assessment, Total Maximum Daily Load Tracking and Implementation System (ATTAINS), many of the nation's waterways are impaired by oxygen depletion (Figure 13.9).

a. b.



L = liters; mg = milligrams

Figure 13.8. Oxygen requirements. Minimum oxygen requirements of several aquatic organisms (a), and progressive changes in fish and invertebrate fauna as the bottom-water oxygen (O_2) concentration decreases from near 2 mg/L to 0 mg/L (b). Sources: CENR (2010) for a. and Rabalais and Turner (2019) for b. (Creative Commons license, https://creativecommons.org/licenses/by/4.0/; no changes made).

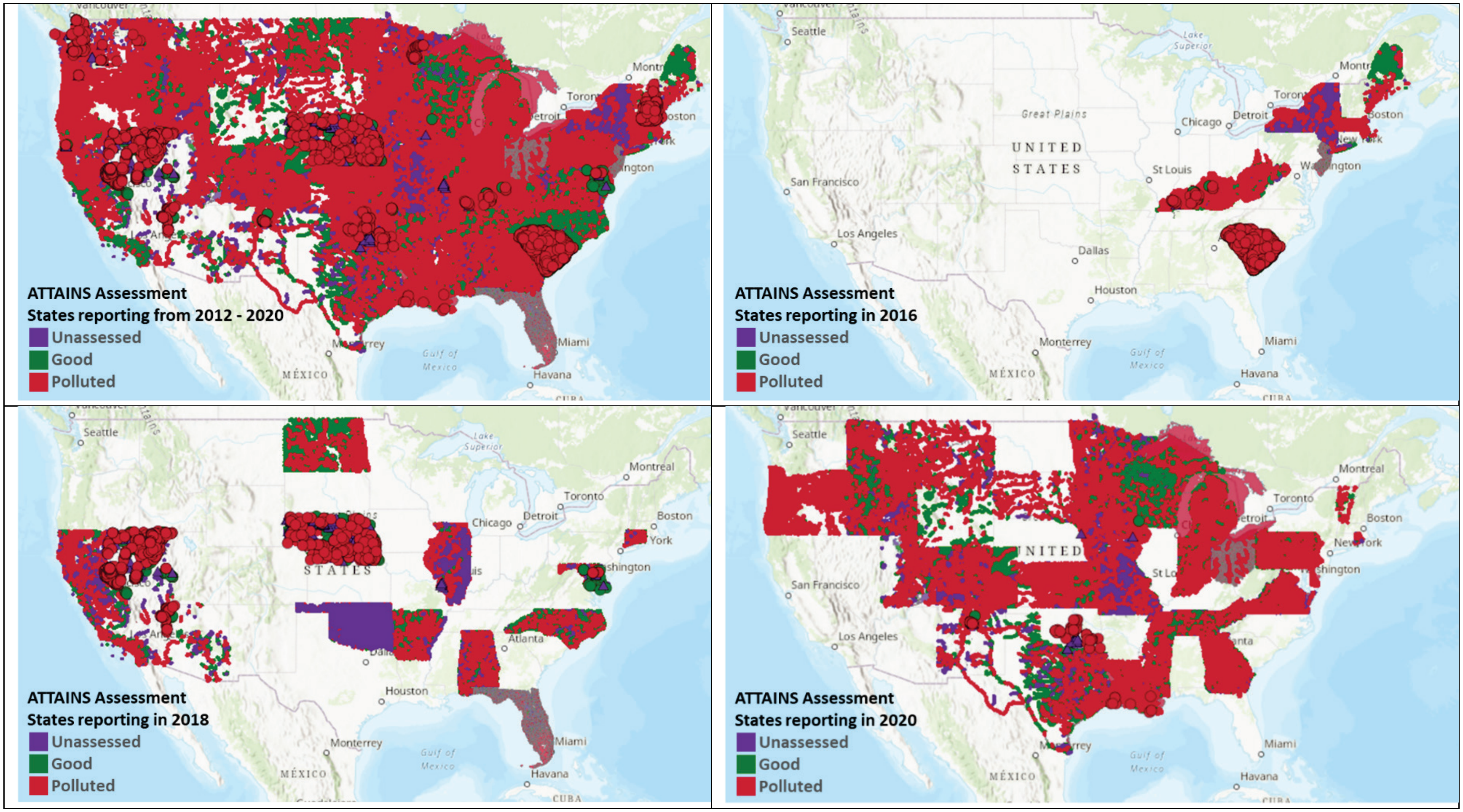
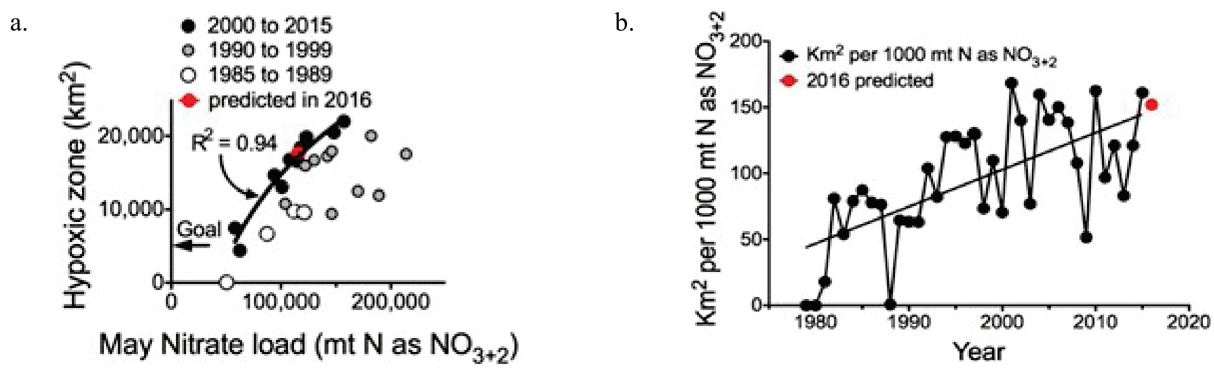


Figure 13.9. Maps of waters where oxygen depletion was identified as a cause of impairment. ATTAINS Assessment geospatial service data from 2012 to 2020, including point, line, and area data. Source: https://www.epa.gov/waterdata/get-data-access-public-attains-data (accessed January 22, 2021).

The size of the Gulf of Mexico hypoxic zone (i.e., area with bottom dissolved oxygen < 2.0 mg/L) is a function of climate, weather, basin morphology, circulation patterns, water retention time, freshwater inflows, stratification, mixing, and nutrient loadings (Dale et al., 2010). The hypoxic zone size is also a function of loading of nitrate-plus-nitrite from the Mississippi and Atchafalaya River system during May, as well as the periodic action of tropical storms to re-aerate the bottom layer (Turner and Rabalais, 2016). However, the nature of this relationship is changing—nitrate/nitrite loading of a given magnitude is causing a larger hypoxic zone in recent years than it did in earlier years (Figure 13.10). The changing sensitivity of the hypoxic zone to nitrate loading could be due to legacy effects as organic matter that accumulated in the sediments in previous years become metabolized in later years (Turner and Rabalais, 2016). The 2017 Gulf of Mexico hypoxic zone was the largest measured since 1985 [Figure 13.11, (LUMCON, 2017)]. The seasonal timing and magnitude of river water flowing into the Gulf of Mexico have important implications on the effects of hypoxia on aquatic life.



km = kilometers; mt = metric tonnes

Figure 13.10. Size of the Gulf of Mexico hypoxic zone. Changes in the measured size of the Gulf of Mexico hypoxic zone (a) as related to the amount of nitrate-nitrate loading (b). Source: Turner and Rabalais (2016) (used with permission).

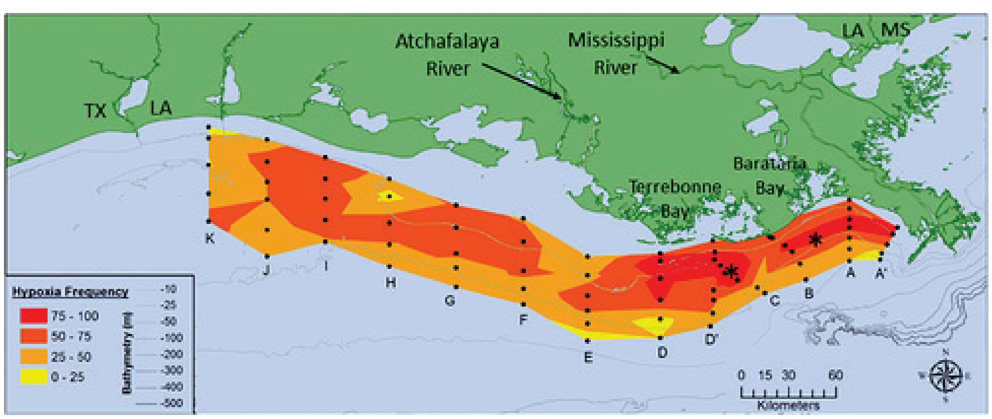


Figure 13.11. Long-term record of hypoxia frequency. Shown are percent of samples with bottom-water dissolved oxygen < 2 mg/L at midsummer (usually mid-July to early August) in the Gulf of Mexico mapped from 1985 to 2014. Source: Rabalais and Turner (2019) (Creative Commons license, https://creativecommons.org/licenses/by/4.0/; no changes made).

13.3.1.6 HABs and Hypoxia Feedback Loops

HABs and hypoxic events are linked phenomena that can interact in positive feedback loops, exacerbating environmental effects on aquatic ecosystems. When algae-eating aquatic life relocate or die (e.g., fish kills due to hypoxia), algal species more likely to become HABs can take over and dominate the system (IWG-HABHRCA, 2016). If key parts of the food web are missing, algal blooms may proliferate unchecked and reinforce hypoxic conditions (IWG-HABHRCA, 2016; Rosenblatt et al., 2013). When hypoxia is more frequent, longer in duration, or more expansive, the oxygen-depleted conditions lead to changes in the chemistry of the bottom sediment that result in release of nutrients into the water column. The additional nutrients in the water further promote HABs, especially in P-limited freshwater systems.

13.3.2 New Analyses

13.3.2.1 SWAT Modeling and Nutrient Thresholds

Two new analyses provide information for Chapter 13: the Missouri River Basin (MORB) SWAT modeling described in Chapter 10 (section 10.3.2), and the geospatial overlays of critical habitat for threatened and endangered (T&E) species with areas of grassland conversion described in Chapter 12 (section 12.3.2). The MORB SWAT modeling did not include pesticides (Chen et al., 2021), thus there are not direct estimates of pesticide yields that may enter streams and rivers in the MORB. The analyses simulated impacts of shifting crop cultivation practices on TN, TP, and suspended sediment yields from MORB for 2008–2016. The simulated scenarios of crop cultivation included baseline, continuous corn, corn/soybean rotation, and corn/wheat rotation.[7] If grasslands in the MORB were converted to corn/soybean rotation, yields of TN and TP from the MORB may increase 6.0% and 6.5%, respectively. Differences between baseline and the three conversion scenarios on streamflow and sediment were trivial at the HUC-8 watershed outlet of the MORB, but the nutrient yields at the watershed outlet were all increased from the three conversion scenarios (see Chapter 10, section 10.3.2).

To better relate the MORB SWAT modeling results to effects on aquatic organisms, the flow-weighted concentrations[8] of TN and TP are compared to EPA NRSA condition classes and state-reported nutrient criteria for rivers and streams within MORB. The NRSA condition classes (least-, moderate, and most-disturbed based on nutrient concentrations) were determined from data and observations from the "best" remaining (i.e., reference) stream/river sites in each ecoregion and the continuous gradient of observed values across the population of streams and rivers in the United States (Van Sickle and Paulsen,

[7] The baseline included the USDA Cropland Data Layer (CDL) for 2008 and 2009 (refer to Chapter 10, Figure 10.6). The scenarios included conversion from grassland to one of three land use types (i.e., continuous corn, corn/soybean rotation, and corn/wheat rotation). Conversion only occurred on the subset of land that were observed to convert from grassland to cropland in Lark et al. (2020).

[8] Flow-weighted concentrations account for the influence of water flow on water concentrations of nutrients by taking the annual nutrient mass flux and dividing it by the annual volume of river discharge.

2008; Stoddard et al., 2006). Nutrients at least-disturbed sites are not different in concentrations from the reference sites, while moderately disturbed sites have somewhat higher concentrations, and most disturbed sites have markedly higher concentrations than reference sites. The NRSA condition classes for nutrients use the 0–75th percentile of the reference distribution in an ecoregion to define the least-disturbed condition class (Table 13.1). The 95th percentile (and above) of the reference distribution in each ecoregion defines the most disturbed condition class (U.S. EPA, 2016c; Herlihy et al., 2008). The moderately disturbed class is in between. State numeric nutrient criteria apply only to specified waterbodies within the state and are not necessarily developed using a reference-based approach like the NRSA condition classes (Table 13.2). Many states do not yet have numeric nutrient criteria, and none have a complete set as of the end of 2020.[9] But, where they exist, state's numeric nutrient criteria (1) provide nutrient goals to protect and maintain the designated uses of a water body (Title 33 of the United States Code [U.S.C.] § 1313(c)), (2) provide thresholds that allow the state to make accurate water quality assessment decisions (33 U.S.C. § 1313(d)), and (3) provide targets for restoration of waterbodies that can guide waste load allocation decisions (33 U.S.C. § 1313(d)).

Table 13.1. Nutrient condition class benchmarks used to characterize least-disturbed, moderately disturbed, and most-disturbed sample reaches in ecoregions surveyed as part of the EPA's NRSA. Modified from table 6.1 in U.S. EPA (2016c).

EPA NARS aggregate ecoregions	TP (mg/L)			TN (mg/L)		
	Least	Moderate	Most	Least	Moderate	Most
Central Plains	<0.06	0.06-0.10	>0.10	<0.62	0.62-1.08	>1.08
Northern Appalachians	<0.02	0.02-0.03	>0.03	<0.35	0.35-0.48	>0.48
Northern Plains	<0.06	0.06-0.11	>0.11	<0.58	0.58-0.94	>0.94
Southern Appalachians	<0.01	0.01-0.02	>0.02	<0.24	0.24-0.46	>0.46
Southern Plains	<0.06	0.06-0.13	>0.13	<0.58	0.58-1.07	>1.07
Temperate Plains	0.09	0.09-0.14	>0.14	<0.70	0.70-1.27	>1.27
Upper Midwest	0.04	0.04-0.05	>0.05	<0.58	0.58-1.02	>1.02
Western Mountains	0.02	0.02-0.04	>0.04	<0.14	0.14-0.25	>0.25
Xeric	0.05	0.05-0.10	>0.10	<0.29	0.29-0.53	>0.53

L = liters; mg = milligrams

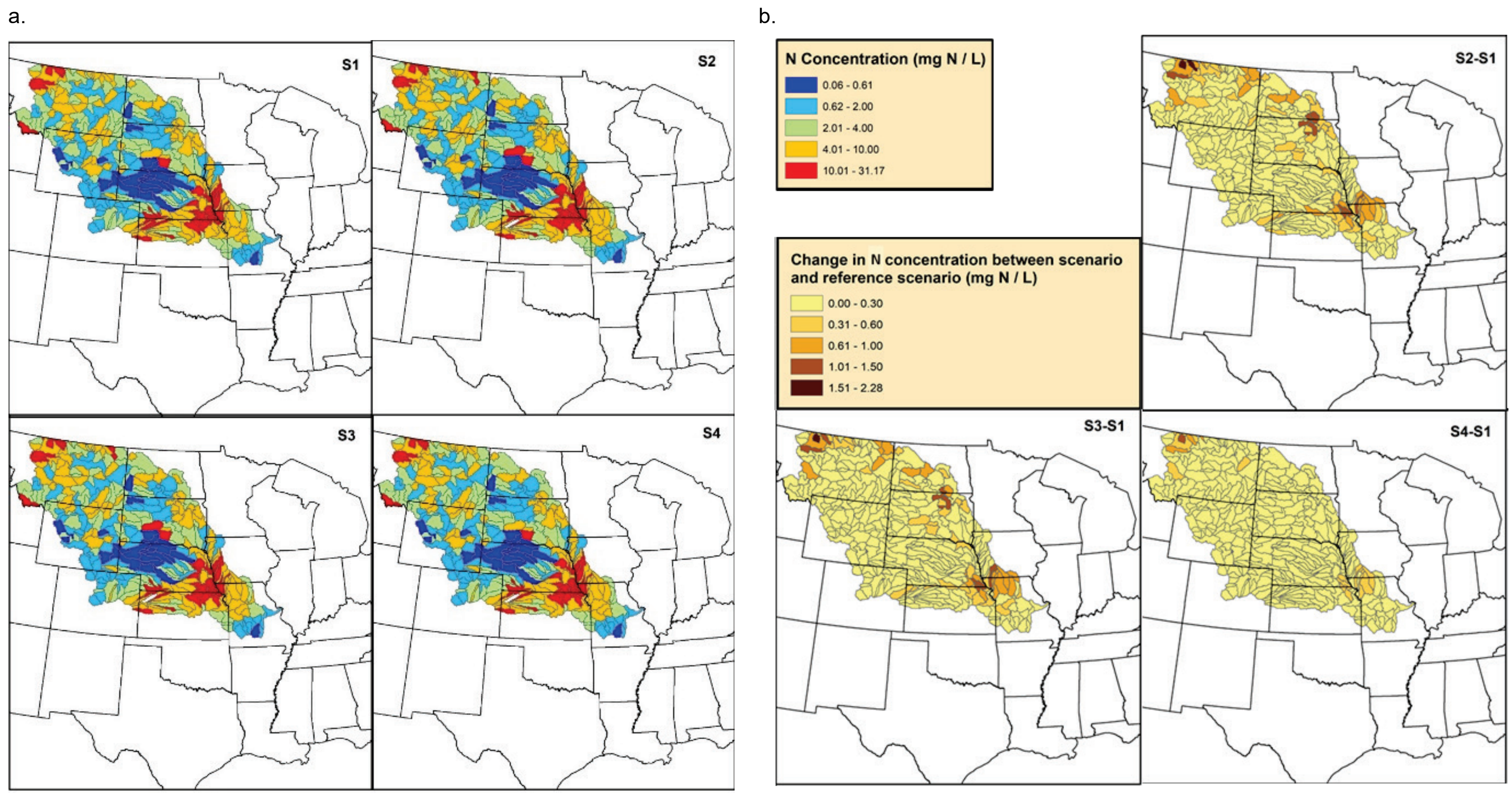
[9] See https://www.epa.gov/nutrient-policy-data/state-progress-toward-developing-numeric-nutrient-water-quality-criteria. A "complete set" is defined as criteria for both N and P for all water types in a state at this website.

Table 13.2. Range of numeric nutrient criteria from states in the Missouri River Basin (as of July 2022).

State	Water body type	Total P criteria (µg/L)	Total N criteria (mg/L)
Colorado*	Lakes/Reservoirs	7.4–36	0.33-0.6
Iowa	All	--	--
Kansas	All	--	--
Missouri	Lakes/Reservoirs	7–31	0.2– 0.62
Montana	Rivers/Streams	20–150	0.3-1.3
Nebraska	Lakes/Reservoirs	40–50	0.8–1
North Dakota	All	--	--
South Dakota	All	--	--
Wyoming	All	--	--

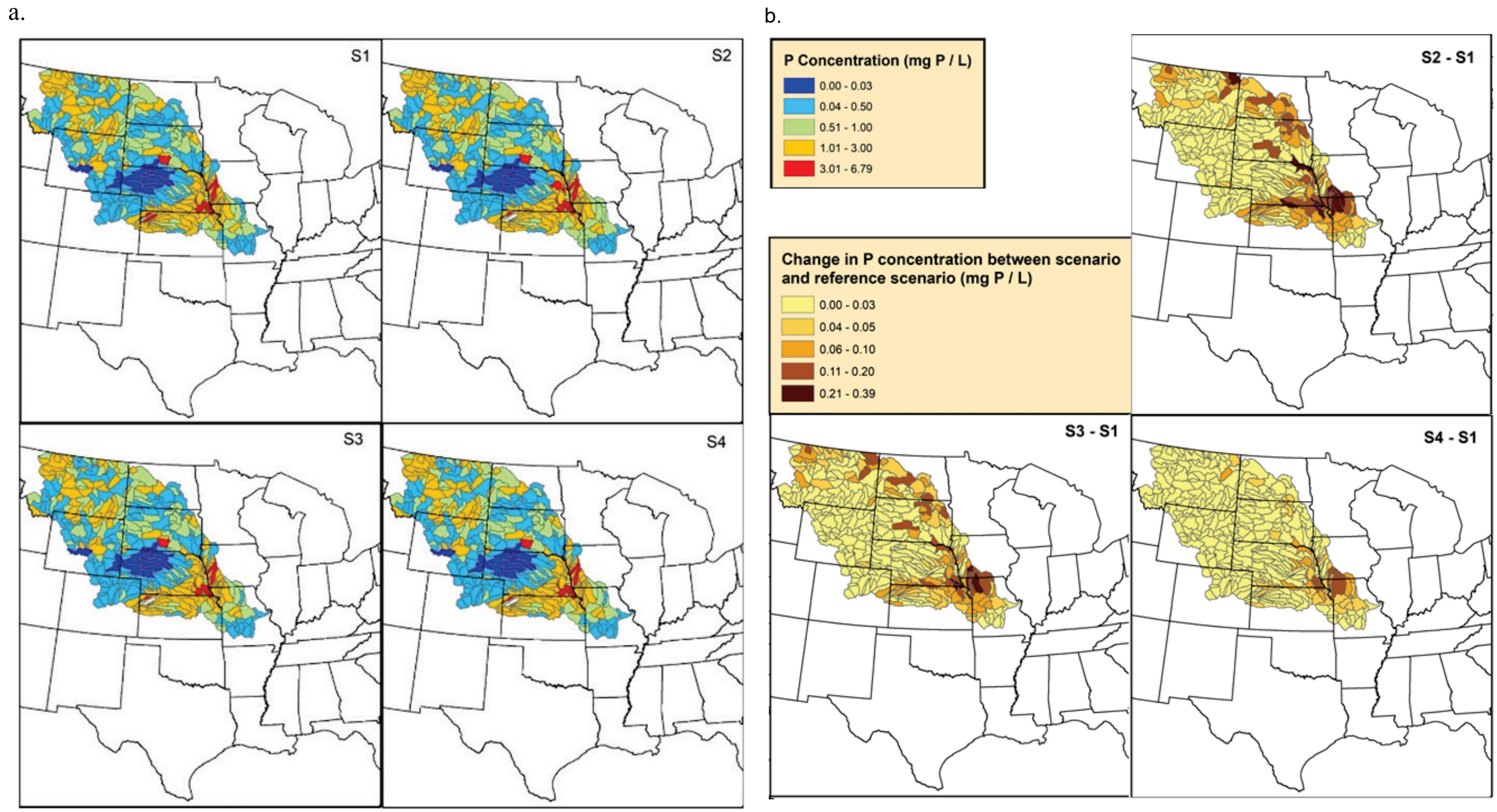
L = liters; µg = micrograms; mg = milligrams
-- = no criterion available. Minnesota omitted since such a small fraction of the state intersects with the MORB (Figure 13.12).
* As of July 2022, Colorado's TP and TN criteria for lakes and reservoirs are those values expected to be adopted after state rulemaking action in November 2022.

Similar to estimates of TN and TP yields, annual flow-weighted concentrations in streams and rivers were highest at the general confluence of Iowa, Nebraska, Kansas, and Missouri across all four scenarios (Figures 13.12a and 13.13a). However, pockets of similarly high nutrient concentrations (>4.0 mg N/L and > 1.0 mg P/L) were also observed in streams and rivers of northern Kansas, as well as central South Dakota and Montana. Other mountainous (Ozarks and Rockies) or generally more arid parts of MORB had concentrations less than 2.0 mg N/L and 0.5 mg P/L. Relative to the rest of the basin, the Sandhills of Nebraska had exceptionally low TP and TN annual flow-weighted concentrations across all four scenarios (Figures 13.12a and 13.13a).



L = liters; mg = milligrams

Figure 13.12. Modeled mean flow-weighted total nitrogen concentrations in the Missouri River Basin (MORB). Shows concentrations in individual HUC-8s (2008–2016) of MORB for four scenarios: S1-Reference, S2-Continuous Corn, S3-Corn/Soy; and S4-Corn/Wheat (a), and change from S1 to remaining three scenarios (b). Refer to Chen et al. (2021) for details and methods. Color legend for (a) provided in (b). Note results in Figures 13.12 and Figure 13.13 are the same as from Chapter 10 (section 10.3.2) but converted to stream concentrations.



L = liters; mg = milligrams

Figure 13.13. Modeled mean flow-weighted total phosphorus concentrations in the Missouri River Basin (MORB). Shows concentrations in individual HUC-8s (2008–2016) of MORB for four scenarios: S1-Reference, S2-Continuous Corn, S3-Corn/Soy; and S4-Corn/Wheat (a) and change from S1 to remaining three scenarios (b). Refer to Chen et al. (2021) for details and methods. Color legend for (a) provided in (b).

Regardless of the specific crop cultivation scenario, even including the reference scenario, the spatial variability of predicted nutrient concentrations can be largely explained by the intensity of agricultural practice. The distribution of crop production is concentrated in the southeastern portion of the MORB because of a combination of amenable soils and climate (or at least access to reliable groundwater reserves) capable of generating reliable crop yields. The rest of the basin crop production is either more limited and opportunistically pursued or not pursued (e.g., in the Sand Hills of Nebraska). Under all three change scenarios, especially the continuous corn scenario, surface water nutrient concentrations are likely to increase rather than decrease (Figures 13.12b, 13.13b).

General increases in flow-weighted TN and TP concentrations are likely tied to increased crop cultivation across all modeled scenarios. Estimated flow-weighted TN and TP concentrations generally increased across the basin for all three cropland cultivation scenarios, but higher predicted increases were observed for the continuous corn and corn/soy rotation scenarios (Figure 13.12b and 13.13b, S2-S1 and S3-S1) compared to the corn/wheat scenario (Figure 13.12b and 13.13b, S4-S1). The changes from the baseline to all three scenarios, however, were relatively small (i.e., <2.28 mg N/L, Figure 13.12b; and <0.39 mg P/L, Figure 13.13b). Increases in estimated N concentrations were most pronounced at the general confluence of Iowa, Nebraska, Kansas, and Missouri, but parts of the Dakotas and northern Montana also had predicted increases in concentrations of 0.3 to 1.5 mg N/L (Figure 13.12b). On average, TN concentrations increased 1.4–4.2% across the MORB with continuous corn having the largest modeled increase followed by corn/soy, and then corn/wheat. There were 37 out of 305 HUC-8s that increased by more than 10% in the continuous corn scenario, whereas 30 HUC-8s and only 1 HUC-8 had similar increases in the corn/soy and corn/wheat scenarios, respectively (Figure 13.12b). Flow-weighted TP concentrations in the Dakotas and Montana (0.06–0.20 mg P/L) were greatest in the southeastern portion of the MORB (0.11–0.39 mg P/L, Figure 13.13a). The TP concentrations increased on average 1.5–4.8% across all HUC-8s in the MORB with continuous corn having the largest increase followed by corn/soy, and then corn/wheat. The continuous corn scenario had 43 HUC-8s displaying greater than a 10% increase in concentration, while 35 and 7 of the HUC-8s had an increase greater than 10% in the corn/soy and corn/wheat scenarios, respectively (Figure 13.13b).

Using data from Figure 13.12 (for N) and from Figure 13.13 (for P) and comparing that with the thresholds in Table 13.1, showed that nutrient conditions for N (Figure 13.14) and P (Figure 13.15) were either most or moderately affected for most HUC8 watersheds regardless of the scenario examined. Thus, overall the watersheds in the MORB are already significantly affected by nutrients, and the additional strain from changes from 2008–2016 are difficult to separate from the baseline conditions.

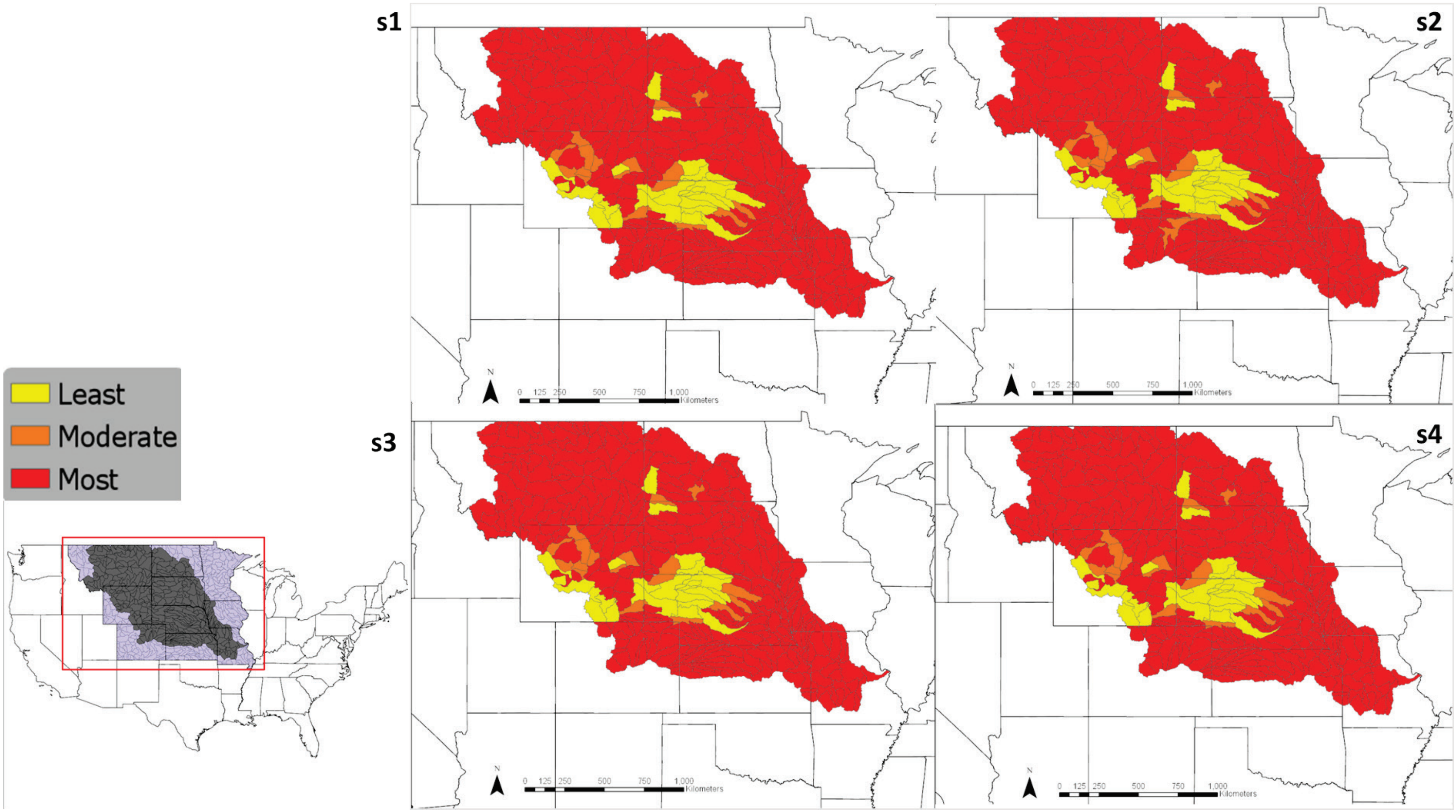


Figure 13.14. Condition classes for total nitrogen (TN). NRSA condition classes (least, moderate, most disturbed) for TN within watersheds in the MORB for the four scenarios: S1-Reference, S2-Continuous Corn, S3-Corn/Soy rotation; and S4-Corn/Wheat rotation. Only a few watersheds changed condition under S2, S3, or S4, compared with S1 (not shown).

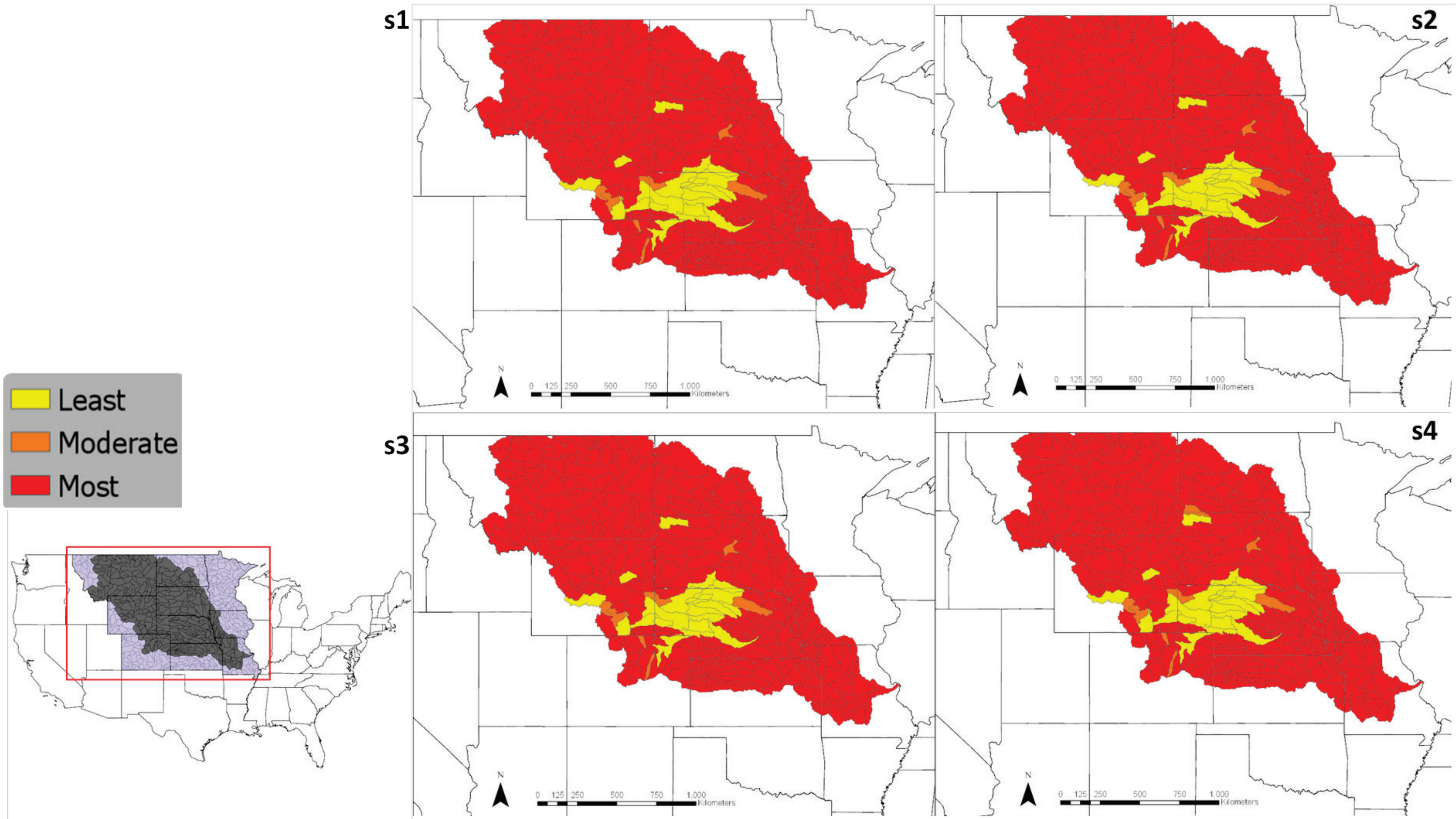


Figure 13.15. Condition classes for total phosphorus (TP). NRSA condition classes (least, moderate, most disturbed) for TP within watersheds in the MORB for the four scenarios: S1-Reference, S2-Continuous Corn, S3-Corn/Soy rotation; and S4-Corn/Wheat rotation. Only a few watersheds changed condition under S2, S3, or S4, compared with S1 (not shown).

Among the states intersecting the MORB, only Colorado, Missouri, Montana, and Nebraska have some numeric nutrient criteria (Table 13.2, Figure 13.16). The annual flow-weighted concentrations of both TN and TP across the MORB largely exceed these numeric nutrient criteria, except in the Sand Hill region. Thus, if these criteria are generally approximate for nutrient criteria in other waterbodies across the basin, then all states likely would have exceedances, even without the small increases associated with the examined scenarios. Due to the existing land management in the MORB, many watersheds in the baseline scenario already have high nutrient concentrations (Figure 13.14 and 13.15, S1). Even for the continuous corn scenario, the risk of new areas surpassing criteria is very low or numeric criteria specifically applicable to the waters in those areas are not available (Figure 13.16).

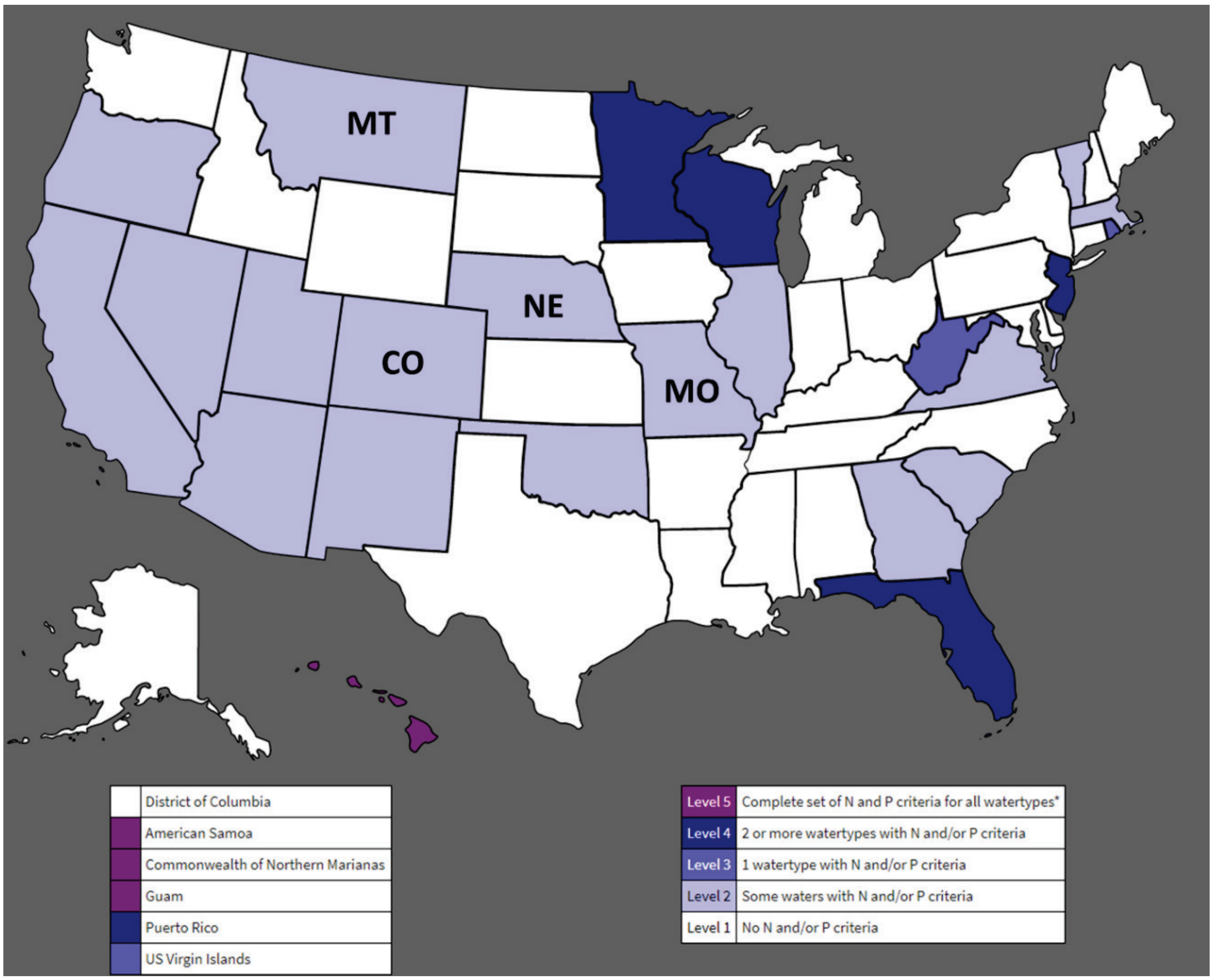


Figure 13.16. National summary of current EPA-approved numeric TN and TP criteria. In the MORB, only Colorado, Missouri, Montana, and Nebraska (labeled states) have numeric criteria for some waters (lake/reservoirs or rivers/streams; refer to Table 13.2). Source: https://www.epa.gov/nutrient-policy-data/state-progress-toward-developing-numeric-nutrient-water-quality-criteria (accessed May 7, 2021).

13.3.2.2 Threatened and Endangered (T&E) Species

T&E aquatic species are sensitive to habitat loss or degradation and excess pesticides, nutrients, and sediments from agriculture and production practices. In many agricultural landscapes, streams are channelized[10] and there is removal of vegetation from riparian areas. The physical and biological changes to streams alter the flow of water and accelerate channel erosion with high flow events. The T&E analysis described in Chapter 12 also considered aquatic species (see section 12.3.2). As shown in Table 12.1, there were 78 aquatic species with 10 acres or more of corn and soybean planted within a 1-mile buffer of their critical habitat. These included a range of aquatic animals, including 8 fish, 19 clams or mussels, 5 other aquatic invertebrates (crustaceans, insects, snails), and the Ozark hellbender, an obligate riverine amphibian (Table 12.1). A full list of T&E species, including aquatic species, occurring in the northern Great Lakes, central plains, and prairie ecoregions is provided in Chapter 12 (Supplemental Table 12.2).

13.3.2.2.1 Fish

The Topeka shiner (*Notropis topeka,* hereafter called "shiners") is a freshwater fish that has been listed as endangered since 1999. Shiner populations and overall range have declined over the past few decades because of streamflow alterations resulting from land use change around their critical habitat (Figure 13.17). Shiners rely on oxbow habitats,[11] and studies have shown that removing sediment from oxbows to restore a groundwater connection and allow more water to remain in the oxbow during droughts has increased the abundance of shiners (Simpson et al., 2019). Additionally, shiners may be indirectly, negatively affected by high atrazine concentrations impacting food resources in agricultural streams (Bartell et al., 2019). Strong evidence for species with critical habitats likely adversely affected by atrazine was found for 34 fish (Supplemental Table 13.4).

The Arkansas River shiner (*Notropis girardi*) is a freshwater fish that has been listed as threatened under the Endangered Species Act since 1998. The Arkansas River shiner originated in the Arkansas River basin, but its range has significantly decreased over the past few decades (Worthington et al., 2014). A variety of physical and chemical factors including reduced stream flow (Durham and Wilde, 2009), warmer temperatures, and increased total dissolved and suspended solids have been shown to influence the abundance and persistence of the Arkansas River shiner (Mueller et al., 2017). Additionally, studies have shown that warmer temperatures and higher total dissolved solids and total suspended solids are associated with earlier development, decreased larval viability, and decreased survival of shiners (Mueller et al., 2017).

[10] Channelized here means artificially straightened, altering the natural path of water in a stream.
[11] Oxbow habitats have water that is free-standing or slowly flowing in a curving, horseshoe-shaped path formed over time by a stream's natural meandering process, which are often channelized in developed areas including those with intensive agriculture. Silt deposits from repeated flooding can reduce water depth over time, making the oxbow prone to hypoxia-related fish kills or drying during droughts.

a. b.

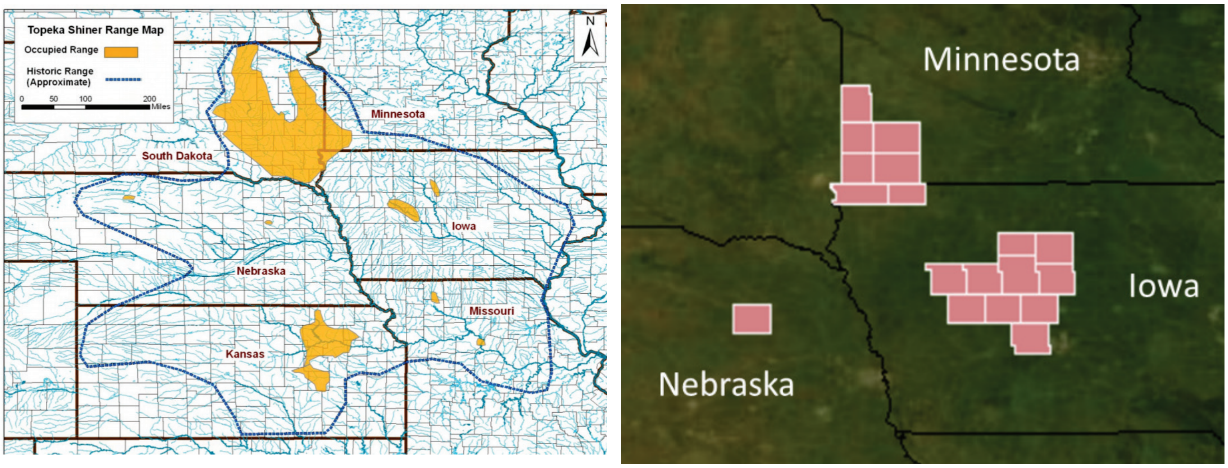


Figure 13.17. Topeka shiner range maps. Historical and occupied range from USFWS 2009 (a). Counties where the final critical habitat generally occur (b). Source of GIS file: USFWS https://catalog.data.gov/dataset/final-critical-habitat-for-the-topeka-shiner-notropis-topeka.

The Gulf sturgeon (*Acipenser oxyrhynchus desotoi*) is an anadromous[12] fish with critical habitat in the Gulf of Mexico (Figure 13.18), that migrates up freshwater rivers to reproduce and find cooler water temperatures in the spring and summer (Sulak and Clugston, 1999). The timing of spring migration does not typically coincide with HABs or the onset of the dead zone in the Gulf of Mexico, which are usually summer or late-summer phenomena. During winters, Gulf sturgeon return downstream and forage in estuarine and marine areas where they may be more vulnerable to HABs, hypoxia, or severe weather events such as hurricanes (Parauka et al., 2011). In comparison to other fish species, sturgeon metabolism, growth, and survival are sensitive to insufficient oxygen levels and sturgeon may become squeezed out of the deeper and cooler waters that they prefer if the oxygen levels become too low (Secor and Niklitschek, 2001). Sturgeon have a few options for dealing with low oxygen: swim away from hypoxic conditions, move vertically to the surface to access more oxygen, or slow down their metabolic rate by reducing swimming (Secor and Gunderson, 1998).

[12] Anadromous fish spend most of their lives in the ocean but return to freshwater spawning areas to reproduce.

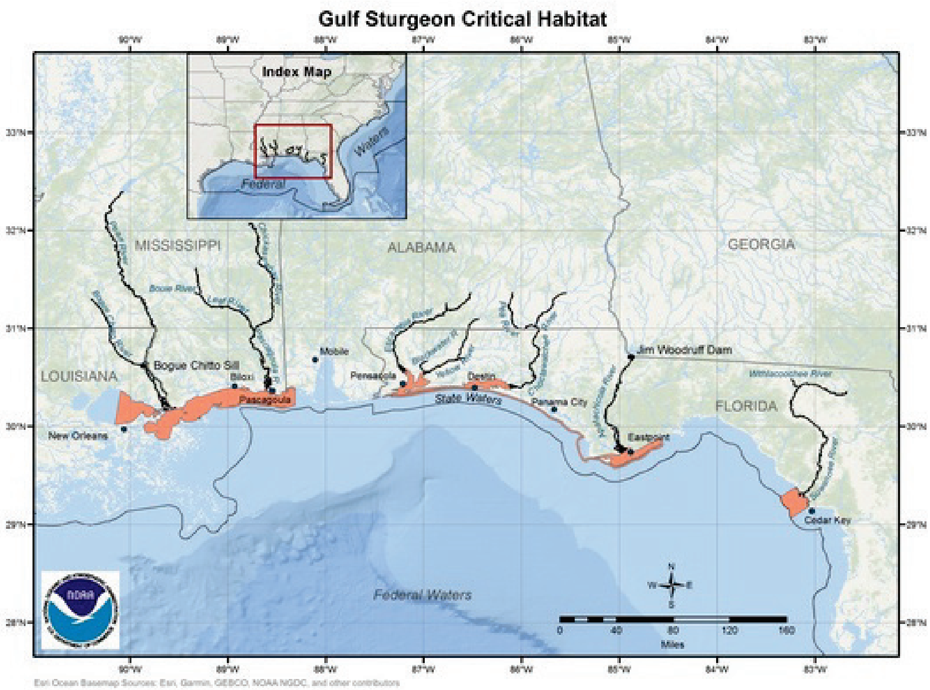


Figure 13.18. Gulf sturgeon critical habitat. Estuarine and marine critical habitat are the orange areas, while the critical habitat in rivers and tributaries are the black lines. Source: NOAA https://www.fisheries.noaa.gov/resource/map/gulf-sturgeon-critical-habitat-map-and-gis-data (accessed May 7, 2021).

13.3.2.2.2 Mollusks

Unlike fish, many aquatic species such as bivalves, have limited ability to move to more suitable habitat following disturbance or in response to pollutants. For example, freshwater mussels are imperiled around the world and underlying reasons for the extinctions or declines in abundance include a variety of factors such as habitat degradation, water quality degradation, climate change, introduction of nonnative species, declines in fish hosts, and overexploitation (Ferreira-Rodríguez et al., 2019). Bivalves are filter feeders[13] and, as noted above, many endangered freshwater mussels are not adapted to high sediment concentrations, which may occur following tillage. An experimental enclosure study in Kentucky rivers found mussel growth rate was negatively correlated with higher row crop agriculture and agricultural contaminants, including nitrate and the pesticides atrazine, metolachlor, and dicamba (Haag et al., 2019). Strong evidence for species with critical habitats likely adversely affected by atrazine was found for 14 aquatic invertebrates and 8 of these were mussels (Supplemental Table 13.4).

[13] Filter feeders strain suspended matter and particles in the water to obtain food.

The reproduction of pearly mussels,[14] such as the endangered pink mucket (*Lampsilis abrupta*), involves the release of sperm by males, fertilization of eggs in females downstream, and attachment of the larvae to host fish to complete their lifecycle. In addition to the more direct impacts of general agricultural land use on flow and water quality, changes to the fish community can indirectly impact endangered mussels if the specific host fish species is no longer abundant or present. The distribution of many T&E mussels is limited to small stretches of rivers, such as those in Missouri (Figure 13.19).

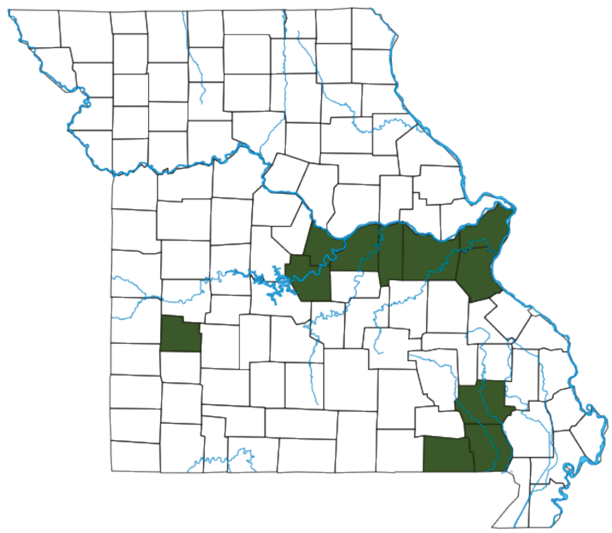


Figure 13.19. Distribution map of the endangered pink mucket mussel (*Lampsilis abrupta*) in Missouri. Source: Missouri Dept. of Conservation, https://nature.mdc.mo.gov/discover-nature/field-guide/pink-mucket.

13.3.3 Attribution to the RFS

The chapter material above focuses on the effects of corn and soybean production and biofuels in general but did not address the effect of the RFS Program specifically. For instance, in the review of the literature (section 13.3.1), studies did not examine how the RFS Program affected corn or soybean production and therefore how it might have affected aquatic ecosystems and biodiversity, but instead focused on the effects of agriculture in general. Where possible, data are from regions (e.g., TPL ecoregion, MORB) that roughly coincide with biofuel feedstock production.

Chapter 5 quantified how much land use change is estimated to be occurring in the United States from all causes and Chapter 6 estimated the subset of that estimated to be attributable to corn ethanol production associated with the RFS Program specifically. Lark et al. (2020) estimated a total of 10.09 million acres of non-cultivated land—mostly grasslands like pasture and CRP grasslands—converted to cropland between 2008 and 2016 in the contiguous U.S. (roughly 1 million acres per year). Based on

[14] Pearly mussel is a common name for about 1,000 species of large (2- to 30-centimeter) bivalves that live in the sediments of rivers, streams, and lakes worldwide; humans have gathered pearly mussels for their meat, pearls, and mother-of-pearl shells for millennia (see Strayer et al. (2004)).

Chapter 6 (section 6.3 and 6.4), 0 to 1.9 million acres of new cropland or approximately 0 to 20% of the total from all causes in Lark et al. (2020) are estimated to be attributable to corn ethanol production associated with the RFS Program during this historical time period (Table 6.10 and 6.11). Thus, the effects from the RFS Program on aquatic ecosystems from the expansion of cropland alone may be up to approximately 0–20% of the results presented in section 13.3.2. As noted in Chapter 6, most years the estimate is no effect of the RFS Program above other factors, but in some years these other factors may not have been sufficient by themselves to support the biofuel volumes and under some assumptions the effect of the RFS Program may have been as high as 1.9 million acres in 2016. Annual estimates for all years include zero, and thus a range of potential effects from the RFS Program is estimated. Notably, these estimates do not include the potential effect of the RFS Program on soybean acreage, since EPA was unable in this report to estimate the land use associated with soy biodiesel volumes (see Chapter 7). Moreover, as noted in section 13.3.2, the actual crops grown on newly converted lands are likely some mixture of the three scenarios examined, with the actual effect likely lower than the high estimate (S2). However, as noted above, many of these watersheds are already in the most disturbed category (Figures 13.14 and 13.15), and although the incremental effect from recent (2008–2016) agricultural expansion from all causes that might have been due to the RFS Program specifically appears to be minor, this represents additional strain on already strained aquatic ecosystems.

13.3.4 Conservation Practices

Opportunities for offsetting negative effects on aquatic biodiversity include setting numerical targets for pollutant criteria and then managing for those targets. Many states do not have numeric nutrient criteria (Table 13.2 and Figure 13.16). Without clearly defined thresholds for many waterbodies, it is difficult to ascertain the environmental effects of biofuels generally, or for the RFS Program specifically, related to increased fertilizer use for feedstock production. Numerical targets for pollutant criteria would make conservation needs less open to interpretation and help drive management decisions that protect aquatic biodiversity. To mitigate the impacts of disturbance to water flow for irrigating feedstock fields, restoring flows that mimic natural hydrologic variability (environmental flows) and removing dams could both be ways to improve connectedness of waterways and the movement of aquatic organisms to suitable habitat (Reid et al., 2019). A review of the literature on low flow ecosystems in agricultural watersheds found that management practices can help overcome some of the negative impacts of agriculture on flow regime. For example, ditch management can account for shading and channel characteristics to promote aquatic habitat (Rideout et al., 2022). In some agricultural landscapes, farm ponds or dams that are managed to improve vegetation structure and water quality could provide habitats that support aquatic invertebrate diversity and even some species of pollinator insects (Westgate et al., 2022; Walton et al., 2021; Law et al., 2016). Without sufficient biotic data, it is difficult to track biodiversity losses or gains. Using environmental DNA methods for high-throughput monitoring of aquatic communities, as well as targeted detection of threatened or endangered species, are opportunities for efficiently tracking biodiversity over large areas and prioritizing conservation efforts in biodiversity hotspots (Deiner et al., 2016).

Conservation tillage, cover crops, and other conservation practices (see Chapters 3 and 12) reduce soil erosion and runoff of sediment, nutrients, and pesticides into waterbodies. Adoption of some of these practices are widespread (e.g., conservation tillage), while others are not (e.g., cover crops) (see Chapter 3, section 3.2.1). The USDA Conservation Effects Assessment Program (CEAP) is a large multiagency effort to quantify the environmental effects of conservation practices and programs and develop the science base for managing the agricultural landscape for environmental quality. The most recent CEAP report (USDA NRCS, 2022) did not address effects on aquatic biota, which is the focus of this Chapter, but effects on water quality and irrigation were addressed (see Chapter 10, sections 10.3.4 and 11.3.4 for water quality and irrigation, respectively). Where conservation efforts are in place, there is a need for improving the ability to measure their effectiveness in protecting aquatic biodiversity. Although there is a large evidence base of primary literature and reports on vegetated strips (e.g., agricultural field margins and riparian buffer strips) used to mitigate habitat loss, soil erosion, and run off of nutrients and pesticides, there is a knowledge gap when it comes to the outcomes and effectiveness of these conservation practices related to aquatic biodiversity (Haddaway et al., 2018). The difficulties of measuring the effectiveness of conservation practices are compounded by other drivers such as a changing climate. A study in the Midwest United States found that fish species were less sensitive to changes in land management practices than to changes in climate regime, which highlights the importance of considering multiple drivers in restoration efforts (Triana et al., 2021).

13.4 Likely Future Impacts

As noted in Chapter 2, corn ethanol and soy biodiesel will likely remain the dominant biofuels in the near future considered in this report (out to 2025). Whether grasslands continue to be converted to corn or soybeans will in large part determine the magnitude of future effects to aquatic ecosystems, since the largest impacts to aquatic ecosystems generally occur from this land cover-land management (LCLM) shift and the associated changes in flow, soil erosion, and runoff of sediments, nutrients, and pesticides to these ecosystems. Lark et al. (2020) reported a slowdown nationally in cropland expansion since 2011, especially since 2015 [Figure 2 in Lark et al. (2020)], in agreement with other sources from Chapter 5, and in connection with reaching the E10 blend wall in 2013. When the Set Rule was finalized on June 21, 2023, EPA published biofuel volumes along with estimates of cropland expansion (see Chapter 6, section

6.5).[15] EPA estimated the biofuel volumes in the Final Set Rule could potentially lead to an increase of as much as 2.65 million acres of cropland in the Midwest by 2025, mostly from soybean expansion in the Midwest, but to a lesser extent from corn expansion in the Midwest and canola expansion in North Dakota. Still, the likely future effects of the RFS Program are highly uncertain. While the projected cropland expansion for 2023–2025 is slightly larger than the estimated historical cropland expansion from the RFS Program, the authors cannot say with reasonable certainty that any particular ecosystem or biodiversity will be impacted, due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes. More information on where these changes are anticipated, and what agricultural practices are employed, are needed to better constrain estimates of the likely future effects to aquatic ecosystems. Even if there are no changes in corn or soybean cultivation in the near future, effects from historical changes could take years to propagate through the connected hydrologic-ecological system. Excess nutrients continue to disturb ecosystems within the Mississippi River Basin and in the northern Gulf of Mexico into which it drains. Strengthened N and P mitigation, altered agriculture practices, and reduction in carbon and nutrient footprints are key to the recovery of these systems (Rabalais and Turner, 2019). Moreover, the potential effects of canola expansion require investigation. Additional information may be found in the associated docket for the Set Rule (EPA-HQ-OAR-2021-0427).

13.5 Comparison with Petroleum

Notably, many of the impacts of biofuels as described in this chapter also occur from the production and use of petroleum. When comparing both industries, it is important to examine impacts across multiple potential pathways. First, the amount of land required for each industry can affect aquatic ecosystems through runoff of sediments, nutrients, and other materials. As discussed in Chapter 12, estimates generally suggest that biofuels require more land currently on a per energy basis than petroleum (see section 12.5 for more details). Projecting in the future, however, Trainor et al. (2016) estimate

[15] On July 26, 2022, the United States District Court for the District of Columbia filed a consent decree, which after stipulations from the parties, required EPA to sign a notice of proposed rulemaking (NPRM) to establish 2023 volumes for the RFS Program on or before November 30, 2022, and to sign a notice of final rulemaking finalizing 2023 volumes by June 21, 2023. *Growth Energy v. Regan et al.*, No. 1:22-cv-01191, Document Nos. 12, 14, 15. EPA's proposed RFS volumes for 2023-2025, 87 FR 80582 (Dec. 30, 2022) available in Docket No. EPA-HQ-OAR-2021-0427 at https://www.regulations.gov, were finalized on June 21, 2023 and published in the Federal Register on July 12, 2023 (88 FR 44468). The estimates of cropland expansion from the RFS Program took considerable time and effort to develop; thus, they are based on the proposed volumes from November 30, 2022, rather than the final volumes from June 21, 2023. Because of that, the acreages discussed here are merely illustrative. However, since the total volumes from crop-based feedstocks changed fairly little between the draft and final rules, they are still relevant here (see Chapter 6, section 6.5).

biofuels and petroleum will become similar in land requirements if the spacing between oil wells is also included.

The more land required for biofuels generally will increase the loading of nutrients and other materials to aquatic ecosystems. The lifecycle assessments from the National Renewable Energy Lab (NREL), discussed in Chapter 10, concluded corn ethanol production exceeded gasoline for both total impacts and impacts on a per megajoule basis for eutrophication potential and freshwater ecotoxicity (see section 10.5 for further details). Compared to diesel, soy biodiesel had a smaller effect on eutrophication potential in total (because soybeans receive less fertilizer and less biodiesel is consumed than diesel), but a larger effect on freshwater ecotoxicity (because of pesticides applied to soybean crops). The same analysis estimated soybean biodiesel had a larger effect per megajoule than diesel. In addition to increased loads of pesticides and nutrients, changes to flow and increased loads of sediment to water bodies are major chronic impacts of agriculture discussed in this chapter.

Aquatic ecosystems can also be impacted through acidification from nitrogen and sulfur deposition. Petroleum production and its supply chain have a relatively smaller contribution (per megajoule [MJ]) to aquatic acidification potential from nitrogen and sulfur deposition compared to corn ethanol production due to low emissions of NO_x and SO_x (Chapter 8, section 8.5). Overall petroleum has a larger impact because more gasoline is consumed than ethanol, but per megajoule, ethanol has a larger effect due to farming and the farming supply chains. Acidification is harmful to aquatic biodiversity because it affects organisms at all trophic levels (Wright et al., 2018). Few or no fish species are found in poorly buffered lakes and streams with pH near 5.0 or lower (U.S. EPA, 2020), There has been gradual recovery from acidification in response to reduced sulfur and nitrogen deposition over the past few decades (Austnes et al., 2018), but chemical recovery has not been consistent with biological recovery (Gray et al., 2016). Reasons for absent or delayed biological recovery include the strong effect of episodic acidification events on sensitive species (Schneider et al., 2018; Kowalik et al., 2007) or that acidification fundamentally alters aquatic food webs (Gray et al., 2016). Aquatic organisms are sensitive to metals, which can accumulate in waterbodies and affect ecosystem diversity and health (Wright et al., 2018). Above certain concentrations of metals, there is significantly more mortality or sublethal effects such as damage to fish gills (Pandey et al., 2008), disruption of photosynthesis, and cell deformities. For example, there were more abnormal diatom cells at seven times above background metal concentrations (Morin et al., 2012).

Lastly, in contrast to chronic impacts, acute events like spills may occur during the petroleum and biofuel lifecycles. These events are difficult to capture in these lifecycle analyses. However, biofuels themselves likely have limited direct environmental effects, especially for biodiesel made from animal fats or plant oils, because these fats and oils break down almost completely in 21–28 days in aquatic environments (Sendzikiene et al., 2007; Zhang et al., 1998), a faster rate relative to petroleum-based diesel. There may still be potential problems with biodiesel, including water-soluble fractions of biodiesel, because these can have acute and chronic toxicity to aquatic organisms (Müller et al., 2019; Pikula et al., 2019; Khan et al., 2007). When biodiesel enters water, transesterification of biodiesel undergoes a hydrolytic reversion process that produces methanol, which is toxic to organisms (Leite et al., 2011; Nascimento et al., 2009). Information about biodiesel's effects in aquatic environments is currently limited to laboratory studies.

13.6 Horizon Scanning

Next generation biofuel feedstocks include cellulosic feedstocks, such as corn stover and switchgrass. Corn stover includes crop byproducts, such as cobs, husks, leaves, and other detritus. These byproducts often remain on fields after harvest, until wind or water transports them to adjacent streams. The corn byproducts can remain in the stream at baseflow and are available as food for aquatic invertebrates, but this may be detrimental to some invertebrates if the corn leaves they consume are from transgenic corn modified to produce Bt endotoxin (Rosi-Marshall et al., 2007). Removal of corn stover from fields for biofuel production may benefit aquatic invertebrates by reducing their exposure to Bt endotoxin. As described in Chapter 10, water quality may significantly improve in areas that grow switchgrass in future scenarios of biofuel feedstock production in the MORB and the Upper Mississippi River Basin by reducing nutrient loads (Wu and Ha, 2017; Wu and Zhang, 2015). Reductions in nutrient loading into waterbodies would allow aquatic ecosystems to start recovering from years of eutrophication and its consequences, such as HABs and hypoxia (Bocaniov et al., 2016). Biological response to nutrient reduction may be slow, especially in shallow lakes with high historical nutrient loads (Reavie et al., 2017), because water column and sediment nutrient concentrations need to reach a new equilibrium before signs of recovery may be observable (Jeppesen et al., 2007).

Also on the horizon are third generation biofuels, such as microalgae, which continue to spur much interest in research and development but are not yet productive at economic scales competitive with petroleum (Correa et al., 2019). Algae could be used to produce several biofuels, such as biodiesel, bioethanol, biogas, and biohydrogen (Schenk et al., 2008). For aquatic ecosystems, the potential advantages of algae-based biofuel production include less pollution from excess nutrients because algae can take up nitrogen, phosphorus, sulfate, and silicon from human or animal wastes (i.e., wastewater remediation); and less pollution from pesticides compared to terrestrial crops (Menetrez, 2012). However, there are still many unknowns about the environmental impacts of algae-based biofuels when production becomes scaled up. Depending on the production process and types of algae cultivated, unintentional consequences could arise from release of algal toxins or genetically modified algal strains into the environment (Slade and Bauen, 2013). Fortunately, risks from genetically modified algae may be low because algal strains grown for biofuel production generally require careful maintenance of water chemistry and constant care to prevent contamination or infection that could cause the population to collapse. Thus, the likelihood that these algal strains could survive and interact with other aquatic organisms in the environment outside of culturing facilities may be low. Best management practices for minimizing the potential impacts of algae-based biofuels would require production processes to prioritize environmental monitoring and to remediate, rather than exacerbate, problems with eutrophication, HABs, and hypoxia in aquatic ecosystems.

Beyond other potential biofuel feedstocks, the effects of multiple stressors on aquatic biodiversity will be amplified by climate change (Dudgeon, 2019). A thorough discussion of the effects of climate change on aquatic biodiversity is beyond the scope of this chapter (for more information see Knouft and Ficklin, 2017; Wells et al., 2015). Direct and indirect effects of increased temperatures were major drivers of cyanobacterial toxin concentrations and number of toxin variants produced by HABs (Mantzouki et al., 2018). In some cases (e.g., cover crops), the conservation practices discussed in section 13.3.4 can promote aquatic biodiversity in waterbodies within and downstream of agriculturally intense regions. In the face of multiple human-induced environmental disturbances, including climate change, conservation of biodiversity is associated with greater resilience and sustainability of ecosystem services (Schindler et al., 2015).

13.7 Synthesis

13.7.1 Chapter Conclusions

- Many watersheds in the Midwest and other U.S. regions have historically been impacted by agriculture generally and by crops used for biofuels specifically, but the incremental effect from recent (2008–2016) agricultural expansion from all causes, including any potential impact from the RFS Program specifically, appears to be minor in comparison.
- Water demand for feedstock production may change stream flow and alter flow patterns that are important for supporting fish diversity.
- Pesticides used in feedstock production, including atrazine, glyphosate, and neonicotinoids, have direct toxicity to some nontarget organisms as well as a variety of sublethal, indirect environmental effects on aquatic ecosystem health and biodiversity. Based on overlap of species ranges and critical habitat with atrazine usage, EPA judged atrazine was likely to adversely affect 180 out of 207 federally listed (i.e., threatened and endangered) aquatic invertebrate species assessed, including mussels, snails, shrimp, amphipods, water beetles, and crayfish.

- Based on data from nationally representative surveys of the nation's wadeable stream miles in 2004 and about 10 years later in 2013–2014, biological and nutrient conditions worsened in the ecoregions roughly coinciding with areas of corn and soybean production compared to the rest of the continental United States. National surveys found that benthic macroinvertebrates were nearly twice as likely to be in poor condition in waterbodies with high nutrient concentrations and/or excess sediments.
- For the scenarios examined in the modeling study on agricultural expansion due to all causes from 2008 to 2016, the flow-weighted nutrient concentrations increased by less than 5% on average across the MORB. For the scenario of conversion from grassland to corn/soy rotation, only 0.11% of watersheds in the MORB had increases in nutrient concentrations that were more than 10% of the baseline scenario. Thus, increases in nutrient concentrations that may be attributable to the RFS Program are unlikely to result in new exceedances of current state numeric nutrient criteria in agricultural regions of the United States, such as the MORB. Total effects may be larger or smaller because this study only included effects from agricultural expansion (expected to be the largest source) and not agricultural intensification or recent improvements in tillage practices.
- Demand for biofuel feedstocks may contribute to increased frequency and magnitude of harmful algal blooms and hypoxia. Altered food webs and changes in nutrient cycling can trigger feedback loops that make it difficult to prevent or mitigate the effects of harmful algal blooms and hypoxia on aquatic ecosystems.
- Adoption and expansion of sustainable conservation practices and technologies remain critically important to reducing impacts on aquatic ecosystems by restoring flow and decreasing loads of nutrients, sediment, and pesticides to levels that are less harmful to aquatic organisms.
- The likely future effects of the RFS Program are highly uncertain. While the projected cropland expansion for 2023–2025 is slightly larger than the estimated historical cropland expansion from the RFS Program, the authors cannot say with reasonable certainty that any particular aquatic ecosystem will be impacted due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes.

13.7.2 Conclusions Compared with the RTC2

The in-depth literature review and additional findings from this chapter strengthen and build upon the conclusions specific to aquatic ecosystem health and biodiversity in the second Report to Congress (U.S. EPA, 2018). The scientific literature continues to emphasize the negative effects of altered water flow and water quality on aquatic biodiversity. Recent studies highlight the negative effects of pesticide use on aquatic organisms, especially macroinvertebrate species. Neonicotinoid insecticides are especially a concern because of their ubiquitous use as corn seed coatings and the stable nature of the chemical compounds in water, resulting in toxicity to aquatic insects. On threatened and endangered (T&E) species, this chapter advances the fundamental understanding beyond that of the RtC2. It is now clear that grassland habitat loss, including to corn and soybeans, has occurred in areas overlapping with the ranges of T&E species. How much of this is due to biofuels and the RFS Program remains an open question, but some fraction of that acreage especially in later years may be due to the RFS Program. Conservation practices can reduce the negative effects of crop production on aquatic biodiversity, but without clearly defined thresholds for many waterbodies, it is difficult to ascertain the environmental effects of biofuels generally or due to the RFS Program specifically related to increased fertilizer use for feedstock production. Setting numerical targets for state pollutant criteria and managing for those targets would make conservation needs less open to interpretation and help drive management decisions that protect aquatic biodiversity.

13.7.3 Uncertainties and Limitations

- There is a lack of studies that target the interactive effects of land use land management change and feedstock production on aquatic habitats.
- Many states do not have numeric nutrient criteria for aquatic ecosystems. Without clearly defined nutrient concentration thresholds for many waterbodies, it is difficult to ascertain the environmental effects of biofuels generally or the RFS Program specifically related to increased fertilizer use for feedstock production.
- Current understanding of the effects of pesticides used in feedstock production on nontarget aquatic organisms is often limited to laboratory studies on model organisms, although data for neonicotinoids also include microcosm studies, as well as laboratory studies on sensitive nonstandard aquatic test organisms including species in the Ephemeroptera, Plecoptera, and Trichoptera orders. Monitoring data on aquatic concentrations of most pesticides are also limited, although for neonicotinoids they are comparatively robust. Understanding how mixtures of pesticides together with other stressors impact aquatic organisms and ecosystems remains a challenge.
- While increased nutrient loads from general agriculture contributes to harmful agal blooms and hypoxia, there is ongoing research on how other factors (e.g., climate, legacy nutrients) may exacerbate impacts on aquatic organisms and the waterbodies they inhabit.

- There are limited data on the implementation and effectiveness of conservation practices to protect aquatic ecosystems near feedstock production areas and biofuel refineries.

13.7.4 Research Recommendations

- More research is needed to quantify with more certainty the effects of land use/land management practices attributable to the RFS and biofuel feedstock production on aquatic biodiversity, including pesticide usage, nutrient pollution, sedimentation, and changes in water flow.
- Trends in neonicotinoid concentrations in surface waters will likely have a positive correlation with trends in seed purchases for general agriculture and biofuel feedstocks, just as trends in concentrations of other pesticides will also tend to correlate with trends in their usage. There is a critical need for research outside of the laboratory on how neonicotinoids affect nontarget organisms, especially aquatic invertebrates in the field or in mesocosms, to assess effects on populations, communities, and ecosystem structure and function.
- Research on legacy nutrients attributable to increased feedstock production is needed to understand the extent of nutrient management measures and lag times needed to observe improvements in aquatic ecosystem health and diversity, especially less frequent harmful algal blooms and hypoxic events.
- Environmental benefit and cost analyses are needed with respect to biofuel refining processes and facilities. While byproducts like glycerin can be captured to make additional products, other wastes (e.g., methanol, trace metals) may enter waterbodies from refineries as point sources or runoff and have potential toxic effects on aquatic organisms.

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13.S Supplemental Tables for Chapter 13

Supplemental Table 13.1. Fish and aquatic invertebrate acute and chronic end points (µg/L) from EPA ecological risk assessments of top corn and soybean pesticides (U.S. EPA, 2017b).

Chemical	Lowest freshwater fish EC50/LC50	Lowest freshwater fish NOAEC/LOAEC	Lowest aquatic invertebrate EC50/LC50	Lowest aquatic invertebrate NOAEC/LOAEC
acetochlor	380 (*O. mykiss*)	130/270 (*O. mykiss*)	8200 (*D. magna*)	22.1/42.7 (*D. magna*)
atrazine	5300 (*O. mykiss*)	5/50 (*O. latipes*)	720 (*C. tentans*)	60 (*G. fasciatus*)
chlorpyrifos	1.8 (*L. macrochirus*)	0.57 (*P. promelas*)	0.10 (*D. magna*)	0.04 (*D. magna*)
clothianidin	>101500 (*O. mykiss*)	9700/20000 (*P. promelas*)	1.85 (*C. dilutus*)	<0.05/0.05 (*C. riparius*)
fipronil	20 (*L. macrochirus*)	6.6/15 (*O. mykiss*)	0.22 (*S. vittatum*)	9.8/20 (*D. magna*)
glyphosate	43000 (*L. macrochirus*)	25700 (*P. promelas*)	53200 (*C. plumosus*)	9220 (*C. plumosus*)
imidacloprid	229000 (*O. mykiss*)	<1200/1200 (*O. mykiss*)	0.65 (*E. longimanus*)	0.01 (*C. horaria*)
lambda-cyhalothrin	0.078 (*L. idus*)	0.031/0.062 (*P. promelas*)	0.007 (*G. pulex*)	0.002/0.0035 (*D. magna*)
methoxyfenozide	>4200 (*O. mykiss*)	530/1000 (*P. promelas*)	60 (*C. riparius*)	3.1/6.3 (*C. riparius*)
metolochlor & metolochlor-S	3800 (*O. mykiss*)	6000/10300 (*L. macrochirus*)	25.1 (*C. dubia*)	3200 (*D. magna*)
propargite	43 (*O. mykiss*)	14/21 (*L. macrochirus*)	14 (*D. magna*)	4/13 (*D. magna*)
pyraclostrobin	6.2 (*O. mykiss*)	2.35/6.42 (*O. mykiss*)	15 (*D. magna*)	4/8 (*D. magna*)
thiamethoxam	>114000 (*O. mykiss*)	20000 (*O. mykiss*)	20 (*C. dipterum*)	0.43/1.4 (*C. dipterum*)

L = liters; µg = micrograms

Supplemental Table 13.2. EPA aquatic-life benchmarks (µg/L) for top corn and soybean pesticides (U.S. EPA, 2017b).

Chemical	Fish, acute	Fish, chronic	Invertebrates, acute	Invertebrates, chronic	Nonvascular plants, acute	Vascular plants, acute
acetochlor	190	130	4100	22.1	1.43	3.4
atrazine	2650	5	360	60	< 1	4.6
chlorpyrifos	0.9	0.57	0.05	0.04	140	
clothianidin	> 50750	9700	11	0.05	64000	> 280000
fipronil	41.5	2.2	0.11	0.011	140	> 100
glyphosate	21500	25700	26600	49900	12100	11900
imidacloprid	114500	9000	0.385	0.01		
lambda-cyhalothrin	0.039	0.031	0.0035	0.002	> 310	
methoxyfenozide	> 2100	530	28.5	3.1	> 3400	
metolochlor & metolochlor-S	1900	30	550	1	8	21
propargite	40.5	16	7	9	19.4	75000
pyraclostrobin	3.1	2.35	7.85	4	1.5	1197
thiamethoxam	> 57000	20000	17.5	0.74	> 99000	> 90200

L = liters; µg = micrograms

Supplemental Table 13.3. Select environmental fate and transport properties from EPA ecological risk assessments of top corn and soybean pesticides (U.S. EPA, 2017b).

Chemical	K_{ow}	BCF, whole fish	K_{OC} (L/kg)	hydrolysis $t_{1/2}$ (d) at pH 7	aerobic soil $t_{1/2}$ (d)	aerobic aquatic $t_{1/2}$ (d)
acetochlor			139	stable	13.3 (upper CL of mean)	13.3 (mean)
atrazine	501.2		75 (mean)	>=742	146	38 (river); 155 (pond)
chlorpyrifos	50118.7	2727	6070	72	76.9	153.8
clothianidin	13.2		311-582	stable	144-5357	178-182
fipronil	3162.3	380	427-1248	stable	128-308	14.5-35.5
glyphosate	<0.001		1600-33000	stable	1.8-109	14.1-518
imidacloprid	3.7		266	stable	139-608	
lambda-cyhalothrin	10^7	4600	333,200 (mean)	stable	46.2	21.1-52.9
methoxyfenozide	5248	9.9-10.5	490 (mean)	stable	336-1100	387, 963
metolochlor & metolochlor-S	1122	69	21.6-119	stable	13.9-67	47
propargite	501187	775	5293-95918	75	53.3, 168	38
pyraclostrobin	15100		9304 (mean)	stable	81.5-330	8.4, 26.4
thiamethoxam	0.7		33-178	stable	34.3-464	16.2-35.1

d =days; kg = kilograms; L = liters

Supplemental Table 13.4. Threatened and endangered aquatic organisms with strongest evidence of likely adverse effect of atrazine on the species' critical habitat (modified from Appendix 4-1 of U.S. EPA (2021)).

Taxa	Scientific Name	Common Name
Aquatic Invertebrates	*Elliptoideus sloatianus*	Purple bankclimber (mussel)
Aquatic Invertebrates	*Lasmigona decorata*	Carolina heelsplitter (mussel)
Aquatic Invertebrates	*Branchinecta lynchi*	Vernal pool fairy shrimp
Aquatic Invertebrates	*Lepidurus packardi*	Vernal pool tadpole shrimp
Aquatic Invertebrates	*Assiminea pecos*	Pecos assiminea (snail)
Aquatic Invertebrates	*Pyrgulopsis roswellensis*	Roswell springsnail
Aquatic Invertebrates	*Juturnia kosteri*	Koster springsnail
Aquatic Invertebrates	*Gammarus desperatus*	Noel's amphipod
Aquatic Invertebrates	*Pleurobema strodeanum*	Fuzzy pigtoe (mussel)
Aquatic Invertebrates	*Quadrula cylindrica cylindrica*	Rabbitsfoot (mussel)
Aquatic Invertebrates	*Villosa choctawensis*	Choctaw bean (mussel)
Aquatic Invertebrates	*Lampsilis rafinesqueana*	Neosho mucket (mussel)
Aquatic Invertebrates	*Elliptio spinosa*	Altamaha spinymussel
Aquatic Invertebrates	*Pleuronaia dolabelloides*	Slabside pearly mussel
Fish	*Etheostoma sellare*	Maryland darter
Fish	*Ptychocheilus lucius*	Colorado pikeminnow (=squawfish)
Fish	*Etheostoma boschungi*	Slackwater darter
Fish	*Notropis mekistocholas*	Cape Fear shiner
Fish	*Menidia extensa*	Waccamaw silverside
Fish	*Scaphirhynchus suttkusi*	Alabama sturgeon
Fish	*Ictalurus pricei*	Yaqui catfish
Fish	*Gila purpurea*	Yaqui chub
Fish	*Eremichthys acros*	Desert dace
Fish	*Cyprinella formosa*	Beautiful shiner
Fish	*Notropis simus pecosensis*	Pecos bluntnose shiner
Fish	*Xyrauchen texanus*	Razorback sucker
Fish	*Catostomus warnerensis*	Warner sucker
Fish	*Percina antesella*	Amber darter
Fish	*Percina jenkinsi*	Conasauga logperch
Fish	*Notropis girardi*	Arkansas River shiner
Fish	*Notropis topeka*	Topeka shiner
Fish	*Catostomus discobolus yarrowi*	Zuni bluehead sucker
Fish	*Notropis oxyrhynchus*	Sharpnose shiner
Fish	*Acipenser medirostris*	green sturgeon
Fish	*Crystallaria cincotta*	Diamond darter
Fish	*Notropis buccula*	Smalleye shiner

14. Wetland Ecosystem Health and Biodiversity

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Key Findings

- The area of wetlands converted to cropland from 2008 to 2016 in the United States is estimated at nearly 275,000 acres, with most losses occurring in the Prairie Pothole Region. Given the lack of national or regional datasets to track changes in acreage attributable to the Renewable Fuel Standard (RFS) Program, the extent of wetland losses solely attributable to the RFS cannot be more accurately estimated in this Third Triennial Report to Congress (RtC3).
- Small, seasonal wetlands are being lost at the fastest rate. The loss and consolidation of small wetlands to promote crop production has negatively impacted amphibians, invertebrates, and other aquatic species that depend on shallow water depths for reproduction. Shifts to longer hydroperiods in large or consolidated wetlands have more uniform (less diverse) invertebrate communities and can support fish that prey on insects and amphibians.
- Small wetlands and ponds are primary sources of water for aquifer recharge in the Northern Prairies. Recent studies in the Canadian Prairie Pothole Region found that while permanent ponds and wetlands are sources for recharge to aquifers, wetlands with surface water ponds that dry out every year play the dominant role in groundwater replenishment.
- While some Endangered Species Act-listed and other waterbirds have declined, waterfowl (ducks, geese, swans) as a group have not experienced declines over the past decade, possibly due to availability of food (grains), increased precipitation, and the interspersion of ponded waters and agricultural fields along migration routes.
- Shifts to corn and soybean production have resulted in more frequent application of chemicals, including pesticides and fertilizers. Increased usage of neonicotinoid insecticides is of particular concern because of their high toxicity to invertebrates, which are important food sources for wetland-dependent taxa.
- Nationally, net wetland gains and losses are not distributed evenly across wetland classes. Since 2007, the nation has lost a net total of 120.3 thousand acres of palustrine (marsh-like) wetlands and gained a net total of 205.9 thousand acres of lacustrine (lake-like) habitats in the conterminous United States. The wetlands within these classes support different species and perform different ecosystem functions. Shifts from palustrine to lacustrine wetlands have resulted in the loss or impairment of functions that impact watershed hydrology, water quality, and water quantity.
- Evidence from the Prairie Pothole Region in the United States and Canada indicates that trends in larger wetland size, shifts to lakes and ponds (vs. vegetated wetlands), and

prolonged and more frequent flooding are due to the combined effects of climate change and increased wetland ditching and consolidation. These trends are highly correlated with increased annual precipitation, which is projected to continue.

- The likely future effects of the RFS Program are highly uncertain. While the projected cropland expansion for 2023–2025 is slightly larger than the estimated historical cropland expansion from the RFS Program, it cannot be said with reasonable certainty that any particular wetland will be impacted, due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes.

Chapter Terms: aquifer, baseflow, biodiversity, biogeochemical cycling, deepwater habitats, ecosystem services, ecosystem, evapotranspiration, fats, oils, and greases (FOGs), groundwater recharge, hydroperiod, lacustrine, natural regulation, palustrine, through flow, water balance, watershed, wetlands, willingness-to-pay (WTP)

14.1 Overview

14.1.1 Background

This chapter updates the assessment of the impacts to date and likely future effects from biofuels and the RFS Program on wetland ecosystem health and biodiversity. It focuses on the feedstock production stage for the two dominant feedstocks (corn and soybean, see Chapters 2 and 3), as these feedstocks are the largest contributor to impacts to wetlands currently. The other two biofuels have a comparatively small effect on U.S. wetlands, with no documented impacts from fats, oils, and greases (FOGs), and no known direct impacts from sugarcane that is cultivated in Brazil and exported to the United States other than climate effects of deforestation for its production, which can influence water levels in wetlands and other aquatic ecosystems. Much of the pertinent research has been done in the Midwest, in low topographic relief landscapes dominated by temperate grasslands and depressional ("pothole") wetlands. Inland (non-coastal) wetlands account for 94% of wetlands in the United States, and many mechanisms of ecosystem change affecting wetlands in the Midwest are applicable to wetlands associated with high-intensity corn and soybean feedstock production in other parts of the country. Therefore, this chapter also provides a general framework for evaluating emerging research on wetlands and feedstock production for future biofuel reports to Congress.

Wetland ecosystems are transitional between terrestrial and aquatic ecosystems. Federal definitions of wetlands vary by agency and program, but all refer to common attributes of wetlands identified by the U.S. Fish and Wildlife Service (Cowardin et al., 1979): (1) inundation or saturation by surface water and/or groundwater at a frequency sufficient to support, at least periodically, plants adapted to wet environments; (2) predominance of undrained hydric soils; or (3) presence of nonsoil substrate saturated by shallow water at some time during the growing season. Wetland ecosystems support uniquely diverse communities of plants and animals, including 5,000 plant species, 190 amphibian species, and a third of all bird species in the United States (Flynn, 1996). They provide critical habitat for many federally listed threatened and endangered species. Of the federally listed species, in 1991 approximately 50% of fish species, 33% of bird species, 25% of plant species, and 17% of mammal species were species found in wetlands (Niering, 1988). Wetlands intercept and store agricultural runoff, trapping sediments and removing nutrients and contaminants that contribute to harmful algal blooms and pollute streams, lakes, reservoirs, and coastal waters. By storing precipitation, snowmelt, and storm runoff, wetlands also are effective at preventing or mitigating floods.

Realistic evaluation of current and future risks to water quality and quantity resulting from loss of wetland resources in the United States can only be evaluated in context with widespread historical wetland losses. Prior to European settlement, the area that became the conterminous United States had approximately 221 million acres of wetlands (Dahl 1990). By the mid-1980s more than half of those wetlands (117 million acres) had been lost, primarily to agriculture (87% nationwide; Figure 14.1) (Dahl and Johnson, 1991; Dahl 1990). Net losses in some states exceeded 90% (Fretwell et al., 1996). Wetlands lost in more recent years (e.g., since 2007) are likely to have had relatively greater ecosystem impacts because the functions of remaining wetlands, while only partially mitigating for adverse effects of cumulative losses, have greater value for maintaining sustainable freshwater resources to support environmental and human needs, including agriculture. Large historical losses have reduced the resilience

Figure 14.1. States with notable wetland loss, 1780s to mid-1980s. Source: USGS Water Supply Paper 2425, Figure 2, modified from Dahl (1990).

of U.S. wetlands, thereby amplifying the environmental risks associated with modern-day disturbances. Two points must be considered: first, the change in derived benefits associated with reduced acreage varies with wetland type, location, history, land use, and other factors; second, incremental change can approach critical thresholds that when exceeded, lead to disproportionate and often irreversible changes in ecosystem state and the environmental services provided (Lane et al., 2022). Therefore, a given incremental decrease in wetland area may have little to no effect on the provision of an environmental service at one point in time, while the next marginal reduction may have a dramatic effect. Cumulative impacts alter ecosystems gradually through chronic multiple stressors acting at local, regional, or continental scales (Pires et al., 2023; Mushet et al., 2020; Zedler and Kercher, 2005; Suding et al., 2004). Good estimates of where such thresholds occur are a subject of developing research, and improved methods for evaluating risk in terms of both prior and projected losses are needed for effective wetland management. Therefore, this chapter begins with a brief overview of historical wetland losses as context for the updated assessment of the effects of biofuels on present-day wetland biodiversity and ecosystem health.

In response to large historical wetland losses, multiple federal initiatives since the 1970s have aimed to slow or reverse these trends. In 1977, President Jimmy Carter issued an Executive Order[1] requiring federal agencies to conserve or minimize impacts to wetlands and compensate for necessary losses through reclamation, mitigation, and restoration. Presidents Clinton, Bush, and Obama all endorsed and updated Carter's policy of "no net loss" of wetlands. Between the late 1980s and late 1990s, gross wetland losses attributable to agriculture had decreased to 26% (USDA, 2000). The five years from 1997 to 2002 was the first reporting period in which a net gain in wetlands was documented by the National Resources Inventory (NRI) (USDA, 2009). During this period, gross gains of wetlands from agricultural lands were greater than gross losses from wetlands on agricultural land (USDA, 2009). These results indicated that "no net loss" policies and related wetland conservation programs[2] were likely having positive effects on wetland recovery in the United States. However, recent data indicate a possible slowing or reversal of those trends. The literature review provided in this chapter will provide an update on NRI data evaluating trends in wetland gains and losses since 2002.

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[1] Executive Order 11990, Protection of Wetlands, 42 FR 26961, 3 CFR, 1977 Comp., p. 121.

[2] Including the 1985 "Swampbuster" provisions of the Food Security Act, which removed incentives to cultivate converted wetlands or highly erodible land for agriculture, the 1990 Wetland Reserve Program, which provided incentives to landowners to encourage the restoration of degraded or drained wetlands, and wetland programs developed by States and Tribes (https://www.epa.gov/wetlands/wetlands-programs-adopted-states-and-tribes-and-analysis-core-components). Also see the 2020 USDA Highly Erodible Land/Wetland Conservation provisions at 85 FR 53137.

14.1.2 Drivers of Change

Biofuels primarily affect wetland ecosystem health and biodiversity through increased feedstock production, as expansion of acreage in biofuel feedstock cultivation and as increased intensity of cultivation within existing agricultural acreage (U.S. EPA, 2018). With increased production, conversion of wetlands as "uncultivated land" and the filling or draining of wetlands for agriculture have resulted in large-scale loss of wetland habitats and functions in ecologically sensitive areas (Lark et al., 2020; Johnston, 2014, 2013). In addition, increased acreage and intensity of biofuel feedstock cultivation has significantly altered the hydrology of watersheds, decreased surface water storage capacity, altered natural water filtration, increased runoff and sedimentation, resulted in shifts from wetland-adapted plants to biofuel crops, increased water consumption for irrigation, added tributary ditches to stream networks, consolidated many remaining wetlands resulting in their conversion to lake-like habitats, and altered the quantity and quality of freshwater available for other uses (Baulch et al., 2021; King et al., 2021; Ameli and Creed, 2019; Mckenna et al., 2019; Evenson et al., 2018; Haque et al., 2018; McKenna et al., 2017; Thorslund et al., 2017; Anteau et al., 2016; Hayashi et al., 2016; McCauley et al., 2015; Van Meter and Basu, 2015; McLaughlin et al., 2014; Wright and Wimberly, 2013; Hoyer, 2011; Welch et al., 2010) (see also the conceptual model of altered watershed hydrology in Figure 14.2).

Wetlands typically have complex shapes, shallow water depths, and high perimeter-to-area ratios that increase the frequency of wet-dry cycling at the edges, giving them greater capacity for coupled nitrification–denitrification processes and enabling them to remove more bioavailable nitrogen that contributes to eutrophication (Marton et al., 2015). In the Prairie Pothole Region, consolidated wetlands are characterized by deeper depths, greater pond permanence, larger surface areas, simplified shapes, and smaller perimeter-to-area ratios. Alteration of the size and spatial distribution of individual wetlands in a watershed (e.g., upslope/downslope; in greater/lesser proximity to streams, lakes, reservoirs, and aquifers; singly or in clusters called wetland complexes) changes their habitat quality, hydrology, biogeochemical functioning, and biodiversity, even when total wetland area remains constant or increases. In addition, the intentional or unintentional rerouting of surface and groundwater flows caused by wetland loss and drainage can increase direct transport of sediments and chemical contaminants to streams, lakes, reservoirs, coastlines, and remaining wetlands. Small wetlands (<7.4 acres) are being drained, filled, or consolidated at a faster rate than larger wetlands (Serran and Creed, 2016; Van Meter and Basu, 2015). These small wetlands provide higher rates per unit area of biogeochemical processing and groundwater recharge than larger, more permanent, wetlands and ponds (Cheng and Basu, 2017; Cohen et al., 2016; Marton et al., 2015), predator-free habitat for amphibians and invertebrates, and conditions that favor high plant biodiversity (van der Valk, 2005a). Therefore, it is particularly important to understand the impacts of losing small wetlands to agricultural expansion and intensification for biofuel feedstock production.

Wetland habitat quality and biodiversity are greatly influenced by hydroperiod (i.e., the length of time a wetland is ponded), which often includes predictable cycles of wetland wetting (seasonal filling by precipitation or groundwater) and gradual drying (through evapotranspiration, percolation, and loss of inputs). It has been shown that wetlands on lands impacted by agriculture lose their natural hydroperiod, with greater fluctuations in water level (14.4 cm in agricultural lands vs 4.7 cm in grasslands) and corresponding shifts in the composition and biodiversity of flora and fauna (Euliss and Mushet, 1996). The water level fluctuations can increase spillage from wetlands as well. Many natural inland wetlands lack the flushing mechanisms typical of flowing waters and therefore retain and transform pollutant runoff from the surrounding landscape. When small, upslope wetlands are drained, the remaining wetlands are exposed to increased, high-volume runoff from fields and drained wetlands, causing them to fill more rapidly and overflow more frequently, thereby releasing accumulated contaminants and sediments into streams (Mckenna et al., 2019), as illustrated in Figure 14.2b. Lastly, local drivers of

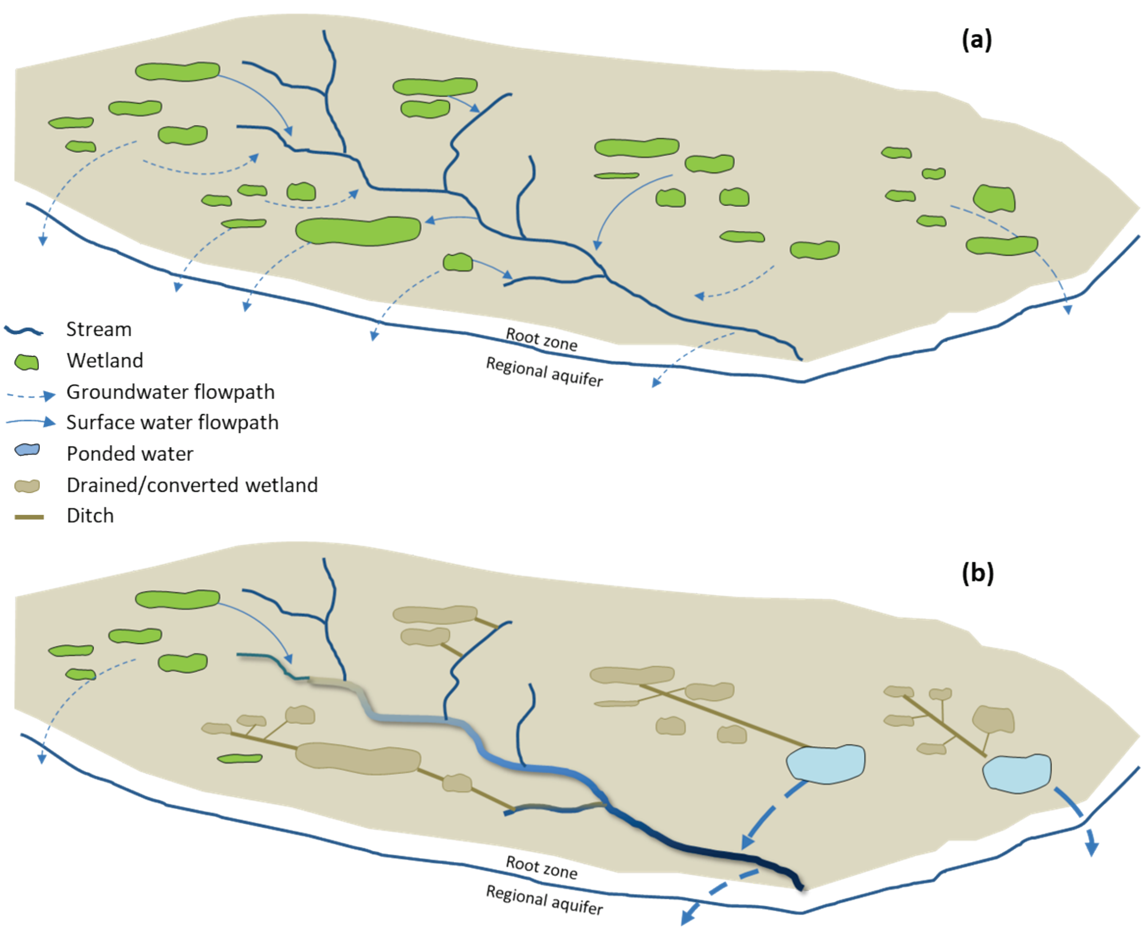

Figure 14.2. Intact wetland-stream landscape (a). Altered wetland-stream landscape for agriculture or other development, illustrating with added drainage, alteration of natural surface and groundwater flowpaths, plus loss of wetland habitat, buffers, and natural surface water storage associated with wetland loss/conversion and consolidation (b).

change that result in the homogenization of wetland types and hydrologic regimes through wetland draining and consolidation and shifts in regional climate conditions have additive or synergistic effects on wetland productivity, biodiversity, and ecosystem functioning at regional or even continental scales (Mclean et al., 2022; Elliott et al., 2020; Cohen et al., 2016; Van Meter and Basu, 2015; Blann et al., 2009; Bedford and Preston, 1988).

14.1.3 Relationships with Other Chapters

The Second Triennial Report to Congress (RtC2) reviewed environmental effects on ecosystems in a single chapter (U.S. EPA, 2018). While terrestrial, aquatic, and wetland ecosystems are inherently interrelated, the unique impacts and risks of biofuels to each merit separate treatment. Therefore, this report has reviewed the available evidence on the health and biodiversity of these three essential ecosystems in separate chapters to more fully address the complexity of potential effects of biofuels and the RFS Program on them.

Because wetlands are transitional between fully aquatic and fully terrestrial ecosystems, it is important to understand that wetland impacts can originate outside of wetland ecosystem boundaries and that wetland loss or degradation has important feedbacks to aquatic and terrestrial ecosystems. Figure 14.3 illustrates the functional relationships of this chapter to other chapters in the current report. The effects of wetland loss and consolidation for biofuel production (shown in green) interact with changes in regional water use and water balance (shown in blue), and land management practices (shown in brown) to produce an ensemble of effects on wetland ecosystem health, species, and services. Habitat loss or impairment directly affects the biodiversity of wetland communities, including threatened and endangered species. In addition, wetland loss and consolidation alter local and large-scale hydrologic and biogeochemical functions of wetland ecosystems, which prevent runoff and flooding, recharge groundwater aquifers, and filter out contaminants that might otherwise enter streams, rivers, lakes, reservoirs, groundwater, and coastal waters. Therefore, this chapter also reviews some of the mechanisms by which incremental wetland losses contribute to changes in water quality (Chapter 10) and water availability (Chapter 11).

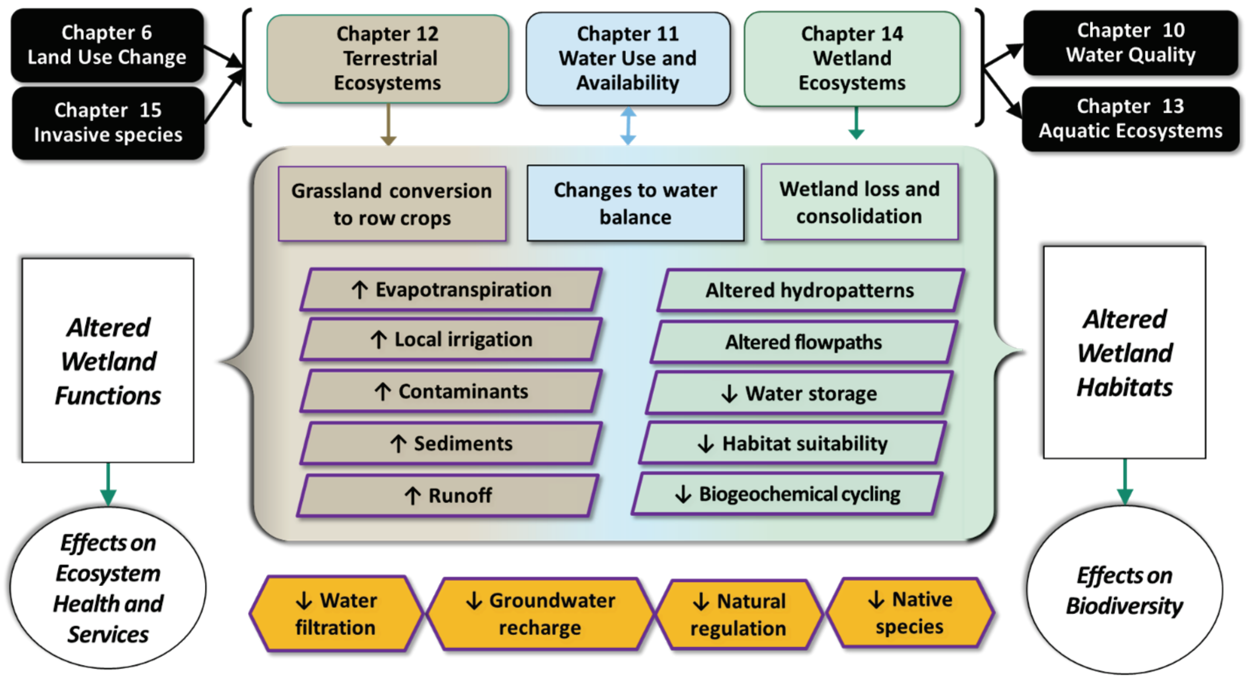

Figure 14.3. Functional relationship to other chapters in the current report.

14.1.4 Roadmap for the Chapter

Section 14.2 repeats the wetland ecosystem and biodiversity conclusions from the RtC2. Section 14.3 reviews the impacts to date for the primary biofuel feedstocks (corn and soybean), including updates of the literature on environmental effects of biofuels on wetland ecosystem health and biodiversity. Specifically, the section reviews effects of biofuels on migratory waterbirds, amphibians, threatened and endangered species, and four regional functions of wetlands: water purification; flood protection; aquifer recharge; and natural regulation of stream baseflow. Following the literature review, the attribution of wetland environmental effects to the RFS Program and opportunities for conservation practices to offset any negative effects are discussed. Section 14.4 provides a brief discussion of likely future impacts to wetlands. Section 14.5 compares effects of biofuels production to the effects of petroleum on wetlands. Section 14.6 considers other biofuel feedstocks and climate change as a horizon scanning exercise. Section 14.7 provides a summary and synthesis of this chapter, including major conclusions about wetland ecosystem health and biodiversity, comparing these with conclusions from the RtC2, an overview of remaining scientific uncertainties, and recommendations for research.

