United States Environmental Protection Agency

DRAFT
DO NOT CITE OR QUOTE

EPA/600/R-10/183A
January 2011
External Review Draft

Biofuels and the Environment:
First Triennial Report to Congress

NOTICE

THIS DOCUMENT IS A PRELIMINARY DRAFT. It has not been formally released by the U.S. Environmental Protection Agency and should not at this stage be construed to represent Agency policy. It is being circulated for comment on its technical accuracy and policy implications.

National Center for Environmental Assessment
Office of Research and Development
U.S. Environmental Protection Agency
Washington, DC

DISCLAIMER

This document is distributed solely for the purpose of pre-dissemination peer review under applicable information quality guidelines. It has not been formally disseminated by EPA. It does not represent and should not be construed to represent any Agency determination or policy. Mention of trade names or commercial products does not constitute endorsement or recommendation of use.

EXECUTIVE SUMMARY

This report is the first of the U.S. Environmental Agency's (EPA's) triennial reports to Congress required under the 2007 Energy Independence and Security Act (EISA). EISA requires EPA to revise the Renewable Fuel Standard (RFS) program to increase the volume of renewable fuel blended into transportation fuel from 9 billion gallons per year in 2008 to 36 billion gallons per year by 2022. The revised standards (RFS2), finalized in 2010, establish new specific annual volume requirements for cellulosic biofuel, biomass-based diesel, advanced biofuel, and total renewable fuel in transportation fuel.

EISA Section 204 calls for EPA to report to Congress on the environmental and resource conservation impacts of increased biofuel production and use, including air and water quality, soil quality and conservation, water availability, ecosystem health and biodiversity, invasive species, and international impacts. This report reviews impacts and mitigation tools across the entire biofuel supply chain, including feedstock production and logistics, and biofuel production, distribution, and use. The report focuses on:

- **Six feedstocks:** The two most predominantly used (*corn starch* and *soybeans*), and four others (*corn stover*, *perennial grasses, woody biomass*, and *algae*) that represent a range of feedstocks currently under development. Because the RFS2 limits the amount of corn starch-derived biofuel that counts toward the volume requirement in 2022 to 15 billion gallons, an increased reliance on other feedstocks is predicted.

- **Two biofuels:** *Ethanol* (both conventional and cellulosic) and *biomass-based diesel*, because they are the most commercially viable in 2010 and/or projected to be the most commercially available by 2022.

This first report represents peer-reviewed information available through July 2010 and reflects the current uncertainty about biofuel production and use. Quantitative assessments are presented, where possible, however, in most cases only qualitative assessments were feasible due to uncertainties and the lack of data and analyses in the peer-reviewed literature. Conclusions, which do not account for existing or potential future mitigation measures or regulations, include:

- **Life Cycle Assessment.** Some segments of the biofuel life cycle result in greenhouse gas (GHG) emissions; however, as noted in EPA's RFS2 Regulatory Impact Analysis, when the entire biofuel life cycle is considered, the EISA-mandated revisions to the RFS2 program are expected to achieve a 138-million metric ton reduction in carbon dioxide-equivalent emissions by 2022 compared to continued reliance on petroleum-based fuels.

- **Water Quality.** Ground and surface water quality can be impacted by erosion and runoff of fertilizers and pesticides when feedstocks, particularly row crops such as corn, are cultivated for biofuel; through pollutants in the wastewater discharged from biofuel production facilities; and from leaks and spills during fuel transport.

- **Water Quantity.** Effects of feedstock production on water availability vary greatly by feedstock, processes used to produce the feedstock, and location. Depending on location, the amount of water required to grow corn and soybean for biofuel can be far greater than that required to grow perennial grasses, woody biomass, and algae. Water used by biofuel production facilities is modest compared to that required to produce biofuel feedstocks, and impacts depend on the location of the facility in relation to water resources.

- **Soil Quality.** Increased cultivation of biofuel feedstocks is likely to affect soil quality in various ways, depending on the feedstock. Some feedstocks may contribute to detrimental effects, including increased soil erosion, decreased soil organic matter content, increased soil GHG emissions, and increased nitrogen and phosphorus losses to ground and surface waters. Other feedstocks may contribute to advantageous effects such as increased soil carbon and reduced erosion.

- **Air Quality.** Air quality may be impacted by pollutants from feedstock production, such as farm equipment emissions and soil/dust particles made airborne during field tillage and fertilizer application; by emissions from combustion equipment used for energy production at biofuel production facilities; and by evaporative and tailpipe emissions from combustion in vehicles.

- **Ecosystem.** Increased cultivation of feedstocks for biofuel could significantly affect biodiversity through habitat alteration when uncultivated land is put into production; from exposure of flora and fauna to pesticides; or through sedimentation and eutrophication in water bodies resulting from soil erosion and nutrient runoff, respectively. Invasiveness potential of cultivated feedstocks is also a concern, but varies by feedstock.

- **International.** Increases in U.S. biofuel production and consumption volumes will affect many different countries as trade patterns and prices adjust in response to global supply and demand. This will result in land use change and effects on air quality, water quality, and biodiversity. Direct and indirect land use changes will likely occur across the globe as the U.S. and other biofuel feedstock-producing countries alter their agricultural sectors to allow for greater biofuel production. Many locations where biofuel production is growing, such as Indonesia, Malaysia, and Brazil, are also areas of high biodiversity value. Depending where biofuel feedstock production occurs, and to what extent the level of production increases with time, impacts to biodiversity could be significant.

Most activities, processes, and products associated with the biofuel supply chain are already regulated, subject to limitations, or mitigated through various approaches. To further address adverse impacts, EPA recommends:

- Developing and evaluating Environmental Life Cycle Assessments for biofuels.

- Ensuring the success of current and future environmental biofuel research through improved cooperation and sustained support.

- Improving the ability of federal agencies (within their existing authorities) to develop and implement best management and conservation practices and policies that will avoid or mitigate negative environmental effects from biofuel production and use.

- Engaging the international scientific community in cooperative efforts to identify and implement sustainable biofuel practices that minimize environmental impact.

Because biofuel impacts cross many topics and Agency responsibilities, EPA likely will address these recommendations through continued and strengthened cooperation with federal agencies and international partners, including the U.S. Departments of Agriculture and Energy.

Table of Contents

Page

TABLE OF CONTENTS (Continued)

Page

TABLE OF CONTENTS (Continued)

Page

TABLE OF CONTENTS (Continued)

Page

TABLE OF CONTENTS (Continued)

Page

LIST OF TABLES

Page

LIST OF FIGURES

Page

1. INTRODUCTION

In December 2007, Congress enacted the Energy Independence and Security Act (Public Law 110-140) (EISA) to reduce U.S. energy consumption and dependence on foreign oil, and to address climate change through research and implementation of strategies to reduce greenhouse gases. Accordingly, EISA requires the U.S. Environmental Protection Agency (EPA) to revise the Renewable Fuel Standard (RFS) program, created under the 2005 Energy Policy Act,[a] to increase the volume of renewable fuel[b] required to be blended into transportation fuel from 9 billion gallons per year in 2008 to 36 billion gallons per year by 2022.

EPA finalized revisions to the RFS program in February 2010. The revised statutory requirements (commonly known as the RFS2) establish new specific annual volume standards for cellulosic biofuel, biomass-based diesel, advanced biofuel, and total renewable fuel that must be used in transportation fuel (see Chapter 2). Meeting RFS2 in 2022 will result in biofuels making up an estimated 7 percent of fuels (by volume) used for transportation (U.S. EPA, 2010a). The purpose of this report is to examine the environmental and resource conservation impacts of this change, as required under EISA Section 204.

EISA Section 204 calls for EPA to report to Congress every three years on the environmental and resource conservation impacts of increased biofuel production and use as follows:

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[c] *on the following:*

1. *Environmental issues, including air quality, effects on hypoxia, pesticides,*[d] *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.*
4. *…. The report shall include the annual volume of imported renewable fuels and feedstocks for renewable fuels, and the environmental impacts outside the United*

[a] The 2005 Energy Policy Act amended the Clean Air Act and established the first national renewable fuel standards. The statute specifies the total volume of renewable fuel that is to be used based on the volume of gasoline sold in the U.S. each year, with the total volume of renewable fuel increasing over time to 7.5 billion gallons in 2012.

[b] To be considered "renewable," fuels produced by biorefineries constructed after EISA's enactment on December 19, 2007, must generally achieve at least a 20 percent reduction in life cycle greenhouse gas emissions compared to petroleum fuels.

[c] EISA 2007 amended Section 211(o) of the Clean Air Act to include the definitions and requirements of RFS2.

[d] Pesticides include antimicrobials, fungicides, herbicides, insecticides, and rodenticides.

> *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 is the first of EPA's triennial reports on the current and potential future environmental impacts associated with the requirements of Section 211(o) of the Clean Air Act. This report reviews environmental and resource conservation impacts, as well as mitigation tools to reduce these impacts, across major components of the biofuel supply chain: feedstock production, feedstock logistics, biofuel production, biofuel distribution, and biofuel use.

This report emphasizes domestic impacts; however, the substantial market created for biofuels by the U.S, Brazil, and other countries has important global implications. For example, countries that produce feedstocks, now or in the future, which are converted to biofuels that qualify for use in the U.S. will experience direct impacts; other countries will have to adapt to changing agricultural commodity distributions that result from diversion of food exports to biofuel production. As required under EISA Section 204, this report describes the impacts of increased feedstock and biofuel production in other countries as a result of U.S. policy.

This first triennial Report to Congress represents the best available information through July 2010 and reflects the current understanding about biofuel production and use, including input from the U.S Departments of Agriculture and Energy, with whom EPA consulted during development of this report. Quantitative assessments are presented, where possible, using 2010 or the most recently available data; however, in most cases only qualitative assessments were feasible due to uncertainties and lack of data and analyses in the peer-reviewed literature. Future reports will reflect the evolving understanding of biofuel impacts in light of new research results and data as they become available. This initial report to Congress serves as a starting point for future assessments and for taking action to achieve the goals of EISA.

2. BACKGROUND AND APPROACH

2.1 EISA and RFS2 Requirements for Biofuel Production and Use

RFS2 (the Renewable Fuel Standard as amended by the Energy Independence and Security Act [EISA]) establishes new specific annual volume standards for four categories of renewable fuels that must be used in transportation fuel[5]: cellulosic biofuel, biomass-based diesel, advanced biofuel, and total renewable fuel (see *Glossary* in Appendix A for fuel definitions). Under RFS2, conventional biofuel (i.e., ethanol derived from corn starch) with a maximum volume target and "additional renewable fuels"[6] are included as eligible fuels to meet the total renewable fuel standard. The revised statutory requirements also include new definitions and criteria for both renewable fuels and the feedstocks used to produce them,[7] including new greenhouse gas emission (GHG) reduction thresholds (as determined by the life cycle assessment that EPA conducted as part of its Regulatory Impact Analysis [RIA] during the final RFS2 rulemaking).

Table 2-1 shows the RFS2 annual renewable fuel standards through 2022. Total renewable fuel under the standard will increase to 36 billion gallons per year (bgy) by 2022 (of which corn starch ethanol is not to exceed 15 bgy).

While EISA establishes the renewable fuel volumes shown in Table 2-1, it also requires the EPA Administrator each November to set the volume standards for the following year based in part on information provided by the Energy Information Administration (EIA) and other data indicating the commercial capacity for producing cellulosic biofuels. EISA therefore requires the EPA Administrator to adjust the cellulosic standard, and potentially the total advanced biofuel and total renewable fuel standards, each year based on this assessment. For 2010, the Administrator adjusted the cellulosic standard from 0.1 bgy (100 million gallons per year) in RFS2 to 5.0 million gallons, but did not adjust the total advanced or total renewable fuel standard.[8] Therefore, the final 2010 standard for total renewable fuel is set at 12.95 bgy, with specific targets for cellulosic biofuel (5.0 million gallons per year), biomass-based diesel (1.15 bgy [combining the 2009 and 2010 standards as proscribed in RFS2]), and total advanced biofuel (0.95 bgy).

[5] Transportation fuel includes fuels used in motor vehicles, motor vehicle engines, non-road vehicles, or non-road engines (except for ocean-going vessels).

[6] EISA defines "additional renewable fuel" as "fuel produced from renewable biomass that is used to replace or reduce fossil fuels used in heating oil or jet fuel." Though RFS2 does not specify a volume standard for this fuel category, it does allow renewable fuel blended into heating oil or jet fuel to count toward achieving the standard for total renewable fuel. (This contrasts with the original RFS [RFS1], which did not provide credit for renewable fuel blended into non-road fuel.) More information about "additional renewable fuel" can be found in Section II.b.e of the final RFS2 rule available at http://www.epa.gov/otaq/fuels/renewablefuels/regulations.htm.

[7] EISA requires that all renewable fuel be made from feedstocks that meet the new definition of renewable biomass, which includes certain land use restrictions. For full details, see Section 3.1.

[8] Although EISA specified a 2010 cellulosic biofuel requirement of 100 million gallons/year, as shown in Table 2-1, EPA determined that this level was not achievable for 2010. EIA projected 5 million gallons/year of cellulosic production for 2010 (6.5 million gallons ethanol equivalent), and EPA accepted this as the 2010 standard. While this is lower than the level specified in EISA, no change to the advanced biofuel and total renewable fuel standards was warranted due to the inclusion of an energy-based equivalence value for biodiesel and renewable diesel.

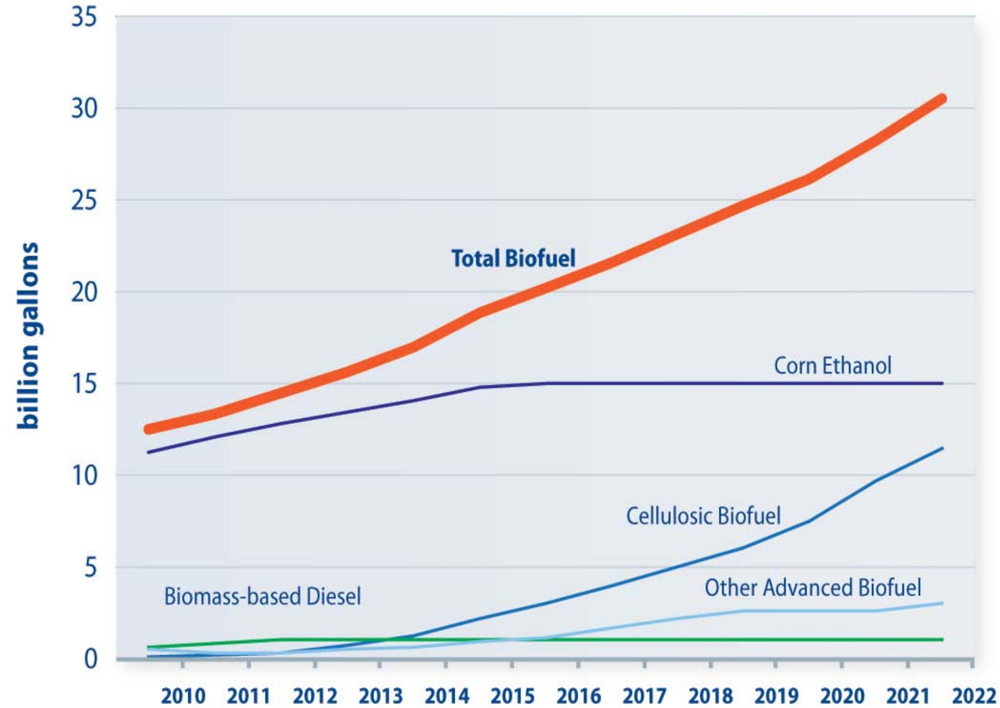
Table 2-1: RFS2 Renewable Fuel Requirements (billion gallons per year) [a, b]

Year	Renewable Fuel				
		Advanced Biofuel			
	Conventional Biofuel	Cellulosic Biofuel	Biomass-Based Diesel	Advanced Biofuel [c]	Total Renewable Fuel
2008	9.0	n/a	n/a	n/a	9.0
2009	10.5	n/a	0.5	0.6	11.1
2010	12.0	0.1 [d]	0.65	0.95	12.95
2011	12.6	0.25	0.80	1.35	13.95
2012	13.2	0.5	1.0	2.0	15.2
2013	13.8	1.0	TBD [e]	2.75	16.55
2014	14.4	1.75	TBD [e]	3.75	18.15
2015	15.0	3.0	TBD [e]	5.5	20.5
2016	15.0	4.25	TBD [e]	7.25	22.25
2017	15.0	5.5	TBD [e]	9.0	24.0
2018	15.0	7.0	TBD [e]	11.0	26.0
2019	15.0	8.5	TBD [e]	13.0	28.0
2020	15.0	10.5	TBD [e]	15.0	30.0
2021	15.0	13.5	TBD [e]	18.0	33.0
2022	15.0	16.0	TBD [e]	21.0	36.0

a – The requirements for cellulosic biofuel, biomass-based diesel, advanced biofuel, and total renewable fuel are minimum required volumes that must be achieved and may be exceeded. The conventional biofuel requirement is a cap that cannot be exceeded.
b – Note that the RFS2 volume requirements are nested: cellulosic biofuel and biomass-based diesel are forms of advanced biofuel; and advanced biofuel and conventional biofuel are forms of total renewable fuel.
c – Note that the sum of the required amounts of cellulosic biofuel and biomass-based diesel is *less* than the required volume of advanced biofuel. The additional volume to meet the advanced fuel requirement may be achieved by the additional cellulosic biofuel and biomass-based diesel (i.e., beyond the required minimum) and/or by other fuels that meet the definition of advanced biofuel (e.g., sugarcane ethanol).
d – As described above, and as allowed under EISA, the EPA Administrator determined that original RFS2 standard of 0.1 bgy for cellulosic biofuel was not achievable for 2010 and therefore decreased this standard to 5 million gallons for 2010.
e – To be determined by EPA through a future rulemaking, but no less than 1.0 billion gallons. This requirement was designated under EISA as “to be determined” with a minimum requirement because of the uncertainty about future capacity to produce fuel that meets the biomass-based diesel definition.
Source: U.S. EPA, 2010b.

2.2 Projected Fuel and Feedstock Use to Meet Required RFS2 Targets through 2022

Figure 2-1 summarizes the fuel types and volumes *projected* to meet the required targets through 2022, as estimated in the RFS2 Regulatory Impact Analysis. Although actual volumes and feedstocks will likely be different, EPA believes the projections are within the range of expected outcomes when the standards are met (U.S. EPA, 2010b).



Source: U.S. EPA, 2010b.

Figure 2-1: Projected Renewable Fuel Volumes to Meet RFS2 Targets

In 2009, corn ethanol constituted 95 percent of total U.S.-produced renewable fuel, with biodiesel made from soybean oil, other virgin vegetable oils, rendered fats, greases, and corn oil from ethanol production accounting for almost all the remaining biofuel consumed (FAPRI, 2010a; EIA, 2010). However, as technologies improve, EPA expects more advanced cellulosic feedstocks, such as agricultural residues (e.g., corn stover, sugarcane bagasse, wheat residue, sweet sorghum pulp), forestry biomass, urban biomass waste, and dedicated energy crops (e.g., switchgrass) to produce biofuels (Figure 2-2) (U.S. EPA, 2010b). Present research is focused on improving technologies to convert different feedstocks to biofuels in an economically viable manner, and determining sustainable biofuel production methods.

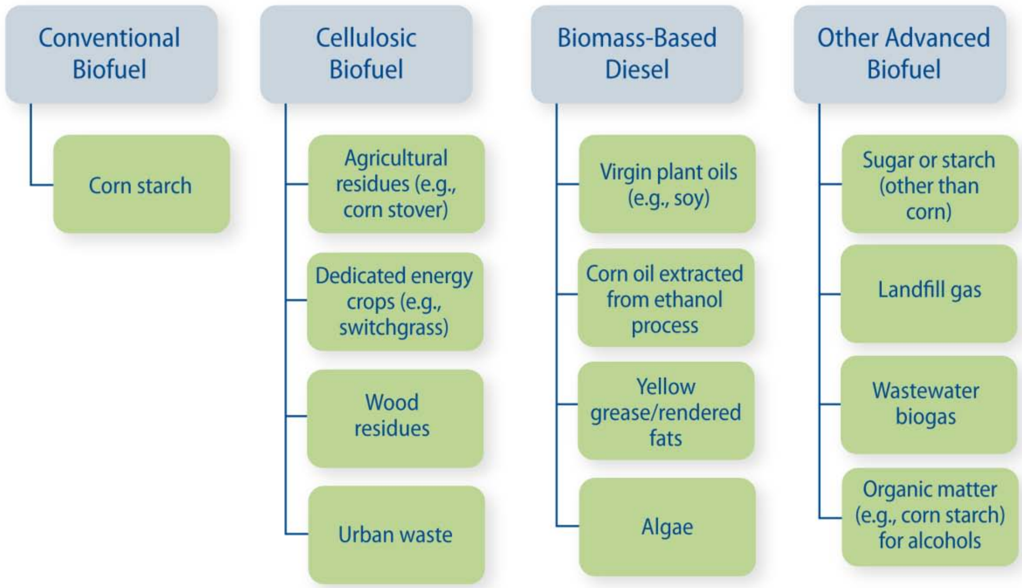


Figure 2-2: Examples of Feedstocks Available for Biofuel Production

With respect to biodiesel, EPA expects continued use of soybean oil, which made up 54 percent of feedstock used for biodiesel in 2009 (EIA, 2010), as well as a varying percentage of other vegetable oils, rendered fats, greases, and corn oil from ethanol production through 2022 (see Table 4-1 for a more detailed breakdown) (U.S. EPA, 2010b). Algae could potentially provide large volumes of oil for the production of biomass-based diesel. However, several hurdles, including technical issues, will likely limit production volumes within the 2022 timeframe (U.S. EPA, 2010b).

Imported sugarcane ethanol, also represents a significant potential supply of biofuel by 2022 (U.S. EPA, 2010b). In 2009, the United States imported 198 million gallons of ethanol (EIA, n.d. [d]). Import volumes are expected to grow as U.S. demand increases to meet the biofuel targets.

2.3 Assessment of the Environmental and Resource Conservation Impacts of Biofuels

2.3.1 Approach

This report presents a comprehensive survey of environmental evaluations across the biofuel supply chain (see below), including current and anticipated future feedstock production and logistics and biofuel production, distribution, and use. It summarizes much of the available information and identifies research needed to evaluate potential environmental impacts from a life cycle perspective and quantify them using more substantive and systematic assessment tools. This report therefore is the first step towards conducting a biofuels environmental life cycle assessment (LCA), which EPA will conduct for its future Reports to Congress (see Chapter 7).

Life cycle assessment evaluates environmental impacts resulting from all stages of a product's development—from extraction of fuel for power to production, marketing, use, and disposal. EPA has begun to collaborate with partners and stakeholders to formulate specific questions, establish boundaries, and identify critical assessment endpoints to be used in modeling the input and output data for comprehensively assessing potential impacts across the biofuel supply chain and integrating environmental risk assessment (ERA) tools. Although this report does not attempt a comprehensive biofuels environmental LCA, as part of the EISA-mandated revisions to the RFS program, EPA conducted a life cycle assessment of GHG emissions from increased renewable fuels use, which projected a 138-million metric ton reduction in CO_2-equivalent emissions by 2022 (U.S. EPA, 2010b). Section 4.3.2.3 provides more details about the LCA methodology and results. This work, which will provide the foundation for future versions of this Report to Congress, will draw from the considerable work that has already been done to develop LCA and other methodologies, including ecological and human health risk assessment, to assess impacts of specific biofuel products and processes.

2.3.2 Biofuel Supply Chain

There are five main stages in the biofuel supply chain: feedstock production, feedstock logistics, fuel production, fuel distribution, and fuel use (Figure 2-3). The specific impacts associated with a particular feedstock or biofuel will vary depending on many factors, including the type, source, and method of feedstock production; the technology used to convert the feedstock to fuel; methods used and distances traveled to transport biofuels; the types and quantities of biofuels used; and controls in place to avoid or mitigate any impacts.

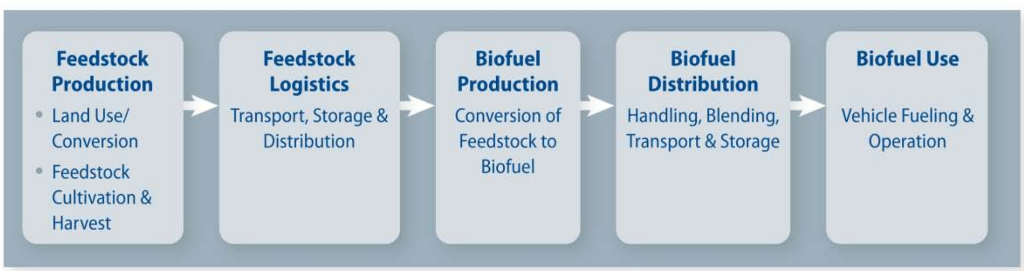


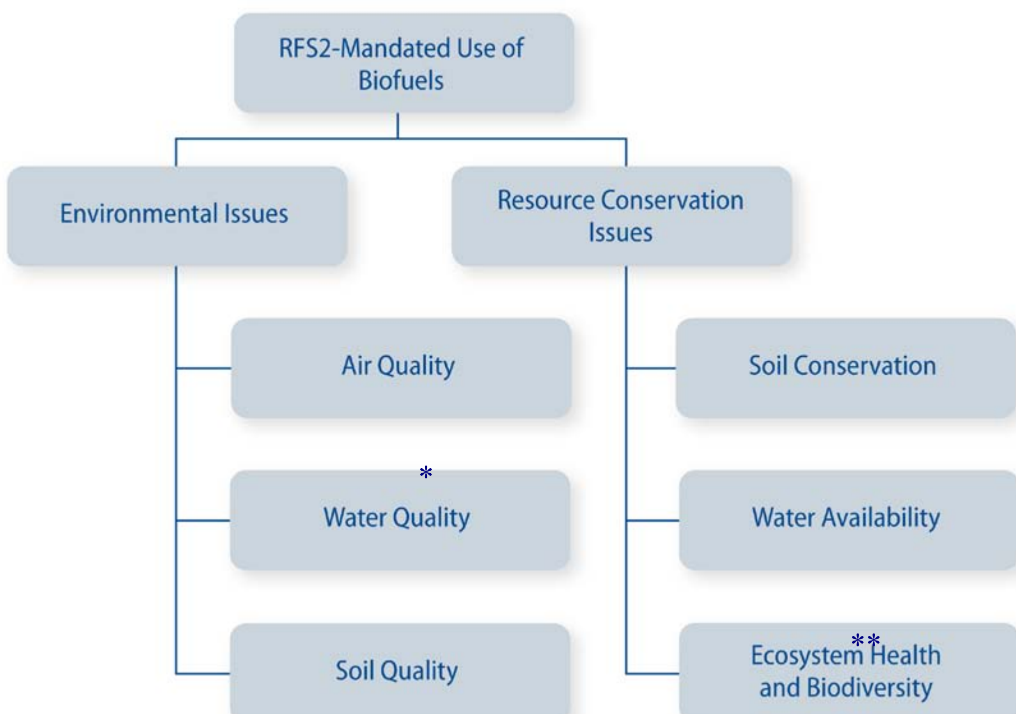
Figure 2-3: Five Stages of the Biofuel Supply Chain

2.3.3 Feedstocks and Fuels Discussed in This Report

There is uncertainty regarding which feedstocks will be used to meet the RFS2 targets in the mid- to long-term time horizon. A few feedstocks are already in use, including primarily corn and soybean, as well as others in smaller quantities. Other feedstocks are in the early stages of research and development or their potential future commercial viability is still unknown. This report focuses on six feedstocks: the most predominantly used (corn and soybeans) and four others (corn stover, perennial grasses, woody biomass, and algae) that represent a range of feedstocks currently under development. The biofuels highlighted in this report are ethanol (both conventional and cellulosic) and biomass-based diesel. Ethanol and biomass-based diesel are the focus because they are currently the most commercially viable and/or are projected to be the most commercially available by 2022, and they are the primary fuels currently projected to meet RFS2. Future reports will analyze other feedstocks and fuels.

2.3.4 Impacts Discussed in This Report

This report focuses on specific environmental and resource conservation impacts specified in EISA Section 204, as shown in Figure 2-4 and described in Tables 2-2 and 2-3. This report does not include extensive discussion of carbon dioxide or other greenhouse gas emissions; interested readers are referred to the EPA's RFS2 Regulatory Impact Analysis (U.S. EPA, 2010b). A short discussion is provided in Section 4.3.2.3 of this report. The environmental and resource conservation impacts discussed in this report reflect a complex set of interactions and feedbacks between land, soil, air, and water; future versions of this report will explore analysis of these important complexities as enhanced data and analysis tools become available. This report does not attempt to conduct a quantitative analysis of the range of impacts associated with increased production of biofuel. Instead, it represents a compilation of available information and analyses that can inform the nature and extent of impacts that might be expected to occur. Thus, this report does not use a baseline year, per se, against which future impacts can be measured. Different impacts have been assessed using applicable baselines cited in the literature or, as appropriate, in the RFS2 RIA. Generally, however, the primary reference point used in this document is consistent with the primary reference case used in the RFS2 RIA. This reference case is a projection made by the U.S. Energy Information Administration prior to EISA in their *2007 Annual Energy Outlook* of renewable fuel volumes expected in 2022.



* Includes pesticides, sediments, nutrients, pathogens, and acreage/function of wetlands
** Includes invasive/noxious plants, forests, grasslands, wetlands, and aquatic ecosystems

Figure 2-4: Environmental and Resource Conservation Issues Addressed in This Report

Chapter 3 focuses on feedstock production, including cultivation and harvest. Chapter 4 covers impacts of feedstock logistics and biofuel production, distribution, and use. Many activities, processes, and products associated with the biofuel supply chain are already regulated, are subject to limitations, or are mitigated through various approaches, as discussed in these chapters. The potential impacts associated with imported biofuels are discussed in Chapter 5. Currently, imported ethanol and biodiesel supply a relatively small percentage of U.S. biofuel consumption—approximately 9 percent in 2008 (EIA, 2009, n.d.[a]; U.S. ITC, 2010; ERS, 2010a). If these percentages increase, future versions of this report may provide expanded analysis of international impacts associated with imported biofuels.

EPA's ability to assess environmental and resource conservation impacts is limited by uncertainties associated with even a qualitative assessment of the direct impacts. Many feedstock technologies are in the early stages of research and development, therefore empirical and monitoring data relevant to environmental impacts are limited and projections of their potential future use are highly speculative. Recommendations in Chapter 6 and the approach to future assessments described in Chapter 7 address how the EPA intends to bolster data availability and analysis to improve understanding of environmental impacts in future reports.

Table 2-2: Overview of Environmental and Resource Conservation Impacts Addressed in This Report [a]

	Feedstock Production and Transportation	Fuel Production, Distribution, and Use
Water Quality	• Pollution of ground, surface, and drinking water due to runoff containing sediments, nutrients, pesticides, and metals • Loss of aquatic habitat due to pollution and sedimentation. • Water quality impacts of converting pasture or marginal or non-cultivated land to feedstock production	• Contamination of surface, ground, and drinking water by wastewater from biofuel production facilities and from leaks and spills during fuel transportation and storage
Water Availability	• Reduced availability of local or regional water due to withdrawals of water needed to irrigate feedstocks • Loss of aquatic habitat due to lowered stream flow	• Lowered stream flow and aquifer levels due to water withdrawals for biofuel conversion. • Reduced availability of water due to contamination (see above)
Soil Quality and Soil Conservation	• Degradation in soil quality due to (1) changes in land use; (2) increased use of nutrients, pesticides, and tillage and (3) harvesting of agricultural and forest residue • Soil contamination from use of pesticides	• Soil contamination from leaks and spills during fuel transportation and storage • Addition of methane to soil gas resulting from biodegradation of spilled biofuel
Air Quality	• Emissions of criteria pollutants, air toxics, and greenhouse gases by farm and transportation vehicles • Fugitive dust from feedstock production operations	• Emissions of criteria pollutants, air toxics, and greenhouse gases during conversion and by transportation vehicles and off-road equipment

Table 2-2: Overview of Environmental and Resource Conservation Impacts Addressed in This Report [a]

	Feedstock Production and Transportation	Fuel Production, Distribution, and Use
Ecosystem Health/Biodiversity (including invasive and noxious plants)	• Impacts on flora and fauna and loss of ecosystem services due to pollution and habitat changes • Establishment and spread of invasive or noxious plants	• Establishment and spread of invasive or noxious plants

a – The impacts in this table are generalized and do not take into account location or effectiveness of mitigation practices.

Table 2-3: Overview of Environmental and Resource Conservation Impacts on Specific Ecosystems Addressed in This Report [a]

	Feedstock Production
Forests	• Short rotation woody crop (SRWC) plantations may deplete soil nutrients over the long run, but appropriate management techniques may increase soil nutrients. • SRWC plantations can sustain high species diversity, although bird and mammal species tend to be habitat generalists. • Some tree species under consideration as feedstocks may invade forests in certain locations. • Forest thinning can reduce the threat of catastrophic wildfires. • Forest thinning can increase nutrient availability in soils over the short term. • Harvesting forest residues decreases nutrient availability, soil organic matter, and habitat for some forest species.
Grasslands	• Conversion of grasslands to row crops impacts grassland-obligate species, potentially leading to declines in wildlife habitat. • Higher proportions of corn within grassland ecosystems leads to fewer grassland bird species. • Growing more switchgrass may improve grassland habitat for some species depending on management regimes. • Conversion of Conservation Reserve Program lands to perennial grasses or harvesting of existing grasslands is likely to have low impacts on grassland species, particularly if harvesting occurs after the breeding season. • Use of native mixtures of perennial grasses can restore some native biodiversity. • Cultivation of switchgrass outside of its native range may lead to invasions of native grasslands. • Cultivation of *Miscanthus* may lead to invasions of pasture and other grasslands.

Table 2-3: Overview of Environmental and Resource Conservation Impacts on Specific Ecosystems Addressed in This Report [a]

	Feedstock Production
Wetlands	• Increased sediments, nutrients, pesticides, and pathogens from runoff can flow into downstream wetlands. • Increased nutrient loadings can lead to changes in wetlands community structure. • Reduced sediment and nutrient loadings can lead to improved water quality, depending on the specific management practice used. • Some grass species under consideration may invade wetlands, including giant reed (*Arundo donax*) and reed canary grass (*Phalaris arundinacea*) . • Harvesting forest residues and forest thinning may increase nutrient loads, depending on slopes, soils, presence of buffer zones, and use of best management practices to reduce runoff. • Algal strains created may escape from cultivation and invade wetlands.

a – The impacts in this table are generalized and do not take into account location or effectiveness of mitigation practices.

2.4 Regulatory Authority Relevant to Biofuel Environmental Impacts

EPA, as well as states, tribes, and local environmental agencies, has statutory responsibility to assess and control air emissions, water discharges, use of toxic substances, microbial and pesticide use, and waste disposal. Many existing environmental regulations and programs are applicable to the biofuel supply chain, including feedstock production and logistics; biofuel production and distribution, and biofuel use.

EPA's primary federal regulatory authority is derived from the Clean Air Act (CAA); the Clean Water Act (CWA), the Federal Insecticide Fungicide and Rodenticide Act (FIFRA); Resource Conservation and Recovery Act (RCRA) and the Toxic Substances Control Act (TSCA). Under the CAA, EPA has broad direct statutory authority to regulate fuel quality and emissions from refining and production facilities for all fuels, including biofuels. The CAA also establishes limits for mobile source (vehicular) emissions. The Clean Water Act requires permits for point source discharges to waters of the U.S, and development of water quality standards, and Total Maximum Daily Loads (TMDLs) for water bodies where water quality standards have not been met. FIFRA establishes standards for storage and use of pesticides in a manner that does not harm human health or the environment. RCRA governs the generation, storage, treatment, transport, and disposal of hazardous waste. TSCA requires manufacturers and importers of new chemicals to submit "pre-manufacture" notices for EPA review prior to manufacture and commercial use of new chemicals, including new fuels, new biological materials, and new genetically engineered microorganisms used to produce biofuels or co-products. Through the CWA's Spill Prevention, Control and Countermeasure rule, EPA has enforceable regulations to control water quality impacts from spills or leaks of biofuel products and by-products. In addition, the Safe Drinking Water Act establishes maximum contaminant levels (MCLs) for more than 90 drinking water contaminants to ensure public health. These statutes provide opportunities within the existing regulatory framework to regulate and mitigate some of the potential adverse health and environmental effects of biofuels. Selected environmental laws relevant to the production and use of biofuels are summarized in Appendix B.

Generally, EPA headquarters offices develop policies and regulations for these federal statutes, while regional EPA offices, in partnership with the states and tribes, implement these programs, ensure compliance, and enforce regulations. EPA and its regional offices work closely with states and tribes to review permit applications for new facilities and to monitor environmental impacts to ensure compliance with all permit conditions. EPA has prepared two documents to help biofuel facilities understand the full range of regulatory requirements (U.S. EPA, 2007a, 2008a). While EPA has oversight authority for federal environmental regulatory programs and regulations, state agencies and tribes are often "delegated" the responsibility for issuing permits, conducting inspections, ensuring compliance, and taking enforcement action. EPA regulations establish minimum requirements. States can enact more stringent standards, although several states have enacted legislation that prohibits adopting requirements stronger than those set by EPA.

3. ENVIRONMENTAL IMPACTS OF SPECIFIC FEEDSTOCKS

3.1 Introduction

The Energy Independence and Security Act (EISA) requires that all renewable fuel be made from feedstocks that meet the definition of renewable biomass (see textbox). Many different feedstocks meet these requirements and can be used to produce ethanol, other biofuels or biofuel components.

In 2009, 95 percent—or 10.9 billion gallons—of total renewable fuel produced in the U.S. was produced from corn and refined almost entirely in the form of conventional corn starch ethanol (FAPRI, 2010a). Soybean oil-based biodiesel accounted for most of the remainder—505 million gallons. EPA expects that corn and soybean feedstocks will continue to account for a large share of biofuel production in the U.S. (U.S. EPA, 2010b) in the near future. As of July 2010, there was no significant commercial-scale production of ethanol from cellulosic or hemicellulosic feedstocks, nor was there significant biodiesel production from oil seed feedstocks other than soybean in the U.S.

As science and technology improves, EPA expects an increase in the use of cellulosic feedstocks to produce advanced biofuel. Such feedstocks include agricultural residues (e.g., corn stover, sugarcane bagasse, sweet sorghum pulp), forestry biomass, urban waste, and dedicated energy crops (e.g., switchgrass) (U.S. EPA, 2010b). Technologies for producing biodiesel from vegetable oils, recycled oils, rendered fats, greases, and algal oils have been developed and tested at various scales from the laboratory to demonstration plants and semi-commercial facilities. EPA expects biodiesel from these feedstocks to gain a wider market share as their production becomes more economically and technologically feasible (U.S. EPA, 2010b).

The feedstocks discussed in this chapter include corn and soybeans, as well as four others currently under development: corn stover, perennial grasses, woody biomass, and algae (see Table 3-1). These feedstocks represent different cultivation and production practices.

Requirements for Renewable Fuels

Under the Energy Independence and Security Act, all renewable fuel must be made from feedstocks that meet the EISA definition of renewable biomass, which includes:

- Planted crops and crop residue from agricultural lands that were cleared prior to December 19, 2007, and were actively managed or fallow on that date.
- Planted trees and tree residue from tree plantations that were cleared prior to December 19, 2007 and were actively managed on that date.
- Animal waste material and by-products.
- Slash and pre-commercial thinnings from non-federal forestlands that are neither old-growth nor listed as critically imperiled or rare by a State Natural Heritage program.
- Biomass cleared from the vicinity of buildings and other areas at risk of wildfire.
- Algae.
- Separated yard waste and food waste.

Currently, as described in the final RFS2 rule, EPA deems renewable fuel producers using domestically grown crops and crop residue as feedstock to be in compliance with the renewable biomass requirements. However, EPA will annually review U.S. Department of Agriculture (USDA) data on lands in agricultural production to determine if these conclusions remain valid.

This chapter reviews the actual (where known) and potential environmental impacts of producing these six feedstocks, including impacts on water quality and quantity, soil and air quality, and ecosystem health/biodiversity. Feedstock production impacts are considered during the cultivation and harvest processes (see Figure 2-3). Impacts associated with the subsequent four stages of the biofuel supply chain are presented in Chapter 4. Row crop feedstocks (corn, corn stover, and soybean), which share many commonalities, are discussed in Section 3.2. Sections 3.3 to 3.5 present potential effects associated with switchgrass, woody biomass, and algae.

Table 3-1: Primary Fuels and Feedstocks Discussed in this Report

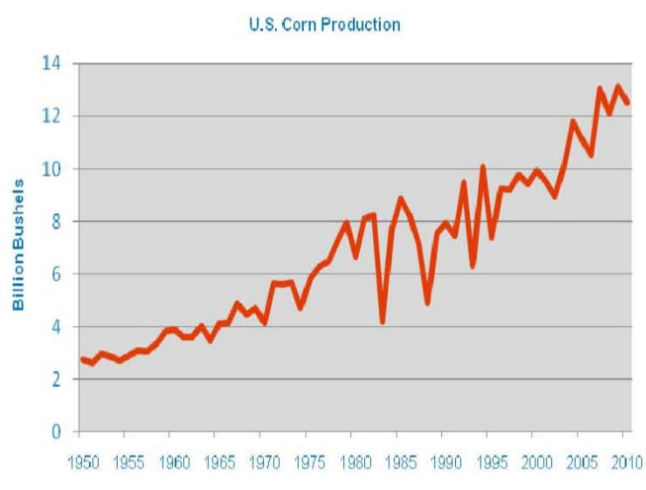
EISA Biofuel Type	Biofuel	Feedstock
Conventional Biofuel	Ethanol	Corn Starch
Cellulosic Biofuel	Ethanol	Corn Stover
		Perennial Grasses
		Woody Biomass
Biomass-Based Diesel	Biomass-Based Diesel	Soybeans
		Algae

In addition to the six primary feedstocks examined in this report, Section 3.6 briefly discusses waste materials as potential emerging feedstocks for biofuels. In addition to general ecosystem impacts, Section 3.7 reviews impacts on specialized habitats (forests, grasslands, and wetlands), as required under EISA Section 204. Section 3.8 reviews environmental concerns associated with genetic engineering of feedstocks, commonly referred to as genetically modified organisms (GMOs).

3.2 Row Crops (Corn, Corn Stover, Soybeans)

3.2.1 Introduction

U.S. corn and soybean production have increased steadily over the past several decades. Increased demand for biofuel provides additional incentive to continue research and development for increasing crop yields. As shown in Figure 3-1, U.S. corn production increased by more than a factor of four between 1950 and 2010. These increases were largely due to gains in efficiency and crop yield. Soybean yields have also increased. For example, soybean yields increased from 21.7 bushels per acre in 1950 to 43.9 bushels per acre in 2010 (NASS, 2010a).



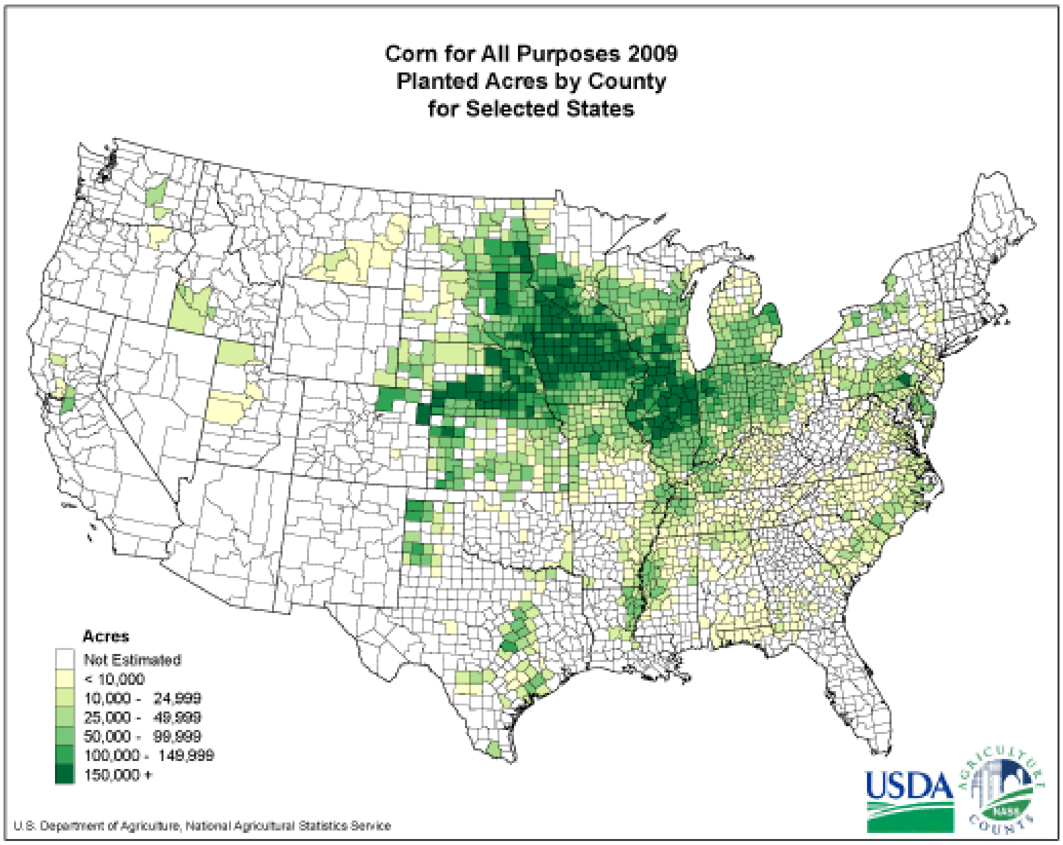
Source: NASS, 2010a.

Figure 3-1: U.S. Corn Production

Actual environmental impacts will vary, depending on the number of acres in production, cropping techniques, implementation of conservation practices, and the location of the crop acreage including hydrology, soils, and other geographic factors.

3.2.1.1 Current and Projected Cultivation

In 2009[9], U.S. farmers planted 86 million acres of corn, harvesting 13.1 billion bushels (NASS, 2010a). Approximately 4.6 billion bushels of corn from the 2009 harvest were used to produce corn ethanol. In 2010, U.S. farmers planted 88 million acres of corn, harvesting 12.5 billion bushels (NASS, 2010a). Approximately 4.8 billion bushels (or 38.4 percent) of corn are projected to be used to produce corn starch ethanol biofuel between September 2010 and August 2011 (ERS, 2010c), up from 11.2 percent in the 2004-2005 harvest year (ERS, 2010b, 2010c). Corn is grown throughout the U.S., but the vast majority of the crop is grown in 12 states: Illinois, Indiana, Iowa, Kansas, Kentucky, Michigan, Minnesota, Missouri, Nebraska, Ohio, South Dakota, and Wisconsin. Figure 3-2 shows a map of planted acres by county (for selected states) in 2009.



Source: NASS, 2010b.

Figure 3-2: Planted Corn Acres by County, for Selected States (2009)

[9] As of January 2011, 2009 was the last year for which USDA NASS and EIA had complete datasets.

EISA establishes 15 billion gallons as the maximum amount of corn starch ethanol that can contribute to meeting the 36 billion gallon per year renewable fuel target in 2022. Domestic production, which totaled 10.9 billion gallons in 2009 (EIA, n.d.[c]), is expected to meet this target through a combination of increased corn yield, increased acreage, and, potentially, improved efficiency in converting corn starch to ethanol. U.S. Department of Agriculture (USDA) estimates that planted corn acreage will remain at 88 million acres through 2021(USDA does not project acreage beyond 10 years), as U.S. demand for biofuel increases (USDA, 2010a). In the RFS2 analysis, EPA estimates that in order to produce 15 billion gallons of corn starch ethanol per year by 2022, the percentage of corn acreage dedicated to ethanol could rise from the current 38 percent to as high as 41 percent in 2022 (U.S. EPA, 2010b).

Concern has been raised that the demand for corn ethanol may put pressure on the USDA's Conservation Reserve Program (CRP) (Secchi and Babcock, 2007). This program provides farmers with financial incentives to set aside a certain portion of their cropland for buffer zones in order to conserve wildlife habitat, reduce erosion, protect water quality, and support other environmental goals. CRP lands are not precluded by the feedstock requirements for renewable fuels (see text box on page 3-1) from being used to grow biofuel feedstocks. Therefore their conversion to biofuel feedstocks could reduce the effectiveness of the CRP program in protecting the environment, and could result in increased environmental impacts, depending on the nature (i.e., crop) and extent of the conversion. One estimate, which examined the state of Iowa, predicted that high corn prices could lead to the recultivation of up to 70 percent of the expiring acreage enrolled in the CRP (Secchi and Babcock, 2007). Other states may not have such high rates of recultivation given that much of the land in the CRP is marginal and would be expensive to cultivate.

The Food, Conservation, and Energy Act of 2008 (Farm Bill) capped CRP acreage at 32 million, reducing enrollment by 7.2 million acres from the 2002 Farm Bill and potentially making more acreage available for corn production (ERS, 2008). In 2007, approximately 28.5 million acres or 78 percent of all CRP lands consisted of some type of grassland (FSA, 2008).

A USDA study estimates that, to meet the renewable fuel standard, total cropland will increase 1.6 percent over 2008 baseline conditions by 2015, with corn acreage expanding 3.5 percent, accounting for most of the cropland increase (Malcolm et al., 2009). While corn acreage is expected to expand in every region, this USDA study estimates that traditional corn-growing areas would likely see the largest increases—up 8.6 percent in the Northern Plains, 1.7 percent in the Corn Belt, and 2.8 percent in Great Lakes States (Malcolm et al., 2009). Historically, corn has been grown in rotation with other crops such as wheat, hay, oats, and especially soybeans. However, high corn prices have created incentives for continuous cultivation of corn (NASS, 2007a), and in some cases conversion of non-cropland to corn.

Corn stover—the stalks, leaves, husks, and cobs that are not removed from the fields when the corn grain is harvested—provides another potential feedstock for meeting EISA requirements. In the RFS2 RIA, EPA estimated that 7.8 billion gallons of ethanol could be produced from corn stover by 2022. Most corn stover harvesting for biofuel is expected to be from the major corn producing states. As of July 2010, there is no commercial production of cellulosic ethanol from corn stover.

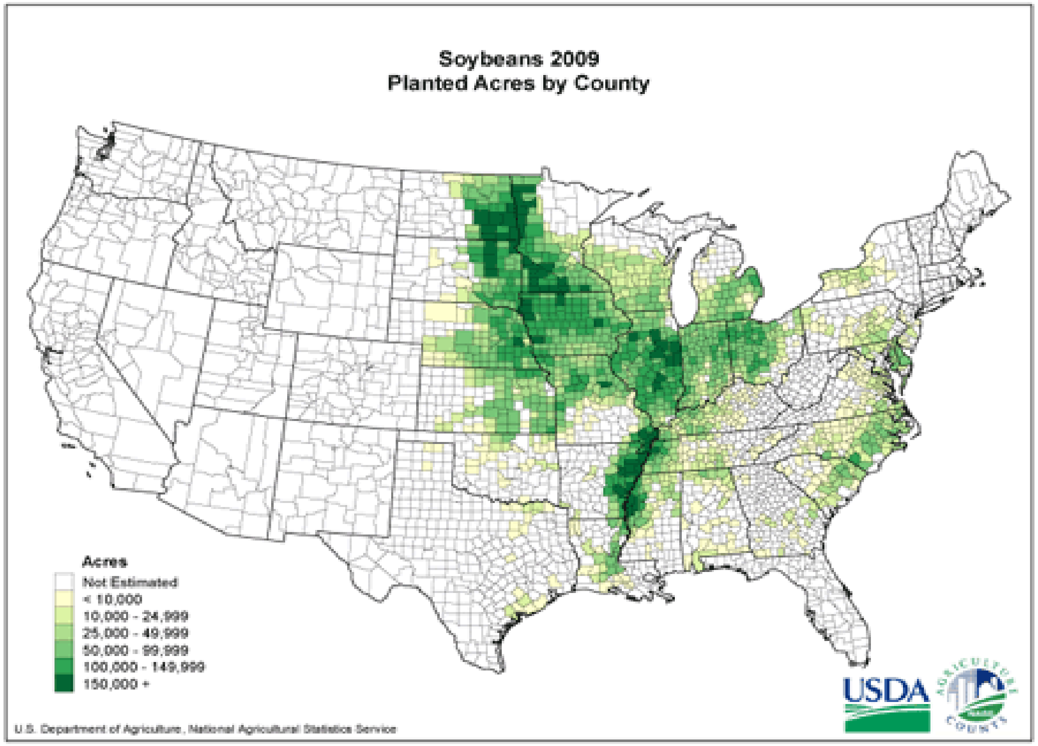
Because corn stover protects underlying soil from wind and water erosion, the use of corn stover as a feedstock could increase soil erosion and environmental impacts compared to existing practices (Sheehan et al., 2004; Williams et al., 2009). The USDA's Natural Resource Conservation Service (NRCS) has established soil loss tolerance levels, and farmers harvesting corn stover are encouraged to maintain a minimum level of groundcover or using other practices to minimize soil loss. Under current rotation and tillage practices, approximately 30 to 40 percent of stover could be collected cost effectively, taking into consideration erosion reduction, soil moisture needs and nutrient replacement (Graham et al., 2007; Perlack et al., 2005).

Maintaining soil carbon levels is another concern that should be taken into account when determining the extent of stover harvesting. To maintain soil carbon levels, a significant portion of the corn stover would need to be left on fields, reducing the amount of the biomass that could be collected for feedstock. Sustainable residue removal rates depend on tillage practices, with no-till allowing for the greatest level of sustainable removal. Developing a single national estimate of the amount of residue that must remain on the ground to meet conservation goals is difficult because much depends on site-specific conditions.

After corn, soybean is the second largest agricultural crop (in terms of acreage) in the U.S. In 2009, American farmers planted 77.4 million acres and harvested 3.4 billion bushels. In 2010, American farmers planted 77.7 million acres of soybeans and again harvested 3.4 billion bushels (NASS, 2010a). Soybean oil is the principal oil used for commercial production of biodiesel in the U.S., responsible for about half of total biodiesel production, with the rest coming from various other vegetable oils such as canola oil as well as waste fats, tallow and greases (see Table 4-1 for more detailed breakdown) (EIA, 2010). In harvest year 2008/2009, approximately 5.6 percent of the soybean harvest, or about 1.9 billion pounds of soybean oil (USDA, 2010b), went to biodiesel production and yielded 505 million gallons in calendar year 2009 (EIA, n.d. [d]). This was a significant decline from the production total in 2008 of 676 million gallons (EIA, n.d. [d]). Nonetheless, USDA expects the percentage of soybean harvest going to biodiesel to increase to 7.8 percent by 2012 and holding steady through 2019. USDA also projects that soybean oil used for biodiesel will represent 13-15 percent of total use of soybean oil–approximately 400 million gallons of biodiesel (USDA, 2010b).[10] USDA estimates that soybean acreage will level off at approximately 76 million acres through 2019 (USDA, 2010b).

In terms of cultivation, soybeans are typically grown in the same locations as corn. Figure 3-3 shows that soybean production is centered in the Upper Midwest and along the Mississippi River Valley, with Iowa, Illinois, Indiana, Minnesota, and Nebraska representing the top soybean-producing states.

[10] Percentages were calculated by multiplying together the percentage of soybean crop converted to soybean oil, percentage of soybean oil converted to biodiesel, and total bushels produced, then dividing by total bushels produced.



Source: NASS, 2010b.

Figure 3-3: Planted Soybean Acres by County (2009)

Much of the recent expansion in corn acreage for ethanol production has come at the expense of land previously used for other crops, especially soybeans (Fargione et al., 2009; Keeney and Hertel, 2009). In 2007, corn acreage expanded 23 percent in response to high prices and the demand for corn ethanol production (Mitchell, 2008). This expansion resulted in a 16 percent decline in soybean acreage, which reduced soybean production and contributed to a 75 percent rise in soybean prices between April 2007 and April 2008 (Mitchell, 2008). Much of the soybean acreage decrease occurred as a result of changing agricultural rotation, for example some corn-soybean-corn rotations were replaced by continuous corn (Fargione et al., 2009). In 2008, corn acreage declined by 7.5 million acres, to 86 million acres, while soybean acreage increased by almost 11 million acres. A large proportion of the soybean acreage increase came from the reduction in corn acreage as well as switching crops other than corn to soybeans, loss of CRP land, and an increase in soybean double cropping (Babcock and McPhail, 2009). Such tradeoffs between food crops, energy crops, and CRP lands may become more critical in the future, especially as climate change affects global cropland area and water availability for irrigation. One study predicted that climate change will reduce global cropland area by 9 percent by the year 2050, although noting that this could be buffered by the increased use of water management strategies (Rost et al., 2009).

3.2.1.2 Overview of Environmental Impacts

Corn and soybean production entails the use of pesticides, fertilizer, water, and fuel/energy, in addition to drainage systems, each of which can affect water quality, water availability, soil quality, air quality, and ecosystem health. Changes in land cover, vegetation, and habitat have additional impacts on the environment. Table 3-2 summarizes these impacts and the factors that influence them. (Note: Because corn stover is essentially a by-product of corn production, only direct environmental impacts from stover harvest are considered for discussion of this feedstock's impacts.)

Table 3-2: Impacts Associated with Biofuel Feedstock Production (Corn Starch, Corn Stover, and Soybean)

Impact/Resource Use Category	Corn Starch [a]	Harvest of Corn Stover	Soybean
Water Quality	Corn production can lead to erosion of sediment and the runoff and leaching of fertilizers such as nitrogen and, phosphorus, and pesticides such as atrazine. Artificial drainage like tile drains increases loss of nitrogen to surface waters. Actual water quality impacts depend on a number of geographic and management factors: for instance, the rate, timing, and method of application of fertilizers, manure, and pesticides; and the use of erosion control practices such as edge of field controls like vegetative buffers, controlled drainage, or constructed wetlands.	Removal of corn stover will require increases in fertilizer application rates, which can result in the pollution of surface and ground waters. Erosion can also increase as more stover is removed, providing less ground cover. Actual water quality impacts will depend on a variety of geographic and management factors.	The majority of soybean acreage is managed with conservation tillage, minimizing erosion. Though soybean production requires smaller amounts of many nutrients—especially nitrogen—than does corn production, it often still requires potassium and phosphorus, which may impact water quality. Actual water quality impacts depend on a variety of geographic and management factors.
Water Quantity	In areas where corn production requires irrigation, surface and ground water quantities may be affected. Irrigation requirements depend on rainfall, relative humidity, soil properties, and crop yield.	Additional water use for stover production is likely to be minimal.	In areas where soybean production necessitates irrigation, surface and ground water quantities may be affected. Irrigation requirements depend on rainfall, relative humidity, soil properties, and crop yield.

Table 3-2: Impacts Associated with Biofuel Feedstock Production (Corn Starch, Corn Stover, and Soybean)

Impact/Resource Use Category	Corn Starch [a]	Harvest of Corn Stover	Soybean
Soil Quality	The conversion of uncultivated land or marginal cropland to corn production may lead to higher soil erosion and lower quantities of soil organic matter. Soil quality is maintained through management practices that include reduced use of tillage, use of crop rotations, and the return of organic matter to the soil through cover crops, manure, or crop residues.	Excess removal of corn stover can increase erosion, decrease soil organic matter and degrade water quality. These impacts depend on management practices and local conditions, including slope, soil type, and prior land use.	The majority of soybean acreage is managed with conservation tillage practices, mitigating erosion and impacts on soil organic matter.
Air Quality	Emissions of criteria and air toxic pollutants are associated with several sources, including combustion of fossil fuels by farm equipment; airborne particles (dust) generated during tillage and harvesting; and the production and application of fertilizers and pesticides. Actual impacts depend on use rates and formulations of fertilizers, and pesticides; tillage methods; the type of fuel and agricultural equipment; and conditions at time of tillage and harvest.	Corn stover harvesting may affect air quality if it requires the combustion of additional diesel or gasoline beyond that used to harvest corn, if it leads to additional fertilization of fields, or if additional dust is released into the air during harvest operations.	Emissions of criteria and air toxic pollutants are associated with several sources, including combustion of fossil fuels by farm equipment; airborne particles (dust) generated during tillage and harvesting; and the production and application of fertilizers and pesticides. Actual impacts depend on use rates and formulations of fertilizers and pesticides; tillage methods; the type of fuel and agricultural equipment; and conditions at time of tillage and harvest.
Ecosystem Impacts	The type and extent of ecosystem impacts depend on local conditions and management. Nutrients, sediment, and pesticides can contaminate surface waters and wetlands, leading to changes in biodiversity.	Corn stover harvest may decrease soil biodiversity. The water quality impacts described above may affect wetland biodiversity. The ecosystem impacts of stover removal depend on a variety of geographic and management factors, including the amount of stover removed.	The type and extent of ecosystem impacts depend on local conditions and management. Nutrients, sediment, and pesticides can contaminate surface waters and wetlands, leading to changes in biodiversity.

Table 3-2: Impacts Associated with Biofuel Feedstock Production (Corn Starch, Corn Stover, and Soybean)

Impact/Resource Use Category	Corn Starch [a]	Harvest of Corn Stover	Soybean
Invasiveness Potential	Corn is non-invasive.	See "Corn"	Soybean is non-invasive.

a – Impacts associated with corn production are described in the "Corn Starch" column.

Cultivation of row crops such as corn and soybeans may lead to high levels of soil erosion, nutrient loss, and pesticide and water use if not managed adequately (Groom et al., 2008, Table 1). Agricultural conservation systems may be used to reduce or minimize the impact of row crop agriculture on the environment. The systems support 1) controlled application of nutrients and pesticides through proper rate, timing, and method of application, 2) controlling erosion in the field (i.e., reduced tillage, terraces, or grassed waterways), and 3) trapping losses of soil at the edge of fields or in fields through practices such as cover crops, riparian buffers, controlled drainage for tile drains, and constructed/restored wetlands (Blanco-Canqui et al., 2004; Dinnes et al., 2002; NRCS, 2010).