14.2 Conclusions from the 2018 Report to Congress

The overall conclusion about ecosystem health and biodiversity from the Second Triennial Report to Congress (U.S. EPA, 2018) was: “The conversion of environmentally-sensitive land to cropland consistent with increased production of current biofuel feedstocks is associated with negative impacts to ecosystem health and biodiversity."

Specific conclusions regarding effects on wetlands, in association with impacts to terrestrial and aquatic systems were:

- Loss of grasslands and wetlands is occurring in ecologically sensitive areas, including the Prairie Pothole Region.
- Loss of habitat and landscape simplification are associated with negative impacts to pollinators, birds, soil-dwelling organisms, and other ecosystem services in both terrestrial and aquatic habitats.
- Increased fertilizer applications of nitrogen and phosphorus have negative effects on aquatic biodiversity.
- Recent literature has emphasized:
    1. impacts to biodiversity and ecosystem health due to the conversion of environmentally-sensitive lands;
    2. the loss of ecosystem services, such as groundwater recharge, reduction in sedimentation, nutrient cycling, biological control of crop pests, and pollination; and
    3. the need for better environmental data collection and monitoring.

The RtC2 stated that fertilizer and pesticide usage and timing, in combination with conservation practices such as constructed wetlands and buffer strips or cover crops, could mitigate impacts to natural aquatic ecosystems, but did not discuss practices aimed at protecting wetlands themselves. This chapter updates this evidence and discusses impacts and practices of biofuels production in relation to wetland ecosystems in more detail in the sections that follow.

14.3 Impacts to Date for the Primary Biofuels

14.3.1 Literature Review

14.3.1.1 Definitions Used by Federal Agencies to Assess Change in Wetlands and Deepwater Habitats

Before presenting the wetland gains and losses reported by the NRI presented in the next section (14.3.1.2), it is necessary to look not only at land use categories (e.g., agriculture, development, forested lands) contributing to resource gains or losses, but also at NRI's subclassification of habitat types included in the resource category of "wetland and deepwater habitats." Deepwater habitats are identified separately because the term "wetland" does not include deep permanent water (Cowardin et al., 1979). The Cowardin et al. (1979) hierarchical classification system contains five classes of aquatic systems at the highest level: *Marine, Estuarine, Riverine, Lacustrine,* and *Palustrine*. Each of these categories supports different species, communities, and biodiversity; has different hydroperiods and substrates; and performs different ecosystem functions. There is overlap in the wetlands and deepwater habitat classes, with most inland wetlands classed as riverine (riparian ecosystems associated with streams and rivers) or palustrine (habitats characterized by emergent vegetation, shallow water, and periodic wet-dry cycles that enhance some types of biogeochemical processes and support high biodiversity of both water-adapted and land-adapted species). Inland wetlands include habitats commonly referred to as marshes, swamps, bogs, fens, mudflats, bottomland flats, floodplains, wet meadows, vernal pools, and potholes, among others. Marine, estuarine, and lacustrine habitats, on the other hand, include environments where surface water is permanent and often deep (>6 feet in non-tidal systems), so that water rather than air is the principal medium within which the dominant species live. As in wetlands, the dominant plants in deepwater habitats are hydrophytes.

The U.S. Geological Survey's (USGS) National Land Cover Database (NLCD) also contains information on wetlands but its use of LandSat satellite imagery results in a spatial resolution (30 m^2 pixel size) that misses many small, but potentially important, wetlands. The U.S. Fish & Wildlife Service (USFWS) Status and Trends (S&T) project (2020), which conducts standardized 5-year surveys for the National Wetlands Inventory, has the most comprehensive data on wetlands in the country, documenting wetland status and trends from the 1970s to 2009 for the 48 conterminous states, Alaska, and the Caribbean. However, the USFWS S&T data could not be used for the RtC3 because the survey results after 2009 still were not yet available. Given the commonalities of the NRI and USFWS S&T datasets, it would be beneficial to assess results from the both for overlapping years. However, because the USDA Natural Resources Conservation Service (NRCS) and the USFWS surveys have different legislative mandates, sampling methodology, data collection processes, and estimation procedures, users are discouraged from making direct comparisons of these datasets (USDA, 2009).

Both the USDA NRCS and USFWS survey programs use the Cowardin classification system (Cowardin et al., 1979) albeit in somewhat different ways. For wetland and deepwater habitats, the NRI reports annual change (gross gain, gross loss) in the *Palustrine wetlands* (inland vegetated), *Estuarine wetlands* (coastal, tidal), *Lacustrine* wetlands (lake-like, typically unvegetated) and "*Other*" (riverine, marine, and other deepwater) habitats. It also reports interannual net change (gross losses + gross gains) in wetland and deepwater habitat by class, plus net gain or loss over the survey period. The classes reported by the NRI are similar to those defined by Cowardin et al. (1979), except that some NRI results combine wetland types (e.g., palustrine with estuarine wetlands). Detailed definitions are available in the NRI.

Palustrine systems (Figure 14.4) are considered the most vulnerable to land use change, including agriculture and development, and are common in the Northern Plains, where corn and soybean cropping is expanding and intensifying. In this chapter, the review and recommendations focus on the U.S. Prairie Pothole Region (U.S. PPR) and on palustrine wetlands, which include most prairie pothole wetlands. The U.S. PPR is characterized by large numbers of depressional palustrine wetlands (prairie potholes), high spatial and temporal variability in climate and hydrology, local and regional variation in soils and substrate, and extensive landscape modification for agriculture (Van Der Valk, 2005b). Despite documented losses of 50–90% of wetlands over the last 150 years (Dahl, 2014), approximately 6 million acres of small (avg. 3.2 acres) wetlands remain in the U.S. PPR, with state and federal initiatives to restore or reclaim more of these valued natural resources. Climate is dynamic across the entire U.S. PPR, where precipitation exhibits high interannual variability within historical multi-decadal wet-dry cycles that affect the distribution and dynamics of surface waters (Winter and Rosenberry, 1998). Spatial variability in climate patterns and soils in the U.S. PPR has given rise to the wide range of functions and habitats of the region's natural wetlands. As discussed in this section, many of these functions and habitats are being impacted by the combined effects of wetland conversion and landscape simplification for agriculture, climate change, and invasive species (Mclean et al., 2022).

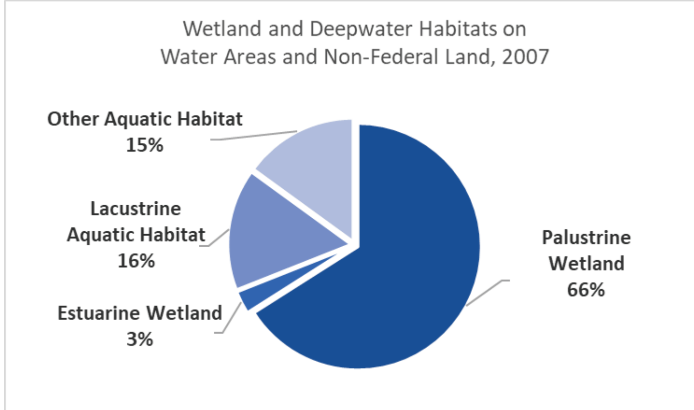

Figure 14.4. Percentage of habitat acreage for each wetland or deepwater habitat class in 2007. Source: USDA (2013).

14.3.1.2 Gains and Losses of Wetland and Deepwater Habitats since 2002 from the NRI

This section reports recent national trends in total wetland gains and losses (1997–2017) and wetland gains and losses categorized by (1) wetland classes and (2) land use class (2002–2017) in the conterminous United States (CONUS). Data on national trends in wetland change inform a broader understanding of the ecosystem health and biodiversity of wetland resources by providing a big-picture perspective for assessing the RFS Program (section 14.3.3) as one of multiple potential stressors on wetlands in the United States.

The USDA's 2017 National Resources Inventory (NRI) dataset was used to quantify national wetland losses and gains (gross and net) on non-federal lands in CONUS (USDA, 2020). The NRI was selected because it provided nationally consistent estimates of wetland and deepwater habitats for a period of 25 years (1992–2017) covering the years of the RFS Program and the years of increased biofuel production in the United States (e.g., 2002–2012). NRI estimates of net gains and losses in CONUS by wetland class and land use class were also used. Wetland class can be used as a surrogate for evaluating losses or gains of the wetland habitats and ecosystem functions uniquely associated with specific classes.

The most recent NRI reports dynamic changes in the acreage of different wetland types from 2002 to 2017. From 2002 to 2007, there was a net increase in total wetland and deepwater habitats of 521 thousand acres, with increases in all types reported except estuarine which decreased. From 2007 to 2012 this trend changed, with much smaller net increases in "wetland and deepwater habitats" that reflect conversion or transition of palustrine wetlands to lake (lacustrine) habitats, with little change in estuarine or other categories. From 2012 to 2017 these trends changed again, with losses of palustrine offsetting increases in lacustrine, and for the first time the NRI reported a net decrease in wetland area. Thus overall from 2002 to 2017, there was a net increase in wetland area, although there were large changes in the composition of those wetlands that harbor different species and perform different ecosystem functions (Figure 14.5).

Looking just at palustrine and estuarine (P&E) wetland gains/losses by land use category, which are combined in the NRI land use dataset, the period from 2002 to 2017 shows persistent net loss of P&E wetlands on cropland, pastureland, and USDA Conservation Reserve Program (CRP) land (Figure 14.6). These losses have offset gains in P&E wetland acreage from other land cover/use categories, as well as an estimated 8,700 thousand additional acres protected and restored through USDA wetland easement programs in the same time period, and resulted in national net losses of 52.8 thousand acres between 2007 and 2012 and 64.3 thousand acres between 2012 and 2017. Net loss since 2002 of P&E wetlands totals 88.6 thousand acres (Figure 14.6). This section also examines some possible causes of the observed shifts in abundance of each habitat type, and their effects on wetland function and wetland-dependent species.

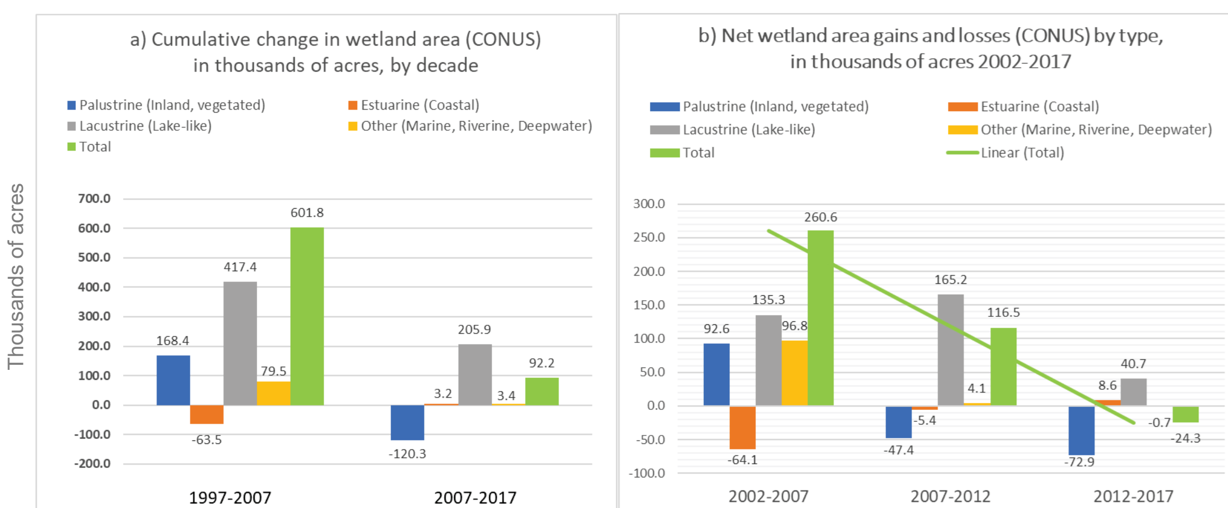

Figure 14.5. Net national change in wetland area, by wetland class (CONUS only). Comparison of net change in wetland area (CONUS) in the decades before and after 2007 (a), and Net change in wetland area (CONUS) over 5-year NRI reporting intervals from 2002 to 2017 (b). Source: 2017 NRI dataset, USDA (2020).

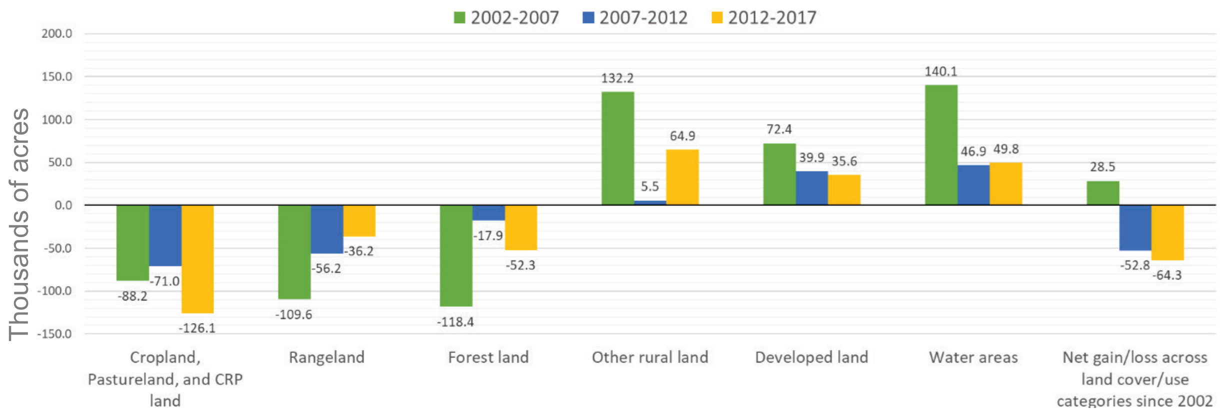

Figure 14.6. Gains/losses of palustrine and estuarine wetlands by National Resources Inventory (NRI) land cover/land use category, in thousands of acres. Source: USDA (2020). Definitions of NRI land use categories can be found online at the NRI Glossary webpage (https://www.nrcs.usda.gov/sites/default/files/2022-10/NRI_glossary.pdf).

14.3.1.3 Migratory Waterbirds

Migratory waterbirds are highly valued avian taxa for commercial and conservation purposes, and thus have been designated as federal trust species (16 U.S. Code Chapter 57B § 3772). Many migratory waterbirds are game species and nearly all are popular with bird watchers and the general public. The Mississippi flyway passes through the Northern Plains in Kansas, Nebraska, and the Dakotas, which are in the central portion of the U.S. PPR. The U.S. PPR in the north-central United States produces 50–80% of ducks that travel along all major U.S. flyways (Batt et al., 1989). In addition, more than 6 million spring-staging crane and waterfowl pass through the Platte River valley, where they now depend on residual crop grains for high-energy food (Sherfy et al., 2011). Wintering habitat is also provided in the Lower Mississippi River basin (Pearse et al., 2012).

Unlike most avian taxa in North America, waterfowl (ducks, geese, swans) as a group have not experienced declines over recent decades. The USFWS reported that relative to long-term averages, population estimates for early breeding migratory waterfowl (ducks, geese, and swans) within surveyed portions of Canada and the United States have been generally stable or increasing since 2007 (USFWS, 2019). However, USFWS estimates for target waterfowl species (spring-breeding waterfowl, primarily mallards) varies among states, with some local populations increasing and others decreasing. This may be related to the effects of interannual variation in precipitation and snowmelt on the number and size of breeding ponds in the north-central United States (USFWS, 2019).

The National Audubon Society estimated that waterbirds increased at a rate of 1.42% per year between 1966 and 2004, and for many species, the trend was stronger in regions dominated by row crops (Butcher et al., 2007). Taxonomic groups with the largest increases were dabbling waterfowl and geese (Butcher et al., 2007). Herbivorous adult waterfowl may be more likely to show positive responses to the proportion of corn or soy in the landscape (Belden et al., 2018), although this is not true for all species [e.g., black ducks (Maisonneuve et al., 2006)]. Wetlands in the Northern Prairie are part of a larger grassland-wetland habitat complex for birds, amphibians, and other species that depend on both ecosystem types to complete their lifecycles. While remaining wetlands on intensively agricultural lands may no longer provide grassland habitat for nesting waterfowl, residual soybean, corn, and grain crops provide high-energy food sources for migrating birds (Sherfy et al., 2011). Additionally, the region has been experiencing a multidecadal wet period that has likely contributed to increased waterfowl production (Mckenna et al., 2019). In the Lower Mississippi River basin, some wintering dabbling ducks preferentially use flooded agricultural fields (rice, soybean, corn, grain sorghum). Flock size is higher in wetlands interspersed with agricultural fields (Pearse et al., 2012).

With rare exceptions, population data for other waterbirds (i.e., not waterfowl) are too limited to determine national or regional trends. Their status cannot be reliably inferred from waterfowl surveys. For example, some marsh birds (rallids, bitterns, and grebes) have different or conflicting habitat requirements from waterfowl, so wetlands that are suitable to or intensively managed for waterfowl likely have limited benefit for this group.

14.3.1.3.1 Effects of Wetland Loss and Consolidation on Migratory Waterbirds

The majority of wetlands in the U.S. Midwest were lost prior to 1965 (Butcher et al., 2007; Samson et al., 2004). Waterfowl abundance is positively related to the density or proximity of ponded wetlands, which is a reliable predictor of waterfowl population density (Fleming et al., 2015; Niemuth and Solberg, 2003). For example, proximity to wetlands was associated with higher nesting densities for the two duck species, blue-winged teal and mallard (Jungers et al., 2015). Migrating mallards preferred stopovers at open-water sites, followed by corn fields adjacent to wetlands (Yetter et al., 2018).

Most waterfowl can forage on residual corn and other grains, making them generally more tolerant of grain agriculture than other waterbirds, which feed only on wetland-associated organisms, including insects, plants or seeds, and amphibians. Although some waterbirds nest on water, others (e.g., mallard, other ducks, geese, cranes) that prefer to nest in agricultural fields (hay, winter wheat, and corn) (Fox and Abraham, 2017; Anteau et al., 2011; Devries et al., 2008) can benefit from the presence of agriculture. It is an open question when conversion or consolidation of wetlands for agriculture begins to have lethal or sublethal effects [if so, this might be due to indirect effects of increased pesticide use on aquatic invertebrate prey (Foth et al., 2014)]. One study in the Canadian Maritime provinces identified thresholds for the proportion of agricultural land use beyond which waterfowl species are not found. In that area, a threshold of 49% was identified for dabbling ducks, whereas for black ducks occurrence declined in watersheds with over 60% of land managed for agriculture (Lieske et al., 2018). However, Janke et al. (2019) found that duck abundances continued to increase even in waterbodies with very high percentages of uplands cropped, possibly due to increased abundance in invertebrate prey resulting from nutrient runoff that increases primary productivity, which then translates into increased invertebrate productivity. Pesticides are generally not sprayed directly on wetlands and typically do not focus on aquatic taxa. If herbicides are applied to wetlands, the resulting dead and decaying vegetation provides additional food sources for invertebrates. This finding supported the "wetland productivity hypothesis," which states that wetlands in intensively farmed landscapes provide better feeding areas for northward-migrating ducks than those in grassland-dominated landscapes (Janke et al., 2019).

While wetlands associated with agriculture can have multiple benefits to waterbirds, wetlands suitable for nesting by breeding duck pairs have declined since 2008. In a recent study of crop expansion and land use change, Lark and others (2020) found that in the U.S. PPR, grassland and wetland habitats estimated to provide 138,000 nesting opportunities for ducks (2.8% of the regional total) were converted to crop production from 2008 to 2016. Nesting opportunities are defined as the estimated number of duck pairs within a one square mile range that have access to the suitable habitats. On average, nesting habitat losses occurred in locations determined to be accessible to an estimated 42.7 breeding pairs per square mile, which is nearly twice as high as the average for existing croplands (22.9 pairs) and 37% greater than other habitat that was not converted (31.2 pairs) (Lark et al., 2020). In addition, 29% of wetlands converted to cropland during the study period were considered "long-term habitat," which is defined as "locations that would not have been cultivated for cropland or pasture for at least a quarter century" (Lark et al., 2020). These results raise concerns about continued wetland loss due to conversion to agriculture, risks to populations of waterbirds that depend on relatively undisturbed grasslands and/or wetlands for nesting and breeding, and about the recovery time needed for long-term habitats temporarily converted to corn or soy production.

14.3.1.3.2 Effects of Sedimentation and Chemical Inputs on Migratory Waterbirds

EPA assesses exposure risks from wetland plants in endangered species biological evaluations, for risk assessments of pesticides used on genetically modified organisms (mostly herbicide-resistant crops), and for assessments of new active ingredients.[3] The Agency's ecological assessments of the five pesticides (all herbicides: glyphosate, atrazine, metolachlor, acetochlor, and 2,4-D) most intensively used (see Chapter 3) on corn and/or soybeans in the five "corn belt" states (Illinois, Indiana, Iowa, Kansas, Missouri, and Nebraska), found that all five posed potential risks to birds, and also to terrestrial-phase amphibians (https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/aquatic-life-

[3] EPA recently released new models and tools for assessments of effects from pesticides on listed species: https://www.epa.gov/endangered-species/models-and-tools-national-level-listed-species-biological-evaluations-triazine

benchmarks-and-ecological-risk). The Agency's assessments of dimethenamid and dicamba—two other herbicides that are among the most-used pesticides on corn and soybeans in the corn belt—also found potential risks from these chemicals to birds and amphibians generally, and to federally listed birds and amphibians specifically.

The potential deleterious effects of pesticides on taxa that rely on wetlands in corn- and soybean-growing regions are a concern that is also supported by findings in the literature. Battaglin (2009) measured glyphosate and other pesticides in vernal pools in three national parks and national wildlife refuges in 2005, including one in Iowa adjacent to a field planted in corn. Of 65 pesticides or pesticide degradates monitored for, residues of 28 were detected, including atrazine, glyphosate and its degradate AMPA, and atrazine and three of its degradates, as well as 2,4-D, dicamba, imazethapyr, metalaxyl, and several others. The authors noted that the results of their study demonstrate "that sensitive aquatic habitats such as vernal pools can be contaminated by the use of herbicides to control weeds in cropped areas." Another study detected neonicotinoid compounds in 30% of sampled pothole wetlands, which tended to be in areas with agricultural drainage (Williams and Sweetman, 2019). Although concentrations were nondetectable in soil samples, some water samples in wetlands draining croplands had concentrations of some compounds that exceeded EPA limits (Main et al., 2014). Pesticide concentrations in wetlands draining grasslands were lower than those draining non-corn/soybean croplands (Main et al., 2014). In summary, because there are thresholds in the benefit from agricultural upland area to foraging waterbirds, larger areas and associated pesticide residue exposure potential may reduce the potential benefits of access by waterbirds to waste grain in fields. In addition, exposure through consumption of treated corn seeds is a relatively new threat (Lopez-Antia et al., 2016).

14.3.1.4 Amphibians

When people think of wetland biota, they often think of the frogs, toads, and other amphibians critically linked to wetland habitats. However, the biphasic (i.e., having life stages with very different resource needs) life history of amphibians also directly links them to surrounding upland habitats where most species live, feed, and hibernate after completing the aquatic phase of their life histories (Mushet et al., 2012). Being dependent upon both wetland and upland habitats makes amphibians especially vulnerable to the influences of agriculture that affect both wetlands and the surrounding uplands, including land use changes associated with the production of biofuel feedstocks such as corn and soybeans.

14.3.1.4.1 Effects of Wetland Loss and Consolidation on Amphibians

The conversion of natural upland and wetland areas for crop production and development has been identified as a primary causal factor in the global decline of amphibians (Houlahan et al., 2000;

Alford and Richards, 1999; Table 14.1). Natural upland plant communities provide abundant insect food resources for amphibians (Stebbins and Cohen, 1995). Grassland cover provides the shade and moisture environment needed by amphibians to maintain hydration and regulation of body temperatures (Semlitsch, 2000). Grasslands also provide the undisturbed soils and layers of dead vegetation that facilitate successful overwintering of amphibians (Naugle et al., 2005). The conversion of grasslands to croplands degrades the amount and quality of upland habitats for amphibians (Gray et al., 2004). This has an influence on the upland areas adult amphibians need to meet foraging, thermoregulation, and overwintering requirements (Semlitsch, 1998; Madison, 1997). Areas converted to corn and soybean production generally have highly controlled and therefore depauperate insect populations; less shaded, drier, and hotter understories; and periodically disturbed (i.e., tilled) soils with little residual vegetation (Kelly et al., 2017; Johnston, 2014).

Since production of corn and soybeans can at times be more economically beneficial than the production of small grains, the transition to corn and soybeans has been associated with increased investments in tile-drain networks that increase yields. The underground, tile drainage of uplands

Table 14.1. Factors and processes contributing to the global decline of amphibians.

Factor	Process(es)
Habitat Destruction, Alteration, and Fragmentation	Habitat loss or degradation attributable to agriculture or development, and fragmentation caused by habitat loss, roads, introduced species, or other factors that separate remaining populations of amphibians from each other, are primary causes of amphibian declines.
Introduced Species	Some non-native (invasive) species prey on or compete with native amphibians.
Overexploitation	Amphibians are removed from the wild and sold as food, as pets, or for medicinal and biological supply markets.
Climate Change	Amphibians are extremely sensitive to small changes in temperature and moisture. Changes in global weather patterns (e.g., El Niño events, global warming) can alter breeding behavior, affect reproductive success, decrease immune functions, and increase amphibian sensitivity to chemical contaminants.
UV-B Radiation	Levels of UV-B radiation in the atmosphere have risen significantly over the past few decades. Researchers have found that UV-B radiation can kill amphibians directly, cause sublethal effects such as slowed growth rates and immune dysfunction, and work synergistically with contaminants, pathogens and climate change.
Chemical Contaminants	Chemical stressors (e.g., pesticides, heavy metals, acidification, nitrogen-based fertilizers) can have lethal, sublethal, and direct or indirect effects on amphibians. These effects may include death, decreased growth rates, developmental and behavioral abnormalities, decreased reproductive success, weakened immune systems, and/or hermaphroditism.
Disease	Diseases (such as chytridiomycosis) or increased susceptibility to existing diseases leads to deaths of adults and larvae. New chytrid diseases such as those caused by *Batrachochytrium salamandrivorans* seem to be particularly lethal to salamanders.
Deformities	There has been a recent and widespread increase of deformities (or malformations) in natural populations of amphibians, not attributable to known diseases; this is now perceived as a major environmental problem.
Synergisms	Multiple factors can act together to cause mortality or sublethal effects.

Modified from AmphibiaWeb. 2021. https://amphibiaweb.org.

surrounding wetlands can have multiple effects on wetlands, including altered hydroperiods and degradation of water quality in wetlands receiving tile-drainage outflow.

Wetland drainage is also associated with the conversion of grasslands to crop production (Figure 14.1). Wetland losses from drainage have been marked (Dahl 1990) and resulted in a substantial reduction on the amount of area available to amphibians for reproduction (Mushet et al., 2012). The smaller number of wetlands on the landscape also has increased the distance between these essential habitat features leading to greater distances that amphibians must travel to repopulate sites in which populations may have become locally extinct and to provide the mixing of genetic materials needed to maintain genetically diverse populations that are more resilient to local extinction events and are able to more readily adapt to changing environmental conditions (Gray et al., 2004; Houlahan et al., 2000; Lehtinen et al., 1999; Findlay and Houlahan, 1997; Dahl and Johnson, 1991; Dahl 1990; Tiner, 1984). Consolidation drainage, that is, the drainage of multiple small wetlands into a single larger wetland (Anteau, 2012), has a two-fold effect on amphibian habitats. The smaller wetlands that are lost in this process typically had the short hydroperiods and shallow water depths needed for amphibian reproduction (Wilbur, 1980; Heyer et al., 1975). Simultaneous with this loss of reproductive habitat is the lengthening of the hydroperiod of the wetlands into which the waters from the drained wetlands are routed. The lengthening of the hydroperiod in these larger, downstream wetlands makes them more likely to support fish populations, having a negative effect on the value of these areas to amphibians (Tyler et al., 1998; Kats et al., 1988; Morin, 1986; Caldwell et al., 1980).

While not all wetlands in agricultural lands have been drained, the habitat quality of those wetlands remaining on the landscape is often degraded in areas with cropped uplands. Water runoff is generally greater from croplands than from grasslands. This can lead to changes in the magnitude of water level fluctuations (Euliss and Mushet, 1996) in these wetlands and influence amphibian reproduction efforts (Petranka, 1989). Increased runoff over bare and disturbed soils leads to greater inputs of sediments into cropland wetlands (Gleason and Euliss, 1998). Similarly, agrichemicals used in the surrounding uplands often make their way to wetlands, where they can negatively influence amphibian populations.

The Conservation Reserve Program (CRP) can provide additional habitat for amphibians in mixed-use landscapes. Results from one study that quantified CRP amphibian habitat showed that from 2007 to 2012, lands in the CRP areas declined 35% across the U.S. PPR and 22% of this land lost was prime amphibian habitat (Mushet et al., 2014). Within this region, the percentage of total CRP land (as of 2012) that is important to amphibians varied between 20% for the Des Moines Lobe (north-central Iowa) to 32% for the Northern Glaciated Plains region (roughly eastern half of the Dakotas) (Mushet et al., 2014).

14.3.1.4.2 Effects of Sedimentation and Chemical Inputs on Amphibians

The growing of corn and soybean crops is generally highly dependent on the use of agricultural pesticides, many of which can have a negative effect on amphibians (e.g., Hayes, 2004). These pesticides can degrade water quality for amphibians, thereby impacting egg development and larval survival (Boyer and Grue, 1995). The synergistic interactions of predation, competition, hydroperiod, and water quality can exacerbate negative influences on amphibian population dynamics, persistence, and community structure (Semlitsch, 2000; Wellborn et al., 1996).

As noted previously, EPA's assessments of the five herbicides (glyphosate, atrazine, metolachlor, acetochlor, and 2,4-D) that are the most intensively used pesticides on corn and soybeans corn belt states, found that all five posed potential risks to terrestrial-phase amphibians (https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/aquatic-life-benchmarks-and-ecological-risk). The Agency's assessments of two other widely used corn belt pesticides—dimethenamid and dicamba—also found potential risks posed by these chemicals to amphibians generally, and to federally listed amphibians specifically in the case of dicamba.

In addition to potential risks to amphibians identified by EPA for the most heavily used corn and soy pesticides, various researchers have documented these chemicals' deleterious effects on larval development and metamorphosis in amphibians, along with neurotoxicity, organ damage, and other sublethal effects (Curi et al., 2019; Lajmanovich et al., 2015; Lenkowski et al., 2008; Cauble and Wagner, 2005). EPA's ECOTOX knowledgebase (https://cfpub.epa.gov/ecotox/) includes thousands of entries of reported effects of corn/soy pesticides on non-target taxa. For example, tabulated results for glyphosate include developmental effects at aqueous concentrations as low as 0.7 micrograms per liter (µg/L), on Criolla frog (*Leptodactylus latrans*) larvae. For paraquat, results include acute mortality to African clawed frogs (*Xenopus laevis*) at median effective aqueous concentrations (EC50) as low as 180 µg/L.

Insecticides, such as endosulfan[4] and imidacloprid, can also have lethal effects on many species of amphibians during their tadpole stage and are often detected at ecologically relevant concentrations (Sievers et al., 2018; Brunelli et al., 2009). Herbicides as well, including glyphosate, have been reported to have lethal effects on amphibians (Jones et al., 2011; Relyea, 2003). These results are supported by a large global metanalysis that found pesticides, in general, as well as fertilizers have a strong negative effect on overall survival of amphibians (Baker et al., 2013).

In addition to negative effects on survival, sublethal effects on amphibians can lead to changes in diversity, community composition, and survival of species over the long term.(Baker et al., 2013) in their

[4] Endosulfan was phased out in the United States between 2010 and 2016 and thus is no longer on the market domestically, but it was used in the country. during the historical period for this report (i.e., since 2005). https://archive.epa.gov/pesticides/reregistration/web/html/endosulfan-agreement.html.

global meta-analysis also found an overall negative effect on amphibian growth. Specifically, this may manifest as lower body mass of larvae and juveniles (Bókony et al., 2018; Egea-Serrano et al., 2012), changes in growth patterns and time to metamorphosis (Relyea, 2012; Brunelli et al., 2009), and increases in deformities or malformations (Egea-Serrano et al., 2012; Brunelli et al., 2009). A more recent sublethal focus has been on changes in behavior. For example, glyphosate has been found to impair antipredator movement behaviors such as decreased swim speed or overall decreased activity (Shuman-Goodier and Propper, 2016; Moore et al., 2015). Insecticides such as malathion, carbaryl, and endosulfan have all been found to have negative effects on movement behavior of many amphibian species. Changes in behavior were similar to responses to herbicides with overall activity and swimming distance and speed negatively impacted (Denoël et al., 2013; Relyea and Edwards, 2010; Brunelli et al., 2009; Bridges, 1997). A recent global meta-analysis found insecticides elicit strong negative responses in amphibians, including an increase in abnormal swimming patterns and reduction of antipredator escape responses (Sievers et al., 2019). These behavioral changes in the presence of pesticides, however, are often mediated by other biotic and abiotic stressors in the environment, making it more difficult to predict the direction and magnitude of effects (Mikó et al., 2017). For example, higher pH (7.5) has been found to interact with and increase the toxic effects of glyphosate (Chen et al., 2004; Edginton et al., 2004). Furthermore, when exposed to glyphosate and increasing competition stress via tadpole density, several species exhibited reduced growth and one species became more susceptible to herbicide toxicity (Jones et al., 2011). While the variations in response to pesticides are partially determined based on factors such as the pesticides in question (including synergistic effects) and the species of interest, the body of work to date demonstrates overall negative effects on amphibians. Conversion from small grain to corn and soybean production also reduces the quality of upland habitats in terms of decreased insect foods and direct exposure of adult amphibians to harmful chemicals. Most amphibians spend the majority of their lives in terrestrial habitats (Semlitsch, 2000) where they can be exposed to direct contact with agricultural pesticides when they are applied. The more frequent application of chemicals in corn and soybean production as compared to small grains increases the chances of this direct exposure occurring. Additionally, some of the pesticides used in corn and soybean production, such as atrazine (Hayes, 2004), have been shown to be especially harmful to amphibians. Use of neonicotinoid pesticides is also more common in corn and soybean as compared to small grain production. The harmful effects of neonicotinoids on amphibians are only recently being explored.

Nitrogen from nitrogen-based fertilizers associated with corn and soybean production can accumulate in wetlands (Rouse et al., 1999) where it typically occurs as nitrate. Nitrate has been identified as a widespread contaminant threat to North American amphibians (Rouse et al., 1999). Nitrate at concentrations found in many agricultural wetlands (>1 milligram per liter [mg/L]) has been shown to cause both acute and toxic effects in amphibians (Bishop et al., 1999; Baker and Waights, 1994; Baker and Waights, 1993; Berger, 1989).

A survey by Evelsizer and Skopec (2018) of drained and reference wetlands in the Des Moines Lobe, where the U.S. PPR extends into Iowa, showed that pesticides are widespread in this landscape. As described earlier, farmed wetlands often drain into other (functional) wetlands and streams. In a four-year study that included sampling for common pesticides, Evelsizer and Skopec (2018) found one or more pesticides in more than 60% of surface water samples from drained wetlands. Concentrations in drained wetlands were high relative to reference wetlands and at times exceeded aquatic life benchmarks, and the study found detectable levels of degradates of one legacy pesticide (Alachlor) for which applications had declined precipitously 20 years prior to the study.

Connectivity between habitats is particularly important for amphibians because they require different types of habitat to complete different life stages. For example, preservation of forested habitat used by adults adjacent to aquatic reproduction sites is vital for maintaining healthy populations for amphibians such as salamanders and frogs (Todd et al., 2009). This connectivity has bidirectional importance as it allows mature adults to move into aquatic habitats for breeding and egg laying but is then needed for emerging tadpoles and larval stages to migrate away from nursery habitat (Cushman, 2006). Therefore, in order to prevent further amphibian decline, not only does the type of preserved habitat matter but accessibility to often disparate (wetland vs. grassland or upland forest) nearby habitat is vital as well as the quality of those habitats.

14.3.1.5 Threatened and Endangered Species

Chapter 12 of this report provides a list of federally threatened and endangered species occurring within 12 U.S. Midwestern states accounting for 80% or more of planted corn and soybean acres (see Chapter 12, Supplemental Tables 12.2 and 12.3 and Figure 12.3). In Supplemental Table 12.2 listings with an asterisk (*) indicate animals that require both upland (e.g., grassland) and wetland habitats to complete their lifecycles, or use wetlands for foraging, refuge, migration, or alternative breeding/rearing habitat. In Supplemental Table 12.3 obligate or facultative wetland plants are identified with an asterisk (*). The list of threatened and endangered species in Chapter 12 includes some well-known wetland-obligate species, including the piping plover (*Charadrius melodus*), whooping crane (*Grus americana*), northern long-eared bat (*Myotis septentrionalis*), the little brown bat (*Myotis lucifugus*), red bats (*Lasiurus borealis*), hoary bats (*Lasiurus cinereus*), silver-haired bats (*Lasionycteris noctivagans*), Hine's emerald dragonfly (*Somatochlora hineana*), and the northern population of the copperbelly water snake (*Nerodia erythrogaster neglecta*), of which only a few hundred individuals remain.

14.3.1.6 Effects on Hydrologic and Biogeochemical Functions of Wetlands

As noted in the introduction to this chapter, the ecosystem functions of wetlands extend far beyond provisioning of food and habitat for wetland-dependent species. One regional function provided by wetlands is water quality purification (Kazmierczak, 2001), including the retention, removal, and transformation of nitrogen and phosphorus (Verhoeven et al., 2006), carbon (Kayranli et al., 2010), and metals (Gambrell, 1994). A second regional function of wetlands is the interception and storage of stormwater and runoff, with gradual release of filtered water into shallow or deep groundwater systems (aquifer recharge) (van der Kamp and Hayashi, 1998; Carter, 1986). The third and final regional function considered here is the capacity for surface water storage in small, distributed wetland complexes to influence subsurface flowpaths that maintain baseflow in stream systems. This section considers the evidence of impacts from corn and soybean production on these wetland ecosystem services.

14.3.1.6.1 Water Purification by Wetlands

A review of 12 studies published between 1981 to 2001 found that valuation of coastal wetlands water quality services varied widely, ranging from $2.85 per acre per year to $5,673.80 per acre per year with a median of $210.93 per acre per year (Kazmierczak, 2001). The magnitude of variation among these estimates is highly dependent on the specific location, the type of water quality service considered, the methods used to estimate value, and whether or not local benefits at the study site were used to estimate water quality services across all existing wetlands. On the other hand, estimates of willingness-to-pay (WTP) for wetland water quality services in these studies were lower and much narrower in range, from $41.71 per acre per year to $101.81 per acre per year (Kazmierczak, 2001).

Nitrogen and Phosphorus

The ability of wetlands to remove nitrogen and phosphorus from through flow has been heavily investigated at both the site- and catchment-level scales (Verhoeven et al., 2006). Wetlands have been engineered and managed to provide tertiary wastewater treatment services (Kadlec and Knight, 1996; Reed, 1991; Conner et al., 1989; Richardson and Davis, 1987) and to remove nutrient fertilizers from agricultural runoff (Doering et al., 1999). Wetlands may reduce nutrient loads through a variety of mechanisms, including storage of nutrient-rich sediments (Johnston et al., 1984; Karr and Schlosser, 1978), nutrient sorption to sediment particles (Khalid et al., 1977), plant uptake of nutrients (Lee et al., 1975), and promotion of denitrification (Lowrance et al., 1984). Wetlands have been shown to effectively remove nitrate from through flow from a variety of land uses and inputs (Hunt et al., 2004; Groffman and Crawford, 2003; Matheson et al., 2003; Clément et al., 2002; Weller et al., 1994), primarily via denitrification, which is the process of nitrate being converted to nitrous oxide that is then converted to atmospheric nitrogen. Wetland plants can take up nitrogen (N) and phosphorus (P) in different forms, creating a short-term pool of stored nutrients (Hefting et al., 2005; Havens et al., 2004; Silvan et al., 2004; Uusi-Kämppä et al., 2000). However, such stored nutrients are only permanently removed from the system if the vegetation is harvested (Addy et al., 1999).

Wetlands have been the focus of large-scale nutrient reduction efforts in the United States. According to Mitsch et al. (2001), 20–50% of the total N load that reaches the Gulf of Mexico from the Mississippi basin could be removed by restoring wetlands covering just 1–2% of the basin's catchment area (11.5 to 23.0 million square miles) primarily located in small (headwater) streams. Strategically restoring wetlands in the headwaters, which have the greatest impact of denitrification, would have greater effect on reducing nutrient export relative to restorations in other parts of the river network. A review of several global case studies concluded that wetlands may significantly contribute to nutrient reductions at the watershed scale if they cover at least 2–7% of the watershed area (Verhoeven et al., 2006).

In a recent synthesis of empirical studies of nutrient retention by prairie pothole wetlands that intercept agricultural runoff in Iowa and Minnesota, Ross and McKenna (2023) reported a strong, near-linear trend of increased wetland removal with increased nitrate inputs. They also determined that wetlands in the studies removed an average of 52% N and 67% P from agricultural runoff water before it was transported downstream (Ross and Mckenna, 2023). However, remaining wetlands and associated headwater streams in some urban and agricultural watersheds are reaching maximum nutrient storage capacity and therefore are passing excess nutrients through to downstream waters. Studies have estimated the "critical load" of nutrients that will saturate a typical wetland's nutrient retention capacity and allow nutrients to pass via through flow. For P, 10 kilograms of P per hectare per year (kg P/ha/yr) has been proposed as a critical loading rate (Richardson and Qian, 1999; Richardson et al., 1996), and 25 kilograms of N per hectare per year has been proposed as a critical loading rate for N (Bobbink and Lamers, 2002; Bobbink et al., 1998; Bobbink and Roelofs, 1995). However, wetlands may be highly heterogeneous with respect to critical loading rates (Verhoeven et al., 2006). Studies in temperate systems have shown that the maximum potential rate of N removal generally ranges from 1,000 to 3,000 kilograms of N per hectare per year, and the maximum potential rate of P removal generally ranges from 60 to 100 kilograms of P per hectare per year (Verhoeven et al., 2006).

After reviewing data from 57 wetlands across the globe, Fisher and Acreman (2004) found that the majority of wetlands reduce nutrient loads, with 80% of wetlands reducing N loads and 84% of wetlands reducing P loads. However, some wetlands may serve as sources of nutrients to adjacent waters particularly under high flow events or over long periods of time (Fisher and Acreman, 2004). Wetlands are more effective at reducing nutrient loads than terrestrial portions of riparian zones because of their higher organic matter content (Cooper, 1990), higher residence (water storage) times (Dettmann, 2001), and because their morphology allows them to easily trap and retain nutrients (Fisher and Acreman, 2004). Studies cite a number of factors affecting the capacity of wetlands to reduce nutrient loads, including oxygen levels, water retention time and volume, and vegetation processes (Fisher and Acreman, 2004).

Several key factors may decrease the potential value of wetland nutrient removal services. First, natural and constructed wetlands can be a source of nitrous oxide (a powerful greenhouse gas that can also destroy stratospheric ozone) to the atmosphere if the reduction of nitrate to atmospheric N is incomplete (Machefert et al., 2002). High nitrate levels may increase the level of nitrous oxide production in wetlands, and studies have shown increased nitrous oxide after N fertilization on agricultural lands (Bouwman et al., 2002; Machefert et al., 2002). Second, some studies have shown that the water-purification functions of wetlands may become degraded over time (Chagué-Goff et al., 1999; Osborne and Totome, 1994). While the N removal potential of wetlands tends to be constant over time, the P removal potential of wetlands tends to decrease over time (Fisher and Acreman, 2004).

Trace and Toxic Metals

A review study found that wetland systems tend to have higher uptake rates, lower leaching losses, and lower surface runoff losses of trace and toxic metals such as lead, cadmium, and zinc, as compared to upland systems (Gambrell, 1994). Many studies have demonstrated that wetland soils can more effectively immobilize trace and toxic metals than can upland soils (Gambrell, 1994). This is largely because wetlands have lower oxygen levels and near-neutral pH levels that create favorable conditions for metal immobilization. Furthermore, flooded soils and sediments tend to have higher organic matter content, including insoluble humic materials that are strongly associated with metals (Gambrell, 1994). Clays and humic materials may adsorb trace and toxic metals, and any sedimentation in wetlands would bury the metals, leading to more stable immobilization (Gambrell, 1994). Leaching rates are low in wetland systems because of the slow water permeability in waterlogged soils (Gambrell, 1994). Ultimately, wetlands provide important water purification services by storing metals that would otherwise reach groundwater, lakes, streams, or rivers.

14.3.1.6.2 Effects on Surface Water Storage on Flood Protection, Groundwater Recharge, and Stream Baseflow

Surface ditching and subsurface (e.g., tile) drainage of wetlands for agriculture have significantly reduced wetland habitats and functions across the United States. In Iowa, the state Department of Natural Resources estimates that 95% (between 3.8 and 5.7 million acres) of historical wetlands have been drained for agriculture (IDNR, 2022). As of 1990, approximately 86% of wetlands in Indiana had been drained or filled for agriculture (Miller, 1990).

The purpose of ditching or tiling is to enhance runoff from fields and "replumb" wetlands to prevent water from accumulating on grasslands and wetlands that have been converted to croplands. Obvious unintended consequences of draining wetlands include increased erosion and transport of fertilizers and pesticides (insecticides, herbicides, fungicides) into rivers, lakes, reservoirs, and coastlines (U.S. EPA, 2015; van der Kamp and Hayashi, 2009). Some less obvious consequences include more frequent and/or damaging floods, reduced groundwater recharge, and changes in the timing, duration, magnitude, and stability of streamflow. How wetland drainage impacts watershed hydrology and associated ecosystem functions depends on characteristics of the wetland or wetland complex, including the wetland types, soils, locations, hydroperiod, and vegetation (Evans et al., 1996), and on regional differences in topographic and geologic controls over vertical and lateral flowpaths that connect wetlands to streams and shallow or deep groundwater systems (van der Kamp and Hayashi, 2009). Draining a few small wetlands may not seem likely to have large impacts on hydrology, but the effects on surface water storage can be large: one acre of natural wetland can hold as much as 1–1.5 million gallons of floodwater (U.S. EPA, 2001) and historically, the cumulative storage capacities of drained wetlands in states like Iowa and Indiana reached into the trillions of gallons. Extensive ditching is not unique to the Midwest. Jones et al. (2018) estimated that plugging the small ditches that drain wetlands on agricultural land on the Delmarva Peninsula (Chesapeake Bay Region) would increase surface water storage capacity across the peninsula by 80%, thereby preventing rapid runoff from directly entering stream networks that drain into the Chesapeake Bay and Atlantic Ocean. Focusing on one watershed, they found that 59% of restorable wetland water storage capacity occurred within 20 meters of the stream/ditch network (Jones et al., 2018). National or state-wide estimates of surface water storage loss to agriculture since 2008 are not available, but wetland draining (i.e., through ditches or tile drains), fills, and consolidation of small wetlands with corresponding increases in larger, more permanent ponds and open waters are highly correlated with agricultural transitions from the growing of small grains to corn and soybean production (Krapu et al., 2018). Recent evidence from the U.S. PPR indicates that the prolonged flooding from increased precipitation and changes in snowmelt that has been occurring since the early 1990s is being exacerbated by high rates of wetland ditching and consolidation for agriculture (Anteau, 2012). In contrast with past patterns of decadal climate shifts between drought and deluge conditions that have occurred for centuries, the current wet phase in portions of the U.S. PPR has been stable and appears likely to continue (McKenna et al., 2017; Johnson et al., 2005).

The cumulative storage capacity of small wetlands in a watershed can be very large (Jones et al., 2018) and spatially distributed. Historically, small, seasonal wetlands are preferentially drained or ditched for development and agriculture (Serran and Creed, 2016). Using a hydrologic model to assess the effects of removing or draining wetlands of different sizes and at different locations relative to streams, Evenson et al. (2018) found that the loss of smaller depressional wetlands (<3.0 hectare) substantially decreased total inundated area and surface water residence times. A wetland management scenario based on protecting wetlands 30 meters and approximately 450 meters from the stream resulted in decreased inundated area and residence times, indicating that wetlands at greater distances from streams enhance these important watershed functions. They also found that the probability of increased downstream flooding from wetland loss was also consistent across all loss scenarios (large vs. small wetlands drained, near vs. far from streams). The authors' results indicate that wetland management plans that weight a single goal (e.g., large wetland protection for flood storage) should balance the effects of benefits of achieving that objective against the cost for other functions that may be lost in the process (e.g., biodiversity, nutrient processing in small wetlands) (Evenson et al., 2018).

Increased residence time allows water purification processes described above to function. In fact, smaller wetlands, which typically have shallow depths and seasonal drying, tend to have higher nutrient removal rates than larger ones; so for the same reduction in wetland area, the loss of small wetlands equates to a greater loss in nutrient removal potential (Cheng and Basu, 2017).

Depressional wetlands (potholes) are focal points for groundwater recharge in the U.S. PPR (LaBaugh et al., 1998). As with nutrient cycling, the seasonal drying of temporary wetlands can enhance groundwater replenishment. A recent study by Bam et al. (2020) in St. Denis, Saskatchewan in the Canadian PPR compared isotope signatures of permanent ponds and temporary wetlands with those of confined aquifers, located in deep glacial till, which supply freshwater to communities and agriculture. They found that permanent ponds had a distinct signature while signatures of temporary wetlands and groundwater aquifers were similar. Their findings indicate that temporary wetlands are the dominant source of groundwater recharge at this location. For this reason, conservation of small, seasonal wetlands is important for groundwater replenishment and supply in some areas.

To see how wetland consolidation might affect streamflow, McLaughlin et al. (2014) modeled water table and streamflow dynamics under scenarios in which wetland area was (1) distributed across a large number of small wetlands, or (2) consolidated into a single, large wetland. They found that increasing total wetland area while decreasing individual wetland size reduced water table and stream baseflow variability by as much as 50%. By intercepting and storing surface water, small, spatially distributed wetlands stabilize water table levels and, therefore, baseflow through a phenomenon the authors call landscape capacitance (McLaughlin et al., 2014). Preserving natural levels of surface water storage in wetlands can mitigate effects of climate change as well. In a model of climate and land use change in the Canadian PPR, Dumanski et al. (2015) estimated that the interaction of wetland drainage, more extreme precipitation events, and altered snowmelt has increased runoff ratios and streamflow volume by an order of magnitude.

14.3.2 New Analyses

There were no additional analyses performed for this chapter supplemental to the habitat conversion analysis already discussed in Chapter 12 (section 12.3.2). That analysis overlayed critical habitat for T&E species in the Midwest with lands that were estimated converted from seminatural and natural cover from 2008 to 2016 (Figure 14.7). As shown in Table 12.1, there were six wetland species with 10 acres or more of corn and soybean planted within a 1-mile buffer of their critical habitat and four with 10 acres or more that were estimated to be directly in the critical habitat. A full list of T&E species, including wetlands species, occurring in the northern great lakes, central plains, and prairie ecoregions is provided in Chapter 12 [Supplemental Tables 12.2 and 12.3, and Lark (2023)].

These areas of wetland conversion in North Dakota, South Dakota, and Minnesota, and areas of grassland conversion additionally in southern Iowa and Northern Missouri, correspond with areas of increased corn and soybean production from 2008 to 2016 (Chapter 5, Figure 5.10). However, due to the lack of national or regional datasets that track changes in acreage of converted wetlands for agricultural production of biofuel feedstocks and more specifically, for the RFS Program, it is not possible to say how many acres of wetlands lost to corn and soy production during this period are directly and solely attributable to the RFS Program.

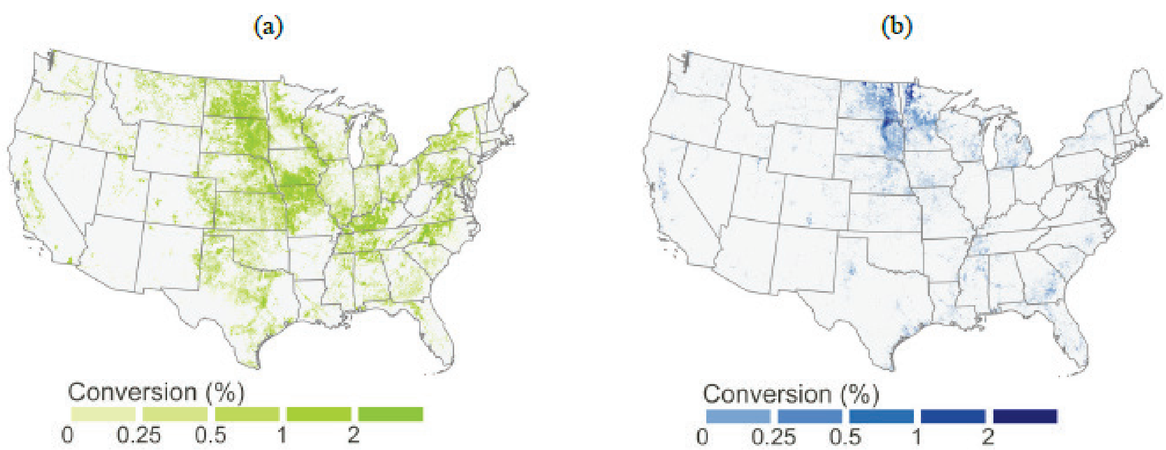

Figure 14.7. Location of gross conversion of grasslands (a) and wetlands (b) to cropland between 2008 and 2016. Source: Lark et al. (2020). (Creative commons license https://creativecommons.org/licenses/by/4.0/).

14.3.3 Attributions to the RFS Program

This chapter reviewed well-documented effects of corn and soy agriculture on wetland biodiversity and ecosystem function. The review focused on the habitats and ecosystem services of wetlands that are ecologically integrated with grasslands in the north-central portion of the United States, where both ecosystems have experienced conversion to cropland between 2008–2016 (Lark et al., 2020; Figure 14.7). This section reports estimates of the percentage of wetland conversions to cropland that can be attributed to the RFS Program as the primary driver of change.

There are two major mechanisms by which the production of corn and soybeans can negatively impact wetlands: (1) conversion of wetlands to croplands; and (2) production practices that increase application and runoff of chemicals, including pesticides and fertilizers, into wetlands. Regarding wetland conversion: between 2008 and 2016 cropland in the conterminous United States expanded by an estimated 10 million acres, of which 275,000 acres were the result of wetland conversion (Lark et al., 2020; Figure 14.7b). In Chapter 6 of this report, an estimated 0 to 1.9 million acres of the total cropland expansion is attributable to corn-ethanol production associated with the RFS Program (approximately 0 to 20% of the observed net increase in U.S. cropland over this period; see Chapter 6 for more information). For the RtC3, the exact locations of where this 0–20% is concentrated are not estimated;[5] thus, the direct impacts of the RFS Program on the area of wetlands lost to cropland between 2008 and 2016 could be as little as 0, as large as 275,000 acres, or more likely, some intermediate amount during this historical period (e.g., if assumed to be 20%, an estimated 55,000 acres).

As of 2009, wetlands covered 5.5% of land area in the conterminous United States (Dahl, 2011). Freshwater wetlands comprise 95% of U.S. wetlands; the rest are marine or estuarine (Dahl, 2011). Wetland densities in the Dakota Prairie Pothole Region (DPPR), exceed the national average by 3% (8.5% of DPPR land area; Johnston, 2013), suggesting that estimated losses may be more important regionally or locally, especially in areas where wetlands are embedded with forests or grasslands (and therefore may be more difficult to classify from remotely sensed data), areas with a higher concentration of converted acres, areas with many wetland-dependent species or wetland-dependent water supplies, and/or where wetlands have historically experienced large losses to agriculture (e.g., Iowa, North and South Dakota).

The data needed to quantify the exact number, area, types, or locations of wetlands drained for biofuel production attributable to the RFS Program on wetlands are not available. However, given what is known about historical and recent rates of loss of palustrine wetlands, concentration of wetland losses in areas with high densities of waterbird breeding habitats (Lark et al., 2020; Figure 14.7b), and colocation of biorefineries with observed areas of wetland conversion (Figure 1 in Wright et al., 2017), it can be inferred that wetland biodiversity and ecosystem health have likely been adversely impacted. Because smaller wetlands are converted to agriculture at higher rates than larger wetlands (Van Meter and Basu, 2015), the habitats and functions of small wetlands—which include higher rates of groundwater recharge, denitrification, carbon storage (not reviewed in this chapter), and surface water storage to mitigate flood peaks and maintain river baseflow—will likely have experienced the greatest impacts. Further, the

[5] More analytical research needs to be conducted before these 0–1.9 million acres can be confidently assigned to geographic locations across the United States.

attributional estimates from Chapter 6 represent the effect of the RFS Program only on corn ethanol and corn, and would likely be larger if the effect on historical soybean acreage were quantified (see Chapter 7).

In addition to wetland conversion, chemical application and runoff (e.g., fertilizers, pesticides) is a second mechanism by which corn and soybean feedstock production can impact wetlands and potentially T&E species. The production of corn on converted wetlands inherently causes an increase in pesticide and fertilizer usage. In addition, much of the conversion from wetland to corn occurred in the northern Midwest, likely at the expense of wheat (see Chapter 5). Producers used almost 1.6 and 6.5 times more pesticides by mass per acre on corn than soybeans and wheat, respectively, in 2008 (Fernandez-Cornejo et al., 2014). This suggests that corn ethanol attributable to the RFS Program likely negatively impacted wetlands through corn production practices in addition to acreage effects, yet again the magnitude of such a potential effect requires further study.

14.3.4 Conservation Practices

Restoring wetlands, wetland complexes, and surrounding grasslands wherever possible, are the primary conservation practices that would lead to improved sustainability of wetland biodiversity and ecosystem function in areas of biofuel feedstock production. The needs of wetland-adapted species throughout their entire life history, not just focusing on the reproductive period, aquatic phases, or migration will lead to increased restoration success rates. Establishing and maintaining grassland buffers around wetlands to provide terrestrial habitat for birds and juvenile and adult amphibians; reducing runoff inputs of sediments, nutrients, and pesticides to wetlands; and limiting the disturbance of soils surrounding wetland habitats will also have beneficial effects. While pesticides are typically an important component of corn and soybean production, limiting their applications in uplands surrounding wetlands to the fullest extent possible will reduce wetland impacts. Likewise, limiting programs that incentivize the drainage of wetlands and upland soils surrounding wetlands, and increasing funding and decision support for programs that incentivize landowners to intersperse functional wetlands within agricultural systems, and to restore and conserve wetlands on low-yield agricultural land will produce positive wetland-related benefits (Box 14.1). There are other ways that land managers can reduce the negative impacts of biofuel feedstock production on wetlands. If not already using integrated pest management (IPM) strategies in pesticide and herbicide applications, they can contact local NRCS offices or extension agents (usually through state agencies and universities) for advice on how to do so. If they are already using IPM methods, they can still check with local support staff to see what else might be done to avoid or limit harm to wetlands.

Box 14.1. Federal Wetland Protection and Restoration Programs

For more than 30 years, the U.S. Department of Agriculture's (USDA) Natural Resources Conservation Service (NRCS) and the Department of the Interior's (DOI) U.S. Fish and Wildlife Service (USFWS) have managed successful programs to conserve important wetland habitat for the benefit of wildlife and people. NRCS' Agricultural Conservation Easement Program – Wetland Reserve Easement (ACEP-WRE), which also includes the former Wetland Reserve Program, protects wetlands on agricultural lands. Through ACEP-WRE, NRCS purchases easements and restores wetland functions and values while providing development protection for the life of the easement. Since its inception in 1991, almost 17,000 applications have been enrolled in all 50 States and Puerto Rico, protecting over 3,000,000 acres. The North American Wetlands Conservation Act (NAWCA), administered by the USFWS, provides funding to conserve wetland habitat. NAWCA provides matching grants to partners to carry out wetland conservation projects in the United States, Canada, and Mexico. Since 1989, it has contributed to the protection, restoration, and enhancement of more than 30.6 million acres of wetlands and associated upland habitats in all 50 U.S. states, 31 Mexican states, 10 Canadian provinces, and multiple territories.

Agricultural Conservation Easement Program – Wetland Reserve Easement

The Wetland Reserve Easement component of the Agricultural Conservation Easement Program (ACEP) is authorized by subtitle H of title XII of the Food Security Act of 1985, as amended by Section 2301 of the 2014 Farm Bill (P.L. 113-79). ACEP-WRE is a voluntary program through which NRCS provides technical and financial assistance directly to private landowners and Indian Tribes who agree to restore, protect, and enhance wetlands through the sale of a permanent or 30-year wetland reserve easement or through a 30-year contract (Tribes only). The goal of ACEP-WRE is to restore wetland functions and values, and wildlife habitat, to the greatest extent possible, on every enrolled acre. Lands primarily used to produce food or fiber including farmed, converted, former or degraded wetlands, along with several other land categories, are eligible for participation. Land eligibility for ACEP-WRE enrollment is determined by NRCS through an onsite evaluation. ACEP is available in all 50 States and U.S. territories.

North American Wetlands Conservation Act (NAWCA)

NAWCA is authorized in 16 USC 4401 et seq., and amendments. It provides matching grants to partners to carry out wetland conservation projects in the United States, Canada, and Mexico. Projects utilize both grant and 1:1 non-federal partner match to conserve wetlands and wetland-associated upland habitat through acquisition (including easements and land title-donation), restoration, enhancement, and/or wetland establishment activities. NAWCA was originally passed to support priorities identified in the North American Waterfowl Management Plan (NAWMP) but has since expanded to support wetland-associated migratory bird populations covered by multiple conservation plans and international treaties. Eligible applicants including federal, state, or local governments, non-profit organizations, private corporations, tribes, and private individuals. The North American Wetlands Conservation Council ranks proposals and the Migratory Bird Conservation Commission makes final funding decisions.

Websites of Interest

- USFWS NAWCA Website: https://www.fws.gov/program/north-american-wetlands-conservation
- ACEP Website: https://www.nrcs.usda.gov/wps/portal/nrcs/main/national/programs/easements/acep/
- Title XII of the Food Security Act of 1985: https://www.agriculture.senate.gov/download/compilation/food-security-act-of-1985
- 2014 Farm Bill: https://www.govinfo.gov/link/plaw/113/public/79

Benefits to waterfowl of riparian buffers planted in native grasses or woody vegetation depend on the relative risks associated with predation compared with crop pesticide effects. On the one hand, some waterfowl prefer to nest in open habitat (e.g., fields) away from high, vegetated riparian buffers where predators can hide (Crimmins et al., 2016). Nest predation rates near shelterbelts (i.e., a line of trees to protect crops and soils from strong winds) are higher than those in open fields (Borgo and Conover, 2016). Shelterbelts in the U.S. PPR also provided corridors for meso (i.e., middle-trophic level) predator movement. On the other hand, buffers can improve wetland habitat by filtering water from agricultural drainage systems before it reaches the pond or wetland, decreasing sedimentation and reducing waterbird exposure to farm chemicals (Williams and Sweetman, 2019). Thus, both effects on filtering water and predation impacts should be considered when making decisions related to the installation of riparian buffers.

Harvest of grasslands or woody crops adjacent to wetlands is of interest as an alternative source of biomass feedstock. The effects of grass harvest on nesting success for ducks (blue-winged teal and mallard) was quantified in Minnesota conservation grasslands (Jungers et al., 2015). These grasses were harvested in late fall and different levels of biomass removal were compared. Birds avoided nesting in recently mowed areas, but the proportion of grass harvested did not affect nesting success when performed late in the season (Jungers et al., 2015). Additionally, many waterbirds have come to depend on access to waste grain during migration. Removal of corn stover has been recommended as a way to make waste corn more accessible (Anteau et al., 2011). Corn stover is also a potential cellulosic feedstock for biofuels (Brandt et al., 2017).

The use of existing habitat quality models to locate and preserve high quality grassland, wetland, and pond habitat needed to complete valued species lifecycle phases (e.g., prebreeding, nesting, brood rearing, foraging, molting, migrating) can aid in the placement of habitat beneficial to these species. Additionally, flooding fields during fallow periods can be used as a way to provide stopover habitat for migrating waterfowl (Heitmeyer, 2006). Leaving shallow wetlands and wetland complexes within agricultural matrices of lands is also beneficial for waterbirds (Berger et al., 2003). To help plan cost-effective interspersion of protected wetlands within agricultural areas growing corn, decision tools exist for prioritizing the protection of wetland habitat across the Dakotas (Hansen and Loesch, 2017). Finally, recommendations for harvest practices to avoid nesting waterfowl as described in the Chapter 12, Terrestrial Ecosystem Health and Biodiversity, can increase waterfowl production.

The cumulative effects of local wetland conversions have regional impacts on biodiversity and ecosystem function, including loss of native species, loss of genetic diversity, decreased nutrient processing, and altered watershed hydrology (e.g., higher volume runoff and lower groundwater recharge) (Hambäck et al., 2023; Mitchell et al., 2022; Preau et al., 2022). Farm-scale management decisions to drain or conserve wetlands are based on local conditions and eligibility for conservation incentives, both of which vary widely across the U.S. PPR. Increased participation in conservation compliance for farm program benefits (Claassen et al., 2017) has potential to mitigate some regional effects of wetland loss, including habitat fragmentation, increased flood risk, and loss of ecosystem resilience in this agriculture- and wetland-dominated landscape.

14.4 Likely Future Impacts

As noted in Chapters 2 and 6, corn ethanol and soy biodiesel will likely remain the dominant biofuels in the near future (out to 2025). Whether grasslands more than wetlands continue to be converted to corn or soybeans will in large part determine the magnitude of direct future effects to wetland ecosystems, since the largest impacts to wetland ecosystems generally occur from drainage and conversion to agriculture. Nonetheless, LCLM of grasslands can have important effects on nearby wetlands, and influence connectivity among wetlands and affect changes in flow, erosion, and runoff of sediments, nutrients, and pesticides to these ecosystems. Lark et al. (2020) reported a slowdown nationally in cropland expansion since 2011, especially since 2015 [figure 2 in Lark et al. (2020)], in agreement with other sources from Chapter 5, and in connection with reaching the E10 blend wall in 2013.

When the Set Rule was finalized on June 21, 2023, EPA published biofuel volumes along with estimates of cropland expansion for 2023–2025 (see Chapter 6, section 6.5).[6] EPA estimated the biofuel volumes in the Final Set Rule could potentially lead to an increase of as much as 2.65 million acres of cropland by 2025, mostly from soybean expansion in the Midwest, but to a lesser extent from corn expansion also in the Midwest and canola expansion mostly in North Dakota. EPA was not generally able to identify specific areas for these estimated conversions, nor the source of lands used in this conversion. The likely future effects of the RFS Program are highly uncertain. While the projected cropland expansion for 2023–2025 is slightly larger than the estimated historical cropland expansion from the RFS Program, it cannot be said with reasonable certainty that any wetland will be impacted, due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes. More information on where these changes are anticipated, and what agricultural practices are employed, are needed to better constrain estimates of the likely future effects to wetland ecosystems.

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[6] On July 26, 2022, the United States District Court for the District of Columbia filed a consent decree, which after stipulations from the parties, required EPA to sign a notice of proposed rulemaking (NPRM) to establish 2023 volumes for the RFS Program on or before November 30, 2022, and to sign a notice of final rulemaking finalizing 2023 volumes by June 21, 2023. *Growth Energy v. Regan et al.*, No. 1:22-cv-01191, Document Nos. 12, 14, 15. EPA's proposed RFS volumes for 2023-2025, 87 FR 80582 (Dec. 30, 2022) available in Docket No. EPA-HQ-OAR-2021-0427 at https://www.regulations.gov, were finalized on June 21, 2023 and published in the Federal Register on July 12, 2023 (88 FR 44468). The estimates of cropland expansion from the RFS Program took considerable time and effort to develop; thus, they are based on the proposed volumes from November 30, 2022, rather than the final volumes from June 21, 2023. Because of that, the acreages discussed here are merely illustrative. However, since the total volumes from crop-based feedstocks changed fairly little between the draft and final rules, they are still relevant here (see Chapter 6, section 6.5).

14.5 Comparison with Petroleum

Biofuels predominantly affect wetland biodiversity, ecosystem health, and ecosystem services through increasing demands on two essential natural resources—fresh water and arable land. Recent demand for row crops like corn and soy, and competition for lands not already in production for food and forest products, has led to high rates of conversion in remaining wetlands, grasslands, and forests. Drained wetlands do not provide the same habitats needed to support desired levels of biodiversity as their natural counterparts, or perform the ecosystem services (denitrification, sediment and contaminant trapping/transformation) needed to maintain and improve water quality in streams, rivers, reservoirs. The impacts of converted wetlands on water quality and quantity can be widespread and long-lived. As discussed in previous sections, small wetlands are dominant sources of groundwater recharge for local and regional aquifers, have high capacity for carbon sequestration (Van Meter and Basu, 2015), and serve as storage "capacitors" (McLaughlin et al., 2014) for shallow subsurface flows that maintain baseflow in rivers and streams.

Studies that have compared the area required by the two industries (e.g., Dale et al., 2014; Parish et al., 2013) predict that the petroleum industry out to 2030 will require more than double the area of biofuels globally, including areas of the ocean and remote locations in the Arctic. In total, these areas overlap with a higher number of threatened species than that of projected biofuel production over the same time period (Dale et al., 2014). Conversely, Elshout et al. (2019) concluded the production of biofuels negatively affected biodiversity more than gasoline and diesel fuel production in most locations considered in a global analysis. The latter study assumes all new biofuel feedstock production leads to habitat loss. In the United States, wetlands have experienced net losses nationally for the first time since 1997 (USDA, 2020) and expansion of cropland for feedstocks has led to high rates of wetland conversions concentrated in environmentally sensitive regions (Lark et al., 2020; Figure 14.7) (Figure 14.7).

In addition to land required, the time or effort to recover from any adverse impacts should be included when comparing the two industries. Among other waste, oil production generates produced water, deep water flowing up through the production well. This water varies widely in chemical composition, and can contain salts, metals, and radioactive materials (U.S. EPA, 2016). Spills of produced water if not contained on the well-pad can cause long-term impacts to the surface environment and to groundwater (U.S. EPA, 2016). In a qualitative weighing of the effects along the supply chain, including spills from produced water, Parish et al. (2013) concluded that the maximum recovery time for petroleum environmental effects would exceed that from biofuels. On the other hand, post-restoration recovery of wetland habitats is slow (years to decades), and wetland functions that depend on organic soils, substrates, hydrology, and vegetation developed over thousands of years may take hundreds or thousands of years to recover pre-conversion conditions, especially in areas where high densities of wetlands have been extensively drained for large-scale agriculture.

The GREET and BEIOM analyses presented in other chapters do not include effects on wetlands specifically, so a quantitative comparison between impacts of the two industries on wetlands and their essential ecosystem service cannot be provided here. However, the freshwater ecotoxicity and eutrophication potentials would also be relevant for wetlands (see Chapter 10, section 10.5 for a discussion of those results).

14.6 Horizon Scanning

Trends in status of the nation's wetland resources suggest that a reversal of gains associated with the establishment of "no net loss" policies may be at risk, with the most recent NRI documenting net wetland losses (2012–2017, Figure 14.5). These losses include direct losses by drainage and filling, in addition to losses resulting from changes in the composition of other freshwater systems away from palustrine wetlands towards more lake-like, lacustrine conditions. The effects on wetlands from agriculture are amplified by climate change. A trend towards climate extremes, including altered timing and intensity of precipitation events in the U.S. PPR (McKenna et al., 2017), is expected to produce higher low and average flow rates (Kelly et al., 2017; Johnston, 2014) with corresponding shifts in wetland condition across the U.S. PPR. The risk to wetlands in regions with high-production corn and soy will likely increase as climate and land use change interact to exacerbate impacts on grassland and wetland ecosystems (Jager et al., 2020; Mckenna et al., 2019).

Perennial grasses could be a cost-effective option to corn and soy feedstocks (Hill and Olson, 2013), but may not provide as much food for migrating waterfowl that have become adapted to (and possibly dependent on) current feedstocks. Furthermore, they still require land to grow, and may have a net negative effect on wetlands if they contribute to further wetland draining and consolidation. Climate projections suggest that climate conditions that support wetlands will shift eastward in the U.S. PPR to areas that have been extensively drained for agriculture (Johnson et al., 2005). Sedimentation under expected future increased precipitation in the Central Plains is predicted to fill many current wetlands, and this should be considered when planning future wetland conservation programs (Skagen et al., 2016).

Incorporation of persistent wetland vegetation (e.g., *Typha* spp. (invasive), *Spartina* spp., mangroves) as biomass feedstocks (Berry et al., 2017; Jakubowski et al., 2010) could potentially motivate preservation of remaining inland and coastal wetlands and restoration of converted wetlands, while improving habitat quality for many species, in addition to fuel production. In addition, federal and state programs can have a positive influence on wetland conservation. For example, the USDA and Department of the Interior (DOI) have multiple conservation programs focused on conserving and enhancing wetlands on agricultural, as well as non-agricultural, lands. Examples of these programs include the USDA NRCS Agricultural Conservation Easement Program (ACEP; includes what was formerly known as the Wetlands Reserve Program) that purchases easements and restores wetland functions and values on enrolled these easement protected lands; the USDA Farm Service Agency's (FSA) CRP that pays farmers to take environmentally sensitive lands out of production and includes several wetland restoration and enhancement practices; and the FSA's Farmable Wetland Program that is designed to restore previously farmed wetlands and wetland buffers to improve vegetation and water flow. Since 1991, restoration of more than 5 million acres of wetland and grassland habitats in the U.S. PPR through the CRP and ACEP has had positive impacts on water storage, reduction in sedimentation and nutrient loading, plant biodiversity, carbon sequestration, and wildlife habitat (Gleason et al., 2011). Another program influencing wetlands on agricultural lands in addition to non-agricultural lands is the North American Wetlands Conservation Act (NAWCA) grant program. Administered through DOI's U.S. Fish and Wildlife Service since 1989, NAWCA grant funds in combination with 2:1 partner matching contributions from more than 6,300 partners have contributed to the protection, restoration, and enhancement of approximately 30.7 million acres of wetlands and associated upland habitats in all 50 U.S. states, 31 Mexican states, 10 Canadian provinces, and multiple territories.

14.7 Synthesis

*14.7.1 **Chapter Conclusions***

- The area of wetlands converted to corn and soy cropland from 2008 to 2016 in the United States is estimated at nearly 275,000 acres, with most losses occurring in the Prairie Pothole Region. Given the lack of national or regional datasets to track changes in RFS-attributable acreage, the extent of wetland losses directly attributable to the RFS Program cannot be more accurately estimated in the RtC3.
- The USDA Natural Resources Conservation Service (NRCS) reported that the total area of "wetland and deepwater habitats" in the conterminous United States decreased by 24.3 thousand acres between 2012 and 2017 (USDA, 2020). This is the first record of a net loss in total area in this resource category since its addition to the National Resource Inventory in 1992.
- Wetland gains and losses are not distributed evenly across wetland types or sizes. Since 2007, the nation has lost 120.3 thousand acres of palustrine (marsh-like) wetlands and gained 205.9 thousand acres of lacustrine (lake-like) habitats in the conterminous United States. The diverse wetlands within these classes support different species and perform different

ecosystem functions, including loss of functions that impact watershed hydrology, water quality, and water quantity.

- Small, seasonal wetlands are being lost at the fastest rate. The loss and consolidation of small wetlands to promote crop production has negatively impacted amphibians, invertebrates, and other aquatic species that depend on shallow water depths for reproduction. Shifts to longer hydroperiods in large or consolidated wetlands, have more uniform (less diverse) invertebrate communities and can support fish that prey on insects and amphibians.
- Small wetlands and ponds are primary sources of water for aquifer recharge in the Northern Prairies. Recent studies in the Canadian PPR found that while permanent ponds and wetlands are sources for recharge to aquifers, wetlands with surface water ponds that dry out every year play the dominant role in groundwater replenishment.
- While some Endangered Species Act (ESA)-listed and other waterbirds have declined, waterfowl (ducks, geese, swans) as a group have not experienced declines over the past decade, possibly due to availability of food (grains), increased precipitation, and the interspersion of ponded waters and agricultural fields along migration routes.
- Shifts to corn and soybean production have resulted in more frequent application of chemicals, including pesticides and fertilizers. Increased usage of neonicotinoid insecticides is of particular concern because of their high toxicity to invertebrates, which are important food sources for wetland-dependent taxa.
- Evidence from the U.S. PPR suggests that trends in larger wetland size, shifts to lakes and ponds (vs. vegetated wetlands), and prolonged and more frequent flooding are due to the combined effects of climate and increased wetland ditching and consolidation. These trends are highly correlated with increased annual precipitation, which is projected to continue.
- Farm-scale management decisions to drain or conserve wetlands are based on local conditions and eligibility for conservation incentives, both of which vary widely across the U.S. PPR. Increased participation in conservation compliance for farm benefits has potential to mitigate some regional effects of wetland loss, including habitat fragmentation, increased flood risk, and loss of ecosystem resilience in this agriculture- and wetland-dominated landscape.
- Pesticides were found in more than 60% of drained wetlands in Iowa. The most common were chloroacetanilide and triazine herbicides, and their degradate compounds. Neonicotinoids were also detected frequently, with clothianidin being the most frequently detected (98% of samples), followed by thiamethoxam (54%) and imidacloprid (48%).

Concentrations in samples exceeded both the acute and chronic aquatic life benchmarks established by EPA. It is not known how export of seasonal and legacy contaminants from drained wetlands in Iowa—where 95% of all wetlands have been drained—or elsewhere is affecting water quality in rivers, streams, and groundwater.

- Amphibians are declining faster than any other vertebrate group globally and habitat loss is one of the primary drivers for this pattern. In the U.S. PPR, one important region undergoing land conversion to corn and soy production, one study quantified Conservation Reserve Program (CRP) amphibian habitat from 2007 to 2012. Results show that from 2007 to 2012, lands in the CRP areas declined 35% across the U.S. PPR and 22% of this land lost was prime amphibian habitat. Within this region, the percentage of total CRP land (as of 2012) that is important to amphibians varied between 20% for the Des Moines Lobe (north-central Iowa) to 32% for the Northern Glaciated Plains region (roughly eastern half of the Dakotas). This illustrates the importance of the conservation of seminatural land as amphibian habitat.

14.7.2 Conclusions Compared with the RtC2

The conclusions of this report are consistent with the second biofuels report to Congress but provide new information documenting (1) recent negative trends in the total area of wetland and deepwater habitats in the conterminous United States, (2) persistent trends in net losses of palustrine and estuarine wetland habitats on cropland, pastureland, and CRP land, (3) preferential loss or consolidation of smaller wetlands and associated and ecosystem services, with concomitant degradation of habitat quality for many wetland species, (4) more frequent application of chemicals (pesticides, fertilizer) that persist in drained wetlands, and increased usage of neonicotinoids, which are harmful to aquatic invertebrates, (5) adverse effects of chemical applications, wetland ditching, and wetland consolidation on amphibian populations, (6) positive trends in surveyed populations of most migratory waterfowl (ducks, geese, swans), albeit with recent (2008–2016) negative trends in wetland habitats suitable for duck breeding pairs in the U.S. PPR, (7) uncertainty about population trends of other migratory waterbirds and declines in some federally endangered or threatened (ESA-listed) waterbirds along historical migration corridors, and (8) effects of wetland ditching and consolidation on critical ecosystem services, including water purification, groundwater/aquifer recharge, and flood prevention/mitigation.

14.7.3 Scientific Uncertainties

- Environmental effects of wetland loss or impairment associated with increased corn and soy production in the United States are well documented in the literature. However, the influence of differing mandates, objectives, and methods used by state and federal agencies to monitor and quantify wetland change limit comparison of results from national surveys, and add uncertainty to attempts at RFS Program attribution for even basic estimates of change (e.g., net gains or losses in area of wetland and deepwater habitats by land use type).

- Some of these discrepancies in results of studies on wetland gains and losses result from uncertainty associated with different methods and standards used for detection and classification of wetland and deepwater habitats. National and regional diversity in wetland ecosystem types, and climate-driven variability in seasonal and interannual patterns of wetland vegetation and inundation, make accurate classification more difficult for this resource category than for more uniform land covers/land uses (e.g., forests, monoculture crops).
- Another source of uncertainty in both national and regional surveys is the difficulty of separating the effects of land use versus climate on wetland change. In the U.S. PPR, for example, a long-term trend towards wetter conditions in the U.S. PPR that began in the early 1990s has contributed to observed shift towards lacustrine (lake-like) habitats and overall decrease in the amount of land dry enough to provide cropland for biofuels or habitats for many wetland species. Over the same period, wetland conversion and consolidation has contributed to shifts from shallow, vegetated wetlands towards deeper, open-water habitats.

14.7.4 Research Recommendations

- Current RFS Program wetland assessments rely on national surveys designed for other (non-RFS) programmatic and management objectives. A national program is needed to identify thresholds related to wetland losses that would greatly increase the marginal damages of additional losses of wetland acreage for biofuel production. Support for the NRCS Wetland Reserves Program and related wetland conservation programs provides some insurance against reaching critical points of functional losses until such thresholds are better understood. For example, increasing the acreage of wetlands protected nationally to roughly 10% of historical wetlands or more (i.e., 20 million acres or more) as opposed to the current enrollment of 1% of historical acreage is predicted to improve sustainable agriculture for biofuel and food production. In addition, interim targets could be established similar to the Hypoxia Task Force for nutrients, and data sharing should be implemented to measure the multiple benefits across programs from such increases.
- Current RFS Program wetland assessments rely on inferences made from changes in wetted area between temporal end points. New research is needed to relate change in wetted area (attributed to the RFS Program) to analyze effects of areal losses or gains to specific wetland functions and communities. Research that includes regional surveys and places wetlands into

a watershed/landscape context with interrelated ecosystems (e.g., streams, lakes, grasslands, forests, other wetlands) and human systems would improve the accuracy of future RFS Program assessments.

- Assessments of "wetland biodiversity" would benefit from the development of metrics for assessing habitat heterogeneity (ecosystem diversity needed to support the range of ecosystem services provided by wetlands, including water purification, aquifer recharge, source water for river baseflow, recreation, and biodiversity), habitat suitability for targeted species and communities (species biodiversity), and landscape attributes that enable wetlands to function (e.g., integration with grassland and stream ecosystems).

14.8 References

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15. Invasive or Noxious Plant Species

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Key Findings

- Direct impacts to date on the environment from the cultivation of invasive or noxious plant species as biofuel feedstocks have not been observed, since most biofuel is produced from a small number of non-invasive feedstock species (i.e., corn and soybean).
- Impacts from the cultivation of corn and soybeans on the evolution of herbicide-resistant weeds do exist, although it is unclear to what extent impacts can be attributed to corn and soybeans grown to meet either biofuel demand generally or the specific requirements of the RFS Program. Since the RFS was enacted, herbicide-resistant weeds have increased production costs for farmers in terms of herbicide expenditures and in their overall investment in technology and production systems. However, this temporal association alone is not sufficient to determine causation.
- The likely future effects of the RFS Program from invasive or noxious feedstocks are uncertain due to many factors. However, if biofuels continue to be produced mostly from corn and soybean, there will be no likely future effects directly from potential invasive or noxious feedstocks. This is because corn and soybean are not invasive. Two potentially invasive feedstocks (i.e., giant reed [*Arundo donax*] and napier grass [*Pennisetum purpureum*]) are part of approved biofuel pathways under the RFS Program. They could produce effects *if* they are grown in the future and *if* additional registration, reporting, and recordkeeping requirements that are in place and designed to limit their spread are not sufficient to prevent escape and invasion. However, as of the publication of this report, no Renewable Identification Numbers (RINs) have been generated that involve these feedstocks nor have incipient invasions or impacts been observed as a result of their production for biofuel.
- Likely future effects from herbicide-resistant weeds will continue to grow if current trends hold in the incidence of new cases and number of weed species that are resistant to multiple herbicide sites of action. As with impacts to date, future impacts from the cultivation of corn and soybeans on the evolution of herbicide-resistant weeds are likely to occur, but it will be challenging to determine what extent of impacts can be attributed to corn and soybeans grown to meet either biofuel demand generally or the specific requirements of the RFS Program.
- It is not possible to reach a firm conclusion regarding the relative overall invasion risk posed by biofuels compared to petroleum. Risks of invasion associated with petroleum exploration and extraction include both the introduction of non-native species via hitchhiking on

machinery and infrastructure and the facilitation of non-native dominance through habitat disturbance across a broad range of habitats, including terrestrial and marine.

Chapter terms: herbicide resistance, invasive plants, naturalized plants, noxious weed, weed risk assessment

15.1 Overview

15.1.1 Background

This chapter addresses the potential effects of the Renewable Fuel Standard (RFS) Program on "the growth and use of cultivated invasive or noxious plants and their impacts on the environment and agriculture." Potentially invasive plants that may be cultivated as biofuel feedstocks are discussed in the context of future impacts and research, because these plants do not serve as significant feedstocks today. As discussed in Chapter 2, the four biofuels that are the focus of the RtC3 are domestic corn ethanol, domestic soybean biodiesel, domestic biodiesel from fats, oils, and greases (FOGs), and ethanol from Brazilian sugarcane. Corn, soybeans, FOGs, and sugarcane cultivated in Brazil and converted to biofuel for export to the United States are not invasive plants. That said, herbicide-resistant weeds in domestic cultivated feedstocks (corn and soybean) are discussed below.

This chapter encompasses species that may be considered invasive or noxious plants. The federal definition of an invasive species is a "non-native organism whose introduction causes or is likely to cause economic or environmental harm, or harm to human, animal, or plant health" (EOP, 2016). In contrast, invasive plant experts define an invasive plant as "naturalized plants that produce reproductive offspring, often in very large numbers, at considerable distances from parent plants,… and thus have the potential to spread over a considerable area" (Richardson et al., 2000). Although these two definitions differ, they describe two fundamental properties of invasive species: high establishment/spread potential and high impact potential. Species that readily naturalize, reproduce in great numbers, and spread through diverse means often cause significant harm where they occur; and species that cause significant harm can only do so if they are readily able to spread and infest natural and managed systems (Cousens, 2008). Thus, both definitions describe invasive plant species.

The term noxious weed is usually used by government agencies to refer to harmful plants they regulate. The U.S. Plant Protection Act (2000) defines a noxious weed as "any plant or plant product that can directly or indirectly injure or cause damage to crops,… livestock, poultry, or other interests of agriculture, irrigation, navigation, the natural resources of the United States, the public health, or the environment." Similar to the federal definition of an invasive species, this term focuses on the consequences of plant invasions. Factors related to all of these definitions are considered within major tools that seek to predict which plants are likely to become invasive or weedy (e.g., weed risk assessments) (e.g., IPPC, 2013; Koop et al., 2012; Pheloung et al., 1999).

Trends in the total number and individual distribution of invasive or noxious plants in the United States and elsewhere are not easily accessible. To the authors' knowledge, there is no single U.S. government entity or program, or nongovernmental group, that collates and makes available this type of information. Furthermore, differences in terminology and how species are categorized as invasive, naturalized, escaped, or introduced can confound efforts to accurately describe the exotic flora of a region. Historically, in North America, the annual rate of first-recorded occurrences of vascular plants (which are not equivalent to plant invasions but provide an upper bound) is estimated to have peaked before 1900 at approximately 150 per year and gradually declined to approximately 50 per year by 2000 (Seebens et al., 2017). Currently in the U.S. flora, there are approximately 16,600 native vascular plant species (USDA NRCS, 2019). An additional 4,300 to 5,100 are reported as naturalized exotic species, while about 1,600 of these are considered invasive (Simpson et al., 2019; USDA NRCS, 2019). Among most U.S. states and regions, about 15–30% of the floras consist of naturalized exotics (FNA Editorial Committee, 1993). The search for new bioenergy plants and the improvement of others is likely to lead to the introduction (VIASPACE, 2012) and possibly establishment of new plant species in the United States. Bioenergy plants may escape from production systems in a number of ways (Figure 15.1).

In agricultural systems, invasive species reduce crop yield and increase costs of production, while in natural ecosystems they negatively impact ecological communities and ecosystem processes in ways that are not easily monetized. Estimates of the economic losses and costs of all invasive species vary, including totals from $11 billion to over $120 billion annually in North America and the United States, respectively (Diagne et al., 2021; Pimentel et al., 2005). According to one study, the roughly 500 non-native plants that have become weeds of crops and forage in the United States, specifically, account for an estimated $24 billion in lost crop productivity and $3 billion in control and management costs annually (Pimentel et al., 2005). Ecological and ecosystem effects of invasive species generally represent changes in species, community, or ecosystem-level measurements. In one global review, invasive plants impacted ecological and ecosystem measurements in a statistically significant way in over 60% of the cases in which they were studied (Pyšek et al., 2012). When fire frequency or intensity was considered, invasive plants had a significant effect 100% of the time (Pyšek et al., 2012). Furthermore, individual invasive species often have multiple, co-occurring economic, ecological, and ecosystem services effects (Vilà et al., 2010).

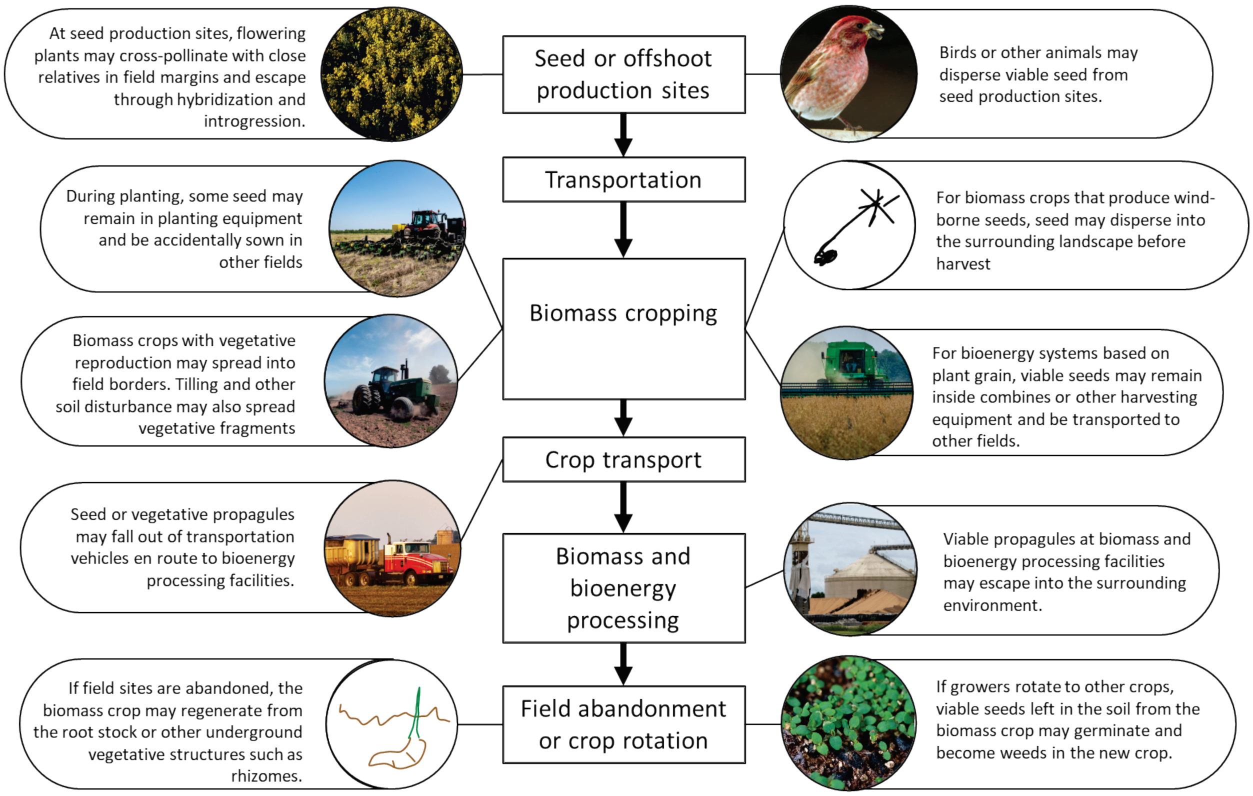

Figure 15.1. Possible ways that bioenergy plants may escape from the production pathway. The production pathway begins with sites where workers propagate the bioenergy species for planting and ends with abandonment or rotation of cropping sites. Image sources (clockwise from top left): USDA–Jack Dykinga; USFWS–Thomas G Barnes; Original graphic–Caroline Ridley; EPA–no photographer named; USDA–Lance Cheung; USDA–Peggy Greb; Original graphic–Caroline Ridley; EPA–no photographer named; USDA–Lance Cheung; USDA–Lance Cheung.

This chapter also addresses herbicide-resistant weeds that arise during biofuel feedstock cultivation. This is the most significant potential effect from biofuels and the RFS Program in the area of invasive plants to date, as the aforementioned feedstocks are not invasive. Herbicide-resistant weeds can be considered a subset of invasive or noxious plants. Herbicide resistance is the inherited ability of a plant to survive and reproduce following exposure to a dose of herbicide normally lethal to the wild type (WSSA, 1998). Herbicide-resistance is relevant to biofuel feedstock production, because the two most important domestically produced biofuel feedstocks (corn and soybean) have associated herbicide-resistant weeds. Herbicide-resistant weeds have been identified as both a result of and a growing threat to agricultural production worldwide (Pannell et al., 2016).

Trends in herbicide-resistant weeds in the United States and elsewhere were largely anecdotal until the International Survey of Herbicide-Resistant Weeds was established in the mid-1990s. From 1970 to the mid-1980s, there was a slow rise in the number of reported cases of herbicide-resistant weeds. The rate of increase accelerated in the mid-1980s, and now there are more than 160 total reported cases (Heap, 2020). Trends in unique cases of herbicide-resistant weed species associated with fields of corn or soy also show steep upward trends since the late-1980s; the incidence of weed species resistance to multiple sites of action is also increasing (Figure 15.2). [1]

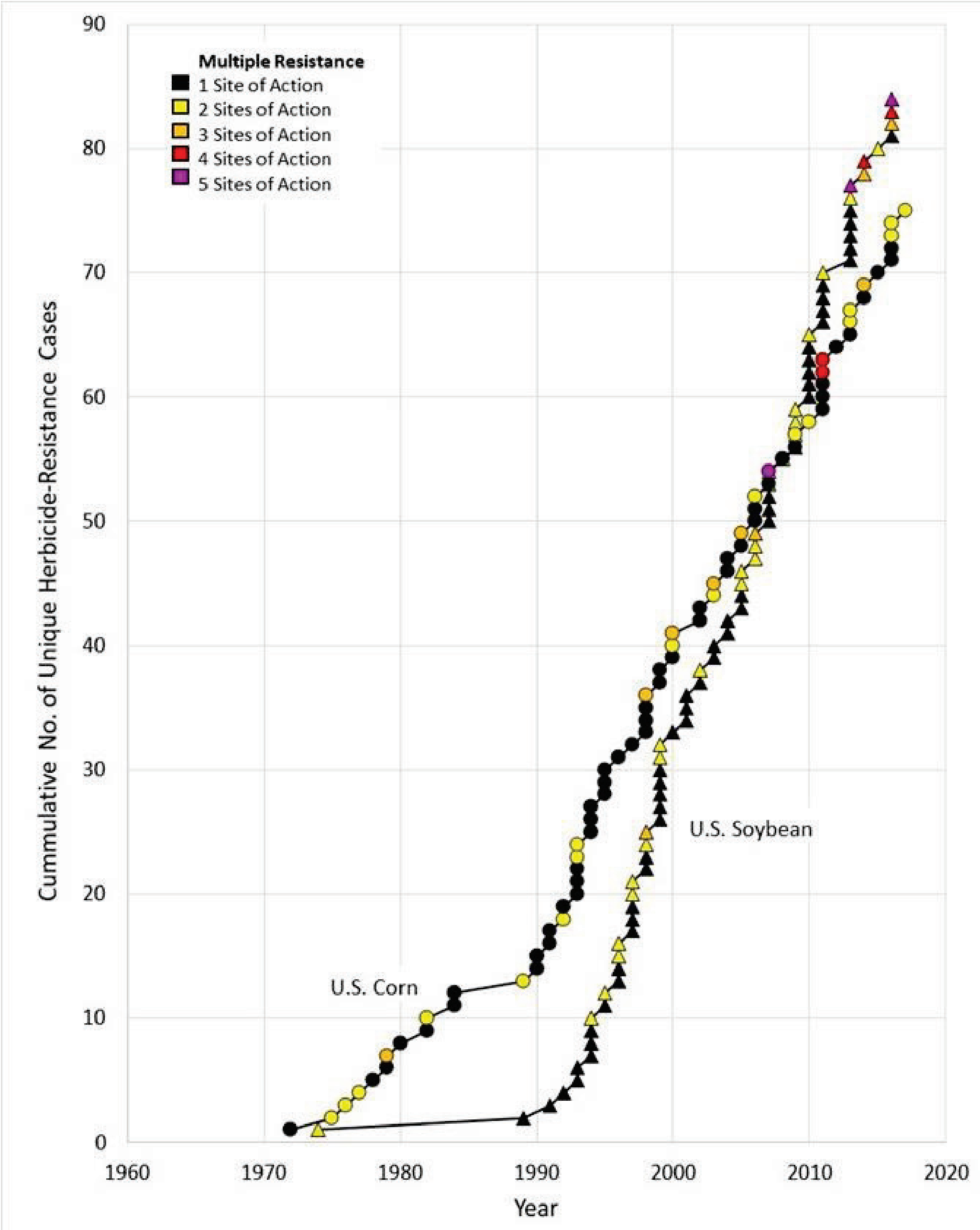

Figure 15.2. Cumulative number of unique herbicide-resistant cases in the United States by major biofuel feedstock. Each unique case is color coded to indicate the number of herbicide sites of action to which the weed was reported resistant. This figure is based on data obtained from the International Herbicide-Resistant Weed Database. Permission to use the data was provided by Ian Heap. Data on other crops and countries can be obtained from the database.

[1] A case is defined as a unique combination of weed species and evolved resistance to herbicide(s) with a particular site of action. Site of action is the specific process in plants that the herbicide disrupts to interfere with plant growth and development.

The impacts of herbicide-resistant weeds are largely felt by farmers. Herbicide-resistant weeds increase production costs for farmers in terms of herbicide expenditures and in their overall investment in technology and production systems (Davis and Frisvold, 2017). These weeds can necessitate more complex weed management programs and may cause a shift in the crops that can be profitably grown (Pannell et al., 2016). The impacts of herbicide-resistant weeds on natural systems are not well-characterized.

A number of legal tools exist aimed at preventing, managing, and controlling invasive species and mitigating their ecological and economic impacts in the United States (Johnson et al., 2017). Current laws are generally tailored to particular species, vectors of introduction, or recipient habitats and in some cases impose specific responsibilities on federal agencies. Executive Order (E.O.) 13751 ("Safeguarding the Nation from the Impacts of Invasive Species") (EOP, 2016), amended a previous E.O. to direct "actions to continue coordinated Federal prevention and control efforts related to invasive species." These two E.O.s establish some of the most comprehensive and unifying frameworks guiding activities of federal agencies with respect to invasive species. Of particular relevance, each federal agency shall "refrain from authorizing, funding, or implementing actions that are likely to cause or promote the introduction, establishment, or spread of invasive species in the United States unless, pursuant to guidelines that it has prescribed, the agency has determined and made public its determination that the benefits of such actions clearly outweigh the potential harm caused by invasive species" (EOP, 2016). Some observers assert that such language specifically constrains actions relevant to the development and cultivation of feedstocks with known histories of invasiveness (Raghu et al., 2006). Understanding the risks of potential feedstock invasions is thus a critical step toward adhering to these E.O.s.

15.1.2 Drivers of Change

The drivers determining the impact of biofuel feedstocks that are potentially invasive include (1) the biological characteristics of the feedstock, (2) the acreage on which the feedstock is grown, and (3) cultivation, harvesting, and transportation practices (Figure 15.1). Biofuel feedstocks vary widely in their biological characteristics, which can influence whether they are likely to escape cultivation and sustain populations in unmanaged settings or become weeds of other crops or forage. Their characteristics also determine the nature of impacts should they escape (e.g., toxicity, whether the feedstock promotes fire). In addition, a larger scale of cultivation will increase the opportunity for escape and establishment. This so-called "propagule pressure" is a major contributor to invasion potential by enabling incipient invasive populations to overcome factors that make it difficult for small populations to persist (Simberloff, 2009; Colautti et al., 2006; Lockwood et al., 2005).

The drivers for the development, spread, and impacts of herbicide-resistant weeds include biological, anthropogenic, and environmental factors (Perotti et al., 2020). Fundamentally, genetic changes must enable a weed to avoid being killed by an applied herbicide, and those genetic changes must be passed on to subsequent generations. In the 1980s, widespread resistance to certain types of herbicides arose from simple genetic changes that were easily passed from parents to offspring and that also spread geographically (Shaner, 2014). After the introduction of herbicide-resistant crop varieties in the 1990s (see Chapter 3, section 3.2.1.5.3), patterns of herbicide use changed. For soybeans, the change was dramatic. There were large increases in the total amount of glyphosate applied per area, the proportion of total herbicide use attributed to glyphosate, and a reduction in diversity of herbicides applied to the crop (Kniss, 2018; Perry et al., 2016). Scientists generally agree that these kinds of changes created environmental conditions in which a new wave of herbicide-resistant weeds began to emerge (Perotti et al., 2020; Green, 2018; Heap and Duke, 2018; Benbrook, 2016; Heap, 2014) but Kniss (2018) recognizes that reducing future impacts from herbicide-resistant weeds will involve more than just reducing herbicide use (see section 15.3.4).

15.1.3 Relationship with Other Chapters

Invasive plants may affect terrestrial (Chapter 12), aquatic (Chapter 13) and wetland (Chapter 14) communities. However, the four primary biofuels examined in the RtC3 are not invasive and the impacts of herbicide-resistant weeds on natural systems are not well-characterized. Additional information about herbicide usage can be found in Chapter 3 (Biofuel Supply Chain; later revisited in Chapter 12) and Chapter 10 (Water Quality).

15.1.4 Roadmap for the Chapter

Section 15.2 contains conclusions from the 2018 Report. Section 15.3 addresses the impacts of biofuel feedstocks to date; it relies on updated literature since the 2018 Report but no new analysis. Section 15.4 addresses likely future impacts. Section 15.5 is a brief comparison of the invasive and noxious weed impacts from petroleum. Section 15.6 scans the horizon for potential impacts from other feedstocks that have been evaluated by EPA for meeting greenhouse gas requirements under the RFS Program or that have received ample attention in the literature. Section 15.7 is a synthesis of the chapter, including conclusions, uncertainties and limitations, and research recommendations.

15.2 Conclusions from the 2018 Report to Congress

The overall conclusions about invasive species and biofuels from the 2018 Biofuels report were:

- Current biofuel feedstocks pose little risk of becoming invasive species. Cultivation of herbicide-resistant feedstock crops (e.g., glyphosate-resistant soybean) and concomitant application of the associated herbicide (e.g., glyphosate) has the potential to contribute to herbicide-resistant weed development, just as herbicide-resistant crops grown for other purposes. [2]

- Biofuels are primarily produced in the forms of bioethanol and biodiesel derived from food crops (i.e., non-invasive first generation biofuels – corn and soy). Hence, current production of biofuel feedstocks poses little risk of invasion, consistent with findings in the 2011 Report.
- Weed risk assessments, which are sometimes part of the biofuel regulatory process, provide information on invasion risk and are designed to inform protective management of species and varieties that are predicted to be invasive.[3]
- Increased cultivation of crops engineered for herbicide resistance (e.g., glyphosate) and concomitant application of the herbicide has led to a widespread increase in the number of glyphosate-resistant weed species.[4]
- Potentially invasive species approved as feedstocks require risk management actions under current RFS requirements. However, invasive species are not presently being used for commercial scale production of biofuels.
- Methodological advancements for weed risk assessments and lessons from other industries (e.g., horticulture) should be incorporated to inform on potential invasiveness of biofuel feedstocks.
- Modeling and field work are needed to investigate the impacts of gene flow between novel feedstock varieties (genetically engineered, selectively bred, or a combination) and local natives.