The effectiveness of conservation practices, however, depends upon their adoption. The USDA Conservation Effects Assessment Project (CEAP) recently released a major study quantifying the effects of conservation practices commonly used on cultivated cropland in the Upper Mississippi River Basin. It found that, while erosion control practices are commonly used, there is considerably less adoption of proper nutrient management techniques to mitigate nitrogen loss to water bodies (NRCS, 2010).

Further, even when erosion practices are reliably implemented, conservation practices and best management practices (BMPs) are not a panacea. A case study in the Chesapeake Bay (CENR, 2010) found that, although the implementation of BMPs since 2000 has significantly lowered loadings of nitrogen (72 percent of sites showed downward trends), total phosphorus (81 percent of sites), and sediment (43 percent of sites), lower nutrient input has not improved dissolved oxygen levels overall in the Chesapeake Bay, with the exception of small-scale reversals in hypoxia.

3.2.2 Water Quality

Water quality impacts from increased corn and soybean production for biofuel may be significant, and are caused by pollution from nutrients, sediment, and pesticides, as well as biological contaminants such as pathogens that are released when animal manure is applied as fertilizer. Multiple studies predict that increased production of crops for biofuels will exacerbate water quality problems in the Gulf of Mexico and other U.S. coastal waters if the crops are not grown under improved agricultural conservation practices and expanded nutrient BMPs (Greene et al., 2009, and Rabalais et al., 2009, cited in CENR, 2010).

3.2.2.1 Nutrient Loading

Nutrients—Surface Water Impacts

Increased production of row crops for biofuel, especially corn, will increase nitrogen and phosphorus loading to surface waters if not managed appropriately. Excessive levels of nutrients in a body of water can cause accelerated algal growth, reducing light penetration and oxygen levels. Low dissolved oxygen (i.e., hypoxia) can kill fish, reducing their populations and the species diversity in the affected area. This nutrient enrichment process (eutrophication) can cause serious deterioration of both coastal and inland water resources. According to a 2008 report by the National Research Council, excess nutrients and sediment from the high corn-producing Midwest are the primary sources of water quality degradation in the Mississippi River Basin and the Gulf of Mexico (NRC, 2008, p. 88). Further, the National Summary of Impaired Waters (U.S. EPA, 2010c)[11] documented that in 2008, nationwide, approximately 50 percent of the 3.5 million miles of stream and rivers and 66 percent of the over 41 million acres of lakes and reservoirs in the U.S. were impaired due to nutrient enrichment. The 2007 National Estuarine Eutrophication Assessment found that the Mid-Atlantic is the region most impacted by hypoxia, with almost 60 percent of the waters affected by anthropogenic land-based sources of nutrient pollution, agriculture being the largest contributor (CENR, 2010). The National Summary of Impaired Waters also reported that in 2008 over 68,000 miles of streams and rivers and over 1.3 million acres of lakes and reservoirs in the Mississippi River Basin states were impaired because of nutrients. Increased corn and soybean production for biofuels could exacerbate this situation due to the nutrients from additional fertilizer or increases in the acreage or extent and density of subsurface tile drainage. The Committee on Environment and Natural Resources cites a 2008 report that predicts the average annual flux of dissolved inorganic nitrogen to the Gulf of Mexico could increase by 10 to 34 percent, based on a “pessimistic” scenario in which corn production acreage increases by up to 9 percent (Donner and Kucharik, 2008, cited in CENR, 2010).

In evaluating the potential for water quality impacts due to increased nutrient loads to surface waters, there is some debate about which nitrogen compounds to consider. Not all nitrogen compounds can be easily used by algae (i.e., are bioavailable) and thus some forms of nitrogen impact eutrophication more than others. Ammonia is the inorganic nitrogen compound that is easiest for most algae to use, followed by nitrate and nitrite (Cole, 1983). Total nitrogen is a measure of the sum of the nitrogen present in both inorganic and organic compounds in water.

Although many studies track total nitrogen, some researchers argue that it is more appropriate to consider inorganic nitrogen compounds only, as those are most likely to impact water quality through eutrophication. When tracking the fate and transport of nitrogen in surface waters within large watersheds (e.g., in the Mississippi River Basin), it is important to remember that inorganic nitrogen compounds are readily converted to organic nitrogen compounds and back to inorganic compounds by organisms present in surface waters and sediments. At the basin scale, measuring total nitrogen provides insight into the potential maximum impact of nitrogen inputs into surface waters.

[11] Numbers in text were calculated by summing miles/acres reported by each state in their 305(b) assessments as impaired by “nutrients”; “ammonia, un-ionized”; “nitrogen, total”; “nutrient/eutrophication”; “phosphorus, total”; “ammonia, total”; “nitrogen, nitrate”; and “ammonia.”

Corn has the highest fertilizer use per acre of any of the biofuel feedstocks, and it accounts for the largest portion of nitrogen fertilizer use among all feedstocks discussed in this report (U.S. EPA, 2010b). By one estimate, which surveyed 19 U.S. states, approximately 96 percent of corn acreage received nitrogen fertilizer in 2005, with an average of 138 pounds per acre (NASS, 2006). An Iowa State University study found that each acre of harvested corn also requires about 55 pounds of phosphorus (in the form P_2O_5) (Iowa State University, 2008). Assuming a yield of 154 bushels per acre (NASS, 2010c) and an ethanol conversion rate of 2.7 gallons per bushel (Baker and Zahniser, 2006), this results in 0.33 pounds of nitrogen and 0.13 pounds of phosphorus applied per gallon of ethanol produced. Nitrogen discharged from corn fields via runoff, sediment transport, tile/ditch drainage, and subsurface flow averages 24 to 36 percent of the nitrogen applied (and can range from 5 percent in drought years to 80 percent in flood years) (Dominguez-Faus et al., 2009).

Nutrients are applied to fewer soybean acres compared to corn and at much lower rates (U.S. EPA, 2010b). However, losses of nitrogen and phosphorus from soybeans can occur at quantities that can degrade water quality (Dinnes et al., 2002; Randall et al., 1997). In 2006, the USDA's National Agricultural Statistics Service estimated that nitrogen and phosphorus fertilizers were applied to 18 percent and 23 percent of soybean acreage, respectively, with an average of 16 pounds of nitrogen and 46 pounds of phosphate applied per acre fertilized (NASS, 2007b). The quantity of nitrogen fertilizer applied to soybean fields ranged from 0 to 20 pounds per acre, while the quantity of phosphate ranged from 0 to 80 pounds per acre. Similar to corn, the conversion of idled acreage to soybeans is estimated to result in losses of nitrogen and phosphorus from the soil (Simpson et al., 2008).

Corn requires less fertilizer when grown in rotation with soybeans. Therefore, crop rotation provides an effective strategy for reducing the amount of fertilizer and pesticide applied to fields, and therefore runoff and leaching of the pollutants to water. One study estimated that 2 to 40 percent of the total nitrogen leached from fields planted alternately with corn and soybeans came from the fields when they were planted with soybeans, meaning that most of the nitrogen runoff was due to corn production (Powers, 2005).

While the total amount of nitrogen lost from corn fields tends to be higher than losses from soybean fields (Powers, 2005), loss of inorganic nitrogen from corn and soybean fields tends to be similar. This suggests that eutrophic effects of nitrogen will be similar for runoff from both corn and soybean fields in surface waters near the fields. However, when considering impacts at the basin scale, it is more relevant to consider the total amount of nitrogen contributed by each crop.

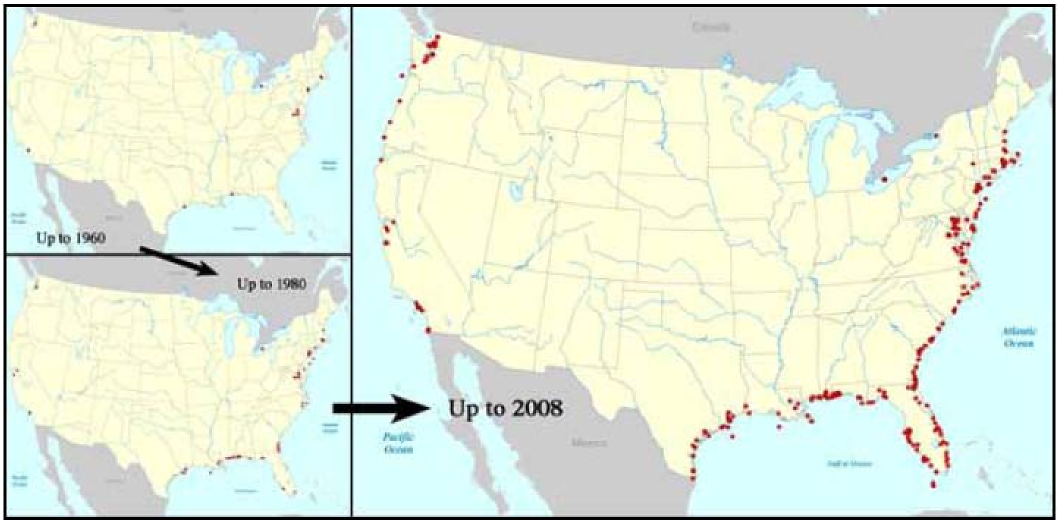
Use of corn stover for ethanol production would not necessarily result in increased corn production. However, the removal of corn stover could lead to loss of soil surface cover if NRCS guidelines are not followed, thereby increasing runoff of nitrogen and phosphorus to surface waters, including wetlands (Kim and Dale, 2005). Even partial removal of corn stover can result in nutrient losses to water due to increased runoff (Kim and Dale, 2005; Lal, 2004). In addition, corn stover removal can lead to the loss of soil nutrients needed for corn growth, and higher fertilizer rates are likely to be required to sustain crop productivity, increasing the likelihood of increased runoff and transport of non-point source pollutants (Blanco-Canqui and Lal, 2009a). Typically, for each ton of corn stover harvested, an additional 16 pounds of nitrogen fertilizer and 6 pounds of phosphorus fertilizer must be added to the soil, though these quantities vary considerably (Sawyer and Mallarino, 2007).

Mitigating the loss of nitrogen and other nutrients to water bodies is a research priority for the USDA. Since drainage systems are a key conduit for nutrient loading, new research is focusing on alternative surface and subsurface drainage solutions. An interagency Agricultural Drainage Management Systems Task Force, formed in 2003 and recently expanded, is working to reduce the loss of nitrogen and phosphorus from agricultural lands through drainage water management (CENR, 2010). One emerging conservation technology that addresses water quality degradation is the creation of wetlands on the perimeter of fields in order to receive surface runoff and filter out nutrients prior to its discharge into streams and rivers. Surface water runoff control, another conservation method used to stop water erosion, reduces the loss of nutrients to the surrounding environment through overland flow, but increases infiltration and loss of soluble nitrogen and phosphorus. A third strategy, lowering the water table during planting and harvesting, has been predicted to lower nitrogen losses in the Chesapeake watershed by 40 percent (CENR, 2010). Other strategies, such as planting perennial grasses over subsurface tile drains or placing wood chips in drainage ditches, are also being explored. Implementing strategies such as these on agricultural lands that contribute a disproportionate share of nitrogen loads will maximize the environmental benefit of their application (CENR, 2010).

However, none of these practices guarantee environmentally sustainable biofuel production on an industrial scale. The interactions between various BMPs need to be investigated more closely, as there can sometimes be unexpected adverse consequences from new technologies. For example, the 2010 report by the Committee on Environment and Natural Resources notes that the introduction of tile-drainage systems in the Midwest has improved agricultural yields but worsened water quality by accelerating nutrient-loaded runoff to streams and rivers without allowing natural processes to filter the nutrients (CENR, 2010).

Nutrients—Coastal Waters Impacts

Nutrient enrichment is a major concern for coastal waters across the U.S., including the Gulf of Mexico, Chesapeake Bay, other estuaries, and the Great Lakes. For example, almost 15 percent of the coastal waters in the Gulf of Mexico and Northeast have poor water quality as measured by nutrient concentrations, extent of hypoxia, and water clarity (U.S. EPA, 2008b). The number of U.S. coastal areas documented as experiencing hypoxia increased from 12 in 1960 to over 300 in 2008 (see Figure 3-4) (CENR, 2010). While these impacts are due to a number of types of nutrient inputs, such as lawn fertilizers, other agricultural uses, atmospheric deposition, and wastewater discharges, increased corn and soybean production for biofuel will likely increase nutrient loading in those watersheds where increased production occurs (CENR, 2010).



Note: Map does not display one hypoxic system in Alaska and one in Hawaii.
Source: CENR, 2010.

Figure 3-4: Change in number of U.S. coastal areas experiencing hypoxia from 1960 to 2008

Hypoxia in the Gulf of Mexico has been a long-standing environmental and economic issue that threatens commercial and recreational fisheries in the Gulf (U.S. EPA, 2010b). The primary cause of hypoxia in the Gulf of Mexico is excess nitrogen and phosphorus loadings from the Upper Midwest flowing into the Mississippi River, suggesting that increased corn and soybean production will exacerbate the problem (U.S. EPA, 2010b). U.S. Geological Survey (USGS) SPARROW[12] modeling of the sources of nutrient loadings to the Gulf of Mexico estimated that agricultural sources contributed more than 70 percent of the delivered nitrogen and phosphorus to the Gulf of Mexico (Alexander et al., 2008). Corn and soybean production accounted for 52 percent of nitrogen delivery and 25 percent of phosphorus delivery. Modeling of the Upper Mississippi River Basin (upstream of Cairo, Illinois) using SWAT[13] modeling indicated that, on average, it contributes 39 percent of the nitrogen load to the Gulf of Mexico, and 26 percent of the phosphorus load (SAB, 2007). One study estimated that corn production contributes between 60 and 99 percent of the total nitrogen load to the Mississippi River from eastern Iowa watersheds (Powers, 2007). Other studies have also determined that the majority of nitrate in the Mississippi River originates in the Corn Belt (Donner et al., 2004; Goolsby et al., 1999). Nitrogen from fertilizers can also volatilize (and then return to water through atmospheric deposition). Atmospheric nitrogen from all sources, including power plant emissions, is estimated to contribute 15 to 20 percent of the nitrogen loading to the Gulf of Mexico (Alexander et al., 2008), and about 30 percent of the nitrogen loading to Chesapeake Bay (Paerl et al., 2002).

[12] SPARROW (SPAtially Referenced Regressions On Watershed) is a watershed model developed by USGS relating water quality measurements at monitoring stations to other watershed attributes. The model estimates nitrogen and phosphorus entering a stream per acre of land, and evaluates the contributions of nutrient sources and watershed properties that control nutrient transport.

[13] The Soil and Water Assessment Tool (SWAT) is a public domain model jointly developed by USDA Agricultural Research Service and Texas A&M University System. SWAT is a river basin-scale model to simulate the quality and quantity of surface and ground water and predict the environmental impact of land management practices on different soil patterns and land use patterns.

A USDA study projects that reaching 15 billion gallons per year of ethanol from corn starch (i.e., not including stover) will result in a 1.7 percent increase in nitrogen loads to surface water, with the greatest increases in nitrogen load occurring in the Corn Belt and Northern Plains (Malcolm et al., 2009). EPA used the SWAT model to predict the impacts of increased corn production to meet the RFS2 corn starch ethanol targets on water quality in Upper Mississippi River Basin, which empties into the Gulf of Mexico. The modeling found a maximum increase in nitrogen load to the Gulf of Mexico of 1.9 percent, and a maximum of 1 percent increase in phosphorus load. The SWAT model also indicated that, by 2022, increased corn yields could reduce the need for increasing the amount of land in corn, so nutrient loads could decrease from earlier peaks (SAB, 2007).

Ecological features such as wetlands and riparian buffers play an important role in absorbing nutrients before they run into surface waters. Conserving wetlands where they exist, or creating artificial vegetated riparian buffers between waters and croplands, is a way to mitigate the impacts of nutrient loading. Riparian buffers and filter strips prevent potential pollutants in agricultural runoff (sediment, nutrients, pesticides, pathogens) from reaching surface waters. While the effectiveness of these buffers can vary depending on many factors, including slope, width, vegetation used, and how well they are maintained, studies have shown that they can remove up to 78 percent of phosphorous, 76 percent of nitrogen, and 89 percent of total suspended solids (TSS) (Schwer and Clausen, 1989).[14]

Nutrients—Ground Water Impacts

Excess nutrients from fertilizers can leach into ground water, which can discharge to surface waters, thereby contributing to surface water nutrient loading. About two-thirds of the nitrogen lost to subsurface flow eventually returns to surface water (U.S. EPA, 2010b, p. 971). Ground water can also be used for public and private drinking water supplies, and fertilizers can increase the concentration of nitrate in ground water wells, especially shallow wells (less than 200 feet deep). USGS sampled 495 wells in 24 well networks across the U.S. in predominantly agricultural areas from 1988 to 2004 and found significant increases in concentrations of nitrate in 7 of the 24 well networks. In 3 of the 7 well networks, USGS found nitrate concentrations that exceeded the federal drinking water standards of 10 mg/L of nitrate-nitrogen (Rupert, 2008). Increased corn and soybean production for biofuels could worsen the problem of contaminated well water because of additional nitrogen inputs from fertilizer used to grow more corn. USDA projects that reaching 15 billion gallons per year of ethanol from corn will result in a 2.8 percent increase in nitrogen leaching to ground water, with the greatest increases occurring in the Great Lakes states and the Southeast (Malcolm et al., 2009). Similar estimates for soybean production were not identified. Studies of nitrate leaching from corn and soybean rotation cropping systems are inconclusive about whether these systems increase or decrease leaching rates (Kanwar et al., 1997; Klocke et al.; 1999; Weed and Kanwar, 1996; Zhu and Fox, 2003).

Fertilizer application management strategies aim to reduce nitrogen leaching by maximizing the efficiency of applied fertilizer. Such strategies focus on collecting precise information on soil nutrient content in order to better inform application rates. The USDA

[14] See also http://cfpub.epa.gov/npdes/stormwater/menuofbmps/index.cfm?action=factsheet_results&view=specific&bmp=82.

reports that phosphorus accumulation on farms has reached levels that often exceed crop needs (ARS, 2003). Better information on these conditions could help reduce nutrient runoff that leads to eutrophication. There may also be economic incentives for implementing fertilizer management strategies. In 2006, the University of Minnesota Extension, an agricultural research partnership between federal, state and county governments, estimated that 86 percent of Minnesota farmers could save more than $6 per acre and 56 percent could save more than $10 per acre in fertilizer costs by following better informed nutrient application rates (Minnesota Department of Agriculture, 2010).

3.2.2.2 Sediment

Nutrients and sediment are the two major water quality problems in the U.S., and much attention has been focused on these issues in the Mississippi River Basin and the Gulf of Mexico (NRC, 2008, p. 88). Use of soil erosion control practices is widespread, yet 15 percent of acres in the Upper Mississippi River Basin experience excessive sediment loss (NRCS, 2010). The National Water Summary of Impaired Waters stated that in 2008 over 70,000 miles of streams and rivers and over 1.2 million acres of lakes and reservoirs in Mississippi River basin states are impaired because of sediments or turbidity (U.S. EPA, 2010c).[15] Nelson et al. (2006) reported that row crops, such as corn and soybean, result in higher erosion rates and sediment loads to surface waters, including wetlands, than non-row crops that might be used as biofuel feedstock, such as grasses. Sedimentation rates in agricultural wetlands can be higher than in natural grassland landscapes; increased sedimentation may, depending on sediment depths, cover viable seeds sufficiently to prevent germination (Gleason et al., 2003). EPA and USDA have evaluated the impact of the RFS2 rule on sediment loads. As reported in the water quality analysis conducted by EPA for the RFS2 rule, it is estimated that annual sediment loads to the Mississippi River from the Upper Mississippi River Basin would increase by 6.22 million tons (15 percent) between 2005 and 2022, assuming corn stover remained on the field following harvest (AquaTerra, 2010). A USDA study estimates that nationally, sediment loads in 2015 will be 1.6 percent greater with implementation of RFS2 than without, assuming ethanol production from corn starch only (Malcolm et al., 2009).

Removal of corn stover from fields for use in biofuel production is expected to increase sediment yield to surface waters and wetlands, but rates are highly variable depending on soils, slope, management of fields, and the proportion of stover harvested (Cruse and Herndl, 2009; Kim and Dale, 2005). Results of SWAT modeling of the Upper Mississippi River Basin (AquaTerra, 2010) indicated that leaving corn stover on fields helps reduce soil erosion and sediment transport, even when the amount of land in corn production increases. However, the amount of soil erosion that agricultural cropland experiences is a function of many factors, including not only residue left on the field, but also field operations (field preparation, tillage, etc.) in preparation for the next crop, timing of field operations, and other site-specific factors noted above (U.S. EPA, 2010b).

[15] Numbers in text were calculated by summing miles/acres reported by each state in their 305(b) assessments as impaired by "sedimentation/siltation" or "turbidity."

Conservation tillage practices, including no-till, strip-till, ridge-till, and mulch-till,[16] can reduce erosion by leaving at least 30 percent of the ground covered by crop residue and by limiting soil disturbance. According to the USDA, 41 percent of planted acreage in the U.S. uses conservation tillage as a mitigation strategy (ARS, 2006). These techniques have been shown to reduce erosion by as much as 60 to 90 percent, depending on the conservation tillage method (Minnesota Department of Agriculture, 2010). In 2002, the USDA Agricultural Research Service studied the effect of ridge tillage on Northern Corn Belt plantations. The study showed that ridge tillage not only reduced erosion and sediment loading but also increased profitability, reduced fuel and labor use, and reduced economic risk relative to conventional tillage for a corn and soybean rotation (ARS, 2006). Additionally, these alternative tillage approaches can reduce trips across the field, lowering fuel use and improving the energy balance of the resulting biofuel. The use of conservation tillage, in combination with BMPs, such as cover crops, may partially compensate for the increase in erosion potential caused by cover stover removal (Blanco-Canqui and Lal, 2009b). Depending on the soil type, these practices may allow a percentage of stover to be harvested sustainably (Blanco-Canqui and Lal, 2009b).

3.2.2.3 Pesticides

According to the National Summary of Impaired Waters (i.e., waters that do not meet the water quality standards) (U.S. EPA, 2009a, 2010d), over 16,000 miles of streams and rivers and over 370,000 acres of lakes and reservoirs in the U.S. were impaired in 2008 because of pesticides, with atrazine (commonly used in corn production) specifically cited by several states (U.S. EPA, 2010c). Atrazine was also estimated to be the most common pesticide lost from agricultural lands in the Upper Mississippi River Basin (NRCS, 2010).

Corn production uses more pesticides than predicted for any other potential biofuel crop produced in the U.S. (Pimentel and Patzel, 2005; Pimentel and Pimentel, 2008, p. 380; Ranney and Mann, 1994). USDA's NASS estimates that insecticides were applied to 16 percent of the 2006 soybean-planted acreage (NASS, 2007b). USDA also estimates that herbicides were applied to 98 percent of the planted soybean acreage in 2006. Soybean production releases less pesticide to surface and ground water per unit of energy gained (Hill et al., 2006).

While effective pest control may be critical to achieving the yield gains that underpin EISA biofuel projections and targets (Perkins, 2009), there are risks associated with the use of pesticides. The FIFRA registration process is intended to minimize these risks. Many factors contribute to the relative risks of pesticides on the environment, including fate and transport characteristics, method of application, depth to ground water, and proximity to receiving waters. To protect consumers against risks posed by ingestion of these pesticides, FFDCA requires the establishment of pesticide residue tolerances on food using a standard of reasonable certainty of no harm.

[16] *No-till* refers to the absence of soil tillage to establish a seed bed, meaning the farmer plants the crop directly into the previous year's crop residue. In *strip-till*, only the portion of the soil that is to contain the seed row is disturbed. In *ridge-till*, plants grow on hills that are the product of cultivation of the previous crop and are not tilled out after harvest. In *mulch-till*, plant residues are conserved but a field cultivator or disks are used to till prior to planting to partially incorporate the residue into the soil.

Growing continuous corn (rather than in rotation with other crops) can increase population densities of pests such as the corn rootworm, resulting in increased pesticide applications to control these pest species (Whalen and Cissel, 2009). A USDA study projects that cropland dedicated to continuous corn will increase by more than 4 percent by 2015 to reach the 15 billion gallons per year of ethanol from corn target (Malcolm et al., 2009). In addition, increases in corn acreage and any conversion to corn of crops other than corn will most likely increase total herbicide use. Increased corn and soybean production can result in the increased use of herbicides that can run off or leach into surface water or ground water sources.

Integrated pest management (IPM) practices may help reduce pesticide use by tailoring treatment to pest infestation cycles, and by more precisely targeting the amount and timing of applications. IPM focuses on extensive monitoring of pest problems, comprehensive understanding of the life cycles of pests and their interaction with the environment, and very precise timing of pesticide applications to minimize pesticide use. In addition to providing environmental benefits of lower pesticide use, IPM often results in lower chemical pesticide expenses and pest damage to crops, as well as preventing the development of pesticide-resistant pests (Minnesota Department of Agriculture, 2010). The use of cover crops is an IPM practice that can dramatically reduce chemical application and soil erosion. USDA research in the Midwest in 2006 demonstrated that autumn-planted small grain cover crops reduced soil erosion, nitrate leaching, and suppressed weeds (Teasdale et al., 2007).

National adoption of IPM strategies varies. Corn and soybean growers reported scouting for weeds, insects, and diseases on 50 percent of acres or more in 2000, but reported adjusting planting or harvest dates to manage pests on less than 20 percent of acres (Weibe and Gollehon, 2006).

3.2.2.4 Pathogens and Biological Contaminants

The use of animal manure as a fertilizer has been tied to an increased risk of viruses and bacteria leaching into the water supply. Pathogens such as *Salmonella* sp., *Campylobacter* sp., and *Clostridium perfringens*—along with additives such as livestock antibiotics and hormones—may be released into surface or ground water when manure is applied to fields (Brooks et al., 2009; Lee et al., 2007a; Unc and Goss, 2004). The USDA Report to Congress on use of manure for fertilizer and energy reports that approximately 12 percent of corn and 1 percent of soybeans are fertilized with manure (MacDonald et al., 2009).

The flow paths by which pathogens can contaminate ground or surface water are the subject of current research. Transport through soil has been shown to remove harmful bacteria in some cases, though this may depend on soil characteristics, the hydrologic regime (i.e., distance to surface or ground water) and the pathogens in question (Malik et al., 2004; Unc and Goss, 2004). Contamination rates likely are greater where there is higher runoff relative to infiltration, a high water table, or a direct surface-ground water connection. Implementation of manure management practices, such as covering or storage at elevated temperatures prior to application can reduce runoff. In addition, applying manure during times of low runoff potential can reduce the risk of water contamination (Moore et al., 1995; Guan and Holley, 2003).

3.2.3 Water Quantity

3.2.3.1 Water Use

Over the entire biofuel supply chain (see Figure 2-3 in Chapter 2), crop irrigation is by far the most significant use of water in the ethanol production process, and it tends to be much higher than water use for most other non-renewable forms of energy on an energy content basis (Wu et al., 2009). In some geographic locations this could lead to serious impacts on already stressed water supplies, while in other locations water supply availability impacts are less likely to occur. Future assessment of biofuel feedstocks will need to consider restrictions on water use due to competing demand for water resources (Berndes, 2002).

For both corn and soybeans, the source for water used to irrigate crops varies from region to region. In the West, surface water is largely used to irrigate crops; in the Great Plains and Midwest, where the majority of corn and soybean production takes place, farmers rely heavily on ground water (Kenny et al., 2009). In the future, as corn production increases to meet ethanol demands, both geographical factors and the type of land/crop conversion will determine water use impacts. Water use will increase as land in pasture or other low- or non-irrigated uses are converted to irrigated corn production, especially in places like the Great Plains, where water demand for corn irrigation is high**.** Converting other crops, soybeans in particular, into corn will have little effect on water use in the Midwest, but could increase the total amount of water used for irrigation in the Plains because of corn's relatively high water use intensity on a per area basis (NRC, 2008).

Corn

Corn is relatively water-intensive compared to other crops. In some parts of the country, water demands for corn are met by natural rainfall, while in other places supplemental irrigation is required. For instance, in Iowa in 2007, less than 1 percent of the more than 14 million acres planted in corn was irrigated. In contrast, over 60 percent of Nebraska's 9.5 million acres of corn was irrigated in the same year (NASS, 2009).

Irrigation use for U.S. corn has been estimated to vary from a low of approximately 8 gallons of water per gallon of ethanol on average in Midwest states in one study (Wu et al., 2009) to a high of up to 1,000 gallons for states in the Great Plains in another study (Dominguez-Faus et al., 2009). While the data and methodology used to calculate these estimates are not uniform across studies, in general, water use is likely to be less than 500 gallons (perhaps substantially less) of irrigation water per gallon of ethanol in the Midwest and greater than 500 gallons per gallon of ethanol in more arid parts of the country (supporting information for Chiu et al., 2009). Taking into account the total volume of corn starch ethanol produced, this might translate into approximately 5 billion gallons of irrigation water in a single season in places like Iowa and Illinois versus 300 billion gallons in Nebraska (Chiu et al., 2009). The 2007 U.S. national ethanol-production-weighted average farm-to-fuel pump water requirement per gallon of ethanol in the U.S. was estimated to be 142 gallons (Chiu et al., 2009).

Corn Stover

Allocation of proportionate water use based on the energy captured from corn starch versus stover may be studied in the future as corn stover becomes a more common biofuel feedstock. Water use for corn stover above and beyond corn cultivation is likely to be minimal or negligible if undertaken with resource conservation practices, especially in the most productive corn-growing regions of the U.S. where corn stover is not functionally necessary for retention of soil moisture. If, however, corn stover is removed from dry corn cultivation areas with supplemental irrigation (states like Nebraska), loss of soil moisture that would have otherwise been retained by corn stover cover and contributed to productivity of the next season's crop (Blanco-Canqui and L al., 2009b) could necessitate additional irrigation.

Soybeans

Water for soybean cultivation, like corn, also comes from both natural precipitation and through irrigation. In some places, the water requirements are largely met with precipitation. For example, in 2007 in the leading soybean-producing state of Iowa, 8.6 million acres of soybeans were grown of which less than 1 percent was irrigated (NASS, 2009). In 2007 Nebraska grew 3.8 million acres of soybeans, of which over 40 percent was irrigated (NASS, 2009).

Average nationwide rates of soybean irrigation are estimated at 3,000 to 6,000 gallons of irrigation water to produce a volume of biodiesel equivalent to a gallon of gasoline (U.S. DOE, 2006; Dominguez-Faus et al., 2009). These rates are not applicable to states such as Iowa, where most soybeans are grown without irrigation. In Nebraska, however, where irrigation is heavily utilized, greater than 4,000 gallons of irrigation water per gallon of gas equivalent is not an unusual investment of water resources for biofuel production (supplemental information to Dominguez-Faus et al., 2009). Overall, irrigation estimates for soybeans tend to be greater than those needed to produce a volume of corn starch ethanol equivalent to a gallon of gasoline (Dominguez-Faus et al., 2009).