[2] In the 2018 report, the text of this conclusion read, "Current biofuel feedstocks pose little risk of becoming invasive species. Cultivation of herbicide-tolerant feedstock crops (e.g., glyphosate-tolerant soybean) and concomitant application of the associated herbicide (e.g., glyphosate) has the potential to contribute to herbicide-resistant weed development, just as herbicide-tolerant crops grown for other purposes." The word "tolerance" was replaced with "resistance" for this version to more accurately reflect the accepted definitions of these terms.

[3] In the 2018 report, the text of this conclusion read, "Weed risk assessments, part of the formal biofuel regulatory process, provide information on invasion risk and are designed to inform protective management of species and varieties that are predicted to be invasive." The phrase "which are sometimes" was added and the word "formal" was removed for this version. This is to more accurately reflect that weed risk assessments conducted then and now to support regulatory decision-making under the RFS are conducted only when deemed appropriate and are not required under any applicable rule or formalized process.

[4] In the 2018 report, the text of this conclusion read, "Increased cultivation of crops engineered for herbicide tolerance (e.g., glyphosate) and concomitant application of the herbicide has led to a widespread increase in the number of glyphosate-resistant weed species." The word "tolerance" was replaced with "resistance" for this version to more accurately reflect the accepted definitions of these terms.

15.3 Impacts to Date for the Primary Biofuels

15.3.1 Literature Review

The primary domestic plant feedstocks used to date (corn, soybean) are not invasive. FOGs are a byproduct of other activities and do not have any known relationship to invasive species. Sugarcane is invasive in parts of the United States, but it is not invasive when grown in Brazil and processed into ethanol prior to export to the United States. The production systems in which biofuel feedstocks are grown likely contribute to the emergence of co-occurring herbicide-resistant weeds and the increasing incidence of weed species that have resistance to multiple herbicide sites of action (Figure 15.2). In the first two triennial Reports to Congress, this conclusion was mostly in reference to production systems that relied on herbicide-resistant crop varieties. A more nuanced understanding of herbicide resistance and its history, drivers, and management have been clarified in recent years. Additional production practices (e.g., crop rotation, tillage) and weed management practices (e.g., herbicide rotation and mixtures) are now widely seen as affecting the incidence, geographic distribution, and severity of impacts from herbicide-resistant weeds (Perotti et al., 2020; Kniss, 2018; Shaner, 2014).

For corn and soybean specifically, cases of herbicide-resistant weeds continue to rise (Figure 15.2), including resistance to commonly applied herbicides by percentage of crop treated (glyphosate and atrazine for corn and glyphosate and sulfentrazone for soybean; see Chapter 3 section 3.2.1.5). Herbicide resistance is documented by site of action (the specific process in plants that the herbicide disrupts to interfere with plant growth and development). Sites of action are more relevant to understand incidence and management of herbicide resistance because an evolved resistant trait can arise from any of the individual herbicides with a particular site of action and affect the future utility of all individual herbicides with a particular site of action. Cases of resistance to multiple herbicide sites of action (including 3, 4, and 5 sites of action) are also rising (Figure 15.2). For instance, Palmer's amaranth (*Amaranthus palmeri*) found in soybean in Arkansas in 2016 was shown to be resistant to five herbicide sites of action.[5] All herbicides with a particular site of action that are applied to corn and soybean are also applied to other crops (Kniss, 2018), and on corn and soybean used for products other than biofuels. Attribution to biofuels or the RFS Program remains a challenge (discussed further in section 15.3.3).

15.3.2 New Analysis

No new analysis was conducted by EPA for this chapter.

[5] https://www.weedscience.org/Pages/Case.aspx?ResistID=18156

15.3.3 Attribution to the RFS

While the phenomenon of herbicide-resistant weeds has been well documented, it is not clear to what degree their emergence might be attributed to the RFS Program per se. Overall, a relatively small fraction of corn acreage (i.e., between 0 and 3.5 million acres in 2016; see Chapter 6 section 6.4 and Table 6.12) and an unquantified fraction of soybean acreage (see Chapter 7) is estimated attributable to the RFS Program. It might be tempting to attribute a proportional fraction of the total cases of herbicide-resistant weeds to the feedstock production associated with biofuels used to satisfy requirements of the RFS Program. However, there is no evidence to suggest that total feedstock acreage or production volume is linearly related to herbicide-resistant weed incidence. The incidence of weed resistance also has a spatial aspect. Resistance occurs in a place and time and potentially spreads locally or regionally via natural and human-assisted dispersal; to date, corn acreage estimated attributable to the RFS Program has not been allocated to the landscape. Furthermore, no apparent causal or quantitative analysis has been undertaken to estimate increases in herbicide application associated directly with plantings for biofuels or other changes in production practices that might give rise to an increase in risks of herbicide-resistant weeds. No evidence exists that would suggest extensification versus intensification of corn and soybean production drive different effects with respect to herbicide-resistant weeds. Both have logical potential to increase the incidence and severity of impacts from these weeds.

15.3.4 Conservation Practices

The lack of direct negative effects of invasive species on the environment from the cultivation of current feedstocks could be maintained by continuing to rely only on corn, soybean, FOGs, and imported sugarcane as feedstocks for biofuel. Additional considerations for avoiding negative effects from potential future feedstocks are discussed in section 15.6.

Offsetting or managing the negative effects of herbicide-resistant weeds that may evolve during the cultivation of corn and soybean feedstocks is more challenging. Published best management practices at the field scale exist to prevent or delay the evolution of herbicide-resistant weeds. Practices include strategic tillage and crop rotation among many others (Beckie and Harker, 2017; Norsworthy et al., 2012), but there is discussion and disagreement among experts about how effective some popular practices are likely to be (e.g., Gressel et al., 2017; Délye et al., 2013). Some practices used to prevent or manage the negative effects of herbicide-resistant weeds have environmental trade-offs (e.g., tillage can impact soil erosion) (Van Deynze et al., 2022). Field or farm-scale practices may also need to be supplemented by regional or landscape-scale management to keep weeds susceptible to herbicides as a public good, although experiments and solutions at the necessary scale are lacking (Bagavathiannan et al., 2019; Gould et al., 2018).

15.4 Likely Future Impacts

The primary domestic plant feedstocks likely to be used in the United States between now and 2025 (corn, soybean, canola) are not invasive, and FOGs and biogas from CNG/LNG are byproducts of other activities and do not have any known relationship to invasive species (see Chapter 2, section 2.3.2 for more on the feedstocks considered in the likely future). The production systems in which biofuel feedstocks are grown will likely continue to contribute to the emergence of herbicide-resistant weeds, given the pace of resistance evolution of these weeds in corn and soybean to date (Figure 15.2). At least one author argues that the number of new species with herbicide resistance is slowing (Kniss, 2018). This may reflect a greater reliance than in the past on herbicides to which weeds have a more difficult time evolving resistance, but could also indicate that there is a continually shrinking fraction of weed species that have not yet evolved resistance (Kniss, 2018). Kniss (2018) does not address the geographic extent of specific cases of herbicide-resistant weeds. When cases are first reported, they may be limited to a relatively small geographic area, but the potential for future spread by farm equipment and/or via harvested material is high (Beckie, 2006). Newly evolved herbicide resistance genes may comingle, producing populations of weeds that either have stronger resistance to an herbicide site of action or multiple resistance. In addition, the lack of new herbicides with alternative sites of action or other simple and scalable non-chemical methods for controlling weeds in corn and soybeans indicates that herbicide-resistant weeds will continue to cause impacts on agriculture beyond 2025.

15.5 Comparisons with Petroleum

Risks of invasion posed by the biofuels sector are typically understood to relate primarily to potential for certain feedstocks to escape cultivation and cause economic or ecological damage and the evolution of herbicide-resistant weeds. In contrast, invasive plant risks associated with the petroleum industry are generally associated with the incidental introduction of species with activities or infrastructure accompanying exploration or extraction. Indeed, multiple aspects of the petroleum industry have been demonstrated to serve or are suspected of serving as vectors for both terrestrial and aquatic invasive species. One example of compelling evidence derives from studies of offshore oil production platforms. One such study identified several known invasive invertebrate species on oil and gas platforms arrayed on the Pacific offshore continental shelf in central and southern California (Page et al., 2006). In another case, a single decommissioned semi-submersible rig, abandoned in 2006 at Tristan da Cunha, Brazil, was found to harbor an intact subtropical reef community including 62 taxa not native to the region (Wanless et al., 2010). One species of invasive marine fish, the violet demoiselle (*Neopomacentrus cyanomos*; now considered likely established on U.S. shores in the Gulf of Mexico), has plausibly been traced to an initial introduction via hitchhiking on mobile oil platforms stationed in the southern Gulf of Mexico (Robertson et al., 2016).

Shipping is also widely recognized as a major pathway for the introduction of non-native species. Oceangoing vessels carry living organisms both in their ballast water and as fouling organisms on their hulls, and these vectors have been responsible for numerous aquatic invasions in the United States and elsewhere. Indeed, recent studies have noted that changes in energy markets may result in substantial shifts in shipping patterns and thus altered risk of invasion for some recipient port systems (Holzer et al., 2017). However, although the transport of petroleum products is a significant component of international vessel traffic, it is nearly impossible to determine which species have been introduced and how many of the total introductions came from that industry. That said, if biofuels reduce the importation of foreign oil as intended by statute, directionally, biofuels should reduce the incidence of aquatic invasive species even though they may have the opposite effect on herbicide resistant species on land. Direct comparison of invasion risk from biofuels vs. petroleum remains extremely challenging.

In terrestrial contexts, the correlation of invasive species with petroleum exploration and exploitation seems to be more anecdotal. One study in Patagonia did observe association of multiple exotic plant species with seismic lines laid to search for oil deposits, suggesting the possibility that the substantial disturbance introduced by exploration may be conducive to the establishment and spread of those species (Fiori and Zalba, 2003). More commonly, published literature simply expresses the conventional wisdom that extractive industries are very likely to contribute to the spread of invasive species, either by directly serving as vectors for propagules or through disturbance-induced dominance of non-natives (Olive, 2018). Unfortunately, no comprehensive examination of the costs of invasive species directly or principally associated with the petroleum industry has been conducted for any region.

15.6 Horizon Scanning

It is uncertain which new feedstocks may contribute to biofuels further in the future. Below, several feedstocks are discussed, including feedstocks that have been evaluated by EPA with respect to greenhouse gas (GHG) requirements under the RFS Program and feedstocks about which a substantial amount of peer-reviewed, published information exists. Discussion is also included of potential improvements to weed risk assessment (WRA) tools, a persistent need identified in the RtC1 and RtC2.

15.6.1 Other Biofuel Feedstocks

Producers indicate interest in growing a biofuel feedstock when they petition EPA to evaluate whether it meets the GHG reduction requirements under the RFS Program. Two potentially invasive feedstocks that went through this process have additional registration, reporting, and recordkeeping

(RRR) requirements that are designed to limit their spread should they be cultivated for conversion to cellulosic ethanol: giant reed (*Arundo donax*) and napier grass (*Pennisetum purpureum*) (U.S. EPA, 2013). These RRR requirements arose in response to public comment. Requirements include a Risk Mitigation Plan (RMP) to be reviewed by EPA in consultation with USDA or information showing such a plan is not necessary (for example, because of specific site conditions). Other potentially invasive feedstocks that may eventually have additional RRR requirements pending public input include Ethiopian mustard (*Brassica carinata*), physic nut (*Jatropha curcas*), field pennycress (*Thlaspi arvense*), common beet (*Beta vulgaris* ssp. *vulgaris*), and a set of short-rotation tree species and hybrids[6] (U.S. EPA, 2017, 2016, 2015a, b, c). These five feedstock types were identified in consultation with USDA through a weed risk assessment process as having some invasive potential that could necessitate mitigation. As of the publication of this report, no RINs have been made with any of the above feedstocks. Furthermore, no incipient invasions or impacts have been observed that are attributable to the RFS Program. Finally, during GHG evaluation of grain sorghum and biomass sorghum (both *Sorghum bicolor,* but bred for different feedstock properties), EPA specifically excluded hybrids of *Sorghum bicolor* and Johnsongrass (*Sorghum halepense*) due to "their potential to behave as an invasive species" (U.S. EPA, 2018, 2014).

Several other feedstocks have been evaluated by EPA for potential inclusion under the RFS Program and received considerable attention as potential large-scale contributors to future biofuels production in the United States, including switchgrass (*Panicum virgatum*), giant miscanthus (*Miscanthus x giganteus*), and various species of algae.[7] Switchgrass is native to the eastern United States, and past studies have suggested that there may be some invasive potential in other regions of the country unless sterility is introduced (Smith et al., 2013; Barney and Ditomaso, 2008). However, recent assessments have generally considered switchgrass at low risk for invasion if utilized as feedstock (Quinn et al., 2014). Unlike *Panicum*, the genus *Miscanthus* is non-native to North America and some species are known invasives (e.g., *M. sinensis* in Tennessee),[8] raising concerns for potential escape and invasiveness of derived biofuel feedstocks. Giant miscanthus, a hybrid between tetraploid *M. sacchariflorus* and diploid *M. sinensis*, has been tested as a potential biofuel in the United States. Giant miscanthus demonstrates increased biomass production compared to parental strains and can be produced as a sterile triploid, thus reducing risks of invasion (Bonin et al., 2017; Quinn et al., 2014). However, vegetative production of the triploid strain limits cost effectiveness, and a fertile, seed-bearing tetraploid strain of *Miscanthus x*

[6] Short-rotation trees are poplars—including the following species, as well as crosses between them: *Populus (P.) deltoides, P. trichocarpa, P. nigra,* and *P. suaveolens* subsp. *Maximowiczii*—and willows—including *Salix (S.) miyabeana, S. purpurea, S. eriocephala, S. caprea hybrid, and S. x dasyclados as well as crosses between S. koriyanagi and S. purpurea, S. viminalis and S. miyabeana, and S. purpurea and S. miyabeana.*

[7] Another prominent feedstock is corn stover. However, because corn stover is a part of corn, there are no additional concerns with respect to invasive plants with corn stover above that already mentioned with corn.

[8] https://www.tnipc.org/invasive-plants/

giganteus, known as "PowerCane," has been proposed as more productive feedstock. Recent studies suggest that escaped PowerCane could prove as or even more invasive than known invasive *Miscanthus* species, suggesting that additional risk assessment may be warranted (Bonin et al., 2017; Miriti et al., 2017). Neither switchgrass nor *Miscanthus* spp. currently have RRR requirements related to potential invasiveness under the RFS Program. Biofuel made from these feedstocks was approved by EPA as meeting GHG reduction requirements under the RFS Program before RRR requirements were routinely considered for potentially invasive feedstocks.

Rapid improvements in genetic and genomic modification technologies raise the possibility of new feedstock varieties that possess traits correlated with increased invasiveness (Allwright and Taylor, 2016). Conversely, feedstocks could also be modified to be less invasive. Academic and commercial laboratories have explored modifications of species that are widely recognized as both promising bioenergy sources and potentially damaging invaders. The Biotechnology Regulatory Services (BRS) of USDA's Animal and Plant Health Inspection Service (APHIS) maintains a database of authorizations related to importation, interstate movement, or environmental release of genetically engineered organisms (APHIS, 2021). Two taxa previously acknowledged to pose invasion risk (Smith et al., 2013) appear in that database: *Camelina* sp. and *Miscanthus* sp. Since 2006, 147 separate permit requests or notifications have been filed for genetically modified *Camelina* sp.; these include strains with modifications for traits such as increased growth rate, enhanced photosynthesis, high yield, and herbicide resistance, all traits potentially linked to invasiveness. It is not clear if any of these modifications of *Camelina* sp. are aimed at increasing utility for biofuel production, as the taxon is utilized in various other industries (e.g., to produce omega-3 fatty acids for human or animal consumption). Similarly, six permits or notifications were filed between 2011 and 2023 for modified *Miscanthus* sp. It is likely that some of these modifications have been made specifically to enhance the potential of this taxon as a biofuel feedstock, as at least one permit was granted to a commercial operation receiving funding from the Department of Energy's Advanced Research Projects Agency-Energy (ARPA-E) to modify perennial grasses specifically for that purpose.

Miscanthus is one of several perennial crops that have been adopted for genetic modifications targeting increased biomass supply for energy production; others include switchgrass (*P. virgatum*), willow (*Salix* sp.), and poplar (*Populus* sp.) (Clifton-Brown et al., 2019). Genetic and genomic modification approaches for these species range from classical transgenic insertion (frequently employing *Agrobacterium*-mediated transformation or biolistic bombardment) to genomic editing using CRISPR-Cas technologies in *Populus*, a taxon for which considerable genomic resources already exist. It is uncertain to what degree introduction of novel traits might alter the risk profiles of these and other species to which similar modifications might be made.

Much interest in algal feedstocks has focused on algae genetically modified to enhance fuel production by improving photosynthetic efficiency, increasing light penetration, or otherwise altering algal metabolism (Abdullah et al., 2019). Substantial cost efficiencies in fuel production can be obtained by producing these modified algae in open systems,[9] which introduces risks associated with release through leakage, interference, or aerosolization. Such releases could lead to competition with or horizontal gene transfer to native algal species, alteration of invaded ecosystems, or even toxicity to exposed organisms (Abdullah et al., 2019; Phang and Chu, 2015). Although algae feedstocks do not have RRR requirements related to potential invasiveness under the RFS Program, understanding the aforementioned potential risks through risk assessments consistent with existing applicable methodologies is important to the future development of these feedstocks (Phang and Chu, 2015).

Despite concerns about invasion risks of some of the feedstocks mentioned in this section (Lewis and Porter, 2014; Quinn et al., 2014; Smith et al., 2013), there appears to be no evidence of any escaping from production sites and causing impacts. This could reflect a lack of data, or it could reflect a lack of effect. At this point, there is no national database of plantings, and few efforts to compile observations of invasive plants beyond published studies and observations by entities working at the local or state levels (Pope, 2015; Daehler et al., 2012). One notable exception is EDDMapS (Early Detection and Distribution Mapping System), which aggregates spatially explicit observations of any invasive species (not limited to biofuel feedstocks) from other databases and individual citizen scientists for the United States and Canada.[10]

For new feedstocks, it may be too early to observe impactful escapes from cultivation due to escape-detection-spread lag times (Smith et al., 2013). For example, of 257 invasive plant-region combinations in the upper Midwest, 197 (77%) showed a statistically discernable lag phase that lasted between 3 and 140 years (Larkin, 2012). Corn and soybeans have not been observed to self-sustain populations in unmanaged settings, so it is unlikely that these feedstocks are in a lag phase and will produce impacts in the future. However, other feedstocks that are being developed and tested (especially those capable of self-sustaining populations outside of cultivation), may not begin spreading for many years. Given the observed uncertainties about invasion post-introduction or establishment and the potentially enormous costs (see section 15.1), utilizing WRA to preclude the use of potentially invasive species is often touted as the most effective way to avoid impacts (Keller et al., 2007).

Finally, the RFS Program could create incentives to conduct research and development on additional novel feedstocks that may pose greater invasion risk, because the traits of a desirable feedstock (e.g., rapid growth, high seed production) are similar to those of an invasive species (Table 15.1). Even

[9] Open systems include ponds and raceways.
[10] https://www.eddmaps.org/

small cultivated acreage of highly invasive feedstocks could lead to considerable negative consequences. Continued future expansion of the number of feedstock species in cultivation would alter the overall risk of invasive impacts.

There are several opportunities for avoiding future negative effects of the cultivation of invasive or noxious feedstocks. First, the methodology for deciding which feedstocks may need additional RRR requirements could be formalized and strengthened. USDA applies a generic WRA methodology, which could undergo changes to make it more relevant to the context of growing and processing feedstocks for biofuels (see below), and the results are used in a case-by-case basis to decide if additional RRR requirements may be necessary. With respect to the feedstocks that already have RRR requirements (*A. donax* and *P. purpureum*), any future RMPs that are implemented during the cultivation of these feedstocks could be evaluated for their effectiveness in preventing incipient invasions and/or impacts. The RRR requirements or specific approaches utilized in the RMP could then be refined as necessary.

Table 15.1. Plant traits under selection for improved biofuel crop performance and economic suitability that overlap with characters of many invasive species. Comparison among traditional field crops, potential biofuel crops, and known invasive species that were introduced for agronomic purposes. "x" indicates presence of a trait and "-" indicates absence of a trait. Table based on Barney and DiTomaso (2010).

	Agronomic crops		Biofuel crops			Invasive species with agronomic origin	
Trait	**Corn**	**Soybean**	**Switchgrass**	***Miscanthus* x *giganteus***	**Giant reed**	**Johnsongrass**	**Kudzu**
Perennial	–	–	x	x	x	x	x
C4 photosynthesis	x	–	x	x	–	x	–
Rapid establishment	x	x	–	x	x	x	x
Highly competitive	–	–	x / –	x / –	x	x	x
Drought tolerant	x	x	x	x	–	x	–
Salt tolerant	–	–	–	–	–	–	–
Reallocation of nutrients to roots	–	–	x	x	x	x	x
No major pests/diseases	–	–	x	x	x	x	x
Disperses readily from aboveground vegetative fragments	–	–	–	x	x	–	x
Prolific viable seed production	x	x	x	–	–	x	x

15.6.2 Opportunistic Harvest of Invasive Plants as Biofuel Feedstocks

Interestingly, given the overlap in traits between invasive plant species and biofuel feedstocks, some researchers have begun to explore the option of using bioenergy production as a control strategy for problematic invaders. Studies from Africa and Europe suggest that some woody and herbaceous invasive plant species (some of which also occur as invasives in North America) could be utilized as feedstocks without additional inputs and agronomic optimization (Van Meerbeek et al., 2015). One recent study of kudzu (*Pueraria montana*) suggested that in the southern United States, yield and carbohydrate content of invasive stands compares favorably to production from corn or sugarcane in terms of potential bioethanol yield per hectare (Sage et al., 2009). Similar analyses indicated that harvest of invasive reed canarygrass (*Phalaris arundinacea*) in Wisconsin could theoretically produce energy surpassing the state's current renewables and would offer additional benefits toward restoration of ecosystem services (Jakubowski et al., 2010). These proposals are part of growing interest in harvest incentives to control invasive species. Success would require addressing challenges common to all such programs, including potentially increased risks of intentional introduction and spread that such incentives might bring. Additional economic, environmental, and regulatory barriers make it unlikely that large-scale harvest of existing invasive biomass for energy production will happen in the near future. Such barriers include the cost of transportation to biorefineries, potential for accidental dispersal along transport routes, or prohibitions on sale and distribution of certain invasive plants species (Quinn et al., 2014).

15.6.3 Improving Weed Risk Assessment Tools

Desirable biofuel feedstocks possess many of the same traits as invasive and weedy plant species (Table 15.1) (Barney and Ditomaso, 2008; Raghu et al., 2006). Scientists have recommended that potential biomass feedstocks, including for biofuels, be carefully evaluated with WRA tools prior to introduction and commercialization (Endres, 2015; Bransby, 2008; Davis et al., 2008). WRAs can be economically beneficial as they can identify potentially costly invasive species before they are introduced to new regions (Keller et al., 2007). In the last decade, researchers have used WRA tools to evaluate dozens of candidate feedstocks and determined that many pose an invasive risk (e.g., Lieurance et al., 2018; Barney et al., 2015; Quinn et al., 2015; Gordon et al., 2011). In response to EPA inquiries about the invasiveness of potential feedstocks, USDA completed WRAs for three species and found that field pennycress (*T. arvense*) has a high risk potential (USDA, 2015b), while the other two (Ethiopian mustard [*B. carinata*] and physic nut [*J. curcas*]) have a moderate risk potential of becoming weedy (USDA, 2015a, 2014).

WRA tools have been shown to accurately identify major- and non-invader species (Koop et al., 2012; Gordon et al., 2008); however, their usefulness has been questioned by some (e.g., Hulme, 2012), because traditional, trait-based, qualitative tools do not consider how abiotic factors and community interactions affect invasive species risk (Smith et al., 2015; Hulme, 2012). Inclusion of these local factors is challenging because most WRAs are done at large geographic scales (e.g., state, country) that encompass a wide range of abiotic and biotic factors. A recent study concluded that broad WRA tools are not able to distinguish between beneficial crops and invasive agricultural species (Smith et al., 2015).

Although the findings of that study were strongly criticized by other scientists (Gordon et al., 2016), the study indicated that broad, trait-based WRAs may have their limitations, particularly when making decisions that could have significant economic impacts (Barney et al., 2016). This can create a lot of uncertainty surrounding predictions (West et al., 2017).

Some researchers have advocated that using tiered weed assessments is more appropriate for decisions about feedstocks (Flory et al., 2012; Davis et al., 2010; Cousens, 2008) and that information from one tier could be used to refine evaluations on other tiers (Barney, 2014). The first tier of such an approach would rely on trait-based qualitative tools discussed above. The second tier would evaluate more detailed information about the species' biology to determine what kinds of conditions (e.g., habitats, climates, inputs) the species needs in order to survive. Finally, in the third tier, quantitative studies would directly measure the ability of the species to establish, grow, reproduce, and spread in habitats and regions where it is proposed for production. Such studies could be carefully conducted under controlled conditions to ensure that no plants escape, similar to those already conducted for transgenic plants (Davis et al., 2010). Davis et al. (2011) demonstrated the value of a tiered approach for the species false flax (*Camelina sativa*), which was rated with a traditional WRA to be high risk. They measured the ability of *C. sativa* to grow and reproduce in two rangeland ecosystems in Montana under different scenarios and concluded that the species is unlikely to become invasive in those habitats. While potentially useful, third tier analyses would need to be conducted across multiple regions, habitats, and years to account for spatial and environmental variation in conditions (Hager et al., 2015; Smith et al., 2015; Flory et al., 2012).

Traditional WRA tools that are currently used to identify potentially invasive species do not necessarily consider the ways and likelihood that plants can escape from the biofuel production pathway (Barney, 2012; Barney and DiTomaso, 2010). For example, plant propagules may escape during planting, crop production, transport, or storage/processing at biomass facilities (Figure 15.1) (Lewis and Porter, 2014; IUCN, 2009). Also, major WRA tools do not consider how normal crop production practices or specific risk management measures can reduce the risk associated with cultivating potentially invasive species (Smith et al., 2015; Buddenhagen et al., 2009). Risk assessment approaches that incorporate risk management strategies are called risk analyses (IPPC, 2017) and may identify critical control points for management [i.e., Hazard analysis and critical control points (HACCP), see U.S. EPA (2013)]. However, despite the potential value of these types of analyses, no evidence was found that such tools are being used in the context of biofuel production.

Plant breeding is a fundamental process in crop improvement programs, including those for second-generation feedstocks (Mohapatra et al., 2019; Thakur et al., 2019; Kandel et al., 2018). Selection for desirable plant traits in biofuel feedstocks may increase their invasive potential [e.g., *Miscanthus* × *giganteus* (Matlaga and Davis, 2013)], not affect it [e.g., *Phalaris arundinacea* (Jakubowski et al., 2011)], or potentially decrease it [e.g., corn vs. its ancestor, teosinte (Vibrans and Flores, 1998)]. Thus, WRAs of potential feedstocks should be done at the level of plant cultivars (Sollenberger et al., 2014; Gómez Raboteaux and Anderson, 2011). Recently there has been some debate about whether qualitative WRAs can accurately assess feedstocks when there may be variation in specific traits among cultivars (Barney et al., 2016; Gordon et al., 2016; Smith et al., 2015). However, if the scope of the WRAs were limited to a specific cultivar and its associated traits, they can produce risk outcomes that differ among cultivars (Leon et al., 2015). For example, WRAs for three types of sorghum (*Sorghum bicolor*) resulted in different risk scores and outcomes: sweet sorghum (risk score = 3; accept); grain sorghum (risk score = 7, reject); and shattercane (risk score = 18, reject) (Gordon et al., 2011). Assessing specific cultivars will be very challenging if detailed descriptions of those cultivars are not available (Gordon et al., 2011). In these cases, it will be important that risk assessors and plant breeders work together to characterize the risk of specific cultivars.

15.7 Synthesis

15.7.1 Chapter Conclusions

- Direct impacts to date on the environment or agriculture from the cultivation of invasive or noxious plant species used as biofuel feedstocks have not been observed since most biofuel is produced from a small number of non-invasive feedstock species (corn and soybean).
- Impacts from the cultivation of corn and soybeans on the evolution of herbicide-resistant weeds do exist, although it is unclear to what extent impacts can be attributed to corn and soybeans grown to meet either biofuel demand generally or the specific requirements of the RFS Program. Since the RFS was enacted, herbicide-resistant weeds have increased production costs for farmers in terms of herbicide expenditures and in their overall investment in technology and production systems. However, this temporal association alone is not sufficient to determine causation. In the RtC2, incidence and impacts of herbicide-resistant weeds were largely attributed to cultivation of herbicide-resistant crop varieties. Literature reviewed in this chapter suggests that additional biological, anthropogenic, and environmental factors determine the existence and extent of impacts from these weeds.
- The likely future effects of the RFS Program from invasive or noxious feedstocks are uncertain due to many factors. However, if biofuels continue to be produced from the feedstock sources identified in Chapter 2 (section 2.3.2), there will be no likely future effects directly from potential invasive or noxious feedstocks because these feedstocks are not invasive.

- Two potentially invasive feedstocks are part of approved biofuel pathways under the RFS Program and could produce effects *if* they are grown and *if* additional registration, reporting, and recordkeeping (RRR) requirements that are in place and designed to limit their spread are not sufficient to prevent escape and invasion. An additional five feedstock types were identified in consultation with USDA through a weed risk assessment process as having some invasive potential that could necessitate mitigation. As of the publication of this report, no RINs have been generated that involve any of the noted feedstocks nor have incipient invasions or impacts been observed as a result of their production for biofuel.
- Likely future effects from herbicide-resistant weeds will continue to grow, if current trends in the incidence of new cases and number of weed species that are resistant to multiple herbicide sites of action continue. As with impacts to date, future impacts from the cultivation of corn and soybeans on the evolution of herbicide-resistant weeds are likely to occur, but it will be challenging to determine what extent of impacts can be attributed to corn and soybeans grown to meet biofuel demand generally, let alone the specific requirements of the RFS Program. Adoption of additional field-scale and regional weed management approaches will likely be necessary to avoid the most severe impacts from these weeds, although complete avoidance will be impossible.
- It is not possible to reach a firm conclusion regarding the relative overall invasion risk posed by biofuels compared to petroleum. Risks of invasion associated with petroleum exploration and extraction include both the introduction of non-native species via hitchhiking on machinery and infrastructure and the facilitation of non-native dominance through habitat disturbance. Furthermore, risks posed by the petroleum industry do clearly impact a broader range of habitats than those posed by biofuel generation, as they also extend to marine and estuarine ecosystems. Nevertheless, direct comparison of the two industries is extremely difficult as the full extent of actual impacts at a national scale remains unknown.

15.7.2 Conclusions Compared with the RTC2

Conclusions from this report are similar, but not identical, to conclusions from the last report. As noted above, more information and analyses have been published recently that reveal incidence of herbicide-resistant weeds are not exclusively the result of corn or soy production systems that rely on herbicide-resistant varieties.

15.7.3 Scientific Uncertainties and Next Steps for Research

Based on the available evidence, there is reasonable confidence that biofuel production from corn and soy feedstocks in the United States to date has not directly resulted in the escape or spread of invasive plants. However, this chapter has uncovered several important uncertainties that should be examined further.

- There is no evidence in the literature of any biofuel feedstocks from pathways approved by EPA under the RFS Program or feedstocks currently under development escaping from production sites and causing impacts. This could reflect a lack of data or lack of effect.
- There is uncertainty in the ability to accurately assess risks from feedstocks under development or consideration and those feedstocks that have been improved through either traditional breeding or genetic engineering.
- While it is clear that herbicide-resistant weeds can significantly reduce crop yields and increase production costs, the extent to which the evolution of herbicide-resistant weeds is attributable to biofuel production or the RFS Program is unknown.

15.7.4 Research Recommendations

- Research should focus on monitoring and data collection. To date, knowledge of the frequency of escapes from fields where new feedstocks are cultivated (that is, feedstocks that are not corn or soybean) is restricted to a small number of case studies. This may indicate that escapes truly are rare and of limited impact, but it also may reflect inadequate surveillance or reporting at broad scales. Without more thorough data collection, it will be difficult to estimate impacts at a national scale and to assess the accuracy of risk assessments to predict the likelihood of invasiveness among future feedstocks.
- Research should also focus on developing weed risk assessment tools specifically relevant for identifying potential invaders in biofuel feedstock production and logistics contexts. New weed risk assessment tools are needed because the application of robust tools and reliance on them to inform decisions about which feedstocks can be grown with minimal risk of invasion will be key to avoiding future impacts.
- In addition, research in the form of causal analysis could be used to understand whether and to what extent biofuel feedstock cultivation has contributed to the evolution of herbicide-resistant weeds. This could include determining the spatial co-incidence of corn and soybean acreage, years, and amounts attributable to biofuels broadly and any amounts attributable to the RFS Program with observations of new herbicide-resistant weeds.

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16. International Effects

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Key Findings

- Attribution of international effects from the RFS Program remains challenging due to complex interrelationships among other major drivers of observed change. There are relatively few studies on this topic specifically, though many on international effects from biofuels more generally, and analyses are impeded by inconsistent data and large uncertainties.
- This chapter considers three causal chains whereby the RFS Program may lead to environmental effects in foreign countries. First, it is considered that the RFS Program may increase biofuel imports from other countries to the United States, and that such imports may be associated with environmental effects related to producing the imported biofuels. Second, the market-mediated effects of the RFS Program on international crop and commodity prices and production, and related environmental effects are considered. Third, it is considered that, by supporting the growth of the U.S. biofuel industry, the RFS Program may increase biofuel exports from the United States to other countries, with potential environmental effects/ benefits for the receiving countries.
- International environmental effects that are clearly attributable to the RFS Program due to U.S. ethanol and biodiesel imports could not be confidently quantified. The lack of empirical evidence to support causal linkages between the RFS and international environmental effects does not necessarily rule out international effects attributable to the RFS Program.
- Imports of biofuels to the U.S.—one of the mechanisms for international effects—have fallen drastically since peaking before the RFS Program in 2004–2006. Evidence supports attribution to the RFS Program for some biodiesel imports since 2007. The value of advanced biofuel (D5) RINs was among many factors that supported sugarcane ethanol imports from Brazil since mid-2010.
- The hypothesis that U.S. demand for sugarcane ethanol attributable to the RFS Program played the dominant role in the observed changes in Brazil's ethanol production and associated environmental effects is not supported by available evidence. Ethanol production in Brazil has been supported by domestic policies in Brazil for decades.
- U.S. ethanol production that exceeds domestic demand is exported to more than 70 nations around the globe, and since 2008, the United States has been a net exporter of biofuel (ethanol + biodiesel) on an annual basis. However, the share of exports attributable to the RFS Program is uncertain. To the degree that the RFS Program encouraged investments that generated surplus ethanol for export, the RFS Program contributed to the international effects

associated with net U.S. exports, which could be environmentally beneficial for importing nations, depending on the feedstocks used for domestic biofuel production.

- Two-way interannual ethanol trade with Brazil (i.e., U.S. corn ethanol and Brazilian sugarcane ethanol) is likely driven by extreme weather events, seasonal differences in the agricultural cycles, pricing dynamics between hydrous and anhydrous ethanol, and regulatory standards (for example, the CA-LCFS could encourage imports of ethanol with lower carbon intensities and exports of ethanol with higher carbon intensities ratings), between the two nations. Brazil's own policy changes to produce and use ethanol (rather than supplying to the U.S. market) are having more of an effect on the increase in production of ethanol in Brazil, such as proposed increase in Brazilian ethanol blend levels in gasoline as well as their own low carbon fuel policy, RenovaBio. Additionally, Brazil's use of a Tariff Rate Quota or a high import tariff has led to a steep decline to de minimis exports of U.S. ethanol to Brazil. However, the relative contributions of these causal factors to the extent of this two-way trade have not been resolved in the scientific literature.
- A portion of the gross biodiesel imports during 2012–2019, averaging approximately 295 million gallons per year, are reasonably attributed at least in part to the RFS Program. However, many factors impact biodiesel trade, and sources of import (i.e., countries) are diverse and irregular, each affected by their own domestic policies which are difficult to assess with current models and data.
- While this chapter concludes that more research is needed on the effect of the RFS Program on international cropland areas and environmental effects more broadly, as well as accounting for foreign countries' own biofuels policies, combining published simulation modeling estimates of the land use change effects of biofuels (section 16.3.3 for corn ethanol) and the estimates for the effect of the RFS Program on corn ethanol (Chapter 6) yields an illustrative range of the effect of the RFS Program on non-U.S. cropland area of 0 to 1.6 million acres. The estimated effect of the RFS Program does not yet include effects on soy biodiesel. As more data become available and are analyzed, historical relationships among U.S. biofuel policies, production, trade, environmental indicators, and other variables may be clarified and uncertainties reduced. Review of potential international effects of the RFS Program associated with biodiesel imports, and on global cropland more broadly, finds that quantification of effects is uncertain but could be significant and merits further research. The relationship of the RFS Program with palm oil expansion, and the environmental costs and benefits of two-way trade, merit further study.

Chapter Terms (see Glossary): advanced biofuel, backfill, bagasse, palm oil, peat soil

16.1 Overview

16.1.1 Background

In the period from 2004 to 2008, several published studies examined the effects of EISA[1] and highlighted the potential for the RFS2 mandates to be increasingly met via imports (Earley, 2009; Kline et al., 2008; Westhoff et al., 2008; Yacobucci, 2008; Wainio et al., 2005).[2] Indeed, gross biofuel imports led by ethanol increased in the 2004–2006 period (Figure 16.1), setting new records for imported renewable fuel volumes that would not be matched until biodiesel imports increased in 2013. Thus, concerns were likely high about the potential international effects from the RFS Program during the drafting of EISA. However, after the 2004–2006 period, when net biofuel imports (ethanol plus biodiesel) represented 10–20% of total U.S. biofuel production, imports declined to where total biofuel (ethanol plus biodiesel) exports exceeded total imports (Figure 16.2). Note that negative values in Figure 16.2 mean that the United States is a net biofuel exporter in terms of gallons (line). The peak net imports of 700 million gallons in 2006 represented nearly 18% of U.S. production in 2006, whereas the peak net exports of 1.6 billion gallons represented about 9% of total U.S. production in 2018. Thus, the United States has been a net exporter of total biofuels every year since 2008. Net exports of biofuels represent, on average, 4% of U.S. annual production during 2007–2019, driven by increasing volumes of U.S. ethanol exports since 2010. Details for annual trade in ethanol, biodiesel, and renewable diesel are presented in Figure 16.3. The observation of net exports does not mean the U.S. demand does not have an effect overseas, but it does mean that the market mediated effects overseas may be at least partly offset by these exports (discussed below).

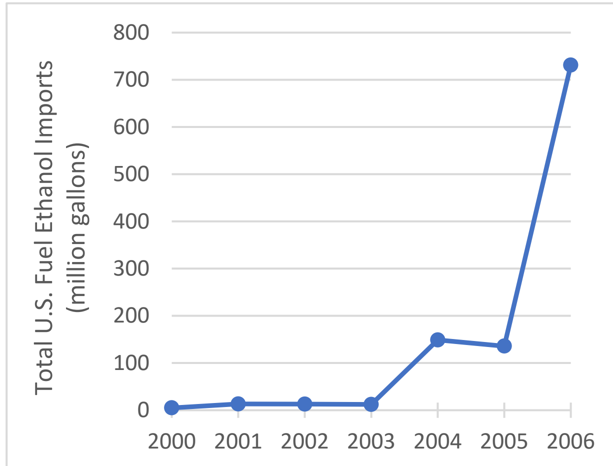


Figure 16.1. Total U.S. fuel ethanol imports, 2000-2006.
Source: EIA (2022).

This chapter relies on U.S. Energy Information Administration (EIA) for data on monthly and annual biofuel production and trade (imports and exports of biofuel) where available. In cases where EIA data are not available, the U.S. Department of Agriculture Foreign Agricultural Service (USDA FAS) or

[1] Energy Independence and Security Act of 2007, Pub. L. No. 110-140, 121 Stat. 1492, preamble (2007).
[2] While some papers were published after implementation of EISA, they rely on historical data from the 1990s–2006, a period when imports to the United States were strong and appeared to be poised to increase in response to RFS Program biofuel mandates.

Economic Research Service (USDA-ERS) reports are the primary source of information. The available data have significant limitations. For example, only U.S. export data from 2010 onward have been compiled by EIA, requiring a combination of data from other sources such as USDA-ERS or U.S. Department of Commerce to characterize the full period. There are discrepancies among data sources in reported annual import and export volumes between 2004 and 2009. The data from EIA show the total biofuel imports (ethanol + biodiesel) as negative in 2002–2003, and then rising sharply with a peak in 2006 and net imports each year 2004–2007. Data from USDA Bioenergy Statistics[3] show a similar pattern occurring two years earlier, with net imports each year 2002–2006, and the peak in net imports in 2004. It is not the purview of this report to update this historical record. Data sources are compatible from 2010 onward. However, calculations of net imports prior to 2010 involve increased uncertainties as they must rely on two distinct data sources and different uses of ethanol (by industry or in transportation). The differences are small for years aside from 2007 and the general trends are consistent. Thus, any differences in the data between these sources do not materially impact the conclusions of this report.

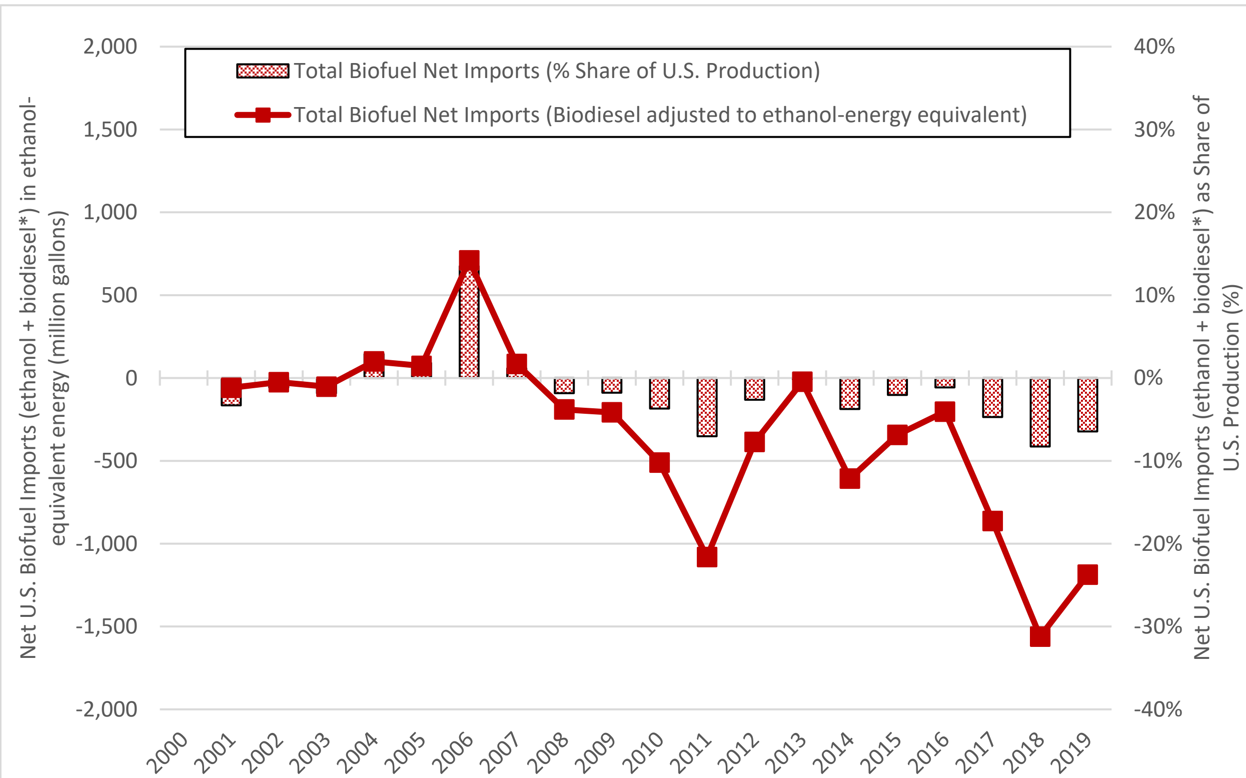


Figure 16.2. Total biofuel (ethanol + biodiesel) net imports (imports – exports) to the United States (red line, left axis), and total biofuel net imports to the United States as a share of total U.S. biofuel production each year (red bars, right axis).[4]

[3] https://www.ers.usda.gov/data-products/us-bioenergy-statistics/

[4] Biodiesel import and export volumes were adjusted to have energy output values equivalent to the same volume of ethanol. The data sources are EIA for all imports and for exports after 2010, and USDA for exports prior to 2010.

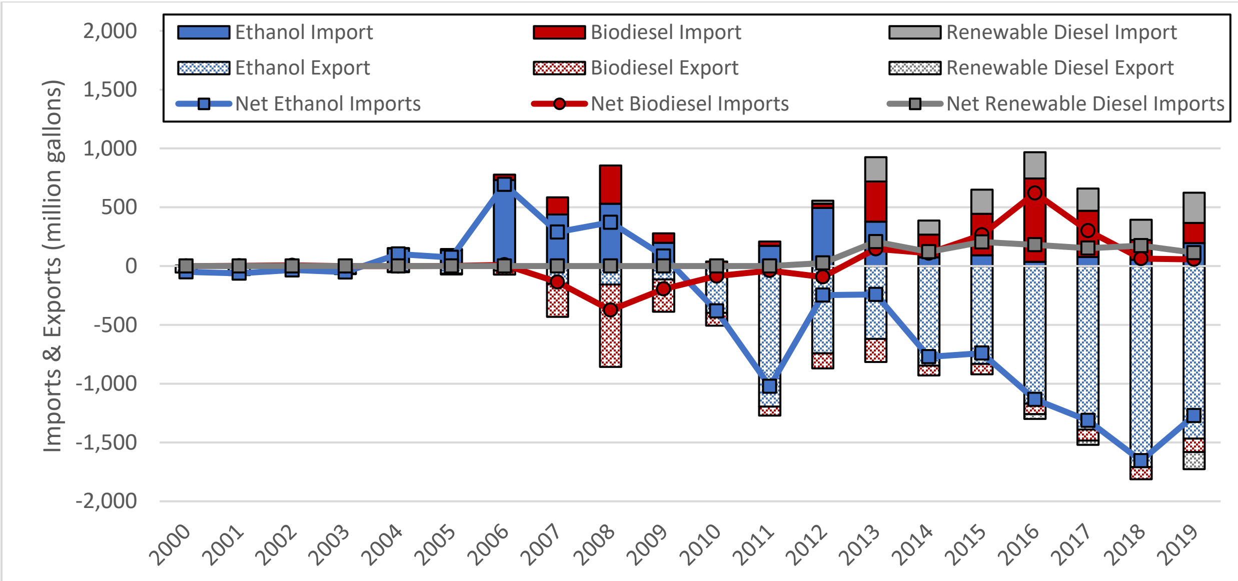


Figure 16.3. Total ethanol, biodiesel, and renewable diesel imports and exports by year from all sources. Same data sources as Figure 16.2, but disaggregated.

The sharp rise in imports during 2004–2006 was predominantly ethanol (Figures 16.1, 16.3), and raised awareness about potential international impacts of the RFS Program, particularly if import trends were to persist. Therefore, potential environmental impacts associated with imports and production in other nations were concerns that were incorporated in EISA (2007) Section 204, which states: "The report shall include the annual volume of imported renewable fuels and feedstocks for renewable fuels, and the environmental impacts outside of the United States of producing such fuels and feedstocks" (see Chapter 1). As discussed in Chapter 2, the placement of the text on international effects after the text on environmental and resource conservation effects implies that the discussion of international effects may be more general in nature (Chapter 2, section 2.4).

Peer-reviewed research articles around that time based on projections and model simulations of demand driven by the RFS Program, crude oil price increases, or other scenarios (e.g., Searchinger et al., 2008) reinforced and highlighted concerns about potential impacts of increasing U.S. biofuel demand on tropical deforestation. This chapter reviews available evidence of linkages between the RFS Program and biofuel imports and exports, and associated international environmental effects. The data show that in the years following passage of EISA, trends in biofuel production and trade evolved differently than some models predicted. For example, exports of commodities including soybeans remained strong, and the United States became a net exporter of biofuels (not importer), driven primarily by ethanol exports in recent years due to increased production capacity and ethanol blending wall.

After 2007, factors impacting terms of trade for biofuels evolved and led to an unexpected role for the United States to emerge as the world's leading ethanol exporter in every year since 2010, except for two years that were impacted by severe drought (2012–2013) (Beckman and Nigatu, 2017). Additionally, since 2013, ethanol exports more than offset the volumes of biodiesel imports (Figure 16.3), resulting in net biofuel exports from the United States when biofuels are considered in total. Imports and exports of biodiesel and renewable diesel also evolved over time due to interactions among domestic and foreign policies. The factors associated with trade in biodiesel and renewable diesel are distinct from those driving trade in fuel ethanol. Because policies, feedstocks, supply chains, markets, and international effects associated with ethanol are distinct from those for biodiesel, the two biofuels are discussed separately.

16.1.2 Drivers of Change

Consistent with the approach to assessing attribution in Chapters 6 and 7, this chapter examines various types of evidence to identify potential international effects of the RFS Program and of U.S. biofuel production in general. The chain of events for international environmental effects are more complex than domestic effects, and it is helpful to explicitly articulate hypothesized mechanisms and relationships (see Box 16.1: Potential Relationships Between the RFS Program and International Effects). Many of these steps and processes are captured in the modeling efforts, but the numerous underlying assumptions and parameters are often not transparently communicated in these studies (Daioglou et al., 2020).

Box 16.1. Potential Relationships Between the RFS Program and International Effects.

> "**→**" signifies "leads to" as a directional link
>
> "Increase," "change," or "decrease" are relative to a counterfactual scenario absent the RFS Program.

1.1. **Effects of biofuel imports to the U.S.:**
RFS Program increases domestic biofuel demand → Increase in U.S. biofuel imports → Increase in biofuel exports from foreign country X to the U.S. → Increased biofuel production in foreign country X to support exports to U.S. → Environmental impacts associated with increased biofuel production in, and exports from, foreign country X. This theoretical chain represents the most direct of the hypothesized international effects of the RFS Program and is the main focus of this chapter.

1.2. **Effects of biofuel exports from the U.S.** (the reverse of 1.1)**:**
RFS Program → Increase U.S. biofuel production → Increase U.S. biofuel exports to foreign country Y → Avoided environmental impacts due to reduced biofuel production or fossil fuel consumption in country.

1.3. **Potential market-mediated effects of RFS Program and U.S. biofuel production:**
Involves several distinct chains and assumed relationships. For example: If the RFS Program causes the price (or U.S. exports) of a global commodity (corn or soybeans) to change → adjustments in foreign production of that commodity and global trade patterns → environmental effects associated with where and how changes in global production occur. Potential market-mediated effects are typically estimated with models (See section 16.4 Horizon Scanning).

16.1.3 Relationship with Other Chapters

This chapter focuses on the potential effects of the RFS Program on imported biofuels, as specified in EISA Section 204, and the associated effects overseas from the production of those biofuels. However, the growth of the biofuels industry in the United States may influence both imports and exports, with attendant net effects overseas due to the two-way nature of trade. Thus, this chapter examines both imports and exports from the United States. EISA Section 204 does not call out the effects from elevated demand in the United States on world markets, and the potential cascading effects from this elevated demand. However, these effects are critically important in order to understand the potential international effects of the RFS Program, and these are emphasized in the scientific literature. Thus, because the RFS Program may also have environmental effects through market-mediated impacts on global cropland area and other factors, these potential effects are also discussed in this chapter. Whereas the other chapters focused on domestic effects that were separated into attribution (Chapters 6 and 7) and environmental and resource conservation effects (Chapters 8–15), this chapter is an overview of those topics, primarily as it pertains to Brazilian sugarcane ethanol (i.e., the one imported biofuel that is a focus of the RtC3), although other biofuels and countries are also discussed (see sections 16.4 and 16.5). Thus, there is not a parallel objective with the other chapters, and as such the chapter follows a slightly different organizational structure to facilitate the communication of findings.

16.1.4 Roadmap for the Chapter

This chapter examines potential environmental effects in foreign countries that could be attributed to the RFS Program in the United States. In doing so, the chapter examines U.S. biofuel trade in general for context. As with Chapters 6 and 7, it is useful to separate the review of biofuels from the review of the feedstocks and the land. The chapter starts with imported ethanol from Brazil as the primary source for U.S. biofuel imports (section 16.3), and the only international source discussed in Chapter 2.[5] The "horizon scanning" section (section 16.4) examines other biofuel imports that were either short-lived historically (e.g., Argentinian biodiesel from soybean) or remained minor by comparison with other biofuels but that came from regions with potentially sensitive ecosystems (e.g., Southeast Asia). Section 16.5 discusses environmental effects associated with palm oil production in Southeast Asia, and potential linkages with the RFS Program. Although the focus of the chapter is on imports due to the RFS Program (as specified in EISA Section 204), the broader market-mediated effects on countries outside the

[5]While this chapter focuses on Brazilian sugarcane ethanol, Brazil has recently begun producing ethanol from corn, with volumes reaching 900 million gallons in 2021. For more information, see section 16.3.2.2.

United States from increased U.S. demand for biofuels are also reviewed. The chapter concludes with a synthesis, review of uncertainties, limitations, and recommendations.[6]

16.2 Conclusions from the RtC2

- Since the 2011 Report, U.S. ethanol imports decreased, while biodiesel and renewable diesel imports increased, leading to potential land use change impacts in countries of origin. Exports of corn, distillers dried grains and solubles (DDGS), soybeans, and ethanol primarily increased or are similar in comparison with 2007 levels.
- Reports suggest that demands for biofuel feedstocks have led to market-mediated land use impacts (both direct and indirect land use changes) in the past decade.
- Cropland expansion and natural habitat loss (including forests) have been observed internationally, and it is likely that increased biofuel production has contributed to these land use changes.
- Quantification and causal attribution of land use change and international environmental impacts due to biofuel production remain uncertain and undetermined.
- Global cropland area has expanded since the year 2000, coinciding with the increase in U.S. biofuel production. During this period, the ratio of area harvested to arable land increased and crop yields increased significantly, due in large part to gains in total factor productivity.
- Agricultural extensification and deforestation have been documented in countries that are major exporters of biofuels to the United States, including Brazil, Argentina, and Indonesia.
- Cropland expansion and natural habitat loss (including forests) have been observed internationally during the implementation of the RFS Program. It is likely that increased biofuel production has contributed to these land use changes, but significant uncertainty remains about the amount and type of land use changes that can be quantitatively attributed to U.S. biofuel consumption.

[6] There is no likely future effects section in this chapter because there were no likely future international effects identified in the Final Set Rule (see docket # EPA-HQ-OAR-2021-0427). Modeling of international effects are discussed extensively in the Model Comparison Exercise (MCE), but since that exercise did not produce estimates of international effects attributable to the RFS Program, the likely future effects internationally are not yet estimated.

16.3 Ethanol Trade and Effects

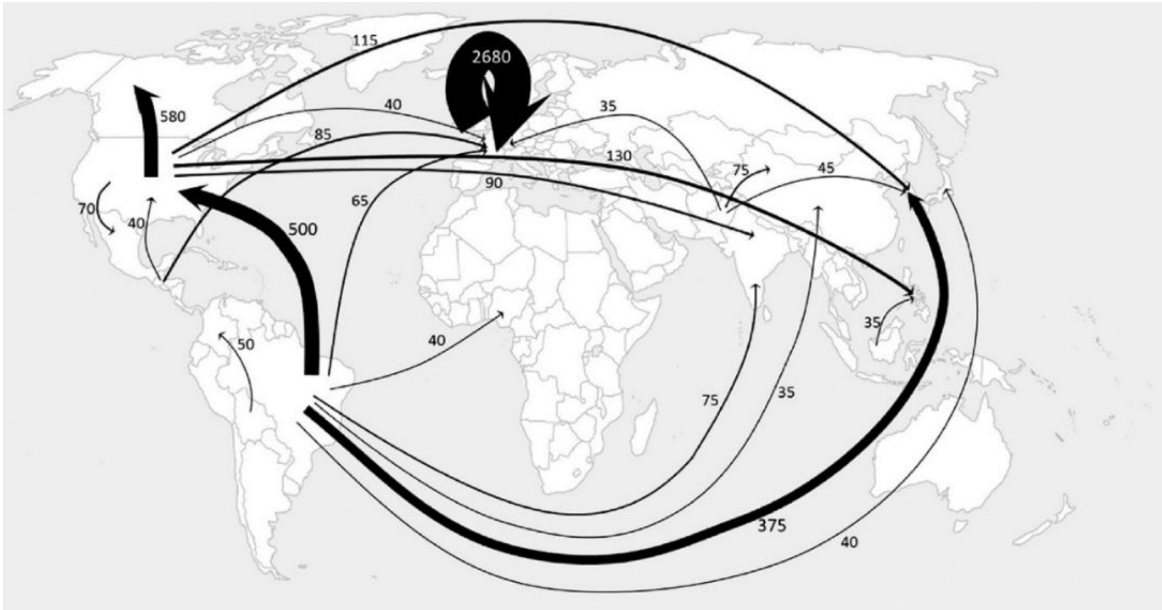
16.3.1 International Ethanol Markets

Global demand for ethanol varies by country, season, and year, depending on numerous factors (Figure 16.4). For example, as discussed in Chapter 2, the movement of ethanol from Brazil through Central American and Caribbean countries in the early 2000s, declined from 2006 to 2009, when this trade route closed due to the adoption of the Central American Free Trade Agreement, which included strict rules-of-origin to prevent transshipment of ethanol from other countries (EOP, 2004). Biofuel exports from the United States to Europe were intermittent but declined since 2012 because of increased EU duties and tariffs on U.S. imports (Figure 16.4) (USDA FAS, 2020b). Overall, as nations developed alternative fuel markets, trade in ethanol more than doubled from 2004 to 2012 (Proskurina et al., 2019a; Proskurina et al., 2019b). Since feedstock prices represent more than half of all costs to produce ethanol, feedstock availability and price are key factors determining supply (Shapouri and Gallagher, 2005). Global ethanol production increased dramatically starting around 2000 (Figure 16.5). The United States and Brazil dominate global ethanol production with a combined production that represented 83% of global supply in 2019 (IEA, 2020). While Brazil and the United States are the two primary net exporters of fuel ethanol, active trade of biofuels is observed all over the world, including in European markets in response to the Renewable Energy Directive (Figure 16.4).[7]

The subsequent sections explore ethanol trade between the United States and Brazil and the extent that this trade and associated environmental effects may be attributable to the RFS Program. Specifically, section 16.3.2 examines the influence of the RFS Program on U.S. ethanol imports, and the drivers of Brazilian ethanol and sugarcane production. Section 16.3.3 discusses potential international environmental effects associated with the RFS Program and the U.S. ethanol market more broadly.

[7] The original Renewable Energy Directive (RED) set a requirement that 20% of the EU's total energy needs will come from renewable energy sources by 2020. The recast RED2, which went into effect in December 2018, requires that at least 32% of total energy needs will from renewable sources by 2030 (https://ec.europa.eu/energy/topics/renewable-energy/renewable-energy-directive/overview_en).

a



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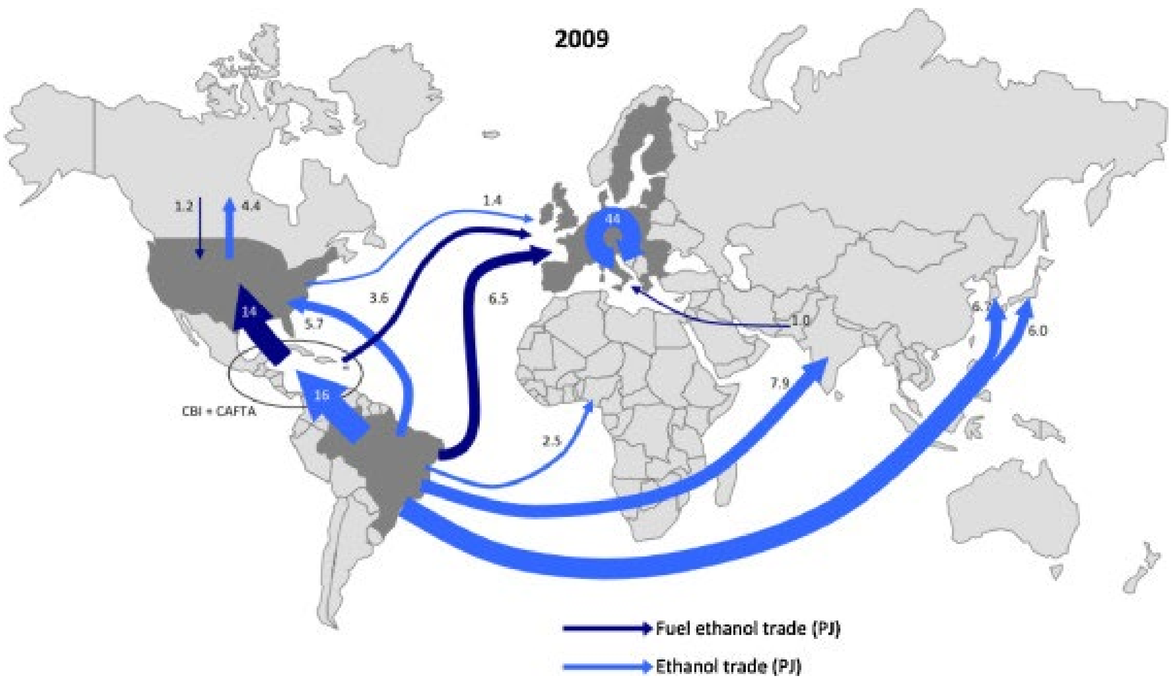


Figure 16.4. Major net total ethanol (industrial and fuel) trade streams (≥35 thousand tons) used for all end uses in 2015 (a) (Proskurina et al., 2019a; Proskurina et al., 2019b) (used with permission), **Ethanol and fuel ethanol trade (in petajoules) in 2009 (b) (Lamers et al., 2011)** (used with permission).

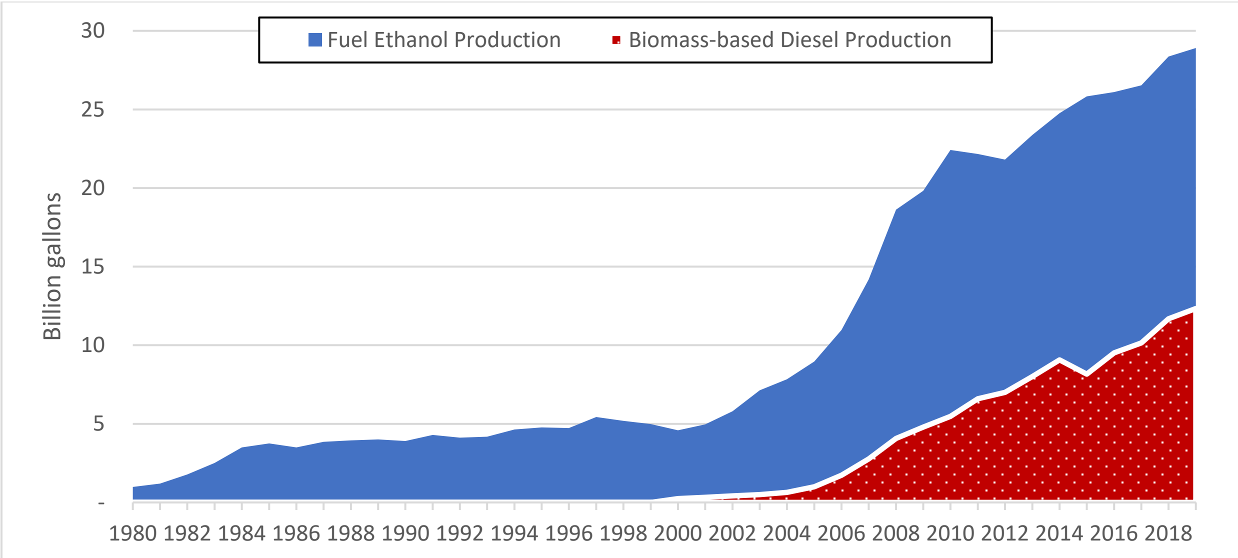


Figure 16.5. Global biofuel production.[8]

16.3.2 Factors Influencing Ethanol Imports to the United States

Total annual ethanol imports to the United States are presented in Figure 16.3. Factors influencing U.S. ethanol import volumes and their proportion of total domestic utilization varied over time. Imports prior to 2010 reflect factors affecting corn ethanol markets that are discussed in some detail in Chapters 4 and 6, such as high oil prices, the need to replace MTBE with ethanol, tax credits and incentives at state and federal levels, including RFS1, tariffs and relative economic advantages (i.e., tariff-free) of imports processed in the Caribbean Basin. Imports responding to these drivers were significant in terms of volume and market share in 2004–2006 (Figures 16.2 and 16.3) but declined rapidly in subsequent years. The United States has been a net exporter of ethanol each year from 2010 to the present, with net exports generally increasing over time (Figure 16.3). During this period, observed variability in gross ethanol imports are associated with distinct factors. First, imports in 2012–2013 are attributed primarily to the significant drought that impacted U.S. agriculture from 2011 to 2013 (Rippey, 2015). Tariffs are important factors for trade, but the net effects were small when U.S. ethanol tax credits and the countervailing 54 cent per gallon special duty on imports ended simultaneously (Jan. 2012). Import tariffs of 2.5% for undenatured and 1.9% for denatured remained in effect and have been estimated to have minimal influence relative to many other factors (NRC, 2013; Devadoss and Kuffel, 2010; Tyner, 2008). Incentives under the California Low-Carbon Fuel Standard (CA-LCFS) are a factor influencing the relatively small gross imports of sugarcane ethanol to the United States observed since 2014. Additionally, as discussed below, spot market opportunities drive seasonal ethanol trade with Brazil. Finally, a portion of U.S. imports from Brazil, which generally arrive in Gulf Coast ports, are

[8] Data compiled by EIA (https://www.eia.gov/international/data/world). Accessed January 28, 2021.

typically processed in the United States and reshipped overseas, such as ethyl tert-butyl ether (ETBE) for Japan (USDA FAS, 2019c). Due to data limitations, such transshipments and emerging ETBE markets are identified as topics for future analysis.

Ethanol imports from Brazil were the only international source that dominated the U.S. pool for 2005–2020 (Chapter 2, Tables 2.1 and 2.2). As shown in Figures 16.6 and 16.7, ethanol sourced from Brazil, including transshipments through Central America under the Caribbean Basin Initiative (CBI[9]), account for about 97% of total ethanol volume imported by the United States from 2007 to 2019. And since 2015, nearly 100% of U.S. imports are sourced directly from Brazil (Figures 16.6 and 16.7). Most of these are destined for the U.S. West Coast (Figure 16.8). Due to this port of entry, it is assumed that ethanol imports since 2015 are primarily in response to incentives under the CA-LCFS, rather than or in addition to such incentives from the RFS Program. One of the likely mechanisms whereby the CA-LCFS could support two-way ethanol trade is due to the way the program incentivizes fuels with low carbon intensities. This system could lead to U.S. exporting corn ethanol while simultaneously importing sugarcane ethanol, which has a lower carbon intensity. While this potential "fuel shuffling" has been discussed in articles concerning interactions between the RFS Program and the CA-LCFS (i.e., Whistance et al. (2017), Whistance and Thompson (2019), Yeh et al. (2016)), this report does not include a thorough assessment of the extent of this phenomenon, its causes, or its impacts.

Ethanol imports from Brazil have been irregular (Figure 16.6) but generally is a small fraction of total U.S. production and total Brazilian production (Figure 16.9). In 10 of the past 15 years (2005–2019), gross ethanol imports from Brazil represented 1% or less of total U.S. biofuel pool (i.e., production plus imports, Table 2.2) and less than 2% of total ethanol produced by the United States (Figure 16.9). However, imports from Brazil represented larger shares of U.S. ethanol production before the RFS Program and in the early years of the Program, for example, 2–3% in 2004, 9% in 2006, 2–3% in 2007–2008, and 2–3% in 2012–2013 (Figure 16.9). The large increase in 2004–2006 was likely due to the phaseout of MTBE in RFG areas outside of California in the summer of 2006, which increased U.S. ethanol demand faster than domestic production (see Chapter 6). These trends continued to a lesser extent in 2007 and 2008 as domestic ethanol production grew and eventually surpassed demand due to the E10 blend wall and other factors (see Chapter 6). The increase in 2012–2013 was likely due to the aforementioned drought in the Midwest (Rippey, 2015). However, aside from 2006, ethanol imports were small relative to U.S. production. Gross ethanol imports from Brazil also generally represent a small fraction of Brazilian production, with most years less than 3% (Figure 16.9). This is consistent with the

[9] Membership in the CBI has changed through time. There are currently 17 countries included: Antigua and Barbuda, Aruba, Bahamas, Barbados, Belize, British Virgin Islands, Curacao, Dominica, Grenada, Guyana, Haiti, Jamaica, Montserrat, St. Kitts and Nevis, St. Lucia, St. Vincent and the Grenadines, Trinidad and Tobago (https://ustr.gov/issue-areas/trade-development/preference-programs/caribbean-basin-initiative-cbi).

diverse trade partners of Brazil (Figure 16.4) and with the strong demand for ethanol domestically in Brazil (see section 16.3.2.2).

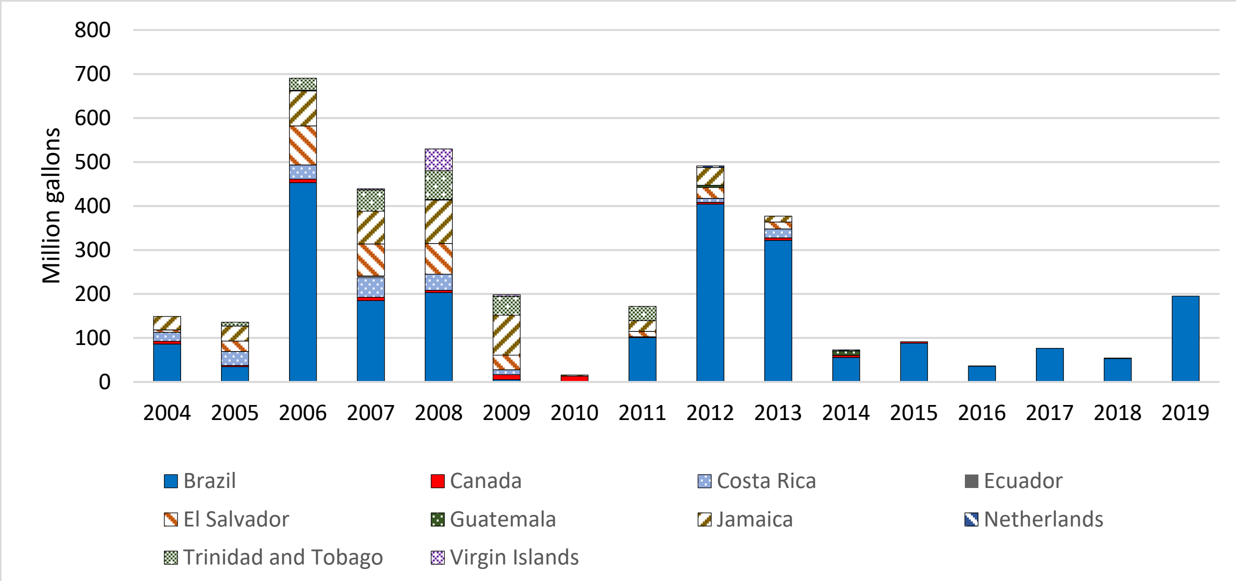


Figure 16.6. U.S. gross fuel ethanol imports from 10 leading countries (99.6% of total volume from all countries. Countries that likely transshipped Brazilian ethanol (see Chapter 2) to the United States under incentives provided by the Caribbean Basin Initiative are shown with patterned fills[10]; non-CBI exporters to the United States include Brazil, Canada, Ecuador, and Netherlands as illustrated, plus smaller volumes from over 50 other nations (EIA, 2022). Note data by country of origin begin in 2004.

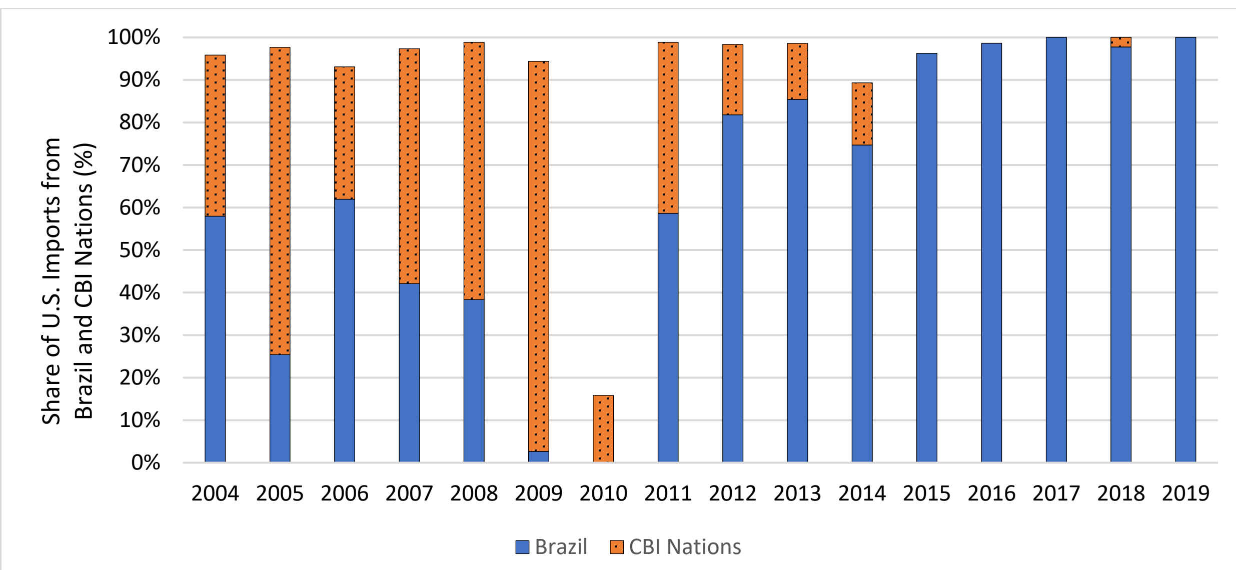


Figure 16.7. Share of total annual ethanol imports to the United States sourced from Brazil (blue, solid) and totals from CBI nations (orange with black dots) by year (EIA, 2022). Imports from CBI nations would increase shares from Brazil in some years (see Figure 16.6).

[10] These countries were members of the CBI during the year of trade even though they may not continue to be members.

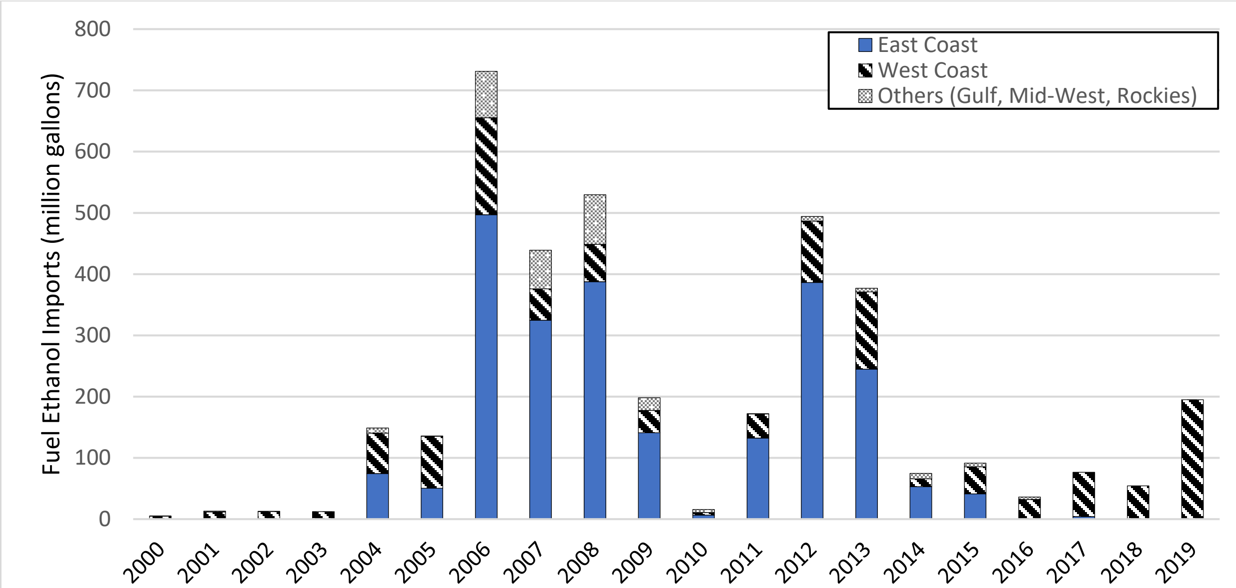


Figure 16.8. U.S. total fuel ethanol imports from all sources, by port of entry (annual, 2000–2019) (EIA, 2022). Virtually all imports from 2000 to 2003 and 2016 to present went to the West Coast.

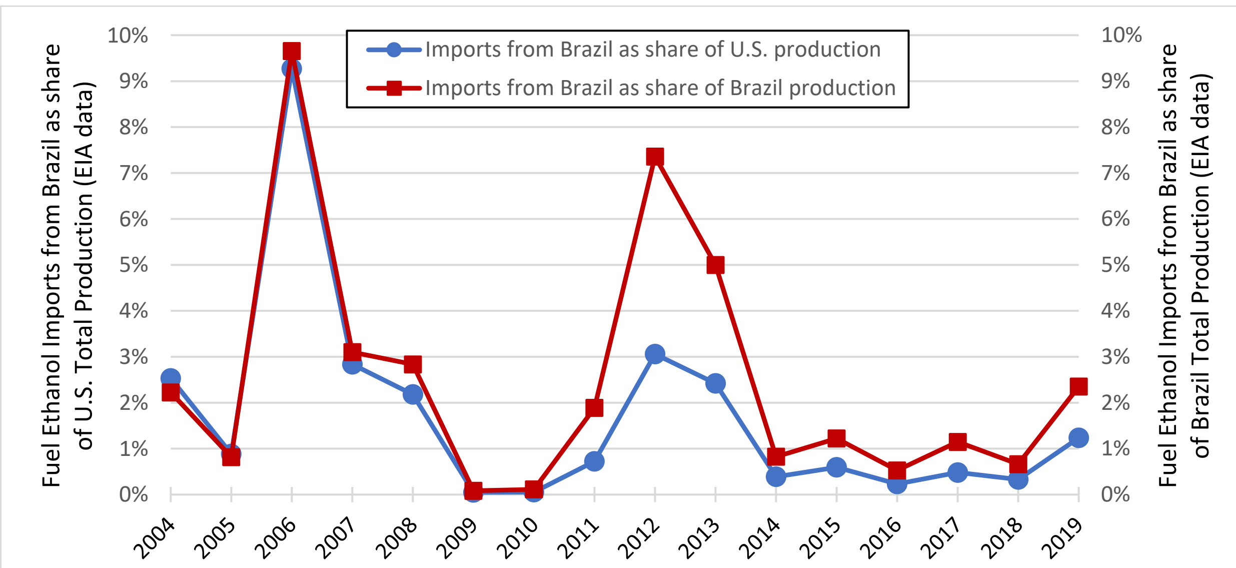


Figure 16.9. Fuel ethanol annual imports from Brazil as share of U.S. and Brazil production (EIA, 2022). Imports peak in 2006 (see Chapter 6 and discussion of MTBE replacement) and 2012–2013 (U.S. drought).

16.3.2.1. Advanced Biofuel D5 RIN Market as Driver of Imports from Brazil

One hypothesis on how the RFS Program drives exports from Brazil to the United States relates to Brazil's sugarcane ethanol, which qualifies as an advanced biofuel under the RFS Program and may generate higher-priced advanced biofuel RINs (D5, see Chapter 1, Figure 1.2). Under this hypothesis, a price premium for D5 RINs could lead to increased sugarcane ethanol imports to the United States. The higher price could create incentives for imports of sugarcane ethanol independent of the broader conventional ethanol fuel markets examined in Chapter 6.

The hypothetical sequence of events connecting environmental effects in Brazil to the RFS Program includes (a) RFS2 creates a demand for advanced biofuels through the advanced biofuel mandate, (b) Advanced biofuels include sugarcane ethanol, which can be registered as one of several "other advanced biofuels" with D5 RINs → (c) Increased demand for advanced biofuel D5 RINs causes an increase in imports of sugarcane ethanol from Brazil to the United States → (d) Increase in ethanol production in Brazil to support exports → (e) Environmental impacts associated with increased ethanol production in, and exports from, Brazil. Any break in this chain of events may influence the estimated environmental effects in Brazil attributable to the RFS Program.

As shown in Chapters 4 and 7, the D5 RIN prices are above transaction costs beginning in year 2010, the first year of the full RFS2 and of the advanced mandate (Chapter 4, section 4.1; Chapter 7, section 7.2.2).[11] This suggests that the RFS2 advanced mandate was binding in these years. Thus, all imports from Brazil from 2010 onward benefit from D5 price incentives which support, to varying degrees, sugarcane ethanol imports. However, imported volumes since 2010 are small (0–200 million gallons, Figure 16.6) in all years except those affected by the U.S. drought (2012–2013; imports rose to 300-400 million gallons). The observed pattern of sugarcane ethanol imports aligns well with three factors: the 2012–2013 drought, seasonal two-way trade (Figure 16.10), and CA-LCFS (suggested in the port of entry for imports in later years; Figure 16.8). Several factors that influence observed U.S.-Brazil trade in ethanol are illustrated in Figure 16.10. The literature review for this report did not identify peer-reviewed studies that examined the U.S.-Brazil ethanol trade in terms of potential explanatory factors for observed variations in trade volumes, or to discern the relative importance of the RFS Program compared to other factors such as those documented in Figure 16.10 (e.g., weather, seasonal variations in supply-demand dynamics, changes in Brazil's domestic or international policies, sugar markets, the CA-LCFS, etc.). However, reviews of literature and data on trade flows identify several additional factors influencing Brazil's ethanol production, consumption, and overall terms of trade for ethanol, including tariffs that are transitory and varying in strength, relative exchange rates, relative prices of sugar and ethanol, and economic growth. Therefore, while D5 RIN prices theoretically offer an additional incentive to export to the U.S. market, volumes imported from Brazil have been small (<200 million gallons) in years other than those affected by the drought. One reason for the low import levels is that, as reported in Chapter 7, use of D5 RINs has been limited because after reaching the E10 blend wall in roughly 2013 (see Chapter 6, section 6.2), domestic biodiesel and renewable diesel appear to serve as the marginal fuel to meet both the

[11] Also see Chapter 1, section 1.1 for an explanation of the years under the RFS1 and RFS2.

advanced biofuel and total renewable fuel volume requirements in those years (Chapter 7, section 7.2.2). In summary, the observed patterns of sugarcane ethanol imports relative to D5 RIN prices do not suggest that the RFS Program played a significant enough role for the timing and volumes of observed imports.

Overall, the RFS Program provided incentives that contributed to observed imports from Brazil over time. However, since imports were relatively small and the RFS Program was one among many variables, a marginal share of U.S. ethanol imports from Brazil uniquely attributable to the RFS Program or D5 RINs could not be quantified with confidence based on the available empirical data.

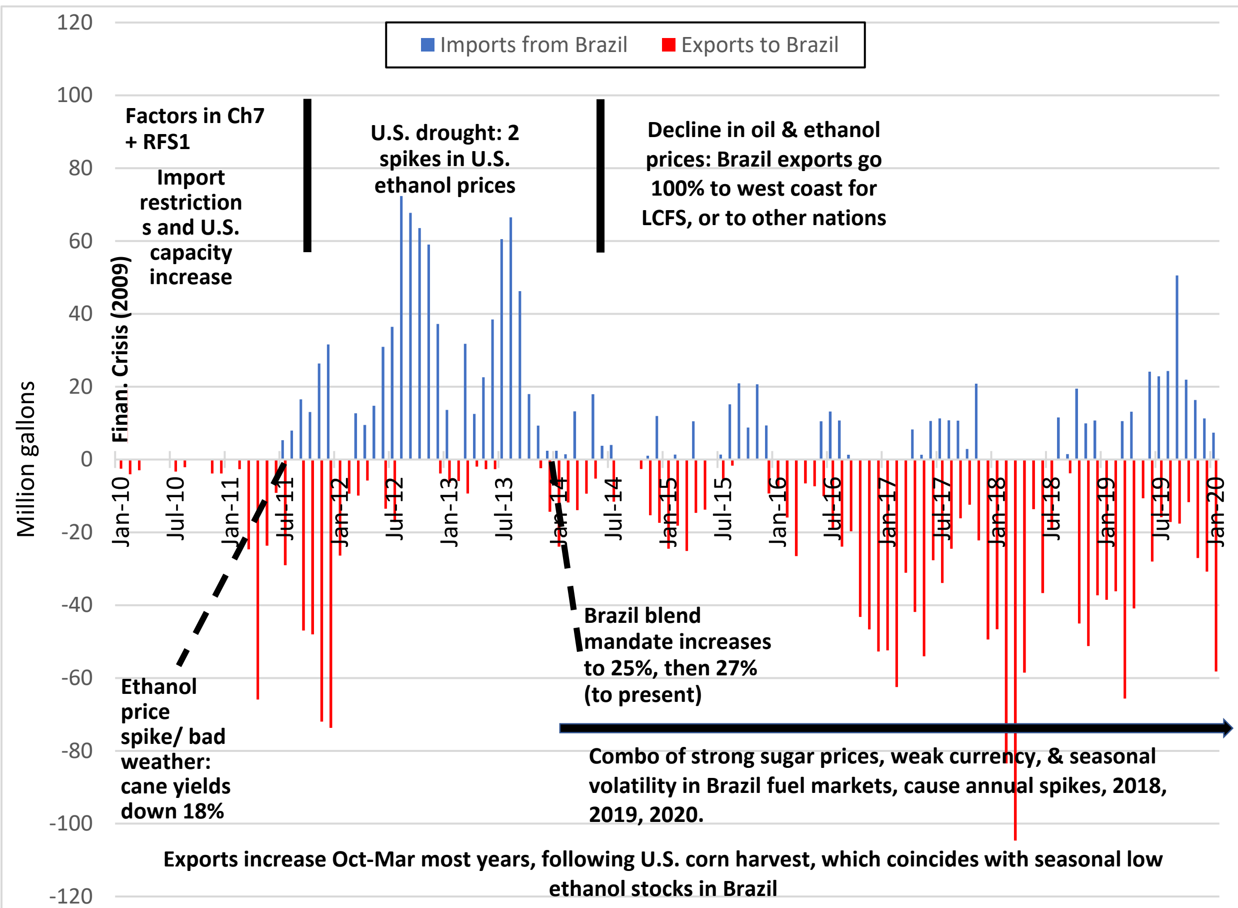


Figure 16.10. Monthly gross U.S. fuel ethanol imports from and exports to Brazil (EIA, 2022) and factors that influenced observed variations in trade volumes. D5 RINs added incentives throughout this period (2010–present) but observed import volumes appear to respond to specific events (see USDA FAS reports) such as those illustrated rather than to changes in D5 RIN price.

16.3.2.2. Factors Influencing Brazil's Production and Exports

Brazil's sugarcane ethanol industry and its transformation over the past 40 years has been examined in over one hundred published studies in the literature and was reviewed in several chapters of the Scientific Committee on Problems of the Environment (SCOPE) Report on Bioenergy &

Sustainability (Souza et al., 2015). Sugarcane has been cultivated in Brazil for over 500 years (de Souza et al., 2014) with few major innovations until the ethanol industry developments began in the 1970s. A government-industry coalition developed a strategy in 2003 to expand the ethanol sector for three reasons: energy security, to establish global leadership in bioenergy (supply foreign markets), and to promote rural development (Sanchez Badin and Godoy, 2014; Goldemberg, 2008). Social, environmental, and technological transformations have been promoted by Brazilian government policies and industry-led initiatives to reduce wastes (generating electricity from the cane residues or bagasse), mechanize planting and harvests (eliminating the requirements for low-wage manual harvests and burning of cane fields), and implement voluntary green certification, which has catalyzed initiatives to better manage waste water, restore riparian areas, and expand private forest reserves (Walter et al., 2014).

Investments in sugarcane production in Brazil have grown since supportive policies were adopted to promote ethanol fuels beginning in the 1970s, which were substantially revised and reinforced in the early 2000s (Antunes et al., 2019). Brazilian sugarcane production is concentrated in the southern and eastern parts of the country (Figure 16.11), away from the Amazon rainforest; although, in 2019 the Brazilian government lifted a moratorium on growing sugarcane in the Amazon (Ferrante and Fearnside, 2020).

1 kilometer (km) = 0.6 miles

Figure 16.11. Brazil's sugarcane growing regions. Source: Caldarelli et al. (2017) (used with permission).

Also of note is the recent increase in the production of corn-based ethanol in Brazil, which has been driven by a surge in corn production in Brazil's Center-West region, poor weather impacting sugarcane production, high sugar prices that favors more sugar relative to ethanol production, and incentives under the RenovaBio, Brazil's most recent major biofuel policy.[12] The volume of ethanol from corn feedstock has increased from close to nil in 2014 to nearly 900 million gallons, or 11% of total ethanol production in Brazil for 2021 (USDA FAS, 2021a). In Brazil, most corn is produced as a second crop in rotation with soybeans in a single calendar year. While Brazil's corn-based ethanol industry is growing, it is likely not contributing to Brazil's ethanol exports. The center of corn ethanol production in the Center-West region

[12] RenovaBio is the name for Brazil's low carbon fuel policy under its National Biofuels Policy (https://www.fas.usda.gov/data/brazil-implementation-renovabio-brazils-national-biofuels-policy)

is far from the existing ethanol export ports of the Southeast. Transporting corn ethanol to these ports would require movement by truck of over 800 miles then transfer to another mode of transport. Due to these challenges, most corn ethanol is consumed locally and not exported overseas (USDA FAS, 2021a). The potential environmental effects of these recent developments (Brazilian RenovaBio incentives began in 2020), and their market-mediated interactions with the RFS Program, were not examined in this study but represent an area for future research.

Ethanol production increased steadily from 2000 up to 2008 in both Brazil and the U.S. but production trends diverged in subsequent years (Figure 16.12). The annual analyses of biofuel developments in Brazil conducted by the USDA FAS in Biofuel Annual Reports[13] document several factors behind changes in Brazilian biofuel markets and production each year, finding similar drivers to those behind U.S. growth from 2000 to 2008, such as increasing world oil prices, a strengthening Brazilian economy, and policies in Brazil designed to increase domestic ethanol consumption (also see discussion of drivers of U.S. ethanol production in Chapter 6). Brazil's incentives (including promotion of flex-fuel vehicle and blending mandates) reportedly were effective and domestic consumption of ethanol increased in parallel with production from 2000 to 2008 (Figure 16.13). In contrast to the United States, after 2009 there was not as much of a constraining blend wall in Brazil because of the replacement of gasoline vehicles by flex-fuel vehicles (FFVs) from 2003–2009 due to the aforementioned government programs. However, unlike the United States, Brazil's production did not grow significantly between 2008 and 2017 likely because of recessions, high sugar prices, sugar yield shortfalls and relatively modest blending mandates (Figures 16.12 and 16.13).

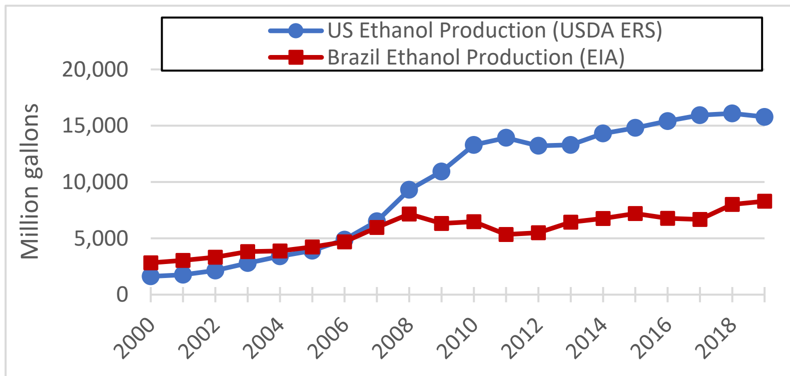


Figure 16.12. Annual ethanol production in United States (blue with circles, USDA-ERS) and Brazil (red with squares, EIA).

Analyzing the timing of imports of ethanol from Brazil to the United States compared to changes in Brazil's production, combined with review of the USDA FAS Biofuel Annual Reports (USDA FAS, 2019a), provides insights into major factors influencing Brazil's production and exports to the United States. For example, Brazil's production and exports reflect distinct periods associated with economic growth and high oil prices (2006–2008), global recession (2009–2011), disruptions to U.S. corn

[13] Available from https://www.fas.usda.gov/data/brazil-biofuels-annual-6.

production due to weather events (2008 and 2012–2013), increased value of sugar relative to sugarcane (2011–2013, 2016–2017), increases to Brazil's domestic consumption mandates (2015), and other factors. As illustrated in Figure 16.13, the global market and economic drivers discussed in Chapter 6 for U.S. corn ethanol also supported expanding production in Brazil, as well as U.S. imports from Brazil, until 2008. However, economic growth, tariffs, exchange rates, and other factors then drove Brazil production downward (and exports to U.S. to zero) by 2010. Exports to the United States rose in 2011–2013, largely attributed to the U.S. drought and facilitated by the elimination of the special duty on imported ethanol. A combination of non-RFS factors appears to explain the observed variability in annual volumes of U.S. imports from Brazil (Figure 16.10) and as discussed above, while the RFS Program helped, its influence was not precisely quantified.

Brazil's ethanol exports to the United States from 2004 to 2008 can best be explained as responses to the same drivers that promoted rapid growth of U.S. ethanol production capacity in those years (Chapter 6), including exports to support MTBE replacement in the United States from 2003–2006, and favorable markets relative to crude oil in transportation. The exports to support MTBE replacement appear to have been mostly in RFG areas outside of California, as evidenced by the lack of imports in 2002 and 2003 (Figure 16.8) when the transition in California occurred, and higher imports in 2004–2006 when the conversion in other RFG areas was taking place (see Chapter 6, sections 6.2.2 and 6.2.3). Possible effects from the RFS1 in 2007 and 2008 cannot be dismissed. However, as discussed in Chapter

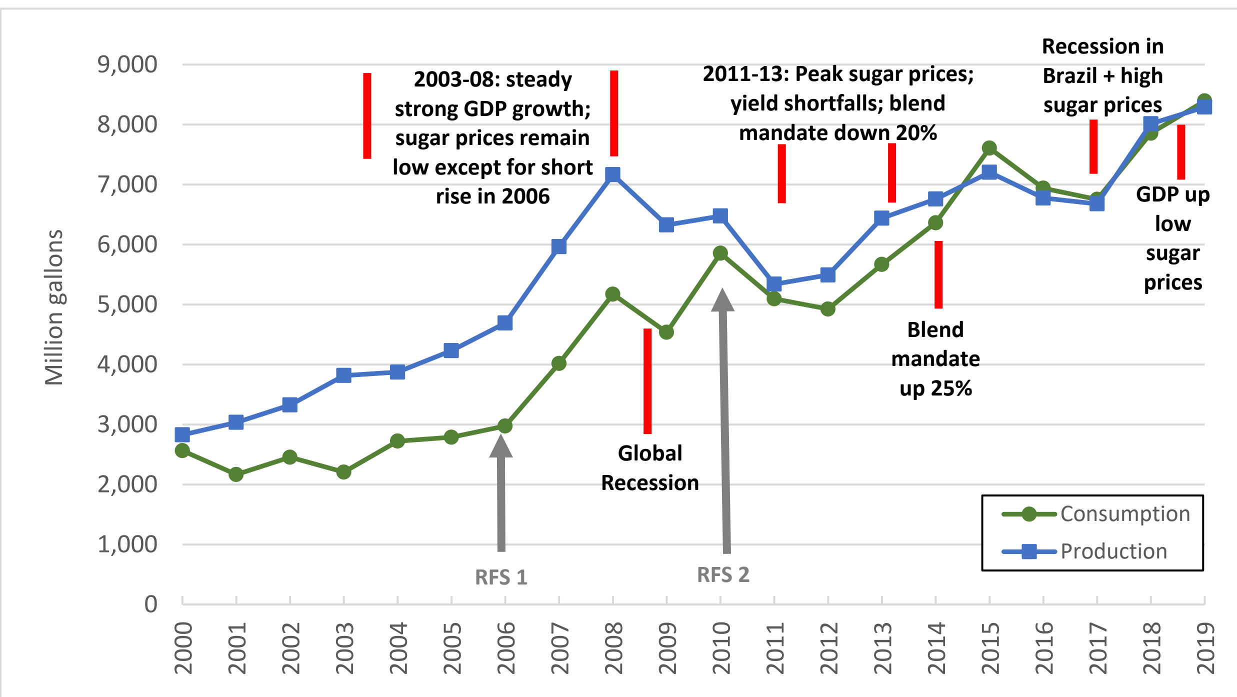


Figure 16.13. Drivers of Brazil ethanol production and events compared to Brazil production and consumption of ethanol (EIA).

6, there is no digital RIN information from 2007 that can be used to evaluate the RFS1 effect, and also there are few models that assess this period (see Chapter 6, sections 6.3.2 and 6.3.3). There appears to have been a small effect in 2008–2009 in the D6 RIN data (which Brazilian ethanol was part of before 2010) and in simulation modeling when oil prices dropped due to the Great Recession. However, aside from the largest year of MTBE phaseout (2006, Chapter 6, Figure 6.5 and Appendix C, Figures C.16 and C.17) imports from 2004 to 2008 were still fairly minor relative to Brazil's total production (3% or less, Figure 16.9). Starting in 2014, and subsequent to the U.S. drought in 2012–2013, ethanol imports from Brazil to the United States declined sharply (Figure 16.10). While most ethanol imports from 2006–2014 arrived in U.S. ports on the East Coast or the Gulf, virtually all imports from 2016 to the present were received on the West Coast, likely serving the CA-LCFS market (Figure 16.8). However, with the exception of the U.S. drought years, ethanol imports from Brazil have been dwarfed by U.S. ethanol exports to Brazil since 2010, reflecting the value to both nations of seasonal, two-way trade (Figure 16.10). In addition to the domestic factors discussed above, and an underlying incentive from the RFS Program, inspection of intra-annual trade reveals strong seasonal and punctuated trade between the United States and Brazil. These shifts in trade dynamics are likely in response to short-term opportunities that arise from weather (drought or excessively wet conditions in either nation that impacts feedstock availability and cost), shifting relative currency exchange rates, the ratio of sugar price to ethanol, and spot prices for ethanol in the United States, among others.

The U.S. EIA began documenting fuel ethanol exports in 2010, and every year since those data have been collected, U.S. exports of fuel ethanol have exceeded imports. Gross exports have represented 1–11% (and on average 6%) of U.S. ethanol production from 2010 to 2019. Even in the drought years (2012–2013), the United States exported 6–9% of U.S. ethanol production. However, annual data (Figure 16.14) mask high variability of imports and exports within each year (Figure 16.10). Under otherwise predictable conditions (e.g., in years when neither nation experiences severe fluctuation in feedstock availability due to weather), seasonal variation in U.S. exports to Brazil roughly follows a calendar determined by sugarcane harvest and processing in Brazil (Figure 16.10). The seasonal cane processing causes Brazil's domestic ethanol prices to fluctuate, rising in December–March prior to sugarcane crush and processing and then falling later in the year (USDA FAS, 2019a). Similarly, the seasonality of U.S. harvests, with peak times for corn being milled and processed to ethanol in October–December, and peak harvest and milling of sugarcane in March–May, contributes to varying ethanol prices. The resulting two-way sub-annual trade offers a ready market outlet for surplus production, which otherwise might need to be stored or shipped at higher cost/lower profit margin to other destinations. It also benefits consumers in both nations with a relatively low-cost alternative for any time when domestic supplies run low or prices spike for other reasons. The response of Brazil to such price spikes in the United States is apparent when the timing of imports from Brazil is compared to U.S. ethanol prices, which have ranged from under $1.30 to $3.60 per gallon.

The EIA monthly reports on ethanol trade between the United States and Brazil show many months with no import/export and single points for months with reported volumes when both prior and subsequent months had zero reported volumes. USDA FAS Biofuel Reports explain that Brazil's exports to the United States increase during "windows of opportunity opened by spikes in U.S. ethanol prices" and that "price spikes in the United States have also limited trade but not sufficiently to prevent imports from the U.S., [to Brazil] even with the 20 percent import tariff paid on imported volumes above the quarterly 150 million liter TRQ [quota]."

As in the United States, domestic factors (such as those illustrated in Figures 16.10 and 16.13) are more obvious, direct, and plausible drivers for observed changes in Brazil's ethanol output than the relatively small export markets (Figure 16.14).

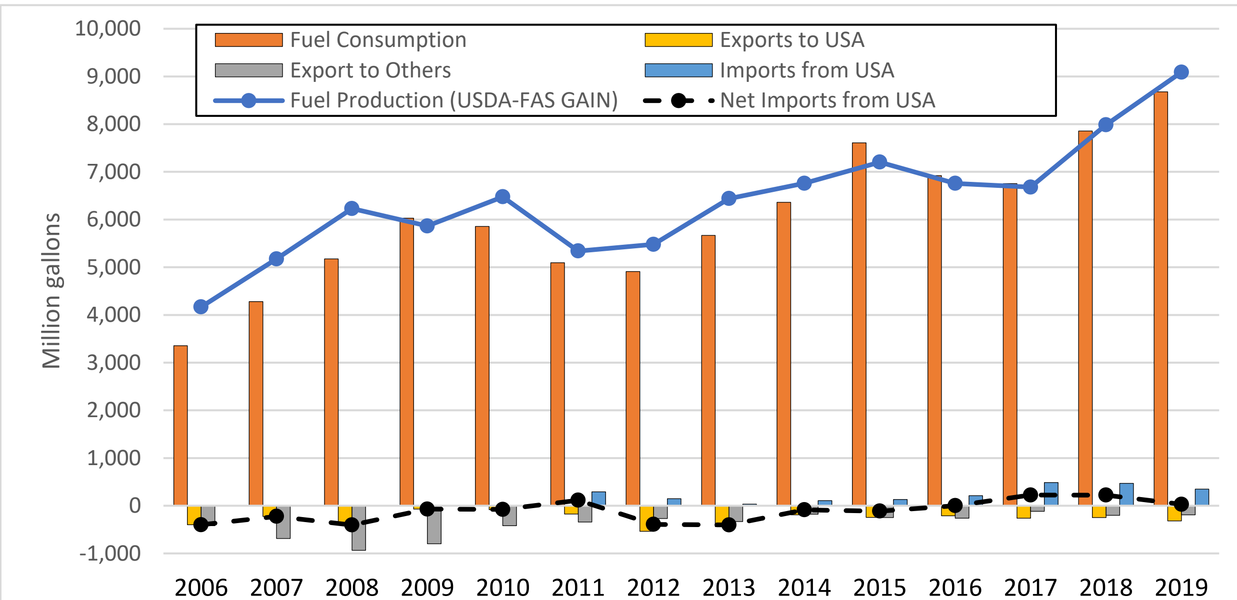


Figure 16.14. Brazil fuel ethanol production and disposition (from USDA FAS-GAIN Brazil: Biofuels Annual Reports 2010, 2012, 2019). Note that exports are illustrated here as negative values (reducing Brazil's domestic ethanol supply pool) while imports from the United States are shown as positive values (adding to the ethanol supply pool).