3.2.3.2 Water Availability

Because agriculture accounts for such a large share of water use in the U.S. (35 percent of withdrawals nationwide in 2005, and a much larger percentage in some parts of the country, according to Kenny et al., 2009), changes in agricultural production could impact future water availability. In particular, land conversion to corn for increased production of ethanol could create more demand for water, adding to existing water constraints and potentially creating new ones. The Great Plains states already have shortages, and water availability may decrease further when typically non-irrigated pasture and CRP land is converted to irrigated corn production. Converting other crops, soybeans in particular, into corn will have little effect on water use and availability in the Midwest, but could increase the total amount of water used for irrigation in the Plains because corn requires more water than soybeans on a per area basis in that region (NRC, 2008).

To a large extent, the current capacity to produce biodiesel from soybeans resides in states with rain-fed soybean cultivation. Such strategic siting of biodiesel production facilities minimizes both demands for irrigation water for biodiesel feedstock and potential conflicts over water availability required for other purposes such as power generation, public water use, and recreation. However, if biodiesel production develops in places requiring greater soybean irrigation such as the Great Plains, water availability could be reduced. This is especially true if irrigated soybean cultivation replaces other low or non-irrigated land uses. Because over 85 percent of irrigation withdrawals come from underground aquifers, ground water availability is likely to be affected the most.

Both surface water and ground water withdrawals can negatively impact aquatic life. Surface water withdrawals can reduce flood flows (or peak flow regimes), as well as reduce total flow (or discharge) during summer months when irrigation requirements are high and surface water levels are low (Poff and Zimmerman, 2010). Ground water availability is largely affected by ground water withdrawals for irrigation. The consequences of excessive ground water withdrawal can include reduced water quality, prohibitive increases in the costs of pumping, reduced surface water levels through hydrological connections, and subsidence (Reilly et al., 2008). Several regions (e.g., High Plains aquifer, Lower Mississippi River alluvial aquifer) that are already experiencing water shortages could be substantially impacted by increased corn production for ethanol. Ground water withdrawals also have indirect impacts on stream flow. Withdrawals from hydrologically connected aquifers can lower base flow to rivers and streams that depend on ground water to maintain year-round stream flow. In some areas, stream flow has been reduced to zero because of ground water depletion, but in other areas, minimum stream flow during the summer has been sustained because of irrigation return flow to streams (Bartolino and Cunningham, 2003).

3.2.4 Soil Quality

3.2.4.1 Soil Erosion

Soil erosion can have substantial negative effects on soil quality by preferentially removing the finest, uppermost soil particles, which are higher in organic matter, plant nutrients, and water-holding capacity relative to the remaining soil (Brady and Weil, 2000). The soil erosion impact of growing corn or soybeans for biofuel will vary, largely depending on the particular land use/land-cover change and tillage practices. Conversion of uncultivated land, such as CRP acreage, to corn or soybeans for biofuels is the land use change scenario most likely to increase erosion and sedimentation. The USDA CEAP report on the Upper Mississippi River Basin found that for land in long-term conserving cover, like CRP, soil erosion and sediment loss were almost completely eliminated (NRCS, 2010). Moreover, CRP acreage in riparian areas slows runoff, promoting the deposition of sediment, nutrients, and other chemicals. The USDA's Farm Service Agency estimated that, in 2008, CRP land collectively prevented 445 million tons of soil from eroding (FSA, 2009). The soil-erosion effects of converting former or current pasture land to corn will vary depending on prior erosion rates. Pasture land in the U.S. Southern Piedmont region, for example, can exhibit soil stability equal to forested or conservation-tilled land; converting this type of land to conventional corn production will increase soil erosion (Franzluebbers et al., 2000). In contrast, if much of the increase in corn or soybean production comes from a shift from other crops (in 2007, for example, the increase in corn acreage came predominantly from a decrease in soybeans), the effect on soil erosion is likely to be much smaller. Allocation of a higher percentage of corn or soybeans for biofuel production to land currently in agricultural use likely will not alter soil erosion rates.

Tillage practices can mitigate soil erosion on current agricultural lands. Conventional tilling[17] breaks up soil aggregates, increasing erosion by wind and water (Lal, 2003). In contrast, conservation tillage—defined as practices that maintain at least 30 percent of the ground covered by crop residue (Lal, 1997)—can considerably reduce soil erosion (Cassel et al., 1995; Shipitalo and Edwards, 1998). No-till agriculture, a type of conservation tillage, disturbs the soil only marginally by cutting a narrow planting slit. According to the CEAP report, conservation tillage is practiced on 96 percent of all crop acreage in the Upper Mississippi River Basin, with 23 percent in no-till, and only 5 percent in continuous conventional tillage (NRCS, 2010). Conservation tillage practices may also partially mitigate the impact of converting CRP acreage to biofuel corn production (Follett et al., 2009). A majority of CRP acreage in areas of the Midwest are classified as highly erodible land, where tillage practices are generally restricted by the conservation compliance provisions of the 1985 Food Security Act (Secchi et al., 2009). These compliance provisions can require corn-soybean rotations with no-till cultivation (Secchi et al., 2009).

Finally, removal of corn stover beyond a certain threshold may increase soil erosion rates. Due to this and cost concerns, a recent study suggested that only approximately 30 percent of corn stover[18] would be available for sustainable harvesting in the U.S. if erosion rates were to be kept lower than soil loss tolerances (T-values) as defined by the USDA NRCS (Graham et al., 2007). Because of wind erosion, the potential for corn stover removal in the Western Plain states may be particularly limited (Graham et al., 2007). Site cultivation practices may partially compensate for the effects of residue removal. If no-till agriculture were universally adopted, sustainably harvested corn stover supplies are estimated to increase from approximately 30 to 50 percent (Graham et al., 2007). Yet, even with no-till management, corn stover removal rates at or higher than 50 percent have been shown to increase erosion potential (Blanco-Canqui and Lal, 2009a).

3.2.4.2 Soil Organic Matter

Soil organic matter is critical because it retains plant nutrients and water, facilitates carbon sequestration, promotes soil structure, and reduces erosion. The impact of corn and soybean production for biofuel on soil organic matter will depend on the cultivated acreage. Corn production will negatively impact soil quality on acreage where organic matter has accumulated over time—for example, grasslands. If conventional tilling is used, a loss of organic matter both to erosion and to the atmosphere as carbon dioxide due to increased microbial decomposition is likely to occur (Reicosky et al., 1995). Estimates of carbon loss following conventional tilling of previously undisturbed soils range from 20 to 40 percent—although how much carbon is respired to the atmosphere versus lost to erosion is unclear (Davidson and Ackerman, 1993). Assuming carbon loss to the atmosphere, it has been estimated that conversion of grasslands in CRP to corn production would create a carbon debt requiring approximately 48 years to repay (Searchinger et al., 2008). In contrast, increased corn or soybean production on currently cultivated land will have a smaller effect on soil organic matter, particularly where substantial amounts of crop residues are returned to the soil or a cover crop is used (Adviento-

[17] Defined as any tillage practice that leaves less than 15 percent of crop residues on the soil surface after planting.

[18] It should be noted that the removal of crop residues by percent mass is not the same as by percent soil coverage. All the percentages from the studies discussed here are by percent mass, unless otherwise noted.

Borbe et al., 2007; Drinkwater et al., 1998; Lal, 2003). While soil quality degrades over time, yields and production can be maintained by the use of fertilizers both commercial and organic.

The harvesting of crop residues, such as corn stover, removes plant material that would otherwise remain on and potentially be incorporated into the soil. The removal of corn stover therefore has important implications for soil quality, chiefly via effects on soil retention, organic matter content, nutrients, and compaction. Stover removal rates of 25 to 75 percent have been shown to decrease soil organic matter across several soil types even under no-till management (Blanco-Canqui and Lal, 2009a). Therefore, there is concern that high stover removal rates may decrease soil carbon sequestration and lower crop yields (Karlen et al., 2009). Whatever the removal rate for a particular site, it has been recommended that soil erosion and organic matter content be periodically monitored to allow stover removal rates to be adjusted accordingly (Andrews, 2006). The effects of crop residue removals on crop yields have been shown to be highly variable depending on soil type, climate, topography, and tillage management, among other characteristics (Blanco-Canqui and Lal, 2009b). Research to date suggests corn stover removal rates should be determined based on site-specific criteria to maximize soil quality.

3.2.5 Air Quality

Air quality impacts during cultivation and harvesting of corn and soybeans are associated with emissions from combustion of fossil fuels by farm equipment and from airborne particles (dust) generated during tillage and harvesting. Soil and related dust particles (e.g., fertilizer, pesticide, manure) become airborne as a result of field tillage, especially in drier areas of the country. In addition, emissions result from the production of fertilizers and pesticides used in corn and soybean production, and the application of fertilizers and pesticides to each crop. Air emissions associated with cultivation and harvesting of corn and soybeans for biofuel will mostly occur in sparsely populated areas. Subsequent stages in the biofuel supply chain (see Figure 2-3), including feedstock logistics and biofuel production, distribution, and use, also affect air quality and are discussed in Chapter 4.

3.2.5.1 Cultivation and Harvesting

Cultivating and harvesting corn and soybeans require a range of mechanized equipment that utilize different fuels, including diesel, gasoline, natural gas, and electric power (Sheehan et al., 1998a). Generally, equipment used to produce corn and soybeans consumes more diesel than for most other crops, while the rate of gasoline consumption is somewhat less than that of other crops. Primary emissions from fuel use include nitrogen oxides (NO_x), volatile organic compounds (VOCs), carbon monoxide (CO), sulfur dioxide (SO_2) (primarily from gasoline), and coarse and fine particulate matter (PM_{10} and $PM_{2.5}$). Gasoline use may also result in benzene, formaldehyde, and acetaldehyde emissions. For corn, approximately 14 gallons of diesel fuel is used per acre for tillage, harvest, and hauling. Fuel use for tillage comprises more than half of this amount; actual usage depends on soil properties and conditions (Iowa State University, 2009). With respect to corn stover, additional fuel use depends on the method of stover harvest. For example, methods that can simultaneously collect grain and stover will use less fuel than those requiring multiple passes with a harvester. For this reason, one-pass harvesters are currently being developed and tested (Shinners et al., 2009).

Emissions are also associated with generation of electricity used for irrigation water pumping. Irrigation power needs are estimated to range from 3 to 11 kilowatt-hours (kWh) per irrigated acre, depending on the region, with a national average of 8 kWh per irrigated acre. For soybean cultivation, electricity use is estimated to be 4.6 kWh per acre (Sheehan et al., 1998a). Emissions associated with this use depend on the source of the electricity consumed. Coal is the predominant fuel source for electricity in the Midwest, accounting for 71.3 percent of generation in the 12 primary corn-producing states. Coal-fired power plants are significant sources of SO_2, NO_x, carbon dioxide (CO_2), and mercury emissions.

Corn with a moisture content of over 18–20 percent may require drying prior to storage to avoid spoilage (South Dakota State University, 2009). Grain driers use liquid petroleum gas (LPG) and electricity. LPG and electricity use depend on grain moisture content at harvest. For example, typical Midwest grain harvest conditions and yields require 20 gallons of LPG per acre harvested. The exact amount depends on grain moisture conditions at harvest.

3.2.5.2 Fertilizers and Pesticides

Pesticides are commonly used on both corn and soybeans, with corn having more intensive application rates (NRC, 2008, p. 3, as cited in U.S. EPA, 2010b) than soybeans. Corn has the highest nitrogen fertilizer use per acre of any biofuel feedstock. Because soybeans are legumes, they require much lower amounts of fertilizer, particularly nitrogen (NASS, 2006, 2007b). Soybeans have the capacity to derive nitrogen from the atmosphere and therefore require less external nitrogen fertilization than corn, resulting in less nitrogen runoff in the surface water.

Air emissions associated with fertilizer manufacturing and transport include NO_x, VOC, CO, and particulate matter (PM_{10} and $PM_{2.5}$), while pesticide production and blending may result in emissions of 1,3-butadiene, benzene, and formaldehyde.

Application of fertilizers and pesticides may result in releases to the air and volatilization of pesticide ingredients. The primary pollutants associated with the releases to air are benzene and acrolein. The results described are consistent with another study, which found increases in benzene, formaldehyde, acetaldehyde, and butadiene emissions, although that study included feedstock transport and so is not directly comparable (Winebrake et al., 2001). Emissions of CO, NO_x, and SO_2 increased with the use of corn stover as a feedstock in a hypothetical system (i.e., a simulation based on corn stover life-cycle data), with higher NO_x emissions mainly due to denitrification of increased amounts of nitrogen fertilizers added to farm soils (Sheehan et al., 2004).

3.2.6 Ecosystem Impacts

3.2.6.1 Eutrophication, Erosion, and Biodiversity Loss

The impact of increased corn and soybean cultivation on ecosystem and biodiversity depends, in large part, on where crop production occurs and what management techniques are used. Major ecosystem-related impacts that could result from additional corn and soybean production are eutrophication, soil erosion and its associated increase in turbidity of receiving waters and sedimentation in basins, and impacts to biodiversity. Eutrophication can occur as fertilizer application increases nutrient loadings (nitrogen and phosphorus) in surface waters such as streams, rivers, lakes, wetlands, and estuaries (U.S. EPA, 2010b). Increased phosphorus concentration has been correlated with declines in invertebrate community structure, and high concentrations of ammonia nitrogen are known to be toxic to aquatic animals. Severe oxygen depletion and pH increases, both of which are correlated with eutrophication, have been known to cause growth problems and mortality in fish and invertebrates (U.S. EPA, 2010b). In addition, as aquatic systems become more enriched by nutrients, algal growth can cause a shift in species composition. Hypoxia threatens commercial and recreational fisheries in the Gulf of Mexico (U.S. EPA, 2010b) and limits biodiversity (Wang et al., 2007a).

Soil erosion can also lead to an increase in wetland sedimentation, which may, depending on sediment depths, cover viable seeds sufficiently to prevent germination (Gleason et al., 2003). In aquatic ecosystems, sediments increase turbidity and water temperatures and bury stream substrates, limiting habitat for coldwater fish (U.S. EPA, 2006a).

In areas where corn production is already significant, increased corn acreage can further reduce landscape diversity (Landis et al., 2008), which might in turn impact other aspects of biological diversity and the ecosystem services associated with biodiversity. In Iowa, Michigan, Minnesota, and Wisconsin, biological control of soybean aphids was found to decline as the proportion of corn in the local landscape increased, resulting in increased expenditures for pesticides and reduced yields (Landis et al., 2008). In the Prairie Pothole region of Iowa, Minnesota, North Dakota, and South Dakota, landscapes with higher proportions of corn acreage had comparatively fewer grassland bird indicator species (Brooke et al., 2009). If landscape diversity decreases (especially in the case of transforming CRP land into corn production), migratory birds will lose habitat and likely decline in numbers. On CRP lands, several grassland bird species have increased in abundance, and it is estimated that, without the 3 million hectares of CRP in the Prairie Pothole region of the U.S., over 25 million ducks would have been lost from the annual fall migratory flights between 1992 and 2004 (Dale et al., 2010).

The removal of corn stover residues from agricultural corn fields for ethanol production has potential consequences on aquatic ecosystems and local biodiversity. Removing crop residues from farm fields has been shown to affect both terrestrial and soil biota. Crop residue removal has been correlated with decreases in the diversity of biota (Lal, 2009; Johnson et al., 2006).

Intensification of soybean production and pesticide use may also threaten biodiversity and nearby biota (Artuzi and Contiero, 2006; Koh and Ghazoul, 2008; Pimentel, 2006). The change in local habitat from corn-soybean-corn rotation to continuous corn production may decrease the support for biological control in soybean cropping systems, as reduced landscape diversity decreases the habitat availability of many insects and animals in the local region (Landis et al., 2008). Also, agricultural herbicides affect the composition of local plant communities, which then affects the abundance of natural enemy arthropods and the food supply of local game birds (Taylor et al., 2006). Fungicide pollution from runoff events has been shown to impact algae and aquatic invertebrates in areas where soybeans are intensively grown (Ochoa-Acuna et al, 2009).

3.2.6.2 Invasive Plants

Modern varieties of corn and soybeans under production today in the U.S. pose little risk of dispersing seeds or regenerative plant parts or creating hybrids with related plants that will become weeds or invasive plants in the future. Corn and soybeans rarely overwinter successfully in major production areas, but on occasion, seed from the previous year's crop can emerge in the following year and the plants persist through a single growing season as a weed. Such populations of plants do not become a chronic problem, however, because they do not sustain themselves (Owen, 2005). To date, no cases of invasive corn or soybeans have ever been reported in natural areas in the U.S. However, since U.S. seed and biotechnology companies working to improve feedstocks may propagate corn in areas such as Mexico where corn and its progenitors originated, it is possible that novel corn cultivars or their hybrids could spread beyond the cultivated fields and survive. This potential for intermixing genetically modified plants with ancestral land acres is the subject of international scientific and regulatory interest (Mercer and Wainwright, 2008).

The extensive cultivation of row crops that are genetically engineered to resist the herbicide glyphosate may result in indirect effects on other weed species and invasive plants. One study correlated the increased use of this herbicide with the appearance of glyphosate resistance in at least ten agricultural weeds in the U.S.; loss of effectiveness of glyphosate could encourage the use of more toxic herbicides (NRC, 2010).

3.2.7 Assessment

Corn and Soybean Acreage: Between September 2010 and August 2011, approximately 38.4 percent of corn consumed domestically is projected to be converted into ethanol biofuel (NASS, 2010a; ERS, 2010c). Corn acreage has increased over 2005 levels in part due to ethanol demand, and planted acreage is expected to increase from 2008/2009 levels of 85.9 million acres to 90 million acres in 2019 to meet the 15 billion gallons per year annual target under EISA (USDA, 2010c). Currently, 5.6 percent of the soybean harvest goes to biodiesel production, and USDA expects this percentage to increase to 7.8 percent by 2012 and hold steady through 2019. USDA also expects that soybean acreages will hold steady at 76 million acres, though this number may be higher to meet the EISA target. Moreover, it may be necessary to increase acreage yield, or the portion of the soybean harvest that is devoted to biodiesel in order to meet EISA targets (FAPRI, 2010a). Use of corn stover for ethanol production is not expected to increase acreage dedicated to corn.

Land Use/Land Cover Change: Much of the environmental impact of corn starch ethanol and soybean biodiesel production depends on the types of land put into cultivation. To date, most additional acreage has originated from lands currently in crop production. Expanding corn crop production to CRP or previously uncultivated acreage will likely have varying degrees of environmental impacts, depending on site-specific characteristics.

Water Quality: Increasing production of corn for ethanol and soybeans for biodiesel may have implications for water quality. Increased corn and soybean production could increase nutrient, sediment, and pesticide loadings to water bodies, including the Gulf of Mexico, Great Lakes, and Chesapeake Bay. Private drinking water wells could see increases in nitrate and public drinking water systems could see increases in their costs to lower nitrate levels. However, some of the potential increased nutrient loadings from corn grown for ethanol might be offset by increasing per-acre corn and soybean yields and by implementing comprehensive conservation systems. Increased risk of pathogens entering surface waters from application of animal manure fertilizers is also possible. Removal of corn stover could lead to loss of soil surface cover, thereby increasing runoff of nitrogen and phosphorus to surface waters; harvesting corn stover may reduce soil nutrient availability, leading to increased fertilizer applications

Water Availability: The magnitude of environmental impact from increased corn and/or soybean production for biofuel will vary geographically. If corn replaces other crops in the Midwest, water availability will be minimally impacted. Increased corn and soybean production in areas requiring irrigation, such as the Great Plains, will increase water usage, potentially decreasing water availability. Removal of corn stover for ethanol will not affect water availability in most parts of the U.S.

Soil: Impacts of expanding corn and soybean production will vary, depending on the converted land use. Negative soil quality impacts will arise from converting acreage protected with perennial vegetation to conventional corn and soybean production, which will likely increase soil erosion, sedimentation, and nutrient losses. Removal of corn stover for ethanol may lead to a decline in organic matter, decreasing soil carbon sequestration and adversely impacting crop yields. Impacts can be minimized through site-specific BMPs that limit soil erosion and ensure that the amount of residue remaining on the field sustains soil quality and nutrient inputs for subsequent crop productivity.

Air Quality: An increase in the production of corn and soybean for biofuel will likely lead to increased pollution from fossil fuels associated with cultivation and harvesting and from airborne particles (dust) generated during tillage and harvesting. Air emissions also result from the production of fertilizers and pesticides used in corn and soybean cultivation, and the application of fertilizers and pesticides for each crop. Increasing their use will likely increase the volume of emissions.

Ecosystem Health/Biodiversity: Ecosystem health/biodiversity impacts include degradation of aquatic life due to eutrophication, impaired aquatic habitat due to sedimentation from soil erosion, and decreases in landscape diversity. Conversion of CRP lands, which are predominantly grasslands, may lead to declines in grassland birds, ducks, and other wildlife that use these lands as habitat.

Invasive Species: Corn and soybean typically are not invasive in the U.S. corn and soybean-growing regions.

3.2.7.1 Key Uncertainties and Unknowns

Uncertainties and a scarcity of data exist in many key areas concerning environmental impacts of biofuel feedstock production. In particular:

- The impacts of additional soybean and corn production are determined by where the production occurs and the types of management practices employed, including the extent of tile drainage. However, it is highly uncertain where production will occur and the extent to which BMPs will be employed. In particular:

- — Increased corn and soybean yields may offset the need for increased acres in production to achieve EISA goals in 2022. However, the extent to which yield increases will occur is currently unknown, and thus the extent to which increased production of corn and soybeans will occur on marginal lands, CRP, and/or via continuous corn production on existing lands now in rotation with other crops is also uncertain.

- — The extent to which BMPs are currently implemented on cropland nationally is unknown, and the potential for future improvements, including improvements in yield; management of nutrients, pesticides, drainage, and energy use; and erosion control systems, is also uncertain.

- The ability to track impacts will depend on the quality and consistency of monitoring fertilizer and pesticide usage, such as data provided by USDA’s National Agricultural Statistics Services.

- The ability to evaluate current and future water shortages associated with ethanol and biodiesel production is limited by the available data. Annual measurements of the extent of irrigation and amounts of surface and ground water used are not systematically collected nationwide, forcing researchers to use incomplete information to calculate crude water use estimates. Estimates of water use to produce soybeans for biodiesel are even less certain than those for corn production. The connection between water use for corn and soybean production and impacts on water availability and water shortages is also surrounded by uncertainty. The availability of fresh water for a particular use is determined by many factors, including rainfall, soil water retention and ground water recharge, water demand for competing uses, and water contamination; attribution of water shortages to a specific use may be difficult to measure without improvements in data collection (Alley et al., 2002; Reilly et al., 2008).

- The uncertainties regarding the effect of corn and soybean production on soil quality arise predominantly from uncertainties regarding the amount and type of land converted to corn or soybeans as a result of biofuel demand. For example, if the USDA soybean acreage projections hold and additional soybean acreage is not required to meet biodiesel demand, then the impact of soybeans for biodiesel on soil quality is likely to be relatively minimal. However, if soybean acreage increases beyond current levels, determining how much land is being converted, the previous crop-type of that land, and its geographical location will be necessary to assess the impact of this increase on soil quality. More studies on land use/land cover changes as a result of ethanol and biodiesel demand are needed.

- Secondarily, uncertainties regarding the effect on soil quality are caused by lack of detailed land management data. For example, more frequent and detailed data—including geographical location—on tillage practices employed would substantially reduce uncertainties surrounding the soil quality response of producing biofuels.

- The key uncertainties with respect to air quality impacts of increased corn and soybean production are similar to water quality with respect to fertilizer and pesticide use and application. In addition, NO_x emission rates from fertilized soil are highly uncertain and variable as they rely on microbial conversion of fertilizer to nitrate which in turn is influenced by environmental conditions. The extent to which cover crops and tillage practices are employed, both of which can reduce fugitive dust emissions, are also highly uncertain. For corn stover, there are a range of assumptions regarding cropping practices, harvest techniques, and farm inputs that require more study.

- Ecosystem health and biodiversity, including fish and wildlife, are highly impacted by uncertain environmental factors such as nutrient and sediment runoff. Nutrient loadings from row crop production into surface waters depend on a variety of factors, including variations due to weather and are therefore widely variable (Powers, 2007). Regardless, the ability to reduce chemical exposure of biota will be beneficial to the ecosystem and local biodiversity. In addition to resolving uncertainties about those factors, more studies are needed on landscape-level associations between corn and soybean production and terrestrial and aquatic biodiversity, as well as biodiversity-related services such as pollination and natural pest control.

- There is substantial uncertainty regarding the impacts of climate change on regional precipitation patterns and temperatures, which could significantly change water demand and availability, crop yield, runoff, and soil loss.

3.3 <u>Perennial Grasses</u>

3.3.1 Introduction

Perennial grasses are herbaceous plants that grow in successive years from the same root system. They lack the sugar and starch content to be converted directly into ethanol using conventional methods, but can be converted using cellulosic conversion technologies. While cultivation of perennial grasses has potential environmental advantages over traditional row crops such as corn and soybeans, major technological challenges exist for the development of these more advanced biofuel conversion technologies. Currently, no commercial-scale facilities for converting perennial grasses to cellulosic ethanol are operating in the U.S.; however, six switchgrass cellulosic ethanol production facilities are under development (RFA, 2010).

The predominant perennial grasses for biofuels are likely to be monocultures of switchgrass (*Panicum virgatum*) or Giant Miscanthus (*Miscanthus x giganteus)*, hereafter referred to as *Miscanthus*. Research suggests that an aggressive genetics program to create fast-growing strains could increase production of both feedstocks dramatically over current production levels (Vogel and Masters, 1998). The research community is also exploring mixtures of native grassland species—referred to as low-input high-diversity (LIHD) mixtures—as a feedstock (see text box on next page). Compared to their constituent monoculture perennial grasses, LIHD mixtures have often demonstrated higher bioenergy yields (i.e., gallons of biofuel produced per unit of land), and a greater ability to grow in infertile soils, although much less is known about their commercial potential (Tilman and Lehman, 2006). Most research and development has been conducted on monocultures of switchgrass and *Miscanthus*, therefore, these species are the focus of this section.

Switchgrass, a native plant of North America, has historically been grown in the U.S. as forage for grazing livestock (Parrish and Fike, 2005). Recently, it has entered breeding programs and agronomic testing as a biofuel feedstock. *Miscanthus*, which is native to Asia, has been developed and tested as a biofuel feedstock largely in Europe. Considerable genetic variation in both these species has yet to be explored to optimize feedstock production and biofuel refining (Keshwani and Cheng, 2009), but promising traits, including low lignin and ash content, and late or absent flowering periods (Jakob et al., 2009), indicate ample potential for high crop yields and efficient conversion to ethanol (Jakob et al., 2009). While standard irrigation, fertilizer, and pesticide use practices have yet to be developed, recent small-scale farming and larger-scale studies, such as those conducted by the U.S. Department of Energy's Regional Biomass Energy Feedstock Partnership, continue to inform estimates of biofuel perennial grass cultivation and resource requirements (Parrish and Fike, 2005). Farm-scale studies have demonstrated that ethanol yield from switchgrass ranges from approximately 240-370 gallons per acre compared to an average of 330 gallons per acre for corn grain (Schmer et al., 2008).

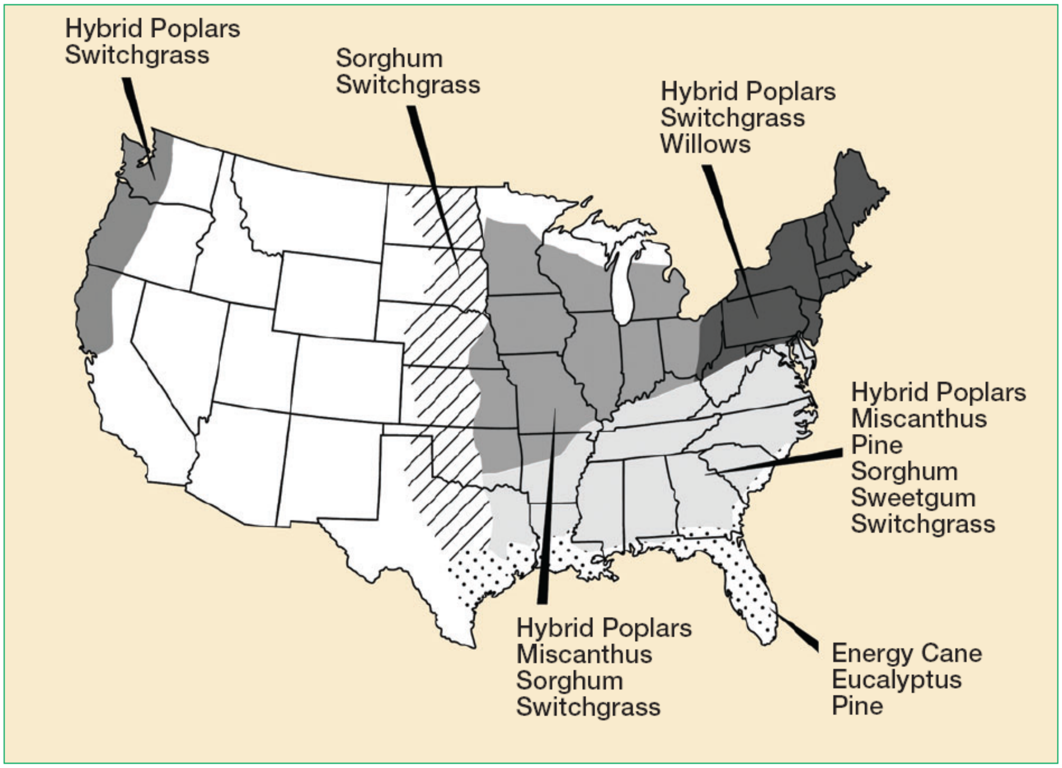
EISA and Section 211(o) of the Clean Air Act limit land conversion for biofuels to existing agricultural land cleared or cultivated prior to Dec. 19, 2007, or land that was non-forested and actively managed or fallow on Dec. 19, 2007 (Clean Air Act, Section 211[o]). As of November 2009, approximately 28 million of the 31.2 million CRP acres were vegetated with mixtures of native or introduced grasses for a variety of environmental purposes, including wildlife habitat, erosion control, and water quality. Economic modeling of global bioenergy markets (POLYSYS) estimates that approximately 8-13 million acres of CRP land and 10-23 million acres of agricultural cropland in the U.S. could possibly (but not necessarily likely) be converted to switchgrass production, depending on economic factors (Walsh et al., 2003). The gross impact on CRP land already growing switchgrass would be minimal, and the estimated combined conversion of "idle" and "pasture" lands to switchgrass production could be between 0.78 and 5.58 million acres (Walsh et al., 2003). Comparable quantitative information is not available for *Miscanthus,* however, high biomass yields on areas with poor soil quality in southern Illinois demonstrate the potential for *Miscanthus* on low fertility lands (Pyter et al., 2004). In addition to CRP land, abandoned cropland is hypothetically available for perennial grass cultivation. Assuming suitable technology and infrastructure exists, an estimated 25 billion gallons of ethanol could potentially be produced annually if switchgrass is grown on the approximately 146 million acres of abandoned agricultural land in the U.S., as long as these lands do not fall under restrictions described in Section 211(o) of the Clean Air Act (U.S. EPA, 2010b, Chapter 6).