16.3.3 Effects of Ethanol Exports from the United States

Another, often overlooked, relationship among U.S. biofuel production, the RFS Program, and ethanol trade is increased ethanol exports from the United States to other countries as noted by relationship 1.2 in Box 16.1. Ethanol exports cannot be directly attributed to the RFS Program, as exports do not generate RINs. However, the RFS Program may have indirectly spurred exports by supporting the growth of the U.S. ethanol industry by providing investors with added confidence and through other means. By exporting U.S. ethanol to other markets, part of the environmental impact of that industry remains in the United States, in part because these exports avoid the need to grow feedstock and produce ethanol in foreign nations but additionally to offset the need for petroleum and its derivatives. Thus, this effect would reduce corresponding environmental impacts in the receiving nations. This is a direct corollary to the concern raised in EISA Section 204 regarding imports to the United States and the estimation of environmental impacts abroad associated with producing biofuels. The global ethanol market is dominated by the United States and Brazil so if environmental effects could be identified, Brazil or other destinations for exported U.S. ethanol, corn, and soybeans are the places to investigate (Figure 16.6). This effect is not specifically requested in Section 204 of EISA, and is not fully developed in the RtC3, but represents a potential area for future investigation.

To illustrate the amount of land potentially affected by ethanol trade with Brazil, an estimate was made for the cropland area in Brazil that would have been required to produce the amount of ethanol exported to and imported from the United States (Figure 16.15) on an annual basis (annual net trade). Based on average sugarcane ethanol yields for 2010–2019 (541 gallons per acre)[14] the annual land area requirements for exports from Brazil to the United States range between zero and 837,000 acres in 2006 (Figure 16.15). Similarly, exports to Brazil from the United States represent land area requirements in Brazil ranging from zero to 924,000 acres in 2018. As shown in Figure 16.14, while imports and exports vary each year, they are eclipsed by both countries' domestic markets. Furthermore, cumulative net

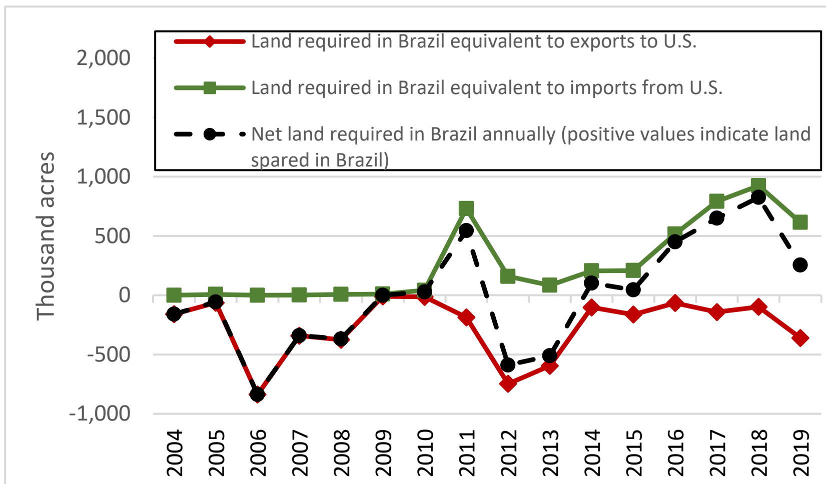


Figure 16.15. Estimate of crop area required in Brazil to produce ethanol volumes traded between the United States and Brazil. Area above zero represents potential land sparing in Brazil and area below zero represents potential land requirements in Brazil associated with net trade (Sugarcane production and harvested area used to calculate yield from UNICA. Net imports of ethanol to Brazil from EIA).

[14] The 2010–2019 average sugarcane ethanol yield = 541 gallons per acre, which was calculated from annual yield obtained from UNICA, and an average sugarcane to ethanol conversion efficiency of 21.13 ga/ton, which was taken from the literature. Compare this to U.S. corn ethanol yield = 476 gallons per acre, which was calculated from 2013-2019 average corn yield = 170 bushels per acre (see Chapter 3, section 3.2.1.1) and an average corn to ethanol conversion efficiency, with a dry-mill process, of 2.8 ga/bushel (see Chapter 3, section 3.4.1.1).

imports to the United States from Brazil for 2005–2019[15] are negative (United States has exported 108 million gallons more to Brazil than it has received from Brazil in imports) and U.S. exports to Brazil relative to imports have been growing over the past decade.

When U.S. ethanol imports from Brazil exceed U.S. exports to Brazil on a land-area-adjusted basis, the black dotted line in Figure 16.15 represents land area in Brazil required to support the net exports from Brazil. When U.S. exports to Brazil exceed U.S. imports from Brazil on a land-area-adjusted basis, positive values of the black dotted line in Figure 16.15 represent the equivalent "land sparing" in Brazil, or land that theoretically was not required for the volume of net imports to Brazil. All these values are theoretical, relatively small compared to overall production (Figure 16.14), and ignore several important factors such as imports and exports to other nations. Net trade since 2010 favors the United States (land sparing in Brazil) for all years except 2012–2013 when net imports from Brazil represent net land requirements in Brazil. However, even in 2012–2013, the United States was a net ethanol exporter when exports and imports to all countries are considered (Figure 16.3). Also, this illustration simply represents total ethanol trade with Brazil, which is driven by many factors.

The analysis for trade with Brazil illustrates some of the challenges in estimating international effects of the RFS Program. Not only is it difficult to quantify the influence of the RFS Program on observed trade patterns, but it is also difficult to estimate international effects when the United States has been a net global exporter of fuel ethanol every year since 2010. Given the diversity of destinations and volumes of U.S. trade, as well as the use of ethanol imports in foreign countries for non-biofuel purposes (e.g., industrial uses), there could be small land sparing or clearing effects in over 70 nations around the globe to the extent that U.S. exports reduced the need to produce ethanol domestically in each case. The degree to which any net environmental benefits could accrue depend on many factors including the supply chains, technologies, and feedstocks used in importing and exporting nations.

16.3.4 Market-Mediated International Effects of U.S. Corn Ethanol and the RFS Program

Two of the primary concerns in the literature on the international effects of the RFS Program and ethanol are related to increased demand of ethanol that induces feedstock production overseas for import to the United States (e.g., of ethanol from corn or sugarcane) (relationship 1.1 in Box 16.1), or increased demand for corn that increases crop prices and induces crop production overseas (relationship 1.3 in Box 16.1). For the former, as discussed above, the ethanol imports over the period of study were primarily from Brazil, the production of which were induced primarily by Brazil's own domestic policies. The latter

[15] Based on EIA data for which export data do not begin until 2010. U.S. exports to Brazil for years prior to 2010 are assumed to be zero. The trade balance will further favor the United States to the degree there were any U.S. exports to Brazil in those years. For years with complete EIA data (2010–2019) trade favors the United States by 974 million gallons.

effect deserves more scrutiny; however, as concluded in Chapter 6, the portion of corn production attributable to the RFS Program was small from 2005 until reaching the E10 blend wall, and reached a high point of 0–3.5 million acres of corn in 2016 (0–3.7% of corn planted acreage in 2016; Chapter 6, section 6.4.2).

Prior to EPA's announcement of the proposed biofuel mandate volumes for 2023–2025, which are outlined in the Set Rule,[16] the Agency conducted an analysis of potential climate change impacts associated with the RFS Program. The Draft Regulatory Impact Assessment (DRIA) for the Set Rule includes a compilation of biofuel lifecycle GHG estimates as well as a review of published estimates of land cover and land management changes resulting from corn ethanol and soy biodiesel production.[17] Details about the methods used for each estimate, including the type of model, the economic and non-economic sectors simulated, the spatial coverage, and the temporal representation used in each model-scenario combination are described in the docket (EPA-HQ-OAR-2021-0427). Estimates of domestic cropland changes from corn ethanol and soy biodiesel production are summarized in Chapters 6 (section 6.4.1.2) and 7 (section 7.3.3), respectively. Here is highlighted the subset of those estimates of cropland expansion outside of the United States from corn ethanol from this same review (cropland expansion outside of the United States from soybean biodiesel discussed in section 16.4.2).

Estimated land area changes were normalized across all model simulations by dividing the total estimated cropland change (million acres) by the biofuel mandate volume used (billion gallons) in the model to show the results on a common scale (Mac/Bgal) (Figure 16.16). The mean or other statistical measure of the collection of estimates are not presented because of the differences in the scenarios and definitions of land types used in the different models (Plevin et al., 2022; Plevin et al., 2015).

[16] See docket EPA-HQ-OAR-2021-0427.

[17] Full text of the DRIA is available online (https://www.epa.gov/renewable-fuel-standard-program/proposed-renewable-fuel-standards-2023-2024-and-2025) and under docket EPA-HQ-OAR-2021-0427. Also see summary in Chapter 6 section 6.4.1.2.

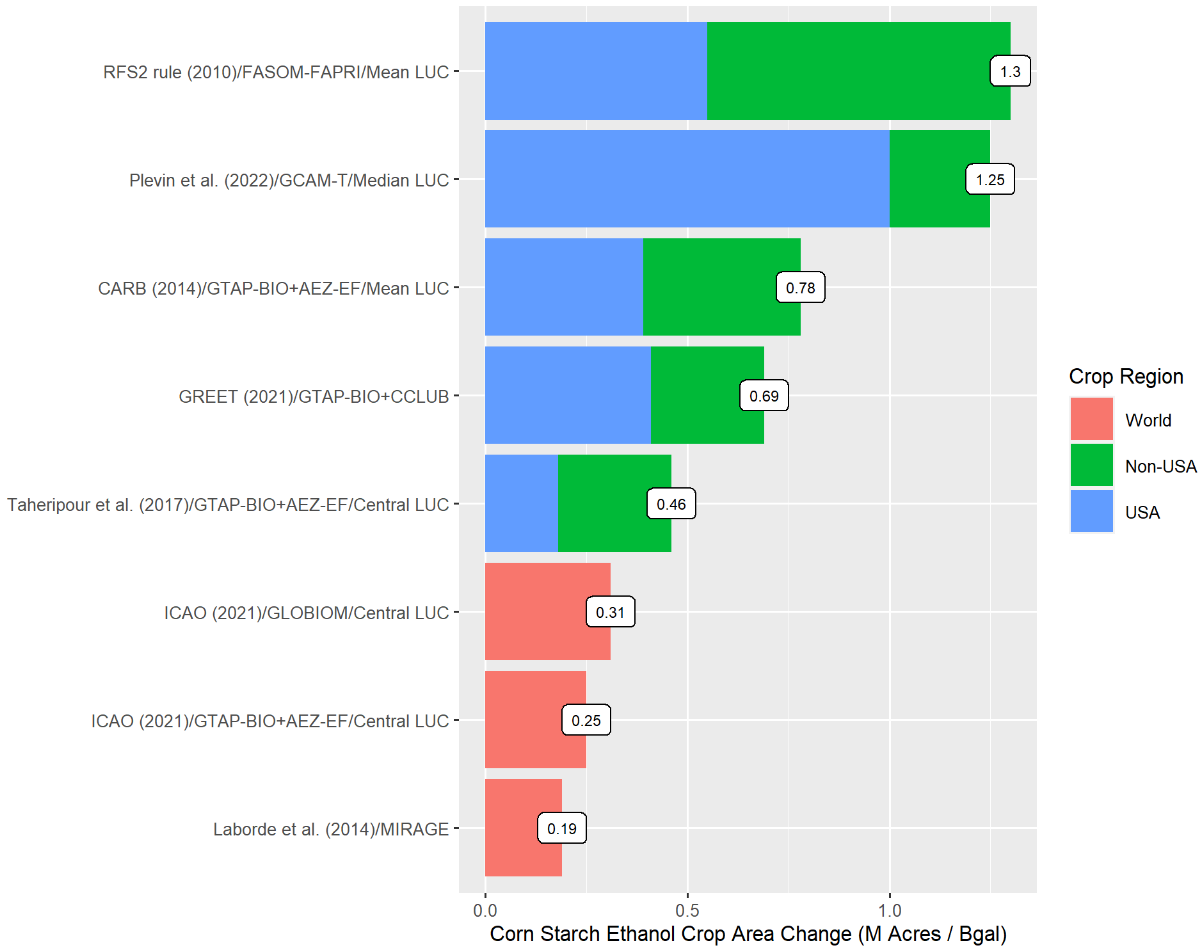


Figure 16.16. Cropland area change estimates per billion gallons of ethanol by study for corn ethanol. The name on the y-axis for each bar/estimate includes multiple descriptors separated by "/". In order, these descriptors are the author or other name (e.g., RFS2 rule), the name of the model used to estimate land use change impacts, and a brief descriptor of the scenario modeled. For studies that did not report disaggregated estimates by USA and Non-USA, only the World total is presented. Scenarios modeled and definitions of cropland differ across studies. ICAO (2021) estimates for corn ethanol to jet fuel were adjusted based on the assumed jet fuel yield. Reproduced from the DRIA (EPA-HQ-OAR-2021-0427). Sources: CARB (2014), U.S. EPA (2010), ICAO (2021), Laborde et al. (2014), Plevin et al. (2022), Taheripour et al. (2017). (Note, this is the same figure as Figure 6.23.)

Within the eight model-scenario combinations analyzed as part of the DRIA (Figure 16.16), the five that distinguish between changes in the United States compared to changes outside the United States estimate non-U.S. cropland expansion due to corn ethanol production of 0.25–0.75 Mac/Bgal. The other three models that estimate only global cropland changes report lower cropland area changes (0.19–0.31 Mac/Bgal), indicating that if these studies reported non-U.S. changes it would decrease the lower end of the 0.25–0.75 Mac/Bgal range. Thus, using the estimated effect of the RFS Program from Chapter 6 (0–2.1 million gallons), suggests that 0–1.6 million acres of cropland expansion may have occurred outside the United States from the effect of the RFS Program on corn ethanol.

As discussed in Chapter 6 and here, there are a relatively wide range of estimates in the literature on the extent of these market-mediated impacts, and this topic deserves further study through statistical analysis and simulation modeling. While there are multiple statistical studies that have estimated the effect of corn ethanol on U.S. cropland area, there are no statistical studies that estimate the global effects (see Chapter 6.4.2). This is an illustrative range based on available simulation modeling estimates, and further research is needed on the international market-mediated environmental effects of the RFS Program and U.S. corn ethanol production. EPA has conducted a Model Comparison Exercise (MCE) to try and better understand the factors in these various modeling approaches that lead to higher or lower estimates (see next section).

16.3.5 Model Comparison Exercise Technical Document

Section 6.4.1.4 of Chapter 6 discusses the results of an EPA Model Comparison Exercise (MCE) Technical Document, which among other metrics, compared international land use change results from a U.S. domestic corn ethanol consumption scenario across four economic models.[18] This report provides the latest assessment of indirect land-use changes potentially attributable to biofuels. The results are preliminary, and EPA's work on this topic is ongoing. The report showed a range of potential future international land use change impacts from the different models. Across these models, there is variation in both the magnitude of additional international cropland cultivated in response to increased consumption of corn ethanol in the United States and also variation in the mix of land covers converted to cropland. This area of scientific inquiry is a critical topic for further research. For a brief summary of the findings, see section 6.4.1.4. For more information, including MCE conclusions, see the full report available on EPA's website.[19] EPA will continue this work through forthcoming relevant regulatory actions and to inform future reports.

16.4 Other Biofuels and Horizon Scanning

16.4.1 Biomass-Based Biodiesel Trade and Effects

As was discussed above and in Chapter 2, whereas ethanol consumption in the United States has been dominated from 2005 to present by ethanol from the United States and Brazil, biomas-based diesel consumption (including biodiesel and renewable diesel) has come from a variety of feedstocks (domestic and international) and countries (Figure 16.17). Like ethanol, biomass-based diesel sources are dominated by domestic sources like fats, oils, and greases (FOGs) and soybean, though less so compared with ethanol. For example, biodiesel imports in 2016 from Argentina soybean and Southeast Asia palm together (734 million gallons) were almost as much as biodiesel from domestic soybean (865 million gallons, Table 2.1). Imports of both of these have since decreased strongly and domestic production has

[18] See EPA-420-R-23-017
[19] See EPA-420-R-23-017

risen. Thus, aside from these high-point years, many of these sources of imports were relatively small and short-lived but may not be insignificant in terms of international environmental effects. As discussed in Chapter 7, biomass-based diesel imports have a distinct pattern and different driving forces than ethanol. The drivers for U.S. imports of biomass-based diesel can best be understood by considering a few time periods that had distinct patterns of imports and exports: 2006–2010, which had high volumes of both imports and exports in the same year and encompasses the "splash & dash phase," and 2013–2017, which was a period of high imports and relatively low exports (see Chapter 7, section 7.3.6). During the 2006–2010 period, the United States was a net exporter of biodiesel, averaging roughly 130 million gallons per year.[20] During the 2013–2019 period, the United States was a net importer of biomass-based, averaging

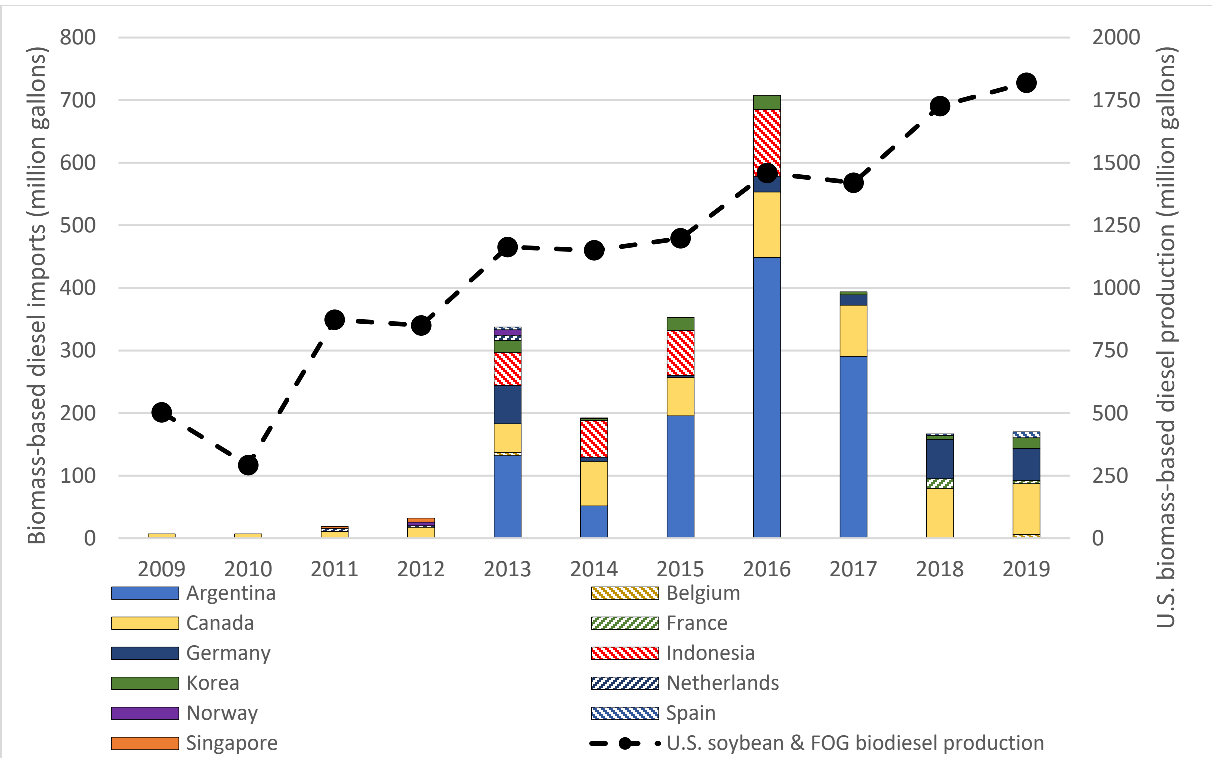


Figure 16.17. U.S. total biomass-based diesel imports by 11 leading (99.5% of total volume from all countries) sources and U.S. soybean and FOG-based biomass diesel production (EIA, 2022).[21]

roughly 220 million gallons per year (see Chapter 7, Figure 7.8).[22] While biodiesel imports have declined in recent years, renewable diesel imports have remained steady (Figure 16.3). Review of trade data finds

[20] This average export level is for the the years 2006–2012.
[21] Data for Figure 16.16 is from EIA and for Table 2.1 is from EPA's EMTS system (see Appendix B), thus the exact values may differ slightly from year to year. Although the majority of biodiesel is used in transportation, some may be used in industrial processes or mixed with other fuels for home heating. The differences are small for years aside from 2013 and the general trends are consistent. Thus, any differences in the data between these two sources do not materially impact the conclusions of this report. Note differences in scale between left and right axes.
[22] This average import level is for the years 2013–2019.

some two-way trade between the United States and partners such as Mexico and Canada, which likely reflect in part the logistical advantages in specific border locations.

Many countries produce and trade biomass-based diesel but, in terms of volume, only a few countries dominate global trade. This section focuses on trade between the United States, Argentina, and Southeast Asia which made up 36% of global production from 2010–2020.[23] U.S. biomass-based diesel imports from all sources peaked in 2016 (Figure 16.3, Table 2.1) and offers a useful demonstration of how the combination of domestic policies, mandates and incentives, along with international trade policies, interact and create large swings in biofuel production, consumption, and trade. Biomass-based diesel imports from Southeast Asia are also discussed briefly, which were between 1-2% of U.S. biofuel pool for three consecutive years (2014–2016, Chapter 2, Table 2.2). Finally, drivers of U.S. biomass-based diesel exports and associated environmental effects are discussed. In general, this section on biomass-based diesel is less detailed than the section above on ethanol. The focus is on U.S. ethanol imports because they have been significantly larger than U.S. biomass-based diesel imports and thus potentially more significant in the context of the RFS Program and the environmental effects under Section 204. However, U.S. biomass-based diesel imports, production, and consumption may have important environmental effects abroad.

16.4.1.1. Biodiesel Imports from Argentina

This section examines the case of soy biodiesel imports from Argentina, which has been intermittent as an import to the United States (Figure 16.16). This appears to be due to the combined effects of policies supporting production and trade in Argentina, strong demand from the United States, and U.S. trade policies.

Argentina has had a strong oil crop industry for decades. In 2001 Argentina established a national biodiesel strategy, which included tax exemptions to accelerate the industry's growth, and later a biofuels law as well as a National Commission on Biofuels (Naylor and Higgins, 2017). These actions succeeded at creating one of the world's most efficient vegetable oil crushing industries (Beckman, 2015). Argentina's biodiesel sector was developed further as a means to meet the country's Paris Agreement and COP23 obligations. Additionally, a growing demand for biodiesel in the EU, the United States, and other countries enabled Argentina to export more than they consumed. From 2008 to 2014, Argentina exported 70% of their biodiesel production (Naylor and Higgins, 2017). From 2014 to 2016, the United States imported increasing volumes of biodiesel from Argentina, reaching an annual maximum of 435 million gallons in 2016 (see Chapter 2, Table 2.1). This increase in imports to the U.S. was driven by tax policies

[23] EIA publishes import data from 17 countries (Argentina, Canada, Indonesia, Germany, South Korea, France, Spain, Norway, Netherlands, Belgium, Singapore, Australia, Panama, Finland, Hong Kong, Taiwan, and Portugal, in order of decreasing import volume) (EIA, 2022).

in Argentina that were favorable to export as well as the increased demand in the United States for biodiesel. A 2017 USDA FAS Biofuels Annual Report for Argentina estimated that 88% of the country's biodiesel export would go to the United States in 2018 (USDA FAS, 2017). Imports from Argentina and Southeast Asia have dropped to zero since 2017, however, due to a U.S. antidumping complaint and countervailing duties announced by the United States in August 2017 to limit unfairly subsidized biodiesel imports from both Argentina and Indonesia (ITA, 2017). Since then, biodiesel exports from Argentina have largely gone to Europe (USDA FAS, 2020a).

The expansion of soybean production in Argentina is likely associated with expanding agricultural frontier and associated environmental effects (Phélinas and Choumert, 2017). However, as discussed above in the case of Brazil, domestic policies and markets are expected to be far more influential factors for land management decisions than a short-term export market partly facilitated by the RFS Program.

16.4.1.2. Biodiesel Imports from Southeast Asia

Other than Argentina, the countries that the U.S. imported biomass-based diesel from, as well as the feedstock types used, varied substantially from year to year (Figure 16.16). However, given that biomass-based diesel from Southeast Asia was relatively small compared with the total U.S. pool (i.e., 0.9-1% in 2015 and 2016, Table 2.2), and the ecological concerns of deforestation of Southeast Asian peatlands for palm oil, only a short discussion on this imported source of biomass-based diesel is included here. A broader discussion of palm oil markets is provided in section 16.5.

U.S. biomass-based diesel imports from Southeast Asia were small prior to 2013 (Figure 16.16; and Chapter 2, Table 2.1, partially recreated in Table 16.1). This was because the E10 blend wall had not yet been reached and either domestic corn ethanol or imported sugarcane ethanol from Brazil was the most cost-effective biofuel for meeting the RFS mandates. As discussed in Chapter 2, the overwhelming majority of biomass-based diesel from Southeast Asia is produced from palm oil feedstock or from waste FOGs, such as used cooking oil or inedible animal tallow. Looking at the biomass-based diesel imports by feedstock, palm oil-based imports reached a high of 178 million gallons in 2016. Conversely, FOG-based imports from Southeast Asia have grown relatively steadily since 2013, reaching a high of 286 million gallons in 2019. The largest source of FOG-based imports has been renewable diesel from Singapore where a large renewable diesel production facility that uses primarily spent cooking oil, residues from vegetable oil production, and animal fats as feedstocks, started production in 2010. The next largest source is biodiesel from Indonesia.

Palm oil-based biofuels do not qualify as renewable fuel under the RFS Program, as they have not been approved as meeting the requisite minimum 20% greenhouse gas (GHG) emissions reductions.

However, biodiesel produced at "legacy" facilities (i.e., facilities under construction or operation prior to December 2007) may qualify for conventional biodiesel (D6) RINs. Biomass-based diesel produced from FOGs may be eligible as biomass-based diesel (D4) or advanced biofuel (D5) RINs, provided they are produced through an approved pathway and meet all other RFS Program regulatory requirements.[24] In roughly 2013, the E10 blend wall was reached yet consumption of biofuels in total continued to increase, largely driven by rising advanced and total renewable fuel mandates. This changed the economics of biomass-based diesel supply and demand. Ethanol was no longer the most cost-effective way to reach additional RFS obligations, so biodiesel or renewable diesel became the marginal fuels (discussed further in Chapter 7, section 7.2.2). In other words, excess domestic and foreign supplies of biodiesel and renewable diesel became the most cost-effective way for obligated parties to meet both the advanced biofuel and total renewable fuel volume requirements.

Table 16.1. U.S. biomass-based diesel imports from Southeast Asia by feedstock and year. (Subset of Tables 2.1 and 2.2 in Chapter 2) (palm Oil production in Malaysia + Indonesia for calculation in bottom row from USDA's Production, Supply, and Disposition database).

Imports	2013	2014	2015	2016	2017	2018	2019
Millions of gallons							
Palm oil	101	54	161	178	2	0	0
FOG	139	129	138	165	197	185	286
Total	240	183	299	343	199	185	286
Percent of U.S. biomass-based diesel consumption							
Palm oil	7.1%	3.8%	10.8%	8.5%	0.1%	0.0%	0.0%
FOG	9.7%	9.1%	9.2%	7.9%	9.9%	9.7%	15.8%
Total	16.8%	12.9%	20%	16.4%	10.0%	9.7%	15.8%
Percent of SE Asia palm oil production							
Palm oil	0.7%	0.4%	1.1%	1.1%	0.0%	0.0%	0.0%

To further evaluate the drivers for biomass-based diesel imports from Southeast Asia, the Biofuels Annual reports and related reports prepared by USDA GAIN from 2012 to the present were reviewed for Indonesia. Indonesian biodiesel capacity and production have grown tremendously in recent years with support from ambitious government programs and subsidies. In August 2018, in response to a weakening exchange rate, an increasing trade deficit, and surplus palm oil supplies, the government of Indonesia expanded its 20% biodiesel blending (B20) mandate that was established in March 2015 and included only the public service fleet, to now include the non-public service transport sector as well (USDA FAS, 2019b). This created a biodiesel fuel demand of over 1 billion gallons per year, with a goal

[24] The RFS Program regulations are located at 40 CFR 80 subpart M and online at https://www.epa.gov/renewable-fuel-standard-program/approved-pathways-renewable-fuel.

of B30 in the near future pending successful on-road testing (a blend-rate for biodiesel nearly three times higher than the 10% mandate in Argentina, the next highest nation) (USDA FAS, 2018a). While production and consumption have increased in recent years, biodiesel exports have been highly variable. Exports to Europe were strong from 2008 to 2012 but dropped precipitously in 2014 due to antidumping duties imposed by the European Commission. As exports to Europe dropped, they increased to China, Malaysia, and the United States. Near the end of 2017, the United States imposed antidumping and countervailing duties that effectively shut Indonesian biodiesel out of the U.S. market for the next five years (USDA FAS, 2018b).[25] By 2020 it appeared that biodiesel exports were all but disappearing due in large part to COVID-19 and lower biodiesel demand from China (USDA FAS, 2020c). Since 2020, biodiesel exports have rebounded modestly to approximately 130 million gallons (USDA FAS, 2023).

Indonesian biodiesel production and exports appear to be highly dependent on trade and domestic policies. When the sector is challenged by declines in export markets or other circumstances that hinder Indonesian biodiesel production, the government of Indonesia has attempted to counter these negative effects by bolstering the sector through various policy mechanisms that either support exports (e.g., lowering export levies), domestic demand (e.g., blending mandates), or lower production costs (e.g., direct subsidies). Most or perhaps all of these relationships are not explicitly included in the currently available suite of global models (discussed in section 16.4.2).[26] Given the many other factors at play, and the fact that palm oil biodiesel does not qualify as renewable fuel under the program, it appears unlikely that the RFS Program has been the direct cause for significant volumes of Indonesia biodiesel production (i.e., observed production likely would have occurred in the absence of the RFS Program). While it is possible that the example set by the RFS Program of promoting biodiesel may have had an effect on Indonesian policy, at this time the extent of this potential effect cannot be evaluated.

Apart from Indonesia, about a third as much biomass-based diesel was imported from South Korea from 2013 to 2019, with no more than 22 million gallons in any single year (EIA, 2022). A larger source was renewable diesel imported from Singapore with imports of approximately 160 to 260 million gallons per year from 2013 to 2019. A large share of these imports is renewable diesel produced from FOGs that have generated biomass-based diesel (D4) RINs under the RFS Program. It is quite possible that these fuel volumes would have found a market in absence of the RFS Program, but that hypothesis was not evaluated for this report. The effect of the RFS Program on FOG-based renewable diesel imports

[25] However, Indonesian exports to other nations remained strong in 2018 in large part because it settled the World Trade Organization (WTO) antidumping case with the EU. Then exports jumped approximately ten-fold in 2019 to approximately 500 million gallons (USDA FAS, 2019b), based on export-driven policies by the Indonesian government (e.g., lower export levies).

[26] See for example Peng et al. (2021) for a discussion of how current models to do not account for political economy.

from Singapore and associated environmental effects are a potential topic for future research and evaluation.

16.4.2 Market-Mediated International Effects of U.S. Soybean Biodiesel and the RFS Program

As discussed in section 16.3.4, as part of the DRIA for the RFS Set Rule, EPA compiled biofuel lifecycle GHG estimates and reviewed published estimates of land cover and land management changes resulting from soy biodiesel production (and other biofuels). This section highlights the subset of those estimates of cropland expansion outside of the U.S. from soybean biodiesel from this review.

For soy biodiesel production, the five model-scenario combinations that distinguish between changes in the United States compared to changes outside the United States estimate non-U.S. cropland expansion of 0.13–0.57 Mac/Bgal. Among the three models that estimate only global cropland area changes, one reports a cropland expansion (0.29 Mac/Bgal) that falls within the estimates from the other five models and two that report cropland expansion values (1.21 and 1.42 Mac/Bgal) that are much higher. While this could mean that two of the three model-scenarios estimate more non-U.S. cropland expansion, the differences in methods and definitions used across the models prevent such a conclusion. The estimated land changes reported here come from model simulations that do not necessarily isolate the U.S. RFS Program from other global drivers of the biofuel industry. The lack of quantifiable volumes of soybean biodiesel attributable to the RFS Program means that the range of land change from model simulations cannot be used to estimate the land use change effects overseas from the RFS Program.

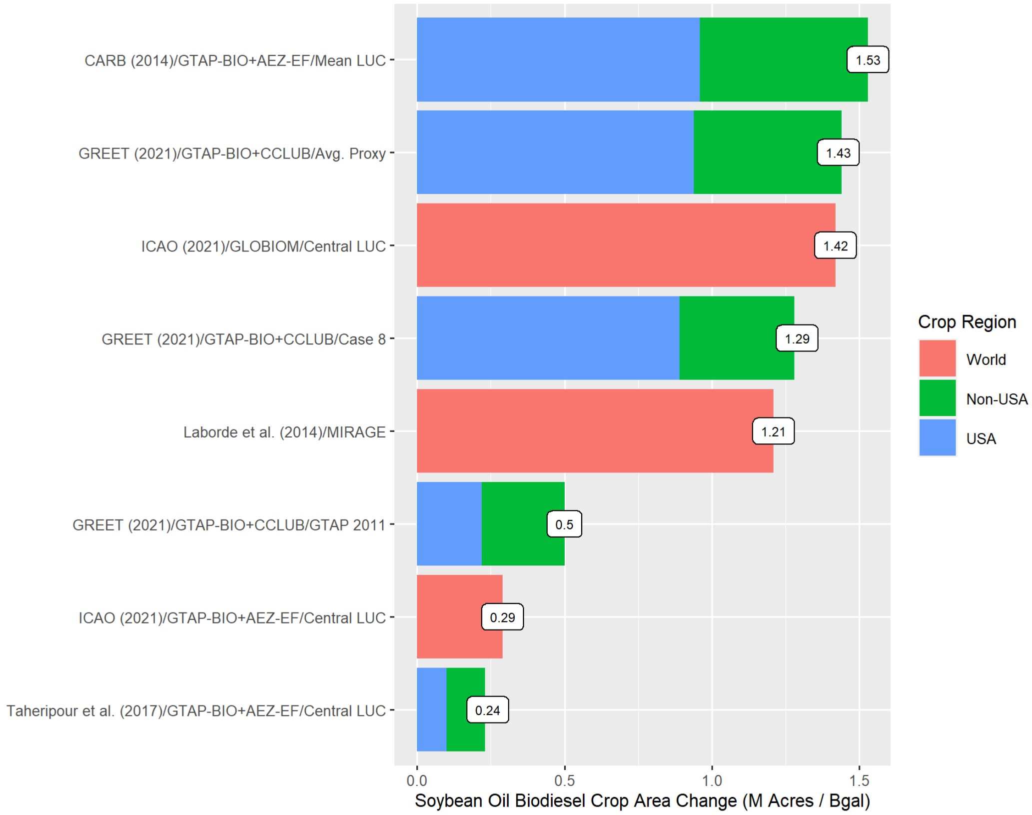


Figure 16.18. Cropland area change estimates per billion gallons of biodiesel by study for soybean biodiesel. The name on the y-axis for each bar/estimate includes multiple descriptors separated by "/". In order, these descriptors are the author or other name (e.g., CARB), the name of the model used to estimate land use change impacts, and in some cases a brief descriptor of the scenario modeled. For studies that did not report disaggregated estimates by USA and Non-USA, only the World total is reported. Scenarios modeled and definitions of cropland differ across studies. ICAO (2021) estimates for soybean oil to jet fuel were adjusted based on the assumed jet fuel yield relative to biodiesel. Reproduced from the DRIA (EPA-HQ-OAR-2021-0427). (Note, this is the same figure as Figure 7.10.).

16.4.3 Model Comparison Exercise Technical Document

Section 7.3.4 of Chapter 7 discusses the results of an EPA Model Comparison Exercise (MCE) Technical Document, which among other metrics, compared international land use change results from a U.S. domestic soybean oil biodiesel consumption scenario across four economic models.[27] This report provides the latest assessment of indirect land-use changes potentially attributable to biofuels. The results are preliminary, and EPA's work on this topic is ongoing. The report showed a range of potential future international land use change impacts from the different models. Across these models, there is variation in both the magnitude of additional international cropland cultivated in response to increased consumption of soybean oil biodiesel in the United States and also variation in the mix of land covers converted to cropland. This area of scientific inquiry is a critical topic for further research. For a brief summary of the findings, see section 7.3.4. For more information, including MCE conclusions, see the

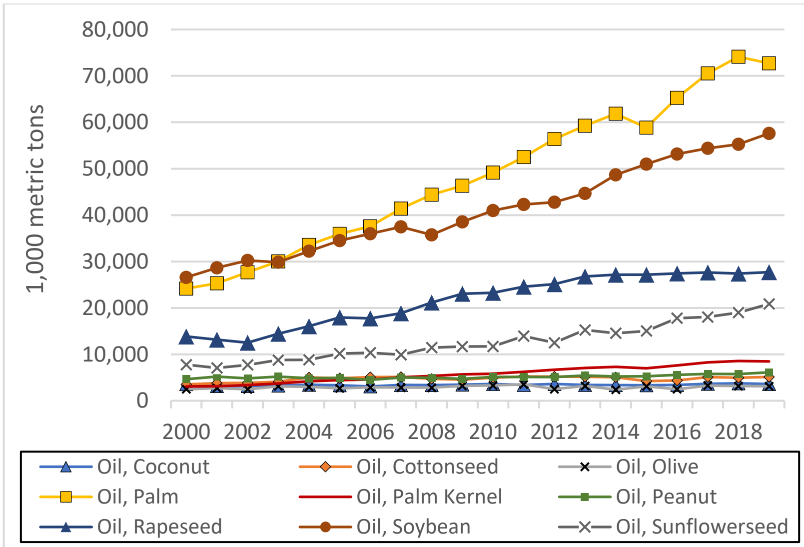
[27] See EPA-420-R-23-017

full report available on EPA's website. [28] EPA will continue this work through forthcoming relevant regulatory actions and to inform future reports.

16.5 Potential Effects on Palm Oil and Associated Environmental Effects

One recent change in land cover and land management that is of global concern is deforestation associated with palm oil production. Since deforestation associated with palm oil is not strictly a biodiesel phenomenon (see section 16.4.1.2), it is expanded on here as an area of potential concern.

Palm oil overtook soybean oil as the largest global source of vegetable oil in 2004, and its production continues to increase (Figure 16.19) (USDA FAS, 2019d). Globally, it is primarily used as a cooking oil or food ingredient (about 70% of use), though it is also used by the oleochemical industry (about 25%) and as a feedstock for biofuels (about 5%) such as biodiesel, renewable diesel, and jet fuel (USDA FAS, 2020e).



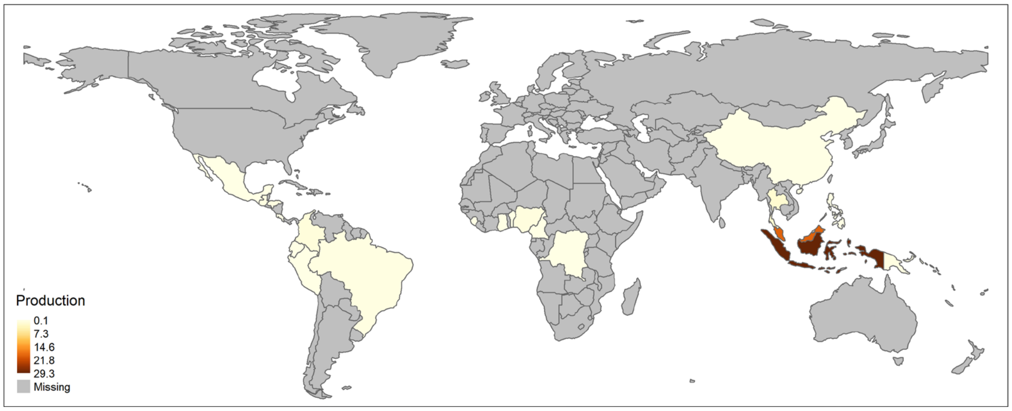
1 metric ton = 2,200 pounds

Figure 16.19. World vegetable oil production by commodity. Years are first year of market year .[29]

Palm oil is mostly produced in the Tropics, with almost 90% of global production from two countries - Indonesia and Malaysia (Figure 16.20). The United States imports a little more than 2% of global palm oil supplies (Figure 16.21), primarily for food and personal care and cleaning products. Exports from Indonesia (Figure 16.21a) and Malaysia (Figure 16.21b) are mostly to India, China, Pakistan, and various countries in the EU, where it is used primarily in food and livestock and, increasingly, in biofuel production (Murphy et al., 2021; Searle, 2019). Palm oil has been linked with a number of environmental impacts, including tropical deforestation, forest fires, methane emissions, and peat soil degradation. While palm oil is not produced in the United States, concerns have been raised about the possibility that the RFS Program is contributing to palm oil expansion and its related environmental effects (Lustgarten, 2018; U.S. EPA, 2018).

[28] See EPA-420-R-23-017

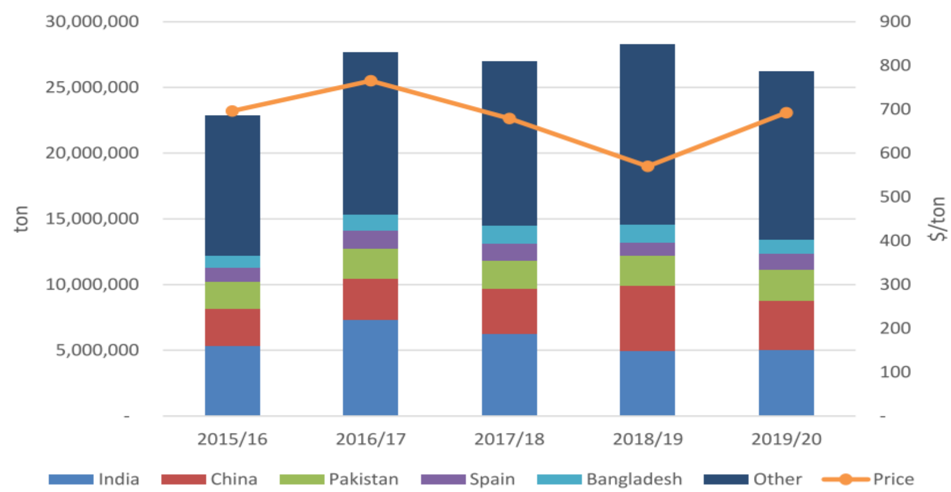
[29] The years listed in the figure are the first year in each market year. For example, the market year from October 2005 to September 2006 is reported as 2005 (USDA FAS, 2019d).



1 tonne = 2,200 pounds

Figure 16.20. Palm oil production by country in 2014 (million tonnes). Data from FAOSTAT,[30] vector and raster map from https://www.naturalearthdata.com.

a



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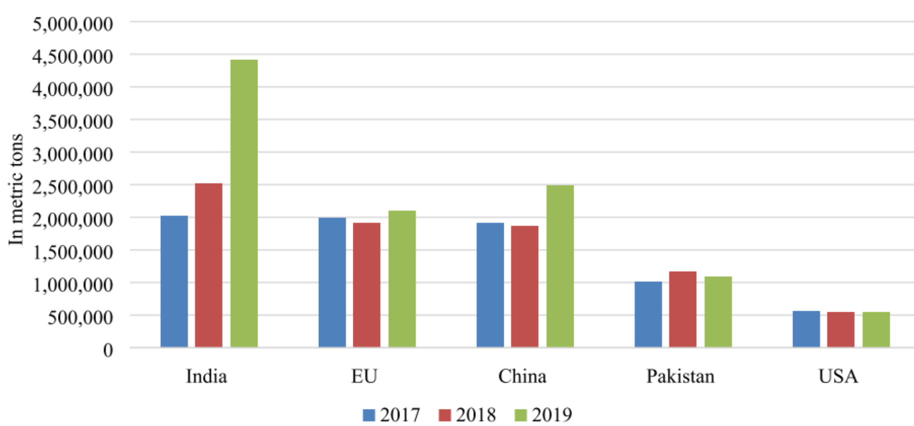

1 metric ton = 2,200 pounds

Figure 16.21. (a) Indonesian and (b) Malaysian palm oil exports by largest destinations (Indonesia export prices in Indonesia). Indonesia figure from USDA FAS (2021b). Malaysia figure from (USDA FAS, 2020d). Both figures are in metric tons, though are labeled differently in the source files.[31]

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[30] FAOSTAT data for palm oil production area by country, accessed December 15, 2018 (FAO, 2022).

[31] Exports to the U.S. were not reported separately in the underlying data for (a).

This section reviews literature on the environmental effects associated with palm oil and potential links between the RFS Program and palm oil production. It starts by reviewing the effects of palm oil production on land use change, deforestation, tropical peatland degradation, and other environmental effects. Although attribution of palm oil production to the RFS Program in particular, and U.S. biofuel consumption more broadly, is uncertain and unresolved, this section ends by reviewing literature that has looked into these attributional questions.

16.5.1 Land Use Change and Deforestation Associated with Palm Oil Production

Palm oil area in Indonesia and Malaysia increased six-fold between 1990 and 2017, from 6.0 million acres in 1990 to 47.3 million acres in 2017 (Figure 16.22).[32] This expansion was associated with environmental consequences including forest loss, peatland drainage, and biodiversity degradation (Koh et al., 2011), and there is ample remaining land for palm oil to continue expanding (Pirker et al., 2016) as demands continue to grow for its use in food, consumer goods, and biofuels. The increase was relatively linear over the period, aside from jumps in 2010 and 2017 associated primarily with large new and replanted plantations coming into production in Indonesia (USDA FAS, 2021b). The increase in production was the result of many factors, the RFS Program likely played a minor role in this overall trend (discussed more in section 16.5.3).

In Indonesia, palm oil was the leading driver of deforestation (23%) from 2001 to 2016 (Austin et al., 2019). From 1990 to 2010, approximately 50–80% of new palm oil plantations replaced forests (Gunarso et al., 2013; Koh and Wilcove, 2008), and this amount was approximately 90% in the Indonesian portion of Borneo (Carlson et al., 2013). Palm oil plantation area has continued to grow, but the share of new plantations coming from previously cleared land instead of primary forest has increased (Gaveau et al., 2016). As a result, the annual area of new plantations associated with deforestation has remained relatively stable at about 289,000 acres per year since 2005 (Figure 16.23), despite higher rates of annual palm oil expansion (Austin et al., 2017a). According to one of the most comprehensive and recent studies, the proportion of plantations replacing forests decreased from 54% during 1995–2000, to 18% during 2010–2015 (Figure 16.21) (Austin et al., 2017b). However, the total acreages of palm plantations increased, with larger and larger acreages from scrubland, swamp, and agricultural lands (Figure 16.21). The share of future plantation development in currently forested areas will depend on regional patterns (e.g., whether development shifts to the heavily forested province of Papua), regulatory structures (e.g., spatial plans for oil palm expansion developed by government planning and permit granting agencies) and other factors (Austin et al., 2017b).

[32] FAOSTAT data for oil palm fruit area harvested, accessed December 26, 2018 (FAO, 2022).

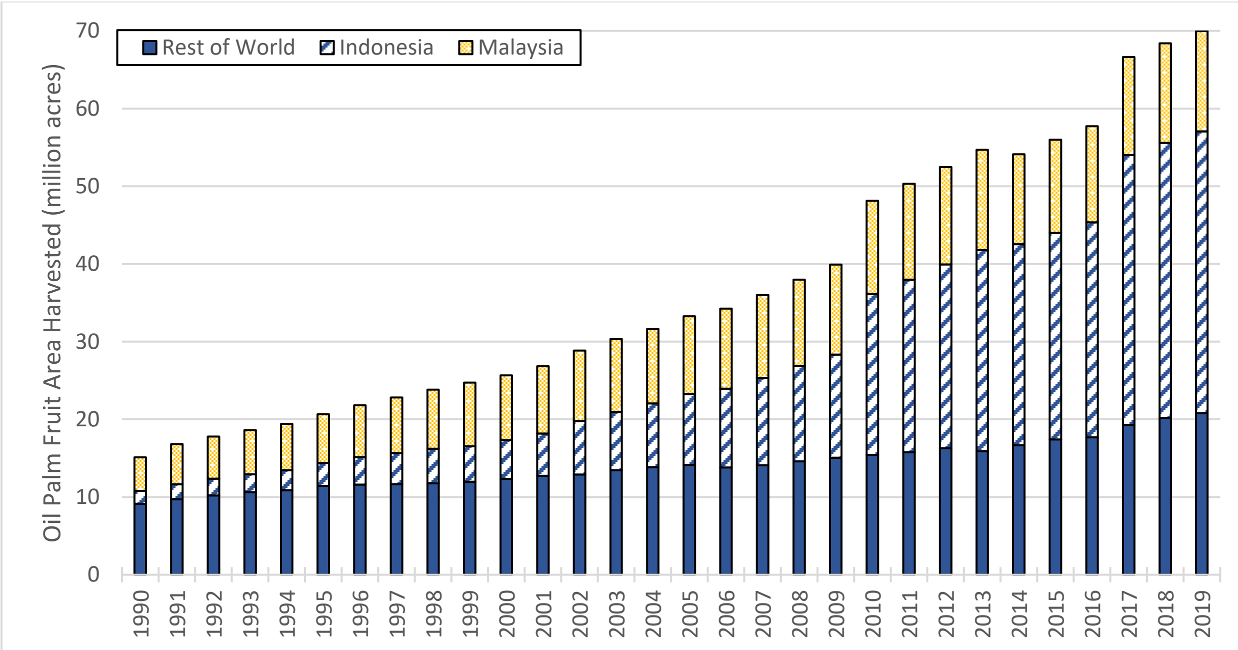
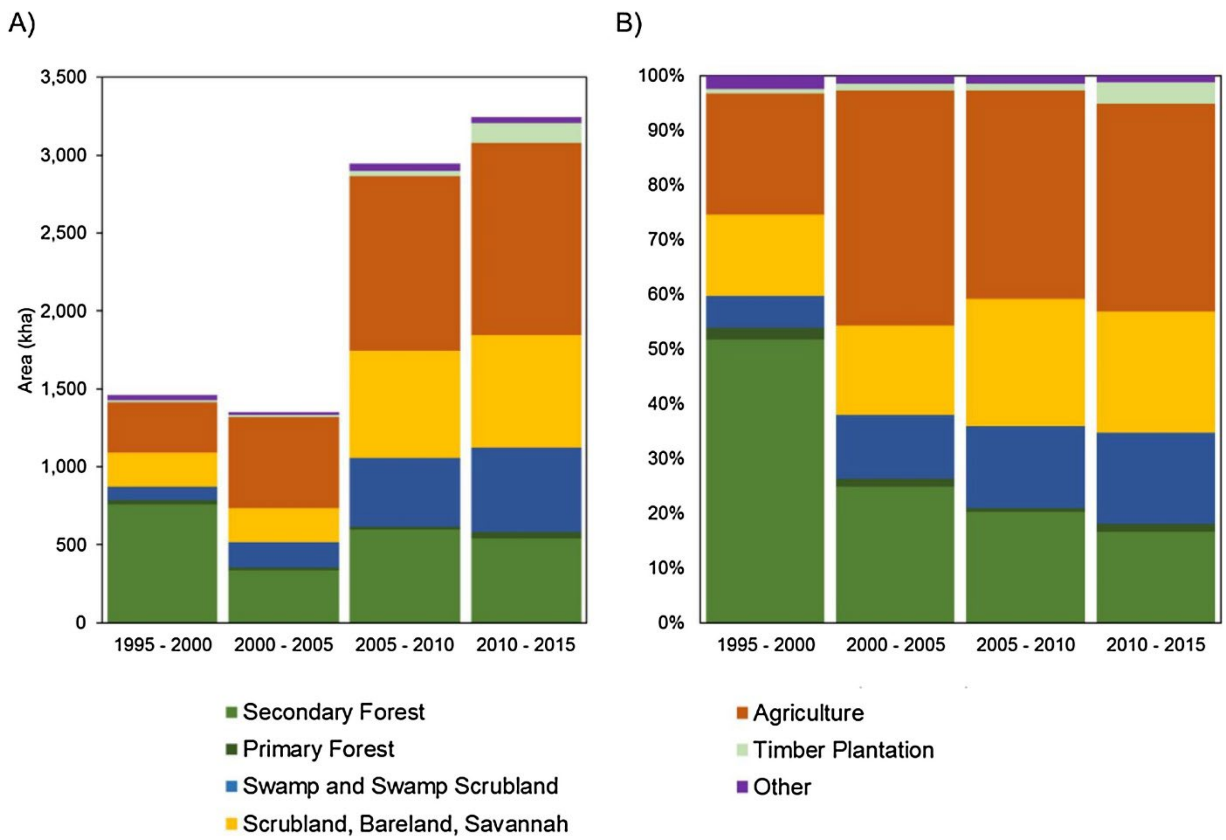


Figure 16.22. Palm oil area harvested (million acres) (FAO, 2022).



1 thousand hectares (kha) = 2,471 acres

Figure 16.23. (A) Area and (B) proportion of each land cover category converted to palm oil plantations in Indonesia for each time period, across all three study islands. Source: Austin et al. (2017b) (Creative Commons license, https://creativecommons.org/licenses/by-nc-nd/4.0/; no changes made).

Palm oil expansion in Malaysia, the second largest producer, has also been steadily increasing and associated with significant deforestation. Studies indicate that before 2002 over half of new plantations replaced forested land (Gunarso et al., 2013), but this dropped to approximately 30% between 2000 and 2010 (Gunarso et al., 2013; Koh et al., 2011).

Interpretation of remote sensing studies can be challenging and are a source for debate, especially when individual land parcels undergo multiple land use transitions over various time scales (Gaveau et al., 2016). While some studies have focused on short time scales with inconclusive results about the link between palm oil and deforestation (Gaveau et al., 2016), other studies looking at long time periods have concluded that almost all palm oil production in Indonesia or Malaysia is on land that was forested within the last 25 years (Vijay et al., 2016). Furthermore, some studies (Austin et al., 2017b) relied on land cover datasets that used a definition of forest that does not include land where forest is regenerating from a previous clearing. Thus, in addition to areas where palm oil directly replaced forests, it may also be cutting off areas of forest regeneration. Also, the amount of deforestation and forest degradation directly or indirectly associated with palm oil may be larger if palm oil expansion on non-forestland resulted in displacement of other agricultural activities (Gatto et al., 2015) or wildlife foraging (Luskin et al., 2017) to the forest frontier. While recent studies using high resolution imagery (Austin et al., 2019) have made progress illuminating the land uses following deforestation in certain regions, additional research, for example through causal analysis and simulation modeling, could provide more information about the extent, location and consequences of deforestation caused by palm oil.

16.5.2 Palm Oil Effects on Soil, Water, and Air Quality

Tropical peatlands are swampy, biodiverse forest and grassland ecosystems that store enormous amounts of organic carbon in their soils. In recent decades tropical peatlands have been drained and used to produce many commodities, including palm oil, timber, food crops, and others. In Indonesia and Malaysia, a share of deforestation has occurred at the expense of peat swamp forests, but non-peatland forests have also been cleared. Additionally, some peat swamp grasslands in these regions have been drained and brought into commercial use. Draining tropical peatlands is an area of particular concern, due to the large environmental effects from draining such areas. Tropical peatland is found across the equatorial tropics including Indonesia, Malaysia, Brazil, Western Africa, and Colombia (Xu et al., 2018). The incomplete decomposition of dead plant material under waterlogged, anaerobic conditions has led to the slow but progressive accumulation of thick deposits of carbon in peat over millennia, giving this ecosystem a very high carbon density (over 7,700 tons of carbon dioxide per acre in the soil). In addition to carbon storage, the peatland areas of Southeast Asia have numerous ecological and hydrological functions such as the regulation of water flow, which reduces flooding in rivers that run through peatlands, ensures water flow during drier periods, and affects regional climate through stabilization of evaporation rates. Remaining peat swamp forests have also become an increasingly important refuge for endangered animal species (Morrogh-Bernard et al., 2003) due to shrinking areas of lowland rain forests.

A portion of palm oil production occurs on drained tropical peatlands. In their natural state, peat swamps are unfavorable for agricultural production because the water table is above or near the surface throughout the year. Despite these challenging conditions, peat swamps have been exploited to make room for plantations for various reasons including diminishing easily accessible land areas in mineral soils, development of working techniques in tropical peat soil, ease of access, low relief, and ease of burning during dry periods (Fuller et al., 2011; Miettinen and Liew, 2010). By one estimate, 6% of tropical peatlands in Indonesia and Malaysia had been changed to palm oil plantations by the early 2000s (Koh et al., 2011) and that figure is certainly higher now. Between 1990 and 2015, 7.8 million hectares of peat swamp forests in Indonesia and Malaysia were converted through forest clearance and land drainage (Miettinen et al., 2016). From 2001 to 2016, approximately 26% of peat swamp deforestation was associated with palm oil expansion (Austin et al., 2019).

Palm oil expansion has also been linked with peat soil degradation in Malaysia. Estimates of the share of palm oil planted on peat soil in Malaysia vary by study and time period including 11% in 2002 (Koh et al., 2011), 13% in 2009 (Gunarso et al., 2013), 24% from 2005 to 2009 (Gunarso et al., 2013), 30% from 2003 to 2009 (Omar et al., 2010), and 40% from 2007 to 2010 (Miettinen and Liew, 2010).

Beyond the loss of soil carbon to the atmosphere, particularly from the draining of peatlands (Hooijer et al., 2012), there are other soil and water quality impacts of palm oil cultivation. These include erosion, and sediment and nutrient loadings to waterways (Guillaume et al., 2015; Hooijer et al., 2012; Babel et al., 2011). Clearing forest land increases soil erosion, which then in turn can increase sedimentation to waterways (Babel et al., 2011). The disposal of palm oil mill effluent as untreated waste into waterways has also degraded water quality in places (Mukherjee and Sovacool, 2014).

Changes in land use associated with palm oil development also affect fire activity and regional population exposure to smoke. Draining tropical wetlands dries out the landscape and increases the risk for large forest fires that burn forest biomass as well as the organic matter in the dried peat soil. As noted by Cattau et al. (2016), fire is a common tool for land conversion and management associated with palm oil production that has implications for air quality and human health in the region. A study looked specifically at the emissions and regional air quality impacts from fires in Indonesia from 2003 to 2013 and found that fires on drained peatlands within palm oil concessions were a major source of smoke emissions (Marlier et al., 2015).

One of the major impacts highlighted by Mukherjee and Sovacool (2014) is that of oil palm on deforestation and the resulting effect on wildlife habitat, which is due to both forest loss as well as fragmentation of forested areas. Much of the concern regarding the impacts on biodiversity are due to the biological richness of the forests in the region. Margono et al. (2014), in their study of primary forest cover loss, noted the high floral and faunal biodiversity contained in Indonesia's forests—including 10% of the world's plant species, 12% of mammal species, 16% of reptile-amphibian species, and 17% of bird species. In addition, a number of species are considered endemic, meaning they are unique to that geographic region. Koh and Wilcove (2008) compared populations of forest bird and forest butterfly species for several land use types and suggested that replacing primary forests and logged forests with oil palm plantations would decrease species richness of forest birds by 77% and 73% respectively. For mammals, much of the focus of biodiversity impacts has been on flagship or iconic species such as orangutans and tigers (Teoh, 2010)—where combined pressures from hunting, logging, forest fires, and both subsistence and plantation agriculture (such as palm oil) can lead to pressure on habitat loss and fragmentation.

Although approximately 90% of palm oil is produced in Indonesia and Malaysia, these two countries only represented 70% of palm oil area in 2019 (Figure 16.20, FAO, 2022), and other regions have been expanding their production. According to FAO, in 2019 Nigeria accounted for 14% of global palm oil area, Thailand for 3%, and a number of other countries accounted for 1–2% each (e.g., Colombia, Ghana, Ecuador, Brazil). Over 40 countries produce palm oil with differing rates of deforestation (Furumo and Aide, 2017; Vijay et al., 2016). Understanding the differences and interactions between palm oil production in different regions is an important area for further study given the potential environmental effects discussed above.

16.5.3 Attribution of Palm Oil Expansion to the RFS Program and U.S. Biofuel Consumption

Although palm oil biofuels do not have an approved pathway under the RFS Program, there are two potential mechanisms for the RFS Program to influence the level of palm oil production. First, as mentioned above U.S. EPA (2012) indicates that biofuels produced from palm oil feedstock do not satisfy the 20% GHG reduction requirement to qualify as renewable fuel under the RFS Program. However, some imported volumes of palm oil biofuels and volumes produced from imported palm oil that are exempt from the GHG reduction requirements, pursuant to the legacy provisions in 40 CFR 80.1403, are eligible to generate D6 RINs. Thus, the RFS Program conventional biofuel volume obligations may provide an incentive for exempted palm oil biofuel production either in the United States or through palm oil biofuels imported to the United States. However, as discussed in section 16.4.1.2 and the next section, no evidence was found that this mechanism has been a significant driver of palm oil biofuel production in Southeast Asia to date, with domestic policies and demand in Asia much more influential. Second, and perhaps more importantly, the RFS Program may increase the demand and price for other vegetable oils (e.g., soybean oil) that are used to produce biodiesel and renewable diesel and, to the extent palm oil is a substitute for those vegetable oils, the RFS Program may indirectly increase the demand and price for palm oil globally. The evidence related to this second mechanism is reviewed in this section in greater detail.

Economic principles suggest that, all else equal, higher renewable biodiesel volumes put upward pressure on the price of vegetable oil by increasing the demand for vegetable oil feedstock.[33] When soybean oil prices increase relative to other vegetable oils, consumers who can, may shift some of their consumption to other oils such as canola, corn, peanut, sunflower and palm oil. Although palm oil tends to be the lowest cost vegetable oil globally, local market prices vary. Also, palm oil is not a perfect substitute for food uses of soybean oil as it has different cooking and taste characteristics[34]. Quantifying these impacts is difficult due to the many confounding factors (e.g., population, income, weather, other market uses) that simultaneously influence the price and supply of soybean and palm oil. Further complicating the issue is that there are a number of potential steps in the causal chain from the RFS Program volume mandates to palm oil production. Based on a review of peer-reviewed literature, some but not all the steps in that chain have been evaluated quantitatively.

A recent study (Santeramo and Searle, 2019) looked at one of the steps in the causal chain by estimating the relationships between the price and supply of soybean oil and palm oil in the United States using country-level data from 1996 to 2016. They found a positive and statistically significant relationship between palm oil imports and the price of soybean oil whereby a 10% increase in the price of soybean oil would have caused a 12.3% increase in the supply of palm oil to the United States (standard error of 4.84%). They found a much weaker, but still statistically significant relationship between the U.S. supply of soybean oil relative to the price of soybean oil whereby a 10% increase in the price of soybean oil was associated with a 1.42% increase in soybean oil supply (standard error of 0.3%). The link between the price and supply of soybean oil may be relatively weak because the oil accounts for only about 33% of the value and 20% of the mass of each soybean, whereas the protein-rich meal, which is in stronger demand, makes up the majority of the value and mass. The authors mentioned that the U.S. supply of soybean oil is much larger than U.S. import of palm oil, suggesting that changes in U.S. soybean oil prices may have a relatively small impact on global palm oil production, but they did not calculate the absolute changes in the supply of each oil from a given change in soybean oil price.

For illustrative purposes, the U.S. EPA (2010) modeling estimates suggest that for every one billion gallon increase of U.S. soybean oil biodiesel production in 2022, the soybean oil price increases by

[33] Note that the RFS Program does not directly mandate vegetable oil biodiesel production, although it may lead to higher levels of vegetable oil biodiesel production than would otherwise be produced.
[34] See Chapter 7, section 7.3 for more information about challenges associated with substituting oils.

47% (FASOM) or 31% (FAPRI) depending on the model used. As discussed above Santeramo and Searle (2019) estimate that a 10% increase in the price of soybean oil causes a 12% ± 10% (range of two standard deviations) increase in the supply of palm oil to the United States. Thus, based on the FASOM and FAPRI estimates of the effect of soy biodiesel production on soybean oil prices, a one billion gallon increase in soybean oil biodiesel production may increase palm oil imports by 57% ± 45%. According to USDA data, the U.S. imported 2.3 million tons of palm oil in 2021.[35] Thus, a 57% ± 45% increase in palm oil imports relative to 2021 levels would be approximately 1.3 ± 1.0 million tons of palm oil imports based on FASOM and 0.9 ± 0.7 million tons based on FAPRI. The FASOM and FAPRI modeling assumed approximately 7.7 pounds of soybean oil per gallon of biodiesel, or 3.9 million tons of soybean oil per billion gallons of biodiesel. Putting this all together, the illustrative estimate suggests that 34% ± 27% of soybean oil used for biodiesel may be backfilled with palm oil imports based on the FASOM price effect estimate, and 22% ± 18% may be backfilled based on the FAPRI price effect estimate.

A number of modeling studies have estimated the effect of U.S. biofuel (ethanol, biodiesel, and other biofuels) consumption on palm oil production and land use in Southeast Asia. Cui and Martin (2017) derived a partial equilibrium model to investigate the market effects of biodiesel expansion on related energy and vegetable oil markets. This model, calibrated to 2014 data, considers two regions (United States and rest of world) and two vegetable oils (soy and palm). Interpreting results from this model must be done with caution because it does not consider the important roles of other vegetable oils in global markets (Taheripour and Tyner, 2020), or important domestic policies outside the United States, but it was developed for the express purpose of exploring interactions between biodiesel, soybean oil, and palm oil markets. The modeling includes assumptions about the prices and supply relationships (elasticities) between soy and palm oil, which they tested through Monte Carlo simulation. Based on this model's assumptions and parameters, increased use of soy oil in biodiesel production in the United States would impact world vegetable oil markets and palm oil would fill most of the gap left by diversion of soy oil to biodiesel. Their result was consistent across different elasticity values for demand as well as substitutability between soy versus palm oil. Modeling of scenarios that evaluated different levels of soy biodiesel production (1.55, 2.0, and 3.4 billion gallons) estimated that the soybean oil feedstock for biodiesel production would be sourced[36] only 13-15% from increased soybean oil production and the rest (85-87%) through diverting soybean oil from other uses to biodiesel. Only 6% of the resulting gap in vegetable oil supply would be filled through increased palm oil production with the rest coming from increased production or reduced demand for other vegetable oils. Soybean end use data from USDA do

[35] USDA PS&D Oilseeds Dataset: https://apps.fas.usda.gov/psdonline/downloads/psd_oilseeds_csv.zip (downloaded 7/12/22, downloaded 7/21/22). Includes both "Oil, Palm" and "Oil, Palm Kernel."
[36] Based on calculations from table 6 in Cui and Martin (2017) evaluating changes from scenarios 2 to 1 and 3 to 1 divided by change in "soybean oils in biodiesel production" for the same scenarios.

not suggest that U.S. soybean oil exports were reduced and diverted to domestic uses, as exports have remained steady aside from market year 2020/2021 (section 3.2.1.1 and Figure 3.14).

U.S. EPA (2010) used the FAPRI-CARD model to estimate international agricultural responses to the RFS Program. Comparing the statutory RFS2 volumes (36 billion gallons of biofuel by 2022) to a reference case with AEO 2007 biofuel volumes (13.6 billion gallons in 2022), U.S. EPA (2010) estimated a roughly 14,000-ton decrease in palm oil production (-0.02%) with the RFS2. A case that only increased U.S. soybean oil biodiesel by 540 million gallons in 2022 (971 million gallons observed in 2019, Table 2.1) estimated a 161,000-ton increase in palm oil production (0.23% globally). In that scenario, soybean oil production increased 593,000 tons, such that the palm oil production response was approximately 27% of the soybean response on a mass basis. However, the increase in palm oil production represents only 8% of the soybean oil needed to produce additional biodiesel in this scenario. In this analysis, palm oil area increased by 77,000 acres (40,000 acres in Malaysia and 30,000 acres in Indonesia), or 143,000 acres of palm oil expansion per billion gallons of U.S. soy biodiesel consumption. For Malaysia and Indonesia, 40,000 acres and 30,000 acres represent 0.3% and 0.1%, respectively, of total palm oil areas in these two countries in 2019.

More recently, Taheripour and Tyner (2020) used the GTAP-BIO model to simulate the effect of the RFS Program ethanol and biodiesel mandates on palm oil in Southeast Asia and found that the production of biofuels in the U.S. generates some land use effects in Malaysia and Indonesia due to market-mediated responses. The estimated responses were rather small—the combined effect of 15 billion gallons of corn ethanol and 2 billion gallons of soybean oil biodiesel were estimated to increase cropland area in Malaysia and Indonesia by less than 150,000 acres, or 0.5% of the observed cropland expansion in those countries from 2000 to 2016. However, as noted in a recent NASEM report, the way this model is currently configured does not allow for conversion of unmanaged forest or grassland to agricultural production (NASEM, 2022; Plevin et al., 2022). The authors evaluated a range of assumptions about the flexibility of substitution (elasticity) between vegetable oils given the relatively small amount of empirical evidence in this area. They found that the inclusion of other potential sources of substitution (other vegetable oils and fats), and the choice of elasticity value used in model simulations, had a large influence on the resultant palm oil demand. This, in turn, has important implications for interpreting other model outputs including estimates of land use change and simulated "backfill."

In summary, available research suggests that U.S. crop-based biofuel production may have had some effect on palm oil and cropland area in Southeast Asia through the indirect effect on global vegetable oil markets; and thus, potentially affected critical peat swamp forest ecosystems. The size of this effect is uncertain due to the complex causal chain involved and the relatively limited body of research, but available estimates suggest an impact of <1% increase in overall palm oil acreages due just to the U.S. biofuel volumes. Part of the uncertainty behind estimated land cover and land management change arises from the different sets of assumptions and parameterizations used in different model-based studies. Some models may have features that are useful when attributing crop area changes to the RFS Program, but also have characteristics that are not ideal for estimating changes in land cover and land management. Especially important in determining model-based results is the choice of price elasticities used in the model. For example, Drabik et al. (2014) estimate a soybean oil price increase of 44%, which is close to the values from FASOM (47%) and FAPRI (31%). Considering the uncertainties associated with simulating land cover and land management changes, a model-based estimate of changes in palm oil area should be seen as a single piece of evidence and not relied on by itself. In general, a more useful approach is to view each model result as part of an ensemble of results and then consider the ensemble alongside other types of evidence. Another area of uncertainty, as discussed above, is regarding how relatively small effects on palm oil production and production practices can have large environmental consequences due to the sensitivity of the potential source ecosystems. There is also uncertainty and a wide range of estimates as to what percentage of soybean oil used for biodiesel production may be or have been backfilled with additional palm oil production. The estimates reviewed suggest this soybean oil backfill percentage may have been approximately 6–14% of the soybean oil diverted to biofuels. However, these studies are limited in their ability to attribute palm oil changes to the RFS Program because they either did not directly study the effects of the RFS Program or did not consider data over the relevant time period (2000 to present). Also, these estimates may not apply to the future as global vegetable oil market conditions change. One mechanism that causes ripples in global vegetable oil markets and palm oil demand is vacillating trade policy. In particular, the EU and United States have attempted to slow palm oil imports through a variety of regulations (Arief et al., 2020; USDA FAS, 2018b) that have had differing enforcement periods. More research on substitution flexibilities between vegetable oils in biodiesel production and domestic food consumption, the role of governmental and other nonmarket drivers in determining the effects of palm oil production, and other factors would help to increase confidence in quantitative estimates on this topic.

16.6 Synthesis

16.6.1 Chapter Conclusions

- Attribution of international effects from the RFS Program remains challenging due to complex interrelationships among other major drivers of observed change. There are relatively few studies on this topic specifically, though many on international effects from biofuels more generally, and analyses are impeded by inconsistent data and large uncertainties, and modeling specifications and assumptions used in empirical analysis of these issues.

- This chapter considers three causal chains whereby the RFS Program may lead to environmental effects in foreign countries. First, it is considered that the RFS Program may increase biofuel imports from other countries to the United States, and that such imports may be associated with environmental effects related to producing the imported biofuels. Second, the market-mediated effects of the RFS Program on international crop and commodity prices and production, and related environmental effects are considered. Third, it is considered that, by supporting the growth of the U.S. biofuel industry, the RFS Program may increase biofuel exports from the United States to other countries, with potential environmental effects/ benefits for the receiving countries.
- International environmental effects that are clearly attributable to the RFS Program due to U.S. ethanol and biodiesel imports could not be confidently quantified. The lack of empirical evidence to support causal linkages between the RFS and international environmental effects does not necessarily rule out international effects attributable to the RFS Program.
- Imports of biofuels to the U.S.—one of the mechanisms for international effects—have fallen drastically since peaking before the RFS Program in 2004–2006. Evidence supports attribution to the RFS Program for some biodiesel imports since 2007. The value of advanced biofuel (D5) RINs was among many factors that supported sugarcane ethanol imports from Brazil since mid-2010.
- The hypothesis that U.S. demand for sugarcane ethanol attributable to the RFS Program played the dominant role in the observed changes in Brazil's ethanol production and associated environmental effects is not supported by available evidence. Ethanol production in Brazil has been supported by domestic policies in Brazil for decades.
- U.S. ethanol production that exceeds domestic demand is exported to more than 70 nations around the globe, and since 2008, the United States has been a net exporter of biofuel (ethanol + biodiesel) on an annual basis. However, the share of exports attributable to the RFS Program is uncertain. To the degree that the RFS Program encouraged investments that generated surplus ethanol for export after reaching the blending wall, the RFS Program contributed to the international effects associated with net U.S. exports, which could be environmentally beneficial for importing nations.
- Two-way interannual ethanol trade with Brazil (i.e., U.S. corn ethanol and Brazilian sugarcane ethanol) is likely driven by tariffs, extreme weather events, seasonal differences in the agricultural cycles, and regulatory standards (for example, the CA-LCFS could encourage

imports of ethanol with lower carbon intensities and exports of ethanol with higher carbon intensities ratings), between the two nations. However, the relative contributions of these causal factors to the extent of this two-way trade have not been resolved in the scientific literature.

- A portion of the gross biodiesel imports during 2012–2019, averaging approximately 295 million gallons per year, are reasonably attributed at least in part to the RFS Program. However, many factors impact biodiesel trade, and sources of import (i.e., countries) are diverse and irregular, each affected by their own domestic policies which are difficult to assess with current models and data.
- While this chapter concludes that more research is needed on the effect of the RFS Program on international cropland areas and environmental effects more broadly, combining published simulation modeling estimates of the land use change effects of biofuels (section 16.3.3 for corn ethanol) and the estimates for the effect of the RFS Program on corn ethanol (Chapter 6) yields an illustrative range of the effect of the RFS Program on non-U.S. cropland area of 0 to 1.6 million acres. The estimated effect of the RFS Program does not yet include effects on soy biodiesel. As more data become available and are analyzed, historical relationships among U.S. biofuel policies, production, trade, environmental indicators, and other variables may be clarified and uncertainties reduced. Review of potential international effects of the RFS Program associated with biodiesel imports, and on global cropland more broadly, finds that quantification of effects is uncertain but could be significant and merits further research. The relationship of the RFS Program with palm oil expansion, and the environmental costs and benefits of two-way trade, merit further study.

16.6.2 Conclusions Compared with the RTC2

In general, the conclusions from this report on international effects are similar to those from RtC2, although the analysis has been extended in this report to cover topics that were not addressed in RtC2. Compared to RtC2, this chapter includes more examination of attribution of international biofuel imports to the U.S. and international environmental effects for specific countries and biofuels. As stated in RtC2, "Quantification and causal attribution of land use change and international environmental impacts due to biofuel production remain uncertain and undetermined." However, additional conclusions have been drawn related to attribution. The RFS Program provided incentives for ethanol imports from Brazil but import volumes are better explained by other factors and on net, imports are increasingly outweighed by U.S. exports to Brazil. Furthermore, there is little evidence linking expanded sugarcane production to the RFS Program because Brazil's sugarcane production is highly influenced by domestic policies in Brazil among other factors (e.g., section 16.3.2.2). This chapter finds that the RFS Program could induce substitution effects in vegetable oil markets that would increase palm oil production in Southeast Asia and other regions. Finally, unlike the last report, it was observed that ethanol exports from the United States and two-way trade may have environmental benefits in other countries that merit further study.

16.6.3 Uncertainties and Limitations

- Many factors contribute to high uncertainty regarding quantitatively estimating international effects of the RFS Program; including but not limited to, differences in how contributing nations record and report volumes of biofuels; inconsistencies between global land cover and land management datasets; reliance on simulation models with limited validation (esp. for changes in land use) and varying specifications and assumptions (especially the choice of elasticity parameterizations), and different levels of complexity; and fluctuating policies and other factors that confound simple statistical analyses.
- Uncertainties are especially large for estimates of indirect or induced impacts of U.S. biofuel policies on tropical forests and areas of high conservation value, such as in the Amazon and Southeast Asia, given the potential for very large environmental effects from small areal changes in these ecosystems.
- International markets are opportunistic, with market shares shifting frequently among exporting nations (Dutta, 2020). Trade is based on opportunities to maximize profits or minimize losses and it is influenced by complex interactions among internal and external markets for ethanol, coproducts (including sugar, distillers grains, feed corn), substitution options (including petroleum products), exchange rates, and the infrastructure and capacities available for transporting corresponding commodities within relatively short time frames (Dutta, 2020; Katrakilidis et al., 2015; Rajcaniova et al., 2013). These factors were not thoroughly analyzed in this report.
- Ample studies exist pertaining to the trade of global commodities (including biofuel feedstocks) and there are several statistical studies of relationships between international commodity prices, oil, and ethanol (Chen and Saghaian, 2015; Katrakilidis et al., 2015; Natanelov et al., 2013; Ciaian and Kancs, 2011). An exhaustive assessment of these findings and the differences in their methodologies is beyond the scope of this report. Research continues to examine these relationships and analyses are improving as more data become available.

- Likely future effects are uncertain and strongly dependent on trade deals, policies in other countries, currency exchange rates, and other factors. These factors are subject to change and are difficult to predict.
- The potential beneficial effects of biofuel policies and sustainability requirements imposed by international biofuel markets on large, established agricultural commodity production systems (sugarcane, palm oil), were not investigated as they fall outside the scope of this report.

16.6.4 Research Recommendations

- An important research objective needed for assessing potential international impacts of the RFS Program is to develop more robust estimates of land area changes overseas, combining modeled and empirical data, associated with U.S. biofuel policies and trade that are separated from domestic policies in other countries.
- The implications of differential environmental effects associated with production in the United States versus Brazil are another area that merits more study.
- Further analysis of where, when, and why land managed for agriculture is changing, and understanding how biofuel policy may interact with the causes for such increases (e.g., changes in land use associated with changes in U.S. exports of commodities such as corn and soybeans), are important topics to address uncertainties associated with current land use change models. For example, in the case of Brazil, environmental effects of a share of ethanol production being exported will depend on the source(s)—sugar plantations and mills and how they are managed—as well as other contextual variables.
- Given uncertainties surrounding impacts of biofuel production on tropical forests and areas of high conservation value such as in the Amazon and parts of Southeast Asia, any potential role the RFS Program might have on these regions represents a priority for further research.
- One of the primary causes of uncertainty in researching international effects of the RFS Program is inconsistent and incomplete datasets. Expanding and improving the current database of biofuel trade flows and associated feedstocks and coproducts would be a good investment for understanding future effects of the RFS Program on international biofuel trade and associated effects. For example, emerging markets for new ethanol products and coproducts merit more analysis in the next report.
- Past studies on the international effects of the RFS Program are heavily reliant on limited and uncertain data and simulation modeling. Data-intensive studies on the connections between international biofuel trade and the resultant changes in land cover and land management, as

well as simulation modeling with different specifications and assumptions, have high potential to shed new light in this area.

- Patterns of two-way trade with Brazil merit research to better estimate the influence of the RFS Program and to identify if and how environmental costs could be reduced, and benefits maximized, through more strategic and efficient trade mechanisms.
- Regardless of the direction of trade, significant volumes of biofuels are being produced and exchanged between numerous countries around the world. It is important to measure the environmental effects of trade of biofuel feedstocks and fuels and identify opportunities to maximize net global benefits.

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17. Compilation of Key Findings

17.1 Chapter 2: Scope of the Report[1]

- The EISA Section 204 reports are intended to examine the "impacts to date" and "likely future effects" of the RFS Program. This may include contextual information on the environmental or resource conservation impacts of biofuel production or agricultural activities more generally, but those subjects are not the intended focus of this report series.
- The authors interpret the impacts to date as the historical effects of the RFS Program from 2005 to about 2020, and interpret the likely future as what may be considered relatively likely to occur over the near term, to approximately 2025, considering current market and technology conditions and trends.
- There were 17 biofuels screened for potential inclusion in the RtC3 based on unique combinations of fuel, feedstock, and production region (e.g., biodiesel-soybean-Argentina). This report focuses on any biofuels that dominated the total U.S. pool from 2005 to 2020 to examine those that potentially have a material effect on the environment. This yielded four biofuels for emphasis in the RtC3: (1) domestic corn ethanol from corn starch, (2) domestic biodiesel from soybean oil, (3) domestic biodiesel from fats, oils, and greases (FOGs), and (4) imported ethanol from Brazilian sugarcane.
- Although these four biofuels are the focus of the RtC3, other biofuels (biodiesel and renewable diesel from other feedstocks, cellulosic biofuels, algae-based fuels, etc.) and considerations are also discussed where appropriate.
- All of the environmental and resource conservation effects specified in EISA Section 204 are included. Effects omitted from EISA Section 204 or covered elsewhere in EISA (e.g., greenhouse gases [GHGs] are addressed in Section 201) are not a focus of this report.

17.2 Chapter 3: Biofuel Supply Chain

- The supply chain of the major biofuels in the RtC3 involve feedstock production (corn and soybean) and collection (fats, oils, and greases [FOGs]), logistics and transport to biorefineries, biofuel production, biofuel logistics, blending and distribution to point of dispensation, and biofuel end use.

[1] Note there are no Key Findings from Chapter 1, which is just the Introduction.

- During feedstock production, fertilizers and pesticides are used for corn and soybean cultivation. On a per acre basis, corn typically uses more nitrogen and phosphorus fertilizer than many other crops, including soybean. Corn grown in rotation with soybean requires less nitrogen fertilizer than when not in rotation.
- Adoption of conservation practices has been steadily increasing since the 1990s. From the most recent estimates available, conservation tillage is practiced on 65% of corn and 70% of soybean acres, while other conservation practices have been less widely adopted (e.g., cover crops are approximately 5–6% of cropland, but are slowly increasing). The extent to which conservation practices are used from one year to another on individual lands is largely unknown.
- Although in early years of the biofuels industry wet- and dry-mill processing were comparable in magnitude, dry-mill operations now make up 91% of the ethanol biorefineries. The production of distillers' grains (DGs) for animal feed through either process is a significant coproduct from ethanol production, which mitigates the effect of ethanol demand on demand for corn which is also used for animal feed.
- FOGs are collected from many different types of operations as a waste product or coproduct (e.g., food-processing or livestock production) and typically purified at rendering facilities into useful commodities that are then processed into fuel or for other purposes.
- Ethanol refineries are concentrated in the Midwest nearer to the major feedstock (corn), whereas biodiesel refineries are smaller and more distributed due to the more diverse number and distribution of feedstocks (e.g., soybean oil, FOGs).
- In the early years of ethanol blending, ethanol was "splash blended" with finished gasoline at the gasoline terminal. For at least the last decade ethanol is now blended into gasoline blendstocks which cannot be legally sold at the pump without the addition of an oxygenate such as ethanol.
- Although the number of E15, E85, and B20 stations are increasing in the United States, they remain a small fraction of total fuel stations and thus are not as widely available as E10 or diesel.

17.3 Chapter 4: Biofuels and Agricultural Markets

- Renewable Identification Number (RIN) prices for renewable (D6) fuels provide evidence that the Renewable Fuel Standard (RFS) Program increased U.S. consumption of renewable biofuels in 2009 (and late 2008) and from 2013 to 2019.

- Advanced (D5), biomass-based diesel (D4), and cellulosic (D3) RIN prices provide evidence that the RFS2 increased U.S. consumption of advanced, biomass-based diesel and cellulosic biofuels in every year of RFS2 for which standards had been set for these fuels (i.e., starting in 2010).
- Studies estimated that the RFS Program could increase corn ethanol production between 0 and 5 billion gallons under scenarios with relatively high oil prices (greater than $60 per barrel in 2018 prices). Oil prices were greater than $60 per barrel for much of the period of growth in the corn ethanol industry.
- A meta-analysis of studies published between 2007 and 2014 on the impact of biofuels estimated that for every billion-gallon increase in corn ethanol production between 2010 and 2019, corn prices would increase about 3–5%.
- Studies of the impact of RFS2 estimated that the Program could increase biomass-based diesel consumption 0.6–1.1 gallon for every gallon in the biomass-based diesel volume obligations. This is equivalent to an increase in biomass-based diesel consumption of 0.4–0.7 gallons for every ethanol equivalent gallon in the advanced volume obligations.
- Studies of the impact of biofuels estimated that for every billion-gallon increase in biomass-based diesel production, soybean prices increased 1.8–8.9%.
- RFS2 was estimated to have a limited impact on soybean meal production (decrease of 1.2% per billion gallons of biodiesel) and put downward pressure on soybean meal prices (decrease of 1–4.1% per billion gallons of biodiesel).
- On average, production decreases in beef, milk, pork, and poultry were estimated to be less than 0.5% per billion gallons of corn ethanol. Producer price increases in these livestock commodities were estimated to be less than 1 cent per pound per billion gallons of corn ethanol. The impact on consumer prices would likely be less than this.
- On average, an estimated 1 million acres of additional corn would be produced, and cropland would expand an estimated 0.7 million acres for each billion-gallon increase in corn ethanol production.

17.4 Chapter 5: Domestic Land Cover and Land Management

- After decades of decline, increases in cultivated cropland have been recorded in multiple federal datasets, using a variety of methodologies, following the 2007 to 2012 period. This increase ranges from 6 to 10 million acres. Despite these recent increases, the extent of current cultivated crop acreage for this period is still below historic levels of crop cultivation.

- Based on the 2012, 2015, and 2017 National Resource Inventory (NRI), there has been a steady increase in agricultural intensity from 2007 to 2017 with a 10 million-acre increase in cultivated cropland coinciding with a 15 million-acre decline in perennially managed land (i.e., sum of lands in Conservation Reserve Program [CRP], pasture, and noncultivated cropland). This increase in cultivated cropland was largely driven by a net 26.5 million-acre increase in corn and soy with small grains and hay in rotation decreasing 16.5 million acres.
- More than half of the corn and soybean increase has largely come from other cultivated cropland (56%), while the rest has come from approximately equal proportions of pasture (13%), noncultivated cropland (20%), and CRP (11%). Corn likely has larger environmental effects than hay, pasture, and other crop types because corn typically uses more fertilizer, pesticides, and other inputs than other crops.
- Many of these changes are taking place throughout the Midwest, with hotspots in northern Missouri, eastern Nebraska, the Dakotas, Kansas, and parts of Wisconsin.
- Based on both the National Agricultural Statistics Service (NASS) and NRI, crop production is becoming less diverse in the United States as cultivated cropland, besides that of the increasing corn/soy acreage, continued to decline from 2000 to present.
- These changes in cultivated cropland acreage have coincided with increased corn and soybean yields and increasing adoption of a variety of best management practices like conservation and no-till practices.
- After short-term disruptions from weather and trade disputes with China, the USDA Long Term Agricultural Projections (LTAP) suggest that corn acreage and corn used for ethanol will remain relatively stable from 2020 to 2025, declining slightly thereafter. This projected decline is driven by increases in fuel efficiency decreasing total gasoline consumption, increasing crop yields, and E10 blend wall issues further exacerbated by slow growth in E15 and E85 consumption. Likewise, soybean acreage is projected to remain stable due to increased yields meeting both domestic and international demand, especially to meet growing international meat consumption.

17.5 Chapter 6: Attribution: Corn Ethanol and Corn

- Many factors have impacted ethanol production and consumption in the United States historically, including higher prices of oil and gasoline, the replacement of methyl tert-butyl ether (MTBE) in reformulated gasoline (RFG) areas, the RFS Program, the Volumetric Ethanol Excise Tax Credit (VEETC), the octane value of ethanol, state programs, and air emission standards.

- The period of rapid growth in the ethanol industry was from 2002 to 2010, and nearly 40% of the increase in ethanol consumption had already occurred by 2006 (the first year of the RFS Program, RFS1), and over 90% of the increase had already occurred by 2010 (the first year of the RFS2).
- Because the factors that affect ethanol production and consumption—including the RFS Program—change through time, so too does the estimated effect of the RFS Program on ethanol production and consumption.
- Evidence from simulation models, observed RIN prices, production exceeding consumption from the RFS standards, and other sources suggest that from 2006 to 2011 the RFS Program—in isolation—accounted for 0–1 billion gallons of ethanol per year, primarily by encouraging market growth and capital investment from the Energy Independence and Security Act (EISA) and to a lesser extent by stabilizing demand during the Great Recession of 2008–2009. In other years of this period, the RFS Program is estimated to have had no effect on ethanol production, with other factors having more influence.
- The synthesis of evidence suggests a dynamic range of effects from the RFS Program from 2012 to 2019 as well, with the largest effect in 2016 (0–2.1 billion gallons per year) primarily due to the RFS Program supporting the industry after other factors had either phased out (e.g., VEETC, MTBE) or diminished in effect (e.g., high oil prices).
- In sum over the entire period assessed, the RtC3 concludes that 0–9% of corn ethanol production and consumption is likely attributable to the RFS Program historically. Lower estimated effects of the RFS Program occur if the effect on market certainty is not considered, or if MTBE replacement by ethanol and transitions to match blending are assumed to be independent of, but coincident with, the RFS Program; larger effects occur if market certainty is included, or if these other factors are omitted or ascribed to the RFS Program.
- Combining these estimated volumes attributable to the RFS Program with literature reviews and a recent statistical analysis suggests that the RFS Program may be attributable for cropland expansion of zero to 1.9 ± 0.9 million acres, and additional acres of corn of zero to 3.5 ± 1.0 million acres, with the largest potential effect estimated in 2016.
- These best available estimates from econometrics of observed trends are consistent with other econometric studies once appropriate adjustments are made and are consistent with estimates from simulation models.
- The likely future effect of the RFS Program was estimated by EPA in the Final Set Rule on June 14, 2023, which estimated 787 million gallons of corn ethanol consumption in 2025 to

be due to the RFS Program, potentially inducing up to 0.46 million acres of cropland expansion. These estimates are highly uncertain, due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes.

- Uncertainties in the estimated effect of the RFS Program on ethanol production remain, including the effect of the RFS Program in establishing market certainty before the mandates were in full effect, the cost, ability, and willingness of refiners to switch back to producing finished gasoline without ethanol if blending ethanol were no longer economical, and others. However, these factors are difficult to quantify with current tools available.
- The RFS Program created a guaranteed market demand for biofuels in the United States that certainly could have helped drive much of the increase in ethanol production and consumption in the United States. However, as events played out, non-RFS factors that also affect ethanol production and consumption (e.g., oil prices, octane value, MTBE bans, tax incentives, state programs) were favorable, and appear to sufficiently explain much or all of the increase in ethanol production and consumption historically in the United States.
- A modest effect from the RFS Program on corn ethanol does not preclude a larger effect on other biofuels (see Chapter 7) as the RFS Program affects many biofuels in addition to corn ethanol.

17.6 Chapter 7: Attribution: Biodiesel and Renewable Diesel

- Some of the same factors that drove ethanol trends in production and consumption in the United States contributed to biodiesel and renewable diesel trends, including federal tax credits, state incentives, and high petroleum prices and low agricultural commodity prices, especially in the early period of growth.
- There is much less information on biodiesel and renewable diesel compared with ethanol, and very few retrospective analyses on the relationship between the RFS Program and biodiesel and renewable diesel production. Therefore, this chapter does not provide a quantitative estimate of the amount of biodiesel and land attributable to the RFS Program in the RtC3 as was done in Chapter 6 for corn ethanol.
- The evidence available suggests that the RFS Program was binding on biodiesel and renewable diesel for the entire period of the RFS2 assessed (2010 to 2019). It does not appear that there was a binding effect prior to this given the lack of an individual biomass-based diesel (BBD) standard from 2006 to 2009 under the RFS1 (2006–2008) or for the first year of the RFS2 (2009), and low RIN prices during these years where data are available (2008–2009).

- Overall, biodiesel and renewable diesel production has been much more strongly dependent on federal and state policies (grants, tax incentives, income tax credits, RIN values, etc.) than has ethanol. The Biodiesel Tax Credit (BTC) and the RFS2 played particularly important roles. A different set of incentives drove production in the early phases compared to more recent years.
- It is not possible in the RtC3 to derive a robust estimate of the volume of soybean biodiesel specifically attributable to the RFS Program. However, economic models suggest that the RFS Program could increase biomass-based diesel consumption 0.6–1.1 billion gallons for every billion gallon increase in the biomass-based diesel volume obligations; and comparison of state and federal mandates suggest that while roughly 0–30% of biodiesel consumption may be due to state programs (mandates and low carbon programs like the Low Carbon Fuel Standard), the remaining 70–100% may be attributable to a combination of other factors, primarily the RFS Program and the BTC. The effects of the RFS Program cannot be isolated at this time for the historical period because most studies do not separate the RFS from other important factors that occurred at the same time such as the BTC and state programs.
- In addition to domestic effects, the RFS Program incentivized the import of foreign biodiesel from different sources in different years (e.g., Argentinian soybean biodiesel, Southeast Asian palm oil). These direct volumes are small on a relative basis but could have important local effects overseas, and diversion of any vegetable oil toward biofuels could have indirect effects on these markets that are difficult to estimate.
- While this and other chapters have discussed the substitutability of different feedstocks into the food, feed, and fuel industries, the authors of this chapter are not aware of sufficiently rigorous studies that have addressed the impact of increasing demand for qualifying feedstocks (such as fats/oils/greases [FOGs] or soybean oil) for biodiesel and renewable diesel production on commodities that may be used as substitutes in other industries (such as other vegetable oils, including palm oil).
- The likely future effect of the RFS Program was estimated by EPA in the Final Set Rule issued on June 21, 2023, which estimated 1,484 million gallons of soybean biodiesel and renewable diesel consumption in 2025 to be due to the RFS Program. Initial estimates from slightly higher volumes suggest that the RFS Program could potentially lead to an increase of as much as 1.9 million additional acres of cropland. These estimates are highly uncertain due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes.

17.7 Chapter 8: Air Quality

- There is no new evidence that contradicts the fundamental conclusions of previous biofuels Reports to Congress. Those conclusions emphasized that emissions of nitrogen oxides (NO_x), sulfur oxides (SO_x), carbon monoxide (CO), volatile organic compounds (VOCs), ammonia (NH_3), and particulate matter ($PM_{2.5}$) can be impacted at each stage of biofuel production, distribution, and usage.
- Increased corn production results in higher agricultural dust and NH_3 emissions from fertilizer use. Improved nitrogen management practices can decrease these NH_3 emissions, however. Increased corn ethanol production and combustion leads to increased NO_x, VOCs, $PM_{2.5}$, and CO. As the increased ethanol volumes are displacing petroleum and its related emissions in each of these areas, the overall impact on the environment is a complex issue.
- Emissions from production of biodiesel from soybean oil vary depending on the oil extraction method, with mechanical expelling the least efficient with the highest emissions of NO_x, VOCs, CO, and $PM_{2.5}$, followed by hexane extraction and then enzyme-assisted aqueous extraction process (EAEP).
- EPA's "anti-backsliding" study (U.S. EPA, 2020b) examined the impacts on air quality from end-use changes in vehicle and engine emissions resulting from required renewable fuel volumes under the Renewable Fuel Standard (RFS). Compared to the 2016 "pre-RFS" scenario, a 2016 "with-RFS" scenario increased concentrations of ozone (eight-hour maximum average) across the eastern United States and in some areas in the western United States, $PM_{2.5}$ concentrations were relatively unchanged in most areas, while NO_2 concentrations increased in many areas and CO decreased. Furthermore, increases in formaldehyde and acetaldehyde were widespread, while benzene and 1,3-butadiene levels went down. Other recent research addressing air quality impacts of biofuels is limited.
- Using the GREET model (Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation), lifecycle emissions from corn ethanol are generally higher than from gasoline for VOCs, SO_x, $PM_{2.5}$, PM_{10}, and NO_x. However, the location of emissions from biofuel production tends to be in more rural areas where there are fewer people. How this translates to effects on human health is complex, as air pollutants can be transported and transformed downwind, and it depends not only on the number of people, but on their demographics and vulnerability, as well as the dose-response relationship, which is pollutant-specific, among other factors.

- On a per unit energy basis over the period analyzed, biofuels manufacturing has a larger impact than their petroleum counterparts on smog formation, acidification, $PM_{2.5}$ exposure, and ozone depletion potentials, but a smaller potential effect in the total U.S. context due to the smaller size of the biofuels industry. Nonetheless, this conclusion needs to be interpreted in the context of each industry: while petroleum refining is a highly optimized, mature industry, biofuels are still reaching maturity as indicated in their emission profile over the 2002–2017 period. The observed trends seem to indicate that, on a per unit energy basis, the biofuel industry is consistently reducing emissions as it matures.
- The likely future effects of the RFS Program on air quality are highly uncertain; there will be an overall increase in biofuel volume but the volumes of various types of biofuels produced and used will be different compared with past RFS volumes.

17.8 Chapter 9: Soil Quality

- Impacts to date on soil quality from biofuels and the RFS Program are almost exclusively due to corn and soybean production for corn ethanol and soy biodiesel.
- Conversion of grasslands to corn and soybeans causes greater negative impacts to soil quality compared to growing these feedstocks on existing cropland. Simulations using the EPIC (Environmental Policy Integrated Climate) model found estimated grassland conversion to corn/soybeans from all causes generally increased soil erosion (-0.9-7.9%), and losses of soil nitrogen (1.2-3.7%) and soil organic carbon (SOC, 0.8-5.6%) in a 12-state, U.S. Midwestern region between 2008 and 2016. The range in losses depended upon the simulated tillage practices.
- Effects were not uniform across the 12-state region. Hotspots of grassland conversion and subsequent soil quality impacts occurred in locations such as southern Iowa and the Dakotas.
- A range of percentages (0–20%) was applied to the EPIC results to estimate the fraction of soil impacts attributable to grassland conversion estimated to be caused by the RFS Program. According to this estimation, the RFS Program increased erosion, nitrogen loss, and SOC loss from 0-1.6%, 0-0.7%, and 0-1.1%, respectively, across the 12-state region between 2008 and 2016. Notably, these modeling estimates represent a RFS-corn-ethanol effect only, and do not include any additional quantitative effect from the RFS Program on soybean biodiesel and soybean acreage as this effect could not be quantified in Chapter 7, nor do the estimates include any effect from crop switching on existing cropland.
- For context, the magnitude of these changes can be compared to the benefits of conservation programs, like the Conservation Reserve Program (CRP). The RFS-associated increase in

nitrogen loss for this 12-state region, for example, represents up to 3.7% of the nitrogen retention benefits of the CRP for the entire United States.

- Additional conservation measures—such as further adoption of conservation tillage and cover crops—would help reduce the impacts of biofuels generally and the RFS Program specifically on soil quality.
- The likely future effects of the RFS Program are highly uncertain. While the projected cropland expansion for 2023–2025 is slightly larger than the estimated historical cropland expansion from the RFS Program, EPA cannot say with reasonable certainty that any particular terrestrial ecosystem or biodiversity will be impacted, due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes.

17.9 Chapter 10: Water Quality

- Water quality impacts to date from biofuel production are almost exclusively due to corn and soybean production for corn ethanol and soy biodiesel. Conversion of grasslands to corn and soybeans causes greater negative impacts to water quality compared to growing these crops on existing cropland.
- A Missouri River Basin (MORB) Soil and Water Assessment Tool (SWAT) model was applied to a 30-year period (1987 to 2016) to assess the effects of recent estimated cropland expansion on water quality, where the highest rate of grassland to cropland conversion have occurred (1.18% of the total land area was converted from 2008 to 2016 basin wide). Conversion to cropland resulted in little change in streamflow basin wide. For total nitrogen (TN) and total phosphorus (TP), grassland conversion to continuous corn resulted in the greatest increase in TN and TP loads (6.4% and 8.7% increase, respectively); followed by conversion to corn/soybean (TN increased 6.0% and TP increased 6.5%); and then conversion to corn/wheat (TN increased 2.5% and TP increased 3.9%). These increases are relatively small on an absolute basis, only approximately 0–20% of which may be due to the RFS Program, but aggravate conditions in watersheds already impacted by nutrients.
- Groundwater and drinking water nitrate concentrations may increase with increasing acreage of corn. Switching from corn or other crops to dedicated biofuel crops (e.g., switchgrass) may lead to reductions in nitrogen losses to water bodies and thereby reduce future drinking water nitrate levels in both groundwater and surface water.
- Pesticides in drinking water could be impacted by increasing acreage of corn or soybean for biofuels or other uses. Certain pesticides, such as atrazine, are more widely used than others on

these crops, and have also been frequently detected in surface and ground water. Pesticides whose usage on corn or soybeans has changed in recent years would presumably see commensurate changes in their detection likelihood in water, including in drinking water supplies. Fewer pesticides may need to be applied to dedicated biofuel crops than corn and soybean crops.

- Lifecycle potential eutrophication effects for both corn ethanol and soybean biodiesel are higher than their fossil fuel counterparts per megajoule and overall in most cases. This is driven primarily by fertilizer application to corn and soybean crops and by the resulting nutrient runoff and leaching.
- Continued implementation of conservation practices has been shown to reduce soil erosion, nitrate loss, and phosphorus release. Integrating landscape design and conservation practices (reduced tillage, riparian buffer, saturated buffer, cover crops) in current corn/soybean land and cropland converted to perennial grass at field tests has been shown to decrease nutrient loss to surface water while maintaining corn/soy productivity. Conservation practices, such as reduced tillage and the use of cover crops, can reduce the negative impacts of corn and soybean feedstock production and improve soil health.
- The likely future effects of the RFS Program on water quality are highly uncertain. While the projected cropland expansion for 2023–2025 is slightly larger than the estimated historical cropland expansion from the RFS Program, it cannot be said with reasonable certainty that water quality will be impacted due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes.

17.10 Chapter 11: Water Use and Availability

- Water use and water availability impacts of biofuels are primarily related to irrigation needs (the feedstock production stage), while water use in biorefineries (the conversion stage) represents a small and declining percentage of lifecycle water use.
- For corn-based ethanol, when accounting for ground and surface water (“blue water”) used for irrigation, 88% of total lifecycle biofuel water use is for irrigation for feedstock production (on a gallon per megajoule [MJ] basis). For soybean-based biodiesel, feedstock irrigation is 98% of total lifecycle biofuel water use.
- The overall irrigated area of corn, according to USDA surveys, increased from between 9.3 and 9.7 million acres before the 2005 Energy Act to between 12 and 13 million acres reported in the 2008 and 2013 surveys, before declining to 11.6 million reported in the 2018 survey (representing 14% of total corn acres in 2018).

- The majority of total irrigation withdrawals (81%) and irrigated lands (74%) in 2015 occurred in the 17 conterminous western states located west of and including the Dakotas, Nebraska, Kansas, Oklahoma, and Texas overlying the High Plains Aquifer (HPA). Some satellite-based studies show irrigated croplands (all crops, all uses) over the HPA increased from approximately 14 million acres to 15 million acres (all crops/uses) between 2000 and 2017.
- Continued irrigation at present rates over the Southern HPA is not sustainable where the extraction rate exceeds recharge, most notably in eastern Colorado, western Kansas, the Texas Panhandle, and eastern New Mexico. However, for the Northern HPA, climate change is expected to increase precipitation, and the projections show that the irrigated area of the "MonDak" region (eastern Montana and western North Dakota) could expand, while irrigation at present rates is considered sustainable in much of eastern Nebraska.
- Water requirements for producing a gallon of corn ethanol (including total irrigation and refinery water) ranges from 8.7 to 160 gal/gal (i.e., gallons of water per gallon fuel) of ethanol (average 76 gal/gal), compared to petroleum-based gasoline, which ranges from 1.4 to 8.6 gal/gal of gasoline (average 5.7 gal/gal). The major factors determining the range are the regional variation in irrigation requirements for these corn-producing regions.
- Though a small fraction of the lifecycle water use, the water intensity of ethanol production in biorefineries decreased by 12% between 2011 and 2017 and by 54% between 1998 and 2017. These reductions have resulted from the adoption of energy-efficient and water-efficient technologies, water reuse and recycling, increased system integration in retrofitting existing plants, and diversification of water sources.
- Combining the GREET (Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation) model with WATER (Water Analysis Tool for Energy Resources) showed that, on a per megajoule basis, corn ethanol requires 0.084 –1.103 gallons (Corn Belt and Northern Plains states, respectively), with a U.S. weighted average of 0.377 gallons per megajoule. In comparison, gasoline averages 0.082 gallons per megajoule. Lifecycle water consumption for soybean biodiesel is slightly higher, from 0.102 to 1.697 gallons per megajoule, compared with 0.057 for diesel.

17.11 Chapter 12: Terrestrial Ecosystem Health and Biodiversity

- Impacts to date from biofuels on domestic terrestrial biodiversity, as an indicator of ecosystem health, are primarily due to corn and soybean feedstock production for ethanol and soy biodiesel. Shifts in perennial plant cover to corn and soybeans, and corn and soybean production practices are the two main drivers of effects.

- Of land in perennial cover shifting to annual crops, the vast majority was from grasslands, ranging from relatively unmanaged to highly managed grasslands (e.g., hay, pasture). The loss of grassland cover to annual crops, such as corn and soybeans, negatively impacts terrestrial biodiversity, including grassland species of birds, bats, pollinators and other beneficial organisms (e.g., insects that provide pest control), and plants.
- Between 2008 and 2016, shifts from land in perennial cover to corn and soybeans due to all causes, including potentially biofuels, occurred in areas adjacent to or within critical habitat of 27 terrestrial threatened and endangered (T&E) species across the contiguous United States, according to an analysis using the USDA Cropland Data Layer (CDL). The CDL is relatively accurate at large spatial scales (e.g., states) but can be more uncertain at local scales. Thus, it may require verification with imagery or direct visitation to confirm these results.
- Beyond change in land cover, crop production practices for corn and soybeans can also negatively affect terrestrial biodiversity, particularly through pesticides.
- The range of possible impacts from the RFS Program likely spanned from no effect to a negative effect on terrestrial biodiversity historically (2008 to 2016). Further refinement of the acreage estimates attributable to the RFS Program are needed to reduce this range of possibilities. These findings do not necessarily apply for years beyond 2016, when the effects of the RFS Program on corn ethanol and soy biodiesel production may have changed.
- Further evaluation would be needed to quantify the magnitude of any historical impacts of the RFS Program on biodiversity. Any effects may be relatively small compared to those of total U.S. cropland, but may be more important regionally or locally. Finally, whether T&E species were impacted by the RFS Program during this period (2008 to 2016) is also possible, but unknown, and requires further evaluation.
- Conservation practices can reduce negative impacts to terrestrial biodiversity. These practices include protecting environmentally sensitive lands, increasing habitat heterogeneity, and decreasing the use of pesticides.
- The likely future effects of the RFS Program are highly uncertain. While the projected cropland expansion for 2023–2025 is slightly larger than the estimated historical cropland expansion from the RFS Program, EPA cannot say with reasonable certainty that any particular terrestrial ecosystem or biodiversity will be impacted due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes.

17.12 Chapter 13: Aquatic Ecosystem Health and Biodiversity

- Many watersheds in the Midwest and other U.S. regions have historically been impacted by agriculture generally and by crops used for biofuels specifically, but the incremental effect from recent (2008–2016) agricultural expansion from all causes, including any potential impact from the RFS Program specifically, appears to be minor in comparison.
- Water demand for feedstock production may change stream flow and alter flow patterns that are important for supporting fish diversity.
- Pesticides used in feedstock production including atrazine, glyphosate, and neonicotinoids, have direct toxicity to some nontarget organisms as well as a variety of sublethal, indirect environmental effects on aquatic ecosystem health and biodiversity. Based on overlap of species ranges and critical habitat with atrazine usage, EPA judged atrazine was likely to adversely affect 180 out of 207 federally listed (i.e., threatened and endangered) aquatic invertebrate species assessed, including mussels, snails, shrimp, amphipods, water beetles, and crayfish.
- Based on data from nationally representative surveys of the nation's wadeable stream miles in 2004 and about 10 years later in 2013–2014, biological and nutrient conditions worsened in the ecoregions roughly coinciding with areas of corn and soybean production compared to the rest of the continental United States. National surveys found that benthic macroinvertebrates were nearly twice as likely to be in poor condition in waterbodies with high nutrient concentrations and/or excess sediments.
- For the scenarios examined in a recent modeling study on agricultural expansion due to all causes from 2008–2016, the flow-weighted nutrient concentrations increased by less than 5% on average across the Missouri River Basin (MORB). For the scenario of conversion from grassland to corn/soy rotation, only 0.11% of watersheds in the MORB had increases in nutrient concentrations that were more than 10% of the baseline scenario. Given the RFS Program may have impacted corn planting by 3.5 million acres or less in 2016 (refer to Chapter 6), increases in nutrient concentrations that may be attributable to the RFS Program are unlikely to result in *new exceedances* of current state numeric nutrient criteria in agricultural regions of the United States, such as the MORB. Total effects may be larger or smaller because this study only included effects from agricultural expansion (expected to be the largest source) and not agricultural intensification or recent improvements in tillage practices.

- Demand for biofuel feedstocks may contribute to increased frequency and magnitude of harmful algal blooms and hypoxia. Altered food webs and changes in nutrient cycling can trigger feedback loops that make it difficult to prevent or mitigate the effects of harmful algal blooms and hypoxia on aquatic ecosystems.
- Adoption and expansion of sustainable conservation practices and technologies remain critically important to reducing impacts on aquatic ecosystems by restoring flow and decreasing loads of nutrients, sediment, and pesticides to levels that are less harmful to aquatic organisms.
- The likely future effects of the RFS Program are highly uncertain. While the projected cropland expansion for 2023–2025 is slightly larger than the estimated historical cropland expansion from the RFS Program, the authors cannot say with reasonable certainty that any particular aquatic ecosystem will be impacted due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes.

17.13 Chapter 14: Wetland Ecosystem Health and Biodiversity

- The area of wetlands converted to cropland from 2008 to 2016 in the United States is estimated at nearly 275,000 acres, with most losses occurring in the Prairie Pothole Region. Given the lack of national or regional datasets to track changes in acreage attributable to the Renewable Fuel Standard (RFS) Program, the extent of wetland losses solely attributable to the RFS cannot be more accurately estimated in this Third Triennial Report to Congress (RtC3).
- Small, seasonal wetlands are being lost at the fastest rate. The loss and consolidation of small wetlands to promote crop production has negatively impacted amphibians, invertebrates, and other aquatic species that depend on shallow water depths for reproduction. Shifts to longer hydroperiods in large or consolidated wetlands have more uniform (less diverse) invertebrate communities and can support fish that prey on insects and amphibians.
- Small wetlands and ponds are primary sources of water for aquifer recharge in the Northern Prairies. Recent studies in the Canadian Prairie Pothole Region found that while permanent ponds and wetlands are sources for recharge to aquifers, wetlands with surface water ponds that dry out every year play the dominant role in groundwater replenishment.
- While some Endangered Species Act-listed and other waterbirds have declined, waterfowl (ducks, geese, swans) as a group have not experienced declines over the past decade, possibly due to availability of food (grains), increased precipitation, and the interspersion of ponded waters and agricultural fields along migration routes.

- Shifts to corn and soybean production have resulted in more frequent application of chemicals, including pesticides and fertilizers. Increased usage of neonicotinoid insecticides is of particular concern because of their high toxicity to invertebrates, which are important food sources for wetland-dependent taxa.
- Nationally, net wetland gains and losses are not distributed evenly across wetland classes. Since 2007, the nation has lost a net total of 120.3 thousand acres of palustrine (marsh-like) wetlands and gained a net total of 205.9 thousand acres of lacustrine (lake-like) habitats in the conterminous United States. The wetlands within these classes support different species and perform different ecosystem functions. Shifts from palustrine to lacustrine wetlands have resulted in the loss or impairment of functions that impact watershed hydrology, water quality, and water quantity.
- Evidence from the Prairie Pothole Region in the United States and Canada indicates that trends in larger wetland size, shifts to lakes and ponds (vs. vegetated wetlands), and prolonged and more frequent flooding are due to the combined effects of climate change and increased wetland ditching and consolidation. These trends are highly correlated with increased annual precipitation, which is projected to continue.
- The likely future effects of the RFS Program are highly uncertain. While the projected cropland expansion for 2023–2025 is slightly larger than the estimated historical cropland expansion from the RFS Program, it cannot be said with reasonable certainty that any particular wetland will be impacted, due to the numerous layers of uncertainty between the finalized RFS annual volumes and on-the-ground, localized land use changes.

17.14 Chapter 15: Invasive or Noxious Plant Species

- Direct impacts to date on the environment from the cultivation of invasive or noxious plant species as biofuel feedstocks have not been observed, since most biofuel is produced from a small number of non-invasive feedstock species (i.e., corn and soybean).
- Impacts from the cultivation of corn and soybeans on the evolution of herbicide-resistant weeds do exist, although it is unclear to what extent impacts can be attributed to corn and soybeans grown to meet either biofuel demand generally or the specific requirements of the RFS Program. Since the RFS was enacted, herbicide-resistant weeds have increased production costs for farmers in terms of herbicide expenditures and in their overall investment in technology and production systems. However, this temporal association alone is not sufficient to determine causation.

- The likely future effects of the RFS Program from invasive or noxious feedstocks are uncertain due to many factors. However, if biofuels continue to be produced mostly from corn and soybean, there will be no likely future effects directly from potential invasive or noxious feedstocks. This is because corn and soybean are not invasive. Two potentially invasive feedstocks (i.e., giant reed [*Arundo donax*] and napier grass [*Pennisetum purpureum*]) are part of approved biofuel pathways under the RFS Program. They could produce effects *if* they are grown in the future and *if* additional registration, reporting, and recordkeeping requirements that are in place and designed to limit their spread are not sufficient to prevent escape and invasion. However, as of the publication of this report, no Renewable Identification Numbers (RINs) have been generated that involve these feedstocks nor have incipient invasions or impacts been observed as a result of their production for biofuel.
- Likely future effects from herbicide-resistant weeds will continue to grow if current trends hold in the incidence of new cases and number of weed species that are resistant to multiple herbicide sites of action. As with impacts to date, future impacts from the cultivation of corn and soybeans on the evolution of herbicide-resistant weeds are likely to occur, but it will be challenging to determine what extent of impacts can be attributed to corn and soybeans grown to meet either biofuel demand generally or the specific requirements of the RFS Program.
- It is not possible to reach a firm conclusion regarding the relative overall invasion risk posed by biofuels compared to petroleum. Risks of invasion associated with petroleum exploration and extraction include both the introduction of non-native species via hitchhiking on machinery and infrastructure and the facilitation of non-native dominance through habitat disturbance across a broad range of habitats, including terrestrial and marine.

17.15 Chapter 16: International Effects

- Attribution of international effects from the RFS Program remains challenging due to complex interrelationships among other major drivers of observed change. There are relatively few studies on this topic specifically, though many on international effects from biofuels more generally, and analyses are impeded by inconsistent data and large uncertainties.
- This chapter considers three causal chains whereby the RFS Program may lead to environmental effects in foreign countries. First, it is considered that the RFS Program may increase biofuel imports from other countries to the United States, and that such imports may

be associated with environmental effects related to producing the imported biofuels. Second, the market-mediated effects of the RFS Program on international crop and commodity prices and production, and related environmental effects are considered. Third, it is considered that, by supporting the growth of the U.S. biofuel industry, the RFS Program may increase biofuel exports from the United States to other countries, with potential environmental effects/ benefits for the receiving countries.

- International environmental effects that are clearly attributable to the RFS Program due to U.S. ethanol and biodiesel imports could not be confidently quantified. The lack of empirical evidence to support causal linkages between the RFS and international environmental effects does not necessarily rule out international effects attributable to the RFS Program.
- Imports of biofuels to the U.S.—one of the mechanisms for international effects—have fallen drastically since peaking before the RFS Program in 2004–2006. Evidence supports attribution to the RFS Program for some biodiesel imports since 2007. The value of advanced biofuel (D5) RINs was among many factors that supported sugarcane ethanol imports from Brazil since mid-2010.
- The hypothesis that U.S. demand for sugarcane ethanol attributable to the RFS Program played the dominant role in the observed changes in Brazil's ethanol production and associated environmental effects is not supported by available evidence. Ethanol production in Brazil has been supported by domestic policies in Brazil for decades.
- U.S. ethanol production that exceeds domestic demand is exported to more than 70 nations around the globe, and since 2008, the United States has been a net exporter of biofuel (ethanol + biodiesel) on an annual basis. However, the share of exports attributable to the RFS Program is uncertain. To the degree that the RFS Program encouraged investments that generated surplus ethanol for export, the RFS Program contributed to the international effects associated with net U.S. exports, which could be environmentally beneficial for importing nations.
- Two-way interannual ethanol trade with Brazil (i.e., U.S. corn ethanol and Brazilian sugarcane ethanol) is likely driven by extreme weather events, seasonal differences in the agricultural cycles, and regulatory standards (for example, the CA-LCFS could encourage imports of ethanol with lower carbon intensities and exports of ethanol with higher carbon intensities ratings), between the two nations. However, the relative contributions of these causal factors to the extent of this two-way trade has not been resolved in the scientific literature.

- A portion of the gross biodiesel imports during 2012–2019, averaging approximately 295 million gallons per year, are reasonably attributed at least in part to the RFS Program. However, many factors impact biodiesel trade, and sources of import (i.e., countries) are diverse and irregular, each affected by their own domestic policies which are difficult to assess with current models and data.
- While this chapter concludes that more research is needed on the effect of the RFS Program on international cropland areas and environmental effects more broadly, combining published simulation modeling estimates of the land use change effects of biofuels (section 16.3.3 for corn ethanol) and the estimates for the effect of the RFS Program on corn ethanol (Chapter 6) yields an illustrative range of the effect of the RFS Program on non-U.S. cropland area of 0 to 1.6 million acres. The estimated effect of the RFS Program does not yet include effects on soy biodiesel. As more data become available and are analyzed, historical relationships among U.S. biofuel policies, production, trade, environmental indicators, and other variables may be clarified and uncertainties reduced. Review of potential international effects of the RFS Program associated with biodiesel imports, and on global cropland more broadly, finds that quantification of effects is uncertain but could be significant and merits further research. The relationship of the RFS Program with palm oil expansion, and the environmental costs and benefits of two-way trade, merit further study.

Appendix A: Procedures and Results for HERO/SWIFT Literature Review

A.1 Overview and Objective

This appendix describes the process for the literature search and screening conducted under Contract EP-C-16-021 WA 3-24, to support the Third Triennial Report to Congress (RtC3). The objectives of this appendix are to (1) describe the literature search approach (section A.2); (2) detail the method used in the literature screening (section A.3); and (3) summarize the results from the screening (section A.4). All literature included in the RtC3 is documented in the U.S. EPA Health and Environmental Research Online (HERO) database. The publicly accessible HERO project page provides the bibliographic information for included studies. References can be viewed individually or filtered by draft or chapter in which they are cited.

A.2 Literature Search Approach

For RtC3, the literature search strategy included a citation network search, which was completed in March 2019 and updated in September 2019 based on references cited in the Second Triennial Report to Congress [RtC2; (U.S. EPA, 2018)]. Results of this citation network search totaled 14,513 peer-reviewed journal articles or book chapters that cited a reference in RtC2 at least once. Results were then uploaded into Swift Active Screener (SWIFT-AS, see section A.3.1) (Howard et al., 2020) to be screened for relevance and potential inclusion in the RtC3 based on inclusion and exclusion criteria (see Table A.1). A small subset of 910 references from the citation network search with missing abstracts were separately screened for inclusion outside of SWIFT-AS. Report authors also relied on their extensive knowledge of the subject area to incorporate any additional peer-reviewed journal articles that met the inclusion criteria.

Following the external peer review of the RtC3 in February 2023, additional articles recommended by the peer review panel were considered for inclusion based on the same inclusion and exclusion criteria used in the initial literature search and screen. Section A.3 describes the screening process and inclusion and exclusion criteria in greater detail.

A.3 Screening Method

A.3.1 Summary

Literature was screened using SWIFT-AS, a widely adopted, web-based literature screening software application that uses active machine learning to allow screeners to efficiently screen literature for relevance. Search results are ranked by descending likely relevance using a bag-of-words approach and Latent Dirichlet Allocation, trained by both the screener's inclusion and exclusion decisions and a positive training set, when supplied (Howard et al., 2016). As references are screened and tagged as included or excluded, the ranking model is further trained to sort the remaining literature, promoting predicted relevant literature in the screening order. Literature was screened until SWIFT-AS reached 90% predicted recall.

The literature was initially screened by the EPA's contractor; papers deemed potentially relevant after initial screening were then shared with the report authors for review and possible inclusion in the report. The screening process was aided by developing a set of *inclusion* and *exclusion* terms in article titles and abstracts. *Inclusion* and *exclusion* criteria highlight keywords to assist with the identification of relevant articles during screening. *Inclusion* criteria for this report were developed to closely track the relevant biofuel types, topics, and content planned for the RtC3. *Exclusion* criteria focused on biofuel topics that are not a focus of the Triennial Reports, specifically identifying terms related to emission coefficients, greenhouse gas accounting, and conversion technologies. See Table A.1 for a list of *inclusion* and *exclusion* terms used in the screening.

Table A.1. Inclusive and exclusive keywords used to aid the screening process.

Inclusive Keywords				Exclusive Keywords
2,4-d	corn	life cycle	policy	emission factor
acetochlor	dichloropropene	maize	soil	generation
air quality	economic	mancozeb	sorghum	GHG accounting
aquatic	ethanol	metam	SOx	GHG
atrazine	ethephon	metolachlor	soy	
bagasse	exotic	MSW	soybean	
beets	fat	neonicotinoid	sugar cane	
biodiesel	feedstock	neonicotinoids	sugarcane	
bioenergy	glyphosate	NLCD	sulfur	
biofuel	grease	non-native	sulfuric acid	
biofuels	intensification	NOx	supply chain	
biota	invasive	oil	terrestrial	
canola	land use	ozone	United States	
chloropicrin	land-use	palm	water quality	
chlorothalonil	LCA	pendimethalin	wetland	
chlorpyrifos	lead	petroleum		

Four screeners participated in the screening process. To ensure internal consistency across screeners, all screeners participated in four shared-screen training sessions, twice weekly check-in meetings, and daily email correspondence. In addition, all reviewers adhered to a series of screening rules established during the shared screening sessions and augmented during the screening process. Once the screening was completed, libraries of relevant articles were then shared with the Chapter Teams for potential inclusion in the RtC3.

A.3.2 Biofuels Definition

This section describes the biofuels and feedstocks that are included in the RtC3 (Table A.2) and served to identify the biofuels and feedstocks to include in the literature screening. This is the same table as in Chapter 2 (i.e., Table 2.1) except that at the time of the SWIFT-AS screening the most recent year available was 2018.

Table A.2. Estimated volumes of biofuel (million gallons) imported or domestically produced from individual biofuel–feedstock–region combinations from 2005 to 2018. Note that biodiesel also includes renewable diesel.[1]

Fuel	Feedstock	Region/Country	Source	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
Ethanol	Corn Starch	U.S.	1	3,904	4,884	6,521	9,309	10,938	13,298	13,929	13,218	13,293	14,313	14,807	15,329	15,845	16,061
Ethanol	Sugarcane	Brazil	2	35	453	185	203	5		101	404	322	56	88	36	77	53
Ethanol	Sugarcane	Central Am./Car.	3	98	228	243	320	182	2	69	82	50	11				1
Ethanol	Mixed	Rest of World	4	3	49	8	6	11	13	2	8	5	8	3	1		
Biodiesel	Canola Oil	U.S.	5						35	113	105	85	145	101	160	205	159
Biodiesel	Corn Oil	U.S.	6					13	16	40	86	141	135	143	185	223	278
Biodiesel	Palm Oil	U.S.	7									83					
Biodiesel	Soybean Oil	U.S.	8					309	161	553	537	726	674	662	865	878	1,004
Biodiesel	FOGs	U.S.	9					194	131	320	315	435	483	534	598	549	721
Biodiesel	Palm Oil	Southeast Asia	10									147	203	275	299	144	33
Biodiesel	FOGs	Europe	11							11	51	78	21	3	24	19	83
Biodiesel	FOGs	Southeast Asia	12							7	17	175	172	154	184	199	192
Biodiesel	Soybean Oil	Argentina	13									66	48	183	434	341	
Biodiesel	Mixed	Canada	14							23	20	23	66	57	101	96	83
Biodiesel	FOGs	Rest of World	15							3	1	2			1		
CNG/LNG	MSW	U.S.	16							1	3	26	53	115	167	208	268
CNG/LNG	MSW	Canada	17											25	21	32	36
Total				4,040	5,614	6,956	9,838	11,652	13,657	15,173	14,848	15,658	16,388	17,3151	18,406	18,815	18,972

[1] Details on the sources of information for Table A.2 are the same as for Tables 2.1 and 2.2 and are described in Appendix B.

A.4 Results

The 90% recall threshold resulted in the screening of 5,911 of the 13,603 (i.e., 43.5%) articles. Of these, 1,555 were identified as relevant to one or more of the chapters in the RtC3 (Table A.3). Relevant articles were then exported from SWIFT-AS and imported into HERO. HERO allows users to view all references included in the RtC3 sorted by various categories including the chapter in which they were cited. Table A.2 details the number of articles imported into HERO and assigned to each category.

At the end of the screening, libraries of relevant papers were assembled and sent to the Chapter Leads for dissemination to the chapter teams for potential inclusion.

Table A.3. Count of screened articles sorted and categorized in SWIFT-AS and imported into HERO.

Category	Count of Articles in HERO Category	Count of Articles in SWIFT-AS Category
Included	1,555	1,555
Dominant four biofuels	682	683[a]
Other biofuels	589	589
Unsure (biofuel)	468	468
Ch. 5–7 (ERD)	333	333
Ch. 8	104	104
Ch. 9	406	407[a]
Ch. 10	210	210
Ch. 11	142	142
Ch. 12	343	343
Ch. 13	101	101
Ch. 14	70	70
Ch. 15	64	64
Ch. 16	438	438

[a] One article was reviewed by two screeners at the same time, creating a duplicate record. However, the article was sorted/binned the same way, so screening results were not impacted by the article being reviewed twice.

A.5 References

Howard, BE; Phillips, J; Miller, K; Tandon, A; Mav, D; Shah, MR; Holmgren, S; Pelch, KE; Walker, V; Rooney, AA; Macleod, M; Shah, RR; Thayer, K. (2016). SWIFT-Review: A text-mining workbench for systematic review. Syst Rev 5: 87. https://dx.doi.org/10.1186/s13643-016-0263-z.

Howard, BE; Phillips, J; Tandon, A; Maharana, A; Elmore, R; Mav, D; Sedykh, A; Thayer, K; Merrick, BA; Walker, V; Rooney, A; Shah, RR. (2020). SWIFT-Active Screener: Accelerated document screening through active learning and integrated recall estimation. Environ Int 138: 105623. https://dx.doi.org/10.1016/j.envint.2020.105623.

U.S. EPA. (2018). Biofuels and the environment: Second triennial report to congress (final report, 2018) [EPA Report]. (EPA/600/R-18/195). Washington, DC. https://cfpub.epa.gov/si/si_public_record_report.cfm?Lab=IO&dirEntryId=341491.

Appendix B: Estimating Renewable Fuel Production and Use in the United States

This appendix explains the data sources and methodology used to estimate renewable fuel production and use in the United States, as shown in Chapter 2, Tables 2.1 and 2.2 of the RtC3. This appendix is organized into sections by fuel type, as the same or similar data sources are generally used for each fuel type.

B.1 Ethanol (Table 2.1, Sources 1–4)

Domestic ethanol production (1) is sourced from the United States Department of Agriculture (USDA) Economic Research Service U.S. Bioenergy Statistics. Data can be found in Table 2 of the following website: https://www.ers.usda.gov/data-products/us-bioenergy-statistics/

Data on ethanol imported from Brazil (2), Central America/Caribbean (CAC) (3), and the rest of the world (4) is sourced from the Energy Information Administration (EIA). These data can be accessed at: https://www.eia.gov/dnav/pet/pet_move_impcus_a2_nus_epooxe_im0_mbbl_a.htm.

For imported ethanol, the focus is mostly on imports from Brazil rather than the CAC because of economic and trade factors that suggest that most of the imported ethanol was originally from Brazil. Prior to 2011, there was a tariff on imported ethanol from countries other than those in the Caribbean Basin Initiative (CBI, included many countries in Central America and the Caribbean) up to a certain volume. EPA concluded that during this period countries such as Brazil likely exported hydrous ethanol to countries in the CBI that were then dehydrated and exported to the United States to avoid the tariffs (U.S. EPA, 2010; Yacobucci, 2008). When the tariff was phased out in 2011, Brazil began directly exporting ethanol to the United States, as evidenced in Chapter 2, Tables 2.1 and 2.2. There were still some direct imports from Brazil from 2005 to 2011 (Tables 2.1 and 2.2). Although the domestic production from the CBI was not zero from 2005 to 2011, much of the feedstock production that contributed to U.S. imports from the region was actually cultivated in Brazil.

B.2 Domestic Biodiesel and Renewable Diesel (Table 2.1, Sources 5–9)

Data for domestic biodiesel production was estimated using data from EIA's Monthly Biodiesel Production Reports. These reports are available at https://www.eia.gov/biofuels/biodiesel/production/. These reports list total domestic biodiesel production (in million gallons) and feedstocks used by domestic biodiesel producers (in million pounds) for each year, starting with 2009. For each year, domestic biodiesel production was estimated by feedstock by taking the ratio of the quantity of each feedstock used to the total quantity of feedstocks used to produce biodiesel, and multiplying that ratio by total biodiesel production that year. If a feedstock did not have an annual feedstock total listed but did have monthly feedstock use listed, annual feedstock use was estimated by adding all the monthly feedstock values that were listed. All animal fats and recycled oils were combined into a more general FOG category.

Data for domestic renewable diesel production was sourced from the EPA Moderated Transaction System (EMTS). This is the electronic reporting system used by all RIN generators in the RFS Program. Public EMTS data can be found at https://www.epa.gov/fuels-registration-reporting-and-compliance-help/rins-generated-transactions. Based on data from EMTS, the majority of domestic renewable diesel is produced from FOG. In the absence of precise data on the feedstocks used to produce renewable diesel a simplifying assumption was made that all domestic renewable diesel was produced from FOG.

B.3 Imported Biodiesel and Renewable Diesel (Table 2.1, Sources 10–15)

Data for total imported biodiesel was sourced from EMTS. All imported D6 biodiesel (legacy conventional biodiesel) was assumed to be palm oil biodiesel from southeast Asia. To allocate all the imported D4 biodiesel, EIA biodiesel import data was used (https://www.eia.gov/dnav/pet/pet_move_impcus_a2_nus_EPOORDB_im0_mbbl_a.htm). The percentage of total biodiesel imports by region (Table 2.1: Europe, Southeast Asia, Argentina, Canada, and the Rest of the World) was calculated. For all regions other than Southeast Asia, the data were used as reported by EIA to calculate each region's share of biodiesel imports. For Southeast Asia, the total import volume listed by EIA was used less the calculated palm oil import volume (described above). These region shares were multiplied by the total imported biodiesel volume from EMTS to estimate the total volume of biodiesel imports from each region by year. Based on the limited feedstock data available from EMTS, imports from Europe and the Rest of the World, along with non-palm oil imports from Southeast Asia, were assumed to be produced from FOGs. Biodiesel imports from Argentina were assumed to be from soybean oil. Biodiesel imports from Canada appeared to be from a variety of feedstocks, and the feedstocks are listed as "mixed" in Tables 2.1 and 2.2. The data and methodology used for imported renewable diesel is similar to that for biodiesel.

Imported biodiesel diesel volumes and percentages, respectively, listed in Table 2.1 and 2.2 (rows 10–15) are the sum of the biodiesel and renewable diesel totals discussed in the preceding paragraphs.

B.4 CNG/LNG (Table 2.1, Sources 16–17)

All CNG/LNG data are from EMTS. All imported CNG/LNG was assumed to be from Canada, based on the location of parties registered to generate D3 RINs from CNG/LNG derived from biogas in the RFS Program. All CNG/LNG (both domestic and imported) was assumed to be from MSW based on the limited available feedstock data in EMTS.

B.5. References

U.S. EPA. (2010). Renewable fuel standard program (RFS2) regulatory impact analysis [EPA Report]. (EPA-420-R-10-006). Washington, DC: U.S. Environmental Protection Agency, Office of Transportation Air Quality, Assessment and Standards Division. https://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=P1006DXP.txt.

Yacobucci, BD. (2008). Ethanol imports and the Caribbean Basin Initiative. (CRS Report No. RS21930). Congressional Research Service. https://www.everycrsreport.com/reports/RS21930.html.

Appendix C: Supplemental Analyses for Ch. 6 (Attribution: Corn Ethanol and Corn)

This appendix describes additional details on various factors assessed in terms of the drivers of increased corn ethanol production and consumption in the United States.

C.1 Inherent Economic Factors Affecting Relative Ethanol and Gasoline Prices

C.1.1 Crude Oil Prices and Ethanol's Relative Blending Value into Gasoline

One reason suggested in the literature for increased ethanol production from 2000–2020 relates to oil price increases (Babcock, 2013; Tyner et al., 2010). As with tax incentives and RIN prices, oil prices contribute to the relative price of ethanol to gasoline, and thus the attractiveness of ethanol versus gasoline. How a refiner decides on whether to produce blendstocks for oxygenate blending (BOBs) versus finished gasoline is part of the process, but is discussed separately in section C.1.3.

Crude oil prices are generally recognized as a principal macroeconomic factor affecting the production costs, and ultimately, the product prices of many different industries. In the context of this report, higher crude oil prices directly increase the price of gasoline. As gasoline prices increase, ethanol prices likely would also increase because higher crude oil prices increase the cost for growing corn; however, all else equal ethanol prices increase less than gasoline prices, making ethanol more favorable for blending into the gasoline pool as oil price increases. Above the blend wall this incentive is diminished for reasons that will be discussed in section C.5.

The effect of crude oil prices on gasoline prices, and likely associated impacts on corn and ethanol prices can be visualized through the price trends of these four commodities over time. The crude oil, gasoline, and ethanol prices are all plotted in Figure C.1 using the same units, while corn prices are plotted on a secondary axis.

Crude oil prices began increasing noticeably in 2004 after many years of being very low. Gasoline prices followed in tandem, rising as the global economy expanded before crashing with the financial crisis of 2008–2009. Crude oil prices increased again after the financial crisis and leveled off from 2011–2014, after which crude oil prices dropped again and stayed lower than the recent year highs. As expected, there is a close association between crude oil prices and gasoline prices. There seemed to be a clear association between corn and corn ethanol prices for many of the years, such as the increase in crude oil prices prior to 2008 and again prior to 2011, although there were times when there seemed to be

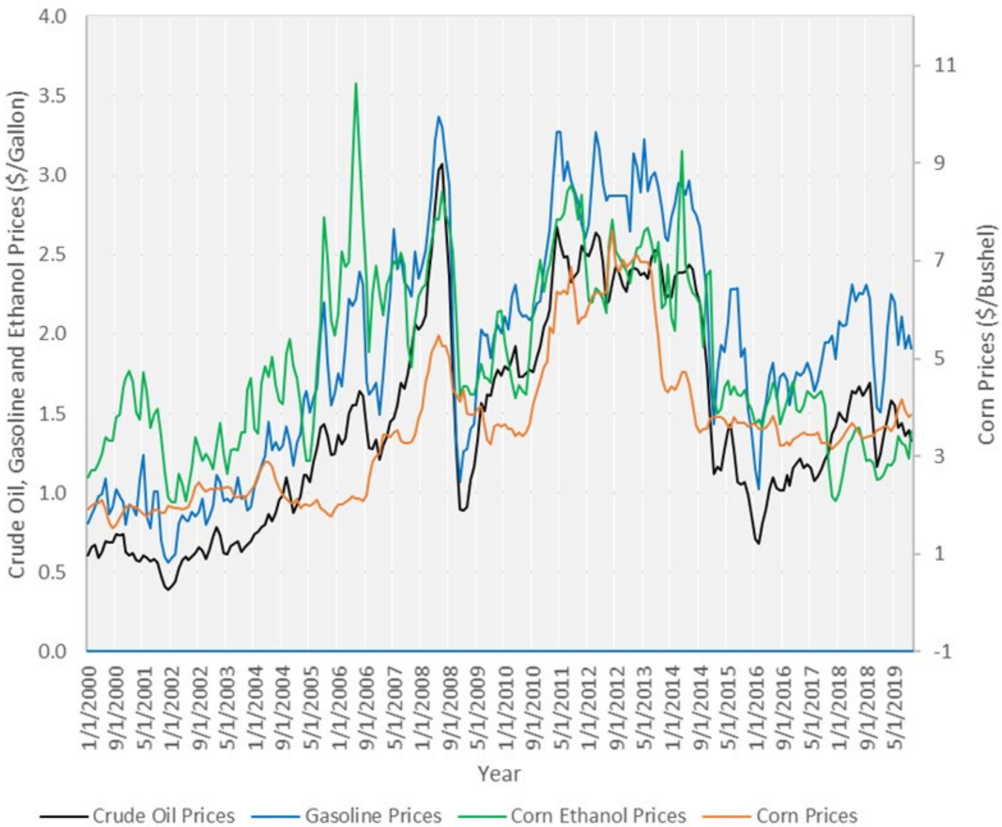


Figure C.1. Monthly crude oil, gasoline, and corn prices over time. Source: Corn and ethanol: USDA Economic Research Service; U.S. Bioenergy Statistics; Table 14, December 2019. Crude oil: EIA's Monthly Energy Review. Gasoline: https://www.eia.gov/dnav/pet/pet_pri_gnd_dcus_nus_m.htm (EIA, 2021b).

no association. Obviously, corn prices can be affected by many other factors, such as the number of acres planted, the productivity of the growing season due to weather, and food demand.

The ultimate impact of high crude oil prices on the economic attractiveness of blending ethanol into gasoline is most clearly understood, among the other factors affecting ethanol's blending value, by evaluating ethanol's blending value into E0 gasoline as E10. A year-by-year analysis was conducted for evaluating ethanol's blending value into the various gasolines (regular and premium grades, conventional and reformulated) sold in each state (U.S. EPA, 2022). This ethanol blending cost analysis included ethanol's estimated distribution cost to each state, accounted for ethanol's octane value and volatility cost, and accounted for the federal and state tax subsidies. As discussed in Chapter 6 section 6.3.5, this retrospective analysis concluded that even without the RFS Program, the economic and market factors were more than sufficient to drive the expansion of corn ethanol as E10 since the mid-2000s.

Another modeling effort also estimated the impact of different gasoline prices on ethanol consumption. Babcock (2013) estimated the effect of different wholesale gasoline prices on the consumption of ethanol without the RFS Program. This effort estimated that at $2 per gallon gasoline with 80–85 million acres of corn harvested as has been observed on average since 2007, it would be profitable to produce 11–12 billion gallons of ethanol without the RFS.[1]

Figure C.2 summarizes several studies that estimate the role that oil prices play in determining the incremental impact of the RFS2 on U.S. corn ethanol consumption discussed in Chapters 4 and 6. As the results across and within studies show, higher oil prices are expected to lead to lower corn ethanol production attributable to the mandate. Therefore, most studies projected that the incremental impact of the mandate would be modest or even nil at oil prices above \$90–\$100 per barrel.

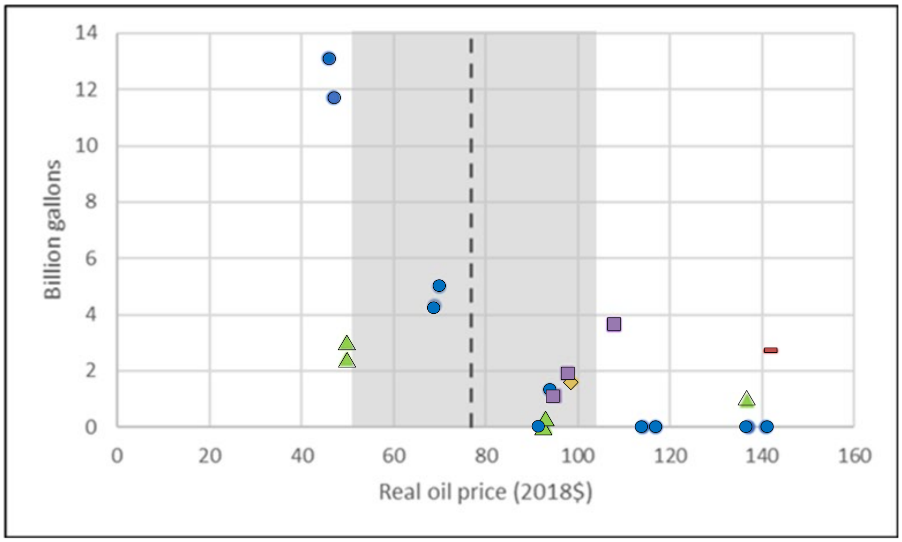


Figure C.2. Incremental effect of RFS2 on U.S. corn ethanol production. Estimates from five prospective studies. Estimates are from Babcock (2013, 2012) (green triangles); Bento and Klotz (2014) (purple squares); U.S. EPA (2010) (red dash); Meyer et al. (2013) (yellow-orange diamond); and Tyner et al. (Tyner et al., 2010; Tyner and Taheripour, 2008) (blue circles). Overlaid on this is the real price from 2002 to 2012 in 2018 dollars for oil from the Cushing OK WTI spot price showing the average (vertical line) and standard deviation (gray band).

C.1.2. Corn Prices

Corn prices appeared to increase roughly two years later than oil, beginning at the end of 2006. Thus, between 2004 and 2006, the price of oil was increasing while the price of corn remained relatively stable. This can be seen when directly comparing the normalized crude oil price to the normalized prices for gasoline and corn (Figure C.3).

There were many factors that contributed to the observed increase in the price of many cereal crops between 2006 and 2008 [Figure C.4, summarized from Wiggins et al. (2010) and other references]. These factors operated on different time scales, and geographies, but converged on the world cereal markets over this time period. Medium term factors (i.e., years to decades) included (1) a decrease in the growth of cereal production from >2.5% per year from 1960–1980 to ~1% thereafter, (2) smaller stocks

[1] This level included a valuation of octane. Without a value on octane and accounting for ethanol's fuel economy cost, production was only 2–3.6 billion gallons.

of grain reserves in the United States, Europe, China, and other countries, (3) devaluation of the U.S. dollar, which can lead to other importers like in Asia to be able to import more, and (4) increasing oil prices. Conventional wisdom suggests that stocks/use ratios less than 12–20% confers vulnerability to potential price spikes, which was seen in the 1970s. Energy costs are a major factor in farming and fertilizer production, thus higher petroleum prices can push up crop prices in addition to making the use of biofuels more attractive as a fuel substitute. Shorter-term factors (i.e., months to a few years) included the RFS Program and other biofuel programs in the United States such as the Low Carbon Fuel Standard and low wheat harvests in Ukraine and Australia in 2006–2007. European biofuel markets are predominantly diesel based and thus not as coupled with cereal markets as U.S. biofuel programs. Short-term factors (i.e., weeks to months) included export bans that further exacerbated worries over food prices and availability, and subsequent over-restocking to prevent future shortages. These short-term factors applied mostly to

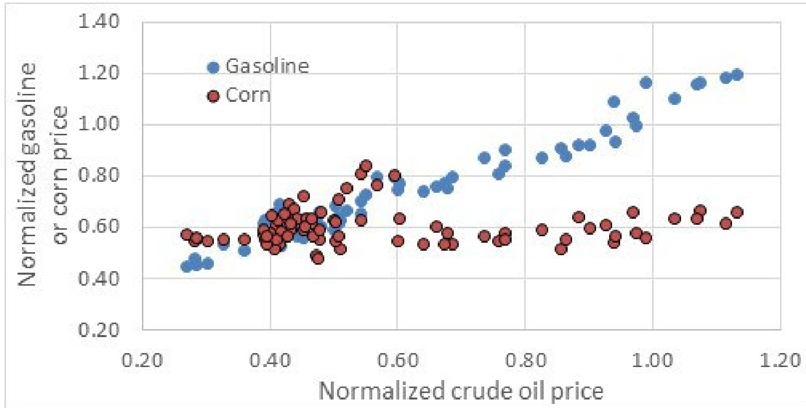
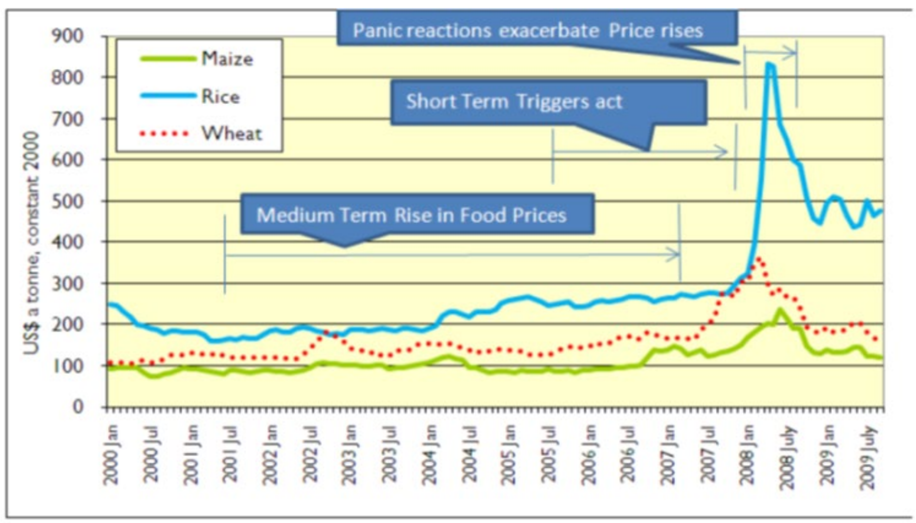


Figure C.3. Monthly normalized crude oil prices for January 2000 through September 2006 compared to normalized prices for gasoline and corn. For each commodity, the normalized price represents the price in a given year divided by the average price over all years shown.[2]



1 tonne = 2,200 pounds

Figure C.4. Prices of three main cereals on world markets (monthly IMF commodity prices, deflated by the U.S. GDP deflator). Source: Wiggins (2010) (used with permission).

[2] Gasoline price is from the EIA, (https://www.eia.gov/dnav/pet/pet_pri_gnd_dcus_nus_m.htm), corn prices are from Macrotrends, " Corn Prices - 45 Year Historical Chart," https://www.macrotrends.net/2532/corn-prices-historical-chart-data, and crude oil price are from EIA, Monthly Energy Review Table 9.1.

wheat and rice, but prices on one cereal can propagate to others because of their mutual effects on land rents. Kazakhstan, Ukraine, Russia, and Argentina all imposed some form of export restriction on wheat (e.g., bans, quotas, taxes). India, one of the primary exporters for rice, banned exports along with Vietnam and Egypt, causing panic in rice markets, and countries like the Philippines, Malaysia, Iran, and the EU all increased imports by 30–71% over prior levels. Thus, there were many factors at play that influenced corn as well as other cereal prices between 2006 and 2008. One of these, however, appears to be domestic biofuel use in the United States, which appears to be responsible for 20–40% of the increase in the price of corn over this period.

Regardless, there was a notable increase in the price of corn between 2006 and 2009, whether measured by corn prices globally (Figure C.4), prices in the U.S. stock market (Figure C.5), or in prices estimated received by farmers in the U.S. (Figure C.6). Corn/soy farmers often make decisions on what to plant based on what they grew the year prior, because these crops are commonly grown in rotation. But, by winter of 2006, corn prices in the stock market had already reached above $4.00 per bushel, levels not seen in any period prior to that aside from the price spikes of 1996.[3] These increases in late 2006 were concurrent with the switch from MTBE to ethanol in summer of 2006. These were also concurrent with the first year of the RFS1 mandates which went into effect in 2006, though production already exceeded the RFS1 mandates by over one billion gallons (see section 6.3.1). Similar price trends are seen in the prices paid to farmers, which began to increase above historic levels in winter 2006 (Figure C.6). Importantly, this increase in corn prices in the winter of 2006 was *before* the extreme price spikes of 2008 (Figure C.4) and immediately prior to the large jump in corn acreage in 2007 (see Chapter 5).

[3] USDA studied these price spikes in the 1970s, 1990, and 2006–2008 and found that they often have similar causes. Unique among these was for 2006–2008 which saw the expansion of the U.S. biofuels market. The spikes in the 1970s and 1990s were comparable to one another, and much smaller than that in 2006–2008 (Peters et al., 2009).

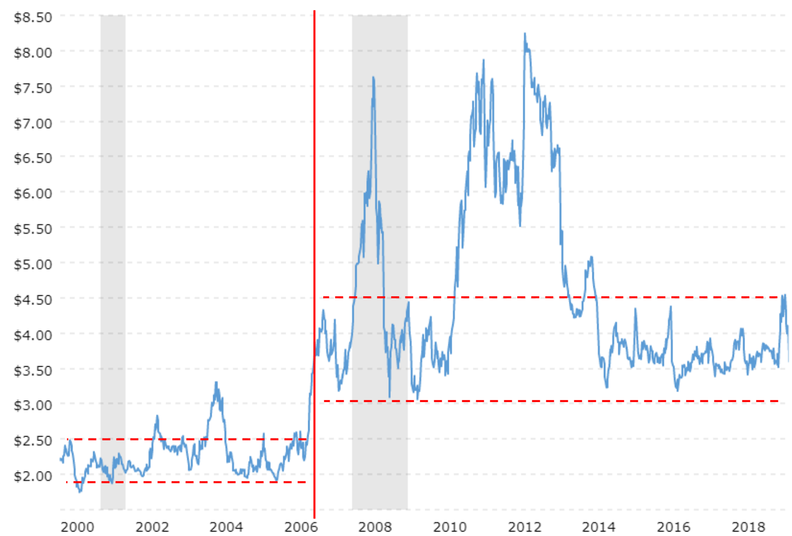


Figure C.5. U.S. stock market daily prices (in dollars per bushel) of corn for 2000–2019.[4] Economic recessions in the United States (in gray) and the price in November 2006 (vertical red line), roughly one year before EISA but the same year as the switch from MTBE to ethanol in much of the United States. Ignoring the peaks, the long-term price from 2000–2006 generally varied from $2 to $2.50 per bushel for most months, while after November 2006, the price varied from $3-$4.50 per bushel (dashed red lines).

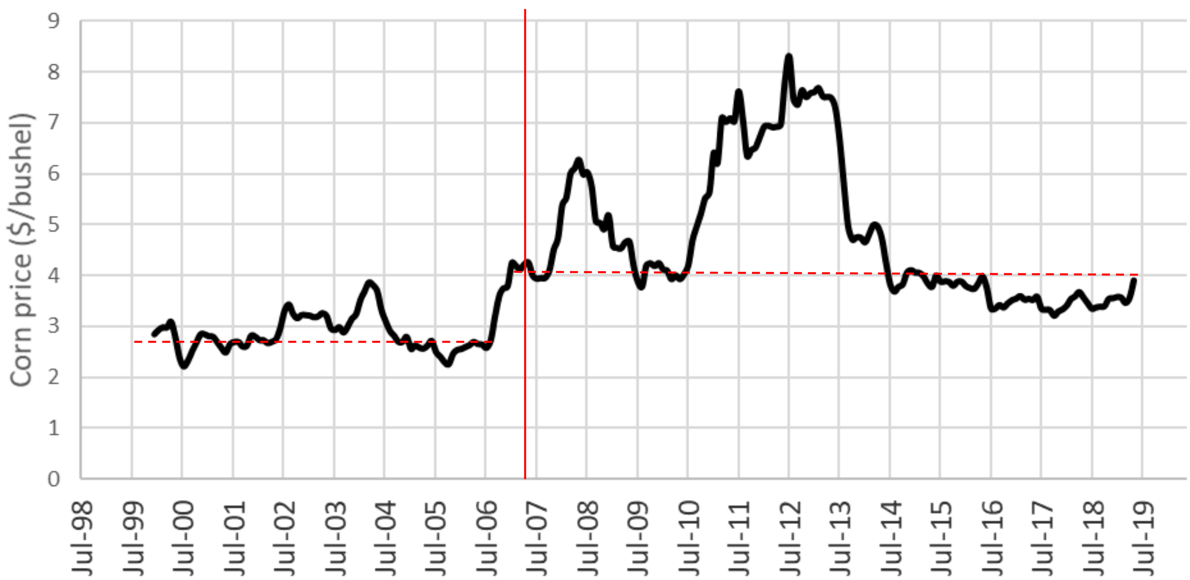


Figure C.6. Monthly prices (in real 2018 dollars per bushel) received by farmers in the United States from 1990 to 2019.[5] November 2006 is shown for reference (vertical red line), roughly one year before EISA but concurrent with the switch from MTBE to ethanol in much of the United States, are shown for reference along with the historical prices of $2.75 and $4.00 (horizontal dashed red lines) and November 2006 (vertical solid red line).

[4] The source for this graphic is the macrotrends database: Corn Prices - 59 Year Historical Chart (https://www.macrotrends.net/2532/corn-prices-historical-chart-data).
[5] Data are from the USDA ERS, specifying the "Prices received by farmers" for "Corn grain" on a "Monthly" basis from "2000-2019" (Source: https://data.ers.usda.gov/FEED-GRAINS-custom-query.aspx#).

However, ignoring the anomalous price spikes of 2008 and 2011–2013, which are driven by more short-term factors, it is clear from both data sources that the historical norms of around $2.75 per bushel received by farmers before late 2006, changed to a minimum of roughly $3.50–$4.00 per bushel thereafter (e.g., 2009–2010, 2014–2019). It is easy to lose sight of the apparent change in the baseline price with such large swings in peak prices. These long-term changes are the type of signal expected from a policy change rather than weather or other punctuated events. A $0.75–$1.25 increase may not appear large, but that represents a 27–45% increase in potential revenue from farming corn, if all else remains equal.

It seems reasonable that farmers were aware of the higher prices for corn in the winter of 2006, one year before the passage of the Energy Independence and Security Act (EISA), and decided to grow more corn in 2007, which was the year of the 15 million acres increase in corn acreage (see Chapter 5, Figure 5.8).

The second period of price increases from 2011 to 2013 is not a primary concern for the RtC3, since it occurred after most expansion of U.S. corn ethanol production. By 2010, ethanol was already at 9.3% of the gasoline pool. A major drought in 2012 is considered responsible for the price increases in 2012 and 2013, and several weather-related events around the world but particularly Russia were responsible for the price increase in 2011 (Trostle, 2011). There was an increase in corn production between 2011 and 2013 associated with this price increase as there was in 2007 (see Chapter 5, Figure 5.8). This increase was not nearly as abrupt as in 2008, and it did not lead to a sustained increase in corn acreage as had been observed previously.

C.1.3. Octane Value of Ethanol

An important change adopted by the fuels industry was a transition in the 2005–2010 period by oil refineries from producing "finished gasoline" (e.g., 87 octane gasoline), which could be sold as-is at a retail station, to a low-octane blendstock, which had to be blended with an oxygenate (e.g., ethanol to make E10) to be legally sold at a retail station. The low-octane blendstocks produced by refiners were called blendstocks for oxygenate blending, or BOBs. These BOBs were only about 84 octane, which would then be blended with ethanol at the terminal to raise the octane value to 87. The process for blending the ethanol with the low-octane BOBs is termed "match blending."[6] This process reduced the production cost of the gasoline they produced and improved the economics for blending ethanol into gasoline.

[6] Under match-blending, the terminal operator would mix the cheaper 84 octane BOB with ethanol: 90% x 84 octane BOB + 10% 115 octane ethanol = E10 at 87 octane. Under splash-blending, the terminal would mix the more expensive 87 octane finished gasoline with ethanol: 90% x 87 octane gasoline + 10% x 115 octane ethanol = E10 at 90 octane. Thus, under splash-blending the E10 is more expensive than under match-blending (i.e., because the 87 octane feedstock is more expensive than the 84 octane feedstock).

This transition from splash-blending (discussed in Chapter 6) to match blending occurred solely for the conventional gasoline pool, because reformulated gasoline (RFG) was already match-blended with an oxygenate. This is because RFG containing 10% ethanol is required to meet the same volatility limit as RFG without ethanol. Ethanol can therefore only be added to special low volatility RFG blendstocks (reformulated blendstock for oxygenate blending or RBOB), as the addition of 10% ethanol to finished RFG would otherwise exceed the volatility limit for RFG. Since refiners must produce a special blendstock for the blending of ethanol in RFG areas to address volatility, they took the opportunity to simultaneously reduce the octane value of that blendstock requiring that high octane ethanol be blended in downstream of the refinery.

Most conventional gasoline receives a 1 psi waiver from the gasoline volatility requirements when blending up E10. Therefore, refiners are not forced to produce a separate gasoline blendstock for ethanol blending. This may have enabled ethanol blending into the gasoline pool early on. This is because gasoline tankage is limited at both refinery and downstream terminals, which does not allow for the storage of an additional gasoline blendstock—a low-octane BOB in addition to a finished gasoline. For this reason, entire conventional gasoline submarkets would need to convert over from producing finished gasoline to producing BOBs for the logistics to work out to converting to BOBs and match blending at terminals. Therefore, a sort of "domino effect" occurred as parts of the conventional gasoline pool converted over to BOBs as more ethanol became available.

As domestic ethanol production and distribution expanded from 2005 to 2010 (Duffield et al., 2015), conventional gasoline markets began to transition from splash-blending ethanol with finished gasoline to match blending ethanol with BOBs to take advantage of the lower cost of producing BOBs relative to finished gasoline. By 2015, effectively all gasoline in the United States, including both RFG and conventional gasoline, was produced by match blending 10% ethanol to BOBs. This transition is evident in Figure C.7, which plots data from

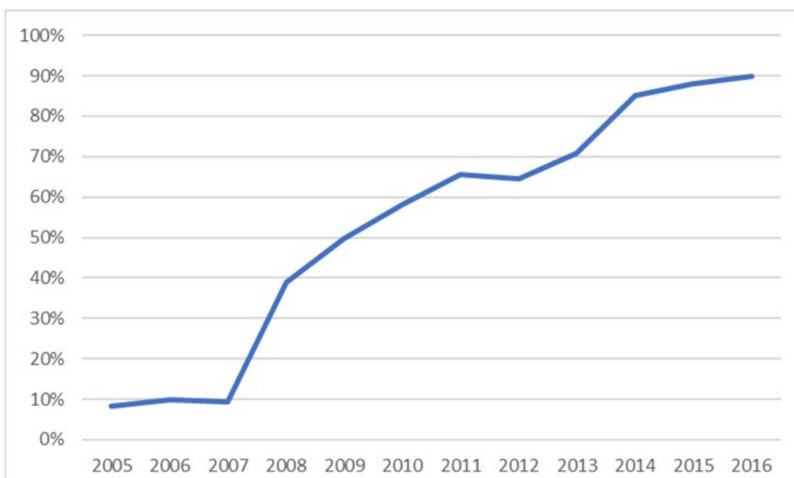


Figure C.7. Fraction of conventional gasoline made from BOBs.[7]

[7] Source: Petroleum Supply Annual, volume 1, Table 19 (2005–2018), Table 19 for BOB and 20 for conventional gasoline (2009 and 2010), and Table 20 for BOB and 21 for conventional gasoline (2011+).

the Energy Information Administration (EIA) on BOBs used by blenders to produce conventional gasoline versus the total volume of conventional gasoline produced.

One key impact of the market transition to match blending is the inertia for continued use of ethanol to produce finished gasoline, even when it may not be cost effective. The refiner-produced BOBs cannot be sold as gasoline without the addition an oxygenate. With MTBE no longer in use after 2006, and ethanol widely available and indeed mandated in many states, ethanol was, and remains, the logical choice for a gasoline blendstock in the United States (see Appendix D for an example refinery analysis). Assuming that the RFS Program was no longer in place, even if these economics did change (i.e., much lower crude oil prices and no federal ethanol policy in place), refiners would likely still continue to blend in ethanol, at least for a period of time, into the future. This is because the refining industry would require significant investments to change their operations to make up the loss in octane, make up for the loss in ethanol volume in the gasoline pool, and perhaps need to make changes to the fuel distribution network as well to remove ethanol from the gasoline pool. Thus, even in situations where ethanol is not cost effective to blend and the RFS Program was not binding, refiners are likely to continue to use ethanol at least in the short term or longer. This means that they would likely continue to produce BOBs and blend ethanol unless market projections suggest it would not be cost effective to blend ethanol for an extended period of time (e.g., several years). For example, after a significant decrease in the price of crude oil in late 2014, ethanol was more expensive than gasoline in many parts of the United States, particularly in 2016. Despite this, ethanol would likely have continued to be blended into nearly all gasoline even without the RFS Program in place (U.S. EPA, 2022).

The octane blending value of ethanol into gasoline can be estimated by refinery models, which can evaluate ethanol's octane value relative to other refinery gasoline streams. One such study estimated the octane value of ethanol for blending into gasoline in the year 2020 and is summarized in Table C.1.[8] The different ethanol values between conventional and reformulated gasolines allowed the estimation of ethanol's volatility cost in addition to its octane value for when ethanol is blended into RFG, which is also summarized in Table C.1.

Table C.1. Octane and Reid Vapor Pressure (RVP) Blending Values by Fuel Type ($ per gallon). Shown are the estimated blending values for two different types of gasoline (Regular, Premium) in either conventional gasoline (CG) or RFG areas, from the value of octane or RVP.[8]

Gas type	Area	Valuation	Summer	Winter
Regular	CG & RFG	Octane Value	0.43	0.29
	RFG	RVP Value	-0.28	
Premium	CG & RFG	Octane Value	0.27	0.26
	RFG	RVP Value	-0.29	

[8] Modeling a No-RFS Case; ICF Incorporated; Work Assignment 0,1-11, EPA contract EP-C-16-020; July 17, 2018 (see Appendix D).

Ethanol's high octane has the highest value in the regular grade gasoline pool, both conventional gasoline (CG) and RFG, at an estimated 43 cents per gallon. This octane value drops to 29 and 27 cents per gallon in winter and for premium gasoline in the summer, respectively. Conversely, ethanol's volatility cost is just under 30 cents per gallon in the RFG gasoline pool. Thus, blending ethanol into summertime RFG has about 15 cents per gallon of net value, and is valued about the same as gasoline when blended into summertime premium RFG.

As discussed in section C.1.1, the octane value of ethanol was also estimated by Babcock (2013), where he noted that the ethanol:gasoline price ratio exceeded what would be expected if ethanol were only valued for its energy content (i.e., 0.7). Babcock found that ethanol's octane value played an important role in improving the blending economics of corn ethanol if the RFS Program did not apply.

C.2. Production Capacity Buildout

While domestic production capacity does not drive domestic ethanol consumption, it does place an upper limit on the volume of ethanol that can be consumed without imports. Insofar as total production capacity in any given year exceeds the RFS volume requirement for that year or near-future years, there is reason to believe that investors expected the market to provide an incentive to use ethanol above that provided by the RFS Program. Thus, existing and under construction production capacity can provide an indirect indication of whether the RFS volume requirements were expected to be binding.

C.2.1. Construction of New Facilities

Investment in new or expanded production capacity is not only a function of the demand for ethanol at a given point in time, but also the longer-term outlook for ethanol demand. Investors must take into account the fact that a typical ethanol plant requires 18–24 months to plan, construct, and become operational, and that market changes many years in the future will impact the return on their investment. An investor's decision to build a new ethanol production facility is a function of many factors, including:

- RFS mandates
- State ethanol mandates
- State and federal support (including multiple USDA initiatives) for building new facilities, such as loans, grants, and tax credits
- Crude oil and corn price projections
- Farm organizations looking to have greater assurance of a market for corn and additional control over when and how corn prices are set

Since these factors applied at different times and to different degrees, investor decisions to build a new ethanol plant could likewise have differed from year to year.

The Energy Policy Act (EPAct), which created RFS1, was enacted in August of 2005. In the remaining months of 2005, it is unlikely that investors, responding to the promise of a future market for ethanol driven by the RFS standards, could have moved beyond the planning stages and begun actual construction of new facilities. Instead, it is more likely that any response to the passage of EPAct in terms of new ethanol plant construction would have begun in 2006. Nevertheless, there was an increase in new plant construction in 2005 (gray band, Figure C.8, and Chapter 6, Figure 6.6). This increase in 2006 would most likely have been driven by other factors such as the impending phaseout of MTBE (see section C.3).

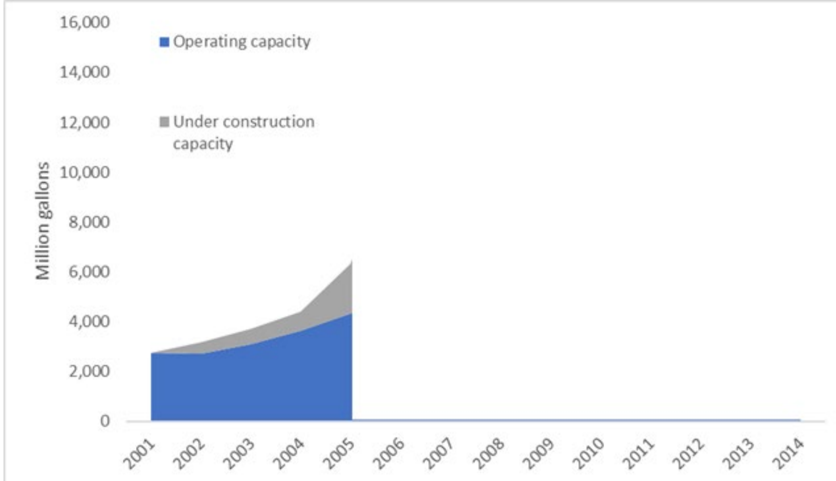


Figure C.8. Ethanol production capacity through 2007, prior to RFS2. (Source: For Figure C.8–C.10, operating capacity and under construction capacity are the same as Figure 6.6 in main text.[9]).

EISA, which established RFS2, was enacted in December of 2007. Thus, for all of 2006 and 2007, investors would only have had the RFS1 volume requirements on which to base their investment decisions, and unless speculating, would likely not have based decisions in those two years on the RFS2 volume requirements as they did not yet exist. Nevertheless, new construction rose dramatically in these two years, far above the highest level under RFS1; the 2012 requirement under RFS1 was 7.5 billion gallons,

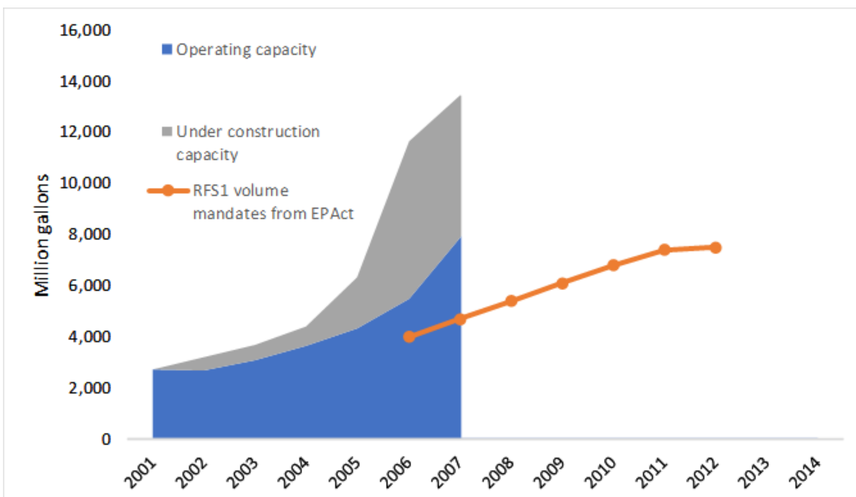


Figure C.9. Ethanol production capacity through 2005, prior to RFS1. (Source: See Figure C.8.).

[9] Renewable Fuel Association's annual "Ethanol Industry Outlook," https://ethanolrfa.org/publications/outlook/. There is no parallel government dataset to the authors' knowledge.

while the sum of operating and under construction capacity at the end of 2007 was 13.4 billion gallons. This suggests that in 2006 and 2007 investors were responding to future outlooks for ethanol demand that were based on factors other than the RFS Program (Figure C.9).

In the first few years following the enactment of EISA at the end of 2007, ethanol production capacity continued to grow, but at a considerably slower rate (Figure C.10). Having already reached 13.4 billion gallons of operating plus under construction capacity in 2007, the industry only needed to add an additional 1.6 billion gallons of production capacity to reach the maximum implied RFS2 conventional renewable fuel mandate of 15 billion gallons. Thus, insofar as future volume requirements under the RFS2 played a role in investor's decisions to build additional construction capacity after 2007, those RFS2 volume requirements could be implicated in providing an incentive only for capacity in excess of 13.4 billion gallons. Incentives for volumes under this amount are possible too, but would have been significantly more risky as they would have been contingent on passage of EISA.

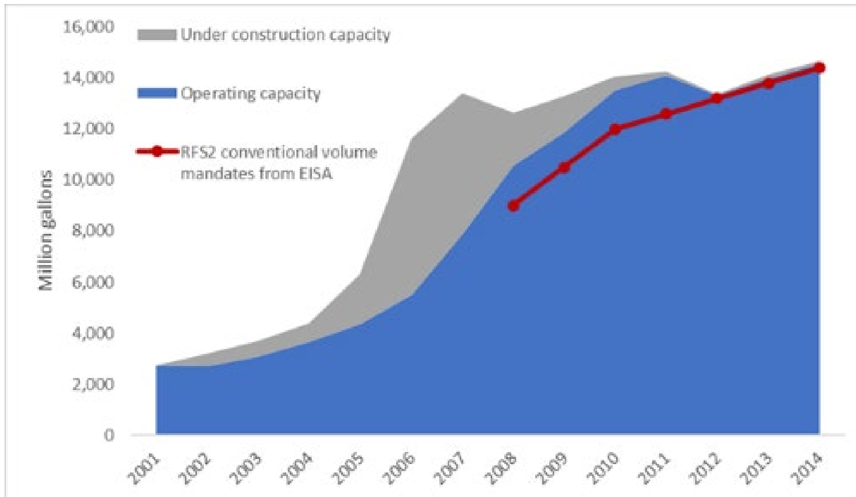


Figure C.10. Ethanol production capacity after RFS2 mandates were established. (Source: See Figure C.8).

C.2.2. State Loans, Grants, and Other Tax Credits

A variety of state programs provided some form of economic incentive to build or expand corn ethanol production facilities between 2005 and 2018 (Duffield et al., 2015). These programs included grants, loans, tax credits, and rebates of varying sizes and applicability, with various beginning and ending dates (Table C.2). These state programs were legally independent of the RFS Program and may or may not have been implemented even if the RFS Program had not existed. Thus, they may have helped to expand ethanol production capacity. It is not possible to determine the dependence of these state economic programs on the RFS Program. These financial incentives are distinct from state mandates for ethanol or low-carbon fuels more generally, which are discussed in section C.4.

Table C.2. State incentives for new corn ethanol production capacity.

State	Title	Type
AL	Agriculturally Based Fuel Production Wage and Salary Tax Credit	Tax credit/exemption
AR	Biofuels Industry Development Grants	Grant
AR	Biofuels Production Incentive	Rebate
CA	Alternative Fuel Production Tax Credits	Tax credit/exemption
FL	Ethanol and Biodiesel Fuel Production Grant	Grant
GA	Ethanol Motor Fuel Production Tax Credit	Tax credit/exemption
GA	Ethanol Production Investment Tax Credits	Tax credit/exemption
IA	Ethanol Production Incentive	Tax credit/exemption
IA	Biofuel Production Facility Tax Credit	Tax credit/exemption
IA	Ethanol Production Incentive	Tax credit/exemption
IL	Alternative Fuel Grants and Rebates	Grant/rebate
IL	Alternative Fuel Loan Program	Loan
IL	Alternative Fuel Production Tax Credit	Tax credit/exemption
IN	Alternative Fuel Production Facility Tax Exemption	Tax credit/exemption
KS	Biofuels Production Tax Credit	Tax credit/exemption
KY	Ethanol Production Tax Credit	Tax credit/exemption
ME	Ethanol Production Tax Credit	Tax credit/exemption
MN	Alternative Fuel Production Loans	Loan
MN	Biofuel Production Facility Tax Credit	Tax credit/exemption
MS	Renewable Fuel Production Facility Tax Credit	Tax credit/exemption
NC	Biofuels Production Tax Exemption	Tax credit/exemption
NC	Biofuels Production Incentive	Grant
ND	Ethanol Production Incentive	Rebate
OH	Alternative Fuel Development and Deployment Grants	Grant
OR	Biofuels Production Tax Credit	Tax credit/exemption
PA	Renewable Energy Property Tax Credit	Tax credit/exemption
PA	Biofuels Investment Tax Credit	Tax credit/exemption
TN	Alternative Fuel Production Tax Incentives	Tax credit/exemption
TX	Biofuels Production Facility Grants	Grant
TX	Biofuels Business Planning Grants	Grant
TX	Ethanol Production Incentive	Rebate
WA	Renewable Fuel Production Grants	Grant
WA	Biofuels Production Incentive Fund	Loan

Source: U.S. Department of Energy (DOE), Alternative Fuels Data Center

C.2.3 Projected Crude Oil Prices

In addition to actual crude oil prices (discussed in section C.1.1), projected crude oil prices likely played a role in the attractiveness of ethanol production as an investment and the production capacity buildout. It is likely that parties considering investing in new production capacity monitored not only historical trends in crude oil prices, but also future projections of crude oil prices. Investors may have looked to crude oil price projections from a variety of sources. Here is a review of the potential impact of price projections from EIA's Annual Energy Outlook (AEO) on investor outlooks for future ethanol profitability.

In the few years just prior to and including the EPAct of 2005, actual crude oil prices had been increasing (Figure C.11). However, the AEOs for 2003, 2004, and 2005 were still projecting that future crude oil prices would be little different than the average of the previous decade (Figure C.11).

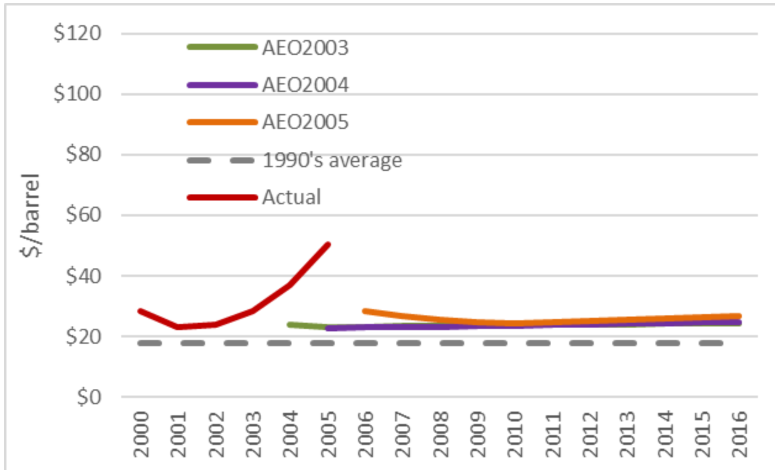


Figure C.11. AEO projections of crude oil prices in 2003, 2004, and 2005, and actual prices. Also shown are the long-term average from the 1990s.[10]

As a result, it seems unlikely that the crude oil price projections available in AEO would have inspired confidence in investors for a future ethanol market, as those projections suggested that ethanol would not be more economically attractive in comparison to gasoline in the coming years than it had been previously. Investors may have had crude oil price projections other than the AEO projections available to them through 2005 that painted a substantially more positive picture for the future of ethanol demand. To the degree that investors relied on AEO alone for their crude oil price projections, it is likely that their decisions to build new ethanol plants in the 2003–2005 time frame were based instead on other factors, such as expectations about the MTBE phaseout.

In 2006 and 2007, EIA's projections of future crude oil prices increased, though not as much as actual prices (Figure C.12). Nevertheless, long-term projections for crude oil prices were more than double the levels of the previous decade. Combined with the fact that actual crude oil prices had increased in every year since 2001, these projections may have given investors additional confidence that ethanol

[10] For Figures C.11–C.13, "Actual" and "1990's average" are based on the EIA Crude Oil Price Summary (Table 9.1, https://www.eia.gov/totalenergy/data/browser/?tbl=T09.01). The AEO projections are based on the AEO archived report for the corresponding year (https://www.eia.gov/outlooks/aeo/archive.php).

would be profitable in the long term. This provides one possible explanation for the significant increase in new construction in these two years (2006 and 2007), before the RFS2 volume mandates were established.

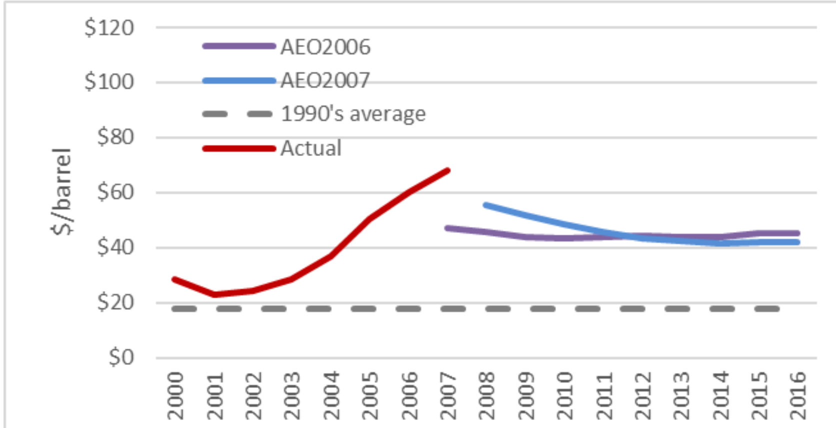


Figure C.12. AEO projections of crude oil prices in 2006 and 2007 and actual prices. Also shown are the long-term average from the 1990s.

In 2008 and 2009 (Figure C.13), EIA's projections of future average crude oil prices continued to increase relative to previous projections. Moreover, in AEO2009 for the first time in the 2000s, EIA projected that crude oil prices would continue increasing in future years, despite the fact that actual crude oil prices decreased in 2009. This projection may have given investors greater confidence in the profitability of ethanol in the long term.

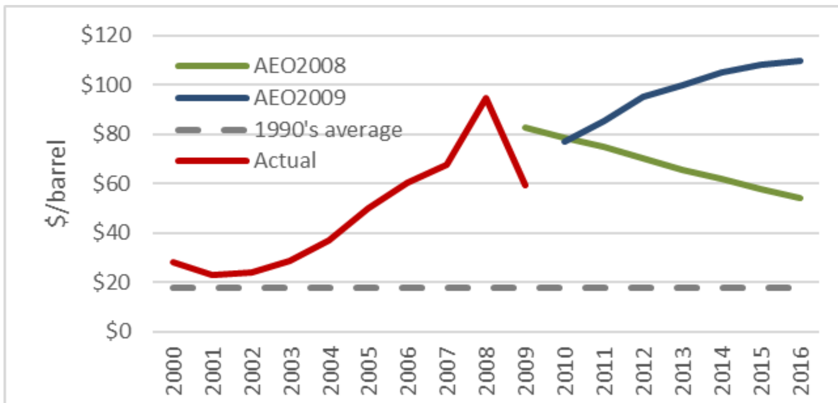


Figure C.13. AEO projections of crude oil prices in 2008 and 2009.

C.3. MTBE Phaseout

C.3.1. Concerns about MTBE and State and Federal Reactions

In the 1990s there was growing concern about the health effects of MTBE, first in the context of exposure to gasoline vapors during vehicle refueling and then in the context of groundwater contamination resulting from leaking underground storage tanks. Related to this concern, the California Air Resources Board (CARB) made a formal request to EPA in 1999 for a waiver from the requirement to use oxygenates in reformulated gasoline. Although that request was ultimately denied in 2001, it was in the context of considering that waiver request that the EPA made an announcement in 2000 that it intended to impose a nationwide ban on the use of MTBE in gasoline (U.S. EPA, 2000a, b). By the end of 2004, 19 states had adopted legislation banning MTBE and 14 of these laws had gone into effect (U.S.

EPA, 2010). Figure C.14 shows that by 2006 (the first year of the RFS1), these state MTBE bans covered roughly 45% of all the gasoline consumed in the United States.

At the federal level, multiple bills banning MTBE were considered by Congress, but none were ultimately adopted (ICIS, 2006). At the same time, Congress also considered providing liability protection for refiners using MTBE under the premise that they had no choice but to use an oxygenate in RFG and Oxyfuels Programs, and that EPA had implicitly approved of MTBE's use inasmuch as EPA knew it was a primary option when the RFG Program was originally implemented (McCarthy and Tiemann, 2006). The potential for some sort of liability protection, as well as the lack of sufficient infrastructure in the early 2000s for distributing and blending ethanol (Duffield et al., 2015), may have given refiners confidence to continue using MTBE despite state bans and concerns expressed by EPA and the public during this time frame.

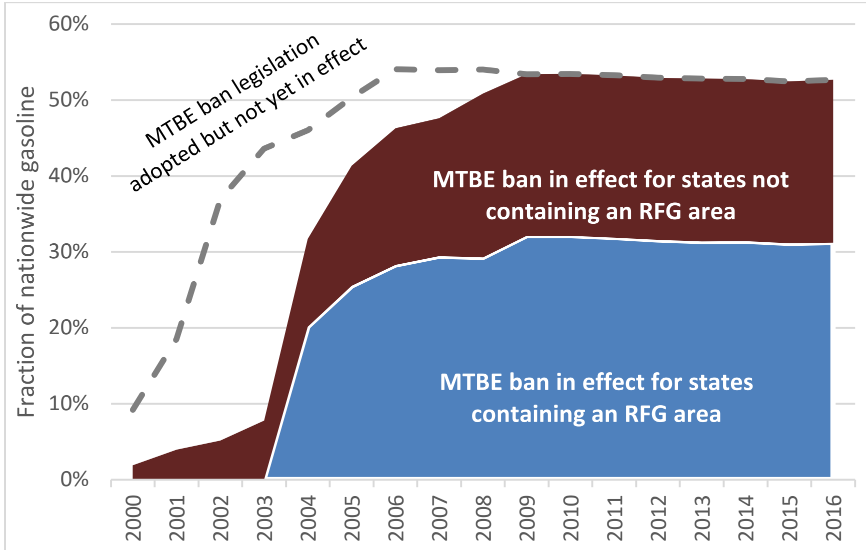


Figure C.14. Fraction of nationwide gasoline covered by state MTBE bans. (Source: State bans from EPA document "State actions banning MTBE (Statewide)" and gasoline consumption by state from EIA's State Energy Data System (SEDS). Source: EIA (2023).

The EPAct of 2005 was signed into law on August 8, 2005, and it established the RFS Program (RFS1). Even though the EPAct went into effect in 2005, the first year in which the volume requirements of the RFS Program applied was in 2006 (discussed in Chapters 1 and 6). Although the EPAct did not include a nationwide ban on the use of MTBE as had previous bills that Congress considered, neither did it include any form of liability protection that had been sought after by refiners who blended MTBE into gasoline. Instead, EPAct eliminated the oxygen requirement for federal RFG and created the RFS Program. Although the oxygen requirement for federal RFG was removed, the emission standards for RFG were neither eliminated nor modified, and the use of an oxygenate continued to be the most economical way to meet those emission standards.[11] The combination of these changes, in addition to the

[11] The removal of the oxygenate requirement in the RFG Program appears to have had almost no impact on the use of oxygenates in RFG areas. This is because compliance with RFG is generally certified through "the Complex

lack of any explicit or implicit liability protection, meant that refiners had little incentive to continue using MTBE and may have faced considerable liability. Consumption of MTBE in all gasoline outside of California dropped by about 80% between 2005 and 2006 ([Figure C.15](#)). In the same time frame, ethanol use increased. This switch is considered here to be largely independent of the actual RFS Program, even though both the RFS1 and the removal of the oxygenate requirement occurred in the same Act.

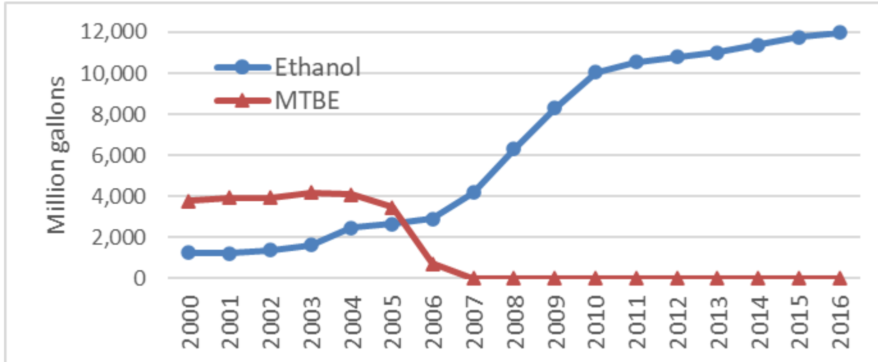


Figure C.16. Consumption of MTBE and ethanol in all gasoline outside of California. Source: EPA batch report data (required under 40 CFR 80.75 and 80.105. See https://www.epa.gov/fuels-registration-reporting-and-compliance-help/public-data-gasoline-fuel-quality-properties).

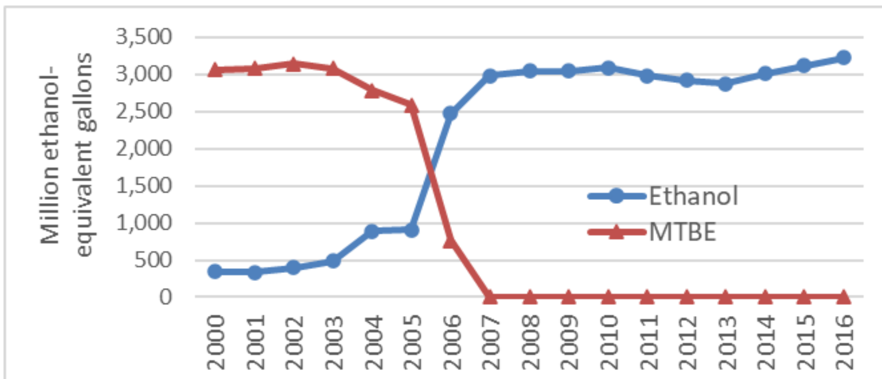


Figure C.15. Consumption of MTBE and Ethanol in RFG outside of California. Source: EPA batch report data (required under 40 CFR 80.75 and 80.105. See https://www.epa.gov/fuels-registration-reporting-and-compliance-help/public-data-gasoline-fuel-quality-properties).

The switch from MTBE to ethanol is even more clearly evident in the RFG pool where oxygenates continued to be used despite the removal of the 2% oxygen requirement (Figure C.16).

Model," which was created in 1994 and translates fuel properties (oxygen, RVP, aromatics, etc.) into a prediction of vehicle emissions. After the EPAct, refiners were free to produce RFG with any fuel properties so long as the Complex Model certified that the resulting fuel met specified emission standards. It is difficult and expensive to produce RFG-compliant fuel without an oxygenate that still complies with the applicable emission standards as predicted by the Complex Model, thus oxygenates though not specifically required, were still the preferred method for compliance. There was virtually no change in the level of oxygenate use in RFG after the 2% oxygen requirement for RFG was removed on May 8, 2006 (71 FR 26691) as evidenced through RFG batch report data (Figure C.16, (U.S. EPA, 2021).

C.3.2. Transition from MTBE to Ethanol in Non-RFS Fuels Programs

The primary uses of MTBE in the United States were in the federal RFG Program, the federal Oxygenated Fuels (Oxyfuels) Program, and the California RFG program. Both RFG programs were designed to address multiple pollutants from both tailpipe emissions and evaporation, while the Oxyfuels Program was designed to address tailpipe CO emissions. The oxygenate requirements of these three programs are summarized in Table C.3.

Table C.3. U.S. programs requiring the use of an oxygenate.

Program	Minimum Oxygen Requirement	Season to Which the Oxygen Requirement Applied
Federal RFG	2.1wt%[a]	Annual
Federal Oxyfuels	2.7wt%	Winter
California RFG	1.8wt%[b]	Winter

wt: weight

[a] The Energy Policy Act of 2005 eliminated the oxygenate requirement for federal RFG, and EPA put this change into effect in 2006.

[b] Applies only to the South Coast area and Imperial County.

Although the requirement for the use of an oxygenate in the federal RFG Program was removed in 2006, meeting the applicable emission standards under the Program was considerably more difficult (i.e., more costly) without an oxygenate. As no other oxygenates were available in sufficient quantities at competitive prices that did not also potentially share the same risks that MTBE had (California Energy Commission, 1999), ethanol replaced MTBE in federal RFG. The rapid need for an oxygenate in the federal RFG Program as MTBE was being phased out is evident in the shift of ethanol consumption from conventional gasoline to RFG in between 2005 and 2006 (Figure C.17). In 2007, as ethanol production increased, its use in conventional gasoline once again began to increase.

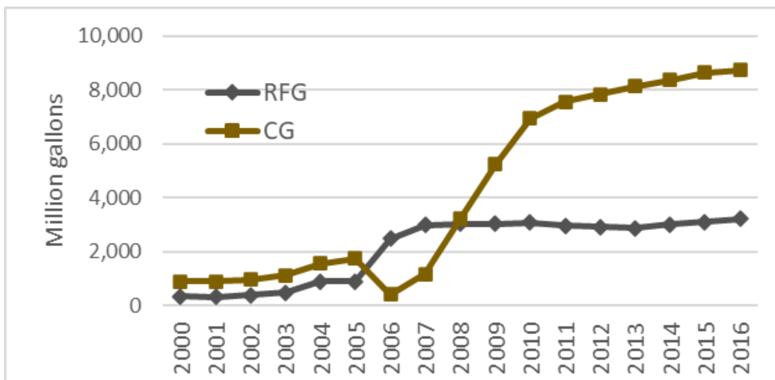


Figure C.17. Consumption of ethanol in reformulated gasoline (RFG) and conventional gasoline (CG) outside of California. Source: EPA batch report data (required under 40 CFR 80.75 and 80.105. See https://www.epa.gov/fuels-registration-reporting-and-compliance-help/public-data-gasoline-fuel-quality-properties).

If the RFS Program had not been enacted through the EPAct, ethanol would still have been an attractive replacement for MTBE that would meet the emission standards for federal RFG. However, the transition from MTBE to ethanol may not have occurred as quickly as it did. The creation of the RFS Program provided an additional reason to use corn ethanol—a biofuel and an oxygenate—to serve the purposes of both the RFG and RFS Programs.

Ethanol also replaced MTBE in the federal Oxyfuels Program. However, the number of areas subject to the federal oxyfuels requirements have declined since the 1990s as areas in nonattainment for CO have come into attainment. By the year 2000, most areas in the United States were in attainment for CO, so the Oxyfuels Program likely only played a minor role in driving the increase in ethanol nationally (Figure C.18). The volume of ethanol consumed in all three programs (i.e., federal Oxyfuel, federal RFG, and California RFG) over time is shown in Figure C.18. All of these replacements had largely occurred by 2007, the year EISA was enacted, totaling 4 billion gallons of ethanol or more.

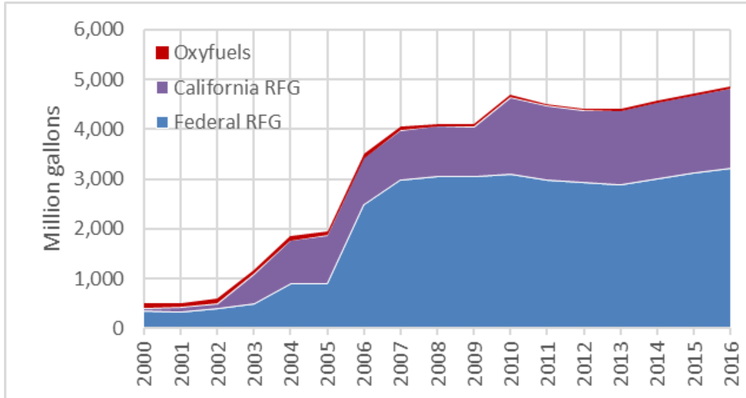


Figure C.18. Use of ethanol in federal RFG, federal Oxyfuels, and California RFG. Source: California RFG: EIA's SEDS. Federal RFG: EPA gasoline batch data. Oxyfuels: independent EPA analysis of program applicability by state.

C.4. Additional Ethanol Mandates and Markets

C.4.1. State Ethanol Mandates

Several states implemented mandates for the use of ethanol in the same time frame that ethanol consumption nationwide was increasing (Table C.4). The effect of mandates is easier to quantify than incentives, since a mandate must be adhered to and an incentive may or may not be used.

Table C.4. State mandates for ethanol.[a]

State	Blend Requirement	First Applicable Year	Last Applicable Year
Minnesota	10%[b]	1997	Still in effect
Hawaii	10%[c]	2006	2015
Oregon	10%	2007	Still in effect
Missouri	10%	2008	Still in effect
Washington	2%	2009[d]	Still in effect
Florida	10%	2011	2013

[a] Does not include biodiesel mandates or mandates for ethanol use in state vehicle fleets.
[b] Between 1997 and 2002, the Minnesota requirement was 2.7 weight% oxygen and was not specific to ethanol. Nevertheless, ethanol was the primary oxygenate used. Between 2003 and 2012, the requirement was for 10 volume% ethanol. For 2013 and thereafter, the requirement was for 10 volume% "conventional biofuel," of which ethanol was the primary option available.
[c] This requirement applied to 85% of gasoline sold in Hawaii.
[d] Actual start date was 12/1/2008.

Most of these state ethanol requirements included some exemptions such as for aviation gasoline, gasoline used in nonroad and marine engines, and/or premium gasoline. There were also ethanol mandates in other states that were conditioned on certain triggers. For instance, both Louisiana and Montana require ethanol in gasoline beginning at the point in time when ethanol production capacity in each state reaches a certain threshold. However, no ethanol facilities have been constructed in these two states by the time of writing. Similarly, Pennsylvania requires 10% ethanol, but the ethanol must be produced entirely from cellulosic feedstocks and the state production capacity of cellulosic ethanol must first reach 350 million gallons. Actual production of cellulosic ethanol in 2018 for the nation as a whole was only 8 million gallons.

EIA's State Energy Data System (SEDS) (EIA, 2021a) provides historical ethanol consumption for each state. These data include ethanol consumption resulting from all programs, incentives, and market conditions, and therefore do not represent ethanol consumption due solely to each state's ethanol mandate. However, these state data were adjusted in an attempt to estimate the possible impacts of the state mandates alone. To do this, it was assumed that ethanol use in a given state and year due to the state mandate is no higher than the state's mandate, but could be lower due to exemptions in each state's regulations for certain categories of gasoline. In practice, this meant using the actual ethanol volume from SEDS for a given state and year only if the ethanol concentration in that state was less than the applicable state mandate. The SEDS data were also adjusted to account for overlap with the federal RFG and Oxyfuels programs. This adjustment only affected two states (Oregon and Missouri). Since federal RFG also applied in some areas of Missouri, accounting for about 30% of all gasoline used, it was assumed that the ethanol mandate in Missouri only applied to the remaining 70% of the gasoline sold in the state. Similarly, since the last year of applicability of the oxyfuels program in Oregon was in 2007, the same year that the state ethanol mandate went into effect, it was assumed that ethanol use in Oregon could not be attributed to the ethanol mandate until 2008. Note that this analysis focuses on binding state mandates and does not include programs (such as in North Carolina, Illinois, Iowa, etc.) that provided other types of support to the industry, or California because the Low Carbon Fuel Standard (LCFS) is not a mandate (discussed later in section C.4.2). Figure C.19 shows the possible impact of the state ethanol

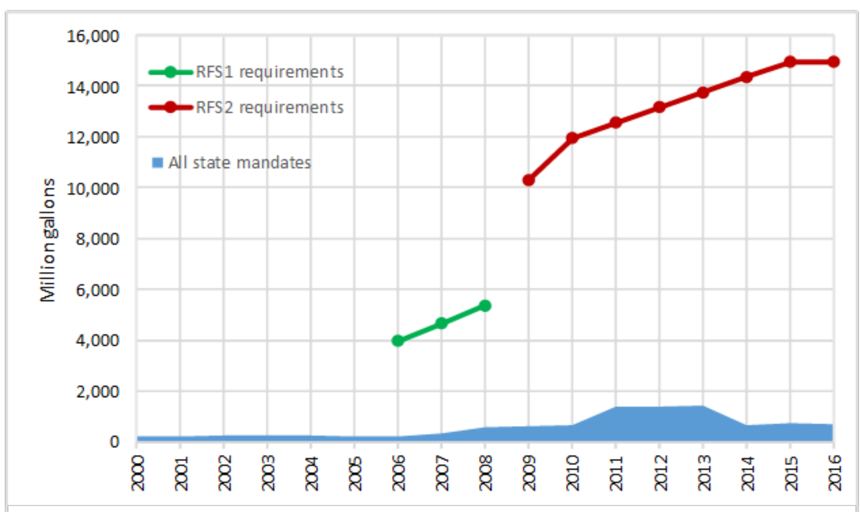


Figure C.19. Comparison of applicable volume requirements under the RFS1 and RFS2 to the sum of state ethanol mandates (2006–2008 volume requirements are for total renewable fuel, while 2009+ volume requirements are for conventional renewable fuel).

mandates on ethanol consumption and Figure C.20 breaks those down by state.

The ethanol volumes associated with state mandates were smaller from 2000–2007 with the exception of Minnesota (the only state with a mandate over that period), amounting to about 200 million gallons. That increased to 600 million gallons from 2006 to 2010 with contributions from five states, and then to 1.4 billion gallons from 2011 to 2014 with the addition of the populous state of Florida. The total from the state mandates was far below the RFS1 or RFS2 standards. This suggests the bulk of the ethanol consumption associated with state mandates was after 2007, at which time the ethanol was already halfway to the blend wall.

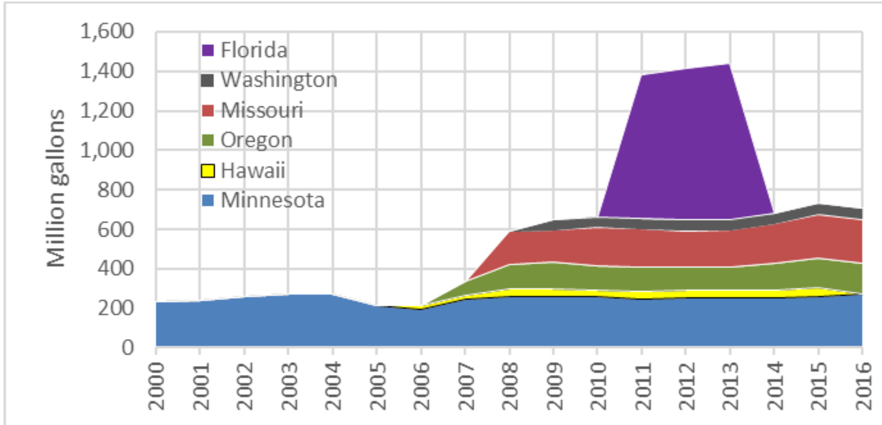


Figure C.20. Ethanol consumption associated with state ethanol mandates.
Source: State ethanol consumption from SEDS for states in Table C.4.[12]

C.4.2. California RFG and LCFS

California's ban on the use of MTBE in gasoline was announced in 1999 and was set to go into effect in December 2002. However, it was subsequently postponed twice, finally going into effect after December 31, 2003. Refiners started transitioning to ethanol in California in 2003 (see Chapter 6 Figure 6.3).

Unlike for federal RFG where the oxygen requirement was removed in 2006, the oxygenate requirement for California RFG has continued to apply to winter gasoline in certain areas in all years. However, as in the case of federal RFG, compliance with the California RFG standards for both summer and winter in all areas has been most cost effective with an oxygenate. Thus, refiners have used ethanol in essentially all California gasoline at the maximum allowed level in all years after 2003. Through 2009 the ethanol content of California RFG was limited to 5.7 volume%,[13] but in 2010 that limit was raised to 10 volume%. These transition points are clearly visible in the SEDS data for California (Figure C.21).

California's LCFS program was legislated in 2007 but did not go into effect until 2011. Thus, beginning in 2011 the LCFS requires that the average carbon intensity of gasoline decrease each year.

[12] Note: Figure C.20 is the same information as Figure C.19 but separated by state and zoomed in on volumes less than 1.6 billion gallons.

[13] The 5.7% limit was from the "Predictive Model," the tool that refiners use to determine eligibility of a particular gasoline formulation.

Ethanol is one means of meeting the applicable requirements, and thus the LCFS provides an additional incentive to use ethanol. However, since by 2010 essentially all gasoline in California already contained ethanol due to the California RFG requirements, the LCFS appears to have had little additional impact on total ethanol use in the California. In more recent years, however, it has created an incentive for refiners to move from ethanol made from corn to ethanol made from other feedstocks, such as sugarcane and cellulose, to meet the decreasing carbon intensity requirements of the LCFS. The LCFS has incentivized corn ethanol refiners to reduce the carbon intensity of their fuels, creating over 200 corn ethanol pathways that can earn emission reduction credits.

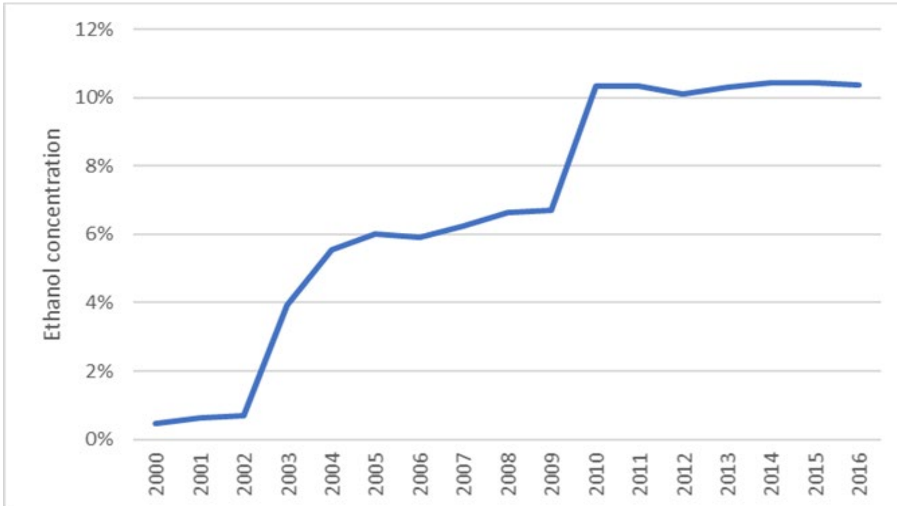


Figure C.21. Ethanol concentration in California gasoline.[14] Source: EIA (2023).

C.5. E10 Blend Wall

The E10 blend wall, or blend wall for short, is a term for the maximum amount of ethanol that can be blended into gasoline as E10. Thus, it is a function of the total amount of gasoline consumed, which is changing as fuel efficiencies increase and people's driving habits change. Without higher consumption of E15 and E85, it represents a constraint on growth of consumption. Higher consumption of E15 has been limited in the past by availability of terminals, legal concerns regarding liability, and other factors (Duffield et al., 2015). Higher consumption of E85 has been limited in the past by limited sales of flex fuel vehicles (FFVs), consumer choice to refuel with E10 rather than E85, and other factors. Thus, historically the blend wall has represented a constraint on domestic consumption of ethanol, though not an absolute limit on ethanol in gasoline as higher volumes are possible with E15 and E85.

[14] Concentration is from the EIA SEDS database, dividing the fuel ethanol consumption in California (i.e., variable "ENTCP," Fuel ethanol, including denaturant, total consumption) by the total gasoline consumption in California for each year (i.e., variable "MGTCP," Motor gasoline total consumption).

The nationwide average ethanol concentration, based on the total volumes of ethanol and gasoline consumption from EIA, suggests that the blend wall does constrain ethanol use. As shown in Figure C.22 (and in the main text in Figure 6.2), the annual increase in the average ethanol concentration decreased dramatically after 2010.

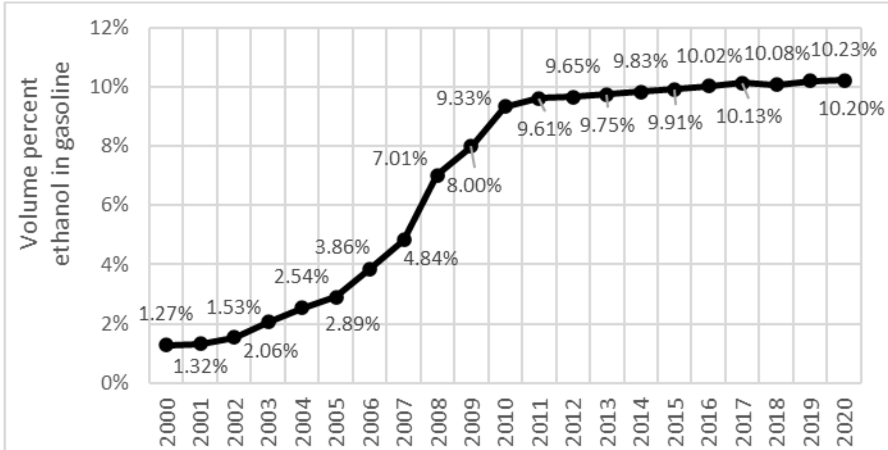


Figure C.22. Ethanol concentration in consumed gasoline. Source: Derived from EIA's Monthly Energy Review for ethanol (Table 10.3) and gasoline (Table 3.5).

C.6. Carryover RINs

The RIN system in the RFS Program was established in accordance with CAA Section 211(o)(5), which authorizes the generation of credits by any person who refines, blends, or imports renewable fuel in excess of the requirements of the statute. These are called "carryover RINs," and they provide liquidity to the RIN market as well as flexibility in the face of a variety of unforeseeable circumstances that could limit the availability of RINs and reduce spikes in compliance costs, including weather-related damage to renewable fuel feedstocks and other circumstances potentially affecting the production and distribution of renewable fuel. Thus, the collective carryover RIN bank provides a programmatic buffer that facilitates individual compliance, provides for smooth overall functioning of the program, and is consistent with the statutory provision allowing for the generation and use of credits.

The total number of carryover RINs available for any given compliance year can and has varied significantly over time. For example, enforcement actions in past years have resulted in the retirement of carryover RINs well after the compliance deadline for a given year to make up for the generation and use of invalid RINs and/or the failure to retire RINs for exported renewable fuel. Future enforcement actions could have similar results and require that obligated parties and/or renewable fuel exporters settle past enforcement-related obligations in addition to complying with the annual standards, thereby potentially creating demand for RINs greater than can be accommodated through actual renewable fuel blending. Conversely, Small Refinery Exemptions (SREs) granted after the compliance deadline for a given year can result in the refunding of RINs retired to meet an exempted party's obligation, which would increase the number of carryover RINs available. Table C.5 summarizes EPA's estimate of available carryover

RINs from the RFS annual rules, as well as the number of carryover RINs actually available for each year based on current data (and see Table C.6 for specific notes). For every year that data are available, there were over 1 billion carry over RINs, representing significant production in excess of the mandates as observed earlier (e.g., Figure C.9 and Figure C.10).