Native Grasslands as a Biofuel Feedstock

Recent research has suggested using mixtures of native perennials as a feedstock on marginal or infertile lands (Tilman et al., 2006; Tilman et. al, 2009; Campbell et al., 2008; Weigelt et al., 2009). This practice is limited by several technological and management hurdles, yet also enjoys many environmental benefits not found to the same degree in other feedstocks discussed in this report. Termed "low-input high-diversity" (LIHD) mixtures, they are essentially comprised of several plant species that perform different functions within the community (e.g., high root mass to prevent soil erosion, nitrogen fixation to reduce fertilizer inputs) potentially at different times (e.g. spring versus fall) or the same function in a different manner (e.g., root growth and soil carbon sequestration at shallow versus deeper soil depths). LIHD mixtures, by definition, have more plant biodiversity than other monoculture-based feedstocks. This higher plant biodiversity is often associated with a variety of benefits, including higher stability of production, higher quality of habitat for wildlife, lower potential for invasion of the community, reduced need for chemical inputs (fertilizers, pesticides), and reduced potential for plant disease and crop losses (Fargione et al., 2009; Hooper et al., 2005; Loreau et al., 2002; Reiss et al., 2009). When systems are viewed as a composite of many co-occurring processes (e.g. primary production, soil stabilization, and decomposition), polycultures sustain higher levels of multiple processes, sometimes termed "ecosystem multifunctionality" (Hector and Bagchi, 2007; Zavaleta, 2010). Diverse mixtures also often produce more biomass than their average constituent species grown in monoculture; however, the productivity of the most productive constituent species is in many cases similar to that of the mixture (Cardinale et al., 2006; Loreau et al., 2002; Cardinale, 2007). Although it seems likely that highly productive feedstocks (e.g., switchgrass and *Miscanthus*) managed for maximum production will produce more biomass for biofuel production than LIHD mixtures, there are no direct field-scale comparisons between LIHD and other feedstocks with which to evaluate this assumption. The only comparison to date found that switchgrass grown on productive lands across the Midwestern corn belt (Nebraska, South Dakota, North Dakota) out-produced LIHD grown on unproductive land in Minnesota (Schmer et al., 2008). However, monoculture crops are expected to require more active management (e.g., to prevent losses from pests) than polycultures such as LIHD (Hill et al., 2006; Tilman et al., 2009; Weigelt et al. 2009). Production of a feedstock composed of a mixture of species will likely face greater technological and management hurdles than production of single-species feedstocks. For example, a mixture of species, having variable tissue densities and arrangements in the cropping system, may be more difficult to harvest, transport, and process into biofuel than a relatively uniform feedstock grown from a single species. Much more research is needed in this area to determine the potential role of LIHD as a biofuel feedstock on marginal or infertile lands.

3.3.1.1 Current and Projected Cultivation

Perennial grasses could thrive across many regions of the contiguous U.S. (see Figure 3-5). Since many of these species, including switchgrass, have historically dominated much of the Midwestern landscape, they are well suited to grow over much of the agricultural region.



Source: Dale et al., 2010, updated from Wright, 1994

Figure 3-5: Generalized Map of Potential Rain-fed Feedstock Crops in the Conterminous United States Based on Field Plots and Soil, Prevailing Temperature, and Rainfall Patterns

3.3.1.2 Overview of Environmental Impacts

As production of biofuel from perennial grass becomes technologically and economically viable, demand for perennial grass will increase. This will result in conversion of qualifying land to perennial grasses, the location and extent of which will depend on region-specific agricultural and economic conditions. Perennial grass production will likely require traditional agricultural activities, including pesticide, fertilizer, water, and fuel/energy usage. The intensity of these activities relative to the land management practices they are replacing will determine the extent to which perennial grass production impacts water quality, water availability, air quality, and soil quality. Finally, perennial grass feedstock transport, which often involves seed movement, may result in unintended dispersal and the spread of invasive grasses.

3.3.2 Water Quality

Perennial grasses, sometimes grown as a conservation practice along the margins of agricultural fields to reduce sediment and nutrient runoff into surface water and wetlands, are expected to have fewer water quality impacts than conventional agricultural crops (Keshwani and Cheng, 2009). This will depend, however, on the agricultural intensity of the perennial grass cropping system (e.g., the extent of fertilizer and pesticide use). Table 3-3 shows inputs needed to grow perennial grasses compared to agricultural intensity metrics associated with growing conventional crops.

Table 3-3: Comparison of Agricultural Intensity Metrics for Perennial Grass and Conventional Crops

Metric	Reduction Relative to Corn-Wheat-Soybean Average
Erosion	125 fold
Fertilizer	1.1 fold
Herbicide	6.8 fold
Insecticide	9.4 fold
Fungicide	3.9 fold

Source: Ranney and Mann, 1994.

3.3.2.1 Nutrient Loading

Nutrients—Surface Water Impacts

Relative to annual crops, such as corn and soybeans, production of switchgrass and *Miscanthus* requires less fertilizer and reduces nutrient runoff. Switchgrass is inherently efficient in its nitrogen use, as well as its use of potassium and phosphorus (Parrish and Fike, 2005). Switchgrass and *Miscanthus* are both nutrient-efficient because they store carbohydrates and nutrients in their roots at the end of the growing season (Beale and Long, 1997; Beaty et al., 1978; Simpson et al., 2008). Therefore, the practice of harvesting the above-ground biomass reduces the need for fertilization in subsequent growing seasons. A recent study reported that *Miscanthus* can fix atmospheric nitrogen, which could be a large benefit to its use as a feedstock (Davis et al., 2010). Studies have shown no response in *Miscanthus* growth to nitrogen additions, suggesting these fertilizers are not needed in its production (Clifton-Brown et al., 2007; Danalatos et al., 2007). In contrast, switchgrass yields increase with nitrogen fertilization, with recommended application rates for switchgrass grown for biofuels ranging from 41 to 120 kg nitrogen/ha/year (37 to 107 lbs nitrogen/acre/year), varying by region (McLaughlin and Kszos, 2005). Data for switchgrass and *Miscanthus* have been generally based on experimental plots, and management and yields may differ at the farm-scale. However, if these lower nitrogen fertilization rates hold, average nitrogen losses to surface waters should be lower relative to the production of corn starch ethanol (ORNL, n.d.).

Nutrients— Coastal Waters Impacts

As mentioned above, switchgrass and *Miscanthus* cropping systems are expected to require fewer fertilizer additions compared to traditional row crops, and have been shown to reduce chemical oxygen demand in runoff when used as filter strips (Keshwani and Cheng, 2009). This will minimize their impact on the hypoxic zones of U.S. coastal waters.

3.3.2.2 Sediment

Switchgrass and other perennial grasses have been used as an erosion control management practice to reduce sediment loads from row crops (Hill, 2007; McLaughlin and Walsh, 1998; U.S. EPA, 2009a). Perennial grasses have been shown to reduce erosion 125-fold when compared to an average of corn, wheat, and soybeans (see Table 3-3). Therefore, assuming good agricultural practices, switchgrass production is not expected to increase sediment loads to surface waters.

3.3.2.3 Pesticides

Perennial grasses, such as switchgrass (native to the U.S.), are generally less susceptible to pests than traditional row crops (Oyediran et al., 2004; Keshwani and Cheng, 2009). A 2004 controlled greenhouse study found that recovery of a dominant pest (western corn rootworm) was 0.2 to 82 times more likely from corn than from 20 other grass species native to the Midwest (Oyediran et al., 2004). However, most species are likely to be more susceptible to pests when grown in monocultures as compared to polycultures. The lack of commercial perennial grass production as biofuel feedstock therefore makes it difficult to predict how much pesticide would be needed for this application and what the environmental impacts would be. In non-commercial production, pesticide releases from perennial grass plantings are much less than from corn or soybeans (Hill et al., 2006). Switchgrass plantings use approximately 90 percent less pesticide than row crops (Keshwani and Cheng, 2009). However, herbicides are used initially to establish and maintain switchgrass plantings for harvest. Switchgrass filter strips have been shown to reduce dissolved atrazine and metachlor concentrations in runoff (Keshwani and Cheng, 2009). Information relevant to potential pesticide use for *Miscanthus* in the U.S. is generally lacking; however, researchers in Europe have reported that pesticide requirements are low compared to row crops (Lewandowski et al., 2000).

Of particular concern is how cellulosic feedstock production may impact the spread of the western corn rootworm (WCR), whose soil-borne larval stage is estimated to be responsible for more than $1 billion in annual losses in the U.S. Corn Belt (Rice, 2003). Recent research reported that WCR is able to use *Miscanthus* and several North American grasses as a host, though not as effectively as corn (Oyediran et al., 2004; Spencer and Raghu, 2009). Similar information on WCR use of switchgrass as a host is not available, though perennial grasses generally are more resistant to pests than corn (Lewandowski et al., 2003; Oyediran et al., 2004).

3.3.2.4 Pathogens and Biological Contaminants

The reviewed literature does not directly discuss the effect of perennial grass plantings on pathogens in runoff or the potential for pathogen loads associated with perennial grass management (i.e., from manure used as fertilizer). Since perennial grasses require fewer inputs and take up more impurities from surface water, fewer contaminants are expected from its growth compared to row crops.

3.3.3 Water Quantity

3.3.3.1 Water Use

Switchgrass is an important native grass in prairies across North America and does not require additional irrigation. As such, studies that calculate water use for ethanol produced from switchgrass often assume that the feedstock is rain-fed, requiring no irrigation, and is capable of tolerating moisture deficits (Dominguez-Faus et al., 2009; Wu et al., 2009). Nonetheless, greenhouse and field studies indicate switchgrass significantly increases biomass production with access to ample water (Barney et al., 2009; Heaton et al., 2004). Thus, farmers may irrigate crops to maximize biomass production, though likely at much lower levels than required for row crops.

Two major subtypes of switchgrass that differ in their water use characteristics have been identified in the wild: an upland and a lowland type. The upland type tends to tolerate dry conditions, though there is considerable variation in growth characteristics based on environment, which is likely due to limited crop selection and improvement. The lowland type requires more water (Parrish and Fike, 2005). Switchgrass farmers may be able to minimize water use by cultivating the upland type of switchgrass.

Miscanthus appears to be at least as efficient at using water for growth as corn and likely more so (Beale et al., 1999), though considerable variation exists in the productivity of *Miscanthus* based on the identity of the cultivar, where it is grown, and the irrigation regime (Clifton-Brown et al., 2001; Richter et al., 2008). Published field studies testing *Miscanthus* in the U.S. are limited, however, and water use practices have not been established.

3.3.3.2 Water Availability

Depending on where perennial grasses are grown, whether irrigation is required, and what crops they replace (if any), perennial grass production could improve water availability. Ground water availability, in particular, could be improved in places like Nebraska, where aquifers provide 85 percent of the water to agriculture (Kenny et al., 2009), if perennial grasses replace more water-dependent crops (NASS, 2009). Water availability will be minimally affected in areas requiring little or no irrigation.

3.3.4 Soil Quality

3.3.4.1 Soil Erosion

Both switchgrass and *Miscanthus* have extensive root systems that prevent the erosion of soil and, unlike corn and soybeans, these perennial grasses are not planted on an annual basis, reducing the frequency of soil disturbance. Currently, switchgrass can be planted in conventional tillage and no-till systems, whereas *Miscanthus* is planted in tilled fields (Heaton et al., 2008; Parrish and Fike, 2005). This one-time tillage can increase erosion risk, particularly in *Miscanthus* where plant growth is slow the first year following planting and does not provide substantial ground cover (Lewandowski et al., 2000). In subsequent years, however, *Miscanthus* stands generally have high yields and dense root mats (Heaton et al., 2008; Lewandowski et al., 2000), and likely provide substantial erosion control benefits relative to annually planted crops. Erosion control by switchgrass has received more study than that of *Miscanthus*. Switchgrass has been extensively planted on CRP acreage for erosion reduction, and planting switchgrass in riparian zone grass barriers and vegetation strips has been shown to substantially reduce runoff, sedimentation, and nutrient loss (Blanco-Canqui et al., 2004).

3.3.4.2 Soil Organic Matter

In general, soil organic matter increases more under perennial species than annual species because of the continuous accumulation of plant material (Sartori et al., 2006). Soil carbon is a primary constituent of soil organic matter. The production of both switchgrass and *Miscanthus* can increase soil carbon, but these organic matter benefits are likely to depend on the particular land use replaced and specific management practices. Where perennials are planted on degraded soils with low organic matter content, soil erosion can be reduced and carbon stocks restored (Clifton-Brown et al., 2007; McLaughlin and Kszos, 2005). For example, on such a soil, switchgrass has been predicted to increase soil carbon by approximately 12 percent following one decade of production and harvesting (Garten and Wullschleger, 2000). If perennial grasses replace annual crops, perennials will likely increase soil organic matter, though direct comparisons are limited (Bransby et al., 1998; Schneckenberger and Kuzyakov, 2007). In one such study, relative to reported values for corn, soil carbon increased under *Miscanthus* cultivation when its above-ground vegetation was harvested annually; however, this result varied according to soil type, with carbon increasing in a loamy soil but not in a sandier textured soil (Schneckenberger and Kuzyakov, 2007). Soil organic matter accumulation under these perennials depends, in part, on harvest frequency, and, in the case of switchgrass, on the potential application of nitrogen fertilizer (Lee et al., 2007b). On the other hand, the effect on soil organic matter of preparing previously undisturbed land for these biofuel feedstocks has received little attention to date. Estimates of carbon loss following conventional tilling of undisturbed soils range from 20 to 40 percent (Davidson and Ackerman, 1993). The amount of time needed for these perennials to restore soil carbon lost following site preparation is uncertain.

3.3.5 Air Quality

As mentioned earlier, little is known overall about the extent to which fertilizer, herbicides, and pesticides will be used to increase perennial grass production. Grasses require significantly less nitrogen fertilizer than corn or soybean, and studies indicate that NO_x emissions should decrease when switchgrass is used as a feedstock (Wu and Wang, 2006). However, switchgrass is not currently grown on large scales under typical farm conditions. Nitrogen fertilizer rates are based on field trials, which are not extensive (Wu and Wang, 2006) and may differ from on-farm conditions (Hill et al., 2009). Similarly, switchgrass has been shown to require lower amounts of phosphorus (P_2O_5) fertilizer, which translates to lower SO_2 emissions (Wu and Wang, 2006)

As described earlier in Section 3.3.2.3, perennial grasses are expected to require less pesticide and herbicide than row crops (except when initially establishing perennial grass plantings); however, the lack of experience with commercial perennial grass production as a biofuel feedstock precludes firm conclusions about potential air quality impacts.

As with corn and soybeans, harvesting of switchgrass will involve use of farm equipment, and thus is expected to generate NO_x and PM emissions. However, VOCs and NO_x and PM emissions associated with switchgrass harvesting have been found to be much lower than those associated with corn harvesting (Hong and Wang, 2009). Decreases in VOCs, CO, NO_x, PM_{10}, $PM_{2.5}$, and SO_2 emissions associated with switchgrass production as compared to corn or soybean have been reported (Wu and Wang, 2006; Hess et al., 2009).

3.3.6 Ecosystem Impacts

3.3.6.1 Biodiversity

Models indicate that a greater diversity of birds are supported by switchgrass than by row crops (corn or soy), though some non-priority species such as horned lark (*Eremophila alpestris)* and killdeer (*Charadrius vociferous*) may decline (Murray and Best, 2003; Murray et al., 2003). One study found that perennial grass crops can provide substantially improved habitat for many forms of native wildlife—including ground flora, small mammals, and bird species—due to the low intensity of the agricultural management system (Semere and Slater, 2006). Increases in avian diversity are insensitive to whether switchgrass is strip harvested or completely harvested (Murray and Best, 2003). However, field studies have shown that different species prefer habitats under different management regimes, suggesting that switchgrass cultivation under a mosaic of field ages and management regimes will maximize total avian diversity over a large landscape (Murray and Best, 2003; Roth et al., 2005). Research from Nebraska and Iowa shows that populations of white-tailed deer are not likely to decline following conversion of land from corn to native grassland (i.e., dominated by switchgrass), but may experience contraction of home ranges to areas near row crops, increasing crop losses and the potential for disease transmission among wildlife (Walter et al., 2009). Though similar studies for *Miscanthus* in the U.S. are lacking, research from the United Kingdom shows that non-crop plants from a wide range of families (*Poaceae, Asteraceae,* and *Polygonaceae*) coexist within young *Miscanthus* cropping systems due to a lack of herbicide applications, and support a greater diversity of bird populations than annual row crops (especially of passerines, game birds, and thrushes) (Bellamy et al., 2009). These effects are likely to be transient as fields mature and crop height and coverage become more homogeneous (Bellamy et al., 2009; Fargione et al., 2009). Similar patterns are likely for the U.S. Use of native mixtures of perennial grasses can restore some native biodiversity (Tilman et al., 2006).

3.3.6.2 Invasive Plants

Grasses are successful at reproducing, dispersing, and growing under diverse environmental conditions. This helps explain their dominance across many areas of the globe, and contributes to their potential risk as agricultural weeds and invasive plants. The risk that switchgrass or *Miscanthus* will become an agricultural weed or invasive plant depends on their specific biology and their interaction with the environments in which they are grown. One study noted that well-managed biofuel feedstock production must not only prevent feedstock crops from invading local habitat, but also prevent the crops from genetically invading native species (Firbank, 2007).

Switchgrass produces large amounts of seed, a trait that correlates with the ability to spread, though it remains unclear how much and how far switchgrass seed can disperse. Switchgrass is being bred for vegetative reproduction, tolerance to low fertility soils, and the ability to grow in dense stands (Parrish and Fike, 2005), all of which could increase invasive potential. On the other hand, breeding for traits like sterility can be utilized to reduce the risk of escape and likelihood of negative impacts. For example, hybrid *Miscanthus* cultivars have been bred to produce almost no viable seed.

The location where a feedstock is grown and the interaction between the feedstock and the local environment will be important for determining its invasion potential. Using species native to the area they are cultivated minimizes the risk of invasion into natural areas. Switchgrass is native east of the Rocky Mountains, although a variety could be bred or engineered to be substantially different from local populations. Switchgrass in any form is not native west of the Rockies. One risk assessment of introducing switchgrass to California indicated that it could become invasive relatively easily (Barney and DiTomaso, 2008). The potential for switchgrass to become a weed of other agricultural crops, even within its native range, is not known.

Unlike switchgrass, *Miscanthus x giganteus* (the variety of *Miscanthus* that has been tested in Europe as a biofuel feedstock) is not native anywhere in the U.S. Little information exists about the ability of *M. x giganteus* to disperse from cultivation and persist as a weed or invade natural areas. One risk assessment recommended no restrictions on planting in the U.S. because the plant produces no living seeds and is therefore unlikely to spread easily (Barney and DiTomaso, 2008). A different study, however, noted that *Miscanthus* can spread vegetatively and could undergo genetic changes to produce seeds once more—making it potentially invasive (Raghu et al., 2006). *Miscanthus sinensis* has been grown in the U.S. for landscaping and horticultural purposes. Herbarium specimens and field observations indicate that it can disperse live seeds and persist in areas beyond where it was originally planted. *Miscanthus sinensis,* a species related to Giant Miscanthus, has been grown in the U.S. for landscaping and horticultural purposes, and is also being developed as a biofuel feedstock. Herbarium specimens and field observations indicate that it can disperse live seeds and persist in areas beyond where it was originally planted, including a variety of habitats like pasture, clearcut forests, and residential areas (Quinn et al., 2010). A recent study found that *Miscanthus sinensis* spreads quickly enough to be labeled invasive (Quinn and Stewart, 2010). Some other grass species that have been considered for use as biofuel currently invade wetlands, including giant reed (*Arundo donax*) (Bell, 1997) and reed canary grass (*Phalaris arundinacea*) (Lavergne and Molofsky, 2004).

While feedstock cultivation poses the greatest risk for invasive impacts, reproductive parts from feedstocks could also be dispersed during transport from the field to storage or ethanol-processing facilities. Roads, railroads, and waterways can act as man-made corridors for non-native and invasive plants. Harvested switchgrass possesses living seed and *Miscanthus* can reproduce vegetatively from plant cuttings, both of which may be dispersed during feedstock transport.

One mitigation option for reducing the potentially negative environmental impacts from perennial grass production is avoiding cultivation of feedstocks with a history of invasiveness, especially in places that are climatically similar to where invasion has already occurred. Another option is to breed feedstocks to limit their dispersal into other fields or natural areas (e.g., the sterile *Miscanthus x giganteus*). For instance, sterile, seedless switchgrass cultivars would be less likely to become invasive than current, seed-bearing cultivars. Often, higher reproduction correlates with lower biomass, so aggressive breeding programs to increase biomass and decrease seed production could produce multiple benefits.

Another strategy for managing potential invasiveness is cleaning harvesting machinery and vehicles used to transport harvested feedstock, which would help to decrease unintended dispersal. Though prevention is most desirable, early detection and rapid response mechanisms could also be put into place to eradicate persistent populations of feedstock species as they arise, but before they have the chance to spread widely (DiTomaso et al., 2010). Such early detection and rapid response mechanisms might involve local monitoring networks and suggested mechanical and chemical control strategies (timing and application rate of herbicides, for example) devised by local agricultural extension scientists for specific feedstocks.

3.3.7 Assessment

Perennial grasses are likely to require less pesticide, fertilizer, and water than traditional row crops used for biofuel production (Downing et al., 1995). The benefits of perennial grasses as a feedstock include reduced soil erosion, enhanced soil structure and carbon sequestration, reduced nitrogen loading and sedimentation to waterways, reduced hypoxia in coastal areas, and greater support for populations of non-crop plants as well as animals and soil biota (Fargione et al., 2009; Hill, 2007; Williams et al., 2009). Use of perennial grasses as a biofuel feedstock carries many advantages to ecosystem services and to biodiversity relative to traditional row crops. The magnitude of these advantages depends on resolving some uncertainties and also on whether perennial grasses are replacing CRP land, row crop farmland, or other lands such as pasture land, and whether they are grown in a monoculture or in a mixture of species. The maintenance of landscape-level biodiversity (e.g., including non-cultivated, protected areas nearby) will depend on the spatial arrangement of reserves promoting connectivity and population persistence, local management practices, and potential for biofuel crops and their pests to spread beyond managed boundaries.

3.3.7.1 Key Uncertainties and Unknowns

- Because no commercial-scale facilities exist for converting perennial grasses to cellulosic ethanol, many uncertainties remain about how growing perennial grasses as a feedstock will affect environmental conditions when grown at commercial scales. This holds for all endpoints documented in this report (soil carbon, leaching, etc.) and highlights the need for large-scale studies comparing perennial grass cultivated under a variety of management regimes with row crops and other feedstocks.

- Most existing literature on switchgrass examines the plant's rangeland and ecological purposes; this literature might not be completely applicable to switchgrass used as a biofuel feedstock.

- Much genetic potential for both *Miscanthus* and switchgrass remains to be explored for increasing their feasibility as feedstocks. If researchers are able to develop novel cultivars of these plants with significantly improved yields, there may be less potential for environmental damage from *Miscanthus* and switchgrass production.

- Little is known about usage of fertilizer and pesticides for increasing perennial grass production. The usage of precision management strategies (e.g., minimal fertilization, irrigation, and pest management at specific times) may potentially increase productivity without deleterious ecological impacts. Depending on where these crops are grown and what crops or other land use they are replacing, they may improve water quality relative to the previous land use.

- The water requirements of different grass species in different areas of the country are not documented, and the use and preferred method of irrigation remains to be determined.

- The role of nitrogen fixation in explaining the productivity of *Miscanthus* requires further study and may have large ramifications on the potential use of *Miscanthus* as a feedstock.

- The potential invasiveness of switchgrass in the western U.S. and *Miscanthus* across all the entire U.S. is relatively unknown. Studies to evaluate feedstocks for the biological characteristics associated invasiveness, including rate of seed production, rate and maximum distance of dispersal from field-scale plots, modes of dispersal (e.g., wind, water, bird), rate of hybridization with already invasive relatives, resistance to chemical or mechanical control, etc., are crucial for anticipating and preventing negative impacts and for determining which alternative feedstocks might pose lower risks.

- It remains uncertain whether the continual removal of above-ground biomass will deplete soil nutrients over the long term, particularly on marginal soils. On these soils, it may be particularly critical to harvest after translocation of nutrients back into the root systems.

- More landscape-level research is needed to understand how the distribution of multiple land use systems across a large landscape (e.g., row crops interspersed with perennial biofuel grasses and native habitat) will affect local and regional biodiversity.

3.4 Woody Biomass

3.4.1 Introduction

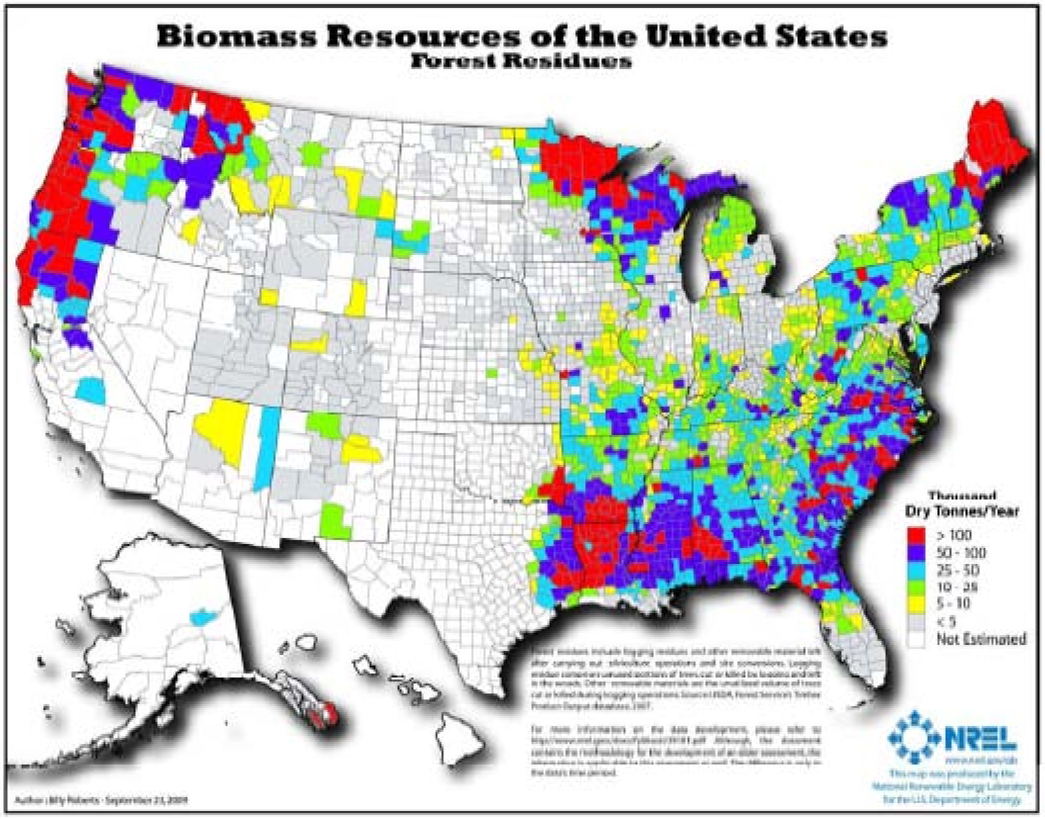
Woody biomass includes trees (e.g., removed or "thinned" from forests to reduce fire hazard or stimulate growth of remaining stands); forest residues (e.g., limbs, tree tops, and other materials generally left on-site after logging); short-rotation woody crops (SRWCs; i.e., fast-growing tree species cultivated in plantation-like settings) and milling residues. Woody biomass is an attractive energy source because of its widespread availability and capacity to store carbon. However, to date woody biomass has been of limited use for energy production, with the exception of pulp and saw mill residues burned to produce heat, steam, and electricity. Woody biomass has been of particular interest as a biofuel feedstock because some forests might benefit from thinning and/or residue removal: removing forest residues from forests could reduce the threat of catastrophic wildfires, at least in some ecosystems, while providing a feedstock for energy production (Gorte, 2009). No commercial-scale biofuel plants using woody biomass as a feedstock are yet in operation, but demonstration and development facilities exist, and woody biomass is projected to be a future source of cellulosic biofuels.

The U.S. has substantial domestic capacity for producing fuel from woody biomass. Estimates of the amount of woody biomass available for biofuel production differ widely and vary by price paid per ton of feedstock. EPA's RFS2 RIA notes that, at $70 per ton, 40 to 118 million dry tons of woody biomass may be available for biofuel production in 2022 (U.S. EPA, 2010b, p. 49). At a currently demonstrated conversion rate of 80 gallons of ethanol per dry ton, up to 9.4 billion gallons of ethanol could be produced from 118 million dry tons (Foust et al., 2009). Additionally, the conversion rate of biomass to ethanol will likely improve in the future.

Under the RFS2 requirements, not all woody biomass would be available. The RFS2 limits the origin of woody biomass to "planted trees and tree residue from actively managed tree plantations on non-federal land cleared at any time prior to December 19, 2007" (U.S. EPA, 2010h, p. 56). Both forest harvesting residues and thinning operations are expected to be the predominant sources of woody biomass for future biofuel use, but SRWCs may be important as well at higher feedstock prices (Perlack et al., 2005; U.S. EPA, 2010b, pp. 38-49; White, 2010). In the following sections, the potential impacts of harvest residues, thinning, and SRWCs are discussed in more detail. For comparison purposes, the environmental impacts of SRWCs are considered in relationship to annual row crops. However, economic analyses suggest that the most likely sources of land for SRWC plantations are CRP or fallow agricultural lands, rather than prime agricultural acres or grasslands; therefore, SRWCs are generally unlikely to replace row crops (Volk et al., 2006; Walsh et al., 2003).

3.4.1.1 Current and Projected Production Areas

The potential sources of woody biomass vary by region of the country, and only SRWC plantations are likely to result in land use/land cover changes. Forest harvest residues are produced in major forest harvesting areas, predominantly in places such as the upper Lake States, the Southeast and the Pacific Northwest (see Figure 3-6). Since these residues will most likely be collected as a by-product of harvesting operations, the use of forest harvest residues is unlikely to produce land use/land cover changes (Williams et al., 2009). However, a rise in price paid per ton for woody biomass may provide an incentive for additional harvesting. Woody biomass from forest thinning will also occur in major forest harvesting areas, and potentially in areas of high wildfire risk. In contrast, SRWC plantations can have substantial land use/land cover effects.



Source: Milbrandt, 2005.