Table C.5. Estimate of carryover RINs (billions).

Compliance Year[a]	RIN Bank Basis[b]	Projected				Actual		
		Date	Carryover RINs	Total RVO[c]	%[d]	Carryover RINs	Total RVO[c]	%[d]
2011	n/a	n/a	n/a	n/a	n/a	2.65	13.65	19.4
2012	n/a	n/a	n/a	n/a	n/a	4.05	15.46	26.2
2013	2011		2.67	16.55	16.1	2.47	16.92	14.6
2014	2012		1.74	16.28	10.7	1.58	16.31	9.7
2015	2012		1.74	16.93	10.3	1.69	17	9.9
2016	2012		1.74	18.11	9.6	1.65	17.93	9.2
2017	2014		1.54	19.28	8	2.48	18.49	13.4
2018	2016		2.22	19.29	11.5	3.13	18.61	16.8
2019	2017		2.59	19.92	13	3.43	20.42	16.8
2020	2018		3.48	20.09	17.3			

[a] Calendar year for which compliance with the applicable standards is determined.
[b] Compliance year data used as the basis for estimating carryover RINs.
[c] Renewable Volume Obligation (RVO) for total renewable fuel.
[d] Carryover RINs as a fraction of the applicable volume requirement for total renewable fuel. A maximum of 20% of the RVO in a given year can be met with previous year RINs.

Table C.6. Notes on carryover RIN estimates for each compliance year.

Compliance Year	Notes
2013	In the final rule for 2013, EPA estimated that there were 2.67 billion carryover RINs available, based on 2011 compliance data. This represented 16.1% of the projected 2013 total RVO. However, the actual number of RVO Final Rule Year carryover RINs that were available for 2013 compliance was 2.47 billion RINs (or 14.6% of the actual 2013 total RVO).
2014	In the final rule for 2014, EPA estimated that there were 1.74 billion carryover RINs available, based on 2012 compliance data. This represented 10.7% of the projected 2014 total RVO. However, the actual number of 2013 carryover RINs that were available for 2014 compliance was 1.58 billion RINs (or 9.7% of the actual 2014 total RVO).
2015	In the final rule for 2015, EPA estimated that there were 1.74 billion carryover RINs available, based on 2012 compliance data. This represented 10.3% of the projected 2015 total RVO. However, the actual number of 2014 carryover RINs that were available for 2015 compliance was 1.69 billion RINs (or 9.9% of the actual 2015 total RVO).
2016	In the final rule for 2016, EPA estimated that there were 1.74 billion carryover RINs available, based on 2012 compliance data. This represented 9.6% of the projected 2016 total RVO. However, the actual number of 2015 carryover RINs that were available for 2016 compliance was 1.65 billion RINs (or 9.2% of the actual 2016 total RVO).
2017	In the final rule for 2017, EPA estimated that there were 1.54 billion carryover RINs available, based on 2014 compliance data. This represented 8% of the projected 2017 total RVO. However, the actual number of 2016 carryover RINs that were available for 2017 compliance was 2.48 billion RINs (or 13.4% of the actual 2017 total RVO).
2018	In the final rule for 2018, EPA estimated that there were 2.22 billion carryover RINs available, based on 2016 compliance data. This represented 11.5% of the projected 2018 total RVO. However, the actual number of 2017 carryover RINs that were available for 2018 compliance was 3.13 billion RINs (or 16.8% of the actual 2018 total RVO).
2019	In the final rule for 2019, EPA estimated that there were 2.59 billion carryover RINs available, based on 2017 compliance data. This represented 13% of the projected 2019 total RVO. However, the actual number of 2018 carryover RINs that were available for 2019 compliance was 3.43 billion RINs (or 16.8% of the actual 2019 total RVO).
2020	In the final rule for 2020, EPA estimated that there were 3.48 billion carryover RINs available, based on 2018 compliance data. This represented 17.3% of the projected 2020 total RVO.

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Appendix D: Modeling a "No-RFS" Case

This appendix constitutes an analysis conducted under subcontract for EPA that examined the effect in 2020 of the hypothetical removal of the RFS Program in 2016. In the "No RFS case" the use of ethanol and biodiesel and renewable diesel fuel in each PADD was governed not by the RFS mandates, but by the economics of gasoline, diesel fuel, and biofuel production and by state and local mandates for ethanol, biodiesel, and renewable diesel fuel use. Appendix D is the Final Report provided to EPA for that work. This Final Report was in support of rulemaking (U.S. EPA, 2020); but, given the subject of simulating ethanol and biodiesel production with and without the RFS Program between 2016 and 2020, it is useful for the RtC3 as well. Full text of the Final Report follows and additional information can be found in the supporting material associated with U.S. EPA (2020). Appendices referenced here refer to appendices associated with the "No-RFS" Case Final Report conducted by ICF, not appendices for the RtC3. Thus, for convenience the term "ICF Report Appendix" is used to refer to those. See the source document below for additional information.

U.S. EPA. (2020). Renewable fuel standard program: standards for 2020 and biomass based diesel volume for 2021 and other changes. Supplementary information for the final rule: Technical support document for modeling a no RFS case prepared by ICF Incorporated, LLC [EPA Report]. (EPA–HQ–OAR–2019–0136). Durham, NC: ICF Incorporated. https://www.regulations.gov/document/EPA-HQ-OAR-2019-0136-2147.

D.1 Introduction and Summary Results

D.1.1 Objective and Purpose

This study was driven by the following purpose, as outlined in the work assignment: "Under Section (211(o)(7)(F)) of the Clean Air Act, the U.S. Environmental Protection Agency (EPA) must modify the statutory volume targets under the Renewable Fuels Standard (RFS) program based on an analysis of the impacts of renewable fuels on the U.S. economy and the environment following actions to waive those statutory volume targets. To facilitate the economic and environmental analysis of the RFS, it is necessary to estimate the volume of renewable fuels which would occur in the U.S. if the RFS requirements were not in place."

Addressing this question requires that the analysis (1) recognize the multiple factors that would influence the economics of blending ethanol and biodiesel in transportation fuels, and (2) utilize the most current information available on the actual properties of fuels, logistics and market costs, individual state biofuel mandates, and refinery and blender characteristics. The project utilized a refinery linear programming (LP) model so that various cases could be examined and compared using consistent assumptions and data to represent the marketplace both with and without the RFS in place.

D.1.2 Executive Summary

In order to develop the analysis for EPA, ICF utilized information in petroleum and biofuels markets to support the development of assumptions and market factors. The analysis assessed the use of only the most prevalent biofuels—corn ethanol and biodiesel. For the refinery model, ICF worked with MathPro Inc., which has a refinery LP model that has been modified and used in multiple studies for EPA, other Federal agencies, and private sector clients.

The cases modeled included the following:

1. A 2016 Calibration case, in which the model was configured to represent actual 2016 biofuels requirements and crude runs, such that model outputs provided a good replication of actual supply and demand outcomes. The calibration was done on a Petroleum Administration for Defense District (PADD)-level basis, and the model results compared with reported 2016 values in each PADD.
2. A 2020 Reference case, in which the model was configured to (1) maintain biofuel requirements specified by the RFS program for 2016 in 2020, (2) reflect product demands in 2020 projected in the U.S. Energy Information Administration's (EIA) 2017 Annual Energy Outlook (AEO), and (3) incorporated any known refinery upgrading projects that would be in place in 2020.

3. A 2020 No-RFS case, in which the use of ethanol and biodiesel and renewable diesel fuel in each PADD was governed not by the RFS mandates, but by the economics of gasoline, diesel fuel, and biofuel production and by state and local mandates for ethanol, biodiesel, and renewable diesel fuel use. The No-RFS case incorporated the Federal Biodiesel Tax Credit of $1.00 per gallon (gal). All cases analyzed represented the refining sector's perspective on the refining economics of gasoline and diesel production. In particular, the results of the analysis do not incorporate the economic effects of ethanol's energy content deficit with respect to hydrocarbon gasoline (≈ 34% lower).

D.1.3 Summary Assumptions

To assess the No-RFS case, the model was configured to recognize numerous policy, logistics, and technical assumptions and premises, including the following:

- Various state and local mandates for ethanol and biodiesel would remain in place even without the RFS, creating a floor for ethanol and biodiesel volumes.[1]
- Reductions in ethanol blending increase the call for octane production by U.S. refineries. The model recognized the blending properties, including octane, of biofuels as well as hydrocarbon blendstocks.
- Economic decisions on ethanol and biodiesel usage in end-use markets recognize the delivered costs and refining values of biofuels and refinery-produced fuels.
- Delivered costs for biofuels and refined fuels differ by PADD, and also within PADD, for major market sectors.
- Delivered costs of ethanol and biodiesel reflected PADD-level cost curves developed to represent ethanol and biodiesel supply costs, based on the biofuels' plant configurations, location, capacity, byproduct values, and operating costs in each region.
- Cost curves were developed for both corn ethanol and biodiesel production to assess the effect that elimination of the RFS might have on less cost-efficient (i.e., smaller plants) segments of the biofuels industry versus more cost-efficient segments.
- The projected prices of biodiesel feedstocks used in the analysis reflect current market prices for these feedstocks. If biodiesel demand were to drop substantially, as in the No- RFS case, biodiesel feedstock prices might be driven down enough to allow more biodiesel production

[1] Note: The study considered mandated volumes, or percentages only. A number of states (Illinois, Texas, and others) offer reductions in sales taxes at the retail level to stimulate sales of biodiesel blends; however, these were not considered as binding as mandated volumes.

to be economical than our analysis indicates (assuming that the $1.00/gal biodiesel tax credit was still in place).

- The properties and production and distribution costs of renewable diesel fuel are assumed to be the same as for biodiesel, based on prior analysis conducted by EPA.[2]
- The delivered cost of ethanol and biodiesel in each end-use market represented depends on the level of biofuel production, biofuel plant characteristics and yields, and the distribution costs to move the biofuels to demand markets.
- E85 volumes in the No-RFS case were assumed to stay the same as in the Reference case, per guidance from EPA.
- The 2017 AEO Reference case for 2020 was used to estimate domestic demand, refinery output, crude oil prices, and various input and output prices for the PADD-level refinery models. The projected average U.S. composite crude oil acquisition cost for 2020 is about $72/barrel (bbl). This is substantially higher than the reported U.S. crude oil acquisition cost of $42.54/bbl for the summer season used in the 2016 Calibration case ($48.24/bbl for the winter season).
- Biofuel feedstock prices are assumed to be the same in 2020 as in 2016.

The full report discusses the technical approach and assumptions in considerably more detail.

D.1.4 Summary Results

The study indicates that removal of the RFS program could have markedly different effects on the use of ethanol and biodiesel.

Ethanol: The volume of ethanol used in gasoline in the No-RFS case is the same as in the Reference case, in which the RFS is in place. This result holds for all PADDs and for all grades of gasoline.

The ethanol volume blended in the 2020 No-RFS case is about 850,000 barrels per day (850 K bbl/d) (ethanol blended into U.S. refinery production for domestic consumption), the same volume as blended in the 2020 Reference case (ethanol blended into imported gasoline adds about another 70 K bbl/d).

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[2] Regulation of Fuels and Fuels Additives: Changes to Renewable Fuel Standard Program, Final Rule; Regulatory Impact Analysis Tables 4.1–41 and 4.1–42; March 26, 2010.

In almost all markets, ethanol's estimated blending value (shadow value[3]) returned by the refinery models is significantly higher—roughly \$15–\$25/bbl—than its estimated net delivered cost. Furthermore, if spot market prices for gasoline and ultra-low sulfur diesel (ULSD) versus crude oil in 2020 are higher than the incremental refining costs returned by the refinery models, as was generally the case for the refinery modeling conducted for 2016, the indicated economics of ethanol use improve further.

The estimated ethanol blending values tend to be higher in regular grade than in premium grade, higher in summer than in winter, and higher in conventional gasoline than in reformulated gasoline (RFG). The only exception to this result pertains to California RFG, for which ethanol's estimated blending value is negative. This result is due to ethanol's unusually large Reid vapor pressure (RVP) uplift in California RFG. However, ethanol blending at 10 vol% is essentially mandated in California, independent of the RFS.

Overall, these results indicate that ethanol blending likely would remain economical in the absence of the RFS, even at crude oil prices significantly lower than those assumed in this study.

Biodiesel and Renewable Diesel: Unlike ethanol use, the estimated use of biodiesel in the No-RFS case is sensitive to the assumed price of crude oil. With a crude oil acquisition cost of \$72/bbl, biodiesel appears to be economical to blend into diesel fuel at levels from about 105 to 155 K bbl/d (1.6 to 2.4 billion gal/year), depending on the assumptions regarding projected spot market margins for ULSD versus crude oil. Biodiesel use would be at the upper end of this range if spot market prices for ULSD versus crude were similar to those in 2016. The incremental production costs returned by the refinery models were lower than the spot market refinery margins in 2016. This volume range is lower than the Reference case volume of about 170 K bbl/d.

However, our results indicated that if crude oil prices were about \$5/bbl lower than the estimated crude oil acquisition price, biodiesel blending could decline to about 45 K to 95 K bbl/day (about 0.7 to 1.5 billion gal/year), again depending on the assumptions regarding spot market prices for ULSD versus crude. The lower end of this range is the minimum volume of biodiesel blending required in states having biodiesel mandates. Our results suggest that another \$5/bbl drop in crude oil prices (\$10/bbl total) likely would reduce biodiesel blending to about 45 K bbl/d, even when spot market margins for ULSD are assumed to be generally higher than the incremental production costs returned by the refinery models.

[3] "Shadow values" are marginal costs or values computed by LP models. In this instance, the shadow value of ethanol computed by the LP model indicates the highest price that a refiner or blender would be willing to pay for ethanol in the case being analyzed. If that price is higher than the prevailing market price of ethanol, then the refiner or blender has an incentive to purchase and blend ethanol.

Of this mandated volume of 45 K bbl/d, approximately 31 K bbl/d would be used in PADD 5 (mainly in California) to meet state Low Carbon Fuel Standard (LCFS) requirements. Biodiesel from waste oil, which is the lowest priced feedstock, appears to be an economical blendstock, even at low crude oil prices; however, it accounts for a relatively small volume of supply (less than the mandate volume). The above estimates of biodiesel use assume that the $1.00/gal tax credit remains in force. Should biodiesel lose its tax credit, crude oil prices would have to increase to around $110/bbl for most biodiesel to become economical for use as a blendstock in diesel fuel.[4]

Unlike ethanol, whose high blending octane allows refiners to reduce the octane of refinery-produced gasoline-based blends (blendstocks for oxygenate blending [BOBs]), biodiesel has no significant blending value to refiners other than as a volume extender in the diesel fuel pool.

When the price is right in the market, biodiesel will be blended. Higher refinery diesel margins (and spot prices) at a given crude oil price could also stimulate additional biodiesel blending. Conversely, higher biodiesel feedstock costs (i.e., soy, canola oil, yellow grease) will increase the biodiesel production cost curve for those feedstocks and tend to reduce biodiesel blending.

These dynamics and the uncertainties that they bring can make it difficult for a biodiesel producer to justifying staying in operation—even with the $1.00/gal tax credit—whenever crude prices approach levels at which the marginal cost of refinery-produced diesel is less than the break-even cost of biodiesel production (net of the tax credit).

Overall: Overall, these results appear to be reasonable. Ethanol's properties as a gasoline blendstock, most notably its high blending octane, give it a market value comparable to that of alkylate, a widely traded high-value gasoline blending component, which often trades at $0.20 to $0.30/gal above the gasoline spot market price. Ethanol's low sulfur and benzene content add to its value as a gasoline blending component. Its use allows refiners to reduce the octane of BOBs produced for ethanol blending, thereby reducing refinery operating costs and improving yields (e.g., by permitting reforming operations at a lower severity).

In contrast, biodiesel, which is generally splash-blended downstream of the refinery, has no blending properties that can support a price premium relative to conventional hydrocarbon diesel fuel. Without premium blending properties, the higher cost to produce and transport biodiesel to market is difficult to overcome, even with the biodiesel tax credit. It is possible that some refiners who may be constrained by diesel fuel cetane standards could benefit by blending higher-cetane biodiesel at the

[4] The biodiesel supply curve was developed using volumetric production data from 2016 blending levels; the supply curve was constrained on a feedstock basis using the inputs to biodiesel production reported by EIA: https://www.eia.gov/biofuels/biodiesel/production/.

refinery (e.g., in California). However, this study did not examine individual refinery constraints, or the full economics of this option.

In the absence of an RFS and with crude prices around $72/bbl (consistent with the AEO Reference case crude price), the sensitivity of biodiesel blending economics to small changes in the differential between the hydrocarbon diesel market price and the biodiesel production cost may make it difficult for biodiesel producers to maintain adequate biorefinery margins in the face of volatility in crude oil prices and refinery margins. This is particularly the case if the $1.00/gal biodiesel tax credit is not in place.

In the absence of an RFS, the dependence of the biodiesel blending economics on the differential between the hydrocarbon diesel market price and the biodiesel production cost may make it difficult for biodiesel producers to ensure consistent biorefinery margins due to the volatility in crude oil prices and refinery margins. This is particularly the case if the $1.00/gal biodiesel tax credit is not in place.

D.2 Study Methodology and Detailed Assumptions

To conduct the analysis for EPA, ICF utilized information in petroleum and biofuels markets to support the development of assumptions and market factors. For the refinery model, ICF retained MathPro Inc., which has a refinery LP model that has been utilized and modified for multiple studies for EPA, other Federal agencies, and private sector clients.

This section of the report contains a discussion of the methodology used to set up the LP model and obtain results. The report outline is as noted here:

- Cases Modeled
  - Summary
  - Detailed Setup of Calibration and Reference Cases
- Biofuel Assumptions
  - Ethanol Summary
  - Biodiesel Summary
  - Ethanol Detailed (Properties, Logistics, Cost Curves, Mandates, Other)
  - Biodiesel Detailed (Properties, Logistics, Cost Curves, Mandates, Other)
- Study Case Setup (2020 No-RFS Case)
- Results
- Appendices (separate documents)
  - ICF Report Appendix A: Calibration Case Data Tables
  - ICF Report Appendix B: Reference Case Data Tables

– ICF Report Appendix C: Case Study Data Tables
– ICF Report Appendix D: Detailed Modeling Case Results
– ICF Report Appendix E: Key Results for a Lower Crude Oil Price Case

D.2.1 Cases Modeled

D.2.1.1 Summary

The cases modeled include the following:

1. A 2016 Calibration case, which assumed actual 2016 biofuels requirements and crude runs and then ran the model to ensure that the results provided a good replication of actual supply and demand outcomes. This case was conducted on a PADD-level basis, and calibrated to actual 2016 results.
2. A 2020 Reference Case, which maintained 2016 biofuel requirements but modified the model to reflect 2020 demand and prices based on EIA's 2017 AEO, and any known refinery upgrading projects that would be in place in 2020.
3. A 2020 No-RFS case, which modified the fixed use of ethanol and biodiesel in the Calibration and Reference cases to allow the refinery models to blend ethanol and biodiesel as dictated by their economics, subject to minimum blending levels representing state mandates. The study assumed an average crude oil acquisition cost of $72/bbl. The study was to assess the effects of removing the RFS program both with and without the Biodiesel Tax Credit of $1.00/gal. However, given that biodiesel blending in the absence of the subsidy was found to be uneconomical at the assumed crude oil price, it was not necessary to formally assess a "no biodiesel subsidy" case.

D.2.1.2 Detailed Calibration Case Setup – 2016

The refinery modeling was conducted at the individual PADD level for both the summer and winter seasons. Our typical approach when conducting refinery modeling is to first develop a Calibration case that pertains to the most recent time period of which substantial information on refining activity and performance is available, mostly from EIA. Setting up the Calibration case is a useful first step in organizing information and in developing a modeling structure that is appropriate for both the Calibration and subsequent Study cases. Furthermore, the generation of refinery modeling results in a Calibration case that is reasonably close (well-calibrated) to the observed performance of the various refining sectors examined, which provides greater assurance that the refinery models will perform appropriately when conducting the Study case analyses.

Information in the following areas was developed to configure each of the five PADD-level models by season. This information is too voluminous to usefully present in the text of this report, so we are providing only a brief discussion of how the information was developed and what is of the most importance. The detailed tables showing the information used in developing the refinery models are provided in ICF Report Appendix A.

- Refining process capacities were developed using EIA's 2016 Refinery Survey.
- Refinery inputs, downstream processing, whole crude oil properties, and the use of ethanol, biodiesel, and renewable diesel: Based primarily on EIA monthly data on refinery inputs, downstream feed throughput for major refining processes (reforming, catalytic cracking, hydrocracking, and coking), and the composite crude American Petroleum Institute gravity and sulfur content. The data on refinery inputs mostly are used to specify input volumes of feeds in the refinery models. However, the volumes of crude oil and downstream processing throughput reported by EIA serve as indicators of how closely the refinery models match the reported performance of the various refining centers (i.e., crude oil throughput and downstream processing throughput are not specified in the refinery models, but are instead results yielded by the models). Likewise, whole crude oil properties are not specified values, but instead are indicators of how closely the PADD-level composite crudes developed for the study match aggregate PADD-level reported properties.
- Estimates of ethanol use (assumed for the purposes of this study to be blended at the refinery rather than downstream) are based on the assumption that all finished gasoline produced for the domestic market, with the exception of small volumes of E0, are blended with denatured ethanol at 10 vol%.[5] Estimates of the use of biodiesel and renewable diesel in the ULSD (and California Air Resources Board [CARB] diesel) produced by refineries in each PADD (this includes estimated volumes of biodiesel and renewable diesel blended in ULSD shipped to other PADDs) are based on renewable identification number (RIN) data and PADD-level allocations developed by EPA, as discussed in more detail below (negative refinery inputs of gasoline blendstocks reported by EIA refer to BOBs and are incorporated in our exhibit detailing refinery outputs).
- Refinery outputs were based on EIA monthly data. Refinery production of finished gasoline, based on BOBs output adjusted for 10 vol% ethanol blending, is reported, along with

[5] In 2016, U.S. refineries reported producing about 1.1 million bbl/d of finished E0, of which about 630 K bbl/d were exported, and 470 K bbl/d were supplied to domestic markets. We assumed that most of the 460 K bbl/d of domestic supply subsequently were blended with ethanol (to produce E10) and that a relatively small amount—75 K bbl/d (less than 1% of U.S. gasoline supply)—was sold at retail as E0. Various studies have estimated E0's share of the gasoline market at between 1% and 6%. Modifying our assumed volume of retail E0 sales up or down by 100% would have little effect on the results of the study, other than slightly decreasing or increasing ethanol use.

estimates of the production of finished ULSD (and CARB diesel) adjusted for estimated regional blending of biodiesel and renewable diesel. Relatively small volumes of E85 were assumed to be produced based on estimates by EPA. All E85 was assumed to be 74% fuel ethanol and 26% natural gasoline.[6] Exports of gasoline and gasoline blendstocks were assumed to be E0 with an average Anti-Knock Index (AKI) of 86.5, an RVP of 8.7 pounds per square inch (psi), and a benzene content of 0.62 vol%.

- Composite crude oil slates were developed using data from EIA on composite crude oil properties, projected domestic crude oil production by type (gravity and sulfur), state-level crude oil production, inter-PADD movements of crude oil, and company-level imports of crude oil, along with representations developed by MathPro using assays for major domestic and foreign crude oils.
- Premium/regular grade shares of gasoline sales by PADD were estimated from EIA Prime Supplier Sales data for reformulated and conventional gasoline.
- PADD-level estimates of finished gasoline octane (AKI) were derived for reformulated and conventional gasoline, by grade, using data from the Alliance 2015 North American Fuel Surveys.
- Ethanol's volumetric blending octane varies depending on the octane of the finished gasoline with which it is blended (i.e., it declines as the octane of the finished gasoline increases). We used the molar blending approach to estimate blending octanes for ethanol in E10 regular and premium grades, accounting for regional differences in octanes within gasoline grades. For example, the AKI of premium generally ranges from about 91 to 93, and the AKI of regular generally ranges from about 85 to 88 (predominantly 87).
- Ethanol's sulfur content (assumed to be 3 parts per million [ppm]), distillation property, and energy density are assumed to be invariant across regions and gasoline types. Ethanol's blending RVP is discussed below.
- Imports and exports of gasoline and distillate were based on monthly EIA data.
- Inter-PADD shipments of gasoline and ULSD are based on monthly EIA data. These data are used to estimate gasoline and ULSD that is produced in one PADD (e.g., PADD 3) but is shipped to another (e.g., PADD 1), and is blended there with ethanol (to make E10) or with biodiesel or renewable diesel. We accounted for such shipments when configuring the

[6] While only a portion of E85 is blended with natural gasoline today, the use of natural gasoline is increasing, and the assumption was not deemed to have a material effect on the results.

refinery models because the economics of blending ethanol or biodiesel and renewable diesel in distant markets supplied by a given PADD may be different from the local economics of such blending.

- Prices for crude oil, other refinery inputs, selected refined products, and renewables were determined as follows: EIA monthly data on crude oil acquisition costs were used to develop the seasonal prices for the composite crudes processed in the various PADDs. Reported crude oil prices during summer 2016 (April through September were about $5/bbl lower than reported prices during the winter (October 2016 through March 2017). EIA regional data on industrial end-use prices for natural gas and power were used to establish refinery prices for natural gas purchases and power (we computed seasonal prices for natural gas, but used an annual cost for power). Hydrogen inputs were priced at twice the price of natural gas. Isobutane and normal butane inputs were priced at zero in the winter (with input volumes specified according to EIA-reported refinery input volumes); in the summer, isobutane and normal butane inputs were priced according to regional spot prices reported in various literature sources, and outputs were priced slightly lower. Propane outputs were priced according to spot prices from various literature sources. Light gases were priced to provide the refinery models with the needed flexibility to accommodate their production in various refinery processes, RVP control in gasoline, and their disposition as finished products (marketable coke also was assigned a nominal price). All other refinery inputs and refined product outputs were priced at zero, with volumes specified according to data we developed on refinery input and output volumes.
- Ethanol inputs were priced at zero with maximum volumes set equal to 10% of the volumes of each finished gasoline type and grade (the model accommodates up to 22 unique ethanol blendstocks). Although ethanol, in fact, is mostly blended at terminals, the refinery models are configured to produce finished E10 gasoline. The ethanol maximum volumes are specified so that the refinery models produce E10 gasolines and generate estimates of ethanol's refining value as a gasoline blendstock for different types and grades of gasoline.
- Biodiesel and renewable diesel are priced at zero with maximum volumes specified (based on data developed by EPA and ICF as discussed further below). As with ethanol, although biodiesel/renewable is mostly blended with ULSD at terminals or large stations, it is represented in this study as being blended into ULSD produced at the refinery so that the refinery model yields an estimate of its incremental value as a ULSD blendstock.

- Properties of biodiesel and renewable diesel are assumed to be the same as ULSD so that the refining values of biodiesel and renewable diesel are the same as the incremental cost of ULSD production. Other than the cetane rating of some biodiesels, neither biodiesel nor renewable diesel have any special properties that would cause refiners or blenders to value them more or less than the cost of ULSD. Unlike ethanol, which has a significant and consistent octane value that allows all refiners to produce BOB product for nationwide distribution and blending, the cetane benefit of some biodiesels does not appear to have driven refiners to lower refinery diesel cetane levels to allow purchase and blending of biodiesel either at the refinery or in the market.
- The RVP of finished gasoline, BOBs, and ethanol were developed using gasoline properties reported in the Alliance 2015 North American Fuel Surveys for the summer and winter seasons, along with state-level standards for low-RVP gasoline. The estimated RVPs were assigned to finished gasolines produced by the PADD-level refining sectors and used within the PADD and to gasolines shipped from the PADD of origin to other PADDs. The RVP of ethanol blended in the various E10 gasolines varies depending on the RVP of the finished gasoline, whether the finished gasoline qualifies for the ethanol RVP waiver, and the season.
- Certain regions of the country have imposed low-RVP standards for summer gasoline, or have disallowed ethanol's RVP waiver (or both). It is important to represent such gasolines in the refinery modeling because, even though the volume of such gasolines is relatively small, lowering summer gasoline RVP standards and, even more so, disallowing ethanol's RVP waiver, reduces the refining value of ethanol as a gasoline blendstock. We estimated the volumes of such gasolines using EPA's tabulation of state-by-state RVPs (at a county level), EIA state-level Prime Supplier sales volumes, and county-level Census data. We assume, to simplify the configuration of the refinery models, that such gasoline volumes are met by refineries within the PADD in which these low-RVP areas are located. The exception is for gasoline supplied by PADD 3 refineries to Georgia and to the Nashville, Tennessee, area (via the Plantation pipeline spur originating in Georgia).
- Allocation of gasoline supplied by PADD 3 to PADD 1: Gasoline shipped from PADD 3 to PADD 1, as indicated by inter-PADD movement data, supplies several regions that can be distinguished by finished gasoline properties (RVP and octane), premium/regular grade splits, delivered costs of ethanol to terminals, and pipeline tariffs for gasoline (the latter two factors are of importance in assessing the economics of ethanol blending). We used Federal Highway Administration data on state-level finished gasoline consumption, state-level gasoline import

data, and inter-PADD shipment data to allocate gasoline originating in PADD 3 to four regions in PADD 1: Florida, Southeast, Mid-Atlantic, and Northeast. This allows differentiation of gasoline shipped to PADD 1 by RVP and octane. It also provides the groundwork for differentiating ethanol supply costs in refinery modeling undertaken later in this study to assess the economics of blending ethanol in the absence of the RFS program.

- Allocation of ULSD supplied by PADD 3 to PADD 1: Differentiation of the ULSD supply from PADD 3 to sub-regions in PADD 1 is not important in configuring the refinery models for the Calibration and Reference cases; however, it is necessary to assess the economics of blending renewables in ULSD in the absence of the RFS program. We used data on (1) state-level ULSD use (from EIA's Prime Supplier Sales data), (2) inter-PADD ULSD shipments, (3) production of ULSD by PADD 1 refineries, (4) exports of ULSD from PADD 1, and (5) state-level imports of ULSD (from company-level import data) to estimate the volume of ULSD supplied by PADD 3 to two regions in PADD 1—Southeast/Mid-Atlantic and Northeast. Those estimates are used in the Study case in conjunction with specifying biodiesel/renewable diesel supply in PADD 1 sub-regions.
- Actual biodiesel and renewable diesel use at the PADD level or in sub-regions within PADDs is unavailable from EIA. Estimates were developed of the regional use of biodiesel and renewable diesel in terms of volume and percent blending in ULSD and CARB diesel using (1) regional estimates of the combined use of biodiesel and renewable diesel prepared by EPA (which was derived using estimates of production, imports, exports, and inter-PADD shipments); (2) estimates of the aggregate net U.S. supply of biodiesel and renewable diesel based on RIN data; (3) estimates of California's use of biodiesel and renewable diesel developed by ICF and production of CARB (hydrocarbon) diesel from the Weekly Fuels Watch reports; and (4) PADD-level data on ULSD production, imports, exports, and inter-PADD shipments from EIA. The calculations and assumptions used in developing the estimates are shown in a series of four exhibits in ICF Report Appendix A. Although the estimates should not be viewed as highly accurate, they appear to be reasonable, and it is not immediately obvious how such estimates could be improved upon.
- Finished gasoline and diesel volumes with renewables are identified in a series of four exhibits in ICF Report Appendix A showing the finished gasoline and diesel volumes used in configuring the PADD-level refining models for the Calibration case. The first set of exhibits detail (1) finished gasoline production volumes by PADD of origin, regional destination, type of gasoline, and grade, along with import volumes by type of gasoline, and export volumes, and (2) finished diesel fuel production volumes by PADD of origin, PADD of destination,

and type of diesel fuel, along with the estimated volumes of biodiesel and renewable diesel blended in finished diesel fuel. The data were developed using the panoply of previously described data. The second set of exhibits shows a consolidated version of gasoline and diesel production for each of the PADDs in which (1) similar gasolines are aggregated, and (2) ULSD production is aggregated, but biodiesel and renewable diesel use (identified in the exhibit as FAME and RENDSL) is broken out by destination of the ULSD (this was done to prepare the model's structure in advance for assessing the economics of biodiesel/renewable diesel blending in the Study case).

- Finished gasoline's and ethanol's blending RVP and octane by type of gasoline and grade, and RVP for summer and winter gasolines are shown in Exhibits A20a and A20b (in ICF Report Appendix A). The estimates are based on information developed regarding regional gasoline properties and gasoline properties for low-RVP gasoline, as discussed above. The blending octanes and RVP for ethanol used in the modeling also are provided. In general, as discussed above, we represent the blending octane of ethanol as declining with the increased octane of the finished gasoline (as is obvious by comparing the blending octane of ethanol in premium versus regular gasoline). Also, the blending RVP for ethanol in the summer is highest for finished gasolines with low RVPs that do not qualify for the RVP waiver and is set equal to the RVP standard (less a safety factor) for gasolines qualifying for the RVP waiver so that ethanol does not have an RVP penalty associated with it. In the winter, all of the RVPs for all ethanol are set equal to the estimated BOB RVPs, again so that there is no RVP penalty associated with blending ethanol in winter gasoline.

The resulting refinery models are configured to represent the results of the current regulatory framework for gasoline and diesel fuel in which the RFS creates incentives to blend most all gasoline with 10 vol% ethanol and to blend significant volumes of biodiesel and renewable diesel in ULSD and CARB diesel. It is important to understand that the refinery models do not assess how the incentives created by the RFS program affect the behavior of the refining sector with regard to the use of renewables. Instead, the refinery models for the Calibration case are set up to require the blending of ethanol, biodiesel, and renewable diesel in the volumes observed (or estimated) during 2016, regardless of their economics.

However, the refinery models have been constructed in a manner that anticipates the model structure and analysis required to assess the economics of renewables in the Study case, where the RFS is assumed to no longer apply.

D.2.1.3 Detailed Reference Case Setup – 2020

The Reference case in this study represents a characterization of PADD-level refining operations in 2020, given (1) continuation of the RFS program and target renewable volumes in effect in 2016, (2) implementation of new standards affecting the refining sector scheduled to be in effect by 2020, (3) changes in refining process capacity, (4) gasoline grade splits remaining as they were in 2016, and (5) changes in petroleum-related supply and demand factors as projected by EIA.

- Continuation of the RFS program with the 2016 renewable targets in place implies that the use of ethanol, biodiesel, and renewable diesel will be similar to the use reported in 2016 (as a percentage of gasoline and ULSD).
- Two new regulatory standards affecting the refining sector will go into effect by 2020. The first is a lowering of the sulfur standard for gasoline from an average of 30 ppm to 10 ppm. The second is the implementation of the MARPOL standards (International Convention for the Prevention of Pollution from Ships) for the marine shipping industry, which could lower the sulfur content of much of the bunker fuel (residual fuel oil) used for marine shipping to 0.5 wt% (5,000 ppm) or less.
- The reduction in the sulfur standard for gasoline was incorporated into the Reference case by reducing the maximum limit on sulfur to 9 ppm for all gasoline types and grades, including exports, which is slightly less than the 10-ppm average standard. To meet the new sulfur standards, many refineries will reduce further the sulfur content of FCC (Fluid Catalytic Cracker) naphtha and, in so doing, reduce somewhat the octane of FCC naphtha. This will have the effect of increasing the refining cost of octane and raising the refining value of ethanol, because it is a high-octane blendstock.
- The MARPOL standards were incorporated into the Reference case by (1) increasing the production of residual oil with 5,000 ppm or less sulfur content and correspondingly reducing the production of high-sulfur-content residual oil (up to 35,000 ppm sulfur content) in PADDs 1, 2, and 5, up to the volume limits consistent with current processing capacity (primarily coking, to reduce the volume of high-sulfur residual, and downstream desulfurization capacity, primarily heavy gas oil hydrotreating); (2) assuming that 68% of total U.S. residual oil production (not including low-sulfur residual oil already being produced) would meet the 5,000 ppm MARPOL standard (based on previous analysis of this issue conducted by EPA); and (3) assigning the remaining volume of MARPOL-compliant residual oil production (after accounting for production in PADDs 1, 2, and 5) to PADD 3.

- The MARPOL requirements may be met by a number of alternatives, including installation of ship SOx scrubbers, use of diesel fuel instead of residual oil, or by refinery investments in desulfurization capacity. It is possible that the use of diesel fuel will increase global diesel fuel demands and create some price escalation, which could impact biodiesel demand. No effect of MARPOL on the diesel market was assumed for this study because it is unclear how the global marine and refining community will determine how the standard will be met over the long term.

EIA's 2017 Refinery Capacity Survey and information from the trade press regarding refinery capacity additions or expansions likely to be in place by 2020 were used to establish PADD- level refining process capacity for 2020. Estimated changes in refining process capacity are small and have only minor effects on the results of the refinery modeling.

Although recently there has been an upward trend in the market share of premium grade gasoline, possibly related to increased market penetration of vehicles using turbo-charging technology and related to lower gasoline prices, we assumed that the premium/regular grade splits estimated for 2016 would persist through 2020. EIA does not provide projections of grade splits in the AEO forecasts. If some increase in premium/regular grade splits had been incorporated in the refinery modeling for 2020, the cost of producing octane would have been somewhat higher than in the modeling actually carried out using the estimated 2016 grade splits. In turn, this would have raised the refining value of ethanol because (1) the cost of producing hydrocarbon gasoline would have been somewhat higher, and (2) the refining value of the octane boost from blending ethanol would have increased.

EIA's Reference case forecast in the 2017 AEO (which was the latest AEO available throughout most of the time that this study was being conducted) was used in this study as the basis for forecasting energy prices (crude oil, natural gas, electricity, and some refined product prices), refinery input and output slates, and imports and exports of refined products. The AEO Reference case forecast projects a significant increase in energy prices from 2016 to 2020.

Based on the AEO price forecasts, the annual average acquisition cost of crude oil increases from about \$41/bbl to about \$72/bbl, an increase of about 75%.[7] End-use prices to electric utilities for natural

[7] In the refinery modeling conducted for this study, for 2016, we used reported composite crude oil acquisition costs averaged over the summer (April through September 2016) and winter (October through March 2016/2017). For 2020, we used a composite crude oil acquisition cost of \$72/bbl based on the Reference case forecast in the 2017 AEO. Unfortunately, the 2017 AEO does not directly report the forecast composite crude oil acquisition cost. However, it does provide the average crude oil acquisition cost for imported crudes and the average wellhead price for Lower 48 domestic crudes. Data reported by EIA for 2016 and 2017 indicate that average domestic crude oil acquisition costs are nearly \$467/bbl higher than average domestic wellhead prices. Using this relationship, along with projected volumes of imported and domestic crudes, the composite crude oil acquisition cost for 2020 would be about \$72/bbl.

gas were projected to rise from an annual national average of about $3.12 per thousand cubic feet (mcf) to about $4.69/mcf, an increase of about 50%. On the other hand, relatively moderate increases of about 3% are projected for the aggregate physical supply of petroleum-related products and the use of refined products.

Unfortunately, the AEO forecasts do not provide projections of U.S. or regional refinery production of specific refined products. Consequently, we used the AEO forecasts to develop more detailed forecasts of aggregate U.S. refinery inputs and outputs of specific refined products and translated those forecasts to PADD-level refining sectors.

The AEO provides forecasts of domestic consumption of refined products by type of product (liquified petroleum gases such as propane, E85, motor gasoline, jet fuel, diesel fuel, other distillate, residual oil, and all other refined products). However, it provides little detail regarding U.S. imports and exports of refined products. Consequently, to estimate U.S. refinery output of specific types of refined products, which may be calculated as consumption plus exports minus imports, we had to estimate the composition of imports and exports of refined products. These estimates, in turn, were used in conjunction with forecasts of refined product consumption to estimate U.S. refinery inputs and outputs of refined products and the resultant changes in volumes between 2016 and 2020. Because the estimated changes in refinery inputs (other than crude oil) were small, we allocated all of them to PADD 3. The moderate changes in estimated aggregate output of refined products were allocated to PADDs 2, 4, and 5 based on regional changes in consumption (as developed by ICF using regional forecasts provided in the 2017 AEO), with the residual allocated to PADD 3. We maintained PADD 1's output of refined products at 2016 volumes because it appears that refining process capacity might be slightly lower in 2020 than in 2016 (the study assumes that there will be no PADD 1 refinery closures through 2020). These estimates are developed in Exhibits B3 through B8 in ICF Report Appendix B.

Exhibits B9 through B12 in ICF Report Appendix B show the volumes of gasoline (by type and grade) and diesel, volumes of renewables, and RVP and octane of finished gasolines and ethanol used in the PADD-level refinery modeling. These exhibits correspond to those shown in ICF Report Appendix A pertaining to the Calibration case, with the only difference reflecting relatively small changes in gasoline volumes.

D.2.2 Biofuel Summary Assumptions

D.2.2.1 Ethanol Summary Assumptions

In order to utilize ethanol as an economical choice in the model for the No-RFS case, it was necessary to ensure that the model incorporated the following:

1. **Ethanol Properties:** Ethanol's value as a gasoline blending component is based on its octane, blending RVP, and low sulfur and benzene content. Ethanol's high-octane value

allows refinery BOBs to be produced and shipped by pipeline and marine methods at octane levels 2–3, octane numbers below the octane levels posted at the pump.

2. **Production Cost Curves:** With 90% of the U.S. ethanol production in PADD 2, the cost of ethanol production at various production volumes in this region needed to be estimated. This process involved consideration of various ethanol plant configurations, sizes, and technologies to estimate the supply cost for different tranches of supply.
3. **Logistics:** The costs to distribute or procure ethanol from the primary production area in PADD 2 (Midwest) to terminals in various markets where refined product is moved to distribution terminals was developed. For example, Florida and the Southeast may have some local ethanol production, but all discretionary ethanol would typically arrive by unit trains from the Midwest into Atlanta, Tampa, and so forth, or through transload facilities in the region. Total distribution costs were estimated from production to primary distribution to local terminal rack costs.
4. **State Mandates:** Mandates by individual states for ethanol blending, either direct or indirect (e.g., California's LCFS), are assumed to remain in place in the No-RFS cases.
5. **Other** key assumptions that were
   a. incorporated in the Model included: E85 volumes in the gasoline pool (fixed)
   b. Demand levels in 2020 versus 2016 and grade mix
   c. Impact of the MARPOL requirements for bunker fuel sulfur level

The model set up using these assumptions would allow an evaluation of the likelihood of ethanol to remain in the gasoline pool on an economical basis reflecting the octane, RVP, and distillation impacts on refinery operations as ethanol was "backed out" of the gasoline pool.

More detail and discussions of each of these areas are included in the body of this report.

D.2.2.2 Biodiesel Summary Assumptions

Biodiesel blending in fuel (biodiesel or renewable diesel) would not be expected to include a significant impact on refinery operations. For example, if more biodiesel is used to meet domestic demands, it will likely simply mean that more petroleum-based diesel fuel is exported. However, biodiesel's relatively similar fuel characteristics to petroleum-based diesel do not provide it with the same degree of additional value as ethanol does with its very high-octane value.

While biodiesel production refineries are primarily located in PADD 2, there is a more diverse mix in other PADDs than for ethanol. Therefore, EPA agreed that it would be prudent to develop specific biodiesel cost curves for each PADD region. Key biodiesel analysis requirements included the following areas:

1. **Biodiesel Properties:** The study assumes that biodiesel has one set of properties regardless of the source feedstock and that these properties (e.g., cetane sulfur content) are similar to hydrocarbon-based diesel fuel, so that all biodiesel is valued as a fuel extender.
2. **Production Cost Curves:** Biodiesel cost curves were developed for all five PADD regions. This process involved consideration of various biodiesel plant sizes and feedstocks to estimate the supply cost for different tranches of supply.
3. **Logistics:** Biodiesel typically moves to markets via rail or truck, and, in some cases, marine. The size of movements is typically smaller than for ethanol shipments, so manifest trains are much more prevalent than unit trains. The logistics analysis assumed that movements between PADDs would occur on manifest trains or via truck movements within a PADD.

The cost of biodiesel in different markets is based on the production cost in that market, the production cost in other source markets (e.g., PADD 2 supply into PADD 1), and logistics costs.

1. **State Mandates:** Mandates by individual states for biodiesel blending, either direct or indirect (e.g., California's LCFS), are assumed to remain in place in the No-RFS cases.
2. **Biodiesel Tax Credit (BTC):** The Biodiesel tax credit of $1.00/gal was assumed to be in place for the No-RFS case.
3. **Other** key assumptions that were incorporated in the model included:
   a. Demand levels in 2020 versus 2016
   b. Impact of the MARPOL requirements for bunker fuel sulfur level

As with ethanol, the PADD-level models were configured such that biodiesel use was determined by its economics, subject to minimum blending levels representing state and local mandate volume requirements, which were assumed to remain in place.

D.2.3 Ethanol Detailed Assumption Descriptions

D.2.3.1 Ethanol Properties

The properties of ethanol that primarily affect its refining value as a gasoline blendstock are:

- Blending octane
- Effect on gasoline RVP

Ethanol is a very-high-octane gasoline blendstock. Table D.1 shows ethanol's volumetric blending octane and the octane increase associated with blending ethanol (at 10 vol%) in regular and premium grade gasolines. Finished regular gasoline ranges in octane from about 85 to 88, whereas premium gasoline ranges in octane from about 91 to 93.[8]

Table D.1. Volumetric Blending Octanes (AKI) of Ethanol, by E10 Gasoline Grade.

Grade	Octane of Finished Gasoline	Volumetric Blending Octane of Ethanol	Implicit Octane of Gasoline BOB	Octane Increase Due to Ethanol
Regular	85	125.3	80.5	4.5
	86	123.8	81.8	4.2
	87	122.3	83.1	3.9
	88	120.8	84.4	3.6
Premium	91	116.3	88.2	2.8
	92	114.8	89.5	2.5
	93	113.3	90.8	2.2

Notes:

(1) AKI stands for "Anti-Knock Index." It is equivalent to (R + M)/2, which represents the average of a gasoline's Research Octane and Motor Octane.

(2) The volumetric blending octane of ethanol and the implicit octane of BOBs are derived using the Molar Blending approach, with ethanol's molar blending RON set at 109 and MON set at 93.2.

These data indicate that blending ethanol at 10 vol% significantly improves the octane of the hydrocarbon BOBs to which it is added. However, ethanol's effective blending octane (and the improvement of octane) diminishes as the octane of the finished gasoline increases. Hence, ethanol's effective blending octane is higher for regular grade gasolines than for premium grades, which, as a corollary, generally means that the octane value of ethanol to refiners will be highest for regular grade gasolines.[9]

Ethanol also has a significant effect on the RVP (in psi) of gasoline. For summer gasoline, the RVP uplift (the delta between the RVPs of the BOB and the finished gasoline) ranges from about 1 psi to almost 1.6 psi, as indicated in Table D.2. The RVP uplift is greatest for finished gasolines with low RVP that do not qualify for the 1-psi RVP waiver.

[8] In this case, octane refers to the anti-knock index (AKI) or its equivalent: (R + M)/2 (the average of research octane and motor octane, which generally is the octane that is posted on gasoline station pumps).

[9] The refinery models are configured to represent the effective blending octanes of ethanol, given the specified AKI of the various finished E10 gasolines included in the models. This results in the refinery model producing BOBs (the hydrocarbon portion of the E10 gasolines) with implicitly lower AKIs than the finished gasoline, consistent with the numbers in Table D.1.

Table D.2. Implicit RVP (in psi) of Ethanol, by Season and Type of Finished E10 Gasoline.

Season and Type of Gasoline	Finished Gasoline RVP	Ethanol RVP Uplift	Implicit BOB RVP	Implicit Ethanol RVP
Summer				
No Ethanol RVP Waiver				
RFG	7.1	1.55	5.55	19.2
Low-RVP	7.6	1.43	6.17	18.9
Conventional	8.7	1.25	7.45	18.8
Ethanol RVP Waiver				
Low-RVP (7.0)	8.0	-	6.8	6.8
Low-RVP (7.8)	8.8	-	7.6	7.6
Conventional (9.0)	10.0	-	8.7	8.7
Winter[1]				
Various Winter RVPs	12.5	0.90	11.60	11.60
PADD 3	13.1	0.86	12.24	12.24
PADD 1 (Southeast)	13.4	0.85	12.55	12.55
PADD 1 (Mid-Atlantic)	13.6	0.84	12.76	12.76
PADD 1 (northeast)	13.7	0.83	12.87	12.87
PADD 2	13.8	0.83	12.97	12.97
PADD 2	14.9	0.78	14.12	14.12

Notes:

The following non-linear formula was used to compute the above estimates of ethanol's RVP uplift.

$RVP_{delta} = 5.4784*RVP_{bob}^{-0.737}$

The formula was derived using data on RVP for finished E10 gasolines and corresponding BOBs from API's 2010 study entitled "Determination of the Potential Property Ranges of Mid-Level Ethanol Blends."

The implicit blending RVP of ethanol is then calculated to yield the estimated RVP uplift, assuming that RVP blends exponentially at the 1.2 power.

Note 1: Winter RVP's for each PADD are assigned to gasoline shipped into the designated PADD from other PADDs. For example, RVP's shown for PADD 1 (Southeast) and PADD 1 (Mid-Atlantic) are assigned to gasoline shipped from PADD 3 to these areas.

In the summer season, RFG (reformulated gasoline) and some low-RVP and conventional gasoline do not qualify for the ethanol RVP waiver. This means that the summer RVP standards set for those gasolines must be met whether or not ethanol is used as a blendstock. As indicated in the table, this requires refiners to significantly reduce the RVP of the hydrocarbon BOBs into which ethanol is blended and, as a corollary, reduces the refining value of ethanol as a blendstock (because it increases the refiner's costs of producing BOBs).[10]

[10] The RVP of ethanol is set at the implicit ethanol RVP values shown in the last column of the table, which forces the refinery model to produce BOBs with lower RVP in order to meet the RVP standard for finished E10 gasoline.

However, much of the country uses finished E10 gasoline that qualifies for a 1-psi RVP waiver (i.e., the RVP of finished E10 can be 1 psi higher than the RVP standard for non-ethanol blended finished gasoline). In that type of gasoline, the RVP properties of ethanol do not negatively affect the refining value of ethanol because refiners do not have to reduce the RVP of BOBs to accommodate ethanol blending. The refining models reflect this by assigning the RVP standard for finished E0 gasolines qualifying for the RVP waiver to the ethanol blended with those gasolines (so there is no RVP effect). For example, conventional E0 summer gasoline has an RVP standard of 9.0 psi (8.7 psi with a safety margin); however, because of the RVP waiver, E10 can have an RVP of 10.0 psi. In the refinery modeling, we set the RVP standard for E10 at 8.7 psi and the blending RVP of ethanol at 8.7 psi, which means that the E10 BOB will have an RVP of 8.7 psi and the model does not register an RVP penalty for ethanol.

In the winter season, ethanol generally is blended into BOBs that meet ASTM regional RVP standards, which means that (1) the RVP of finished winter E10 will be higher than the corresponding BOB RVPs, and (2) there is no adverse RVP effect for refiners blending ethanol in winter gasolines. However, to more accurately represent the average RVPs of winter BOBs, the RVP uplift was backed out of the average RVP of finished E10 gasolines as derived from the Alliance 2015 North American Fuel Survey for various geographic regions. Those calculations are shown in the Winter section of the above table.

Ethanol also has two other properties that can significantly affect its value:

- Ethanol is pipeline incompatible because it readily absorbs water (hygroscopic).
- Ethanol’s energy content is significantly lower (about 34% lower) than hydrocarbon gasoline.

The former means that ethanol cannot be blended in gasoline at the refinery and shipped via pipeline to terminals. Instead, ethanol must be shipped separately, by train or truck, from ethanol plants to terminals and blended there with BOBs to produce finished gasoline for final distribution to gasoline stations. The latter affects the relative fuel economy of finished hydrocarbon gasoline versus ethanol blended gasoline (and E85). To the extent that consumers are, or become, aware of the adverse effects of ethanol on fuel economy, they generally would expect to pay less for ethanol-blended gasoline than pure hydrocarbon gasoline. The adverse effects of ethanol on fuel economy become increasingly noticeable as the ethanol content of the blended gasoline increases.

D.2.3.2 Ethanol Production Cost Curve

BASIS FOR COST CURVE

Corn production has thrived in the Midwest due to fertile lands and climate, and, as a result, the Midwest is the epicenter of ethanol production due to the cost-effective benefits of locating ethanol facilities close to corn production. In 2017, the Midwest produced more than 91% of U.S. ethanol.[11]

Table D.3. 2017 Ethanol Production by PADD.

PADD	Ethanol Production (million gal/year)	Ethanol Production (bbl/d)	Percentage
1	437	28,506	2.8%
2	14,177	924,788	91.1%
3	444	28,963	2.9%
4	187	12,198	1.2%
5	310	20,222	2.0%
Total	15,555	1,014,677	100.0%

Because ethanol production is so heavily leveraged to the PADD 2 region, the relative cost variations in ethanol price (e.g., due to plant size, efficiency, byproduct values) will be examined using one ethanol cost curve with multiple tranches of supply cost based on Midwest facilities.[12] Because other PADDs have less than 3% of the ethanol market, it was assumed that those ethanol plants would likely be providing base ethanol supply in those markets, with all economical ethanol supply being driven by the cost curve and logistics.

COST CURVE MODEL DEVELOPMENT

The cost curve model utilizes assumptions to determine the supply curve for ethanol production, starting by characterizing the existing infrastructure of ethanol-producing facilities across the United States. The Renewable Fuels Association lists ethanol-producing facilities in their *Ethanol Industry Outlook*.[13] In this report, the association has facility locations, feedstock, production capacity, and production. This listing was supplemented with additional facilities from other resources. The model split these facilities into quintiles in order to apply economy-of- scale assumptions to different size facilities across the industry. Each quintile had unique assumptions for efficiencies, capital costs, and feedstock costs, with the median capacity of each quintile equal to 25, 50, 60, 100, and 130 million gal/year. The

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[11] https://ethanolrfa.org/resources/annual-industry-outlook
[12] Note that 2017 U.S. ethanol production was not all blended into U.S. refinery gasoline. About 90 K bbl/d was exported, and about 60 K bbl/d was blended into imported gasoline.
[13] https://ethanolrfa.org/resources/annual-industry-outlook

ethanol production supply curve was then determined by aggregating the cost per gallon associated with capital costs, operating and maintenance costs, feedstock costs, and accounting for other co-products at each facility when producing ethanol.

According to studies conducted by Iowa State University and the University of Illinois, the annualized capital cost for a 100-million gal/year facility is $0.21/gal.[14] This value was scaled to the various facility sizes using the 6/10ths scaling factor as outlined in the equation below.

$$C_A = \frac{[0.21 * 100] * (\frac{F}{100})^{0.6}}{F_A}$$

where:

C_A = Cost per gallon for the capital costs of facility A

0.21 = Average cost per gallon due to the capital costs at a 100-million gal/year facility

100 = Size of the facility in million gallons per year

F = Median size of facility A for a given quintile in million gallons per year

0.6 = Constant used for applying economies of scale to capital expenditures

The cost per gallon was then applied to all facilities within the quintile. In addition, the variable operating and maintenance costs per gallon of ethanol produced was assumed based on studies to be $0.17/gal of ethanol produced, and this value was applied to all facilities.[15]

The model assumes that the unit cost of inputs, including corn, natural gas, denaturant, and co-products such as distillers dried grains with solubles (DDGS) vary, depending on the size of the facility. Larger facilities oftentimes have economies of scale that provide additional efficiencies from buying in bulk, stable contractual arrangements, and additional transportation infrastructure in place. How these economies of scale were applied to estimating the supply curve varied based on the input as described below:

- The model took the average 2016 ethanol price, which was estimated assuming a corn price of $3.48 per bushel,[16] and divided this by the average ethanol conversion from a bushel of 2.86 gal of ethanol per bushel. The yield was then assumed to deviate approximately 5.5% to

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[14] Iowa State University, retrieved from https://www.extension.iastate.edu/agdm/energy/xls/d1-10ethanolprofitability.xlsx

[15] University of Illinois, retrieved from https://farmdocdaily.illinois.edu/2017/02/the-profitability-of-ethanol-production-in-2016.html

[16] Retrieved from the U.S. Department of Agriculture's Economic Research Service, Bioenergy Statistics, Table 14; https://www.ers.usda.gov/data-products/us-bioenergy-statistics/

account for conversion technologies that ethanol-producing facilities have implemented.[17] The difference in yield was varied by plant size, as follows:

- The median facility production capacity of 60 million gallons per year (MGPY) with a range of 40–70 MGPY was assumed to produce 2.86 gal of ethanol per bushel of corn.
- Facilities with a production capacity of greater than 25 MGPY and less than 40 MGPY were assumed to operate 3% less efficiently than the median production facility.
- Facilities with a production capacity of greater than 40 MGPY and less than 60 MGPY were assumed to operate 1.5% less efficiently than the median production facility.
- Facilities with a production capacity of greater than 70 MGPY and less than 110 MGPY were assumed to operate 1.3% more efficiently than the median production facility.
- Facilities with a production capacity of greater than 110 MGPY were assumed to operate 2.5% more efficiently than the median production facility.
- The model set the natural gas price at $3.39 per million British thermal units (MMBtu) and varied the volume of natural gas necessary to produce a gallon of ethanol, assuming that some facilities have installed and use combined heat and power and other energy-efficient equipment. The natural gas necessary per gallon of ethanol produced varied from 16.5 to 29.4 MMBtu, which equated to $0.06 to $0.10/gal.
- In order for ethanol to be transported as a fuel and forego the taxes to which consumable ethanol is subjected, fuel ethanol adds a denaturant before transportation. Ethanol is then transported, typically as E98, where it will then be blended with gasoline at a later point. This denaturant, which is typically natural gas liquids, accounts for 2% of the final product.

Data from EIA indicate an average price of natural gas liquid at $5.04/MMBtu, or the equivalent of about $0.51/gal.[18] With the assumption that this represents only 2% by volume of the denatured product, adding the denaturant and displacing a corresponding volume of clear ethanol leads to a small reduction in the cost of fuel ethanol (on the order of $0.01/gal).

- When estimating the ethanol supply curve, it was necessary to recognize that, in the ethanol production process, other products, including corn oil and DDGS, are produced and sold as co-products. These other products, similar to the cost impacts, influenced the break-even cost

[17] Based on ICF analysis of publicly available presentations from Christianson, PLLP, regarding their Biofuels Benchmarking™ service. For instance, *Ethanol Plant Performance & Co-Product Quality in 2017*, presented by C. Lindstrom and J. Cline at the Distillers Grain Symposium, May 2017.
[18] EIA, U.S. Natural Gas Liquid Composite Price; https://www.eia.gov/dnav/ng/hist/ngm_epg0_plc_nus_dmmbtum.htm

of ethanol in the various tranches in the cost curve so that the cost curve shows how the ethanol costs vary with the size categories. Corn oil yield was assumed to vary from 0.35 to 0.82 pounds per bushel (lbs/bushel), while DDGS was assumed to vary from 13.75 to 16.05 lbs/bushel.[19] We assumed that the plants with higher conversion efficiency for corn bushel to ethanol production had lower DDGS yields. The model assumed corn oil prices of $0.30/lb and DDGS prices of $116/ton.[20]

- Given the limited data availability at the production facility level for the varying efficiencies introduced for natural gas consumption and corn oil yield, ICF assumed a random distribution of efficiencies for these co-products across the five representative production facilities. The high price tail to the ethanol production cost curve shown in Figure D.1, below, is due to the confluence of smaller plant size and randomly applied higher operating cost factors.
- ICF assumed a constant cost of $0.05/gal of ethanol produced for marketing, regardless of facility size. The smallest ethanol facilities were removed from consideration in the analysis, with a cutoff of any facility producing less than 25 million gal annually. This was to remove any potential production facilities that rely on waste sugars and starch, which sources its feedstocks primarily from the beverage industry. The lower pricing of the feedstocks from waste was unknown; however, it offsets the higher fixed costs of these smaller plants. Because of their very small volume, ignoring these facilities did not impact our analysis.Table D.4 illustrates two examples of the production costs and co-product revenues at two different ethanol plants with production capacities of 50 and 100 million gal per year, respectively.

Table D.4. Ethanol Production Costs ($/gal of ethanol produced).

Production Capacity MGPY (million gal/year)	Production Costs						Co-Product Revenue		Total
			Plant Costs						
	Corn	NG	Var.	Fixed	Denat.	Market.	Corn Oil	DDGS	
50	1.24	0.10	0.28	0.17	0.01	0.05	0.07	0.30	1.47
100	1.20	0.06	0.21	0.17	0.01	0.05	0.09	0.28	1.33

[19] Based on ICF analysis of publicly available presentations from Christianson, PLLP, regarding their Biofuels Benchmarking™ service. For instance, *Ethanol Plant Performance & Co-Product Quality in 2017*, presented by C. Lindstrom and J. Cline at the Distillers Grain Symposium, May 2017.
[20] Prices reported for corn oil and DDGS are from The Jacobsen.

COST CURVE

The resulting ethanol supply-cost curve is presented in Figure D.1.

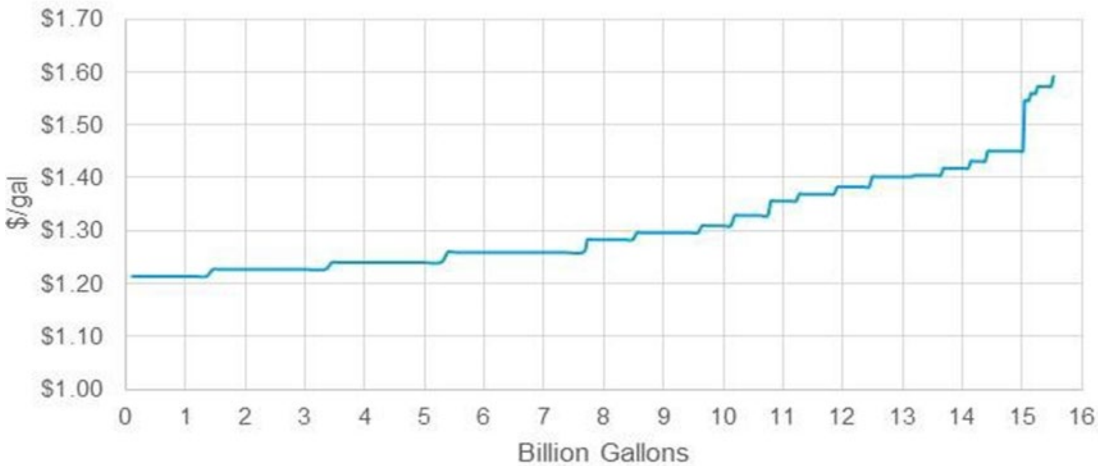

Figure D.1. 2016 Ethanol Supply Curve.

The ethanol supply-cost curve in the figure above reflects the diverse range of production costs at ethanol facilities as a function of myriad parameters outlined previously, including corn starch to ethanol conversion efficiency, co-product yields, and fixed and variable cost differences between facilities. The supply-cost curve also supports the concept that marginal cost producers will generally set the market ethanol price (according to the U.S. Department of Agriculture's [USDA] Economic Research Service [ERS], the average ethanol price in 2016 was $1.55/gal).[21]

D.2.3.3 Ethanol Logistics

Ethanol logistics costs were developed to estimate the additional cost to move ethanol from production sites in PADD 2 to market demand areas (e.g., East Coast, West Coast). Once the ethanol is in the destination market area, there are additional costs for moving ethanol from those receiving hubs into distribution terminals, where the ethanol price would be comparable to refinery BOB product at the terminal rack.

The overall approach and analysis follows.

Cost Estimation Approach BASIS:

1. The ethanol cost into markets is based on the ethanol production cost curve plus total cost to the distribution terminal rack. The bulk of the cost to distribution terminals outside PADD 2

[21] The USDA ERS indicates that they retrieve ethanol prices from the Nebraska Energy Office, https://neo.ne.gov/.

is the unit train cost to that market area. The distribution cost will, over time, mirror the spot transaction differentials in those markets versus the Midwest farm FOB[22] price.

2. Ethanol transactions from producers to obligated parties are primarily term transactions based on the regional spot barge/rail prices as reported in the Oil Price Information Service (OPIS) weekly report, Argus, and so forth.
3. Rail deliveries of ethanol can occur at a number of terminals in destination markets, or ethanol can be transloaded from railcars into trucks. Ultimately, all ethanol would need to be delivered to a blending terminal to be loaded into trucks for consumers as E10.
4. The spot transaction costs determine the price of ethanol in the specific location; deliveries before that location (e.g., Phoenix before Los Angeles) or after (San Diego versus Los Angeles) can be lower or higher, respectively (although local market issues can affect that, the logic is that, on average, the cost in a given region will be higher or lower than that regions spot marker based on distance).

SPOT BARGE/RAIL MARKET PRICE ANALYSIS

EPA's Argus data, showing spot bulk/rail prices for 2016/2017 (averages), and OPIS data based on an October 12, 2017, OPIS weekly report are presented in Table D.5.

Table D.5. Spot Bulk/Rail Prices for 2016/2017 (averages).

PADD	State	Ethanol prices in U.S. $/gal.	Wkly. Avg.	Argus-2016	Argus-2017
1	NY	New York	1.49	1.596	1.590
1	GA	Atlanta		1.636	1.588
1	FL	Tampa	1.56	1.707	1.655
2	IL	Chicago	1.43	1.519	1.518
2	IL	Chicago Rule 11	1.40	1.515	1.501
2	NE	Nebraska	1.25	1.411	1.406
3	TX	Gulf Coast	1.50		
3	TX	Dallas	1.44	1.564	1.547
5	CA	Los Angeles	1.58		
5	CA	San Francisco	1.58		
5	WA	Washington	1.54		

[22] FOB means "free on board." Simply put, FOB prices exclude all insurance and freight charges. Most fuel is sold either FOB (effectively priced at the loading port) or CIF (cost, insurance, and freight charges for shipping products; effectively priced at the delivery port); https://www.platts.com/glossary#Free on board

The spot bulk/rail pricing reported by Argus and other pricing services is the basis for transactions between buyers and sellers in the identified market. In most cases, these prices are similar to the Midwest plant-gate price plus unit train costs to the destination market (although these can vary over time based on local market issues in the destination market).

The chart shows for 2016 and 2017 a relatively flat average price market, although Tampa prices dropped slightly in 2017. The October 12, 2017, OPIS data (single day) shows more bulk spot/rail locations, and clearly shows that ethanol prices have softened versus the Argus 2016 and 2017 averages, with benchmark Chicago and Nebraska prices down about $0.09/gal and $0.16/gal, respectively. New York, Tampa, Dallas, and Los Angeles prices are all down about $0.10/gal from the 2017 average from Argus. Note: In general, the OPIS and Argus spot prices should be very close on any given day as they reflect transactions done in the reporting markets.

The relative cost between the Nebraska plant-gate cost and the destination market cost reflect the bulk acquisition prices in those markets. The differential from the Nebraska plant-gate cost and the destination market approximate the cost for rail movements (e.g., the market differential from a Nebraska facility to Tampa in 2016 was about 29.6 cents/gal, and about 10.8 cents/gal to Chicago. The full distribution cost then adjusts the costs for distribution, handling, and blending within each PADD (see below).

ADD-ON COSTS AFTER SPOT MARKET

Spot market purchases of ethanol are similar to spot market purchases of gasoline RBOB/CBOB and CARBOB (Reformulated, Conventional and California Blendstocks for Oxygenate Blending). The buyer acquires the commodity at the spot market location (e.g., New York Harbor) and, in most cases, the buyer is required to transport the product to the blending terminal for delivery of the fuel to service stations.

In most cases, the BOB supply to blending terminals is via marine or pipeline transport. Trucks are rarely used because the petroleum infrastructure provides economical supply by marine and pipeline (although trucks can be used during times of disruption if there are infrastructure problems).

For ethanol, the situation is more difficult because, in almost all cases, pipeline is not an option. Consequently, gasoline sellers must secure ethanol at blending terminals in the most economical manner. Most gasoline sellers will purchase ethanol at term conditions from major ethanol producers on a spot bulk rail/barge price basis. In other words, they have ensured supply at a market location, but the price will vary based on the market conditions. They may or may not pay a premium or discount for the volume based on their contract with the producer.

There appear to be several options for moving the ethanol from spot sources to blending terminals:

- Truck movements from a unit train destination hub (e.g., Lomita, Stockton, Tampa, Atlanta, Dallas) to the blending terminal. Cost is a function of distance and time to load/deliver/return in 9,000-gal trucks.
- Truck movements from an ethanol plant directly to a blending terminal. Obviously, this is used primarily in the Midwest where ethanol plants are located from Ohio to the Central Plains states.
- Barge movements from the unit train destination to the blending terminals (primarily used in New York and Albany, but also from the Sauget, Illinois, area, as well as the Chicago market. In addition, marine movements from the Gulf Coast and New York areas into Florida markets are used.

Gasoline sellers also can purchase ethanol from producers for loading on manifest trains for delivery to more remote terminals. These volumes would supply markets without the ability to receive unit trains, and would be more costly for rail delivery.

- The full cost of this option would include the manifest train cost plus the cost to transload the ethanol into the blending terminal (most terminals do not have rail delivery capability, so either the ethanol would be loaded into a terminal in the area with rail capability and trucked over, or would utilize a transflow facility to directly load into trucks to move into the blending terminal).
- The more remote terminals also could simply receive truck deliveries from unit train distribution hubs, although the distance and truck cost would be weighed against other options such as manifest train supply.
- Some locations (e.g., Charleston, South Carolina) could get marine deliveries, manifest rail deliveries, or unit train plus truck deliveries.

To try to arrive at some analytical assessment of multiple options, the terminal rack prices for ethanol in PADD 2 versus the Chicago hub price (OPIS data) were examined. PADD 2 was appropriate because more than 80% of all OPIS-reported terminal rack ethanol prices were in the Midwest versus other PADDs. The average rack prices at these terminals for the week ending October 12, 2017, were compared with the OPIS spot barge/rail price in Chicago for the same week (which was $1.43/gal).

Terminal rack locations were chosen that had, at a minimum, several producers/suppliers selling ethanol at the rack, so there was a competitive market for the ethanol. Locations in Oklahoma were excluded because they were far removed from Chicago and likely would be priced off of the Dallas spot barge/rail price.

The average mark-up from the Chicago market is about $0.11/gal, which includes an odd data point from the small Lemars, Iowa, location in western Iowa, possibly reflecting a tight supply at that terminal on this date.

Table D.6. PADD 2 Ethanol Rack Prices versus Chicago and Nebraska Spot (cents/gal).

Selected Locations	State	PADD	5-Day Average	Difference Between OPIS Rack and OPIS Chicago Spot Price	Difference Between OPIS Rack and OPIS Nebraska Spot
Alexandria	MN	2	150.8	7.9	25.5
Argo	IL	2	149.0	6.0	23.7
Bettendorf	IA	2	152.9	10.0	27.6
Carthage	MO	2	158.5	15.5	33.2
Columbia	MO	2	152.2	9.3	26.9
Columbus	NE	2	153.3	10.3	28.0
Concordia	KS	2	159.9	16.9	34.6
Des Moines	IA	2	149.3	6.4	24.0
Doniphan	NE	2	152.0	9.0	26.7
Fargo	ND	2	153.3	10.3	28.0
Geneva	NE	2	154.4	11.4	29.1
Grand Forks	ND	2	159.6	16.7	34.3
Great Bend	KS	2	156.7	13.8	31.4
Iowa City	IA	2	152.7	9.7	27.4
KC/Magellan	KS	2	154.5	11.6	29.2
Lemars	IA	2	164.7	21.7	39.4
Lincoln	NE	2	151.0	8.1	25.7
Mankato	MN	2	153.8	10.9	28.5
Milford	IA	2	153.2	10.3	27.9
Minneapolis	MN	2	154.0	11.1	28.7
Oklahoma City	OK	2	165.4	22.5	40.1
Omaha	NE	2	153.7	10.8	28.4
Rochester	MN	2	153.1	10.1	27.8
Roseville/Magellan	MN	2	154.0	11.1	28.7
Tulsa	OK	2	162.81	19.9	37.5
Wichita	KS	2	158.0	15.0	32.7

Otherwise, the data are surprisingly consistent. As noted in prior correspondence, the terminal rack prices do not necessarily (or at all) reflect the ethanol price paid by the major blenders, but it is a decent indicator of the ethanol market at that location.

Consequently, for PADD 2, an ethanol add-on of $0.11/gal to the Chicago spot barge/rail price was appropriate (it should be noted that the ethanol sellers at the rack likely are including some profit in their price).

In addition, the add-on versus the Nebraska spot also is fairly consistent. However, for PADD 2, the Chicago spot is best for a basis because the Nebraska price is FOB ethanol plant gate and is a cost for rail movements (i.e., unit train origins).

For other PADDs, OPIS publishes minimal blending terminal rack price data. It appears that this may be because most of the ethanol moved to coastal markets is already contracted for by either the large gasoline sellers (e.g., ExxonMobil, BP) or exporters. The geographic disparity around the Argus spot locations (e.g., Dallas, Tampa, New York) is not dissimilar to the disparity around the Chicago spot location, which reflected the $0.11/gal average spread versus the spot price. Therefore, one option (clearly the simplest) is to apply the $0.11/gal add-on to all market regions. For a lack of more detailed information, we chose to apply the $0.11/gal add-on to most of the ethanol sales. This is a simplification; terminals close to the unit train receiving terminals would have a lower distribution cost and terminals farther away would have a higher cost. However, the gasoline prices in each PADD also are simplified in a similar manner, thus the analysis is consistent by using these assumptions.

ESTIMATED TOTAL DELIVERED COST FROM PLANT GATE

Table D.7 shows the total estimated cost in origin and destination markets for ethanol at the distribution terminal.

Table D.7. Ethanol Terminal Costs ($/gal).

PADD/Region	Spot Rail/Barge Basis	2016 Spot*	Add-On	Total Cost
PADD 2 - Midwest	Chicago Argo	$1.519	$0.110	$1.629
PADD 1 - NY Region	NYH Barge	$1.596	$0.110	$1.706
PADD 1 - Florida	Tampa Rail	$1.707	$0.110	$1.817
PADD 1 - Southeast	Atlanta Rail	$1.636	$0.110	$1.746
PADD 3 - Texas/La	Dallas Rail	$1.564	$0.110	$1.674
PADD 4	Nebraska Rail	$1.411	$0.280	$1.691
PADD 5	Carson (LA) Low CI	$1.683	$0.090	$1.773

** Argus 2016 Average*

The full breakdown of these costs as differentials to the Nebraska plant-gate price is shown in Table D.8. The table shows the locational cost to the market and the add-on cost to the blending terminal. Note that this table provides several destination locations in large-volume PADDs 1 and 5, which are needed by the MathPro model to estimate ethanol costs for markets within a PADD region.

Table D.8. Total Distribution Cost.

Location		Distribution Cost to:			Total	
		Hub/Terminal (¢/g)		Blending Terminal (¢/g)		
PADD	Area	To Chicago	From Chicago		(¢/g)	($/b)
PADD 1	Florida/Tampa		17.8	11.0	35.8	15.0
	Southeast/Atlanta		11.7	11.0	29.7	12.5
	VA/DC/MD		9.7	11.0	27.7	11.6
	Pittsburgh		6.2	11.0	24.2	10.2
	New York		7.7	11.0	25.7	10.8
PADD 2	Chicago	7.0	0.0	11.0	18.0	7.6
	Tennessee		9.7	11.0	27.7	11.6
PADD 3	Dallas		4.5	11.0	22.5	9.5
PADD 4			6.2	11.0	24.2	10.2
PADD 5	Los Angeles		16.4	9.0	32.4	13.6
	Arizona		16.4	9.0	32.4	13.6
	Nevada		12.4	9.0	28.4	11.9
	Northwest		12.4	9.0	28.4	11.9

Specific PADD Assumptions PADD 1 Market Assumptions

1. Northeast – The New York harbor spot price (barge) in 2016 applies to Pennsylvania to Maine markets, plus an additional inter-PADD movement cost. The intra-PADD movement cost was estimated to be $0.11/gal. Ethanol is supplied by barge from New York Harbor to coastal/river blending terminals, and by truck to regional inland terminals. Ethanol is supplied to New York City, Albany, and Philadelphia by unit train.
2. Southeast – The Atlanta spot market price (rail) applies to the Colonial/Plantation corridor from Alabama to Baltimore. The intra-PADD movement cost to blending terminals was estimated to be $0.11/gal. This region would be supplied by unit train to Atlanta, Baltimore,

and Charlotte (Baltimore and Charlotte spot rail prices should be very close to that for Atlanta as they would travel similar distances). Some markets (Nashville, Richmond, Norfolk, Charleston, and Savannah) could be supplied by manifest trains from ethanol plants at a cost above the Atlanta spot price, or the coastal locations could be supplied by barge from New York or Baltimore. A number of locations in the Southeast are also supplied by manifest trains that drop off ethanol railcars at transflow locations for direct loading into ethanol trucks to nearby terminals (e.g., Birmingham, Alabama). Each of these alternative options can vary in cost at some level above the Atlanta spot price, and it is believed that the $0.11/gal Midwest spread should cover this market.

3. Florida – The Tampa spot market rail/barge price should apply to Florida. Florida receives supply by rail into the Tampa Kinder Morgan hub and in Miami by marine deliveries from the Gulf Coast and New York on Jones Act vessels. Ethanol moves from Tampa to Orlando by pipeline. The competitive nature of the Florida ethanol market and the concentration of demand in major population centers near marine and rail/pipeline delivery locations may imply that the same add-on from the Midwest used in other markets may be several cents per gallon lower in Florida; however, without data to confirm this possibility and to make the analysis consistent with how the distribution of gasoline was modeled, the $0.11/gal add-on was used here as well.

PADD 2 Market Assumptions

The Midwest should reflect the Chicago Argo hub spot (barge/rail) price plus the $0.11/gal added seen in PADD 2 for Midwest rack locations.

PADD 3 Market Assumptions

The PADD 3 market should utilize the Dallas spot rail price from Argus. Dallas likely sources ethanol into the highly populated Dallas/Fort Worth region, as well as Oklahoma (PADD 2) and Houston. Houston also receives unit trains and has the ability to supply ethanol by truck into Texas Gulf Coast markets and by marine to Louisiana. As with PADD 1, there are virtually no OPIS rack postings in PADD 3; however, the blending terminal add-on should, on average, be similar to the Midwest at $0.11/gal.

PADD 4 Market Assumptions

About 40% of PADD 4 ethanol demand is met from in-PADD ethanol plants. Most of the PADD 4 plants are located in central and northeastern Colorado with reasonable access to the Denver and

Cheyenne demand centers. Only one plant, albeit a large one, is located in Burley, Idaho, some distance from Boise and Pocatello demand centers.

The balance of ethanol needs appears to be supplied by manifest rail deliveries from Nebraska, Kansas, and North Dakota ethanol plants into terminals and/or refineries (storage at rail-capable terminals is too small for unit train deliveries). Based on how OPIS rack prices in the region behave, PADD 4 ethanol contracts were assumed to be between ethanol refiners and local (and Nebraska) ethanol suppliers at some premium for the Nebraska FOB price. Manifest rail cost (500 to 1,000 miles) would be $0.15–$0.19/gal, plus cost for railcar lease, and then the additional rack distribution premium similar to the Midwest of $0.11/gal.

PADD 5 Market Assumptions

PADD 5 was based on the Argus LA (Los Angeles) spot rail price. This price is for ethanol rail deliveries into the Los Angeles market at Lomita and Colton, while similar rail hubs exist in the San Francisco area in Concord and Stockton. Data from OPIS indicated that market prices in Los Angeles and San Francisco are similar, and that the Pacific Northwest prices are lower by about $0.04/gal. The bulk of PADD 5 gasoline production and demand is based in California, although California production also feeds Phoenix and Nevada markets. In addition, the Nevada/Arizona market was assumed to follow a similar trend to the Pacific Northwest as the production was starting in California and traveling similar distances. The intra-PADD movement costs were estimated to be $0.09/gal, with select regions maintaining discounted rates due to their proximity to production. This is estimated to be somewhat lower than the Midwest benchmark due to a relatively high concentration of demand in markets near the major ethanol hubs (e.g., Lomita, Concord).

D.2.3.4 Ethanol State Mandates

In the No-RFS cases, individual state mandates are assumed to remain in place. Table D.9 lists the states and associated mandates incorporated in the model for 2020. Based on 2016 state volumes, about 156 K bbl/d of ethanol demand will be fixed in the model, even if ethanol blending is uneconomical. Volumes will be adjusted to 2020 based on the 2017 AEO demand changes from 2016 to 2020. These programs are assumed to remain in place in the No-RFS case.

Table D.9. State-Mandated Volumes – Ethanol.

PADD	State	Volume (K bbl/d)	Volume (million gal/year)	Ethanol Requirements
	Total	**156.3**	**2,396**	
PADD 2	Minnesota	17.7	271	10% minimum
	Missouri	21.2	325	10% minimum if ethanol is priced lower than gasoline
PADD 3	Louisiana	1.6	25	2% minimum only in ozone attainment areas
PADD 5	California	102.9	1,577	10% minimum required by LCFS
	Oregon	9.2	141	10% minimum, but only in regular gasoline; assume that 90% of the pool is regular
	Washington	3.7	57	2% minimum

Note that several states have incentive programs that may, under certain circumstances, result in additional ethanol blending. These were not included as an option in the No-RFS case.

D.2.4 Biodiesel and Renewable Diesel Fuel Detailed Assumption Descriptions

D.2.4.1 Biodiesel Properties

As noted, the study assumes that biodiesel and renewable diesel fuel have one set of properties regardless of the source feedstock. The model includes assumptions on specific properties, such as cetane value, sulfur content, and so forth, that are similar to hydrocarbon-based diesel fuel. None of the biodiesel properties are assumed to have any blending value that differ from the hydrocarbon diesel fuel it displaces.

D.2.4.2 Biodiesel Production Cost Curves

Biodiesel cost curves were developed specific to each PADD using assumptions regarding given feedstocks and various other plant costs. Biodiesel plants and capacities were first identified by PADD. Soy oil, corn oil, and yellow grease were considered as feedstocks, and a production cost per gallon was determined using annual average feedstock costs,[23] as well as natural gas and methanol prices as fuel and additional feedstocks, respectively. Fixed and other variable costs[24] were also included in the cost per gallon. These production costs were generated for the various feedstocks and for different facility sizes. Facilities with larger capacities were given higher efficiencies because larger plants typically produce more economically due to scale. Table D.10 summarizes the parameters for the production cost estimates.

[23] Biodiesel feedstock pricing was retrieved via The Jacobsen, a biofuel industry reporting publication.
[24] The Profitability of Biodiesel Production in 2016; https://farmdocdaily.illinois.edu/wp-content/uploads/2017/04/fdd010317.pdf

Table D.10. Biodiesel Production Cost Parameters.

Parameter		2016 Avg Price	Notes
Oil Feedstock	Soy Oil (¢/lb)	31.65	▪ Based on average monthly pricing reported by The Jacobsen. ▪ Assumed that 7.55 lbs/gal of biodiesel is produced.
	Corn Oil (¢/lb)	27.89	▪ Based on average monthly pricing reported by The Jacobsen. ▪ Assumed that 8.20 lbs/gal of biodiesel is produced.
	Yellow Grease (¢/lb)	< 20	▪ Used this feedstock as a proxy for yellow grease and used cooking oil. ▪ Assumed to be priced significantly lower than other feedstocks. ▪ Assumed that 8.00 lbs/gal of biodiesel is produced.
Conversion Rates			▪ Assumed a 5.5% spread in efficiency of plant operation, applied to the feedstock conversion factors (lbs/gal) included for each feedstock. ▪ The values noted above are for the Reference case. ▪ Included facilities assumed to be 3% and 1% less efficient. ▪ Included facilities assumed to be 1.25% and 2.5% more efficient.
Biodiesel Inputs	Natural Gas ($/MMBtu)	3.00	▪ Based on EIA reported data for 2016. ▪ Assumed that 7 standard cubic feet of natural gas used per gallon of biodiesel produced.
	Methanol ($/gal)	0.84	▪ Based on non-discounted reference prices posted by Methanex. ▪ Assumed that 0.71 lbs of methanol per gallon of biodiesel is produced.
Plant Balance Costs	Fixed Costs ($/gal)	0.26	▪ Based on numbers reported by farmdoc Daily.[25] ▪ Adjusted based on the size of the plant and the assumed financing schedule.
	Other Variable Costs	0.25	▪ Based on the numbers reported by farmdoc Daily.[26] ▪ Held constant across facility size and location.
By- Products	Glycerine (¢/lb)	5.05	▪ Based on average monthly pricing reported by The Jacobsen. ▪ Assumed that 0.9 lbs is produced per gallon of biodiesel produced.

Using PADD-specific production capacities, tranches of supply were generated based on the various costs per gallon from the parameters mentioned above.

Figure D.2, below, shows the supply curves generated as a function of price for each PADD.

We did not estimate separate production and distribution costs for renewable diesel, but included the renewable diesel's plant capacity and production volumes with biodiesel volumes. In doing so, we effectively assume that renewable diesel production and distribution costs are similar to those for biodiesel. We recognize that these assumptions may be conservative given the potential for co-processing

[25] Irwin, S. The Profitability of Biodiesel Production in 2014. https://farmdocdaily.illinois.edu/2015/01/profitability-of-biodiesel-production-in-2014.html

[26] Ibid.

renewable feedstocks at refineries to produce renewable diesel. This type of process would reduce significantly both the production cost and the distribution cost compared with those estimated for biodiesel. We opted to exclude this type of consideration from our analysis given that this type of renewable diesel production is not done at commercial scale today.

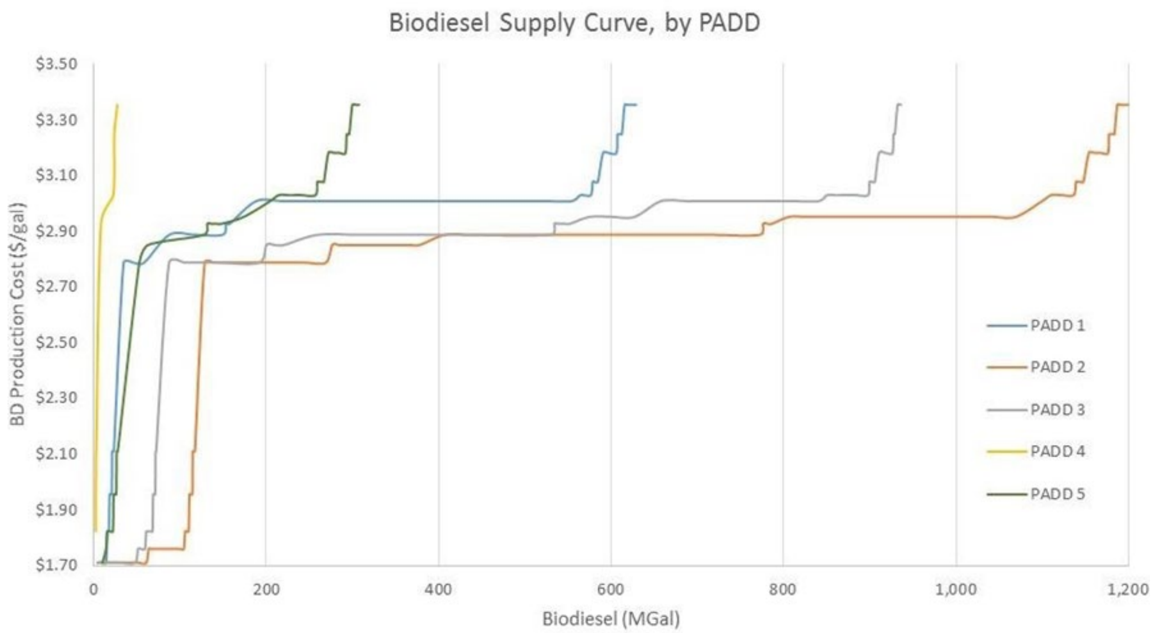

Figure D.2. 2016 Biodiesel Supply Curves.

Table D.11 illustrates three examples of the production costs and co-product revenues at three different biodiesel plants with production capacities of 11, 25, and 40 million gal/year using yellow grease, corn oil, and soy oil, respectively. All costs shown in the table below are in dollars per gallon of biodiesel produced.

Table D.11. Biodiesel Production Costs.

Feedstock	Capacity MGPY (million gal/yr)	Production Costs ($/gal)					Co-Product Revenue ($/gal)	Total ($/gal)
					Plant Costs		Glycerine	
		Oil	NG	MeOH	Var.	Fixed		
Yellow Grease	11	1.20	0.02	0.09	0.40	0.25	0.05	1.91
	25	1.18	0.02	0.09	0.28	0.25	0.05	1.78
	40	1.17	0.02	0.09	0.23	0.25	0.05	1.71
Corn Oil	11	2.32	0.02	0.09	0.40	0.25	0.05	3.03
	25	2.29	0.02	0.09	0.28	0.25	0.05	2.88
	40	2.26	0.02	0.09	0.23	0.25	0.05	2.81
Soy Oil	11	2.43	0.02	0.09	0.40	0.25	0.05	3.14
	25	2.39	0.02	0.09	0.28	0.25	0.05	2.98
	40	2.36	0.02	0.09	0.23	0.25	0.05	2.91

D.2.4.3 Biodiesel Logistics

Biodiesel and renewable diesel fuel can move between markets via rail, truck, or, in some cases, marine. The associated transportation costs were determined using a combination of both publicly available data and a number of assumptions.

The total volume of biodiesel sold in each PADD was estimated by summing the PADD biodiesel and renewable diesel fuel production and net volume shipped into or out of the PADD as reported by EIA (in 2016, all reported volume was by rail). Movements of biodiesel by rail are available from EIA[27] at both the intra- and inter-PADD levels. Rail costs in cents per gallon were determined as a function of distance using a cost basis provided in the Bates White report[28] and assumed transportation distances between PADDs. Internal PADD rail transportation costs were assumed to be $0.15/gal. Costs between PADDs are presented in Table D.12 (for the movements where volume was reported by EIA).

[27] https://www.eia.gov/dnav/pet/pet_move_railNA_a_EPOORDB_RAIL_mbbl_a.htm
[28] https://www.bateswhite.com/media/publication/116_2016.07.11%20Biodiesel%20paper%20final.pdf

Table D.12. External Rail Costs (cents/gal) Between PADDs.

	Origin				
Destination	**1**	**2**	**3**	**4**	**5**
1		15	25	No Movements	No Movements
2	15		18	No Movements	No Movements
3	No Movements	18		No Movements	32
4	No Movements	25	No Movements		18
5	No Movements	32	32	No Movements	

Rail movements do not account for all transportation of biodiesel. To determine biodiesel transported by internal trucking (within 300 miles), biodiesel production numbers[29] were considered. The difference between total biodiesel production at a PADD level and the volume of biodiesel transported by internal and external rail movements was assumed to be moved by truck.

Truck transportation costs were assumed to be $80/hour for truck and driver, plus an additional $0.80/mile for fuel and maintenance, divided by gallons hauled at 7,000 gal per truck. Truck round-trip miles and average speed and loading/unloading times were estimated for each PADD.

Table D.13 shows the internal truck movement costs for each PADD.

Table D.13. Internal PADD Trucking Costs ($/gal).

PADD	Costs*	Average Round-Trip Hours	Round-Trip Mileage
PADD 1	0.1219	7.67	200
PADD 2	0.1429	8.50	300
PADD 3	0.1429	8.50	300
PADD 4	0.1600	9.00	400
PADD 5	0.1429	8.50	300

* Any barge movements of biodiesel within a PADD were assumed to be approximately equivalent to trucking costs, meaning that any barge costs were accounted for by internal trucking estimates.

Summary of Costs

The following tables contain total cost estimates by PADD determined using the above analysis for the mentioned transportation modes.

[29] https://www.eia.gov/biofuels/biodiesel/production/

Table D.14. Rail Volumes and Costs by PADD.

Location	Rail Volume Within PADD (K gal)	Internal Rail Cost ($)	External Volume Received (K gal)	External Rail Received Cost ($)
PADD 1	23,352	3,502,800	70,644	10,693,200
PADD 2	51,534	7,730,100	12,054	2,056,500
PADD 3	28,182	4,227,300	94,458	16,981,500
PADD 4	8,862	1,329,300	23,478	5,746,500
PADD 5	5,334	800,100	115,416	37,098,000

Table D.15. Trucking Volumes and Costs by PADD.

Location	Volume Trucked (K gal)	Trucking Cost ($)
PADD 1	62,288	7,593,204
PADD 2	744,194	106,313,429
PADD 3	222,920	31,845,714
PADD 4	0	0
PADD 5	117,146	16,735,143

Table D.16. Overall Volumes and Costs by PADD.

Location	Total Volume (K gal)	PADD Total Transportation Cost ($)	PADD Total Transportation Cost (cents/gal)
PADD 1	156,284	21,789,204	13.94
PADD 2	807,782	116,100,029	14.37
PADD 3	345,560	53,054,514	15.35
PADD 4	32,340	7,075,800	21.88
PADD 5	237,896	54,633,243	22.97

D.2.4.4 Biodiesel State Mandates

In the No-RFS cases, individual state mandates are assumed to remain in place. The following table lists the states and mandate volumes incorporated in the model for 2020. Total fixed biodiesel demand based on 2016 volumes is about 42 K bbl/d, or 630 million gal/year.

Estimated 2020 state-mandated biodiesel/renewable biodiesel volume is 44.5 K bbl/d, with the increase driven by the following: (1) In New York, heating oil sales in Westchester, Nassau, and Suffolk counties were considered to be included starting in 2018 at a 5% biodiesel mandate; (2) Minnesota's summer biodiesel mandate in diesel sales increases to 20% in 2018; and (3) In California, the same percentage of biodiesel in on-road diesel fuel in 2016 was assumed for 2020 (about 10.6%), but volumes are slightly higher in 2020 due to the AEO-forecasted on-road demand growth.

State-mandated volumes are presented in Table D.17.

Table D.17. State-Mandated Volumes – Biodiesel.

Location	State	Volume (K bbl/d)	Volume (million gal/year)	Biodiesel Requirements
	Total	44.5	682	
PADD 1	New York	2.3	35	5% of heating oil demand in NYC, Westchester, Nassau, and Suffolk (Bill S 5422 signed by the Governor)
	Pennsylvania	2.0	31	2% minimum for ULSD
	Rhode Island	0.3	5	5% minimum on heating oil demand
PADD 2	Minnesota	6.7	103	Assume an average of 20% / 5% summer/winter minimums as of 2018 for ULSD (the planned increase to 20% for summer 2018 is reflected in modeling)
PADD 3	Louisiana	0.8	12	2% minimum for ULSD only in ozone attainment areas
	New Mexico	1.7	26	5% minimum for ULSD
PADD 5	California	27.7	425	Assumes the same % use as in 2016, and a small increase in the projected use of CARB diesel
	Oregon	2.03	31	5% minimum for ULSD (and a small increase in ULSD use)
	Washington	0.97	15	2% minimum for ULSD (and a small increase in ULSD use)

Note that several states have incentive programs that can allow diesel retailers to reduce state taxes on sales to end users (for example, Illinois and Texas). These could provide incentive to increase biodiesel sales, but may not provide blender or producer benefits unless there is an arrangement to share the tax savings. Because these are not specific volume mandates, they were not included in the mandate volume.

D.2.5 Study Case Setup

The Study case developed for this study represented a situation in which the RFS program no longer applies. The premise is that refiners/blenders still must supply the same volume of gasoline and diesel fuel (on an energy-adjusted basis) regardless of the level of ethanol or biodiesel blending. Absent the regulatory requirements of the RFS, continued use of ethanol and biodiesel/renewable diesel then would be driven by economics and state and local mandates. Assessing the economics of the use of renewable fuels requires integration of the supply functions for ethanol and biodiesel/renewable diesel developed by ICF into the PADD- level refinery models.

D.2.5.1 Ethanol

As shown earlier, the major source of ethanol production is the Midwest. Hence, the conceptual framework used in this study for the supply of ethanol is that the marginal source of ethanol for all PADDs is centered in the Midwest. The cost of ethanol blended with BOBs at terminals in various regions is then determined by (1) the incremental cost of ethanol production in the Midwest, (2) the cost of shipping ethanol from the Midwest via rail to transport hubs in regional markets, and (3) the cost of moving ethanol from the receiving transport hubs to terminals for blending with gasoline. Because the refinery models used in this study reflect the cost of producing hydrocarbon gasoline at the refinery gate (not its delivered cost to terminals), we adjust the delivered cost of ethanol by subtracting the estimated pipeline tariffs for gasoline to various regional markets. This puts the delivered cost of ethanol on an even footing with the delivered cost of hydrocarbon gasoline.

The derived ethanol supply curve, shown earlier and reproduced below (with ethanol costs converted to $/bbl) and the x-axis converted to K bbl/d (to agree with the metrics used in the refinery modeling), shows the initial Midwest plant-gate ethanol price ($60.90/bbl) used for the Study case. This cost is represented by the intersection of estimated ethanol production in 2020, assuming that the RFS is in place (the red line), with the ethanol supply curve (blue line).[30] The approach for the Study case is that, after running all of the PADD-level refinery models, if the volume of ethanol backed out at the specified delivered ethanol prices is sufficiently large, we would re-establish a new equilibrium plant-gate price consistent with the ethanol supply curve and a lower level of total ethanol production. We then would adjust the delivered ethanol prices in each of the PADD-level refinery models and re-run the cases.[31]

[30] There is a fair amount of uncertainty regarding the exact shape of the ethanol supply curve and the precise volume of ethanol production to be expected in 2020. In light of this, we set ethanol production (including exports) at about 980 K bbl/d, slightly less than our initial estimates of production volumes, so that it intersects the supply curve just before it turns steeply upward. The resulting equilibrium price with ethanol volumes reflecting the RFS being in place is very close to the average annual spot price reported in 2016 for Nebraska of a little more than $59/bbl.
[31] This could be accomplished through automatic iterations of the PADD-level refinery models if they were formally linked together mathematically. However, the MathPro refinery models are stand-alone models (not linked) and the approach is to make such price adjustments manually as called for. A last step in such model runs would be to adjust the volume of gasoline production so that it was equivalent, in terms of total energy supplied, to the amount implicitly specified in the Reference case.

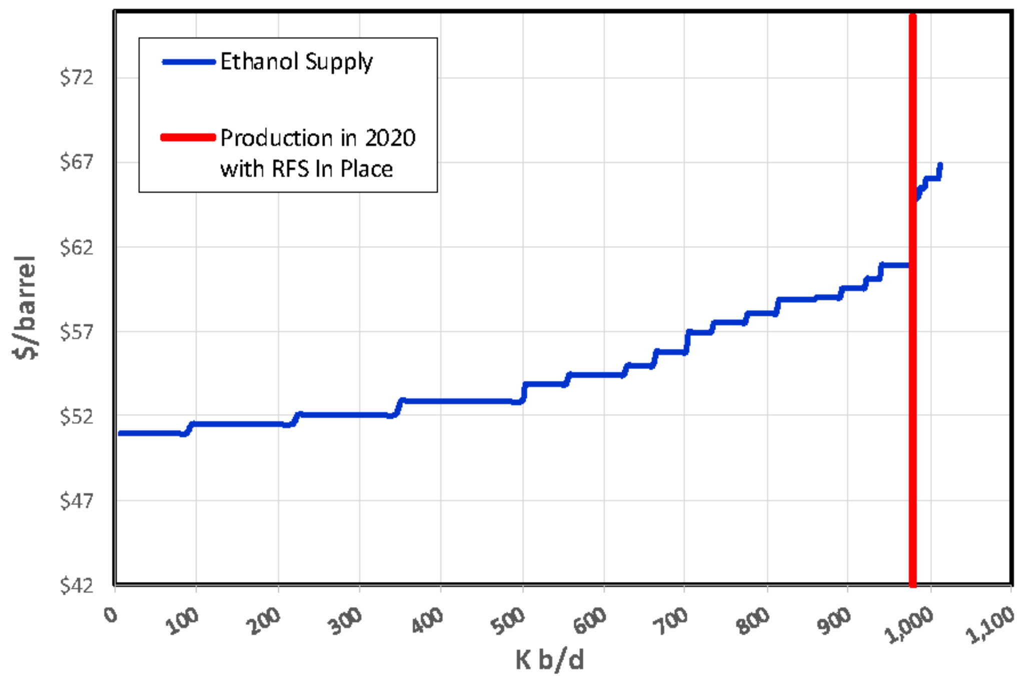

Figure D.3. Ethanol Supply Curve[32]. Note: excludes a small number of small plants with capacities of less than 25 M g/y.

Delivered prices of ethanol to terminals were calculated as the equilibrium plant-gate ethanol price in the Midwest (e.g., Nebraska) plus (1) the transport costs to Chicago, (2) the transport costs from Chicago to major regional hubs/terminals, and (3) the local transport costs of moving ethanol to terminals for final blending with gasoline (and a final adjustment for pipeline/barge tariffs for hydrocarbon gasoline). Estimated transport costs from Midwest ethanol plants to Chicago, rail costs from Chicago to regional markets, and local transport costs to blending terminals used in calculating delivered ethanol prices are shown in Table D.18. Those costs are fairly substantial, adding about $7 to $15/bbl to the price of ethanol.

[32] Note: 500 K bbl/d is about 7.65 billion gal/year; 1,000 K bbl/d is about 15.33 billion gal/year.

Table D.18. Estimated Distribution Costs for Ethanol.

Location		Distribution Cost to: Hub/Terminal (¢/g) To	Distribution Cost to: Hub/Terminal (¢/g) From	Distribution Cost to: Blending Terminal	Total	
PADD	Area	Chicago	Chicago	(¢/g)	(¢/g)	($/b)
PADD 1	Florida/Tampa		17.8	11.0	35.8	15.0
	Southeast/Atlanta		11.7	11.0	29.7	12.5
	VA/DC/MD		9.7	11.0	27.7	11.6
	Pittsburgh		6.2	11.0	24.2	10.2
	New York		7.7	11.0	25.7	10.8
PADD 2	Chicago	7.0	0.0	11.0	18.0	7.6
	Tennessee		9.7	11.0	27.7	11.6
PADD 3	Dallas		4.5	11.0	22.5	9.5
PADD 4			6.2	11.0	24.2	10.2
PADD 5	Los Angeles		16.4	9.0	32.4	13.6
	Arizona		16.4	9.0	32.4	13.6
	Nevada		12.4	9.0	28.4	11.9
	Northwest		12.4	9.0	28.4	11.9

In assessing the economics of ethanol blending, we want to compare the estimated delivered price of ethanol (plant-gate price plus distribution costs, as shown above) to the estimated delivered cost of hydrocarbon gasoline (refinery production cost plus pipeline/barge tariffs to the destination terminal). Estimates of pipeline/barge tariffs for gasoline are shown in Exhibit C1 in ICF Report Appendix C. Depending on the origin of the gasoline and its final destination, such charges range from as little as about $0.20/bbl to more than $4.00/bbl.

The refinery models are configured to estimate the refining cost of producing gasoline, not its delivered cost. Consequently, to put hydrocarbon gasoline and ethanol on an even footing with regard to distribution costs, we have *subtracted* estimated pipeline/barge tariffs for gasoline from estimated distribution costs for ethanol to develop estimates of the *net* distribution cost for ethanol. These calculations are shown in detail in Exhibits C2a and C2b. The exhibits are too lengthy to be included here. They show ethanol distribution costs, pipeline/barge tariffs for gasoline, and the net delivered cost of ethanol assigned to the various gasolines specified in the PADD-level refinery models, along with required finished gasoline volumes, maximum ethanol blending volumes, RVPs for finished gasoline, and ethanol's blending RVP. After these adjustments, net distribution costs for ethanol range from about $5/bbl to more than $13/bbl.