Figure 3-6: Estimated Forest Residues by County

3.4.1.2 Overview of Environmental Impacts

Several activities associated with woody biomass as a feedstock may impact the environment. In the case of forest thinning and residue removal, there may be a direct environmental impact of biomass removal, as well as an impact from operation of forestry machinery. In the case of SRWCs, traditional forestry and agricultural activities undertaken during feedstock cultivation and harvest, such as pesticide, fertilizer, water, and fuel/energy use, have the potential to impact the environment. In addition, the choice of tree species may influence the risk of establishment, invasion, and impact during both feedstock production and transport. All these activities can alter air quality, water quality, water availability, and soil quality, with resulting impacts on ecosystems, though the extent of the impacts depends on each activity’s intensity.

3.4.2 Water Quality

Use of woody biomass as a feedstock can impact water quality, primarily through nutrient runoff and sedimentation. However, the impacts of harvesting trees or removing forest residues can be limited through implementation of forestry best management practices. The extent to which SRWCs have a lower water quality impact than conventional crops will depend on the agricultural intensity of the short rotation woody crops production system (e.g., the extent of fertilizer, pesticide use and replanting interval). Table 3-4 shows inputs needed to grow SRWCs compared to agricultural intensity metrics associated with growing conventional crops.

Table 3-4: Comparison of Agricultural Intensity Metrics for Short-Rotation Woody Crops and Conventional Crops

Metric	Reduction Relative to Corn-Wheat-Soybean Average
Erosion	12.5 fold
Fertilizer	2.1 fold
Herbicide	4.4 fold
Insecticide	19 fold
Fungicide	39 fold

Source: Ranney and Mann, 1994.

3.4.2.1 Nutrients

The literature is mixed on whether residue removal increases (Kreutzweiser et al., 2008) or decreases (Lundborg, 1997) nutrient loads to surface water bodies, including wetlands. The impacts of removing tree harvest residues on nutrient loads vary depending on topography (slope), soil nutrient content, and the chemistry of the residues themselves (Titus et al., 1997). Compared to forest residue removal, moderate forest thinning typically does not increase loss of soil nutrients to ground or surface waters (Baeumler and Zech, 1998; Knight et al., 1991).

Forestry Best Management Practices (BMPs) such as buffer zones (vegetated setbacks from water bodies) are used to reduce water quality impacts. Careful planning to minimize the construction of roads and stream crossings or the use of portable stream crossing structures can help reduce erosion and sedimentation (Aust and Blinn, 2004; Shepard, 2006). Other BMPs include: using energy efficient machinery, minimizing traffic in buffer zones and choosing low-impact equipment that is of the appropriate size and scope for the site (Phillips et al., 2000). The draft 2010 National Report on Sustainable Forests by USDA's U.S. Forest Service suggests widespread adoption of forestry BMPs to protect water resources, although many states failed to respond to a request for data (U.S. Forest Service, 2010). If practices are followed, impacts can be minimized; outreach, education, and monitoring to ensure implementation and effectiveness are ongoing.

As described above, SRWCs are unlikely to directly replace row crops; however, for comparative purposes, it is noted that nutrient losses from SRWCs are in general considerably less than in annually cropped systems, depending in part on the harvesting and replanting interval. In willow plantations, the recommended fertilization rate is 89 pounds of nitrogen per acre (100 kg/hectare) every 3 years, which equates on an annual basis to approximately 22 percent of the average rate for corn production (Keoleian and Volk, 2005; NASS, 2006). In the first year or two following planting, SRWC plantations can exhibit losses of nitrogen at rates comparable to conventional grain production, yet following this initial establishment phase, nitrogen losses decline to low levels (Aronsson et al., 2000; Goodlass et al., 2007; Randall et al., 1997). A comparison of nutrient exports from a short-rotation poplar stand and a native forest found no difference (Perry et al., 1998), and measurements of nitrogen in ground water and leaching from established willow plantations generally show little eutrophication potential for aquatic ecosystems (Keoleian and Volk, 2005). In coppiced systems, where trees are harvested at the ground level and re-grow from the stump, the harvesting of the aboveground portion of the tree appears to have little impact on nitrogen leaching (Goodlass et al., 2007). Losses can be substantially higher when the stand is replanted (Goodlass et al., 2007). Longer rotation lengths would likely improve nutrient retention on-site and reduce losses to waterways.

3.4.2.2 Sediment

Forest soils generally exhibit low erosion rates and thus small sediment losses to surface waterways (Neary et al., 2009). However, when forests are harvested and the soil prepared for the next stand without using BMPs, erosion rates can increase significantly (McBroom et al., 2008). Harvesting residues left on-site physically shield soil particles from wind and water erosion, and promote soil stability through the addition of organic matter. Thus, removal of harvest residues is an element of harvest operations that could increase erosion and associated sediment loading to surface waters, especially on steeper slopes (Edeso et al., 1999). Thinning can also increase erosion and sediment loads to surface waters, depending on the site characteristics and the methods used (Cram et al., 2007; U.S. Forest Service, 2005; Whicker et al., 2008). Research indicates that proper use of BMPs, such as road design and buffer zones, can significantly reduce sediment impacts to surface waters (Aust and Blinn, 2004; Shepard, 2006). In addition, erosion rates at harvested sites decline once vegetation re-colonizes the site (Aust et al., 1991; Miller et al., 1988). See Section 3.4.4.1 for discussion of impacts of SRWCs on soil erosion and sedimentation.

3.4.2.3 Pesticides

Pesticides might be used with SRWCs; for purposes of comparison, it is noted that the amount used would be significantly less than that for corn or soybeans (Ranney and Mann, 1994).

3.4.3 Water Quantity

3.4.3.1 Water Use

The utilization of harvest residues from mature stands of trees and thinning does not require additional water use at the feedstock production stage.

For the most part, growth of SRWCs will likely occur in areas with high water availability, such as the Northeast, Southeast, and Northwest. Because they are usually not irrigated, trees require less total water than row crops (Evans and Cohen, 2009). However, they can still have a large impact on regional water availability due to their much higher evapotranspiration rate. In places where high-intensity tree plantations replace existing ecosystems with lower evapotranspiration rates, the potential for increased water consumption is significant. A study of southern pine in the Southeast found that an additional 865 gallons of water is consumed per gallon of ethanol produced from woody biomass (roughly 1,300 gallons of water per gallon of gasoline equivalent), due to land conversion for woody biomass production (Evans and Cohen, 2009). Further, in certain locations and in some years, additional irrigation water may be required to maintain high biomass accumulation (Hansen, 1988). Precision application systems can reduce the amount of water applied.

3.4.3.2 Water Availability

Use of forest harvest residues and biomass from thinning should have little or no effect on water availability at the feedstock production stage. Plantations of SRWCs may reduce runoff into streams and rivers compared to traditional row crops like corn and soybeans, potentially benefiting water quality (Updegraff et al., 2004). However, some experts warn that reduced runoff coupled with high water requirements could reduce or eliminate stream flow (Jackson et al., 2005). In places with seasonal flooding, modulation of surface water flow closer to pre-agricultural development levels could possibly mitigate flooding (Perry et al., 2001).

3.4.4 Soil Quality

3.4.4.1 Soil Erosion

The soil erosion impacts of SRWCs will depend on harvesting and planting frequencies; impacts are lower when time between planting intervals is longer. Short-rotation woody crops require intensive soil preparation for successful establishment, and it is during this brief establishment phase that erosion rates can be a high (Keoleian and Volk, 2005). For example, higher sediment losses were observed within the first 3 years of seedling establishment in sweetgum (*Liquidamber styraciflua*) plantations compared to no-till corn or switchgrass (Nyakatawa et al., 2006). The slow-developing canopy failed to provide adequate ground cover to protect against erosion as a result of rainfall (Nyakatawa et al., 2006). However, in established SRWC plantations, soil erosion rates are likely much lower than those of annually harvested row crops. The use of a cover crop can also significantly reduce erosion caused by SRWC establishment (Nyakatawa et al., 2006), and the soil erosion effects of SRWCs are likely to be lower under a coppicing system, which reduces the frequency of soil disturbance by keeping the root systems intact. Willows are generally managed by the coppicing system and harvested at 3- to 4-year intervals for a total of 7 to 10 harvests (Keoleian and Volk, 2005). This allows 21 to 40 years between soil disturbances.

3.4.4.2 Soil Organic Matter

Harvesting of forest residues removes plant material that could otherwise become soil organic matter. A review analysis suggested that, on average, a complete, one-time removal of forest residues slightly decreases soil organic matter in coniferous forests, but may not affect levels in hardwood or mixed stands (Johnson and Curtis, 2001). Leaving logging residues is important for soils with low organic matter content, and repeated harvesting of residues in the same location could lead to overall declines in soil organic matter (Thiffault et al., 2006). Further research is needed to determine the cumulative effect of repeated removals. Thinning of forests has been shown to reduce carbon in forest floor layers, but less evidence is available regarding its impact on mineral soil organic matter levels (Grady and Hart, 2006; Jandl et al., 2007). The effect of thinning over the long-term will depend on both the frequency and intensity of the specific thinning operations.

Production of SRWCs can add organic matter to the soil, sequestering carbon, but the net soil organic matter benefits of these crops depend on land-use change and time between harvests. Generally, soil organic matter, including carbon, is initially lost when a forest is planted because the amount of organic matter entering the soil from the reestablishing plants is typically small and is exceeded by decomposition (Paul et al., 2002). Over time, substantial amounts of organic matter accumulate in the trees, the forest floor layer and the soil, greatly exceeding the carbon contained in abandoned agricultural systems (Schiffman and Johnson 1989; Huntington 1995; Richter et al., 1999). The amount of time it takes for soil carbon to re-accumulate varies. In hybrid poplar plantations in Minnesota, it was estimated to take 15 years to meet the carbon levels of the agricultural field replaced (Grigal and Berguson, 1998). A review study suggested that on average it can take 30 years to exceed those of abandoned agricultural fields; though when the forest floor was also considered, carbon accumulation rates were higher, reducing the time needed to regain carbon from the initial forest establishment (Paul et al., 2002). Overall, if frequently harvested SRWCs replace longer rotation, managed forest lands, then the net effect on soil organic matter is likely to be negative; but, if they are grown using longer rotations, particularly on degraded former agricultural land, substantial amounts of organic matter are likely to added to the soil (Schiffman and Johnson, 1989; Huntington 1995).

3.4.4.3 Soil Nutrients

Use of harvesting residues removes a potential source of soil nutrients that can be utilized by the regenerating forest. Harvesting with residue removal generally leads to declines in soil nutrients and forest productivity, but in some cases, it can be sustainable for at least one rotation (McLaughlin and Phillips, 2006; Thiffault et al., 2006). The cumulative effects of repeated removals from the same site are likely negative, but require further study. Residue removal has been suggested as a management technique to reduce nitrogen in forests that receive high atmospheric deposition, such as in the northeastern U.S. (Fenn et al., 1998). However, this may lead to depletion of calcium and other nutrients critical for plant growth (Federer et al., 1989). Overall, residue removal may be less problematic on high fertility soils compared to coarser-textured, low fertility soils. The risk posed to soil nutrients by thinning is likely to be much smaller than that of the removal of harvesting residues (Luiro et al., 2010).

There is concern that continual harvesting of SRWCs will deplete soil nutrients over the long-term (Adegbidi et al., 2001). Commercial fertilizers or organic waste products, such as municipal effluent, can be used to offset these losses (Stanton et al., 2002). Nutrient removal from such effluents by SRWCs may provide an additional environmental benefit, though it remains unclear how much nitrogen, other nutrients, or contaminants might leach from these systems if this technique is used.

3.4.5 Air Quality

Few data are available for evaluating air emissions from SRWCs such as hybrid poplar and willow. As with switchgrass, SRWCs require less tillage (reducing fugitive dust emissions) and fewer applications of fertilizer (reducing emissions associated with fertilizer production and application). However, some species such as poplar and willow that are potential feedstocks for either cellulosic ethanol or biodiesel are significant emitters of biogenic VOCs such as isoprene. Compared to non-woody crops that emit relatively little isoprene, extensive plantations of these trees have the potential to significantly impact ozone concentrations, although this effect will be highly sensitive to environmental conditions, preexisting vegetative cover, and the presence of other atmospheric chemicals, especially NOx (Hess et al., 2009; U.S. EPA, 2006b).

3.4.6 Ecosystem Impacts

3.4.6.1 Biodiversity

Tree harvesting activities can impact aquatic biodiversity in a number of ways. For example, removal of woody biomass by harvesting of forest residues or thinning in riparian areas may reduce the woody debris in headwater streams, which is an important component for aquatic habitat (Angermeier and Karr, 1984; Chen and Wei, 2008; Stout et al., 1993; Thornton et al., 2000). In addition, tree canopies over streams help maintain cooler water temperatures conducive to cold-water smallmouth bass, trout, or salmon populations (Binkley and Brown, 1993; U.S. EPA, 2006c). These benefits may be lost when trees are harvested.

There is some evidence that planting SRWCs can improve species habitat relative to agricultural crops (Christian et al., 1998). Several studies have documented that bird species diversity on woody biomass plantations is comparable to that of natural shrubland and forest habitats (Dhondt et al., 2007; Perttu, 1995; Volk et al., 2006), though this is not always the case (Christian et al., 1998). Bird and small mammal species found on SRWC plantations tend to be habitat generalists that can also use open habitats like agricultural lands, while birds and small mammal species in mature forests are more specialized and require forest cover (Christian et al., 1998). If understory plants become prevalent in SRWC plantations, species diversity can increase due to increases in habitat complexity (Christian et al., 1998).

3.4.6.2 Invasive Plants

Like perennial grasses, woody plants cultivated for biofuel feedstock can become invasive. However, because many woody plants have a longer life cycle than many (though not all) grasses, they tend to reproduce and spread more slowly, making the evidence of their invasion and effects on natural areas less immediate. Trees used in forestry can sometimes be highly invasive, negatively affecting biodiversity and water availability (Richardson, 1998).

Proposed SRWCs, such as willow or poplar, are native or hybrids of natives in the U.S., but *Eucalyptus* species, which are non-native, may pose an invasive risk. *Eucalyptus* is an important genus of forestry plants worldwide, and its future development as a biofuel feedstock in plantations has been discussed for Florida (Rockwood et al., 2008). Several species of *Eucalyptus*, including *E. globulus* and *E. grandis*, have been introduced to Florida and bred conventionally and using biotechnology for traits like cold tolerance. The intent is to expand their future cultivated range to include much of the Southeast. While introduced *Eucalyptus* or their improved varieties have not become invasive in the Southeast, *E. globulus* is a listed invasive plant in California, and recently, several cultivars of *E. grandis* were found to be potentially invasive by the Institute of Food and Agricultural Science at the University of Florida and are recommended for planting only under limited conditions. Reassessment of the species will take place again in two years after continued monitoring for invasion.

3.4.7 Assessment

Current environmental impacts of production and use of woody biomass as a biofuel feedstock are negligible, since no large-scale, commercial operations are yet in existence to create demand for this feedstock. However, estimates suggest that the potential for biofuels made from woody biomass is substantial, with predominant sources coming from forest harvest residues, thinning, and SRWCs. Of these, the removal of harvesting residues from logging sites is likely to have the most negative impacts on soil and water quality. Complete removal of residues poses the risk of increased nutrient and sediment losses to waterways, and decreased plant nutrient availability and forest productivity. In comparison, moderate thinning regimes will have relatively few impacts on soil and water quality, particularly on stable slopes and finer-textured soils.

The environmental effects of SRWCs as a source of woody biomass are more complex, since these require a shift in land use/land cover type. In general, SRWCs are expected to result in lower nutrient and sediment loads to surface waters relative to that of row crops, especially once the canopy is established. Woody biomass species require fewer inputs of fertilizer and pesticides, resulting in reduced runoff of these substances into surface and ground water. Woody biomass production requires considerable water use, but if undertaken in appropriate regions with adequate water supplies, water quality benefits may outweigh possible water availability drawbacks.

3.4.7.1 Key Uncertainties and Unknowns

- Woody biomass is not yet converted to biofuel on a large-scale; this creates considerable unknowns and uncertainties when projecting the potential environmental effects, both positive and negative, of this feedstock.

- Specific environmental impacts will vary, depending on soil type, soil chemistry, topography, climate, and other factors (e.g., the land use SRWCs would replace).

- Lack of information about the amount and relative proportion of woody biomass that would come from harvest residues, thinning, and SRWCs to support large-scale operations creates substantial uncertainty. The potential effects of harvest residues and thinning are easier to assess because a body of literature from other forestry applications does exist. Even so, uncertainties arise from variations in the percent of residues removed during harvesting and in the degree of thinning, which can range from small to large proportions of the existing stand.

- Quantifying impacts of SRWCs to ecosystems and biodiversity will depend on knowing where and under what agronomic conditions SRWCs are grown and how they are managed. Uncertainty about these factors limits understanding of the potential impacts of this feedstock. For example, it is not known whether repeated removal of biomass from SRWCs will deplete soil nutrients over the long term, particularly on marginal soils.

3.5 Algae

3.5.1 Introduction

Algae are of interest as a biofuel feedstock because of their high oil content, low water demand, ability to recycle waste streams from other processes, and ability to grow on marginal lands (EPA, 2010b). Algae production demands much less land area per gallon of fuel produced compared to most other feedstocks.

Research and pilot studies have shown that the lipids and carbohydrates in microalgae[19] can be refined and distilled into a variety of biodiesel- and alcohol-based fuels, including diesel, ethanol, methanol, butanol, and gasoline. Algae also have the potential to serve as feedstock for other types of fuels, including bio-oil, bio-syngas, and bio-hydrogen. This section focuses on the use of algae for biodiesel, because biodiesel is the most likely near-term pathway for algal use as biofuel.

There are many different types of algae, methods to cultivate them, and processes to recover oil from them. Algae grown photosynthetically are limited to growth during daylight hours and require carbon dioxide. Heterotrophic algae, which do not utilize photosynthesis, can be grown continuously in the dark, but require a fixed carbon source such as sugars because they cannot use carbon dioxide directly (Day et al., 1991). Cultivation of algae feedstocks can take place in photobioreactor facilities with closed-cycle recirculation systems or in open-system-style impoundments. *Open systems* use pumps and paddle wheels to circulate water, algae, and nutrients through shallow, uncovered containments of various configurations. *Closed systems* employ flat plate and tubular photobioreactors and can be located outdoors or indoors. Variations include hybrid (combined open and closed) cultivation and heterotrophic cultivation (which uses organic carbon instead of light as an energy source). Different algae cultivation strategies are being studied to determine which is most suitable for supporting large-scale biofuel production (Chisti, 2007; U.S. EPA, 2010b).

Harvesting requires that the algae be removed, dewatered, and dried. Dewatering is usually done mechanically using a screw press, while drying can use solar, drum, freeze, spray, or rotary techniques (NRDC, 2009). After harvesting, the biofuel production process begins when oil is extracted from the algae through chemical, mechanical, or electrical processes (U.S. EPA, 2010b, p. 61). Algal oil can then be refined with the same transesterification process used for other biofuel feedstocks such as soybeans.

While the different methods of algae cultivation and recovery will clearly have very different environmental impacts, such as energy consumption and chemical use and disposal, it is premature to draw definitive conclusions about these impacts, given the nascent state of cultivating algae for biofuel.

[19] The term “microalgae” refers to photosynthetic and heterotrophic organisms too small to be easily seen with the naked eye—distinguished from macroalgae, otherwise known as seaweed. Macroalgae is generally not grown as an energy crop. In this report the terms “algae” and “microalgae” are used interchangeably.

3.5.1.1 Current and Projected Cultivation

Land use consideration is one of the primary drivers behind interest in algae as a biomass feedstock. Relative to other feedstock resources, algal biomass has significantly higher productivity per cultivated acre. For example, meeting half the current U.S. transport fuel demand with soybean-based biodiesel would require an arable land area in excess of 300 percent of the area of *all* 2007 U.S. cropland, while the land area required to meet those same fuel needs using algal biodiesel would be less than 3 percent of all U.S. cropland in 2007 (Chisti, 2007). Moreover, algae cultivation does not require arable land. Algae's lack of dependence on fertile soil and rainfall essentially eliminates competition among food, feed, and energy production facilities for land resources (Muhs et al., 2009). Because algae-based biofuel production facilities do not require specific land types, they may be sited closer to demand centers, reducing the need to transport significant quantities of either biofuel or feedstock from one region of the country (e.g., the Midwest) to another (e.g., coastal population centers).

Proximity to input sources, such as carbon dioxide sources, and output markets, as well as the availability of affordable land, will likely drive algae production facility siting decisions. The U.S. Southwest is viewed as a promising location for economic algae-to-biofuel cultivation due to the availability of saline ground water, high exposure to solar radiation, and low current land use development. Based on pilot studies and literature on algae cultivation, likely areas for siting algae-based biofuels facilities also include coasts, marginal lands, and even co-location with wastewater plants (Sheehan et al., 2004; U.S. EPA, 2010b). Algae grown in conjunction with animal and human wastewater treatment facilities can reduce both freshwater demands and fertilizer inputs, and may even generate revenue by reducing wastewater treatment costs. U.S. companies are already using wastewater nutrients to feed algae in intensively managed open systems for treatment of hazardous contaminants (Munoz and Guieysse, 2006).

3.5.1.2 Overview of Environmental Impacts

Algae-based biofuel production systems are still being investigated at the pilot stage using smaller-scale prototype research facilities. Evaluating the potential environmental and resource impacts of full-scale production is highly uncertain because much of the current relevant data is proprietary or otherwise unavailable, and many key parameters are unknown, including where and how algae will be produced and what species and strains of algae will be used as feedstocks.

Algae cultivation can require the use of pesticides, fertilizers, water, and fuel. Each of these activities, in turn, can impact air quality, water quality, and water availability. (Soil quality is not a concern.) In addition to these impacts, there is potential for invasive algae strains to escape from cultivation (NRDC, 2009). Industrial oil extraction and biodiesel production, biodiesel and byproduct transport and storage, and biodiesel and byproduct end use also entail environmental impacts, which are discussed further in Chapter 4.

3.5.2 Water Quality

Scaled production of algae oil for biofuels has not yet been demonstrated; therefore, water quality impacts associated with large-scale use of algae-based biofuels are currently speculative. Wastewater is a key factor influencing water quality impacts of algae production facilities, including whether wastewater is used as a water source for algae cultivation, and whether wastewater is discharged from the algae cultivation site. According to the National Resources Defense Council, wastewater from the dewatering stages of algae production could be released directly, sent to a treatment unit and released, or recycled back as make up water, depending on the process (NRDC, 2009). Depending on the treatment requirements, release of wastewater could potentially introduce chemicals, nutrients, additives (e.g., from flocculation), and algae, including non-native species, into receiving waters.

Co-locating algae production facilities with wastewater treatment plants, fossil fuel power plants, or other industrial pollution sources can improve water quality and utilize waste heat that contributes to thermal pollution, while reducing freshwater demands and fertilizer inputs (Baliga and Powers, 2010; Clarens et al., 2010). By co-locating these facilities, partially treated wastewater acts as the influent to the algae cultivation system. Algae remove nutrients as they grow, which improves the quality of the wastewater and reduces nutrient inputs to receiving waters. If fresh water or ground water is used as the influent, nutrients must be added artificially in the form of fertilizer.

Significant environmental benefits could be associated with the ability of algae to thrive in polluted wastewater. Algae can improve wastewater quality by removing not only nutrients, but also metals and other contaminants, and by emitting oxygen. Thus, algae can effectively provide some degree of "treatment" for the wastewater (Darnall et al., 1986; Hoffmann, 1998).

3.5.3 Water Quantity

3.5.3.1 Water Use

Water is a critical consideration in algae cultivation. Factors influencing water use include the algae species cultivated, the geographic location of production facilities, the production process employed, and the source water chemistry and characteristics. Estimates for water consumption vary widely, ranging from 25 to 974 gallons of water per gallon of biodiesel produced (U.S. EPA, 2010b). EPA has estimated that an open-system-type biofuel facility generating 10 million gallons of biofuel each year would use between 2,710 and 9,740 million gallons of saline water each year; a similar scale photobioreactor-type facility would use between 250 and 720 million gallons of saline water annually (U.S. EPA, 2010b, Table 2.4-56, p. 426).

The harvesting and extraction processes also require water, but data on specific water needs for these steps are limited (U.S. EPA, 2010b). Compared to the water required for algae growth, however, demands are expected to be much lower.

3.5.3.2 Water Availability

Depending on the cultivation system, algae production could exacerbate or create water availability problems, especially in promising locations like the Southwest, which are already experiencing water shortages. However, the water used to grow algae does not have to be high-quality fresh water. Algae can thrive in brackish water, with salt concentrations up to twice that of seawater (U.S. EPA, 2010b), as well as in contaminated wastewater such as agricultural, animal, or municipal effluent; or even coal, pharmaceutical, or metal plating wastewater (NRDC, 2009). Thus, competition for freshwater resources may be mitigated by siting facilities in areas that can provide suitable brackish or wastewater sources.

In addition to the water quality benefits described above, co-locating algae production facilities with wastewater treatment plants can reduce water demands (Clarens et al., 2010). The water availability impacts of algae production for biofuel can also be mitigated in large part using photobioreactors, which require less water and land area than open systems (U.S. EPA, 2010b, Table 2.4-56, p. 426).

3.5.4 Soil Quality

Very little peer-reviewed literature exists on the soil impacts of algae production because these impacts are likely to be negligible and have therefore not been the subject of much study. Presumably, the primary mechanism affecting soil quality would be transport and migration into soil of wastewater, particularly highly salinated wastewater, that has been released into freshwater ecosystems (NRDC, 2009).

3.5.5 Air Quality

The effects of algae-based biofuels on air quality have received little attention to date in peer-reviewed literature. As a result, additional research is needed to determine whether anything unique to algae production processes would raise concern about air emissions.

Open or hybrid open systems appear to have greater potential to impact air quality compared to enclosed photobioreactors, given the highly controlled nature of the latter systems. No studies are yet available, however, to characterize or quantify emissions associated with open systems used to produce algae for biofuel. Studies have measured air emissions of open-system algae ponds that are part of wastewater treatment systems (Van der Steen et al., 2003), but these studies may have very limited applicability to open systems for commercial-scale production of algae oil for biodiesel. Additional research will be required to estimate and characterize emissions from pumping, circulation, dewatering, and other equipment used to produce algae for biofuel.

3.5.6 Ecosystem Impacts

3.5.6.1 Biodiversity

Algal production has fewer biodiversity impacts than production of other feedstocks because algae typically require less fertilizer, pesticide, and water than do other feedstocks, and because algal production plants may be co-located with wastewater treatment plants. As mentioned above, the location of algae production facilities will be a key factor affecting the potential for impacts. Using wastewater to capture nutrients for algal growth could help reduce nutrient inputs to surface waters (Rittmann, 2008). Algae also require low inputs of fertilizers and pesticides compared to other feedstocks, which may translate into fewer ecological impacts to aquatic ecosystems (Groom et al., 2008). Production facilities for algae that need sunlight to grow could be located in arid regions with ample sunlight (Rittmann, 2008); however, growing algae in areas with limited water resources could impact the amount of water available for the ecosystem because of draws on ground water. It is unknown what impacts an accidental algae release might have on native aquatic ecosystems, particularly if the algae released have been artificially selected or genetically engineered to be highly productive and possibly adaptable to a range of conditions.

3.5.6.2 Invasive Algae

The potential for biofuel algae to be released into and survive and proliferate in the environment is, at present, highly uncertain. This potential will vary, depending on what species and strains of naturally occurring, selectively bred, or genetically engineered algae are used and how they are cultivated.

The risk of algae dispersal into the environment is much lower in closed bioreactor systems than open system production, though unintentional spills from bioreactors in enclosed production facilities are possible. High winds blowing across open systems may carry algae long distances, depositing them in water bodies, including wetlands, near and far. Designers of coastal algae production plants must take into account hurricanes and other severe storms that could disperse algae over large areas. Wildlife, including birds, may also disperse algae. Closed systems, in addition to limiting algae dispersal, have the benefit of protecting algal media from being contaminated with other microbes, which could compete with the cultivation strains for nutrient resources.

Effluent from algal biomass dewatering processes may contain residual algae, which could thrive in receiving waters. Treatment strategies will need to be developed to prevent algae in effluent from contaminating the surrounding ecosystem.

The ability of cultivated algae to survive and reproduce in the natural environment is unknown: one theoretical study suggests that some, but not all, strains with the most desirable commercial characteristics would be out-competed by native algae (Flynn et al., 2010). Further empirical work is critical to determine competitive and hybridizing abilities of biofuel algae in the natural environment and to measure possible effects on algal community dynamics and ecosystem services.

3.5.7 Assessment

3.5.7.1 Current and Future Impacts

With the exception of nutrients, open system cultivation requires far more resources than photobioreactor systems. Regardless which type of system is used commercially, future increases in production of algal biomass have the potential to impact water availability and quality, air quality and atmospheric climate, and ecosystem health and biodiversity. However, due to the lack of data on commercial-scale algal biofuel production processes, it is uncertain what these impacts will be.

In some cases, the algal production process could prove environmentally beneficial. For example, use of wastewater effluent, particularly partially treated wastewater, to cultivate algae provides benefits of removing nutrients from the wastewater and reduces the environmental impact of the production process. Combining commercial-scale algae production facilities with wastewater treatment plants may therefore create synergies that increase algae yields while decreasing environmental impacts of both facilities.

Key points to understanding the potential environmental impacts of using algae for biofuel production are described below.

Water Quality: The water quality impacts of algal biofuel production will depend on both the source of water used for cultivation and the quality of the water released from the production facility. Algae production facilities co-located with water treatment plants, fossil fuel power plants, or other industrial sources of pollution could have a positive impact on water quality. However, release of wastewater from an algal biofuel production process—especially salinated wastewater released into a freshwater environment—could adversely affect water quality.

Water Quantity: Water is an important input in the algae cultivation process; increased production of algal biofuels will impact water availability, especially in areas where water is already scarce. However, algae may be less water intensive than other feedstocks. Moreover, because algae can thrive in brackish and untreated waters, they are an ideal feedstock for water-stressed locations.

Soil Quality: Soil quality impacts are likely to be minimal, based on existing studies.

Air Quality: Little is known about how algal biofuel production will affect air quality. However, preliminary data suggest that open system cultivation systems have a greater potential than photobioreactor systems to adversely affect air quality.

Ecosystem Health/Biodiversity: Little is known about how increases in algal biofuel production might affect biodiversity. Because algae demands substantial amounts of water, its cultivation could adversely impact native species in water-stressed locations. Compared with other feedstocks, however, algae is not water intensive. Also unknown are what impacts an accidental release of algae might have on native aquatic ecosystems. Algae require low inputs of fertilizers and pesticides compared to other feedstocks, which may also translate into fewer ecological impacts to aquatic ecosystems

Invasive Species: The ability of cultivated algae to escape into and survive in the natural environment is uncertain. Experts speculate that photobioreactor cultivation systems would be superior to open systems in preventing the escape of cultivated algae.

3.5.7.2 Key Uncertainties and Unknowns

- Most of the uncertainties related to the production of algae for biodiesel stem from a lack of knowledge about which technologies may be used in future commercial applications, where they will be located, and what species and strains of algae will be used.