One last issue pertaining to the representation of ethanol in the refinery modeling is that certain states, as indicated in Table D.19 (repeated from Table D.9 earlier), have adopted mandates which require that ethanol be blended in gasoline at certain specified percentages or which impose minimum limits on ethanol blending. These mandates are incorporated in the refinery modeling as minimum constraints on the volume of ethanol blended in certain gasolines. For example, in PADD 2, Minnesota and Missouri have set minimum blending standards for ethanol at 10 vol%. We represent this in the refinery modeling by setting a minimum use of ethanol in PADD 2 of 38.9 K bbl/d, allocated to premium/regular grades of conventional gasoline produced in PADD 2. The refinery model can elect to use more ethanol in PADD 2 than the mandated volume, depending on blending economics (up to a specified maximum reflecting 10 vol% blending), but must use at least the minimum volume of ethanol specified. For PADD 5, all California RFG is required to be blended at 10 vol% due to the California Reformulated Gasoline Standards and the LCFS.

There are several states with incentive- or contingent-based mandates (e.g., Montana requires 10% ethanol in gasoline if in-state ethanol production exceeds 40 million gal). These were not included in the 2020 mandated volume.

Table D.19. State Mandates for Ethanol Use, 2020.

PADD	State	Minimum Volume (K bbl/d)	Ethanol Blending Requirements
Total		**156.2**	
PADD 2	Minnesota	17.7	10% minimum
	Missouri	21.2	10% minimum if ethanol priced lower than gasoline
PADD 3	Louisiana	1.6	2% minimum only in ozone attainment areas
PADD 5	California	102.9	10% minimum required by Predictive Model and LCFS
	Oregon	9.2	10% minimum, but only in regular gasoline (assume it affects the entire pool)
	Washington	3.7	2% minimum

Ethanol is represented in the refinery model as would be any purchased blendstock, such as reformate or alkylate, that could be blended directly in gasoline. The refinery model can buy any volume of ethanol at the specified price (subject to minimum and maximum constraints). If, at the specified price, the refinery model chooses to buy less ethanol, it must make up the lost volume by producing additional hydrocarbon gasoline. The incremental cost of producing BOBs increases as ethanol is backed out of the gasoline pool because (1) more hydrocarbon gasoline must be produced (increasing crude throughput and processing capacity utilization), and (2) the hydrocarbon gasoline must have higher octane to replace the octane lost when high-octane ethanol is removed from the gasoline pool. The refinery models reach an equilibrium when the incremental refining value of ethanol (which increases as the incremental cost of producing BOBs increases) equals the net delivered cost of ethanol

However, ethanol decidedly is not like other gasoline blendstocks:

- First, ethanol's RVP is set in the refinery model to reflect its effective blending RVP *when blended at 10 vol%*. If blended at a lower percentage, such as 5 vol%, its effective blending RVP would more than double. This is because the RVP delta (the increase in RVP above that of the BOB) is actually somewhat higher at 5 vol% blending than at 10 vol% blending, and the increase in RVP would be caused by half the volume of ethanol.
- Second, the array of regulatory standards affecting gasoline and the consequent design of the distribution system lead to a gasoline pool that is, for the most part, either E0 or E10 (with some E15, which is not assessed in this study, and small volumes of E85), but does not consist of gasoline blended with ethanol between 0 and 10 vol%. Thus, if the refinery model chooses to reduce the volume of ethanol blended below 10 vol% for certain gasoline types and grades, what is being produced would be a mix of E0 and E10, not a gasoline with an intermediate volume of ethanol.
- Because of the implications on the distribution system of reductions in ethanol use in the gasoline pool, it is by no means clear that a low, refinery-based valuation of ethanol for some gasoline types and grades, in practice, would lead to less ethanol use and more production of E0. For example, suppose that the refinery modeling suggested that some ethanol could be backed out of a premium grade. Doing so, by producing more E0 premium, might be of such high cost—for changes needed in the distribution system in order to segregate E10 and E0—that ethanol would continue to be blended at 10 vol% in all premium grades. Costs could be incurred all along the distribution system in terms of needed extra product segregations. At the station level, stations probably would have to carry either E0 or E10 premium, but not both, due to limited tankage, and those that carried E0 premium likely would have difficulty offering mid-grade (which generally is blended at the station) because a mid-grade blend of E0 premium and E10 regular would exceed applicable RVP standards. For these reasons, entire gasoline marketing regions would likely need to transition away from E10 to avoid these logistical complications.

The upshot is that the results of the refinery modeling should indicate where ethanol's continued use might be at some risk because of low refining valuations relative to its net delivered cost (hypothetically). However, a determination of whether ethanol use likely would decline would have to take other factors into account, primarily the implications for the distribution system and any associated costs to handle additional types and grades of E0 gasoline. It is possible that the motor vehicle and fuel industries could settle on an intermediate level of ethanol (e.g., E5).

However, potential examination of such a case was beyond the scope of this study because it would entail the introduction of a new motor fuel into the market. In addition, the No-RFS Study case found that blending E10 would be economical for refiners and blenders even without the RFS.

D.2.5.2 Biodiesel and Renewable Diesel

Unlike ethanol, which has most production centered in the Midwest and with limited imports, local biodiesel/renewable diesel production, along with regional imports, are capable of meeting regional demands in PADDs 1, 2, and 3. In PADDs 4 and 5, which rely on supply from other regions (primarily PADD 2), local production still accounts for a sizable portion of the supply. In view of this, biodiesel/renewable diesel supply is represented in the refinery modeling with PADD-level supply curves.[33]

The PADD-level supply curves are based on the regional supply curves developed by ICF and discussed earlier in this report. Exhibits C3a–e in ICF Report Appendix C delineate the estimated regional biodiesel supply curves for each region (excluding the supply of renewable diesel, but including potential supply from imports of biodiesel), along with estimated biodiesel/renewable diesel use in 2016, the latter to indicate the extent to which various PADDs must rely on production based in other PADDs or are capable of supplying other PADDs after meeting internal demands.

The next step in developing PADD-level supply curves was to compile for 2016 estimates of the annual supply of ULSD and biodiesel/renewable diesel by source for each PADD (or sub-region for PADD 1). Table D.20a–c show (1) the volume of ULSD used in each PADD, along with the source of the supply; (2) sources and dispositions of biodiesel/renewable diesel to the various PADDs; and (3) biodiesel production capacity by PADD (or sub-region). These data were developed from estimates of regional biodiesel/renewable diesel use (based on RIN data), regional biodiesel supply curves developed by ICF, production, EIA import and export data, and inter-PADD movement data reported by EIA adjusted for sub-PADD regions in PADD 1. The table shows the flows of biodiesel/renewable diesel between regions and provides a starting point for considering how supply to the various regions might be affected by the absence of the RFS.

The data in this table also were used to set biodiesel use in the Calibration and Reference cases refinery modeling. We allocated the biodiesel volumes in each of the consuming areas shown in Table D.20a–c to the various ULSD supply sources. For example, biodiesel use in the Northeast was

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[33] Note: A large portion of the inter-PADD biodiesel/renewable diesel movements are to PADD 5. The PADD 5 demand is driven by California's LCFS, which is treated as a mandated demand, and the study is not concerned about the economics of serving mandated areas.

allocated to ULSD supplied by PADDS 1, 2, and 3 and to imports; biodiesel use in PADD 2 was allocated to ULSD supplied by PADDs 2, 3, and 4 (imports were negligible); and so forth. This procedure results in biodiesel volumes for *producing* PADDs differing from that shown in Table D.17 for *consuming* PADDs. For instance, although we estimated that the Northeast used about 22 K bbl/d of biodiesel in 2016, we allocated only about 11 K bbl/d to ULSD produced by PADD 1 refineries (after accounting for exports).

Table D.20a. Estimated ULSD and Bio/Renewable Diesel Supply and Use, 2016 (K bbl/d): PADD 1.

Region/Source/ Destination	ULSD	Combined Bio/ Renewable Diesel Supply		Biodiesel Production Capacity	Region/Source/ Destination	ULSD	Combined Bio/ Renewable Diesel Supply		Biodiesel Production Capacity
PADD 1 Northeast					**PADD 1 Southeast**				
ULSD Supply	**486**				**ULSD Supply**	**538**			
Northeast Production	303				Southeast Production	0			
To PADD 1	250				From PADD 3	516			
Exports	53				Imports	22			
From PADD 2	24								
From PADD 3	150								
Imports	62								
Bio/Ren Diesel Supply		21.9			**Bio/Ren Diesel Supply**		**16.5**		
Northeast Production		6.3		10.1	Southeast Production		3.3		5.0
To Northeast			6.1		To Southeast			3.1	
Exports			0.2		To PADD 2			0.2	
From PADD 2		4.5			Exports			0.0	
Imports		11.3			From PADD 2		0		
					From PADD 3		0.3		
					Imports		13.1		

Table D.20b. Estimated ULSD and Bio/Renewable Diesel Supply and Use, 2016 (K bbl/d): PADD 2.

Region/Source/ Destination	ULSD	Combined Bio/ Renewable Diesel Supply		Biodiesel Production Capacity	Region/Source/ Destination	ULSD	Combined Bio/ Renewable Diesel Supply		Biodiesel Production Capacity
PADD 2					**PADD 3**				
ULSD Supply	1162				**ULSD Supply**	707			
PADD 2 Production	1017				PADD 3 Production	2460			
To PADD 1	24				To PADD 1	666			
To PADD 2	946				To PADD 2 directly	119			
To PADD 3	28				To PADD 2 via PADD 1	84			
To PADD 4	17				To PADD 3	678			
Exports	2				To PADD 4	0			
From PADD 3 directly	119				To PADD 5	25			
From PADD 3 via PADD 1	84				Exports	888			
From PADD 4	12				From PADD 2	28			
Imports	1				Imports	1			
Bio/Ren Diesel Supply		53.2			**Bio/Ren Diesel Supply**		41.3		
PADD 2 Production		71.4		76.5	PADD 3 Production		22.6		47.1
To PADD 1			4.5		To PADD 1			0.3	
To PADD 2			50.6		To PADD 2			0.6	
To PADD 3			6.4		To PADD 3			16.6	
To PADD 4			1.4		To PADD 4			0.0	
To PADD 5			5.2		To PADD 5			3.9	
Exports			3.3		Exports			1.2	
From PADD 1		0.2			From PADD 2		6.4		
From PADD 3		0.6			From PADD 5		0.3		
Imports		1.8			Imports		18.0		

Table D.20c. Estimated ULSD and Bio/Renewable Diesel Supply and Use, 2016 (K bbl/d): PADD 4 and U.S.

Region/Source/ Destination	ULSD	Combined Bio/ Renewable Diesel Supply		Biodiesel Production Capacity	Region/Source/ Destination	ULSD	Combined Bio/ Renewable Diesel Supply		Biodiesel Production Capacity
PADD 4					**PADD 5**				
ULSD Supply	190				**ULSD Supply**	496			
PADD 4 Production	196				PADD 5 Production	512			
To PADD 2	12				ULSD	253			
To PADD 4	173				To PADD 5	188			
To PADD 5	11				Exports	65			
Exports	0				CARB Diesel	259			
From PADD 2	17				From PADD 3 to Arizona	25			
Imports	0				From PADD 4 to Northwest	11			
					Imports				
					California	0			
					Northwest & Hawaii	13			
Bio/Ren Diesel Supply		2.7			**Bio/Ren Diesel Supply**		34.5		
PADD 4 Production		0.9		1.8	PADD 5 Production		10.6		18.1
To PADD 4			0.7		To PADD 3			0.3	
Exports			0.2		To PADD 4			0.1	
From PADD 2		1.4			To PADD 5			9.4	
From PADD 5		0.1			Exports			0.8	
Imports		0.4			From PADD 2		5.2		
					From PADD 3		3.9		
U.S.					Imports				
ULSD Supply	3579				California		15.4		
Bio/Ren Diesel Supply		170.1			Northwest & Hawaii		0.6		

Certain states[34] have imposed mandates for the use of biodiesel/renewable diesel in ULSD, as well as heating oil. Table D.21 shows estimated volumes of biodiesel/renewable diesel mandated by states for 2020 (this is repeated from Table D.17 shown earlier). These mandates are incorporated in the refinery models by setting lower limits on the volumetric use of biodiesel/renewable diesel.

Note that the mandated volumes do not include any biodiesel blending that may become economical due to state subsidies; for example, Illinois and Texas have reductions in the state diesel tax

[34] In some cases, the mandated biodiesel is for cities or counties.

that increase as the biodiesel percentage increases. Since these are retail-based incentives, they are not included in the mandate volume because it is unclear how they would directly affect producers/blenders.

Table D.21. State Mandates for Bio/Renewable Diesel Use, 2020.

PADD	State	Minimum Volume (K bbl/d)	Biodiesel Blending Requirements
Total		**44.5**	
PADD 1	New York	2.3	5% of heating oil in NYC, Westchester, Nassau, & Suffolk (Bill S 5422 signed by Governor)
	Pennsylvania	2.0	2% minimum for ULSD
	Rhode Island	0.3	5% minimum on heating oil
PADD 2	Minnesota	6.7	Assume average of 20% / 5% summer/winter minimums as of 2018 for ULSD
			(the summer standard increases from 10% to 20% in 2018)
PADD 3	Louisiana	0.8	2% minimum for ULSD only in ozone attainment areas
	New Mexico	1.7	5% minimum for ULSD
PADD 4			
PADD 5	California	27.7	Assumes same % use as in 2016 and a small increase in projected use of CARB diesel
	Oregon	2.03	5% minimum for ULSD (and a small increase in ULSD use)
	Washington	0.97	2% minimum for ULSD (and a small increase in ULSD use)

The PADD-level supply curves incorporated in the refinery models must include the distribution costs associated with transporting biodiesel/renewable to blending terminals or large stations that blend on site (adjusted for pipeline tariffs, as for ethanol). Table D.22 provides our estimates of those distribution costs. Distribution costs associated with local biodiesel production that supply in-region, local blending terminals are significantly lower than those associated with shipping biodiesel long distances by rail and then moving the material from transport hubs to local blending terminals. Such long-distance shipping occurs primarily for biodiesel supply originating in PADD 2 and shipped to the West Coast, and in PADD 3 and shipped to the Southeast.

Table D.22. Estimated Biodiesel/Renewable Diesel Distribution Costs for Use with Supply Curves in the Refinery Models.

PADD of Origin (Source)	Regional Destination	Estimated Volume in 2016 (K b/d)		Fraction of Hydrocarbon ULSD in 2016 (%)		Bio/Renewable Diesel Distribution Costs ($/b)						ULSD Distri-bution Costs ($/b)
						Inter-PADD Supply		Intra-PADD Supply	Imports			
		Mandated	Total1	Mandated	Total	Rail Transport to Hubs/ Terminals	Transport to Local Terminals	Transport from Biodiesel Plants to Local Terminals	Port Handling Charges	Transport to Local Terminals	Total	
PADD 1		4.6	38.4									
Northeast		4.6	21.9	0.9%	4.5%							
	Northeast		6.1					5.12			5.12	1.00
Southeast			16.5		3.1%							
	Southeast		3.1					5.12			5.12	1.50
	PADD 2		0.2									
PADD 2		6.7	53.2	0.6%	4.6%							
	PADD 1 (NE)		4.5			6.30	4.62				10.92	1.00
	PADD 2		50.6					6.00			6.00	1.50
	PADD 3		6.4			7.50	4.62				12.12	1.65
	PADD 4		1.4			10.50	4.62				15.12	0.50
	PADD 5		5.2									
	California		5.2			13.50	3.78				17.28	1.00
	Northwest		0.0			13.50	3.78				17.28	1.00

(continued)

Table D.22. Estimated Biodiesel/Renewable Diesel Distribution Costs for Use with Supply Curves in the Refinery Models. (continued)

PADD of Origin (Source)	Regional Destination	Estimated Volume in 2016 (K b/d)		Fraction of Hydrocarbon ULSD in 2016 (%)		Bio/Renewable Diesel Distribution Costs ($/b)						ULSD Distri-bution Costs ($/b)
						Inter-PADD Supply		Intra-PADD Supply	Imports			
		Mandated	Total1	Mandated	Total	Rail Transport to Hubs/ Terminals	Transport to Local Terminals	Transport from Biodiesel Plants to Local Terminals	Port Handling Charges	Transport to Local Terminals	Total	
PADD 3		2.5	41.3	0.4%	5.8%							
	PADD 1 (SE)		0.3			7.50	4.62				12.12	1.50
	PADD 2											
	Direct		0.6			7.50	4.62				12.12	1.50
	Via PADD 1		-									
	PADD 3		16.6					6.00			6.00	1.65
	PADD 5		3.9									
	California		-			13.50	3.78				17.28	1.00
	Ariz & Nev		3.9			10.50	3.78				14.28	1.00
PADD 4			2.7		1.4%							
	PADD 4		0.7					6.72			6.72	0.50
PADD 5		29.6	34.5	6.0%	7.0%							
California		26.7	26.7	10.3%	10.3%							
	California		6.1					6.00			6.00	1.00
	PADD 3		0.3	-	-			-			-	-
All Other		2.9	7.9	1.2%	3.3%							
	All Other		3.3					6.00			6.00	1.00
	PADD 4		0.1	-	-			-			-	-

(continued)

Table D.22. Estimated Biodiesel/Renewable Diesel Distribution Costs for Use with Supply Curves in the Refinery Models. (continued)

PADD of Origin (Source)	Regional Destination	Estimated Volume in 2016 (K b/d)		Fraction of Hydrocarbon ULSD in 2016 (%)		Bio/Renewable Diesel Distribution Costs ($/b)						ULSD Distri-bution Costs ($/b)
						Inter-PADD Supply		Intra-PADD Supply	Imports			
		Mandated	Total1	Mandated	Total	Rail Transport to Hubs/ Terminals	Transport to Local Terminals	Transport from Biodiesel Plants to Local Terminals	Port Handling Charges	Transport to Local Terminals	Total	
Imports			60.2									
	PADD 1		24.4									
	Northeast		11.3						1.00	4.62	5.62	1.00
	Southeast		13.1						1.00	4.62	5.62	1.50
	PADD 2		1.8						1.00	4.62	5.62	1.50
	PADD 3		18.0						1.00	4.62	5.62	1.65
	PADD 4		0.4			7.50				4.62	12.12	0.50
	PADD 5		16.0									
	California		15.4						1.00	3.78	4.78	1.00
	Northwest		0.6						1.00	3.78	4.78	1.00

[1] Detail does not add up to the Regional totals, as the former reflects local supply and the latter reflects total use, including inter-PADD shipments and imports.

The distribution cost estimates were used in combination with the regional supply curves shown in ICF Report Appendix C, estimates of biodiesel supply and disposition for 2016 (Table D.20a–c, above), and estimates of mandated use of biodiesel for 2020 (Table D.21 above) to develop supply functions for the PADD-level refinery models under the assumption that the RFS no longer is in force. The logic used in developing such supply functions is as follows:

- For the Northeast, PADD 2, and PADD 3, the lowest cost biodiesel locally produced (primarily from yellow grease) remains in the region and is used to fully meet the regional biodiesel mandate volumes.
- For PADD 5, all mandate volumes for California and the Pacific Northwest are met using the lowest cost sources (in terms of delivered cost) of biodiesel and renewable diesel from (1) local supply, (2) imports, and (3) inter-PADD shipments of biodiesel to California and the Northwest from PADDs 2 and 3 not already used to meet mandated volumes in those PADDs. The rationale for including biodiesel from PADDs 2 and 3 in the supply function for PADD 5 is that (1) inter-PADD movement data reported by EIA indicate that such shipments occurred in 2016, and (2) the mandates in PADD 5, especially in California, would allow blenders to bid biodiesel away from PADDs 2 and 3 due to its significantly higher, regulatory-induced value on the West Coast. Biodiesel would be worth less to blenders in PADDs 2 and 3 because, if the RFS program was no longer in effect, they would not receive compensation from RIN-generations.
- Biodiesel use in PADDs 2 and 3 up to the volume occurring in 2016 with the RFS in place (after mandates were satisfied) would be met with the remaining indigenous supply and imports.
- Biodiesel supply to PADD 4 would come from indigenous production and the lowest delivered cost sources remaining in PADD 2 (up to the volume of use attained in 2016).
- Remaining use of biodiesel (after mandates) in the southeast and northeast parts of PADD 1 first are met with indigenous supply and imports (ordered by delivered cost) and then are supplemented with supply from PADD 3 to the Southeast (based on whatever supply remains after PADD 3 demands are met and movements to PADD 5 are considered) and to the Northeast from PADD 2 (after accounting for previous allocations of PADD 2 production). Local PADD 1 biodiesel production (after accounting for mandate volumes) was allocated to the Southeast and Northeast on a 1:2 basis.
- Because PADD 3 accounts for large volumes of inter-PADD ULSD shipments, four separate biodiesel supply functions are incorporated in the PADD 3 refinery model, each representing

biodiesel supply in the final region of destination of the ULSD—PADD 3, Southeast, Northeast, and PADD 2. For the regions outside of PADD 3: (1) residual (high-cost) biodiesel supply in PADD 2 was allocated to ULSD projected to be produced by PADD 3 refineries and shipped to PADD 2; (2) biodiesel production in the Southeast, plus imports to the Southeast, were allocated to ULSD produced by PADD 3 refineries and shipped to the Southeast (imports of ULSD were small enough to ignore); and (3) biodiesel supply in the Northeast was allocated to ULSD produced by PADD 1 refineries and ULSD shipped to the Northeast from PADD 3 on a proportional basis. Other inter-PADD shipments of ULSD were small enough to ignore in terms of allocating biodiesel supply.

Table D.23 shows the biodiesel supply functions in the refining models. Minimum volumes were established to ensure that mandate volumes were met. Maximum volumes were set at each of the price levels shown so that increasing use of biodiesel required accessing higher cost sources of biodiesel. The refining models chose to blend biodiesel in ULSD as long as the net delivered cost of biodiesel was equal to or less than the refining value of biodiesel (which is equal to the incremental cost of producing ULSD).

Table D.23. Estimated Biodiesel/Renewable Diesel Supply Functions for Refinery Modeling, 2020.

Category	Bio/Renewable Diesel					ULSD Distribution Cost ($/b)	Net Bio/Ren Cost		Source of Biodiesel/ Renewable Diesel
	Volume (K b/d)	Cumulative Volume (K b/d)	Production Cost ($/b)	Distribution Cost ($/b)	Total Cost ($/b)		Without Subsidy ($/b)	With Subsidy ($/b)	
PADD 1									
Total Supply	0.91	0.91	71.83	5.12	76.95				PADD 1
	0.44	1.35	82.16	5.12	87.28				PADD 1
	0.16	1.51	88.64	5.12	93.76				PADD 1
	2.28	3.80	117.09	5.12	122.21				PADD 1
	5.94	9.73	121.30	5.12	126.42				PADD 1
	0.55	10.29	122.96	5.12	128.08				PADD 1
	25.86	36.15	126.38	5.62	132.00				Imports
	0.4	36.55	121.30	12.12	133.42				PADD 3
	4.5	41.05	124.01	10.92	134.93				PADD 2
Northeast	0.91	0.91	71.83	5.12	76.95				PADD 1 (NE)
4.6	0.44	1.35	82.16	5.12	87.28				PADD 1 (NE)
(mandated)	0.16	1.51	88.64	5.12	93.76				PADD 1 (NE)
	1.53	3.04	117.09	5.12	122.21				PADD 1 (NE)
	3.98	7.02	121.30	5.12	126.42				PADD 1 (NE)
	0.37	7.39	122.96	5.12	128.08				PADD 1 (NE)
	11.90	19.29	126.38	5.62	132.00				Imports (NE)
	0.00	19.29	121.30	12.12	133.42				PADD 3
	4.50	23.79	124.01	10.92	134.93				PADD 2

(continued)

Table D.23. Estimated Biodiesel/Renewable Diesel Supply Functions for Refinery Modeling, 2020. (continued)

Category	Bio/Renewable Diesel					ULSD Distribution Cost ($/b)	Net Bio/Ren Cost		Source of Biodiesel/ Renewable Diesel
	Volume (K b/d)	Cumulative Volume (K b/d)	Production Cost ($/b)	Distribution Cost ($/b)	Total Cost ($/b)		Without Subsidy ($/b)	With Subsidy ($/b)	
Southeast	0.75	0.75	117.09	5.12	122.21	1.50	120.71	78.71	PADD 1 (SE)
	1.96	2.71	121.30	5.12	126.42	1.50	124.92	82.92	PADD 1 (SE)
	0.18	2.89	122.96	5.12	128.08	1.50	126.58	84.58	PADD 1 (SE)
	13.97	16.86	126.38	5.62	132.00	1.50	130.50	88.50	Imports (SE)
	0.40	17.26	121.30	12.12	133.42	1.50	131.92	89.92	PADD 3
	0.00	17.26	124.01	10.92	134.93	1.50	133.43	91.43	PADD 2
Northeast: P1 ref.									
2.9	0.57	0.57	71.83	5.12	76.95	1.00	75.95	33.95	PADD 1 (NE) 62.5%
(mandated)	0.27	0.84	82.16	5.12	87.28	1.00	86.28	44.28	PADD 1 (NE) 62.5%
	0.10	0.95	88.64	5.12	93.76	1.00	92.76	50.76	PADD 1 (NE) 62.5%
62.5%	0.96	1.90	117.09	5.12	122.21	1.00	121.21	79.21	PADD 1 (NE) 62.5%
(allocation)	2.49	4.39	121.30	5.12	126.42	1.00	125.42	83.42	PADD 1 (NE) 62.5%
	0.23	4.62	122.96	5.12	128.08	1.00	127.08	85.08	PADD 1 (NE) 62.5%
	7.44	12.06	126.38	5.62	132.00	1.00	131.00	89.00	Imports (NE) 62.5%
	0.00	12.06	121.30	12.12	133.42	1.00	132.42	90.42	PADD 3
	2.81	14.87	124.01	10.92	134.93	1.00	133.93	91.93	PADD 2 62.5%
Northeast: P3 ref.									
1.7	0.34	0.34	71.83	5.12	76.95	2.00	74.95	32.95	PADD 1 (NE) 37.5%
(mandated)	0.16	0.51	82.16	5.12	87.28	2.00	85.28	43.28	PADD 1 (NE) 37.5%
	0.06	0.57	88.64	5.12	93.76	2.00	91.76	49.76	PADD 1 (NE) 37.5%

(continued)

Table D.23. Estimated Biodiesel/Renewable Diesel Supply Functions for Refinery Modeling, 2020. (continued)

Category	Bio/Renewable Diesel					ULSD Distribution Cost ($/b)	Net Bio/Ren Cost		Source of Biodiesel/ Renewable Diesel
	Volume (K b/d)	Cumulative Volume (K b/d)	Production Cost ($/b)	Distribution Cost ($/b)	Total Cost ($/b)		Without Subsidy ($/b)	With Subsidy ($/b)	
37.5%	0.57	1.14	117.09	5.12	122.21	2.00	120.21	78.21	PADD 1 (NE) 37.5%
(allocation)	1.49	2.63	121.30	5.12	126.42	2.00	124.42	82.42	PADD 1 (NE) 37.5%
	0.14	2.77	122.96	5.12	128.08	2.00	126.08	84.08	PADD 1 (NE) 37.5%
	4.46	7.23	126.38	5.62	132.00	2.00	130.00	88.00	Imports (NE) 37.5%
	0.00	7.23	121.30	12.12	133.42	2.00	131.42	89.42	PADD 3
	1.69	8.92	124.01	10.92	134.93	2.00	132.93	90.93	PADD 2 37.5%
PADD 2									
P2 Refineries	3.96	3.96	71.83	6.00	77.83	1.50	76.33	34.33	PADD 2
6.7	2.84	6.80	73.91	6.00	79.91	1.50	78.41	36.41	PADD 2
(mandated)	9.90	16.70	117.09	6.00	123.09	1.50	121.59	79.59	PADD 2
	7.09	23.79	119.75	6.00	125.75	1.50	124.25	82.25	PADD 2
	20.21	44.00	121.30	6.00	127.30	1.50	125.80	83.80	PADD 2
P3 Refineries	5.53	5.53	121.30	6.00	127.30	1.30	126.00	84.00	PADD 2
	0.67	6.20	122.96	6.00	128.96	1.30	127.66	85.66	PADD 2
	12.61	18.81	124.01	6.00	130.01	1.30	128.71	86.71	PADD 2
PADD 3									
2.5	3.11	3.11	71.83	6.00	77.83	1.65	76.18	34.18	PADD 3
(mandated)	7.78	10.89	117.09	6.00	123.09	1.65	121.44	79.44	PADD 3
	1.88	12.77	119.75	6.00	125.75	1.65	124.10	82.10	PADD 3
	20.23	32.99	121.30	6.00	127.30	1.65	125.65	83.65	PADD 3

(continued)

Table D.23. Estimated Biodiesel/Renewable Diesel Supply Functions for Refinery Modeling, 2020. (continued)

Category	Bio/Renewable Diesel					ULSD Distribution Cost ($/b)	Net Bio/Ren Cost		Source of Biodiesel/ Renewable Diesel
	Volume (K b/d)	Cumulative Volume (K b/d)	Production Cost ($/b)	Distribution Cost ($/b)	Total Cost ($/b)		Without Subsidy ($/b)	With Subsidy ($/b)	
	1.44	34.43	122.96	6.00	128.96	1.65	127.31	85.31	PADD 3
	4.88	39.30	124.01	6.00	130.01	1.65	128.36	86.36	PADD 3
	10.03	49.33	126.38	6.00	132.38	1.65	130.73	88.73	PADD 3
PADD 4									
	0.14	0.14	76.54	6.72	83.26	0.50	82.76	40.76	PADD 4
	0.03	0.18	88.64	6.72	95.36	0.50	94.86	52.86	PADD 4
	0.36	0.54	122.96	6.72	129.68	0.50	129.18	87.18	PADD 4
	0.93	1.47	127.28	6.72	134.00	0.50	133.50	91.50	PADD 4
	1.33	2.80	124.01	15.12	139.13	0.50	138.63	96.63	PADD 2
PADD 5									
30.7	0.65	0.65	71.83	6.00	77.83	1.00	76.83	34.83	PADD 5
(mandated)	0.26	0.91	73.91	6.00	79.91	1.00	78.91	36.91	PADD 5
	0.53	1.44	76.54	6.00	82.54	1.00	81.54	39.54	PADD 5
	0.25	1.69	82.16	6.00	88.16	1.00	87.16	45.16	PADD 5
	0.75	2.44	73.91	17.28	91.19	1.00	90.19	48.19	PADD 3
	0.57	3.02	76.54	17.28	93.82	1.00	92.82	50.82	PADD 3
	0.37	3.39	76.54	17.28	93.82	1.00	92.82	50.82	PADD 2
	0.11	3.50	88.64	6.00	94.64	1.00	93.64	51.64	PADD 5
	0.21	3.71	82.16	17.28	99.44	1.00	98.44	56.44	PADD 3
	0.29	3.99	82.16	17.28	99.44	1.00	98.44	56.44	PADD 2
	0.07	4.06	88.64	17.28	105.92	1.00	104.92	62.92	PADD 3

(continued)

Table D.23. Estimated Biodiesel/Renewable Diesel Supply Functions for Refinery Modeling, 2020. (continued)

Category	Bio/Renewable Diesel					ULSD Distribution Cost ($/b)	Net Bio/Ren Cost		Source of Biodiesel/ Renewable Diesel
	Volume (K b/d)	Cumulative Volume (K b/d)	Production Cost ($/b)	Distribution Cost ($/b)	Total Cost ($/b)		Without Subsidy ($/b)	With Subsidy ($/b)	
	0.20	4.26	88.64	17.28	105.92	1.00	104.92	62.92	PADD 2
	1.63	5.89	117.09	6.00	123.09	1.00	122.09	80.09	PADD 5
	0.65	6.54	119.75	6.00	125.75	1.00	124.75	82.75	PADD 5
	4.24	10.78	121.30	6.00	127.30	1.00	126.30	84.30	PADD 5
	1.32	12.10	122.96	6.00	128.96	1.00	127.96	85.96	PADD 5
	1.70	13.80	124.01	6.00	130.01	1.00	129.01	87.01	PADD 5
	15.96	29.76	126.38	4.78	131.16	1.00	130.16	88.16	Imports
	3.43	33.20	127.28	6.00	133.28	1.00	132.28	90.28	PADD 5
	3.90	37.10	126.38	14.28	140.66	1.00	139.66	97.66	PADD 3

Note: Above estimates indicate the supply of bio/renewable diesel available for blending with ULSD produced within the various PADDs in the absence of the RFS program. PADD 3 has four biodiesel supply curves:
for ULSD sold within PADD 3
for ULSD shipped to the Southeast
for ULSD shipped to the Northeast, and
for ULSD shipped to PADD 2.
Other inter-PADD shipments of ULSD are small and, for simplicity, are ignored when establishing supply curves.

D.3 Study Results

D.3.1 Calibration Case

A summary of selected results from the 2016 Calibration case modeling for each PADD, along with reported data for the various measures, is shown in Table D.24 (more detailed results are shown for the Calibration and other cases in ICF Report Appendix D). Three general types of measures are included in the table: volume, price/marginal cost, and property measures. The volume measures include crude oil inputs, charge rates to major downstream processes, coke production, and energy use. Price/marginal cost measures include spot prices for conventional regular gasoline, RBOB, and distillates, and the incremental refining costs returned by the refinery model for finished gasoline and distillates (the Crude Acquisition costs shown were specified as the price for composite crudes in the various refinery modeling cases). Property measures are for the finished gasoline pools produced in each PADD. However, Reported properties provided by EPA include imported gasoline and Calibration properties include exported gasoline.

In general, the Reported data and Calibration results for volumes are reasonably close. Crude oil throughputs match up well, as do charge rates for downstream processes in most of the PADDs. Energy use, by type, does not line up as closely. On the whole, the PADD-level refinery models tend to use somewhat less energy than is reported. Part of this reflects lower power use in the refinery models, partly because the refinery models only account for refinery process-related power use. Looking only at energy use from natural gas, still gas, and catalyst coke, the PADD-level models, on average, use about 5% less energy than reported.

Gasoline properties also match up well, albeit with some exceptions. Some properties are specified (as maximums) in the refinery models, such as RVP, benzene, and sulfur, whereas values for others, such as aromatics, olefins, E200, and E300, are returned by the refinery models and reflect the properties of the mix of blendstocks forming the various types and grades of gasoline. Regional octane levels in terms of (R + M)/2 for the various gasoline types and grades are specified as lower limits in the refinery models (based on the Alliance 2015 North American Fuel Survey data). However, the RON and MON (Research and Motor Octane Number) values are returned by the refinery models, again reflecting the octanes of the mix of gasoline blendstocks. The calculated octane sensitivities of the gasoline pool (RON minus MON), averaging about 9 for the summer and 8.5 for the winter, are consistent with gasoline pool octane sensitivities derived from the Alliance 2015 North American Fuel Surveys (about 8.7 for the summer and 8.2 for the winter).

Table D.24. Selected Calibration Modeling Results, 2016.

Measures	PADD 1				PADD 2				PADD 3			
	Summer		Winter		Summer		Winter		Summer		Winter	
	Reported	Calib	Reported	Calib	Reported	Calib	Reported	Calib	Reported	Calib	Reported	Calib
Crude Inputs (K b/d)	1,122	1,104	1,022	1,006	3,621	3,673	3,670	3,701	8,589	8,626	8,456	8,347
Operations												
Charge Rates (K/b/d)												
Reforming	189	188	173	134	662	669	639	617	1,468	1,485	1,369	1,364
Fluid Cat Cracking	407	388	394	397	1,115	1,172	1,080	1,094	2,616	2,643	2,486	2,381
Hydrocracking	35	36	35	36	273	293	298	293	1,047	970	919	970
Coking	59	57	57	69	471	485	477	503	1,383	1,352	1,328	1,245
Operating Indices												
FCC Conversion		66.7		66.9		69.3		73.2		70.7		71.3
Reformer Severity		94.8		92.6		95.6		92.6		95.5		95.5
Coke Make (K b/d)												
Marketable	14	14	15	19	140	166	143	174	393	419	385	382
Catalyst	23	20	22	20	49	59	46	53	122	131	115	116
Prices/Marginal Cost ($/b)												
Crude Acquisition Cost	44.98		50.97		41.97		46.92		42.23		48.17	
Gasoline[1]												
RFG: Premium		61.5		60.9		58.1		58.1		48.8		55.4
Regular		58.9		59.4		56.2		56.6		47.2		54.2

(continued)

Table D.24. Selected Calibration Modeling Results, 2016. (continued)

	PADD 1				PADD 2				PADD 3			
	Summer		Winter		Summer		Winter		Summer		Winter	
Measures	Reported	Calib	Reported	Calib	Reported	Calib	Reported	Calib	Reported	Calib	Reported	Calib
Conventional: Premium		59.8		60.9		57.1		57.7		48.2		55.5
Regular	57.22	57.6	61.53	59.4		55.2		56.4	55.45	46.6	59.82	54.3
RBOB												
Distillates												
Jet Fuel		56.8		63.7		50.6		56.4	54.00	47.5	61.67	56.6
ULSD/CARB Dsl[1]	54.82	56.7	62.66	63.7		50.5		56.2	53.49	47.4	61.48	56.4
Finished Gasoline												
Pool Properties[2,3]	**1291**	**621**		**615**	**2088**	**2161**		**2313**	**3885**	**4537**		**4493**
RVP (psi)	7.8	7.9		13.7	8.9	9.1		14.6	8.8	8.8		13.1
Oxygen (wt%)		3.3		3.3		3.4		3.4		2.9		2.8
Aromatics (vol%)	17.9	20.4		16.4	19.3	18.5		15.3	19.3	19.0		16.5
Benzene (vol%)	0.61	0.54		0.54	0.57	0.54		0.54	0.54	0.54		0.48
Olefins (vol%)	10.2	9.0		10.0	6.3	7.5		7.8	9.7	8.0		7.7
Sulfur (ppm)	25	19		20	19	17		16	25	18		16
E200 (vol% off)	54.9	46.3		53.7	54.9	52.3		57.7	54.2	50.8		54.7
E300 (vol% off)	87.7	81.3		84.0	85.5	83.2		85.0	85.3	82.5		85.2
Octane												
(R+M)/2		88.0		87.9		88.0		88.0		87.8		87.8
MON		83.2		83.4		83.4		83.7		83.3		83.5
RON		92.7		92.5		92.5		92.3		92.3		92.0
Sensitivity		9.5		9.1		9.1		8.6		9.0		8.5

(continued)

Table D.24. Selected Calibration Modeling Results, 2016. (continued)

Measures	PADD 1				PADD 2				PADD 3			
	Summer		Winter		Summer		Winter		Summer		Winter	
	Reported	Calib	Reported	Calib	Reported	Calib	Reported	Calib	Reported	Calib	Reported	Calib
Energy Use (B btu/d)[4]	**637**	**548**		**502**	**2109**	**1990**		**1928**	**5269**	**4960**		**4690**
Natural Gas (K foeb/d)	22	28		24	89	91		94	267	232		237
Still Gas (K foeb/d)	41	29		27	139	126		119	299	321		295
Catalyst Coke (K b/d)	23	20		20	49	59		53	121	131		116
Power (MM Kwh/d)	11	6.9		6.0	38	27.1		26.3	97	68.3		64.4

Reported prices for gasoline and ULSD/CARB are adjusted downward by the estimated cost of a RIN bundle (Summer $3.62/b; Winter $3.41/b).

Reported properties from EPA -- ethanol adjusted, annual average (except RVP is for Summer) including imports, and PADD 5 excludes California.

Calibration results include exports (substantial in PADD 3).

Reported data are for annual operations.

Table D.24. Selected Calibration Modeling Results, 2016. (continued)

Measures	PADD 4				PADD 5				U.S.			
	Summer		Winter		Summer		Winter		Summer		Winter	
	Reported	Calib	Reported	Calib	Reported	Calib	Reported	Calib	Reported	Calib	Reported	Calib
Crude Inputs (K b/d)	**604**	**593**	**605**	**597**	**2,439**	**2,501**	**2,235**	**2,207**	**16,375**	**16,497**	**15,988**	**15,859**
Operations												
Charge Rates (K/b/d)												
Reforming	101	98	92	94	440	445	390	415	2,860	2,885	2,661	2,624
Fluid Cat Cracking	163	168	164	171	731	774	678	678	5,032	5,144	4,803	4,721
Hydrocracking	24	24	21	24	487	504	452	542	1,864	1,826	1,725	1,865
Coking	66	62	72	65	465	475	435	389	2,444	2,431	2,368	2,270
Operating Indices												
FCC Conversion		67.6		68.7		70.9		71.6		70.0		71.3
Reformer Severity		91.5		90.4		97.8		90.9		95.7		93.7
Coke Make (K b/d)												
Marketable	17	19	19	20	120	136	110	107	684	754	673	703
Catalyst	8	8	8	8	35	36	33	31	237	254	225	228
Prices/Marginal Cost ($/b)												
Crude Acquisition Cost	40.01		45.15		43.95		50.20					
Gasoline[1]												
RFG: Premium						64.8		62.5				
Regular						61.4		61.6				
Conventional: Premium		50.0		54.3		56.1		62.2				
Regular		48.7		53.1		52.2		60.9				
RBOB					60.55		62.76					

Table D.24. Selected Calibration Modeling Results, 2016. (continued)

Measures	PADD 4				PADD 5				U.S.			
	Summer		Winter		Summer		Winter		Summer		Winter	
	Reported	Calib	Reported	Calib	Reported	Calib	Reported	Calib	Reported	Calib	Reported	Calib
Distillates												
Jet Fuel		48.6		54.0		60.9		63.7				
ULSD/CARB Dsl[1]		48.5		53.9	56.10	57.8	65.22	63.3				
Finished Gasoline												
Pool Properties[2,3]	**334**	**341**		**348**	**510**	**1492**		**1452**	**8107**	**9,152**		**9,221**
RVP (psi)	8.9	9.5		13.8	8.7	7.8		12.7	8.7	8.7		13.5
Oxygen (wt%)		3.4		3.4		3.2		3.2		3.1		3.1
Aromatics (vol%)	19.8	17.8		16.7	21.6	21.4		16.2	19.2	19.3		16.1
Benzene (vol%)	0.80	0.54		0.54	0.61	0.68		0.70	0.57	0.56		0.54
Olefins (vol%)	8.7	8.7		9.5	6.7	5.7		5.4	8.6	7.6		7.6
Sulfur (ppm)	21	17		17	18	5		7	23	16		15
E200 (vol% off)	54.0	50.8		57.8	53.3	50.8		58.2	54.4	50.8		56.0
E300 (vol% off)	87.8	83.8		85.0	87.3	84.4		88.0	85.9	83.0		85.5
Octane												
(R+M)/2		87.6		87.5		88.2		88.2		87.9		87.9
MON		83.2		83.1		83.7		84.1		83.4		83.7
RON		92.0		92.0		92.7		92.3		92.4		92.2
Sensitivity		8.8		8.8		9.0		8.2		9.0		8.5

(continued)

Table D.24. Selected Calibration Modeling Results, 2016. (continued)

Measures	PADD 4				PADD 5				U.S.			
	Summer		Winter		Summer		Winter		Summer		Winter	
	Reported	Calib	Reported	Calib	Reported	Calib	Reported	Calib	Reported	Calib	Reported	Calib
Energy Use (B btu/d)[4]	**368**	**282**		**282**	**1808**	**1628**		**1465**	**10191**	**9408**		**8867**
Natural Gas (K foeb/d)	18	18		17	91	67		74	487	436		445
Still Gas (K foeb/d)	22	14		14	123	114		90	624	604		546
Catalyst Coke (K b/d)	8	8		8	33	36		31	235	254		228
Power (MM Kwh/d)	6	3.6		3.5	26	27.0		24.7	178	132.9		124.9

1. Reported prices for gasoline and ULSD/CARB are adjusted downward by the estimated cost of a RIN bundle.
2. Reported properties from EPA -- ethanol adjusted, annual average (except RVP is for Summer) including imports, and PADD 5 excludes California.
3. Calibration results include exports (substantial in PADD 3).
4. Reported data are for annual operations.

The degree to which incremental costs for gasolines and distillates align with reported spot prices, however, is a more complicated and different story. In the recent past, refining margins (as measured by the 3-2-1 crack spread) have varied considerably, the market value of RINs has increased, and premium/regular price deltas in spot and wholesale markets have fluctuated over a wide range. Figure D.4 and Figure D.5 show trends in the crack spread for PADD 3, trends in the premium/regular wholesale price deltas for PADD 3, and bulk price deltas for the United States.

Variations in crack spreads and in premium/regular price deltas reflect the interaction between market demand for refined products and the refining sector's capability of expanding the output of refined products and producing additional octane. Closely matching observed spot market prices and premium/regular price deltas (as calculated using spot prices) for some specified period generally requires iteratively tightening (or loosening) constraints on selected refining process capacity so that the refinery model is cable of producing the required product volumes at the target incremental production costs (determined by spot market prices). The drawback of this sometimes useful approach is that publicly available information on the U.S. refinery sector usually is not sufficient to establish the degree to which the various refining centers may be process capacity constrained, or whether such constraints may persist or be moderated over time. When refining crack spreads are large or premium/regular price differentials are wide, matching those targets by constraining refining processes in the regional refining models can result in the production costs for refined products returned by the models being highly sensitive to small changes in required output volumes. This is an inherent difficulty with aggregated models in representing refining centers comprised of multiple refineries, each with their own unique combinations of process units and processing capabilities.

However, we moved partially in this direction by imposing capacity constraints on crude distillation, conversion processes, and octane-making processes to improve the calibration results in terms of refining cost and throughput.

- For atmospheric crude distillation capacity, we set capacity limits just below the volume of crude throughput required to produce the desired product slate, thereby forcing the models to add atmospheric distillation capacity. This increased the incremental cost of producing all refined products and brought refining margins closer in line with margins observed during the 2016 calibration period (crude oil processing capacity may be constrained due to relatively high capacity utilization rates and the recent increase in the domestic supply of light crude oils, which are problematic for many U.S. refineries configured to process heavier crude oils).
- For the major conversion processes, we limited capacity utilization to (1) 90% for fluid catalytic cracking in PADDs 1–4, and (2) 85% in PADD 1 and 90% in PADD 3 for hydrocracking. These adjustments brought throughput rates closer to those reported by EIA.

- For the major upgrading (octane-producing) processes, such as alkylation, pen-hex isomerization, and reforming, we limited capacity utilization to 80% to 90%, varying by region and process. These adjustments served to constrain the refining sectors' octane-production capabilities and to increase the premium/regular cost spread, but not by as much as was indicated by reported premium/regular spot price spreads.

These constraints moved the production costs for refined products returned by the refinery models in the right direction. However, the constraints on conversion and upgrading processes were not so tight as to call for investments in new process capacity. The subsequent Reference and Study cases also employ these constraints.

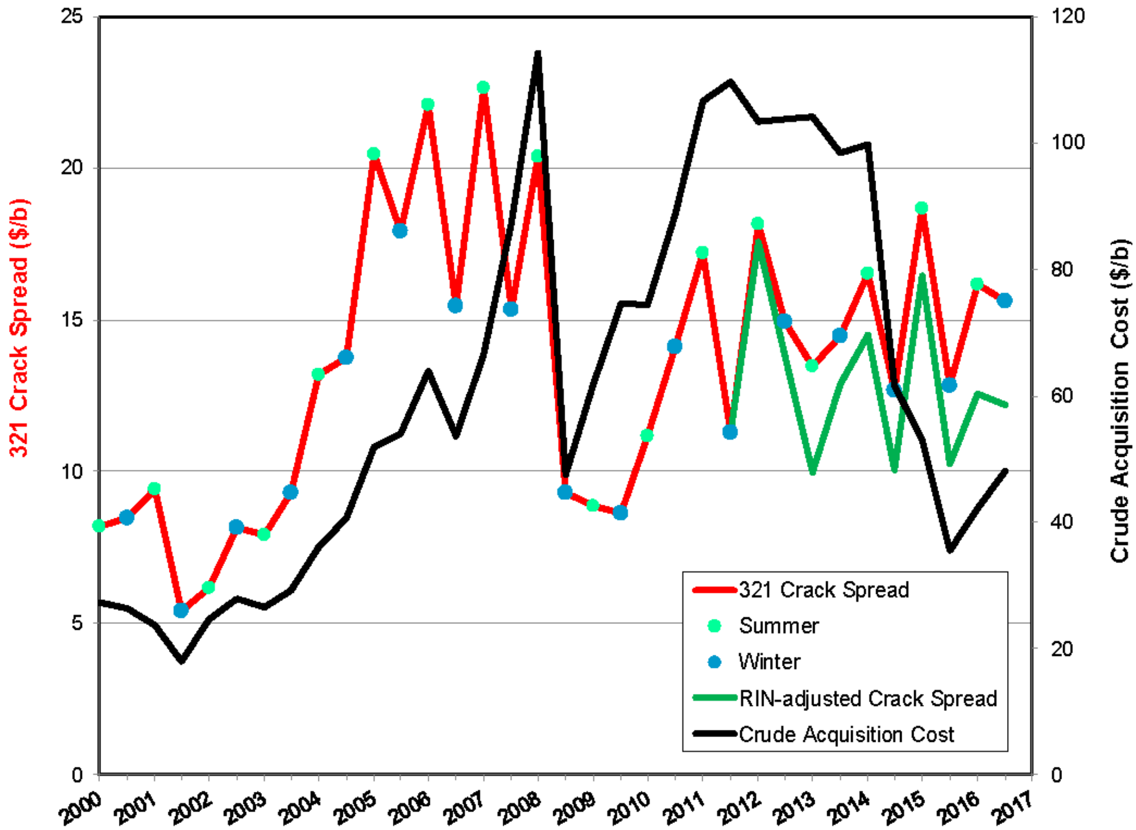

Figure D.4. PADD 3 Seasonal 3-2-1 Crack Spread, Crude Oil Acquisition Cost, and Crack Spread Adjusted for the Cost of RIN Bundle. Note: Uses crude oil acquisition cost to calculate crack spreads.

Included in the above figure is a RIN-adjusted Crack Spread. Under the RFS program, refiners must acquire specified volumes of RINs in proportion to their production of BOBs (for domestic use) and ULSD (but not jet fuel). This is a line-item cost to refiners that has grown substantially in recent years and is thought by many observers of the U.S. refining sector to be mostly, if not completely, passed forward through increases in the prices of those refined products affected by the RFS program. The refinery models used in this study do not incorporate the cost of RINs.

Consequently, we adjusted downwards in Table D.24, above, the reported spot market prices for refined products by the estimated cost of a RIN bundle for the summer and winter seasons in 2016 so that they may be compared directly with the incremental costs for refined products returned by the refining models. These comparisons suggest the following:

- PADD 1: The model's incremental refining costs for gasoline and ULSD are reasonably close to reported spot prices.
- PADD 2: Spot prices were not available from EIA; however, average wholesale/resale prices for regular gasoline reported by EIA suggest that the model's incremental refining costs are around \$3/bbl lower. EIA wholesale/resale price data for No. 2 diesel suggest that the model's incremental refining costs for ULSD may be on the order of \$6/bbl lower.
- PADD 3: The model's incremental refining costs for gasoline and ULSD are about \$4/bbl lower than reported spot prices in the summer and about \$5/bbl lower in the winter.
- PADD 4: Spot prices were not available from EIA; however, average wholesale/resale prices are as much as \$10/bbl higher than the model's estimated incremental refining costs.
- PADD 5: The model's incremental refining costs for CARB regular gasoline and CARB diesel are close (within \$2/bbl) to reported spot prices in the summer and winter.

The likely trend in future refining margins is not something on which we should speculate. However, if refining margins in the various PADDs persist at about those observed for 2016, the results from refinery modeling for the Study case likely would understate the refining value of ethanol and biodiesel/renewable diesel by about the price/incremental cost differences identified above for the various PADDs. Note that Figure D.4 also shows that the 3-2-1 crack spread margin in 2016 was typical, if not slightly below, the historical trend from 2004, so it is reasonable to examine the impact of 2016 margins on the study results.

Figure D.5 shows that premium/regular price deltas have varied over a wide range in recent years. We think that this has resulted from (1) an increase in the relative volumes of light, tight oil crudes processed by refineries (thus increasing the supply of light, low-octane blending components and increasing the proportion of lower quality naphtha in reformer feeds); (2) increases in gasoline production and higher premium gasoline demand, requiring production of more octane barrels; (3) constraints in refinery processing capacity, particularly in reforming and alkylation capacity; and (4) changes in crude oil prices (higher crude oil prices tend to increase premium/regular price deltas, and lower crude oil prices tend to reduce them).

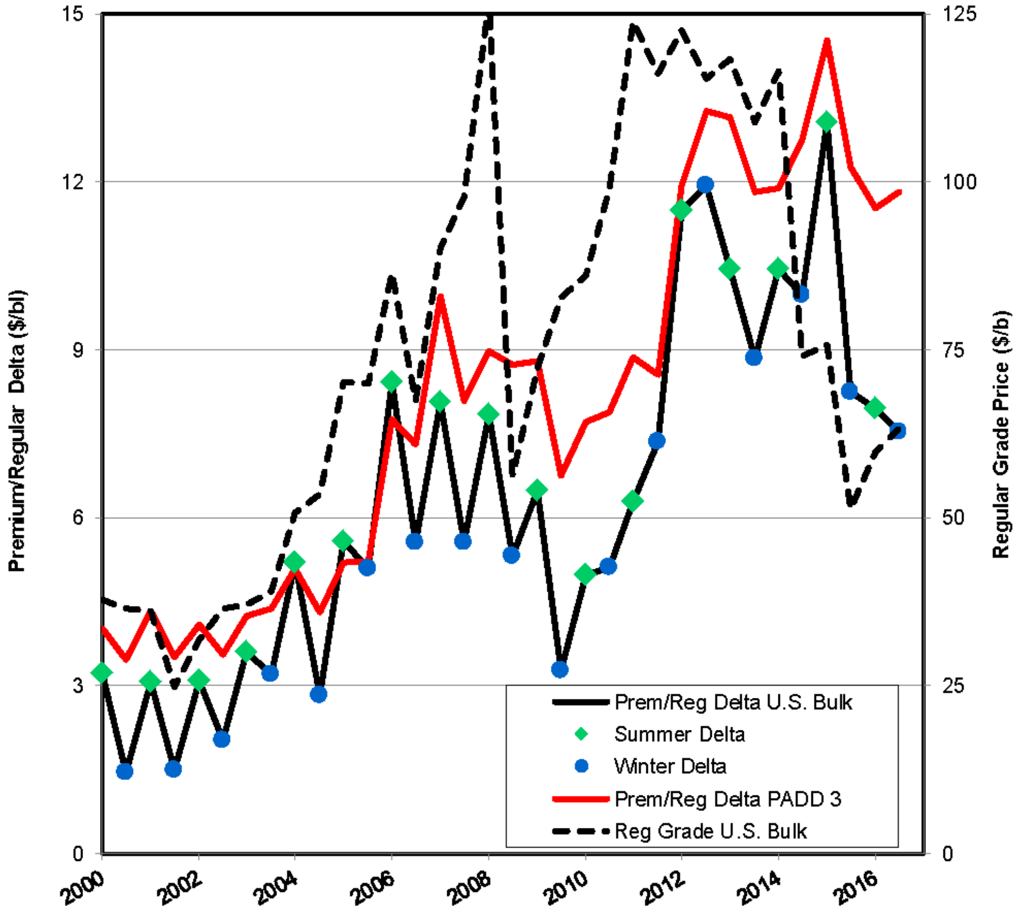

Figure D.5. Premium/Regular Price Deltas – U.S. Bulk and PADD 3 Wholesale Markets. Source: Derived from EIA Refiner Gasoline Prices by Grade and Sales Type.

Premium/regular price deltas at the bulk/spot level appear to have declined in 2016, but they still are higher than the premium/regular cost deltas returned by all of the PADD-level refinery models. Some of the reason for this might be due to how premium gasoline is priced in the market—for example, incremental octane cost may be set by the cost of high-octane blendstocks, such as reformate and alkylate, rather than according to the incremental refining cost of the premium grade.

It appears that the premium/regular deltas returned by the PADD-level refinery models consistently are about $4/bbl lower than the apparent market prices. The implication is that the refining value of ethanol returned by the refining models for premium grade gasolines in the Study case probably also could be $4/bbl too low (in addition to whatever adjustments might be made to account for refining margins).

D.3.2 Reference and Study Cases

As discussed in a previous section, the most important changes from the Calibration case that are reflected in the Reference case include:

- Higher prices for crude oil—an increase of about 75% in the national average crude oil acquisition costs, from about $42/bbl in summer 2016 to about $72/bbl in 2020.
- Higher prices for natural gas—about a 50% increase, from about $3.12/mcf in summer 2016 to about $4.69/mcf in 2020.
- Modest changes in refined product outputs and capacity.
- Tightened sulfur standards for gasoline and MARPOL standards for marine bunker fuel.

Because the RFS program is assumed to remain in place in the Reference case, we continued to specify the volume use of ethanol and biodiesel, and set prices for ethanol and biodiesel at zero. The total volume of projected annual ethanol use, including ethanol blended in imported gasoline, declines by about 10 K bbl/d, from about 918 K bbl/d in 2016 to about 909 K bbl/d in 2020. The total volume of projected annual biodiesel use increases by about 7 K bbl/d, from about 169 K bbl/d in 2016 to about 176 K bbl/d in 2020. Projected total ethanol use declines due to a slight decline in gasoline use; biodiesel use increases because of a slight increase in ULSD use.

In general, the results for the Reference case, in terms of refining operations, are similar to those of the Calibration case, primarily because required refinery outputs of refined products change only modestly and gasoline grade splits (which affect gasoline pool octane) were assumed to remain unchanged. What does change significantly, however, are the incremental production costs for refined products. These refining costs are driven upwards by the higher crude oil and natural gas prices assumed in the Reference case. In turn, the refining values of ethanol and biodiesel increase significantly because of the substantial increase in the cost of the refined product into which they are blended and of the gasoline and distillate blendstocks they implicitly replace. Key results of the refining analysis for the Reference case are shown in Table D.25, below; more detailed results are provided in ICF Report Appendix D.

Study Case

The primary difference between the Reference and Study cases is that the RFS program is assumed to not be in place for the Study case. However, we assumed that the biodiesel subsidy of $1.00/gal continues. Without the RFS program in place, the use of ethanol and subsidized biodiesel would be driven by economics, not by Federal regulatory requirements. Consequently, we introduced the biofuels supply functions discussed earlier into the Study case and allowed the PADD-level refinery models to purchase whatever volumes of ethanol and biodiesel were justified on economic grounds, subject to some minimum volume requirements to account for state volume mandates.

The results from the refinery model runs for the Study case (and the Calibration and Reference cases) are summarized in Table D.25, Table D.26, and Table D.27, below. More detailed results are provided in ICF Report Appendix D. Table D.25 provides the results regarding refining volumes (inputs, process feed rates, and outputs) and the incremental production costs for gasoline and distillates. Table D.26 shows the refining valuations of ethanol in the Calibration and Reference cases (in which ethanol was priced at zero) and its refining value relative to its delivered cost in the Study case. Table D.27 shows the biodiesel supply functions used in the refinery modeling and the refining value of biodiesel relative to its estimated delivered cost (net of subsidy). The refining value of biodiesel in the Calibration and Reference cases is equal to the incremental production cost of ULSD.

In these latter two tables, we provide Study case results for Refining Valuation Relative to Net Delivered Cost under two alternatives.

- In the first table, we report the results directly returned by the refining models (i.e., with no adjustments made to refining valuations relative to the net delivered cost of ethanol or biodiesel).
- In the second table, we report refining valuations that have been adjusted upwards to reflect the degree to which the refining models in the 2016 Calibration case tended to understate spot market prices (i.e., returned incremental refining costs that were less than spot prices for gasoline and ULSD in the various PADDs [including a $4/bbl upward adjustment for premium gasoline]). If the differences between the observed spot market prices and the incremental refining costs returned by the refining models for 2016 persist in 2020, especially for PADD 4, the adjusted refining valuations would provide more appropriate and more favorable indications of the viability of ethanol and biodiesel in the absence of an RFS program.

Our findings regarding how the absence of an RFS program might affect the future use of ethanol and biodiesel are as follows.

Ethanol

Our primary finding with regard to ethanol is that its refining value is high enough relative to its net delivered cost (given projected crude oil prices of about $72/bbl) that it would continue to be used in the gasoline pool in the same volumes in the absence of the RFS as with the RFS remaining in place.[35] In 2020, that would amount to about 850 K bbl/d blended in gasoline produced by U.S. refineries and destined for markets in the United States and almost 70 K bbl/d blended in imported gasoline for a total of about 920 K bbl/d.

On average, across PADDs and seasons (excluding the California summer RFG[36]), ethanol's refining value exceeds its net delivered cost by about \$20 to \$25/bbl, although for specific PADDs, gasoline types, and seasons, those values could be higher or lower. For example, ethanol is valued higher during the summer in conventional gasoline that benefits from the ethanol RVP waiver than in RFG (or low-RVP gasoline not subject to the waiver). Ethanol tends to be valued lower in the winter than in the summer, especially for conventional gasoline (because the relaxed RVP standards in the winter permit the use of more butanes in gasoline, thereby reducing the value of ethanol's octane). Ethanol also appears to be valued lower for premium gasoline than for regular gasoline because the molar blending model indicates that ethanol's effective blending octane declines as the octane of the hydrocarbon gasoline increases.

As noted previously, the premium/regular cost differentials returned by our refinery models understate observed market spot price differentials, on average, by about \$4/bbl. Adding this amount to the refining cost of premium grade gasoline returned by the models results in ethanol having about the same refining value for both premium and regular gasoline grades.

The high refining valuations for ethanol relative to its estimated net delivered cost suggest that, all else being equal, crude oil prices would have to be substantially lower than assumed in this study for refiners to begin backing ethanol out of the gasoline pool, perhaps on the order of \$20/bbl lower. Even then, refiners would have to confront distribution issues that would arise when considering supplanting

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[35] Our analysis focused on the refining value of ethanol as a gasoline blendstock and did not consider consumer perceptions regarding the value of E10 relative to E0 and how that potentially could play out if the RFS program were not in place.

[36] In setting up the PADD 5 refinery model, we specified the properties of finished California RFG based on the Alliance North American survey data (except for oxygen content) rather than employing the Predictive Model, and allowed the model to blend ethanol up to a maximum volume set to represent 10% ethanol blending. Ethanol's blending RVP was set such that it would result in an RVP uplift of about 1.6 psi when blended at 10 vol%. Other approaches to modeling California RFG would be to employ the Predictive Model to determine either BOB properties or finished gasoline properties. In the former, ethanol would not appear in the refinery modeling; in the latter, blending ethanol at 10 vol% (3.5 wt% oxygen) would be required rather than being optional; consequently, the model would not return its incremental value as a gasoline blendstock. Our results suggest that California refineries have high RVP control costs at the margin, and that the high effective RVP of ethanol reduces its refining value and would be a deterrent to blending it at 10 vol% in the entire gasoline pool if distribution and regulatory issues were not a concern. Under different circumstances, refiners might prefer to produce a portion of the RFG pool as E10 and the remainder as E0. However, this is not a realistic option for California refiners, given the regulatory structure (California RFG3 oxygen mandate, the LCFS, and the Predictive Model) under which they operate. Thus, for the purposes of this study, we considered ethanol to be mandated at 10 vol% in California RFG, and the negative refining value of ethanol (relative to its delivered price) had no effect on the results of the study.

E10 with E0. Furthermore, as discussed earlier, backing out ethanol, a high-octane blendstock, from the gasoline pool would tend to raise both the cost of producing octane and the incremental cost of gasoline, diminishing incentives to replace ethanol with hydrocarbon blendstocks.

Biodiesel

Our findings regarding biodiesel use in the absence of an RFS program are different from those for ethanol. As discussed above, biodiesel and renewable diesel, although suitable blendstocks for ULSD, have no special properties that would increase their refining value beyond the refining cost of ULSD.[37] Thus, in this study, they are treated as fuel extenders, meaning that they would be blended in ULSD only if their net delivered cost was the same or lower than that of refinery- produced ULSD.

Our results indicate, given the $72/bbl projected 2020 price of crude oil and the continuation of the $42/bbl biodiesel subsidy, that, in the absence of an RFS program:

- Biodiesel use would decline by about 65 K bbl/d from the Reference case—from about 170 K bbl/d to about 105 K bbl/d—if spot market prices for ULSD in 2020 were similar to the incremental cost of ULSD returned by our regional refinery models. (Biodiesel is blended as long as the refining valuations, relative to net delivered cost, are positive. When the relative valuations turn negative, there is no longer an economic incentive to blend more biodiesel in ULSD.)
- Biodiesel use would decline by only about 15 K bbl/d from the Reference case (to about 155 K bbl/d) if the differences between observed spot market price margins for ULSD and the incremental costs of producing ULSD returned by our refinery models found for 2016 persist into 2020. These spot price/incremental cost differences are shown in the rightmost column in Table D.27. They were used to adjust upwards the biodiesel valuations returned directly by the regional refinery models. Because our analysis limited PADD-level supply of biodiesel to the volumes required to be used under the RFS program, volumes greater than that could be used in PADD 2 if crude oil prices rose to about $72/bbl.
- Biodiesel use appears to be sensitive to changes in the market price of crude oil. For example, our results suggest that a decline in crude oil prices of about $5/bbl could reduce biodiesel use to almost as low as the mandate volume of 45 K bbl/d to about 95 K bbl/d, depending on the assumptions regarding the relationship of spot market prices to the incremental cost of

[37] Renewable diesel's high cetane number would increase its value in California relative to the cost of CARB diesel, when blended at the refinery, because California refineries can be constrained on cetane. But this does not factor into the study results because of the implicit mandates imposed by California on the use of biodiesel. Refineries in other parts of the country do not appear to be constrained on cetane.

ULSD production returned by the refinery models. These results are shown in Exhibit E2 in ICF Report Appendix E (the two exhibits in ICF Report Appendix E report the results of refining analysis conducted for a 2020 No-RFS case using a composite crude oil acquisition cost of $67/bbl [$5/bbl lower than the $72/bbl price used in our primary analysis]). The refining valuations shown in Exhibit E2 also indicate that a further reduction in crude oil prices of about $5/bbl ($10/bbl total) would reduce biodiesel use to the mandate volume.

- Both the 2017 and 2018 AEO project wholesale ULSD prices to increase relative to gasoline prices over time, possibly due to projected increases in exports of ULSD (the AEOs do not break down exports of individual refined products, and EIA does not make their estimates of refined product exports publicly available). If this projected trend in ULSD prices occurs, the economics of blending biodiesel in ULSD would improve.

If biodiesel were to lose its $42/bbl subsidy, our results indicate (given the assumed price for crude oil of $72/bbl) that biodiesel use would decline to mandated levels (should the mandates persist in the presence of a large disparity between the refining cost of ULSD and the cost of producing biodiesel). In the absence of a subsidy, it would take an increase in the market price of crude oil of around $40/bbl, that is, to a crude oil price in the range of $110/bbl, to make biodiesel attractive as a fuel extender for ULSD.

There are many moving parts in the analysis performed for this study, leading to a fair degree of uncertainty regarding the refinery valuations of ethanol and biodiesel relative to estimates of their delivered cost. Nonetheless, we consider our finding robust that even without the RFS, ethanol use would continue at about the current RFS level. However, our findings regarding the extent to which biodiesel would continue to be blended in ULSD absent the RFS are less certain, with biodiesel use (above state mandate volumes) being at the mercy of changes in crude oil prices.

Table D.25. Selected Refinery Modeling Results for the Calibration, Reference, and Study Cases.

Category	PADD 1						PADD 2					
	Calibration		Reference		Study		Calibration		Reference		Study	
	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter
Price of Crude	44.98	50.97	74	74	74	74	41.97	46.92	71	71	71	71
Crude Oil Throughput												
Volume (K b/d)	1,104	1,006	1,108	1,006	1,115	1,005	3,673	3,701	3,694	3,726	3,718	3,734
As % of Distillation Capacity	86.4%	78.7%	88.2%	80.2%	88.8%	80.1%	93.6%	94.4%	92.4%	93.2%	93.0%	93.4%
Processes												
Feed Volume (K b/d)												
Reforming	188	134	179	137	179	136	669	617	664	609	664	609
Catalytic Cracking	388	397	392	388	391	389	1,172	1,094	1,111	1,097	1,122	1,096
Catalytic Hydrocracking	36	36	36	36	36	36	293	293	321	321	321	321
Delayed and Fluid Coking	57	69	57	69	58	69	485	503	483	518	488	515
Feed as % of Capacity												
Reforming	77.9%	55.4%	83.6%	64.0%	84.0%	63.8%	82.3%	76.0%	80.9%	74.2%	80.8%	74.2%
Catalytic Cracking	81.5%	83.4%	84.4%	83.7%	84.3%	83.8%	94.8%	88.6%	90.0%	88.8%	90.9%	88.7%
Catalytic Hydrocracking	86.0%	86.0%	86.0%	86.0%	86.0%	86.0%	100.1%	100.1%	100.0%	100.0%	100.0%	100.0%
Delayed and Fluid Coking	76.1%	91.6%	76.0%	92.4%	77.8%	92.1%	93.0%	96.5%	91.4%	98.0%	92.4%	97.4%
Output as % of Capacity Alkylation Isomerization	90% 81%	78% 81%	90% 81%	76% 81%	90% 81%	76% 81%	90% 81%	86% 81%	90% 80%	84% 80%	90% 80%	84% 80%
Process Operations												
Reformer Severity	94.8	92.6	96.1	92.3	96.1	92.3	95.6	92.6	95.2	92.3	95.1	92.3
FCC Conversion	66.7	66.9	66.6	66.9	66.6	66.9	69.3	73.2	71.8	73.2	71.3	73.2

(continued)

Table D.25. Selected Refinery Modeling Results for the Calibration, Reference, and Study Cases. (continued)

Category	PADD 1						PADD 2					
	Calibration		Reference		Study		Calibration		Reference		Study	
	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter
Renewable Inputs (K b/d)												
Ethanol	66	65	65	64	65	64	216	231	215	229	215	229
Biodiesel/Renewable Diesel	11	11	11	11	5	12	45	49	47	51	24	44
Refined Products[1] (K b/d)	1,229	1,173	1,230	1,173	1,230	1,173	3,894	4,067	3,912	4,082	3,912	4,082
Gasoline:	621	615	621	615	621	615	2161	2313	2137	2287	2137	2287
E10 RFG	386	388	386	388	386	388	312	321	301	308	301	308
E10 Conv. & Low -RVP	219	207	219	207	219	207	1822	1955	1809	1942	1809	1942
Clear Finished	5	5	5	5	5	5	21	21	21	21	21	21
Exported	11	15	11	15	11	15	6	16	6	16	6	16
E85	7	7	6	6	6	6	4	4	5	5	5	5
Jet Fuel	87	87	87	87	87	87	258	263	262	267	262	267
Diesel Fuel	355	331	355	331	355	331	1025	1130	1072	1178	1072	1178
ULSD & CARB Diesel	331	277	331	277	331	277	1021	1130	1066	1176	1066	1176
Other	24	54	24	54	24	54	4	0	6	2	6	2
Refining Cost ($/b)												
Gasoline												
RFG & CARB: Premium	61.5	60.9	95.8	83.8	94.8	83.8	58.1	58.1	88.1	81.5	88.1	81.7
Regular	58.9	59.4	91.6	81.3	90.7	81.3	56.2	56.6	84.8	79.6	84.8	79.8
Conventional: Premium	59.8	60.9	92.8	83.8	91.6	83.8	57.1	57.7	85.5	81.1	85.5	81.2
Regular	57.6	59.4	89.0	81.3	87.9	81.3	55.2	56.4	82.5	79.3	82.5	79.4
Distillates												
Jet Fuel	56.8	63.7	88.0	89.3	88.7	89.3	50.6	56.4	84.0	84.4	84.0	84.3
ULSD/CARB Diesel	56.7	63.7	87.8	89.3	88.5	89.3	50.5	56.2	83.7	84.3	83.7	84.3

Note: Does not include ethanol and biodiesel blended in imported gasoline BOBs and ULSD.
1 Total excludes coke and sulfur.

Table D.25. Selected Refinery Modeling Results for the Calibration, Reference, and Study Cases. (continued)

Category	PADD 3						PADD 4					
	Calibration		Reference		Study		Calibration		Reference		Study	
	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter
Price of Crude	42.23	48.17	72	72	72	72	40.01	45.15	70	70	70	70
Crude Oil Throughput												
Volume (K b/d)	8,626	8,347	8,964	8,687	8,994	8,750	593	597	601	607	605	611
As % of Distillation Capacity	90.7%	87.7%	91.3%	88.5%	91.6%	89.1%	87.4%	88.0%	85.0%	85.9%	85.5%	86.4%
Processes												
Feed Volume (K b/d)												
Reforming	1,485	1,364	1,518	1,438	1,517	1,441	98	94	101	102	101	103
Catalytic Cracking	2,643	2,381	2,588	2,303	2,588	2,328	168	171	135	138	136	140
Catalytic Hydrocracking	970	970	1,059	1,059	1,059	1,059	24	24	58	58	58	58
Delayed and Fluid Coking	1,352	1,245	1,411	1,320	1,417	1,331	62	65	62	67	63	68
Feed as % of Capacity												
Reforming	87.3%	80.2%	86.4%	81.9%	86.4%	82.1%	82.5%	79.2%	84.1%	84.8%	84.2%	86.0%
Catalytic Cracking	92.2%	83.0%	92.3%	82.1%	92.3%	83.0%	85.7%	87.3	66.4%	68.2%	66.9%	68.9%
Catalytic Hydrocracking	90.0%	90.0%	90.0%	90.0%	90.0%	90.0%	84.2%	84.2%	100.1%	100.1%	100.1%	100.1%
Delayed and Fluid Coking	92.7%	85.4%	95.0%	88.9%	95.4%	89.6%	74.8%	78.7%	732%	79.2%	74.2%	80.5%
Output as % of Capacity												
Alkylation	90%	86%	90%	86%	90%	87%	90%	78%	89%	65%	89%	65%
Isomerization	80%	80%	80%	80%	80%	80%	69%	69%	69%	69%	69%	69%
Process Operations												
Reformer Severity	95.5	95.5	96.5	95.6	96.5	95.7	91.5	90.4	91.7	91.3	92.0	91.3
FCC Conversion	70.7	71.3	70.8	71.5	70.8	71.5	67.6	68.7	71.6	69.4	70.7	67.8
Renewable Inputs (K b/d)												
Ethanol	393	373	398	378	398	378	35	35	35	36	35	36
Biodiesel/Renewable Diesel	68	73	71	76	44	18	3	4	3	4	0	0

(continued)

Table D.25. Selected Refinery Modeling Results for the Calibration, Reference, and Study Cases. (continued)

Category	PADD 3						PADD 4					
	Calibration		Reference		Study		Calibration		Reference		Study	
	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter
Refined Products[1] (K b/d)	9,521	9,332	9,800	9,614	9,800	9,615	637	649	648	662	648	662
Gasoline:	4537	4493	4604	4561	4604	4561	341	348	344	352	344	352
E10 RFG	858	693	916	752	916	752	0	0	0	0	0	0
E10 Conv. & Low -RVP	3050	3012	3054	3015	3054	3015	338	345	341	349	341	349
Clear Finished	32	31	32	31	32	31	3	3	3	3	3	3
Exported	597	757	602	763	602	763	0	0	0	0	0	0
E85	3	3	2	2	2	2	1	1	1	1	1	1
Jet Fuel	880	859	903	883	903	883	41	34	42	35	42	35
Diesel Fuel	2806	2748	2973	2915	2973	2915	196	211	207	221	207	221
ULSD & CARB Diesel	2578	2488	2742	2652	2742	2652	195	210	205	220	205	220
Other	228	260	231	263	231	263	1	1	2	1	2	1
Refining Cost ($/b)												
Gasoline												
RFG & CARB: Premium	48.8	55.4	85.4	78.3	85.4	79.2						
Regular	47.2	54.2	81.9	76.4	81.9	76.8						
Conventional: Premium	48.2	55.5	83.6	78.8	83.6	79.7	50.0	54.3	79.8	77.0	79.6	77.1
Regular	46.6	54.3	80.0	76.8	80.0	77.1	48.7	53.1	77.4	75.1	77.1	75.1
Distillates												
Jet Fuel	47.5	56.6	84.3	83.4	84.4	83.8	48.6	54.0	81.8	82.1	82.1	82.1
ULSD/CARB Diesel	47.4	56.4	84.1	83.2	84.1	83.6	48.5	53.9	81.7	82.0	82.0	82.0

Note: Does not include ethanol and biodiesel blended in imported gasoline BOBs and ULSD.
1 Total excludes coke and sulfur.

Table D.25. Selected Refinery Modeling Results for the Calibration, Reference, and Study Cases. (continued)

Category	PADD 5						U.S.					
	Calibration		Reference		Study		Calibration		Reference		Study	
	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter
Price of Crude	43.95	50.20	73	73	73	73	42.54	48.23	72	72	72	72
Crude Oil Throughput												
Volume (K b/d)	2,501	2,207	2,490	2,197	2,491	2,198	16,497	15,859	16,856	16,224	16,923	16,298
As % of Distillation Capacity	85.5%	75.5%	84.9%	74.9%	84.9%	75.0%	88.2%	84.8%	90.1%	86.7%	90.5%	87.1%
Processes												
Feed Volume (K b/d)												
Reforming	445	415	400	415	402	413	2,885	2,624	2,862	2,701	2,864	2,704
Catalytic Cracking	774	678	795	645	789	645	5,144	4,721	5,020	4,571	5,026	4,598
Catalytic Hydrocracking	504	542	482	533	474	533	1,826	1,865	1,956	2,007	1,948	2,007
Delayed and Fluid Coking	475	389	513	432	513	432	2,431	2,270	2,526	2,406	2,539	2,415
Feed as % of Capacity												
Reforming	86.9%	81.1%	76.0%	78.8%	76.3%	78.4%	83.9%	76.3%	83.2%	78.6%	83.3%	78.6%
Catalytic Cracking	94.2%	82.5%	96.6%	78.3%	95.9%	78.4%	93.0%	85.4%	90.8%	82.7%	90.9%	83.1%
Catalytic Hydrocracking	92.9%	100.0%	90.4%	100.0%	89.0%	100.0%	85.7%	87.5%	91.8%	94.2%	91.5%	94.2%
Delayed and Fluid Coking	92.4%	75.7%	99.5%	83.6%	99.3%	83.7%	90.4%	84.4%	93.9%	89.5%	94.4%	89.8%
Output as % of Capacity												
Alkylation	100%	81%	95%	80%	95%	81%	90%	83%	91%	83%	91%	84%
Isomerization	63%	71%	82%	100%	83%	100%	74%	76%	81%	86%	81%	86%
Process Operations												
Reformer Severity	97.8	90.9	98.8	90.9	98.7	90.9	95.7	93.7	96.3	93.8	96.3	93.9
FCC Conversion	70.9	71.6	69.0	68.9	69.2	68.9	70.0	71.3	70.4	71.1	70.3	71.0
Renewable Inputs (K b/d)												
Ethanol	146	141	145	141	145	141	856	845	858	847	858	847
Biodiesel/Renewable Diesel	34	31	35	33	33	31	160	168	167	174	106	105

(continued)

Table D.25. Selected Refinery Modeling Results for the Calibration, Reference, and Study Cases. (continued)

	PADD 5						U.S.					
	Calibration		Reference		Study		Calibration		Reference		Study	
Category	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter
Refined Products[1] (K b/d)	2,742	2,592	2,728	2,571	2,728	2,571	18,022	17,812	18,318	18,102	18,318	18,103
Gasoline:	1492	1452	1485	1443	1485	1443	9152	9221	9191	9258	9191	9258
E10 RFG	1045	1028	1041	1024	1041	1024	2601	2430	2644	2472	2644	2472
E10 Conv. & Low-RVP	369	342	366	337	366	337	5798	5861	5789	5850	5789	5850
Clear Finished	14	14	14	14	14	14	75	74	75	74	75	74
Exported	64	68	64	68	64	68	678	856	683	862	683	862
E85	5	5	5	5	5	5	20	20	19	19	19	19
Jet Fuel	445	424	445	425	445	425	1711	1667	1739	1697	1739	1697
Diesel Fuel	580	534	598	552	598	552	4962	4954	5205	5197	5205	5197
ULSD & CARB Diesel	555	511	572	529	572	529	4680	4616	4916	4854	4916	4854
Other	25	23	26	23	26	23	282	338	289	343	289	343
Refining Cost ($/b)												
Gasoline												
RFG & CARB: Premium	64.8	62.5	96.3	84.0	96.4	84.1	58.2	59.6	91.5	81.9	91.4	82.2
Regular	61.4	61.6	91.4	82.4	91.5	82.5	55.7	58.5	87.4	80.1	87.3	80.2
Conventional: Premium	56.1	62.2	78.1	83.6	78.0	83.6	52.0	56.7	84.0	79.9	83.9	80.4
Regular	52.2	60.9	73.6	81.7	73.5	81.8	50.2	55.5	80.5	77.9	80.5	78.2
Distillates												
Jet Fuel	60.9	63.7	94.0	86.4	93.7	86.4	52.0	58.7	86.9	84.6	86.9	84.8
ULSD/CARB Diesel	55.5	61.8	88.3	85.9	88.5	85.8	49.7	57.4	84.7	84.1	84.8	84.3

Note: Does not include ethanol and biodiesel blended in imported gasoline BOBs and ULSD.
1 Total excludes coke and sulfur.

Table D.26: Refining Valuations for Ethanol ($/bbl).

PADD of Gasoline Origin	Gasoline		Refining Valuations of Ethanol				Refining Valuations of Ethanol Relative to Ethanol Prices Set in Study Cases				
			Calibration		Reference		Returned by Refining Models		Adjusted for Spot Price/ Incremental Cost Differences Found in 2016 Calibration		
	Type	Grade	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter	Adj.
PADD 1	RFG	Prem	56.90	66.49	99.83	93.90	24.61	23.01	28.61	27.01	4
		Reg	60.68	68.82	105.44	97.13	29.92	26.24	29.92	26.24	0
	Conv.	Prem	68.99	66.49	110.22	93.90	37.77	23.50	41.77	27.50	4
		Reg	72.73	68.79	115.76	97.09	43.02	26.69	43.02	26.69	0
PADD 2	RFG	Prem	59.15	65.08	92.25	91.00	25.29	24.19	32.29	31.19	7
		Reg	62.09	67.19	96.53	93.51	29.57	26.68	32.57	29.68	3
	Conv.	Prem	65.72	65.08	101.84	90.95	34.88	24.13	41.88	31.13	7
		Reg	68.47	67.06	105.85	93.29	38.89	26.47	41.89	29.47	3
PADD 3	RFG	Prem	49.62	59.81	88.72	86.43	18.85	19.27	26.85	28.27	8 to 9
		Reg	52.09	61.69	93.66	89.02	23.79	22.63	27.79	27.63	4 to 5
	Conv.	Prem	54.14	59.85	98.03	86.49	25.07	16.25	33.07	25.25	8 to 9
		Reg	56.58	61.70	102.91	89.05	29.96	19.57	33.96	24.57	4 to 5
PADD 4	Conv.	Prem	58.1	61.8	93.8	88.0	23.2	17.7	37.18	31.68	14
		Reg	60.6	64.0	97.7	91.0	27.1	20.8	37.08	30.77	10
	Low RVP	Prem	58.3		94.5		23.8		37.84		14
		Reg	60.7		98.3		27.7		37.74		10
PADD 5	RFG	Prem	38.39	68.43	44.41	95.65	-29.40	21.46	-24.40	26.46	5
		Reg	41.41	69.34	44.79	97.07	-28.89	22.87	-27.89	23.87	1
	Conv.	Prem	77.58	69.78	105.20	95.77	33.00	23.81	38.00	28.81	5
		Reg	72.49	69.20	108.80	96.79	36.34	24.86	37.34	25.86	1
Average[2]							30.00	23.00	35.00	28.00	

Notes:

(1) Refining valuations for ethanol represent averages of similar gasoline types that may have different destinations.

(2) Refining valuations for Low-RVP gasoline are not reported, except for PADD 4, because they are similar to those for RFG when not qualifying for the ethanol RVP waiver or to conventional gasoline when qualifying for the RVP waiver.

1. Includes a $4/b upward adjustment for premium gasoline.
2. Excludes California RFG for the summer.

Table D.27. Refining Valuations for Biodiesel (assumes that $1.00/gal [$42/bbl] biodiesel subsidy applies).

ULSD		Biodiesel Supply Function		Biodiesel Mandate Volume (K b/d)	Refining Valuations of Ethanol Relative to Ethanol Prices Set in Study Cases				
PADD of					Returned by Refining Models		Adjusted for Spot Price/ Incremental Cost Differences Found in 2016 Calibration		
Origin	Destination	Price with Subsidy ($/b)	Cumulative Volume (K b/d)		Summer	Winter	Summer	Winter	Adj.
PADD 1	PADD 1	83.00	4.4	2.9	5.1	5.8	5.1	5.8	0
		84.66	4.6		3.5	4.2	3.5	4.2	
		88.58	12.1		-0.4	0.3	-0.4	0.3	
		91.51	14.9		-3.4	-2.7	-3.4	-2.7	
PADD 2	PADD 2	35.99	6.8	6.7	47.3	47.9	53.3	53.9	6
		79.17	16.7		4.1	4.7	10.1	10.7	
		81.83	23.8		1.5	2.0	7.5	8.0	
		83.38	44.0		-0.1	0.5	5.9	6.5	
PADD 3	Northeast	82.00	2.6	1.7	1.7	1.2	5.7	6.2	4 to 5
		83.66	2.8		0.1	-0.5	4.1	4.5	
		87.58	7.2		-3.9	-4.4	0.1	0.6	
		90.51	8.9		-6.8	-7.3	-2.8	-2.3	
	Southeast	78.71	0.75		5.4	4.9	9.4	9.9	4 to 5
		82.92	2.71		1.2	0.7	5.2	5.7	
		84.58	2.89		-0.4	-1.0	3.6	4.0	
		88.50	16.86		-4.4	-4.9	-0.4	0.1	
		89.92	17.26		-5.8	-6.3	-1.8	-1.3	

(continued)

Table D.27. Refining Valuations for Biodiesel (assumes that $1.00/gal [$42/bbl] biodiesel subsidy applies). (continued)

ULSD		Biodiesel Supply Function		Biodiesel Mandate Volume (K b/d)	Refining Valuations of Ethanol Relative to Ethanol Prices Set in Study Cases				
PADD of		Price with Subsidy ($/b)	Cumulative Volume (K b/d)		Returned by Refining Models		Adjusted for Spot Price/ Incremental Cost Differences Found in 2016 Calibration		
Origin	Destination				Summer	Winter	Summer	Winter	Adj.
	PADD 2	83.58	5.53		0.1	-0.4	4.1	4.6	4 to 5
		85.24	6.20		-1.5	-2.0	2.5	3.0	
		86.29	18.81		-2.6	-3.1	1.4	1.9	
	PADD 3	33.76	3.11	2.5	50.0	49.4	54.0	54.4	4 to 5
		79.02	10.89		4.7	4.2	8.7	9.2	
		81.68	12.77		2.0	1.5	6.0	6.5	
		83.23	32.99		0.5	0.0	4.5	5.0	
		84.89	34.43		-1.2	-1.7	2.8	3.3	
		85.94	39.30		-2.2	-2.7	1.8	2.3	
		88.31	49.33		-4.6	-5.1	-0.6	-0.1	
PADD 4	PADD 4	40.34	0.14		41.3	41.3	51.3	51.3	10
		52.44	0.18		29.2	29.2	39.2	39.2	
		86.76	0.54		-5.1	-5.1	4.9	4.9	
		91.08	1.47		-9.5	-9.5	0.5	0.5	
		96.21	2.80		-14.6	-14.6	-4.6	-4.6	
PADD 5	PADD 5	87.74	29.76	30.7	2.5	-1.6	3.5	-0.6	1
		89.86	33.20		0.4	-3.8	1.4	-2.8	
		97.24	37.10		-7.0	-11.1	-6.0	-10.1	

(continued)

Table D.27. Refining Valuations for Biodiesel (assumes that $1.00/gal [$42/bbl] biodiesel subsidy applies). (continued)

ULSD		Biodiesel Supply Function		Biodiesel Mandate Volume (K b/d)	Refining Valuations of Ethanol Relative to Ethanol Prices Set in Study Cases				
PADD of		Price with Subsidy ($/b)	Cumulative Volume (K b/d)		Returned by Refining Models		Adjusted for Spot Price/Incremental Cost Differences Found in 2016 Calibration		
Origin	Destination				Summer	Winter	Summer	Winter	Adj.
Estimated Biodiesel Use (K b/d)				45	106	105	152	163	
PADD 1					5	12	5	12	
PADD 2					24	44	44	44	
PADD 3					44	18	68	75	
PADD 4					0.2	0.2	1	1	
PADD 5					33	31	33	31	

Note: Biodiesel volumes in the Reference case were set at 167 K b/d in the Summer and 174 K b/d in the Winter.

Appendix E: Supplemental Analysis for Ch. 7 (Attribution: Biodiesel and Renewable Diesel)

E.1 Estimating Biodiesel and Renewable Diesel Use from State Mandates and Related State Programs (2010–2019)

E.1.1 Introduction

This document outlines the methods used to estimate the minimum volume of biodiesel and renewable diesel that may have been consumed in the United States in the absence of the RFS Program each year from 2010 through 2019. During this time period biodiesel and renewable diesel has generally been priced higher than petroleum diesel, and thus it likely has not been cost effective to blend these fuels into diesel fuel without the $1 per gallon Biodiesel Tax Credit (BTC, Chapter 7, Figure 7.5) and other incentives. In addition to the RFS Program and the federal BTC, several states also have implemented mandates for the use of biodiesel and renewable diesel or other programs that provide significant incentives for the use of these fuels. To estimate the minimum volume of biodiesel and renewable diesel likely to be used in the absence of the RFS Program, state-level fuel mandates and low carbon fuel programs (i.e., the California Low Carbon Fuel Standard [LCFS] and Oregon's Clean Fuels Program) were examined and their likely impact on the use of biodiesel and renewable diesel each year from 2010 to 2019 was estimated. There are numerous other state incentives in the form of tax benefits that likely also play a role, but it is difficult to estimate the amount of biodiesel attributable to those incentives and thus these are omitted in the RtC3. Future reports may examine these incentives in greater detail. The methods used to estimate the impact of these state-level programs and the total volume of biodiesel and renewable diesel attributable to these programs (in the absence of the RFS Program) are described in the following sections of this paper.

E.1.2 Assessing State Mandates and Incentives

To identify state mandates for the use of biodiesel and renewable diesel or state incentives that may be significant enough to result in the use of these fuels in the absence of the RFS Program, the database of state laws and incentives compiled by the U.S. Department of Energy Alternative Fuels Data Center (AFDC)[1] was searched. From the many mandates and incentives in this database the incentives listed in Table E.1 were identified as the most likely to be significant enough to incentivize the use of biodiesel and renewable diesel in the absence of the RFS Program. Table E.1 does not include mandates or incentives that applied to only a portion of the diesel fuel used in a state (for example, incentives or

[1] https://afdc.energy.gov/fuels/laws/BIOD

mandates that only applied to diesel used by schools, state and local government, or in heating oil applications).

Table E.1. State mandates and incentives for biodiesel and renewable diesel from the AFDC database.

State	Mandate/Incentive	Approximate Start Date
California	LCFS program in place since 2011	2011
Hawaii	State tax rate for biodiesel is 0.25 times the tax rate for diesel	2009
Illinois	Sales and use tax (normally 6.25%) does not apply to biodiesel blends B11–B99	2003
Iowa	Has varied over time and by biodiesel blend rate; currently $0.035 per gallon of biodiesel for B5–B10 and $0.055 per gallon for biodiesel for B11+	2006
Kansas	$0.03 per gallon tax credit per gallon of biodiesel for biodiesel blends that exceed specified threshold; threshold is B20 in 2020	2009
Kentucky	Biodiesel producers and blenders are eligible for a credit of up to $1 per gallon of biodiesel produced or blended, subject to a "tax credit cap." The tax credit cap for all producers and blenders of biodiesel has been $10 million since 2009.	2005
Louisiana	All diesel fuel must contain at least 2% biodiesel when biodiesel production in Louisiana reaches 10 million gallons per year[2]	2010
Maine	Tax rate for biodiesel blends B90+ is $0.025 lower than the tax rate for diesel	2005
Minnesota	All diesel must contain at least 20% biodiesel during the summer months (April–September) and at least 5% biodiesel during the winter months (October–March)[3]	2005
Montana	Distributors are eligible for a $0.02 per gallon tax rebate and retailers are eligible for a $0.01 per gallon tax rebate for biodiesel produced from feedstocks from Montana	2005
New Mexico	All diesel must contain at least 5% biodiesel	2012
North Dakota	Tax credit of $0.05 per gallon for fuel containing at least 5% biodiesel	2009
Oregon	All diesel must contain at least 5% biodiesel; Oregon also has a clean fuels program in place since 2016[4]	2009
Pennsylvania	Increasing biodiesel use mandates as state biodiesel production thresholds are met; currently all diesel fuel must contain at least 2% biodiesel	2010
Rhode Island	Biodiesel is exempt from the $0.30 per gallon motor fuel tax	2009
South Dakota	Motor fuel excise tax is $0.02 lower for biodiesel blends containing at least 5% biodiesel than it is for diesel fuel; only activated when production capacity and production thresholds have been met (not currently active)	2015
Texas	Biodiesel portion of a biodiesel blend is exempt from state excise tax (normally $0.20 per gallon)	2005
Washington	All diesel must contain at least 2% biodiesel; minimum requirement increases to 5% when certain state production thresholds are met (not met for years assessed)	2009

[2] Based on publicly available data, Louisiana appears to have reached the production threshold in 2010 with the opening of the Dynamic Fuels renewable diesel production facility (https://afdc.energy.gov/fuels/laws/BIOD).
[3] Minnesota's mandated minimum level for biodiesel in the summer months was 5% starting in 2009, increased to 10% in 2014, and increased again to 20% in 2018 (data from https://afdc.energy.gov/fuels/laws/BIOD). The winter month minimum was unchanged with each update.
[4] Oregon's mandated minimum level increased from 2% to 5% in 2015 (data from https://afdc.energy.gov/fuels/laws/BIOD).

E.1.3 Estimating Biodiesel and Renewable Diesel Use from State Mandates and Low Carbon Fuel Programs

Having identified states with mandates or incentives for the use of biodiesel and renewable diesel, an assessment was performed of the likelihood these mandates or incentives would be significant enough to result in the use of biodiesel and renewable diesel in the absence of the RFS Program. While it is possible that states that currently mandate or incentivize the use of biodiesel and renewable diesel would not have enacted these mandates in the absence of the RFS Program, for this analysis it was assumed that all the state-level programs would have existed in the absence of the RFS Program.

To estimate the volume of biodiesel and renewable diesel that may have been used from 2010 through 2019 in the absence of the RFS Program, the focus was on states with use mandates for biodiesel and renewable diesel and states with low carbon fuels program. As both these types of programs function as mandates (for either biodiesel/renewable diesel blending or the reductions in the carbon-intensity of transportation fuel) it is reasonable to assume that they would have resulted in biodiesel and renewable diesel use absent the RFS Program. It is also possible that other state financial incentives may have been large enough during at least a portion of the time period being examined to result in the use of biodiesel and renewable diesel in the absence of the RFS Program. Determining the degree to which these programs would have resulted in the use of biodiesel and renewable diesel would require a sophisticated economic analysis, which is beyond the scope of this approach in the RtC3.

For each of the states with a mandate for the use of biodiesel or renewable diesel or a clean fuels program in place for at least one year from 2010 (i.e., the first year of a binding effect from the RFS Program on biodiesel or renewable diesel, see Chapter 7) through 2019 (the last year data were available when the analysis was conducted), the volume of these fuels expected to be used each year was estimated. For states with mandates for the minimum use of biodiesel and renewable diesel, it was estimated that in the absence of the RFS Program biodiesel and renewable diesel use would be equal to the minimum mandated volume. There were five states in this category (Table E.2; Louisiana, Minnesota, New Mexico, Oregon, Pennsylvania, Washington). Table E.2 summarizes the state mandates for biodiesel and renewable diesel use in states with mandates, the total volume of ultra-low sulfur diesel (ULSD) used in these states, and the minimum volume of biodiesel and renewable diesel needed to satisfy these mandates each year from 2010 through 2019.

Table E.2. State biodiesel and renewable diesel mandates and ULSD consumption. Note the first sub-table, is multiplied by the second sub-table to yield the third.

Mandated Biodiesel/Renewable Diesel Blend Level										
State	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**2018**	**2019**
Louisiana	2%	2%	2%	2%	2%	2%	2%	2%	2%	2%
Minnesota	5%	5%	5%	5%	7.5%	7.5%	7.5%	7.5%	12.5%	12.5%
New Mexico	0%	0%	2.5%	5%	5%	5%	5%	5%	5%	5%
Oregon	2%	2%	2%	2%	2%	5%	5%	5%	5%	5%
Pennsylvania	2%	2%	2%	2%	2%	2%	2%	2%	2%	2%
Washington	2%	2%	2%	2%	2%	2%	2%	2%	2%	2%
ULSD Consumption (Million Gallons)[5]										
State	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**2018**	**2019**
Louisiana	1,331	1,641	1,781	1,795	1,833	2,135	2,115	2,286	2,228	1,807
Minnesota	823	856	929	942	963	966	929	969	971	982
New Mexico	479	516	545	564	600	672	657	696	758	764
Oregon	571	599	551	641	737	725	753	760	780	780
Pennsylvania	1,439	1,447	1,306	1,885	1,959	1,668	1,615	1,619	1,669	1,681
Washington	803	867	907	957	1,004	1,121	1,145	1,152	1,208	1,235
Estimated Biodiesel and Renewable Diesel Use (Million Gallons)										
State	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**2018**	**2019**
Louisiana	27	33	36	36	37	43	42	46	45	36
Minnesota	41	43	46	47	72	72	70	73	121	123
New Mexico	0	0	14	28	30	34	33	35	38	38
Oregon	11	12	11	13	15	36	38	38	39	39
Pennsylvania	29	29	26	38	39	33	32	32	33	34
Washington	16	17	18	19	20	22	23	23	24	25
Total	124	134	151	181	213	241	238	247	300	294

For states with low carbon fuels programs, it was estimated that in the absence of the RFS Program use of these fuels would be equal to actual use for years in which the incentives were in place. For California and Oregon,[6] biodiesel and renewable diesel use was estimated based on publicly reported data from the LCFS and Clean Fuels programs, respectively. These volumes are summarized in Table E.3.

[5] Data from EIA Prime Supplier Sales Volumes (https://www.eia.gov/petroleum/marketing/prime/).
[6] Oregon had both a fuel mandate and a low carbon mandate (data from https://afdc.energy.gov/fuels/laws/BIOD). To avoid double counting, the biodiesel from the fuel mandate was counted first, and then any biodiesel above the fuel mandate was assumed attributable to the low carbon fuel program.

Table E.3. Biodiesel and renewable diesel use in states with incentives (million gallons).[7]

State	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
California	0	14	29	177	180	292	419	505	568	830
Oregon	0	0	0	0	0	0	47	51	53	76
Total	0	14	29	177	180	292	466	557	621	906

Finally, the combination of the volumes of biodiesel and renewable diesel estimated that would have been used in the absence of the RFS Program in states with mandates and in states with low carbon fuels programs were compared to the total volume of biodiesel and renewable diesel used in the United States. These data are shown in Table E.4 and Figure E.1.

Table E.4. Estimated biodiesel and renewable diesel use without the RFS Program (million gallons).

Usage Category	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
States with Mandates	124	134	151	181	213	241	238	247	300	294
States with Clean Fuels Programs	0	14	29	177	180	292	430	519	582	867
Total (Mandates + Clean Fuel Programs)	124	148	180	358	393	533	668	766	882	1,161
Actual U.S. Use[8]	260	886	1,034	1,791	1,681	1,867	2,570	2,271	2,349	2,436
Total from State Mandates & Clean Fuels Programs as a Percent of U.S. Actual	48%	17%	17%	20%	23%	29%	26%	34%	38%	48%

[7] Data are only for years in which the incentives applied. In all other years Table E.3 reflects no use of biodiesel or renewable diesel. Estimates for California and Oregon are based on publicly available data from the LCFS (https://ww2.arb.ca.gov/resources/documents/lcfs-data-dashboard) and Clean Fuels programs (https://www.oregon.gov/deq/ghgp/cfp/Pages/CFP-Overview.aspx), respectively.

[8] Data for the actual use of biodiesel and renewable diesel in the U.S. is sourced from EIA's Monthly Energy Review (for 2010 and 2011; https://www.eia.gov/totalenergy/data/monthly/) and data collected in the RFS Program accessed through the Electronically Moderated Transaction System (EMTS) (https://www.epa.gov/fuels-registration-reporting-and-compliance-help/public-data-renewable-fuel-standard).

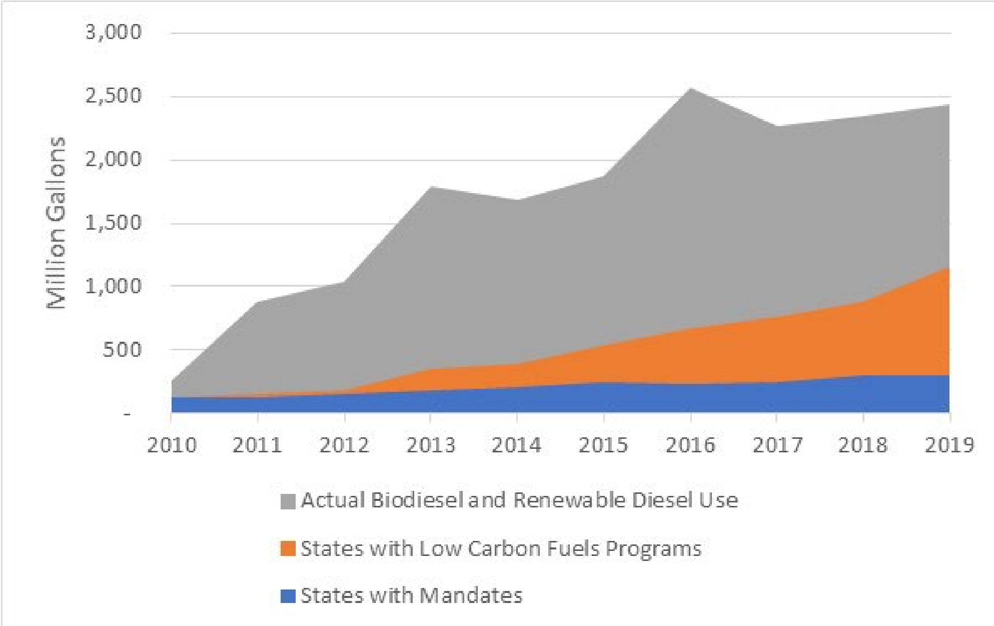


Figure E.1. Estimated biodiesel and renewable diesel use without the RFS Program vs. actual use.