- Water availability impacts from feedstock growth will depend on where the algae are grown, if open or closed systems are used, and whether water is recycled.

3.6 Waste-Based Feedstocks

3.6.1 Introduction

Diverse wastes, including construction debris, municipal solid waste (MSW), yard waste, food waste, and animal waste, have the potential to serve as biofuel feedstocks. Depending on the waste, conversion system, and product, potential exists for municipalities, industries, and farmers to transform a material with high management costs to a resource that generates energy and profits. In some instances, diverting waste that cannot be recycled or reused to fuel has been found to reduce land-filled materials by 90 percent, helping to also extend the lifetime capacity of the landfill (Helou et al., 2010).

Tapping into waste energy sources has many challenges, including dispersed locations and potentially high transport costs; lack of long-term performance data; the cost of converting waste to energy, and the possibility that the resulting biofuel might not meet quality or regulatory specifications for use (Bracmort and Gorte, 2010).

Use of wastes as biofuel feedstocks will vary based on their availability, the ability of conversion technologies to handle the material, and the comparative economics of their use for fuel versus power and other products. Types and quantities of wastes used will vary by region. A large number and variety of waste-based materials are being investigated and implemented as feedstocks for ethanol and biodiesel, mostly on local scales. For example, several states—e.g., Massachusetts (Advanced Biofuels Task Force, 2008; Timmons et al., 2008), California (Chester et al., 2007), and Ohio[20]—have explored waste availability and its potential to meet regional energy needs, either for power or for transportation fuel. Feedstocks may be converted to biofuel or used as an energy source to power a biorefinery.

3.6.2 Municipal Solid Waste

The biogenic portion of municipal solid waste (paper, wood, yard trimmings, textiles, and other materials that are not plastic- or rubber-based), has the potential to be a significant feedstock for ethanol and other biofuels. Using 2005 data, the U.S. Energy Information Administration calculated that 94 million tons (MT) (about 56 percent) of the 167.8 MT of MSW waste generated that year had biogenic BTU content (EIA, 2007). This estimate included food waste (the third largest component by weight), which is also potentially viable as a biofuel feedstock in addition to biogenic material listed above. Some producers claim to have thermochemically-based technology that can yield 120 gallons of ethanol per ton of MSW, and therefore, this could serve as a rough MSW yield estimate. (Fulcrum Bioenergy, 2009). Accepting this estimate, the 94 MT of biogenic MSW generated in the U.S. in 2005 would have the potential to generate up to 11 billion gallons of ethanol. While this is not a likely scenario (since, for example, some of the biogenic fraction—paper, wood, etc.—would be recycled or reused), it demonstrates that MSW could be a significant source for biofuel. In addition, there are significant environmental co-benefits associated with using MSW for biofuel, including

[20] Specifically, a partnership between the Solid Waste Authority of Central Ohio and Quasar Energy Group to produce ethanol from municipal solid waste (see http://www.quasarenergygroup.com/pages/home.html), as well as the "Deploying Renewable Energy—Transforming Waste to Value" grant program (see: http://www.biomassintel.com/ohio-10-million-available-waste-to-energy-grant-program/).

diverting solid waste from landfills and incinerators, extending their useful life, and reserving that capacity for materials that cannot be recycled or reused.

3.6.3 Other Wastes

Several types of waste materials that currently present environmental and economic challenges have the potential to be harnessed as feedstock for biofuel. These materials include waste oil and grease, food processing wastes, and livestock waste (Antizar-Ladislao and Turrion-Gomez, 2008; Helou et al., 2010).

The U.S. Department of Energy estimates that the restaurant industry generates 9 pounds of waste oil per person annually, and that the nation's wastewater contains roughly 13 pounds of grease per person per year (Wiltsee, 1998). Several municipalities and industries have implemented collection programs, and are converting these wastes to biodiesel.

Annually, the U.S. generates an estimated 48 million tons of food processing wastes (i.e., food residues produced during agricultural and industrial operations), not including food waste disposed and processed through wastewater treatment plants (Kantor et al., 1997). These wastes have potential a biofuel feedstocks.

The U.S. generates over 1 billion tons of manure, biosolids, and industrial by-products each year (ARS, n.d.). The amount of manure generated at confined and other types of animal feeding operations in the U.S. is estimated to exceed 335 million tons of dry matter per year (ARS, n.d.). While much of this manure is applied to cropland and pasture as fertilizer, excess is often available and could be tapped as a biofuel feedstock. It has been estimated that around 10 percent of current manure production could be used for bioenergy purposes under current land use patterns once sustainability concerns are met (i.e., this manure is available after primary use of manure on soils to maintain fertility) (Perlack et al., 2005). Methane emissions from livestock manure management systems, which account for a significant percentage (10 percent, or 17.0 million metric tonnes of carbon equivalent [MMTCE] [3.0 teragrams, or Tg] in 1997) of the total U.S. methane emissions, are another potential energy source (U.S. EPA, 1999).

Using any of these excess waste materials as biofuel feedstocks has potential to create a higher value use with significant environmental and economic benefits.

3.6.4 Environmental Impacts of Waste-Based Biofuel

Waste-based biofuels are expected to have both environmental benefits and impacts. On the positive side, for example, diverting waste from landfills avoids generation of landfill methane gases, and diverting waste and trap greases from wastewater treatment plants helps avoid costly plant upsets that contribute to combined sewer overflow. Biorefineries that use wastes, particularly MSW, tend to be located in proximity to the waste source, which correlates well with the densely populated end-users of transportation fuels and helps reduce the GHG lifecycle footprint of waste-derived fuels (Antizar-Ladislao and Turrion-Gomez, 2008; Williams et al., 2009).

More information is needed to understand and evaluate the environmental effects of waste-based biofuels. Different wastes have different characteristics, including size, volume, heterogeneity, moisture content, and energy value. These characteristics will, to a large degree, determine feasible and appropriate collection, processing, and conversion methods, which in turn will determine net energy gain, as well as environmental impacts such as air and GHG emissions. Research is needed to compare the benefits and impacts of various technological options for converting MSW to biofuel, and to compare, on a regional basis, the environmental benefits and impacts of MSW to other biofuel feedstocks. Currently, data are lacking for such comparisons. Comparative life cycle assessments that consider both the direct impacts or benefits and indirect impacts or benefits (e.g., impacts of reduced landfilling of MSW) are needed to understand the true value of waste as an alternative feedstock.

Assessment and Uncertainty

There have been comparatively few attempts at assessing the environmental impacts associated with the production and use of waste-based biofuels (Williams et al., 2009). In general, waste as a feedstock is expected to have a smaller environmental impact than conventional feedstocks. However, the choice of waste management options and the particular technology for energy recovery will influence the environmental medium impacted and the level of impact (Chester and Martin, 2009; Kalogo et al., 2007). As the number of waste conversion facilities increases, environmental monitoring and research will be needed to address the information gaps that currently limit environmental assessment.

3.7 Summary of Feedstock-Dependent Impacts on Specialized Habitats

EISA Section 204 requires an assessment of the impacts of biofuels on a variety of environmental and resource conservation issues, including impacts on forests, grasslands, and wetlands. This section provides an overview of impacts on these specific habitats.

3.7.1 Forests

Woody biomass is the feedstock most likely to affect forests; row crops, algae, and most perennial grasses are unlikely to have an impact on this habitat.

Section 211(o) of the Clean Air Act limits planting of short-rotation woody crops and harvesting of tree residue to actively managed tree plantations on non-federal land that was cleared prior to December 19, 2007, or to non-federal forestlands; and limits removal of slash and pre-commercial thinning to non-federal forestlands. However, as described in Table 3-5, a variety of activities associated with producing woody biomass feedstock may impact forests.

Table 3-5: Overview of Impacts on Forests from Different Types of Biofuel Feedstocks

Feedstock	Forest Impact	Report Section
Row Crops	Unlikely to have impacts.	
Perennial Grasses	Most grass species are unlikely to have impacts.	3.3.6.2
Woody Biomass	SRWC plantations may deplete soil nutrients with repeated, frequent harvesting, particularly on marginal soils, but may sustain levels with coppicing, longer-rotations, and strategic use of cover crops.	3.4.4.3
	SRWC plantations can sustain high species diversity, although bird and mammal species tend to be habitat generalists.	3.4.5.1
	Some tree species under consideration, like *Eucalyptus*, may invade forests in certain locations.	3.4.5.2
	Harvesting forest residues may decrease nutrient availability, soil organic matter, and woody debris available for species habitat.	3.4.4.2; 3.4.4.3; 3.4.6.1
Algae	Unlikely to have impacts.	

3.7.2 Grasslands

Production of row crops and perennial grasses for biofuel feedstocks can impact grasslands, although perennial grasses may also have some positive effects on grasslands.

In addition to the restrictions on forested sources of renewable biomass mentioned above, Section 211(o) of the Clean Air Act more broadly limits the lands on which any biofuel feedstock can be produced to those that were cleared or cultivated at any time prior to December 19, 2007, either in active management or fallow and non-forested. Therefore, grassland that remained uncultivated as of December 19, 2007, may not be converted to grow biofuel. Most of lands that would be eligible for renewable biomass production under the Clean Air Act, because they were cultivated at some point prior to December 19, 2007, are now part of the Conservation Reserve Program (see Section 3.2.1.1). The USDA estimates that the vast majority (78 percent) of lands that will be taken out of the CRP to grown biofuel feedstock will be grasslands (FSA, 2008). Therefore, conversion of CRP lands to grow biofuels will impact grassland ecosystems, as will other aspects of biofuel feedstock production (Table 3-6).

Table 3-6: Overview of Impacts on Grasslands from Different Types of Biofuel Feedstocks

Feedstock	Grasslands Impact	Report Section
Row Crops	Conversion of grasslands to row crops impacts grassland-obligate species, potentially leading to declines, including declines in duck species.	3.2.6.1
	Higher proportions of corn within grassland ecosystems leads to fewer grassland bird species.	3.2.6.1
Perennial Grasses	Conversion of row crops to switchgrass may improve grassland habitat for some species depending on management regimes.	3.3.6.1
	Conversion of CRP lands to perennial grasses or harvesting of existing grasslands is less likely to have negative impacts on grassland species, particularly if harvesting occurs after the breeding season.	
	Use of native mixtures of perennial grasses can restore some native biodiversity.	3.3.6.1
	Cultivation of switchgrass outside its native range may lead to invasions of native grasslands.	3.3.6.2
	Cultivation of *Miscanthus* may lead to invasions of pasture and other grasslands.	3.3.6.2
Woody Biomass	Unlikely to have impacts.	
Algae	Depends on siting.	

3.7.3 Impacts on Wetlands

Provisions in both the Food Security Act of 1985 (commonly known as the Swampbuster Program) and Clean Water Act (Section 404 Regulatory Program) offer disincentives that limit the conversion and use of wetlands for agricultural production. Nevertheless, impacts to wetlands are still expected from the feedstocks assessed in this report (Table 3-7).

Table 3-7: Overview of Impacts on Wetlands from Different Types of Biofuel Feedstocks

Feedstock	Wetlands Impact	Report Section
Row Crops	Increased sediment, nutrients, chemicals, and pathogens from runoff flow into downstream wetlands.	3.2.2
	Increased nutrient loadings, leading to changes in wetlands community structure.	3.2.2.1
Perennial Grasses	Reduced sediment and nutrient loadings, leading to improved water quality (but dependant on specific management practice).	3.3.2.1
	Some grass species under consideration may invade wetlands, including giant reed (*Arundo donax*) and reed canary grass (*Phalaris arundinacea*).	3.3.6.2
Woody Biomass	Harvesting forest residues and thinning may increase nutrient loads, depending on slopes, soils, any buffer zones, and use of best management practices to reduce runoff.	3.4.2.1
Algae	Algal strains created may escape from cultivation and invade wetlands. As noted in Section 3.5.2, use of algae for biofuel may also have positive impacts: Co-locating algae production facilities with wastewater treatment plants, fossil fuel power plants, or other industrial pollution sources can improve water quality and utilize waste heat that would otherwise contribute to thermal pollution, while reducing freshwater demands and fertilizer inputs.	3.5.6.2

3.8 Genetically Engineered Feedstocks

Genetic engineering of crops has a history of research, development, and commercialization that extends back for more than 15 years. Along with the growth of this biotechnology industry, the U.S. established a coordinated framework for regulatory oversight in 1986 (OSTP, 1986). Since then, the relevant agencies (EPA, USDA, and Food and Drug Administration) have implemented risk assessment programs that allow informed environmental decision-making prior to commercialization. These programs have been independently assessed over the years (NRC, 2000, 2001, 2002) and improvements made to ensure the safety of the products. At the same time, the methodology for biotechnology risk assessment has been scrutinized and general frameworks created to facilitate robust approaches and harmonize the processes internationally (Craig et al., 2008; Auer, 2008; Nickson, 2008; Romeis et al., 2008; Raybould, 2007; Andow and Zwalen, 2006; Conner et al., 2003; Pollard et al., 2004). This section describes environmental concerns associated with Genetically Modified Organisms (GMOs) that are currently used as biofuel feedstocks, as well as anticipated concerns for GMOs that will be developed for the next generation of biofuel feedstocks.

As indicated earlier in this chapter, great advantage has already been taken for genetically engineered corn and soybean, which are now grown worldwide, along with other engineered crops. Brookes and Barfoot have conducted a series of extensive post-commercialization assessments of genetically engineered maize, soybeans, cotton, sugar beets, and canola varieties at 10-year intervals (Brookes and Barfoot, 2006, 2008, 2009, 2010). In these analyses, the authors found consistent reductions in the amounts of pesticides used and a reduction in GHG emissions for agricultural systems where these GMO crops are grown. These results are supported by others (Brinmer et al., 2005; Knox et al., 2006), although regional differences in the reductions have been noted (Kleter et al., 2008). The results for corn and soybean independently are consistent with the general trends (Brookes and Barfoot, 2010). Assuming that current genetically engineered varieties of corn and soybeans receive continued regulatory oversight, no additional environmental concerns are anticipated with these organisms in their current genetic configuration, even with an increase in their production. However, as feedstocks for biofuel change to accommodate cellulosic technologies and algae production, the range of environmental considerations, including impacts from GMO varieties, will change as well (Wilkinson and Tepfer, 2009; Lee et al., 2009).

To harness the full potential of biomass, the genetic engineering of feedstocks has been recognized as a key technology (Gressel 2008; Antizar-Ladislao and Turrion-Gomez, 2008; Sexton et al., 2009). The approaches being considered include increasing plant biomass by delaying flowering, altering plant growth regulators, and manipulating photosynthetic processes; modifying traits (e.g., herbicide tolerance, insect resistance) in non-row crop plants that reduce cultivation inputs; and modifying cellulose/lignin composition and other traits that result in cost reductions in bioprocessing (i.e., facilitating the biorefinery process) (Sticklen 2007, 2009; Ragauskas et al., 2006; Gressel 2008). These new varieties may have implications for the environment beyond what has been considered in first generation biotechnology crops, and the scientific community has begun to examine whether and how well existing risk assessment procedures will work for bioenergy crops (Wolt, 2009; Chapotin and Wolt, 2007; Wilkinson and Teper, 2009; Firbank 2007; Lee et al. 2009). For example, some first attempts have been made at evaluating feedstocks with particular traits; suggestions for minimizing environmental impacts are incorporated (Kausch et al., 2010a, 2010b).

4. Biofuel Production, Transport, Storage and End Use

4.1 Introduction

This chapter addresses potential environmental impacts of post-harvest activities of the biofuel supply chain (see Figure 4-1). These activities comprise feedstock logistics (discussed in Section 4.2) and biofuel production (Section 4.3), distribution (Section 4.4), and end use (Section 4.5).

Production of biofuel from feedstock takes place at biofuel production facilities through a variety of conversion processes. The resulting biofuel is transported to blending terminals and retail outlets by a variety of means, including rail, barge, tankers, and trucks. Biofuel distribution almost always includes periods of storage. Once dispensed at the final outlet, biofuel is combusted in vehicles and other types of engines, usually as a blend with gasoline or diesel, or in some cases in neat form.

Biofuel production, distribution, and end use result primarily in impacts to air and water. Air emissions may be released by a variety of sources (see Figure 4-1). Many factors affect the quantity and characteristics of these emissions, including the type and age of equipment used, and operating practices and conditions.

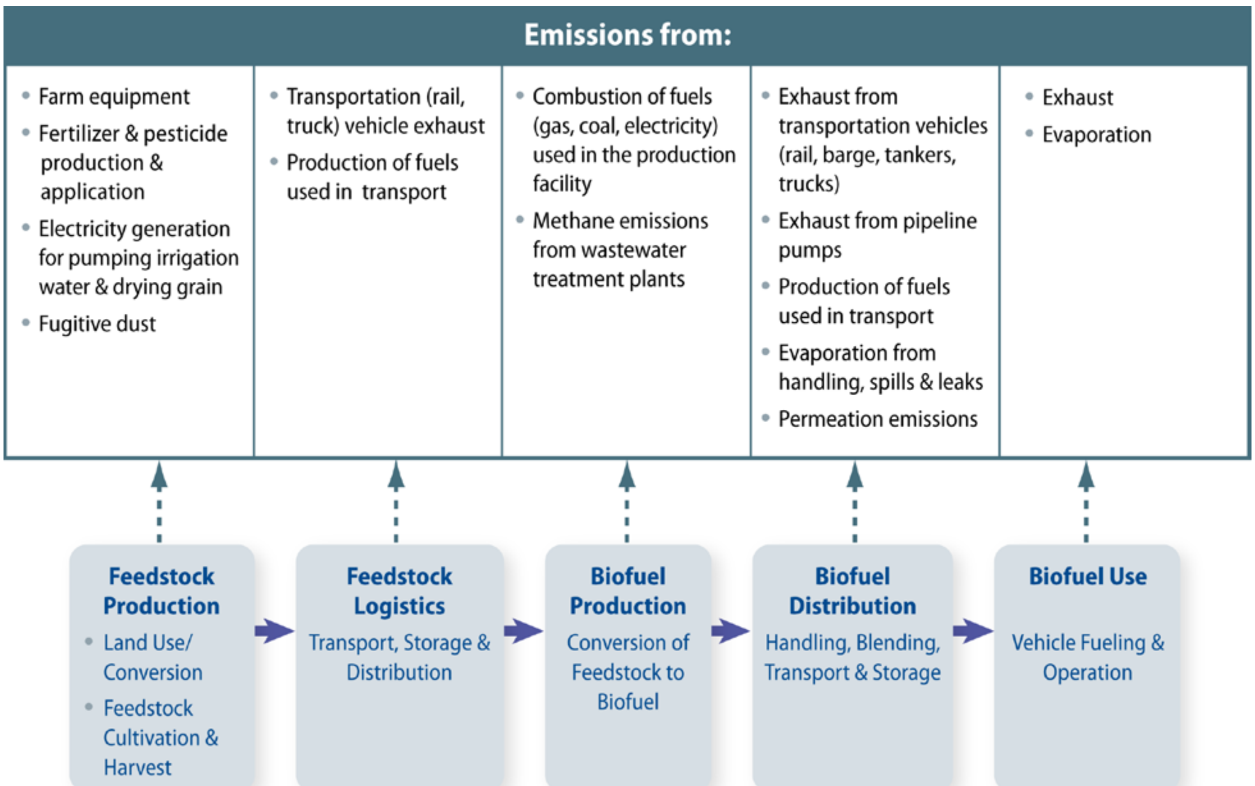


Figure 4-1: Sources of Criteria Air Pollutant and Toxics Emissions Associated with Production and Use of Biofuel

Air emissions associated with end use of ethanol combustion are relatively independent of feedstock or conversion process, whereas biodiesel emissions are highly dependent on feedstock type. As discussed later in the chapter, biofuel combustion may result in higher emissions of some pollutants compared to gasoline combustion, and lower emissions of others.

Biofuel production may require use of water, which may contribute to ground water depletion or lower surface water flow, depending on the amount of water withdrawn and water availability. Potential water quality impacts include wastewater discharge during the conversion process and the potential for leaks and spills to surface and groundwater during biofuel handling, transport, and storage. Additionally, phosphorus runoff from the manure of animals that have been fed an ethanol by-product, dried distillers grains with solubles, which has a high phosphorus content (Regassa et al., 2008), may have the potential to impact water quality.

Possible air and water impacts associated with ethanol and biodiesel, as well as opportunities for mitigation, are discussed in Sections 4.2 to 4.5. Discussion focuses primarily on the impacts of corn ethanol and diesel from soybean, since these constitute the vast majority of biofuel produced and used in the U.S. as of July 2010.

4.2 Feedstock Logistics

4.2.1 Handling, Storage, and Transport

Feedstock logistics comprise activities associated with handling, storing, and transporting feedstocks after harvest to the point where the feedstocks are converted to biofuel. The most significant environmental impacts of these activities are the emissions associated with energy use. Both greenhouse gases (GHG) and criteria pollutant emissions result from the combustion of fuels used during transportation. In general, feedstock logistics may be optimized, and emissions reduced, by integrating feedstocks, processing facilities, and consumer demands at a regional scale to minimize transport distances.

4.2.1.1 Ethanol

Harvested corn is transported to a biorefinery where it is converted to ethanol and a number of co-products. Air quality will be impacted by emissions from the combustion of fuels used for transportation vehicles and equipment.

4.2.1.2 Biodiesel

After harvest, soybeans are transported from fields to the drying site, storehouse, or collection center, followed by transport to the biodiesel refinery. Air quality may be affected by emissions from the combustion of fuels used for transportation vehicles and equipment.

4.3 Biofuel Production

4.3.1 Biofuel Conversion Processes

4.3.1.1 Ethanol

As of November 2009, there were 180 corn starch ethanol facilities in the U.S. with a combined capacity of 12 billion gallons per year (bgy) (U.S. EPA, 2010b, footnote 250).[21] At that time, 27 of these (representing 1,400 million gallons per year (mgy) of capacity) were idled, and another 10 facilities, with a combined capacity of 1,301 mgy, were under construction (U.S. EPA, 2010b, p.137). These facilities are located in the major corn-producing region of the country: Iowa (the largest production capacity and the greatest number of plants) followed by Nebraska, Minnesota, Indiana, and Illinois (U.S. EPA, 2010b).

Conventional ethanol is produced from the fermentation of corn starch. Two methods are currently utilized:

- **Dry milling,** in which the corn kernel is first ground into a meal, usually without separating out the various component parts of the grain. The meal is then slurried with water and cooked at high temperatures to form a mash, which then undergoes fermentation. Dry milling is more commonly used than wet milling.

- **Wet milling,** in which the kernels are steeped in water to separate out the germ, fiber, and gluten (fractionation). From this initial separation, co-products such as corn meal, corn gluten meal, and corn gluten feed are recovered. The remaining mash contains the water-soluble starch, which undergoes further processing for biofuel.

In both processes, soluble starch is subsequently converted to a simple sugar (glucose) through saccharification, an enzyme-catalyzed hydrolysis reaction. This is followed by yeast fermentation of the glucose to ethanol. Following fermentation, the mash is distilled to collect the ethanol as a mixture of 95 percent alcohol and 5 percent water. A subsequent dehydration step is required to remove the aqueous portion to yield 99.5 percent pure ethanol.

Substantial efforts are under way to develop processes to convert feedstocks containing cellulose into biofuels. These cellulosic feedstocks are primarily composed of cellulose, hemicellulose, and lignin polymers. Currently, two major pathways exist for converting cellulosic feedstocks into biofuel:

- **Biochemical conversion** using a physical and chemical process to liberate tightly bound cellulose and hemicellulose from lignin. The process uses strong acid or enzymes (cellulases) to hydrolyze the cellulose and hemicelluloses to glucose and other simple sugars, followed by microbial fermentation of the sugars into ethanol.

[21] Sources include the Renewable Fuels Association's Ethanol Biorefinery Locations (updated October 22, 2009) and *Ethanol Producer Magazine*'s producing plant list (last modified on October 22, 2009), in addition to information gathered from producer websites and follow-up correspondence.

- **Thermochemical conversion** involving gasification or pyrolysis.

  — In the *gasification process*, biomass is heated at high temperatures with a controlled amount of oxygen to decompose the cellulosic material. This yields a mixture comprised mainly of carbon monoxide and hydrogen known as syngas.

  — In *pyrolysis*, the biomass is heated in the absence of oxygen at lower temperatures than used in gasification. The product is a liquid bio-oil that can be used subsequently as a feedstock for a petroleum refinery.

Other cellulosic conversion processes are in various stages of development, from concept stage to pilot-scale development, and to demonstration plant construction. Although no U.S. commercial-scale plants are operating as of July 2010, several companies are expected to have facilities operating within the next few years.

4.3.1.2 Biodiesel

As of November 2009, there were approximately 191 biodiesel facilities in the U.S., with a combined capacity of 2.8 billion gallons per year (bgy) (U.S. EPA, 2010b). Total domestic production of biodiesel in 2009 was 505 mgy—much less than domestic production capacity. The dominant technology used to produce biodiesel involves a transesterification reaction in which triglycerides (fats) from the oil are converted to esters in the presence of an alcohol and a catalyst, such as potassium hydroxide. Plant oils (soy, rape, palm, algae, etc.) and other feedstocks (e.g., animal-derived oil such as lard and tallow, recycled oil and grease from restaurants and food processing plants) provide sources of triglycerides for conversion to biodiesel. Free glycerol, or glycerin, is a major co-product in transesterification, comprising an estimated 10 percent of the final product (U.S. EPA, 2010b). Table 4-1, below shows the breakdown of feedstocks used to produce biodiesel in the U.S. in 2009. Soybean oil made up the majority of biodiesel feedstock, comprising 54 percent.

Table 4-1. 2009 Summary of Inputs to U.S. Biodiesel Production

Input			2009 Total (lbs.)	Percentage of Total
Feedstock Inputs	Vegetable Oils	Corn Oil	84	2.3%
		Soybean Oil	1,974	54.0%
		Other Vegetable Oil	7	0.2%
	Animal Fats	Poultry Fat	127	3.5%
		Tallow	524	14.3%
	Animal Fats	White Grease	307	8.4%
		Other Animal Fats	82	2.2%
	Recycled Feedstock	Yellow Grease	156	4.3%
		Other Recycled Feedstock	13	0.4%

Table 4-1. 2009 Summary of Inputs to U.S. Biodiesel Production

Input		2009 Total (lbs.)	Percentage of Total
Other Inputs	Alcohol	328	9.0%
	Catalysts	56	1.5%

Source: EIA, 2010

Commercial processes for large-scale algae production and algal oil collection are currently being developed as another plant oil source for biodiesel (U.S. EPA, 2010b). Lipid extraction and drying currently are energy-intensive steps in the algae diesel production process. Other processing techniques are currently being investigated, including enzymatic conversion and catalytic cracking of algal oil, pyrolysis, and gasification of algae. However, lipid extraction via solvents followed by transesterification remains the most commonly used method for algal oil processing (U.S. DOE, 2010). Until commercial facilities using mature technologies go into production, the impacts from algal conversion will be uncertain.

In addition to transesterification, other methods for converting seed oils, algal oils, and animal fats into biofuel have been developed recently using technologies that are already widely employed in petroleum refineries (Huo et al., 2009). Hydrotreating technologies utilize seed oils or animal fats to produce an isoparaffin-rich diesel substitute referred to as “green diesel” or renewable diesel (Huo et al., 2008).

Although the transesterification process can generate a much larger amount of diesel product than the other processes, as noted above, it requires more energy and chemical inputs (Huo et al., 2008). In the case of biodiesel hydro-generation technology, inputs such as hydrogen, which are very energy-intensive to produce, must be taken into consideration in a full life cycle assessment in order to adequately evaluate energy efficiency of each fuel production process. Compared with conventional diesel and biodiesel, renewable diesel fuels have much higher cetane numbers (a measure of diesel fuel quality).

4.3.2 Air Quality

Air quality impacts associated with the production of biofuels occur throughout the production chain (Figure 4-1). EPA’s Regulatory Impact Analysis (RIA) of the revisions to the national Renewable Fuel Standard Program (RFS2) assessed the air pollutant emissions and air quality impacts of the biofuel volumes (see Table 2-1 in Chapter 2) required under the 2007 Energy Independence and Security Act (EISA). The discussion below summarizes the RIA results; the interested reader is referred to the RIA for additional details (U.S. EPA, 2010b).

The RIA analysis[22,23] focused on the projected impact of the renewable fuel volumes required in 2022, and accounted for GHG life cycle emissions, as well as the displaced petroleum consumption associated with use of biofuels. The emission impacts of the 2022 RFS2

[22] The RIA’s assessment of air quality relied on data from the EPA’s 2005 National Emissions Inventory, supplemented with the most up-to-date information where possible.

[23] For assessment of electrical energy demand, the RIA used a national energy source profile (i.e., coal, hydro, wind, natural gas) rather than regional source profiles, which can vary across the country.

volumes were quantified relative to two reference cases: 1) the original RFS program (RFS1) mandate volume of 7.5 billion gallons of renewable fuel (6.7 billion gallons ethanol), and 2) the U.S. Department of Energy (DOE) Annual Energy Outlook (AEO) 2007 projected 2022 volume of 13.6 billion gallons of renewable fuels. The RIA analysis found decreases in overall emissions for carbon monoxide (CO) and benzene, and increases in overall emissions for nitrogen oxide (NO_x), volatile organic compounds (VOCs), PM, and several air toxics, especially ethanol and acetaldehyde. Overall emissions of sulfur dioxide (SO_2) exhibited mixed results, depending on the fuel effect specified.

For biofuel production and distribution, the net change in VOC, CO, NO_x, and PM emissions can be attributed to two effects: 1) emission increases connected with biofuel production, and 2) emission decreases associated with reductions in gasoline production and distribution as ethanol displaces gasoline. Increases in fine particles less than 2.5 micrometers in diameter ($PM_{2.5}$), sulfur oxide (SO_x), and especially NO_x were determined to be driven by stationary combustion emissions from the substantial increase in corn and cellulosic ethanol production. Substantial fugitive dust and particulate increases are also associated with agricultural operations.

The RIA found that increasing the production and distribution of ethanol would lead to higher ethanol vapor emissions. To a lesser degree, the production and distribution of greater amounts of ethanol would lead to increases in emissions of formaldehyde and acrolein, as well as very small decreases in benzene, 1,3-butadiene, and naphthalene emissions relative to the total volume of these emissions in the U.S. Emissions of ammonia are expected to increase substantially due to increased ammonia from fertilizer use. Additional details on EPA's analysis of changes in emissions associated with the RFS2 volumes can be found in www.epa.gov/otaq/fuels/renewablefuels/regulations.htm.

Air pollutant emissions associated with the conversion of biomass to fuel may be mitigated through the use of cleaner fuels during the conversion process and more efficient process and energy generation equipment. The majority of ethanol plants built in recent years, and expected to be built in the near future, utilize dry mill technology (Wang et al., 2007a). Because most ethanol plants utilize similar dry milling production processes, differences in environmental impacts among plants are primarily due to each plant's choice of fuel. EPA's RIA assumes a dry mill for the base scenario.