E.2 Conclusions

Based on the estimates for the volume of biodiesel and renewable diesel that would have been used in the absence of the RFS Program, the RFS Program likely has been a significant driver of the production, import, and use of these fuels since the RFS2 was fully implemented in 2010. State policies, including mandates and other incentives, are estimated to have been responsible for approximately 30% of all biodiesel and renewable diesel use in the United States since 2010. This percentage had varied from year to year, from a low of 17% in 2011 to a high of 48% in 2010 and 2019. At this time the effect of the RFS Program cannot be separated from that of the BTC and other potential factors; thus, a quantitative estimate of the effect of the RFS Program on biodiesel and renewable diesel is not provided in the RtC3. However, this analysis suggests that the RFS Program and the BTC together accounted for roughly 70–100% of the biodiesel and renewable diesel used in the United States over the entire period assessed. The lower end of the range corresponds to assumptions of primacy to state programs, the upper end of the range corresponds to assumptions of primary from the RFS and the BTC. State mandates and incentives appear to be playing an increasingly significant role, with the estimate of the percentage of biodiesel and renewable diesel attributable to non-RFS factors increasing each year since 2016 (Table E.4). Finally, as discussed in more detail in the following section, the estimates of the volume of biodiesel and renewable diesel that would have been used in the absence of the RFS Program may be best interpreted as minimum values, as the volumes of these fuels that would have been used in the absence of the RFS Program in states without significant incentives or mandates for the use of biodiesel or renewable diesel were not considered.

E.2.1 Limitations

The estimate of biodiesel and renewable diesel that may have been used in the absence of the RFS Program presented in this appendix is based on estimates of the use of these fuels in states with significant incentives or mandates. It was assumed that these incentives and mandates would have been enacted in their existing forms in the absence of the RFS Program. States with mandates or incentives for the use of biodiesel and renewable diesel in only a portion of the diesel fuel used in that state (such as diesel used by state fleets, schools, or in heating oil) were not considered. However, these are likely small by comparison with the light- and heavy-duty vehicular fleet shown here. Further, it was assumed that biodiesel and renewable diesel use in states with mandates would have been equal to the minimum mandated volumes, and that biodiesel and renewable diesel use in states with low carbon fuel programs would have been equal to the observed use in years when the incentives were in place. These estimates may underestimate the use of biodiesel and renewable diesel in the absence of the RFS Program (especially in states with mandates, as actual use of these fuels may well have exceeded the mandates in this case) or they may overestimate the use of these fuels (especially in states with low carbon fuels programs, as obligated parties may have sought alternative compliance approaches without the RFS Program).

Finally, and perhaps most importantly, potential use of biodiesel and renewable diesel was not estimated for states without mandates or significant incentives in the absence of the RFS Program. It is likely that at least in some years the relative economics of biodiesel and renewable diesel vs. petroleum diesel, in combination with the federal BTC, would have resulted in the use of some quantity of biodiesel or renewable diesel in these states. In light of this limitation, the estimate is best interpreted as the minimum volume of biodiesel and renewable diesel that would have been used in the absence of the RFS and other programs, rather than a central estimate of the volume of these fuels that would have been used. A state-by-state assessment of the economics of producing and distributing biodiesel and renewable diesel relative to the price of petroleum diesel (as was done for ethanol, see Chapter 6, section 6.3.5), including the impacts of all federal and state incentives, is needed to further refine the estimate of the volume of biodiesel and renewable diesel used in the United States that is attributable to the RFS Program versus other factors.

Appendix F. Bio-Based Circular Carbon Economy Environmentally-Extended Input-Output Model (BEIOM)

F.1 Introduction

Chapters 8 (Air Quality), 10 (Water Quality), and 11 (Water Use and Availability) provide analysis across different life cycle impact analysis midpoints and resource use indicators that were derived through an environmentally extended input-output model described in Avelino et al. (2021). This appendix provides a short summary of the model framework and some additional analyses. Details are provided in the associated peer reviewed publications (Avelino et al., 2021, Lamers et al., 2021)

The Bio-based circular carbon economy Environmentally-extended Input-Output Model (BEIOM) encompasses 16 different environmental effects in a single framework (Table F.1). The environmental effects that are within scope for the RtC3 are water withdrawals, smog formation potential, eutrophication potential, acidification potential, freshwater ecotoxicity potential, PM2.5 exposure potential, and ozone depleting potential. The reader is referred to the peer reviewed publications for results on other effects. It applies a hybrid framework, linking environmentally extended input-output (EEIO) and life cycle assessment (LCA), capturing direct and indirect feedbacks between biofuel supply chains and the U.S. economy, and providing a comprehensive accounting of environmental effects related to the production and consumption of specific products or product portfolios in the United States (Avelino et al., 2021).

Table F.1. Overview of metrics and abbreviations including units. (See Avelino et al., 2021, Lamers et al., 2021).

Abbreviation	Metric	Unit
GWP	GHG emissions	kg CO_{2eq}
H_2O	Water withdrawals	m^3
LOC	Land occupation	m^2
SFP	Smog formation potential	kg O_{3eq}
EUP	Eutrophication potential	kg N_{eq}
ACP	Acidification potential	kg SO_{2eq}
FEP	Freshwater ecotoxicity potential	CTU_e
NEU	Non-renewable energy use	MJ
HTP	Total human toxicity potential	CTU_h
PEP	PM2.5 exposure potential	kg $PM_{2.5eq}$
ODP	Ozone depleting potential	kg CFC-11_{eq}
MIN	Total mineral use	kg
JOB	Total number of jobs	Person
GDP	Gross Domestic Product	USD

The approach necessitates the use of harmonized national datasets and should be considered complementary to the data and literature reviews in the individual RtC3 chapters. BEIOM's retrospective, ex-post analysis focuses on the effects from the consumption of domestically produced corn ethanol and soybean biodiesel for specific years across the period 2002–2017 and compares them to their respective substitute transport fuels (gasoline and diesel) on a well-to-wheel (WTW) basis. Other biofuels and processes have also been reported on in the source literature.

Integrated or multi-variate analyses of biofuel pathways have been included qualitatively in the First Triennial Report to Congress [RtC1, Chapter 6 (U.S. EPA, 2011)]. The Second Report to Congress (RtC2) was primarily a literature review and did not perform an integrated analysis. While the framework applied in this chapter could be described as an "attributional approach" from a methodological standpoint (Sonnemann and Vigon, 2011), it does not study the "attribution" of the RFS Program. The former term describes a system modeling approach in which inputs and outputs are attributed to the functional unit of a product system by linking and/or partitioning the unit processes of the system within physical boundaries (Sonnemann and Vigon, 2011), that is, in the context of this chapter, it estimates what share of the national environmental effect results from the production and consumption of a product. The latter term describes the causal effect of the RFS Program on biofuel production and associated effects (see Chapters 6-7 for information on this topic).

The idea of modeling economy-wide industrial supply chains incorporating environmental consequences was pioneered by Leontief (1970). Bullard et al. (1978) were one of the first to combine process-based and input-output analysis to assess net energy flows in the U.S. economy. Later, Lave (1995) proposed integrating the EEIO and LCA methodologies into a single framework (Lifset, 2009). This allowed expanding the system boundaries of process-based LCA studies by considering the feedbacks and environmental effects across more sectors of the economy, avoiding the need to truncate upstream production chains and potentially omit larger upstream polluters (Lenzen, 2000). Since then, EEIO models have been used more widely to evaluate the relationship between economic activities and associated environmental effects and, by now, several EEIO databases for both national and global levels are publicly available (Malik et al., 2019). A trade-off for EEIO modeling is its large underlying data requirements. To portray every economic transaction for every sector in the economy, governmental data is usually only available at more aggregated classifications (to avoid revealing confidential information and to reduce uncertainty) and spatial units (national, state, county). An analysis of individual supply chains requires a sectoral disaggregation, which depends on data availability including market (e.g., product value), technical (e.g., process yield), and life cycle information (e.g., energy use) of the processes in focus.

In comparison to process-based (bottom-up) LCA, the hybrid EEIO-LCA, a top-down approach, can circumvent partial boundaries by considering the interdependencies among sectors in an economy-wide perspective. EEIO-LCA does not require the definition of system boundaries. Instead, all direct and indirect effects related to the system are automatically incorporated. This puts the system to be evaluated in the context of the broader system (or the whole economy), traces structural changes, and identifies cross-sectoral impacts. The basic mechanics of an EEIO framework are that, to supply a given demand, a sector purchases inputs from other sectors in the economy, which in turn purchase inputs from other sectors, and so on. Thus, EEIO is inherently economic in nature, rather than physical like GREET (though these economic fluxes are later converted into physical potential effects). Such "ripple" effects generate impacts across the economy as the initial demand spreads throughout the different supply chains. Hence, the total impact of a change in demand (e.g., to produce biofuels) is not only its direct effect (in economic and environmental terms), but also its indirect effects generated by these sectoral feedbacks. Thus, provided the underlying economic datasets capture these interactions between sectors, the approach captures potential ripple effects and impacts outside the direct supply chain(s) in focus. Moreover, comparing the effects for the same system over time captures not only the potential shifts or changes of the environmental effects within the system or supply chain but also changes in the rest of the economy. The approach is related to, but distinct from the partial equilibrium (PE) and computable general equilibrium (CGE) economic modeling discussed in Chapters 4 and 6. While PE models offer a detailed microeconomic perspective focusing on a limited set of sectors, EEIO and CGE models offer a macroeconomic perspective with economy-wide detail. Similar to the CGE approach, EEIO accounts for all interactions between sectors in the economy. While CGE focuses on modeling substitution decisions when there are shocks in the system, EEIO focuses on the observed economic structure *per se*,[1] with an emphasis on understanding how a sector/product is linked to the rest of the economy at a given time. For example, from a product's perspective, EEIO highlights its precursor raw materials, as well as the environmental and economic effects from the acquisition and processing of these materials into the target product throughout all existing sectors of the economy.

A rich literature of process-based, bottom-up LCA studies has determined the environmental effects of various biofuel technologies in a U.S. context, including corn ethanol (Yang, 2013; Wang et al., 2012), corn kernel fiber ethanol (Qin et al., 2018), and cellulosic ethanol (Wang et al., 2012). Results remain sensitive to, among others, how coproducts such as distillers' dried grains with solubles (DDGS) are allocated, land use change, and energy grid assumptions (Canter et al., 2016; Daystar et al., 2015; Wang et al., 2011). Few integrated hybrid EEIO-LCAs have been conducted for biofuel pathways, which evaluate the effects of biofuel in the context of the U.S. economy. None has yet comprehensively described the evolution of impacts for U.S. corn ethanol and soybean biodiesel. Harto et al. (2010) evaluated the effects of U.S.-produced corn ethanol and soybean biodiesel in a hybrid framework, but only related to the water consumption profile per passenger vehicle mile traveled. Strogen and Horvath (2013) compared environmental releases from the construction, manufacturing, operation, and maintenance of the U.S. distribution infrastructure for petroleum and lignocellulosic ethanol. Liu et al. (2018) used a similar approach as BEIOM, but for a fast pyrolysis hydro-processing biofuel pathway. As

[1] Because of this, there is also no counterfactual in an EEIO analysis as there is for a CGE.

an economy-wide model, it does not define domestic system boundaries, but it is limited to domestically produced goods and services (i.e., imports are not accounted for in the main analysis). The results reflect total direct and indirect impacts linked to the production and consumption of the respective fuel (and its coproducts). For additional metrics not covered in this report and detailed methodology, see Avelino et al. (2021).

F.2 Additional Information

F.2.1 Structural Path Analysis

The BEIOM framework accounts for all direct and indirect effects related to the technology in focus. The relevance of indirect effects depends on the directly related sectors and their sector connectivity (i.e., the input and output linkages between the supply chains and the other sectors in the economy). A structural path analysis (SPA) (Defourny and Thorbecke, 1984) provides insights in how important those cross-sector relations are for the fuels in focus. SPA reveals all the consumption "paths" a product takes from the moment it is produced until it reaches the final demand (each "path" is one different way in which a product is incorporated in different industries in the production chain).

Figure F.1 shows the SPA for corn from its production (BEA code #111150) to its total final demand in the U.S. economy in 2012. Corn ethanol (325193) is the second largest demand sector after food manufacturing (311FT). Figure F.2 shows a similar degree of downstream paths for soybeans (111110), but with a smaller share entering biodiesel production (32519A). Figure F.3 shows that the economic cross-links between oil and gas extraction (211) and final demand in the United States in 2012 are much more manifold than corn and soybeans, with transport fuels (324111–324113, 32411B) being the most significant single demand sectors. Crude oil and gas partake in almost every sector of the economy, primarily via transportation (as fuel) or via petrochemical products, recirculating in the economy longer than corn/soybeans, which are embedded mainly in food related industries.

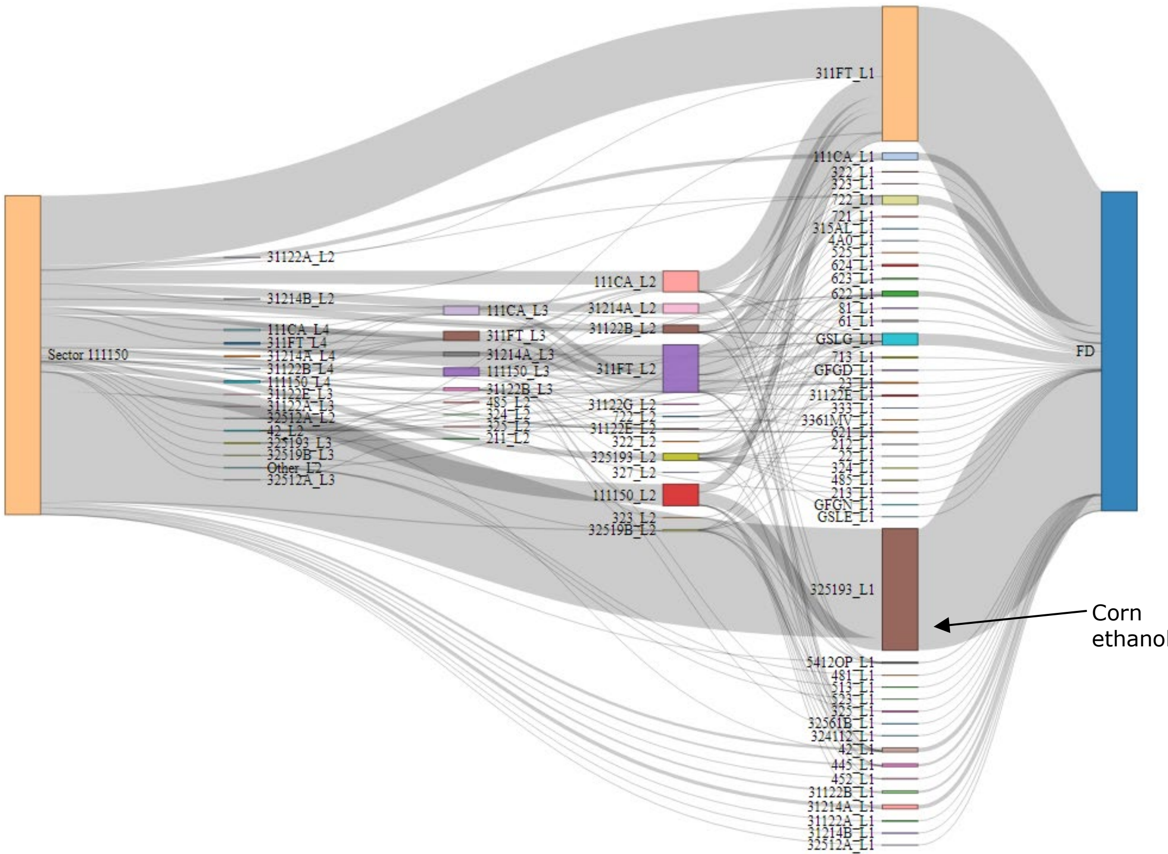


Figure F.1. Structural path analysis from U.S. corn farming (111150) to final demand (FD) for corn in 2012 across all sectors including corn ethanol (325193). Left-hand side box represents the total production (in dollars) of corn (111150) in the United States in 2012 and it is the same size as the right-hand side box (total consumption, FD). The size of intermediate boxes represents the amount of corn (in dollars) used in each sector either directly or embedded in other products. Each box is labeled *product_level*, where *product* is the BEA Summary Level code and *level* represent the number of intermediate processing steps necessary to get to final consumption.

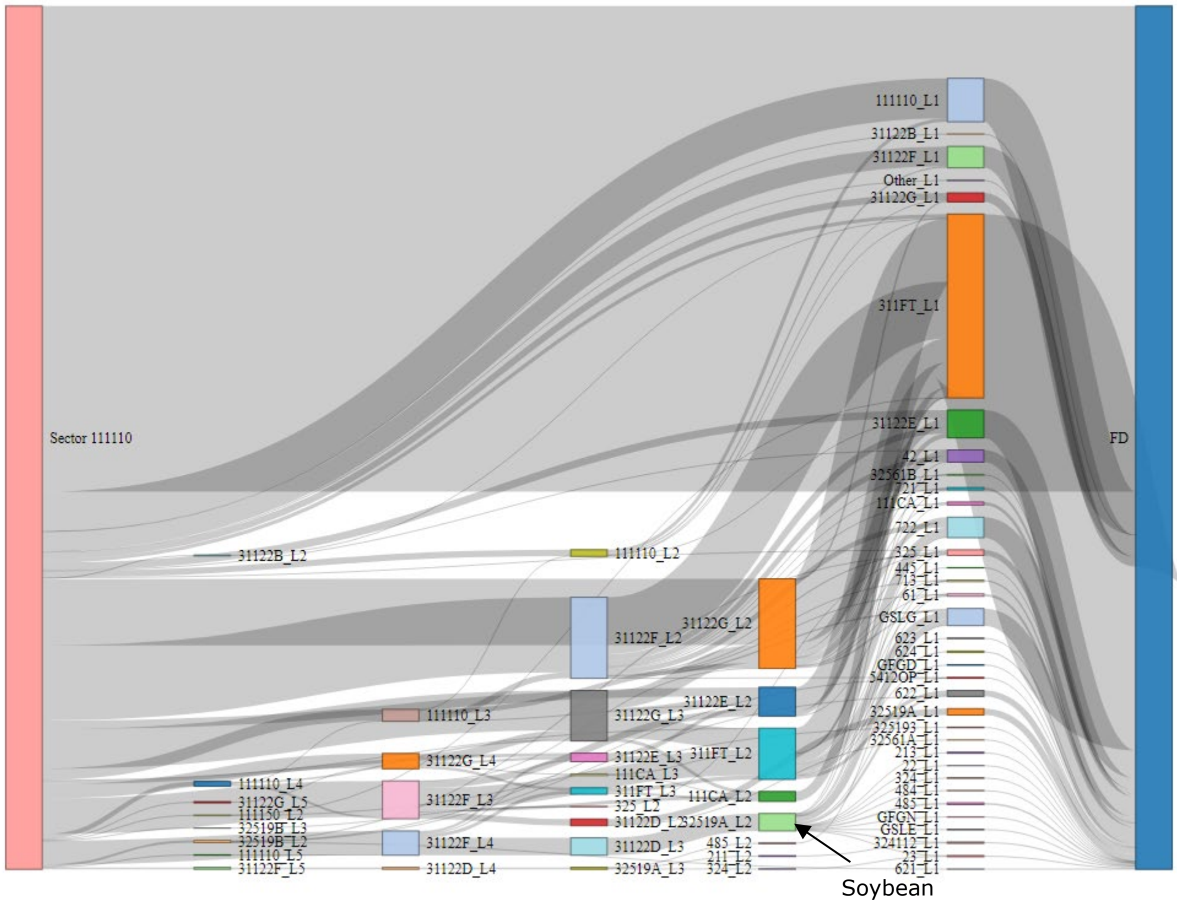


Figure F.2. Structural path analysis from U.S. soybean farming (111110) to final demand (FD) in 2012 across all sectors including soybean biodiesel (32519A). Left-hand side box represents the total production (in dollars) of soybean farming (111110) in the United States in 2012 and it is the same size as the right-hand side box (total consumption, FD). The size of intermediate boxes represents the amount of soybeans (in dollars) used in each sector either directly or embedded in other products. Each box is labeled *product_level*, where *product* is the BEA Summary Level code and *level* represent the number of intermediate processing steps necessary to get to final consumption.

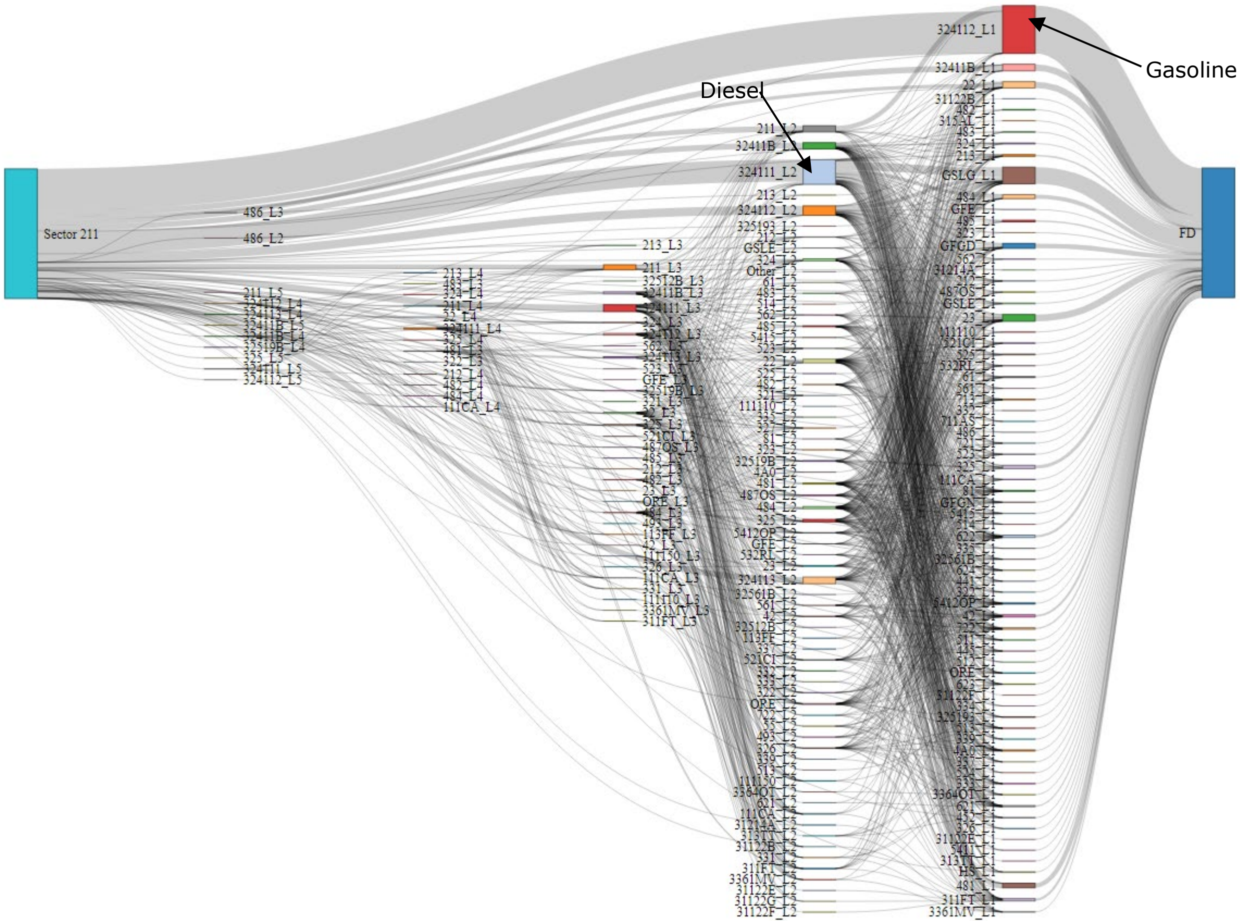
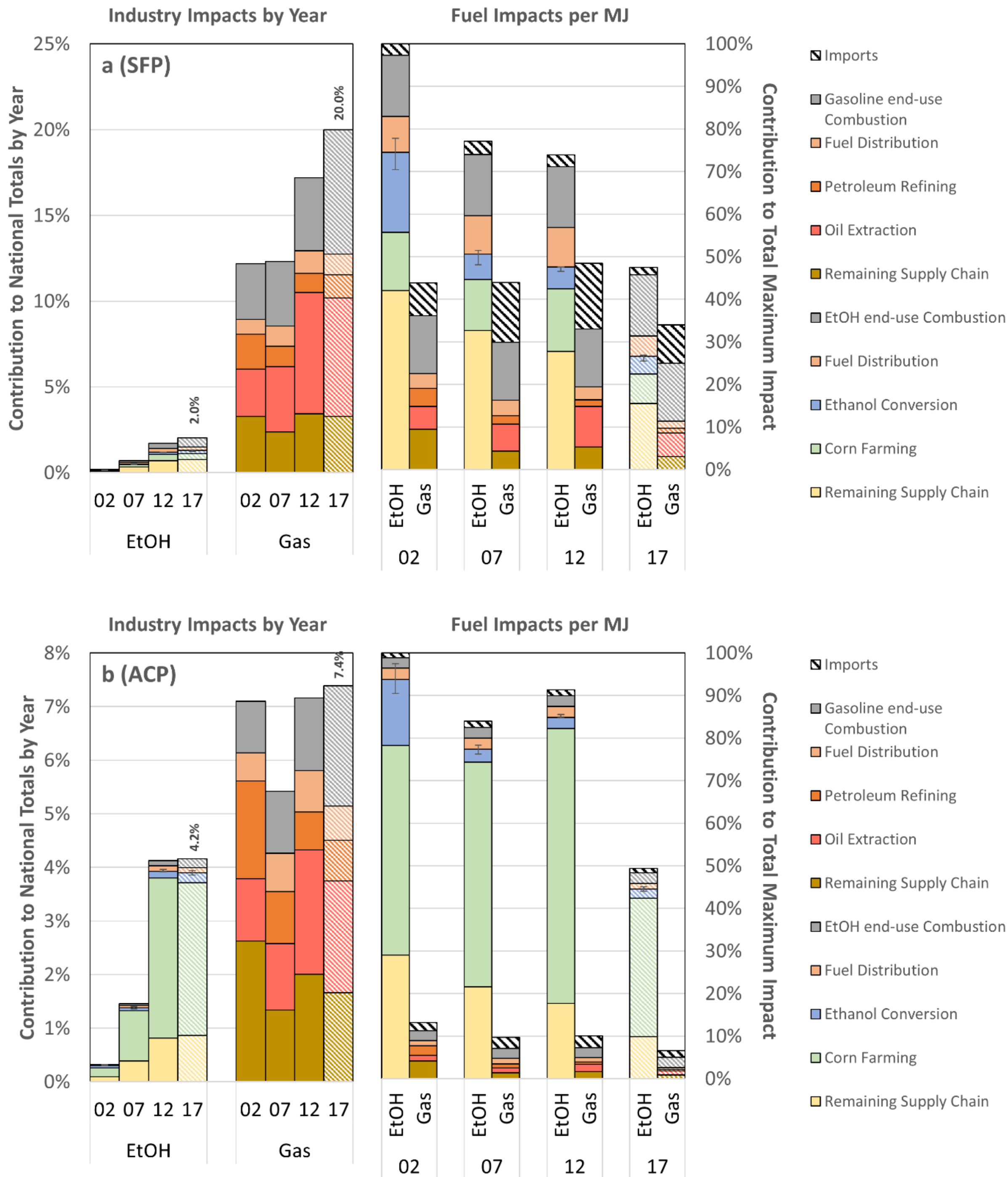


Figure F.3. Structural path analysis from U.S. oil and gas extraction (211) to final demand (FD) in 2012 across all sectors including gasoline (324112) and diesel (324111). Left-hand side box represents the total production (in dollars) of oil and natural gas extraction (211) in the United States in 2012 and it is the same size as the right-hand side box (total consumption, FD). The size of intermediate boxes represents the amount of oil and natural gas (in dollars) used in each sector either directly or embedded in other products. Each box is labeled *product_level*, where *product* is the BEA Summary Level code and *level* represent the number of intermediate processing steps necessary to get to final consumption.

F.2.2 Non-Domestic Effects due to Imports

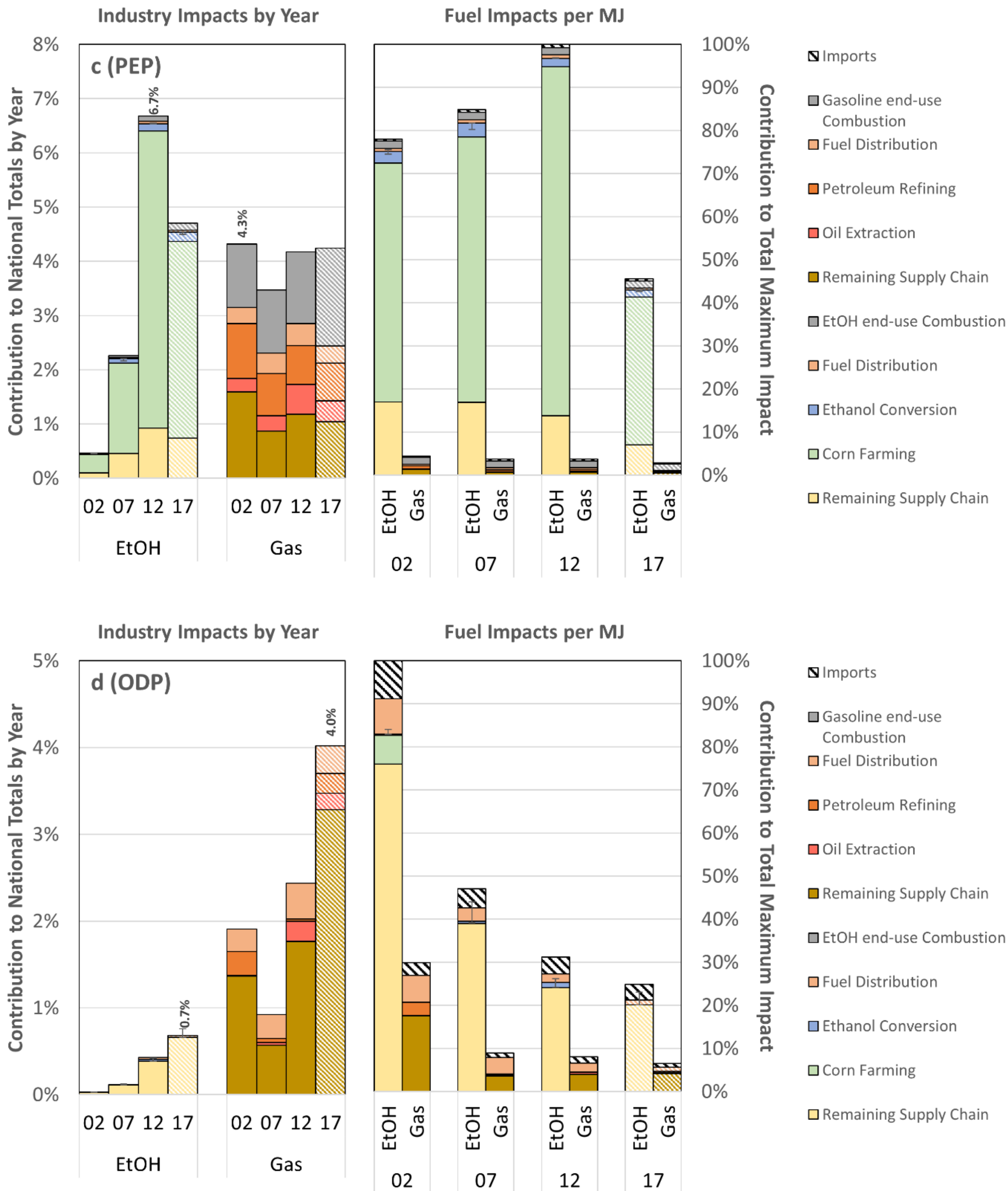
The default BEIOM model only considers domestic intersectoral linkages and does not represent international trade feedbacks. However, both biofuels and fossil fuel counterparts also rely on imported inputs, particularly crude oil. Thus, a supplemental analysis of international effects is provided, in which foreign environmental releases and resource uses are estimated by assuming that foreign sectors pollute at the same rate as domestic sectors. The following results (Figure F.4 and Figure F.5) provide the effects per megajoule if imports to the sectors are given the same environmental effect as domestic effects.

This supplemental analysis reveals that the domestic model boundary affects the results for both domestic corn ethanol and soybean biodiesel as well as their respective fossil substitutes. For most metrics, however, the inclusion of international effects did not increase the estimated effects dramatically.



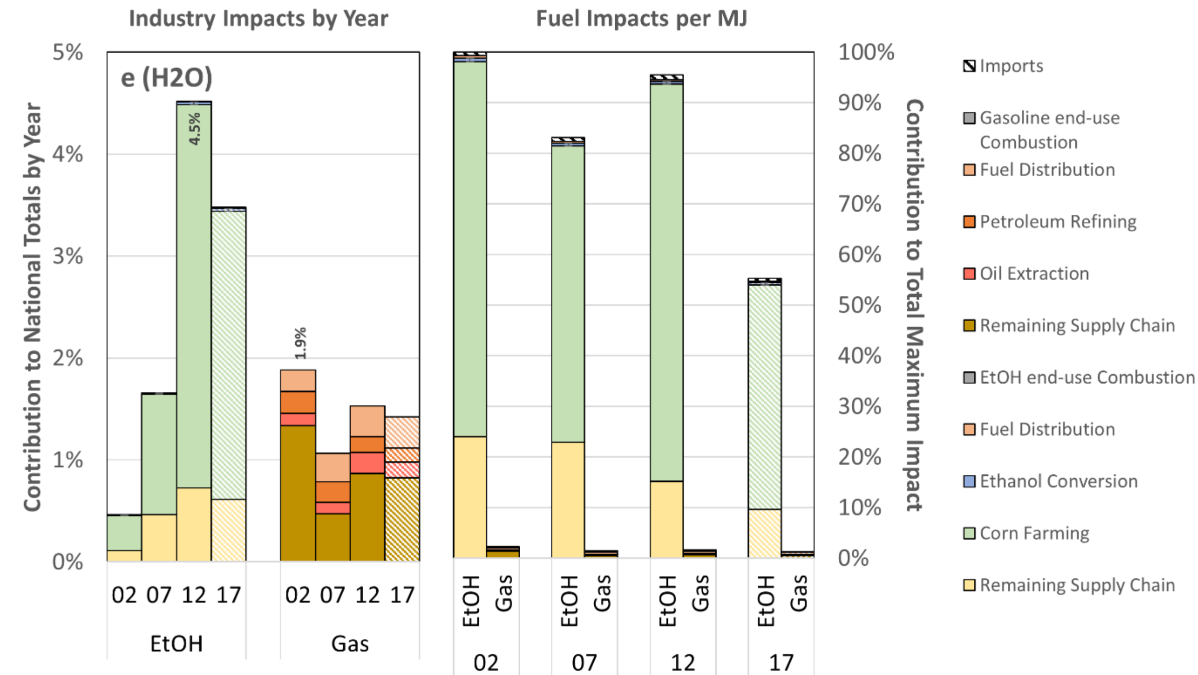
MJ = megajoules

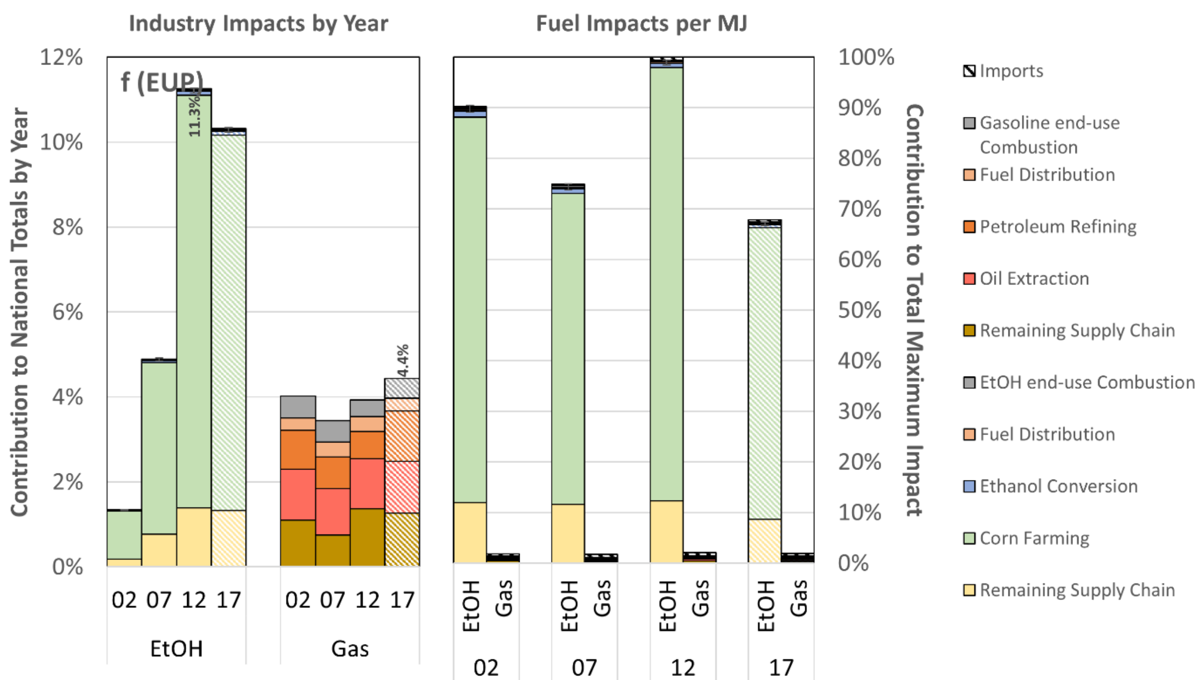
Figure F.4. Comparisons of corn ethanol (EtOH) vs. gasoline (Gas) for smog formation potential (a, SFP), acidification potential (b, ACP), $PM_{2.5}$ exposure potential (c, PEP), ozone depletion potential (d, ODP), freshwater withdrawals (e, H_2O), eutrophication potential (f, EUP), and freshwater ecotoxicity potential (g, FEP). Total industry contributions to total U.S. national emission level per year (left panel) and impacts per energy unit of fuel including foreign effects. (continued)



MJ = megajoules

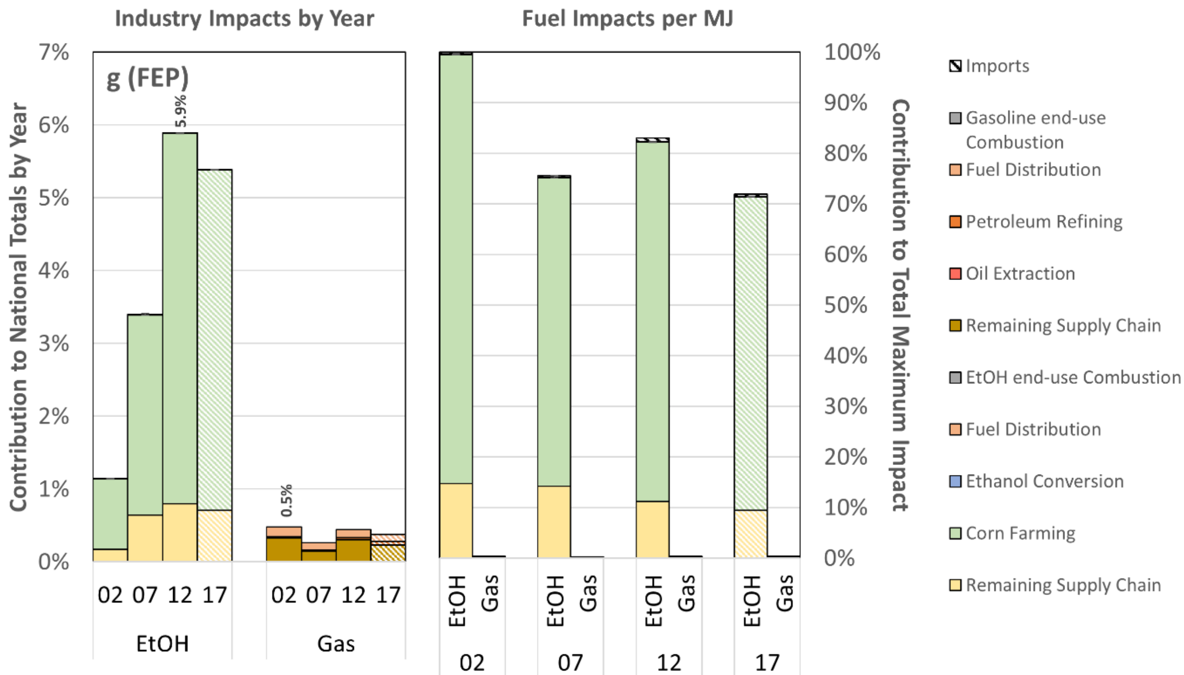
Figure F.4 (continued). Comparisons of corn ethanol (EtOH) vs. gasoline (Gas) for smog formation potential (a, SFP), acidification potential (b, ACP), $PM_{2.5}$ exposure potential (c, PEP), ozone depletion potential (d, ODP), freshwater withdrawals (e, H_2O), eutrophication potential (f, EUP), and freshwater ecotoxicity potential (g, FEP). Total industry contributions to total U.S. national emission level per year (left panel) and impacts per energy unit of fuel including foreign effects.





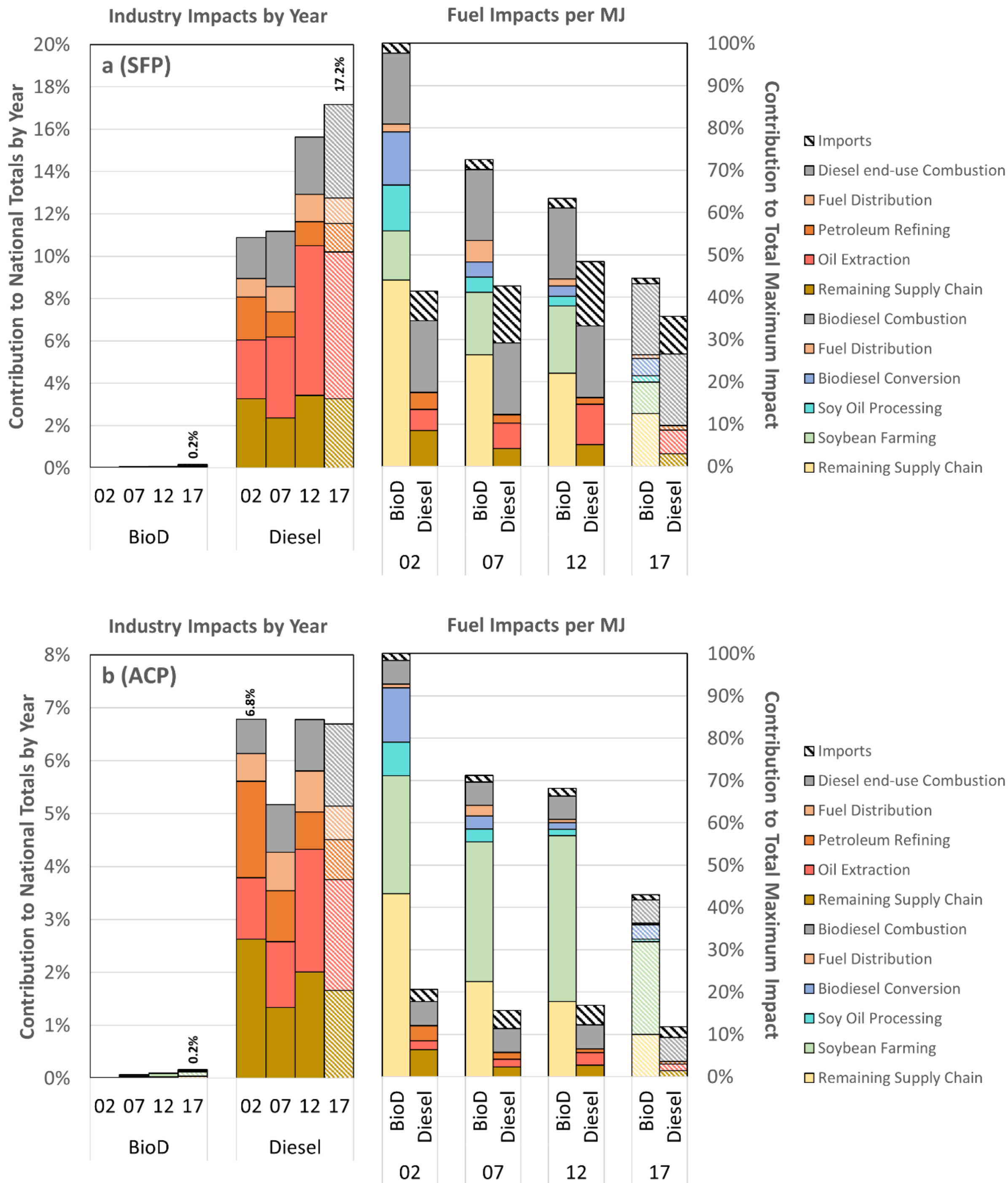
MJ = megajoules

Figure F.4 (continued). Comparisons of corn ethanol (EtOH) vs. gasoline (Gas) for smog formation potential (a, SFP), acidification potential (b, ACP), $PM_{2.5}$ exposure potential (c, PEP), ozone depletion potential (d, ODP), freshwater withdrawals (e, H_2O), eutrophication potential (f, EUP), and freshwater ecotoxicity potential (g, FEP). Total industry contributions to total U.S. national emission level per year (left panel) and impacts per energy unit of fuel including foreign effects. (continued)



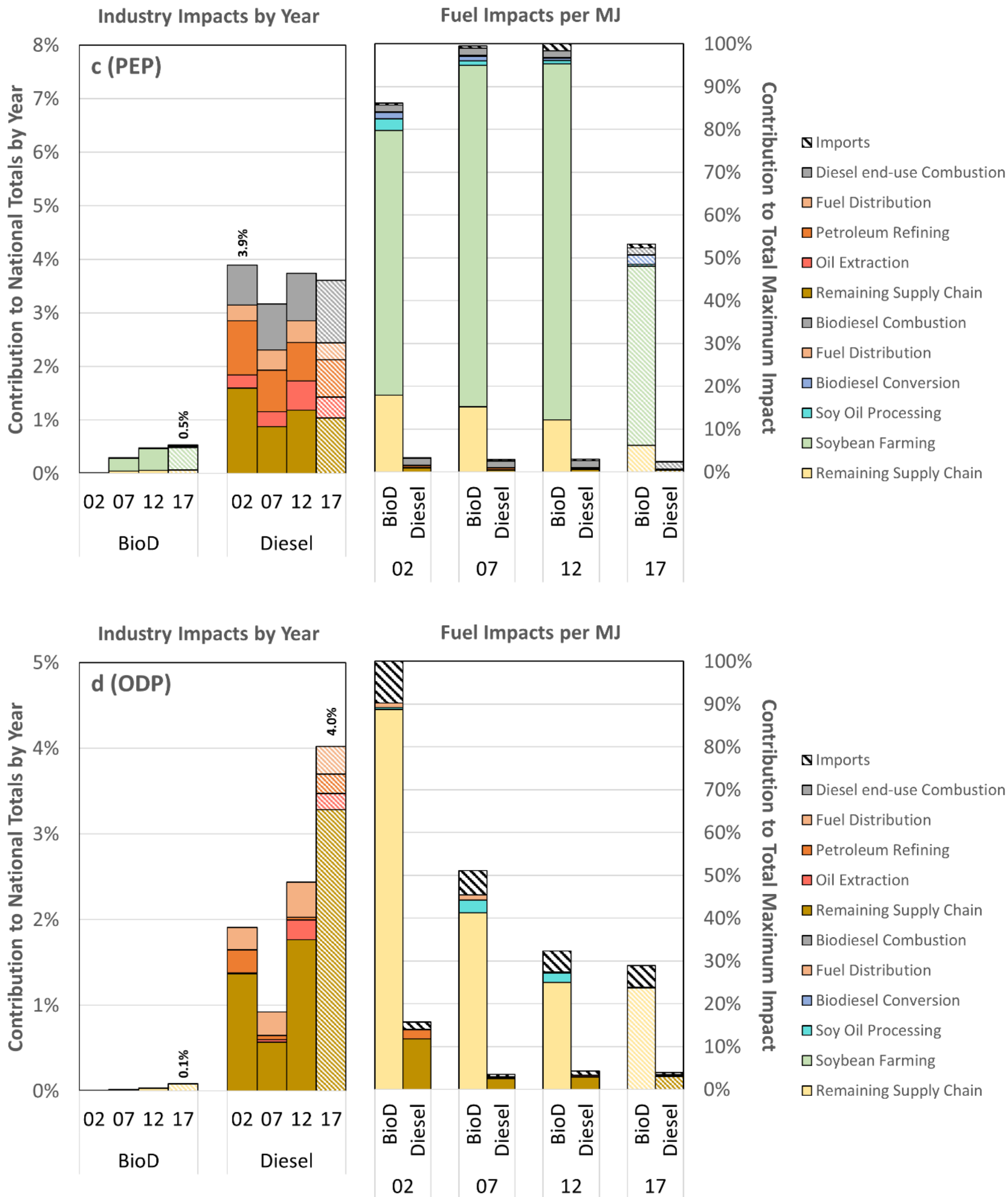
MJ = megajoules

Figure F.4 (continued). Comparisons of corn ethanol (EtOH) vs. gasoline (Gas) for smog formation potential (a, SFP), acidification potential (b, ACP), $PM_{2.5}$ exposure potential (c, PEP), ozone depletion potential (d, ODP), freshwater withdrawals (e, H_2O), eutrophication potential (f, EUP), and freshwater ecotoxicity potential (g, FEP). Total industry contributions to total U.S. national emission level per year (left panel) and impacts per energy unit of fuel including foreign effects.



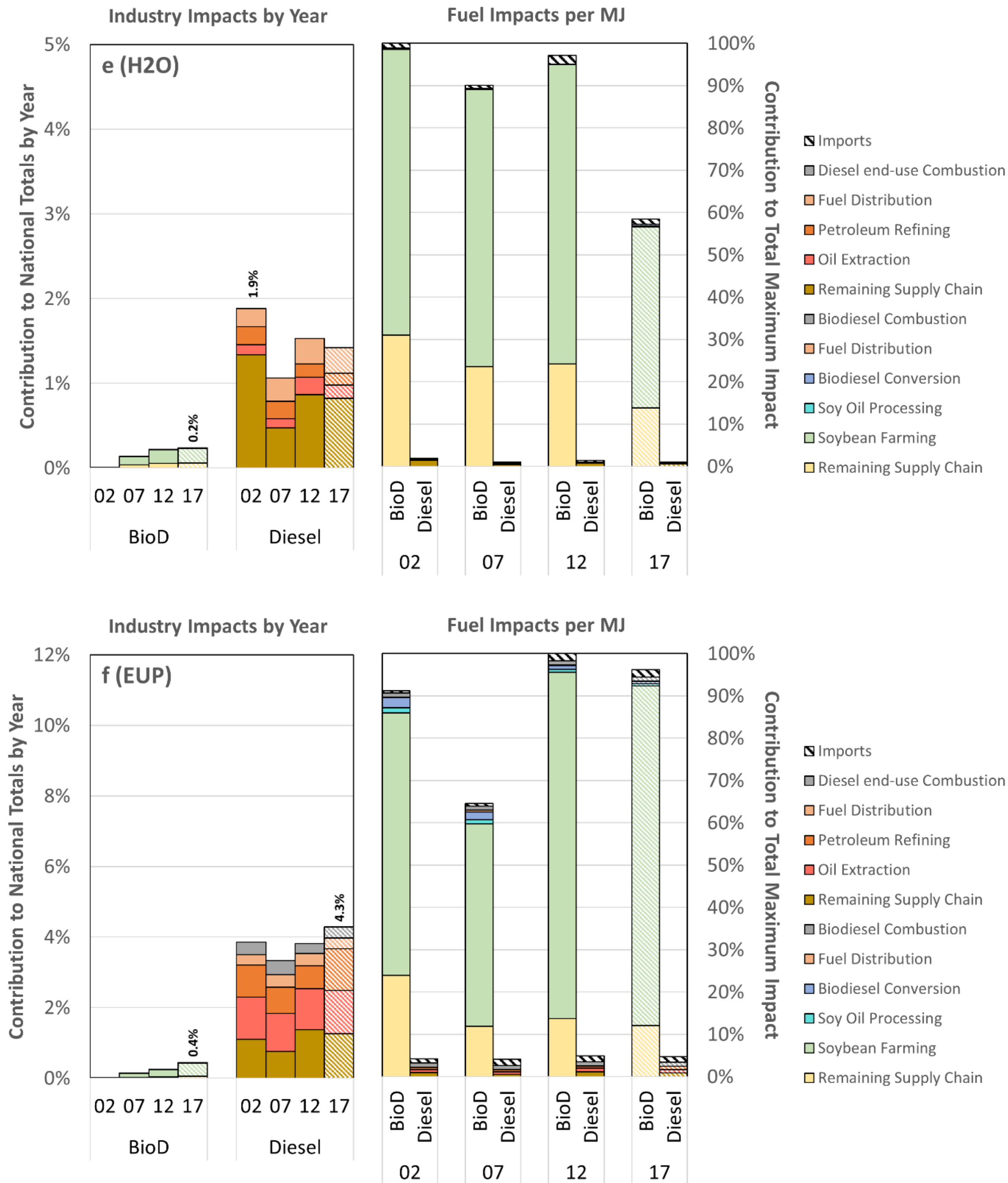
MJ = megajoules

Figure F.5. Comparisons of soybean biodiesel (BioD) vs. diesel (Diesel) for smog formation potential (a, SFP), acidification potential (b, ACP), $PM_{2.5}$ exposure potential (c, PEP), ozone depletion potential (d, ODP), freshwater withdrawals (e, H_2O), eutrophication potential (f, EUP), and freshwater ecotoxicity potential (g, FEP). Total industry contributions to total U.S. national emission level per year (left panel) and impacts per energy unit of fuel including foreign effects. (continued)



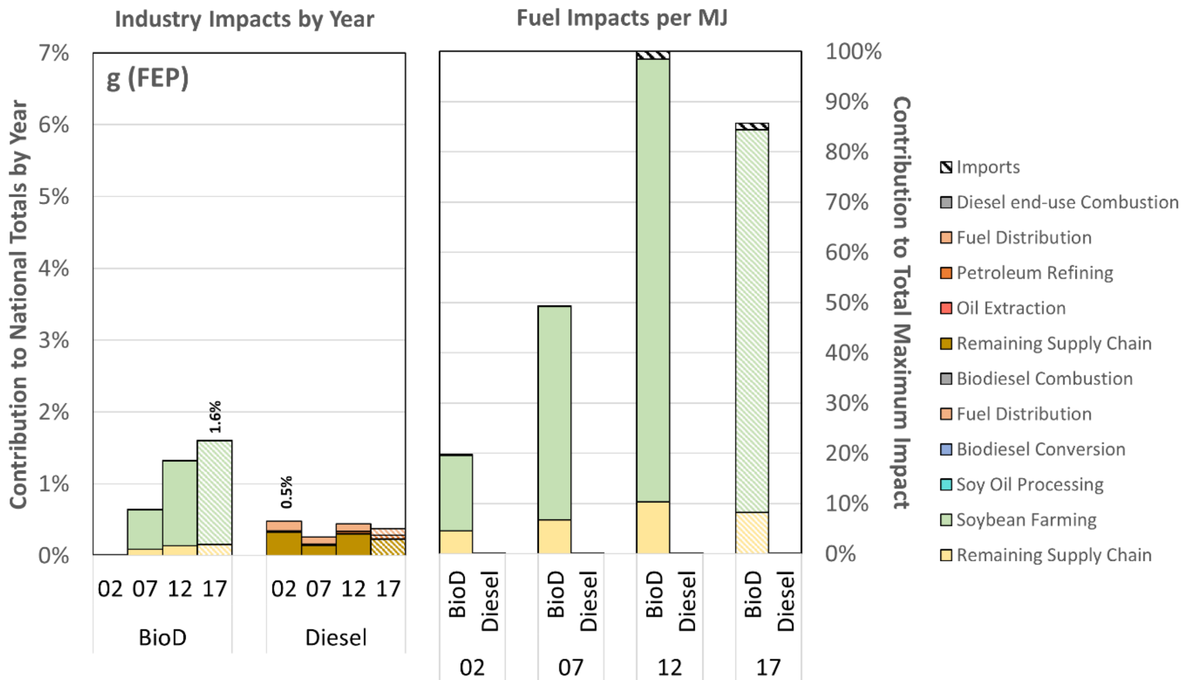
MJ = megajoules

Figure F.5 (continued). Comparisons of soybean biodiesel (BioD) vs. diesel (Diesel) for smog formation potential (a, SFP), acidification potential (b, ACP), $PM_{2.5}$ exposure potential (c, PEP), ozone depletion potential (d, ODP), freshwater withdrawals (e, H_2O), eutrophication potential (f, EUP), and freshwater ecotoxicity potential (g, FEP). Total industry contributions to total U.S. national emission level per year (left panel) and impacts per energy unit of fuel including foreign effects. (continued)



MJ = megajoules

Figure F.5 (continued). Comparisons of soybean biodiesel (BioD) vs. diesel (Diesel) for smog formation potential (a, SFP), acidification potential (b, ACP), $PM_{2.5}$ exposure potential (c, PEP), ozone depletion potential (d, ODP), freshwater withdrawals (e, H_2O), eutrophication potential (f, EUP), and freshwater ecotoxicity potential (g, FEP). Total industry contributions to total U.S. national emission level per year (left panel) and impacts per energy unit of fuel including foreign effects. (continued)



MJ = megajoules

Figure F.5 (continued). Comparisons of soybean biodiesel (BioD) vs. diesel (Diesel) for smog formation potential (a, SFP), acidification potential (b, ACP), $PM_{2.5}$ exposure potential (c, PEP), ozone depletion potential (d, ODP), freshwater withdrawals (e, H_2O), eutrophication potential (f, EUP), and freshwater ecotoxicity potential (g, FEP). Total industry contributions to total U.S. national emission level per year (left panel) and impacts per energy unit of fuel including foreign effects.

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Glossary

advanced biofuel: A renewable fuel, other than ethanol derived from corn starch, that has life cycle greenhouse gas emissions that are at least 50 percent less than life cycle greenhouse gas emissions from petroleum fuel.

agricultural residue: Plant parts, primarily stalks and leaves, that are not removed from fields used for agriculture during harvesting of the primary food or fiber product. Examples include corn stover (stalks, leaves, husks, and cobs), wheat straw, and rice straw.

algae: Photosynthetic organisms that form the base of most aquatic food webs, ranging from microscopic, single-celled diatoms, to macroscopic, filamentous green algae and large seaweeds.

anti-backsliding study: Study required under Section 211(v)1 of the Clean Air Act (CAA), to examine impacts on air quality of renewable fuel volumes mandated by United States (U.S.) Environmental Protection Agency (EPA) Renewable Fuel Standard (RFS).

aquifer: A geologic formation, group of formations, or portion of a formation capable of yielding usable quantities of groundwater to wells or springs.

B100: Pure (i.e., 100 percent) biodiesel. See also **"neat biofuel"**.

B20: A fuel mixture that includes 20 percent biodiesel and 80 percent conventional diesel and other additives. Similar mixtures, such as B5 or B10, also exist and contain 5 and 10 percent biodiesel, respectively.

bagasse: The fibrous material that remains after sugar is pressed from sugarcane.

baseflow: Sustained flow of a stream or river in the absence of precipitation or snowmelt. Natural baseflow is sustained by discharge from local or regional aquifers; "baseflow" also can be sustained by human sources (e.g., irrigation recharge to groundwater). See also **"aquifer"**.

benthic invertebrates: See **"macroinvertebrates"**.

best management practices (BMPs): The techniques, methods, processes, and activities that are commonly accepted and used to facilitate compliance with applicable requirements, and that provide an effective and practicable means of avoiding or reducing potential environmental impacts.

biochar: The product of heating biomass in the absence of- or with limited air, with the resulting material rich in organic carbon. This material can be used as a soil amendment.

biodiesel: A renewable fuel produced through transesterification of organically derived oils and fats. It may be used as a replacement for, or component of, diesel fuel. According to 40 CFR 80.1401, "biodiesel" means “a mono-alkyl ester that meets ASTM D6751 (Standard Specification for Biodiesel Fuel Blend Stock (B100) for Middle Distillate Fuels).”

biodiversity: The variety of life on Earth at all its levels, from genes to ecosystems, and can encompass the evolutionary, ecological, and cultural processes that sustain life.

biofuel blend: Fuel mixtures that include a blend of renewable biofuel and petroleum-based fuel. This is opposed to neat form biofuel that is pure, 100 percent renewable biofuel. See also **"neat biofuel"**.

biofuel consumption: The usage of biofuel in the transportation, heating, and other sectors. The biofuel can be corn ethanol, soy biodiesel, or any other biofuel.

biofuel distribution: Transportation of biofuel to blending terminals and retail outlets by a variety of means, including rail, barge, tankers, and trucks. This almost always includes periods of storage.

biofuel end use: Combustion of biofuel in vehicles and various types of engines, usually as a blend with gasoline or diesel, or in some cases in neat form. See also **"neat biofuel"**.

biofuel feedstock: Any biogenic material that is converted into fuel.

biofuel life cycle: All the consecutive and interlinked stages of biofuel production and use, from feedstock generation to biofuel production, distribution, and end use by the consumer.

biofuel production: The process or processes involved in converting a feedstock into a biofuel.

biofuel supply chain: The five main stages involved in the life cycle of a biofuel: feedstock production, feedstock distribution/transport, fuel production, fuel distribution, and fuel use.

biofuel: Any fuel made from organic materials or their processing and conversion derivatives.

biogenic: Any material having its origin in animals or plants, and which is not fossil fuel-based.

biogeochemical cycling: Describes the chemical, physical, biological, and geological processes in the environment.

biomass: Any organic matter which, in the context of biofuels, is usually plant-based (e.g., agricultural crops and crop wastes; wood and wood wastes and residues; aquatic plants; and perennial grasses) or non-plant based (e.g., fats, oils, and greases)**.**

biomass-based diesel: In the context of the Renewable Fuel Standards (RFS) Program, "biomass-based diesel", according to 40 CFR 80.1401, is "a renewable fuel that has lifecycle greenhouse gas emissions that are at least 50 percent less than baseline lifecycle greenhouse gas emissions and meets all of the following requirements:

- Is a transportation fuel, transportation fuel additive, heating oil, or jet fuel;
- Meets the definition of either biodiesel or non-ester renewable diesel; and
- Registered as a motor vehicle fuel or fuel additive under 40 CFR part 79, if the fuel or fuel additive is intended for use in a motor vehicle. Renewable fuel that is coprocessed with petroleum is not biomass-based diesel."

biorefinery: A facility that converts biomass into fuels, heat, chemicals and other products using a variety of processes and equipment.

blend wall (also known as **"blendwall"** or **"E10 blendwall"):** The E10 blend wall is the maximum amount of ethanol that can be consumed if all gasoline contains 10 percent ethanol and there are no non-ethanol gasoline (E0) or higher ethanol blends (e.g., E15 or E85). The less precise term "blend wall" does not specify a percent, but usually refers to the maximum amount of ethanol that can be consumed if all gasoline contains 10 percent ethanol, plus any additional amount from higher ethanol blends (e.g., E15 or E85).

blue water: Freshwater sourced from surface and groundwater.

cellulosic biofuel: A renewable fuel derived from lignocellulose (a plant biomass composed of cellulose, hemicellulose, and lignin that is a main component of nearly every plant, tree, and bush in meadows, forests, and fields). According to 40 CFR 80.1401, "cellulosic biofuel" is “renewable fuel derived from any cellulose, hemicelluloses, or lignin that has lifecycle greenhouse gas emissions that are at least 60 percent less than the baseline lifecycle greenhouse gas emissions.”

Census of Agriculture (Census): Provides a complete count of United States (U.S.) farms and ranches and the people who operate them at the county scale across the U.S. The "Census of Agriculture", taken only once every five years, looks at land use and ownership, operator characteristics, production practices, income and expenditures.

Census of Agriculture, select terms[1]**:**

- **total cropland:** This category includes cropland harvested, other pasture and grazing land that could have been used for crops without additional improvements, cropland on which all crops failed or were abandoned, cropland in summer fallow, and cropland idle or used for cover crops or soil improvement but not harvested and not pastured or grazed.
- **harvested cropland:** This category includes land from which crops were harvested and hay was cut, land used to grow short rotation woody crops, Christmas trees, and land in orchards, groves, vineyards, berries, nurseries, and greenhouses. Land from which two or more crops were harvested was counted only once. Land in tapped maple trees was included in woodland not pastured. The 2017 census definition for harvested cropland is the same as the 2012 definition.
- **other pasture and grazing land that could have been used for crops without additional improvements:** This category includes land used only for pasture or grazing that could have been used for crops without additional improvement. Also included are acres of crops grazed by livestock, but not harvested prior to grazing. However, cropland that was pastured before or after crops were harvested in 2017 was included as harvested cropland rather than cropland for pasture or grazing.
- **other cropland:** This includes all cropland other than harvested cropland or other pasture and grazing land that could have been used for crops without additional improvements. It includes cropland idle or used for cover crops or soil improvement, cropland on which all crops failed or were abandoned, and cropland in summer fallow.
- **cropland idle or used for cover crops or soil improvement, but not harvested and not pastured or grazed:** Cropland idle includes any other acreage which could have been used for crops without any additional improvement and which was not reported as cropland harvested, cropland on which all crops failed, cropland in summer fallow, or other pasture or grazing land that could have been used for crops without additional improvements.
- **cropland on which all crops failed or were abandoned:** No separate definition.
- **cropland in cultivated summer fallow:** No separate definition.

[1] For a full list of terms, see the *2017 Census of Agriculture*, by U.S. Department of Agriculture, National Agricultural Statistics Service, AC-17-A-51, April 2019, https://www.nass.usda.gov/Publications/AgCensus/2017/Full_Report/Volume_1,_Chapter_1_US/usv1.pdf

compressed natural gas (CNG): A gas containing primarily methane, with lesser amounts of ethane and propane and only trace amounts of heavier hydrocarbons, typically extracted from underground wells and compressed to several thousand pounds per square inch (psi).

Conservation Reserve Program (CRP): A program administered by United States Department of Agriculture (USDA) Farm Service Agency. In exchange for a yearly rental payment, farmers enrolled in the program remove environmentally sensitive land from agricultural production and plant species to improve environmental health and quality.

conservation tillage: a tillage practice leaving at least 30 percent of the soil surface covered by crop residues. Examples of "conservation tillage" include no-till and mulch tillage. See also **"tillage"**, **"no-till"**, and **"mulch tillage"**.

consumptive water use (also known as **"water consumption"**): Represents the part of water withdrawn that is evaporated, transpired, incorporated into products or crops, consumed by humans or livestock, or otherwise not available for immediate use.

continuous corn: The farming practice by which corn is grown year-after-year on the same land.

continuous saccharification: A process designed for continuous enzymatic liquefaction of corn starch at high concentration and subsequently saccharification to glucose.

conventional biofuel: In the context of this report, "conventional biofuel" refers to fuels that qualify to generate D6 RINs under EPA's RFS program. Historically, the vast majority of conventional biofuel has been ethanol derived from corn starch. However, other fuels (including grandfathered fuels) also qualify as conventional biofuel.

conventional tillage: A tillage practice leaving less than 15 percent of the soil surface covered by crop residues. See also **"tillage"**.

coproduct: A product that is produced during the production of some other product (e.g., distillers dried grains with solubles (DDGS) are a co-product of corn ethanol production).

corn stover: The stalks, leaves, husks, and cobs that are *not* removed from the fields when corn is harvested.

criteria air pollutants: Pollutants for which United States (U.S.) Environmental Protection Agency (EPA) has set National Ambient Air Quality Standards (NAAQS)**.**

crop residue: Plant material remaining after harvesting, including leaves, stalks, and roots.

crop yield: The quantity of grains or dry matter produced from a particular area of land. In this report, "crop yield" is most often measured in bushels per acre of corn or soybean.

Cropland Data Layer (CDL): A raster, geo-referenced, crop-specific land cover map for the continental United States (U.S.).

Cropland Reporting Districts (CRD): National Agricultural Statistics Service (NASS) spatial survey unit that aggregates multiple counties within a state. See **"National Agricultural Statistics Service (NASS)"**.

cultivated cropland: Includes what is commonly considered cropland, row crops, and other land used in rotation with row crops.

cyanobacteria: Photosynthetic bacteria that frequently form harmful algal blooms in marine and fresh waters; also called blue-green algae.

dedicated biofuel crop: Any crop that may be cultivated primarily for biofuel production and not for food or feed (e.g., switchgrass, algae).

deepwater habitats: (1) Permanently flooded lands that lie below the deepwater boundaries of wetlands. (2) Any open water area in which the mean water depth exceeds 6.6 feet in nontidal areas or at mean low water in freshwater tidal areas, or is covered by water during extreme low water at spring tides in salt and brackish tidal areas, or covers the deepest emerging vegetation, whichever is deeper. See also **"wetlands"**.

direct land cover and land management change: The changes in land cover and land management in order to produce feedstocks for use as a biofuel. See **"land cover and land management (LCLM)"**.

disinfection by-products (DBP): Chemical, organic, and inorganic substances that can form during a reaction of a disinfectant with naturally present organic matter in the water.

dissolved organic carbon (DOC): The fraction of organic carbon in solution, operationally defined being able to pass through a filter with a pore size typically between 0.22-0.7 micrometers.

distillers dried grains with solubles (DDGS): A coproduct from the conversion of corn to corn ethanol during either wet or dry mill process. DDGSs are extensively used as an animal feed.

double cropping: The process of planting two different crops (not including cover crops) on the same piece of land over the course of a growing season.

drinking water: Water used for human consumption that comes from a variety of sources including public water systems, private wells, or bottled water.

dry milling: A process for producing conventional corn starch ethanol in which the kernels are ground into a fine powder and processed without fractionating the grain into its component parts. Most ethanol currently comes from "dry milling".

E0: Gasoline containing no ethanol.

E10 blendwall (also known as "blendwall" or "blend wall"): The maximum amount of ethanol that can be consumed if all gasoline contains 10 percent ethanol, and there are no non-ethanol gasoline (E0) or higher ethanol blends (e.g., E15 or E85).

E10: A fuel mixture of 10 percent ethanol and 90 percent gasoline based on volume.

E15: A fuel mixture of 15 percent ethanol and 85 percent gasoline based on volume.

E85: A fuel mixture of 85 percent ethanol and 15 percent gasoline based on volume.

ecosystem health: The condition of ecological systems, including their physical, chemical, and biological characteristics, and the processes and interactions connecting them. Can also refer to the ability of an ecosystem to maintain its internal structure and organization, and to resist external stress.

ecosystem services: The benefits people obtain from ecosystems. These include provisioning services such as food and water; regulating services such as regulation of floods, drought, land degradation, and disease; supporting services such as soil formation and nutrient cycling; and cultural services such as recreational, spiritual, religious, and other nonmaterial benefits. The term "ecosystem goods and services" is synonymous with ecosystem services.

ecosystem: A dynamic complex of all the living organisms in a particular area and the non-living environmental surroundings, such as air, water, and mineral soil, with which the organisms interact.

effluent: Liquid or gas discharged in the course of industrial processing activities, usually containing residues from those processes.

Endangered Species Act (ESA): Passed by the United States (U.S.) Congress in 1973, the "Endangered Species Act (ESA)" seeks to protect and recover imperiled species and their ecosystems. Under the ESA, species may be listed as threatened or endangered. See also **"Threatened and Endangered (T&E) species"**.

enzyme loading: The amount of enzyme is effectively used in the enzymatic hydrolysis process. "Enzyme loading" is determined by the amount of cellulose present in the hydrolysate and the specific activity of the enzyme.

EPA Moderated Transaction System (EMTS): A system that was designed to allow companies to report and track transactions for Fuel Programs. Currently, there are two types of transactions that are reported in EMTS: (1) Renewable Identification Number (RIN) transactions under the Renewable Fuel Standard, and (2) Fuels Averaging, Banking, and Trading (ABT) credit transactions under the Gasoline Sulfur program. For more information, see: https://www.epa.gov/fuels-registration-reporting-and-compliance-help/how-use-emts-report-transactions-fuel-programs.

ethanol: A colorless, flammable liquid with the chemical composition C_2H_5OH that is most commonly produced by fermentation of sugars. "Ethanol" is generally blended with gasoline and used as a fuel oxygenate.

ethanol production: The industrial process by which ethanol is produced, usually by fermentation.

eutrophication: Nutrient enrichment of aquatic ecosystems, in which excessive nutrient levels cause accelerated algal growth, which in turn can reduce light penetration and oxygen levels in water necessary for healthy aquatic ecosystems. "Eutrophication" can cause serious deterioration of both coastal and inland water resources and can lead to hypoxia.

evapotranspiration: The combined processes by which water is transferred to the atmosphere from soil by evaporation and from vegetation by transpiration.

extensification: The expansion of agricultural land, like row crops, onto previously uncultivated land.

fats, oils, and greases (FOGs): In the context of biofuels, "fats, oils and greases (FOGs)" are a descriptive term that covers waste and byproduct lipids, and generally excludes virgin vegetable oil. FOGs include: animal fats (e.g., tallow, white grease, poultry fat) obtained from slaughterhouse and livestock farm waste; used cooking oil (UCO) generated at commercial and industrial cooking operations; and grease recovered from traps/interceptors installed in the sewage lines of restaurants/food-processing plants and wastewater treatment plants.

feedstock logistics: All activities associated with handling, storing, and transporting feedstocks after harvest to the point where the feedstocks are converted to biofuel.

feedstock production: All activities associated with cultivation and harvest of biofuel feedstock.

feedstock: In the context of biofuel, “feedstock” refers to any biogenic material that is converted into fuel.

filter strip: A strip or area of herbaceous vegetation that may reduce nutrient loading, soil erosion, and pesticide contamination by removing soil particles and contaminants from overland water flow.

flood irrigation: Also called surface irrigation, a broad class of irrigation systems where water is distributed over the field surface by gravity flow.

forest residue: Includes tops, limbs, and other woody material *not* removed in forest harvesting operations in commercial hardwood and softwood stands.

forest thinning: Removal of trees from overgrown forests to reduce forest fire risk or increase forest productivity. These trees are typically too small or damaged to be sold as round wood but can be used as biofuel feedstock.

fuel terminal: A waypoint in the fuel distribution system where fuels from different sources are collected and blended, and from which further distribution to retail outlets is managed.

furrow irrigation: A type of flood or surface irrigation method where farmers flow water through small trenches running through their crops.

General Equilibrium (GE) economic models: "General Equilibrium (GE) economic models" are typically global in scale and have more coarse economic resolution than Partial Equilibrium (PE) economic models. GE models account for more feedbacks to the broader economy, but have less industry detail than PE models. GE and PE models solve for the new equilibrium state following some perturbation of interest (e.g., a new policy, drought, etc.). See also **"Partial Equilibrium (PE) economic models".**

genetically engineered feedstock: Plants, trees, and other organisms that have been modified by the application of recombinant DNA technology and produce the biomass-based material converted for use as a fuel or energy product.

grassland: An open area of land dominated by herbaceous plants, including grasses. For the purposes of this Report, "grassland" includes pasture and Conservation Reserve Program land in perennial grasses. Hence, the definition is based on cover type, not use.

green water: Soil moisture from precipitation.

greenhouse gases (GHGs): Gases that trap the heat of the sun in the Earth’s atmosphere, producing the greenhouse effect. Greenhouse gases include water vapor, carbon dioxide, hydrofluorocarbons, methane, nitrous oxide, perfluorocarbons, and sulfur hexafluoride.

groundwater recharge: Natural or artificial introduction of water into the saturated zone of an aquifer.

groundwater: Water found underground in the cracks and spaces in soil, sand, and rock.

harmful algal blooms (HABs): Growths of a subset of algal species (including diatoms, dinoflagellates, and cyanobacteria) that produce toxins or grow excessively, harming humans, other animals, and the environment.

hemicellulose: Any of various plant polysaccharides less complex than cellulose and easily hydrolysable to monosaccharides (simple sugars) and other products.

herbicide resistance: The inherited ability of a plant to survive and reproduce following exposure to a dose of herbicide normally lethal to the wild type. In a plant, resistance may be naturally occurring or induced by such techniques as genetic engineering or selection of variants produced by tissue culture or mutagenesis.

High Plains Aquifer (HPA): An aquifer that underlies an area of about 174,000 square miles that extends through parts of eight states of the Midwest. This aquifer is the principal source of water in one of the major agricultural areas of the United States (U.S.). It is sometimes called the "**Ogallala Aquifer**".

hybrid: A plant species created from the offspring of genetically different parents, both within and between species. "Hybrids" combine the characteristics of the parents or exhibit new ones.

hydropattern: Changes in wetland extent, water level, and duration produced by seasonal variability in hydrologic inputs and outputs and hydraulic controls within the wetland landscape. Because the reproduction and development of wetland species are tightly synchronized with natural cycles of wetland inundation, hydropattern is a key determinant of wetland biodiversity. See also **"water balance"**.

hypoxia: The condition of waters when they are severely depleted of oxygen.

indirect land cover and land management change: A change in land cover and land management (LCLM) to fill an unmet demand in the market as a result of direct land cover land management. See **"land cover and land management (LCLM)"**.

indirect land use change (ILUC): Land use changes that occur as the result of the price-induced, or market-mediated, effects of a change in the demand for biofuels and biofuel feedstocks. For example, when biofuel feedstock is sourced from existing farms or plantations diverting land-based products from other markets, this could lead to expansion of agricultural land elsewhere through changes in agricultural prices.

integrated pest management (IPM): An environmentally sensitive approach to pest management that uses current, comprehensive information on the life cycles of pests and their interaction with the environment to manage pest damage by the most economical means, and with the least possible hazard to people, property, and the environment.

intensification: Increased intensity of cultivation with no change in total agricultural land acreage.

invasive plants: Naturalized plants that produce reproductive offspring, often in very large numbers, at considerable distances from parent plants, … and thus have the potential to spread over a considerable area. Their introduction causes or is likely to cause economic or environmental harm, or harm to human, animal, or plant health.

irrigation water applied: Water withdrawn from surface and groundwater that are applied to cropland for irrigation. A portion of the applied water is consumed by crops and a portion is lost to unnecessary evaporation, deep percolation, and runoff.

lacustrine: Wetlands and deepwater habitats with all of the following characteristics: (1) situated in a topographic depression or a dammed river channel; (2) lacking trees, shrubs, persistent emergents, emergent mosses, or lichens with greater than 30 percent areal coverage; and (3) total area exceeds 8 hectares (20 acres). Similar wetland and deepwater habitats totaling less than 8 hectares are also included in the "lacustrine" system if an active wave-formed or bedrock shoreline feature makes up all or part of the boundary, or if the water depth in the deepest part of the basin exceeds 2 meters (6.6 feet) at low water. Lacustrine waters may be tidal or nontidal, but ocean derived salinity is always less than 0.5 per mil (‰).

land cover and land management (LCLM): Land cover (LC) strictly describes the physical cover of the land surface (e.g., grassland) irrespective of what it is used for (e.g., pasture). Land management (LM) describes how the land is managed, which may include many factors which may be agronomic (e.g., fertilizer application, irrigation), or in some cases even geopolitical (e.g., zoning, land rights).

land cover: Vegetation, habitat, or other material covering a land surface.

land use: See **"land cover and land management"**.

land use change (LUC): A broad term that includes any changes in the way that areas of land are managed for productive or non-productive uses. For example, converting grassland or pasture to row crops (or the reverse) is a type of LUC.

legumes: Plants belonging to the Fabaceae family (commonly called "pea family") that typically host symbiotic nitrogen-fixing bacteria.

life cycle assessment: A comprehensive systems approach for measuring the inputs, outputs, and potential environmental impacts of a product or service over its life cycle, including resource extraction/generation, manufacturing/production, use, and end-of-life management.

life cycle greenhouse gas emissions: The aggregate quantity of greenhouse gas emissions (including direct emissions and significant indirect emissions such as significant emissions from land use changes), as determined by the United States (U.S.) Environmental Protection Agency (EPA) Administrator, related to the full fuel life cycle, where the mass values for all greenhouse gases are adjusted to account for their relative global warming potential. See also **"biofuel life cycle"**.

lignin: A group of complex organic polymers that form key structural materials in the support tissues of most plants (e.g., stems, bark, wood).

lignocellulosic biomass: Produced from atmospheric CO_2 and water using the sunlight energy through the photosynthesis process, "lignocellulosic biomass" is the most abundant raw material on the earth for biofuel productions. It is composed of polysaccharides (or carbohydrate polymers) and phenolic polymers (lignin). These carbohydrate polymers contain different sugar monomers (six and five carbon sugars) and they are tightly bound to lignin.

liquified natural gas (LNG): A gas containing primarily methane, with lesser amounts of ethane and propane and only trace amounts of heavier hydrocarbons, typically extracted from underground wells and cooled to approximately -162 degrees Celsius (-260 degrees Fahrenheit) so that it becomes a liquid.

Long Term Agricultural Projections (LTAP): Annually released, departmental consensus on a long-run representative scenario for the agricultural sector for the next decade. The projections are based on specific assumptions about macroeconomic conditions, policy, weather, and international developments, with no domestic or external shocks to global agricultural markets.

macroinvertebrates: Small organisms lacking vertebrae that live on or in the substrate (benthic invertebrates) or in the water column of lakes (zooplankton), such as dragonfly larvae and water fleas. Abundances of pollution-tolerant and pollution-sensitive macroinvertebrate species provide one indication of biological condition in aquatic ecosystems.

Major Land Use (MLU) Series: Long-term accounting of all major land uses of public and private land in the United States (U.S.). Annual estimates are updated every five years, coinciding with the Census of Agriculture. See **"Census of Agriculture"**.

Major Uses of Land Uses in the United States (U.S.), select terms[2]:

- **cropland:** Total cropland includes five components: cropland harvested, crop failure, cultivated summer fallow, cropland used only for pasture, and idle cropland.
- **cropland used for crops:** Three of the cropland acreage components—cropland harvested, crop failure, and cultivated summer fallow—are collectively termed cropland used for crops, or the land used as an input to crop production.
  - **cropland harvested:** Includes row crops and closely sown crops; hay and silage crops; tree fruits, small fruits, berries, and tree nuts; vegetables and melons; and miscellaneous other minor crops. This category includes Christmas tree farms.
  - **crop failure:** Consists mainly of the acreage on which crops failed because of weather, insects, and diseases but does include some land not harvested due to lack of labor, low market prices, or other factors.
  - **cultivated summer fallow:** Refers to cropland in subhumid regions of the West that are cultivated for one or more seasons to control weeds and accumulate moisture before small grains are planted.
- **cropland pasture:** —Generally is considered to be in long-term crop rotation. This category includes acres of crops hogged or grazed but not harvested and some land used for pasture that could have been cropped without additional improvement.
- **idle cropland:** —Includes land in cover and soil-improvement crops and cropland on which no crops were planted. Some cropland is idle each year for various physical and economic reasons.
- **grassland pasture and range:** Grassland pasture and range encompass all open land used primarily for pasture and grazing, including shrub and brush land types of pasture; grazing land with sagebrush and scattered mesquite; and all tame and native grasses, legumes, and other forage used for pasture or grazing—regardless of ownership.
- **forested land:** As defined by the Forest Service, the 766 million acres of forested land in 2012 consist of "land at least 120 feet (37 meters) wide and at least 1 acre (0.4 hectare) in size with at least 10 percent cover (or equivalent stocking) by live trees, including land that formerly had such tree cover and that will be naturally or artificially regenerated".

[2] For a full list of terms, see *Major Land Uses (MLU) in the United States, 2012*, by D.P. Bigelow and A. Borchers, EIB-178, U.S. Department of Agriculture, Economic Research Service, August 2017, https://www.ers.usda.gov/webdocs/publications/84880/eib-178.pdf

- **timberland:** Forestland that produces or is capable of producing crops (in excess of 20 cubic feet per acre per year) of industrial wood and not withdrawn from timber use by statute or administrative regulation.
- **reserved forestland:** Forestland withdrawn from timber use through statute, administrative regulation, or designation without regard to productive status. Forested wilderness areas and parks are included in this category.
- **other forestland:** Forestland other than timberland and productive reserved forestland. It includes available forestland, which is incapable of annually producing 20 cubic feet (1.4 cubic meters) per acre (0.4 hectare) of industrial wood under natural conditions because of adverse site conditions, such as sterile soils, dry climate, poor drainage, high elevation, steepness, or rockiness.

match blending: The process by which refiners produce lower octane blendstocks for oxygenate blending (BOBs) rather than finished gasoline, which is then blended with an oxygenate (commonly ethanol) before being sold as finished gasoline.

methyl tert-butyl ether (MTBE): A flammable liquid produced from petroleum refining that has been used as an oxygenate additive for gasoline.

milling residues (primary and secondary): Wood and bark residues produced in processing (or milling) logs into lumber, plywood, paper, furniture, or other wood-based products.

mulch tillage: A type of conservation tillage in which some crop residue is incorporated into the soil, but at least 30 percent of the soil surface remains covered. See also **"tillage"** and **"conservation tillage"**.

National Agricultural Statistics Service (NASS): Annual summaries that capture the production and supplies of food and fiber, prices paid and received by farmers, farm labor and wages, crop acreage, livestock populations, farm finances, chemical use, and changes in the demographics of United States (U.S.) producers. Crop acreage survey were leveraged in the land cover and land management (LCLM) analysis.

National Resource Inventory (NRI): A statistical survey of land use and natural resource conditions and trends on United States (U.S.) non-Federal lands.

National Resources Inventory (NRI), select terms[3] (see USDA (2020) for full list of terms):

- **cropland:** A land cover/use category that includes areas used for the production of adapted crops for harvest. Two subcategories of cropland are recognized: cultivated and non-cultivated:
  - **cultivated cropland:** Comprises land in row crops or close-grown crops and also other cultivated cropland, for example, hayland or pastureland that is in a rotation with row or close-grown crops.
  - **noncultivated cropland:** Includes permanent hayland and horticultural cropland:
    - **hayland:** A subcategory of cropland managed for the production of forage crops that are machine harvested. The crop may be grasses, legumes, or a combination of both. Hayland also includes land in set-aside or other short-term agricultural programs.

[3] For a full list of terms, see *Summary Report: 2017 National Resources Inventory*, by USDA Natural Resources Conservation Service and Center for Survey Statistics and Methodology, 2020, https://www.nrcs.usda.gov/sites/default/files/2022-10/2017NRISummary_Final.pdf

- **horticultural cropland:** A subcategory of cropland used for growing fruit, nut, berry, vineyard, and other bush fruit and similar crops. Nurseries and other ornamental plantings are included.

- **land cover/use:** A term that includes categories of land cover and categories of land use. Land cover is the vegetation or other kind of material that covers the land surface. Land use is the purpose of human activity on the land; it is usually, but not always, related to land cover:
  - **pastureland:** A land cover/use category of land managed primarily for the production of introduced forage plants for livestock grazing. Pastureland cover may consist of a single species in a pure stand, a grass mixture, or a grass-legume mixture. Management usually consists of cultural treatments: fertilization, weed control, reseeding or renovation, and control of grazing. For the NRI, includes land that has a vegetative cover of grasses, legumes, and/or forbs, regardless of whether or not it is being grazed by livestock.
  - **rangeland** A land cover/use category on which the climax or potential plant cover is composed principally of native grasses, grass-like plants, forbs or shrubs suitable for grazing and browsing, and introduced forage species that are managed like rangeland. This would include areas where introduced hardy and persistent grasses, such as crested wheatgrass, are planted and such practices as deferred grazing, burning, chaining, and rotational grazing are used, with little or no chemicals or fertilizer being applied. Grasslands, savannas, many wetlands, some deserts, and tundra are considered to be rangeland. Certain communities of low forbs and shrubs, such as mesquite, chaparral, mountain shrub, and pinyon-juniper, are also included as rangeland.
  - **row crops:** A subset of the land cover/use category cropland (subcategory, cultivated) comprising land in row crops, such as corn, soybeans, peanuts, potatoes, sorghum, sugar beets, sunflowers, tobacco, vegetables, and cotton.

naturalized plants: Alien plants that reproduce consistently … and sustain populations over many life cycles without direct intervention by humans (or in spite of human intervention); they often recruit offspring freely, usually close to adult plants, and do not necessarily invade natural, seminatural, or human-made ecosystems.

neat biofuel: Any biofuel that is not blended with fossil-based fuel such as gasoline or diesel. See also **"B100"**.

net energy balance: In the context of biofuel, refers to the energy content in the resulting biofuel minus the total amount of energy used over the production and distribution process.

nitrate (NO_3^-): Nitrate is a ionic compound that is formed naturally when nitrogen combines with three oxygen atoms and exists in the environment in highly water-soluble forms.

nitrogen (N): Chemical element with atomic number, found as N_2 in the atmosphere but can be "fixed" into forms available for plant growth. Nitrogen is essential for life and is a main element in amino acids, proteins, and DNA.

nitrogen fixation: The transformation of atmospheric nitrogen into nitrogen compounds that growing plants can use. Nitrogen-fixing plant species, such as soybeans, can accomplish this process through symbioses with bacteria often in their root nodules.

no-till: A type of conservation tillage, disturbing the soil only marginally by cutting a narrow planting strip, leaving most crop residue on soil surface. See also **"tillage"** and **"conservation tillage"**.

noxious weed: Any plant or plant product that can directly or indirectly injure or cause damage to crops (including nursery stock or plant products), livestock, poultry, or other interests of agriculture, irrigation, navigation, the natural resources of the United States (U.S.), the public health, or the environment.

nutrient loading: A process in which compounds from waste and fertilizers, such as nitrogen and phosphorus, enter a body of water. This can happen, for example, when sewage is managed poorly, when animal waste enters ground water, or when fertilizers from residential and agricultural runoff wash into a stream, river, or lake.

nutrients: Nutrients are chemical compounds used by living organisms, needed to grow and reproduce.

NWALT (U.S. Conterminous Wall-to-Wall Anthropogenic Land Use Trends), USGS 1974-2012, select terms[4]:

- **production, pasture/hay:** Areas of grasses, legumes, or grass-legume mixtures planted for livestock grazing or the production of seed or hay crops, typically on a perennial cycle. Identical definition to NLCD 2011 class 81.
- **production, grazing potential:** Areas of good grazing potential beyond what is indicated by the NLCD. Information suggests the land could and has been used at least on a seasonal or occasional basis for animal grazing, including woodland pasture.

octane number (also known as **"octane rating"** or **"octane value"**): A standard measure of the ability of a fuel to resist engine knock.

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Ogallala Aquifer: See "**High Plains Aquifer**". The Ogallala formation is one of the geologic units that make up the High Plains Aquifer.

oxygenate: A gasoline additive whose chemical structure includes oxygen. Most commonly refers to alcohols and ethers.

oxygenated fuels: Fuels, typically gasoline, that have been blended with an oxygenate. Sometimes used to refer specifically to the Oxygenated Fuels Program which targets reductions in carbon monoxide.

ozone (O_3): A form of oxygen consisting of three oxygen atoms. In the stratosphere (7 to 10 miles or more above the Earth's surface), ozone is a natural form of oxygen that shields the Earth from ultraviolet radiation. In the troposphere (the layer extending up 7 to 10 miles from the Earth's surface), ozone is a widespread pollutant and major component of photochemical smog.

palustrine: Nontidal wetlands dominated by trees, shrubs, persistent emergents, emergent mosses or lichens, and all such wetlands that occur in tidal areas where salinity due to ocean-derived salts is below 0.5 per mil (‰). It also includes wetlands lacking such vegetation, but with all of the following four characteristics: (1) area less than 8 hectares (20 acres); (2) active wave-formed or bedrock shoreline

[4] For a full list of terms, see U.S. Conterminous Wall-to-Wall Anthropogenic Land Use Trends (NWALT), 1974–2012 by J.A. Falcone, U.S. Geological Survey Data Series 948, https://pubs.er.usgs.gov/publication/ds948.

features lacking; (3) water depth in the deepest part of basin less than 2 meters at low water; and (4) salinity due to ocean-derived salts less than 0.5 per mil (‰).

Partial Equilibrium (PE) models: Partial Equilibrium (PE) economic models are typically regional or national in scale and have more detailed economic resolution than General Equilibrium (GE) economic models. PE models have more industry detail than GE models, but account for fewer feedbacks to the broader economy. GE and PE models solve for the new equilibrium state following some perturbation of interest (e.g., a new policy, drought, etc.). See also **"General Equilibrium (GE) economic models".**

peat soil: Soil material consisting largely of undecomposed, or only slightly decomposed, organic matter accumulated under conditions of excessive moisture.

perennial grass: A species of grass that lives more than two years and typically has low nutrient demand and diverse geographical growing range, and offers important soil and water conservation benefits.

photobioreactor: A vessel or closed-cycle recirculation system containing some sort of biological process that incorporates some type of light source. Often used to grow small phototrophic organisms such as cyanobacteria, moss plants, or algae for biodiesel production.

PM_{10}: Particles that are 10 microns or smaller in diameter.

$PM_{2.5}$: Particles that are 2.5 microns or smaller in diameter.

post emergent: In the context of a pesticide (often an herbicide), this is applied after the plant emerges from the soil.

renewable biofuel: See **"renewable fuel"**.

renewable biomass: As defined by the 2007 Energy Independence and Security Act, "renewable biomass" is any of the following:

- Planted crops and crop residue harvested from agricultural land cleared or cultivated at any time prior to the enactment of this sentence [December 19, 2007] that is either actively managed or fallow, and nonforested.
- Planted trees and tree residue from actively managed tree plantations on non-federal land cleared at any time prior to enactment of this sentence [December 19, 2007], including land belonging to an Indian tribe or an Indian individual, that is held in tryst by the United States or subject to a restriction against alienation imposed by the United States.
- Animal waste material and animal byproducts.
- Slash and pre-commercial thinnings that are from non-federal forestlands, including forestlands belonging to an Indian tribe or an Indian individual, that are held in trust by the United States or subject to a restriction against alienation imposed by the United States, but not forest or forestlands that are ecological communities with a global or State ranking of critically imperiled, imperiled, or rare pursuant to a State Natural Heritage Program, old growth forest, or late successional forest.
- Biomass obtained from the immediate vicinity of buildings and other areas regularly occupied by people, or of public infrastructure, at risk of wildfire.
- Algae.
- Separated yard waste or food waste, including recycled cooking and trap grease.

renewable diesel: Diesel fuel derived from renewable biomass, generally using a thermal depolymerization process, which meets the requirements of the American Society of Testing and Materials D975 or D396 standards.

Renewable Fuel Standard Program: The "Renewable Fuel Standard (RFS) Program" was created under the Energy Policy Act of 2005 (EPAct), which amended the Clean Air Act (CAA). The Energy Independence and Security Act of 2007 (EISA) further amended the CAA by expanding the RFS Program. The United States (U.S.) Environmental Protection Agency (EPA) implements the Program in consultation with U.S. Department of Agriculture (USDA) and the Department of Energy (DOE). The RFS Program is a national policy that requires a certain volume of renewable fuel to replace or reduce the quantity of petroleum-based transportation fuel, heating oil, or jet fuel.

Renewable Fuel Standard 1 (RFS1): The version of the Renewable Fuel Standard (RFS) Program created under the Energy Policy Act of 2005. The RFS1 was in effect from 2006 to 2008.

Renewable Fuel Standard 2 (RFS2): The Renewable Fuel Standard (RFS) Program as revised in response to requirements of the 2007 Energy Independence and Security Act. RFS2 increased the volume of renewable fuel required to be blended into transportation fuel to 36 billion gallons per year by 2022. The RFS2 has been in full effect since 2010, with 2009 being a transition year between the RFS1 and RFS2 (see Chapter 2).

renewable fuel: A fuel produced from renewable biomass that is used to replace or reduce the use of fossil fuel.

Renewable Identification Numbers (RINs): Credits used for compliance with the Renewable Fuel Standard (RFS) Program. Different biofuels produce different kinds of "Renewable Identification Numbers (RINs)" including cellulosic biofuel (D3 or D7 RINs) biomass-based diesel (D4), other advanced biofuels (D5, e.g., sugarcane ethanol), and conventional biofuels (D6).

Renewable Volume Obligation (RVO): The number of Renewable Identification Numbers (RINs) that a producer or importer of gasoline or diesel is obligated to acquire to demonstrate compliance with the applicable standards under the Renewable Fuel Standard (RFS) Program. See also **"Renewable Identification Numbers (RINs)"** and **"Renewable Fuel Standard (RFS) Program".**

RFS2 Regulatory Impact Analysis (RIA): United States (U.S.) Environmental Production Agency's (EPA) analysis of the impacts of the Renewable Fuel Standard 2 (RFS2) volume targets established by Congress in the 2007 Energy Independence and Security Act. See also **"Renewable Fuel Standard 2 (RFS2)".**

richness: The number of species or other biological organization units in a particular area.

row crop: A crop planted in rows wide enough to allow cultivators between the rows. Examples include corn, soybeans, peanuts, potatoes, sorghum, sugar beets, sunflowers, tobacco, vegetables, and cotton.

saccharification: "Saccharification" is also called enzymatic hydrolysis. It is a process converting carbohydrate polymers (cellulose) to glucose and xylose monomers using cellulase enzymes.

sediment: Eroded material such as silt, sand, and gravel.

sedimentation: Soil particles, clay, sand, or other materials settle out of a fluid suspension into the bottom of a body of water.

short-rotation woody crop (SRWC): Fast-growing tree species grown on plantations and harvested in cycles shorter than is typical of conventional wood products, generally between three and 15 years. Examples include hybrid poplars (*Populus* spp.), willow (*Salix* spp.), and eucalyptus.

soil environmental quality: See **"soil quality".**

soil erosion: The movement and loss of soil by the action of wind or water or a combination thereof.

soil health: The continued capacity of soil to function as a vital living ecosystem that sustains plants, animals, and humans. See also **"soil quality"**.

soil organic matter (SOM): The organic fraction of the soil that includes plant and animal residues at various stages of decomposition, cells and tissues of soil organisms, and substances synthesized by the soil population.

soil quality: The capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation. In short, the capacity of the soil to function. See also **"soil health"**.

Soil Tillage Intensity Rating (STIR): A numerical index used to evaluate the degree of soil disturbance, with higher values associated with increased erosion risk**.**

splash blending: In contrast with match blending where blendstocks for oxygenate blending (BOBs) are blended with ethanol, "splash blending" is the process by with finished gasoline is blended with ethanol. Splash blending pre-dated match blending in areas not required to use reformulated gasoline (RFG), and generally occurred at the retail station or terminal.

spray irrigation (also known as **"sprinkler irrigation"**)**:** Application of water in a controlled manner that is similar to rainfall.

sprinkler irrigation (also known as **"spray irrigation"**)**:** Application of water in a controlled manner that is similar to rainfall.

sterols: Any of various solid steroid alcohols (such as cholesterol) widely distributed in animal and plant lipids.

sulfur oxides (SOx): Compounds of sulfur and oxygen molecule**s.**

Threatened and Endangered (T&E) species: Organisms in danger of extinction throughout all or a significant portion of its range (termed "Endangered"). "Threatened" means a species is likely to become endangered within the foreseeable future. See also **"Endangered Species Act (ESA)"**.

tillage: The mechanical disturbance of the soil for planting. The two main categories of tillage practices are conventional and conservation tillage. Conservation tillage includes practices such as no-till and mulch tillage. Tillage types can also be defined by Soil Tillage Intensity Rating (STIR) values. See also **"conventional tillage"**, **"conservation tillage"**, **"no-till"**, **"mulch tillage"**, and **"Soil Tillage Intensity Rating (STIR)".**

total renewable fuel: In the context of the Renewable Fuel Standard (RFS) Program, this is the total amount of biofuel mandated by the Program. Under the Renewable Fuel Standard 1 (RFS1), there was only one total renewable fuel standard. Under the Renewable Fuel Standard 2 (RFS2), nested within the volume requirement for total renewable fuel are three other volume requirements: advanced biofuel, biomass-based diesel, and cellulosic biofuel. See also **"Renewable Fuel Standard (RFS) Program"**, **"Renewable Fuel Standard 1 (RFS1)"** and **"Renewable Fuel Standard 2 (RFS2)"**.

transaction cost: The minimal costs of recording and trading Renewable Identification Numbers (RINs), roughly a few cents per RIN. See **"Renewable Identification Numbers (RINs)";** discussed in more detail in Chapter 6.

transesterification: In the context of biofuel, the chemical process that reacts an alcohol with triglycerides in vegetable oils and animal fats to produce biodiesel and glycerin.

transloader: A vehicle or mechanism that transfers goods from one mode of transportation (e.g., ship) to another (e.g., truck) such as a crane.

transmodal facility: A facility where goods are transferred from one mode of transportation (e.g., ship) to another (e.g., truck).

turbidity: A cloudy condition in water due to suspended silt or organic matter.

underground storage tanks (USTs): A tank and any underground piping connected to the tank that has at least 10 percent of its combined volume underground. United States (U.S.) Federal UST regulations apply only to systems storing either petroleum or certain hazardous substances.

vegetative reproduction: A form of asexual reproduction in plants by which new individuals arise without the production of seeds or spores. It can occur naturally or be induced by horticulturists.

volatile organic compounds (VOCs): per 40 CFR Part 51.100(s) is any compound of carbon, excluding carbon monoxide, carbon dioxide, carbonic acid, metallic carbides or carbonates, and ammonium carbonate, which participates in atmospheric photochemical reactions.

water availability: In the context of this report, "water availability" refers to the amount of water that can be appropriated from surface water sources (e.g., rivers, streams, lakes) or groundwater sources (e.g., aquifers) for consumptive uses.

water balance: Describes the state where inflows to any water system or area are equal to outflows plus the change in water storage during a specific time interval. See also **"hydropattern"**.

water consumption (also known as "**consumptive water use"**): Represents the part of water withdrawn that is evaporated, transpired, incorporated into products or crops, consumed by humans or livestock, or otherwise not available for immediate use.

water footprint: The volume of both direct and indirect water use required to produce a specific good, such as a volume of fuel, across the full supply chain. Life cycle assessment takes a similar approach to assessing the water use impacts, but often is done in comparison to alternative supply chains (such as petroleum-based fuels) and may include other resource issues and environmental impacts. The "water footprint" approach also differentiates between blue water (ground and surface water) and green water (rainwater) requirements.

water quality: A measure of the suitability of water for a particular use based on selected physical, chemical, and biological characteristics. "Water quality" is most frequently measured by characteristics of the water such as temperature, dissolved oxygen, and pollutant levels, which are compared to numeric standards and guidelines to determine if the water is suitable for a particular use.

water use: The total volume of water that can be estimated for a specific purpose. It can be used collectively as a term to represent withdrawals, deliveries, consumption and returns of water as it is moved through hydrologic and anthropogenically-designed systems.

water withdrawal: Water removed from the ground or diverted from a surface-water source for use.

watershed: The area drained by a stream, river, or other water body; typically defined by the topographic divides between one water body and another. Synonymous with catchment and drainage basin.

weed risk assessment: An evaluation of the likelihood that a plant will be introduced, escape, establish, spread, and cause harm.

wet milling: In the context of biofuel, a process for producing conventional corn starch ethanol in which the corn is soaked in water or dilute acid to separate the grain into its component parts (e.g., starch, protein, germ, oil, kernel fibers) before converting the starch to sugars that are then fermented to ethanol.

wetlands: Lands that are transitional between terrestrial and aquatic systems, where the water table is usually at or near the surface, or the land is covered by shallow water (e.g., from precipitation). "Wetlands" must have one or more of the following three attributes: (1) at least periodically, the land supports predominantly hydrophytes (i.e., plants that only grow in water); (2) the substrate is predominantly undrained, hydric soil (i.e., soils that are saturated, flooded, or ponded long enough to develop conditions for the growth and regeneration of hydrophytic vegetation); and (3) the substrate is nonsoil and is saturated with water or covered by shallow water at some time during the growing season of each year. There are many different classification systems for wetlands. Variation in methods can make results from different surveys (e.g., from the United States Department of Agriculture (USDA) and United States Fish and Wildlife Service (USFWS)) difficult or impossible to compare directly. See also **"deepwater habitats"**.

woody biomass: Tree biomass thinned from dense stands or cultivated from fast-growing plantations. This also includes small-diameter and low-value wood residue, such as tree limbs, tops, needles, and bark, which are often byproducts of forest management activities.

zooplankton: See **"macroinvertebrates"**.