EPA's RIA examines the impacts of using energy-saving technologies such as combined heat and power (CHP). CHP is an effective means to reduce air emissions associated with biofuel production (both ethanol and biodiesel). CHP generates electricity by burning natural gas or biogas, and then employs a heat recovery unit to capture heat from the exhaust stream as thermal energy. Using energy from the same fuel source significantly reduces the total fuel used by facilities along with the corresponding emissions of carbon dioxide (CO_2) and other pollutants. Fractionation, membrane separation, and raw starch hydrolysis are additional technologies examined in EPA's RIA that increase process efficiencies by enabling producers to sell distillers grains (a co-product of the corn-ethanol conversion process) wet rather than dry, thereby reducing greenhouse gas emissions and other possible environmental impacts (since drying distillers grains is an energy-intensive process).

4.3.2.1 Ethanol

Ethanol production requires electricity and the use of steam. Electricity is either purchased from the grid or produced onsite, and steam is typically produced onsite from natural gas. Power and the energy used to fuel boilers are responsible for emissions of VOCs, PM, CO, SO_x and NO_x (U.S. EPA, 2010b; Wang et al., 2007b). For corn-based ethanol, fossil fuels such as natural gas are typically used to produce heat during the conversion process, although a number of corn ethanol facilities are exploring new technologies with the potential to reduce their energy requirements.

A number of processes at ethanol production facilities result in emissions of air toxics. These processes include fermentation, distillation of the resultant mash, and drying of spent wet grain to produce animal feed. Emissions of air toxics vary tremendously from facility to facility due to a variety of factors, and it is difficult to determine how differences in the production processes individually impact emissions (U.S. EPA, 2010b). Ethanol vapor and air toxic emissions associated with biofuel production were projected to increase in EPA's RFS2 RIA, but these would be very small compared to current emissions (U.S. EPA, 2010b).

4.3.2.2 Biodiesel

While the production process for biodiesel is fundamentally different from ethanol, thermal and electrical energy are still required for production. The thermal energy required for biodiesel production is usually met using steam generated using a natural gas boiler. In certain situations, the glycerol co-product may also be burned to produce process heat, or a biomass boiler may be used to replace natural gas.

Air quality issues associated with a natural gas-fired biodiesel production process are similar to those for other natural gas applications such as ethanol production, and include emissions of VOCs, PM, CO, SO_x, and NO_x. Glycerol or solid fuel biomass boilers have emissions characteristics similar to those anticipated for cellulosic ethanol plants, including increased particulates and the potential for VOCs, NO_x, and SO_x.

Biodiesel production using a closed hot oil heater system would have none of the air emissions associated with traditional steam production. Air emissions associated with these systems would be those associated with the production of the electricity, which would take place outside the biodiesel plant boundary.

Additionally, the extraction of vegetable oil to create biodiesel in large chemical processing plants is typically achieved using hexane, a VOC that EPA has classified as a hazardous air pollutant. Hexane is also commonly used to extract algal oils. Fugitive emissions of hexane may result from increased biodiesel manufacture (Hess et al., 2009).

4.3.2.3 Greenhouse Gases

Fuel combustion at ethanol and biodiesel facilities releases greenhouse gases. Fermentation to ethanol also releases CO_2. Combustion-related greenhouse gas emissions are released to the atmosphere, but some ethanol facilities capture, purify, and sell their CO_2 associated with fermentation to the beverage industry for carbonation or to food processors for flash freezing. Opportunities for CO_2 reuse depend on the proximity of ethanol refineries to local users. The industrial gas supplier Airgas estimates that the ethanol industry captures and recovers 5 to 7 percent of all CO_2 that it produces from the ethanol fermentation process (spreadsheet provided to EPA by Bruce Woerner at Airgas on August 14, 2009, cited in U.S. EPA, 2010b, p. 133). Other sources of GHG emissions, such as those from electricity generation produced from coal or other GHG-emitting sources, are not currently recovered by the industry.

EISA Section 204 does not include GHG emissions in the set of environmental issues to be examined in this report. However, EPA did analyze life cycle GHG emissions from increased renewable fuels use as part of the EISA-mandated revisions to the RFS program. The Act established specific life cycle GHG emission thresholds for each of four types of renewable fuels, requiring a percentage improvement compared to life cycle GHG emissions for gasoline or diesel (whichever is being replaced by the renewable fuel) sold or distributed as transportation fuel in 2005. These life cycle performance improvement thresholds are listed in Table 4-2.

Table 4-2. Lifecycle GHG Thresholds Specified in EISA (percent reduction from 2005 baseline)

Renewable fuel [a]	20%
Advanced biofuel	50%
Biomass-based diesel	50%
Cellulosic biofuel	60%

a – The 20% criterion generally applies to renewable fuel from new facilities that commenced construction after December 19, 2007.

EPA's methodology for conducting the GHG life cycle assessment included use of agricultural sector economic models to determine domestic agriculture sector-wide impacts and international changes in crop production and total crop. Based on these modeling results, EPA estimated GHG emissions using DOE's GREET model defaults and Intergovernmental Panel on Climate Change (IPCC) emission factors. The GHGs considered in the analysis were CO_2, methane (CH_4), and nitrous oxide (N_2O). Biofuel process energy use and associated GHG emissions were based on process models for the different pathways considered. For ethanol and biodiesel, EPA's RFS2 RIA projected that (U.S. EPA, 2010b):

- Ethanol produced from corn starch at a new (or expanded capacity from an existing) natural gas-fired facility using advanced efficient technologies will comply with the 20 percent GHG emission reduction threshold.

- Ethanol produced from sugarcane will comply with the 50 percent GHG reduction threshold for the advanced fuel category.

- Biodiesel from soybean oil and renewable diesel from waste oils, fats, and greases will comply with the 50 percent GHG threshold for the biomass-based diesel category.

- Diesel produced from algal oils will comply with the 50 percent GHG threshold for the biomass-based diesel category.

- Cellulosic ethanol and cellulosic diesel (based on the modeled pathways) will comply with the 60 percent GHG reduction threshold applicable to cellulosic biofuels.

Based on the assessment described above, EPA projected a reduction of 138 million metric tons of CO_2-equivalent emissions by 2022 compared to projected 2022 emissions without the EISA-mandated changes (see the RFS2 RIA [U.S. EPA, 2010b] for details).

4.3.3 Water Quality and Availability

All biofuel facilities utilize process water to convert biomass to fuel. Water used in the biorefining process is modest in absolute terms compared to the water applied and consumed in growing the plants used to produce biofuel. The impacts associated with water use at conversion facilities depend on the location of the facility in relation to water resources. In some regions where water is abundant, increased withdrawals may have little effect. Ground water depletion may result in increased costs to pump water from deeper wells, loss of stream flow, and subsidence of the overlying land (Reilly et al., 2008). Several areas of the country that are already experiencing lowered ground water levels (e.g., High Plains aquifer, Lower Mississippi River alluvial aquifer) correspond with regions where increased biofuel production is expected. In addition, minimum in-stream flow for aquatic life can be affected by ground water depletion because ground water discharge into streams is a major source of stream base flow. In some areas, streams have run dry due to ground water depletion, while in other areas, minimum stream flow during the summer has been sustained because of irrigation return flow to streams (Bartolino and Cunningham, 2003). In the case of sole source aquifers, ground water depletion may severely impact drinking water availability, since these areas have no readily available alternative freshwater sources (Levin et al., 2002).

Comprehensive local, state, and regional water planning, as well as state regulatory controls, are critical to ensure that facilities are located in watersheds that can sustain the increased withdrawal without affecting other uses. The first step in mitigating water availability concerns associated with increased biofuel production is to locate production facilities where water sources are adequate to meet production needs without impacting other uses. Siting of biofuel facilities may also be influenced by state laws and regulations designed to avoid or mitigate conflicts among water uses. These vary by state. For example, withdrawals associated with biofuel production facilities may need a state permit to ensure that the proposed withdrawal does not result in unacceptable impacts on other users or on aquatic life. In addition, different states assign water rights in different ways. Some exercise the prior appropriation rule (i.e., water rights are determined by priority of beneficial use, meaning that the first person to use the water can acquire individual rights to the water); some are based on the English law of absolute ownership (i.e., rights to use water are connected to land ownership); some limit withdrawals based on stream flow requirements for aquatic life; and some have a hierarchy to prioritize uses of the water.

Like water quantity impacts, water quality impacts depend on a number of factors including facility location, water source, receiving water, type of feedstock used, biorefinery technology, effluent controls, and water re-use/recycling practices. Water quality impacts are associated with the wastewater discharge from the conversion process. Biological oxygen demand (BOD), brine, ammonia-nitrogen, and phosphorus are primary pollutants of concern from ethanol facilities, while discharges of BOD, glycerin, and to a certain extent, total suspended solids (TSS) pose the major water quality concerns from biodiesel facility effluent. Regulatory controls placed on the quality of biofuel production wastewater discharge can mitigate water quality impacts. Discharges to publicly owned wastewater treatment works (POTWs) are subject to general pre-treatment standards in the Clean Water Act (40 CFR 403.5). Biofuel facilities that discharge their wastewater to POTWs are subject to whatever pre-treatment limitations are in force for the receiving POTW. For those facilities that treat and discharge their own wastewater, EPA has enforceable regulations to control production facility effluent discharges of BOD, sediment, and ammonia-nitrogen through the National Pollutant Discharge Elimination System (NPDES) permit program.

Whether effluent is discharged to a POTW or treated onsite at the production facility, BOD can lead to methane emissions during the wastewater treatment process. To mitigate the release of methane to atmosphere, facilities can install anaerobic digesters as a treatment step. Anaerobic digesters treat the biosolids contained in wastewater effluent, generating biogas that is approximately 60 to 65 percent methane. This biogas can then either be flared or captured and used as a clean energy source at the biofuel production facility or elsewhere.

Currently there are no effluent limitation guidelines or categorical pretreatment standards that regulate process wastewater discharges from ethanol and biodiesel manufacturing facilities.

4.3.3.1 Ethanol

In 2007-2008, EPA evaluated biodiesel and corn ethanol manufacturing facilities. No major effluent quality issues were found from corn ethanol plants discharging to either surface waters or to wastewater treatment plants.

While some ethanol facilities get their process water from municipal water supplies, most use onsite wells (Wu et al., 2009). However, most untreated ground water sources are generally not suitable for process water because of their mineral content. Ground water high in mineral content is commonly treated by reverse osmosis, which requires energy and concentrates ground water minerals into reject water, with potential water quality impacts upon their release. For every two gallons of pure water produced, about a gallon of brine is discharged as reject water (U.S. EPA, 2010b). Methods to reduce the impact associated with reject water high in mineral concentration include (1) further concentration and disposal, or (2) use of in-stream dilution. Some ethanol facilities have constructed long pipelines to access additional water sources to dilute the effluent to levels that meet water quality standards.

Once process water is treated, most is lost as steam during the ethanol production process. Water use varies depending on the age of the facility and the type of milling process. Older generation production facilities use 4 to 6 gallons of process water to produce a gallon of ethanol; newer facilities generally use less than 3 gallons of water in the production process. Most of this water savings is gained through improved recycling of water and heat in the process. Dry milling facilities consume on average 3.45 gallons of fresh water per gallon of ethanol produced (Wu et al., 2009); newer facilities tend to consume about 27 percent less water. Wet mill facilities consume an average of 3.95 gallons of fresh water per gallon of ethanol produced

(Wu, 2008). Most estimates of water consumption in ethanol production are based on the use of clean process water and do not include the water discharged as reject water.

Ethanol plants are designed to recycle water within the plant, and improvements in water use efficiency of ethanol facilities are expected through steam condensate reuse and treated process water recycling (Wu et al., 2009). New technologies that improve water efficiency will help mitigate water quantity impacts.

Wastewater effluent from corn starch ethanol facilities is high in BOD (Powers, 2007). For example, one report found that ethanol production from corn produces wastewater with BOD from 18,000 to 37,000 mg/L (Pimentel and Pimentel, 2008, p. 380). Ethanol wastewater effluent can also contain ammonia-nitrogen and phosphorus.

Because no large-scale cellulosic ethanol production facilities are currently operating, water demand for production of cellulosic ethanol is not certain. However, for most cellulosic feedstocks, including agricultural residues like corn stover and dedicated energy crops like switchgrass, water demand is estimated to be between 2 and 10 gallons of water per gallon of ethanol, depending on the conversion technology, with volumes greater than 5 gallons of water per gallon of ethanol cited more often (NRC, 2008; Williams et al., 2009; Wu et al., 2009). Some studies assume water demand for processing woody biomass will be similar to processing cellulosic material from agricultural residues or dedicated energy crops (up to 10 gallons of water per gallon of ethanol) (Evans and Cohen, 2009). Other studies state that new technologies like fast pyrolysis will require less than half that amount of water per gallon of ethanol (Wu et al., 2009). Consumptive use of water is declining as ethanol producers increasingly incorporate recycling and other methods of converting feedstocks to fuels that reduce water use (NRC, 2008).

Cellulosic ethanol facilities that employ biochemical conversion would be expected to have similar water requirements and brine discharges as the current operating corn ethanol facilities. The additional steps required to separate the lignin from the cellulose could produce wastewater streams high in BOD that would require treatment onsite or at wastewater treatment plants.

4.3.3.2 Distillers Grain with Solubles

One important co-product of ethanol production is dried distillers grain with solubles (DDGS). Due to the increase in ethanol production and the price of corn, DDGS has become an increasingly important feed component for confined livestock. About one-third of the corn processed into ethanol is converted into DDGS; therefore, approximately 45 million tons of DDGS will be produced in conjunction with the 15 billion gallons of corn ethanol produced by 2015.

Livestock producers may partially replace corn or other feeds with DDGS for both economic and production reasons. Different livestock species can tolerate varying amounts of DDGS in their diets. Although specific analysis of DDGS can vary among ethanol plants, DDGS are higher in crude protein (nitrogen) and three to four times higher in phosphorus compared to corn (Regassa et al., 2008).

The increase in nitrogen and phosphorus from DDGS in livestock feed has potential implications for water quality. When nitrogen and phosphorus are fed in excess of animals' needs, excess nutrients are excreted in urine and manure. Livestock manure may be applied to crops, especially corn, as a source of nutrients. When manure is applied at rates above the nutrient needs of the crop or when the crop cannot use the nutrients, the nitrogen and phosphorus can runoff to surface waters or leach into ground waters. Excess nutrients from manure nutrients have the same impact on water quality as excess nutrients from other sources.

Livestock producers may limit the potential pollution from manure applications to crops through a variety of techniques. USDA's Natural Resources Conservation Service (NRCS) has developed a standard for a comprehensive nutrient management plan to address the issue of proper use of livestock manure (NRCS, 2009).

4.3.3.3 Biodiesel

Biodiesel facilities use much less water than ethanol facilities to produce biofuel. The primary consumptive water use at biodiesel plants is associated with washing and evaporative processes. Water use is variable, but is usually less than one gallon of water for each gallon of biodiesel produced (U.S. EPA, 2010b); some facilities recycle washwater, which reduces overall water consumption (U.S. EPA, 2010b). However, water use has been reported as high as three gallons of water per gallon of biodiesel (Pate et al., 2007). Larger well-designed facilities use water more sparingly, while smaller producers tend to use more water per production volume (U.S. EPA, 2010b). New technologies that improve water efficiency will help mitigate water quantity impacts.

In addition to water use in the washing and evaporation processes, other sources of wastewater include steam condensate; process water softening and treatment to eliminate calcium and magnesium salts, iron, and copper; and wastewaters from the glycerin refining process (U.S. EPA, 2008c). In a joint DOE/USDA study, it was estimated that consumptive water use at a biodiesel refinery accounts for approximately one-third of the total water use, or about 0.32 gallon of water per gallon of biodiesel produced (Sheehan et al., 1998b). New technologies have reduced the amount of wastewater generated at facilities. Process wastewater disposal practices include direct discharges (to waters of the United States), indirect discharges (to wastewater treatment plants), septic tanks, land application, and recycling (U.S. EPA, 2008c).

Most biodiesel manufacturing processes result in the generation of process wastewaters with free fatty acids and glycerin (a major co-product of biodiesel production). Despite the existing commercial market for glycerin, the rapid development of the biodiesel industry has caused a glut of glycerin production, resulting in many facilities disposing glycerin. Glycerin disposal may be regulated under several EPA programs, depending on the practice. However, there have been incidences of glycerin dumping, including an incident in Missouri that resulted in a large fish kill (U.S. EPA, 2010b).

Other constituents in the biodiesel manufacturing process wastewater include organic residues such as esters, soaps, inorganic acids and salts, traces of methanol, and residuals from process water softening and treatment (U.S. EPA, 2008c). Solvents used to extract lipids from algae, including hexane, alcohols, and chloroform, could also impact water quality if discharged to surface or ground water. Typical wastewater from biodiesel facilities has high concentrations of conventional pollutants—BOD, TSS, oil, and grease—and also contains a variety of non-conventional pollutants (U.S. EPA, 2008c).

Some biodiesel facilities discharge their wastewater to municipal wastewater treatment systems for treatment and discharge. In some cases, wastewater with sufficiently high glycerin levels has disrupted municipal wastewater treatment plant function (U.S. EPA, 2010b). There have been several cases of municipal wastewater treatment plant upsets due to high BOD loadings from releases of glycerin (U.S. EPA, 2010b). To mitigate wastewater issues, some production systems reclaim glycerin from the wastewater. As another option, closed-loop systems in which water and solvents can be recycled and reused can reduce the quantity of water that must be pretreated before discharge.

4.3.4 Impacts from Solid Waste Generation

Biofuels may also lead to significant environmental impacts stemming from solid waste generated by various production processes. EPA defines "solid wastes" as any discarded material, such as spent materials, by-products, scrap metals, sludge, etc., except for domestic wastewater, non-point source industrial wastewater, and other excluded substances (U.S. EPA, 2010i). A type of solid waste that is of particular interest in the case of biofuel production is the diatomaceous earth that is used as a filter to remove impurities from methyl esters, such as biodiesel. Several reports have indicated that diatomaceous earth may be spontaneously combustible, and disposal sites consider it a potential hazardous waste (Missouri Department of Natural Resources, 2008; Nebraska Department of Environmental Quality, 2009). The high surface area of the diatomaceous earth and the oil sets up a rapid decomposition that creates heat. Further study is needed to investigate this potential hazard and look into mitigation strategies.

4.4 Biofuel Distribution

The vast majority of biofuel feedstocks and finished biofuel are currently transported by rail, barge, and tank truck. Ethanol and biodiesel are both generally blended at the end of the distribution chain, just before delivery to retail outlets. Storage of biofuels typically occurs in above-ground tanks at blending terminals, in underground storage tanks (USTs), and at retail outlets (as a petroleum-biofuel blend).

The primary impacts related to transport and storage of biofuels relate to air quality (i.e., emissions from transport vehicles and evaporative emissions) and water quality (i.e., leaks and spills).

4.4.1 Air Quality

4.4.1.1 Ethanol

Air pollution emissions associated with distributing fuel come from two sources: 1) evaporative, spillage, and permeation emissions from storage and transfer activities, and 2) emissions from vehicles and pipeline pumps used to transport the fuels (see Figure 4-1). Emissions of ethanol occur both during transport from production facilities to bulk terminals, and after blending at bulk terminals.

Although most ethanol facilities are concentrated in the midwestern United States, gasoline consumption is highest along the east and west coasts. Fleet transport of biofuel, often by barge, rail, and truck, increases emissions of air pollutants, such as CO_2, NOx, and PM due to the combustion of fuels by transport vehicles. EPA's RFS2 RIA found relatively small increases in criteria and air toxics emissions associated with transportation of biofuel feedstocks and fuels (U.S. EPA, 2010b). In addition, transport and handling of biofuel may result in small but significant evaporative emissions of VOCs (Hess et al., 2009). With the exception of benzene emissions, which were projected to decrease slightly, EPA's RFS2 RIA projected relatively small increases in emissions of air pollutants associated with evaporation (U.S. EPA, 2010b).

Pipeline transport decreases air emissions associated with fleet transport of biofuel because fuel is not combusted in the transport process. However, transport of biofuels by pipeline raises potential technical issues, including internal corrosion and stress corrosion cracking in pipeline walls, and the potential to degrade performance of seals, gaskets, and internal coatings. Additionally, ethanol's solvency and affinity for water can generate product contamination concerns (U.S. EPA, 2010b). Dedicated ethanol pipelines may alleviate these issues; however, they are costly to construct. Due to the incompatibility issues with the existing petroleum pipeline infrastructure, the growth in ethanol production is expected to increase emissions of criteria and toxic air pollutants from freight transport, while a corresponding decrease in gasoline distribution would decrease emissions related to pipeline pumping (Hess et al., 2009).

4.4.1.2 Biodiesel

Air pollution emissions from fuel combustion in transport vehicles related to biodiesel feedstocks and fuels are not materially different than those associated with ethanol. Currently, pipeline distribution of biodiesel is still in the experimental phase. Significant evaporative emissions are not expected from storage and transport of biodiesel fuel due to its low volatility (U.S. EPA, 2010b).

4.4.2 Water Quality

Leaks and spills from above-ground, underground, or transport tanks may occur during biofuel transport and storage, potentially contaminating ground water, surface water, or drinking water supplies.

For bulk transport, the major concern is based on an accident scenario in which the transport tank is damaged and a large amount of fuel is spilled. In addition, leaks might occur during transport because of certain fuel-related factors, such as the fuel's corrosivity. Ethanol is slightly acidic and can corrode some active metals; biodiesel is also slightly corrosive. The possibility of leaks during transport is minimized by the selection of appropriate materials and proper design in accordance with the applicable material standards.

Leaks from storage tanks are also a major concern. Most states report that underground storage tanks (USTs) are a major source of ground water contamination (U.S. EPA, 2000). Preliminary research has shown that any concentration of biofuel in an underground tank also containing petroleum-based fuels increases the potential for groundwater migration (EPA

OSWER Biofuels Compendium, http://www.epa.gov/oust/altfuels/bfcompend.htm). A leaking UST can also present other health and environmental risks, including the potential for fire and explosion.

EPA's Office of Underground Storage Tanks is working with other agencies to better understand material compatibility issues associated with older UST systems, in order to assess the ability of these systems to handle new fuel blends (U.S. EPA, 2009b). Because most of the current underground storage tank equipment, including 617,000 active USTs, was designed and tested for use with petroleum fuels, many UST systems currently in use may contain materials that are incompatible with ethanol blends greater than 10 percent (U.S. EPA, 2009c) or biodiesel. Although it is not possible to quantify the risk at this time, EPA is developing modeling software to assess fuels of varying composition on ground water (U.S. EPA, 2010b).

Biodiesel and ethanol blend fuels degrade many non-metallic materials, such as natural rubber, polyurethane, older adhesives, certain elastomers and polymers used in flex piping, bushings, gaskets, meters, filters, and materials made of cork. Biodiesel and ethanol blend fuels also degrade soft metals such as zinc, brass, and aluminum. If a fuel system does contain these materials and users wish to fuel with blends over B20 (i.e., with fuel containing more than 20 percent biodiesel), replacement with compatible elastomers is needed. In many instances, especially with older equipment, the exact composition of elastomers cannot be obtained and it is recommended they be replaced if using blends over B20.

Several measures are already in place to help prevent and mitigate potential water quality impacts. Under the Resource Conservation and Recovery Act (RCRA), owners and operators of regulated UST systems must comply with requirements for financial responsibility, corrosion protection, leak detection, and spill and overfill prevention. Federal regulations require that ethanol and biodiesel storage containers are compatible with the fuel stored. For USTs, leak detection equipment is required and must be functional. Through the Spill Prevention, Control, and Countermeasure (SPCC) rule, EPA has enforceable regulations to control water quality impacts from spills or leaks of biofuel products and by-products.

Further testing and certification of the acceptability of storage tanks and leak detection systems performance will be crucial to safe storage of biofuels. In addition, developing storage materials that are resistant to biofuel leaks and spills locating will help prevent spills, and locating USTs away from ground water supplies, and will mitigate water quality impacts in case a spill or leak does occur.

Additional details specific to ethanol and biodiesel are discussed below.

4.4.2.1 Ethanol

Ethanol is stored in neat form at the production facility, in denatured form at terminals and blenders, and as E85 (85 percent ethanol and 15 percent gasoline) and E10 (10 percent ethanol and 90 percent gasoline) mixtures at retail. Ethanol is water soluble and can be degraded by microorganisms commonly present in ground water (U.S. EPA, 2009d). In ground water, ethanol's high oxygen demand and biodegradability changes the attenuation of the constituents in gasoline/ethanol blends. This can cause reduced biodegradation of benzene, toluene, and xylene (up to 50 percent for toluene and 95 percent for benzene) (Mackay et al., 2006; U.S. EPA, 2009d). The presence of ethanol can restrict the rate and extent of biodegradation of benzene, which can cause the plumes of benzene to be longer than they would have been in the absence of ethanol (Corseuil et al., 1998; Powers et al., 2001; Ruiz-Aguilar et al., 2002). This could be a significant concern to communities that rely on ground water supplies with the potential to be impacted by leaks or spills (Powers et al., 2001; Ruiz-Aguilar et al., 2002). In surface waters, rapid biodegradation of ethanol can result in depletion of dissolved oxygen with potential mortality to aquatic life (U.S. EPA, 2010b).

There are other potential hazards in addition to those associated with chemical toxicity. Some spills of gasoline with ethanol may produce methane concentrations in the soil that pose a risk of explosion (Da Silva and Alvarez, 2002; Powers et al., 2001).

4.4.2.2 Biodiesel

In general, if biodiesel is blended with petroleum diesel, another petroleum product, or a hazardous substance, state UST regulations may apply to those blends. One-hundred percent biodiesel contains no petroleum-based products or hazardous substances. Therefore, UST regulations generally do not apply to 100 percent biodiesel.

Biodiesel is not water soluble. Biodiesel degrades approximately four times faster than petroleum diesel. In aquatic environments, biodiesel degrades fairly extensively (Kimble, n.d.). Results of aquatic toxicity testing of biodiesel indicate that it is less toxic than regular diesel (Kahn et al., 2007). Biodiesel does have a high oxygen demand in aquatic environments, and can cause fish kills as a result of oxygen depletion (Kimble, n.d.). Water quality impacts associated with spills at biodiesel facilities generally result from discharge of glycerin, rather than biodiesel itself (Kimble, n.d.).

4.5 Biofuel End Use

Most vehicles on the road today can operate on E10. Nearly half of U.S. gasoline is an E10 mixture to boost octane for more complete combustion or to meet air quality requirements (Alternative Fuels and Advanced Vehicles Data Center, 2010). E85 is another form in which ethanol is consumed, but it can only be used in flex-fuel vehicles, which can run on either conventional gasoline or ethanol blended into gasoline up to 85 percent. Under current market circumstances, greater deployment of flex-fuel vehicles may be needed to meet the EISA mandated volume standards. Because of biodiesel's chemical properties, it is not interchangeable with petroleum-based diesel fuel. For this reason, its blend level is specifically labeled when it is blended with petroleum diesel (U.S. EPA, 2010b). Biodiesel can be used in its pure form, known as "neat biodiesel" or B100, but most vehicle and engine manufacturers do not recommend its use in non-approved engines and vehicles. There are some concerns regarding maintenance issues related to engines operated on biodiesel, because the fuel has been shown to soften and degrade certain types of elastomers and natural rubber compounds over time. This will impair fuel system components such as fuel hoses and fuel pump seals. Such component degradation can lead to leaks, poor performance, and other problems that are likely to result in increased emissions and subsequent environmental impacts.

Biofuels for jet aircraft require additional refining or need to be blended with typical jet fuels to meet the standards of commercial aviation fuels. There are few long-term studies of biofuel performance on large diesel engines such as stationary power generators, ships, locomotives, and jet engines.

4.5.1 Air Quality

The primary impact associated with biofuel end use is air quality. Section 211 (v) of the Clean Air Act requires EPA to study the air quality impacts associated with the use of biofuel and biofuel blends. EPA has already adopted mobile source emission control programs that reduce air pollution emissions and improve air quality. If necessary, EPA will issue further regulations to mitigate adverse air quality impacts as a result of increases in biofuels.

4.5.1.1 Ethanol

The following discussion is based on E10, because considerably more information is available about its use. A wide variation in evaporative and tailpipe emissions have been reported due to a range of factors, such as the age of the vehicle, the power output and operating condition of the engine, the fuel characteristics, how the vehicle is operated, and ambient temperatures (Ginnebaugh et al., 2010; Graham et al., 2008; Yanowitz and McCormick, 2009).

As stated in section 4.3.2, the emission impacts of the 2022 RFS2 volumes in the RFS2 RIA were quantified relative to two reference cases: 1) the original RFS program (RFS1) mandate volume of 7.5 billion gallons of renewable fuel (6.7 billion gallons ethanol), and 2) the U.S. Department of Energy (DOE) Annual Energy Outlook (AEO) 2007 projected 2022 volume of 13.6 billion gallons of renewable fuels. In the RFS2 RIA, EPA projected decreases in emissions of carbon monoxide, benzene, and acrolein in 2022 under the RFS2-mandated volumes of biofuels, while NO_x, HC, and the other air toxics, especially ethanol and acetaldehyde, were projected to increase. The inclusion of E85 emissions effects would be expected to yield larger reductions in CO, benzene, and 1,3-butadiene, but more significant increases in ethanol, acetaldehyde, and formaldehyde (U.S. EPA, 2010b).

4.5.1.2 Biodiesel

Air emissions from combustion of some biofuels, such as ethanol, are relatively independent of feedstock or conversion process. However, biodiesel emissions may be highly variable depending on the feedstock type (Lapuerta et al., 2008; U.S. EPA, 2002). With respect to carbon content, plant-based biodiesel is slightly higher percentage-wise than animal-based biodiesel in gallon-per-gallon comparisons. For NO_x, PM, and CO, plant-based biodiesel tends to have higher emissions than animal-based biodiesel for all percent blends (U.S. EPA, 2002).

Studies of biodiesel and biodiesel blends show varying results depending on the fuel (i.e., type of biodiesel, biodiesel blend, type of base diesel), the vehicle being tested, and the type of testing. In general, combustion of biodiesel has been shown to decrease PM, CO, and HC emissions, increase NO_x emissions, and increase ozone-forming potential (Gaffney and Marley, 2009; U.S. EPA, 2002). It should be kept in mind that petroleum-based diesel-fueled vehicles are expected to emit significantly lower amounts of SO_2 because of the Heavy-Duty Engine and Vehicle Standards and Highway Diesel Fuel Sulfur Control Requirements (2007 Heavy-Duty

Highway Rule) and the availability of low-sulfur diesel fuel in the marketplace, which must be accounted for when considering the emission benefits of low SO_x biodiesel (U.S. EPA, 2001). Blending biodiesel in low percentages will not have much impact on sulfur emissions.

With respect to carbon content, plant-based biodiesel is slightly higher than animal-based biodiesel percentage-wise. For NO_x, PM, and CO, plant-based biodiesel tends to have higher emissions than animal-based biodiesel for all percent blends (U.S. EPA, 2002).

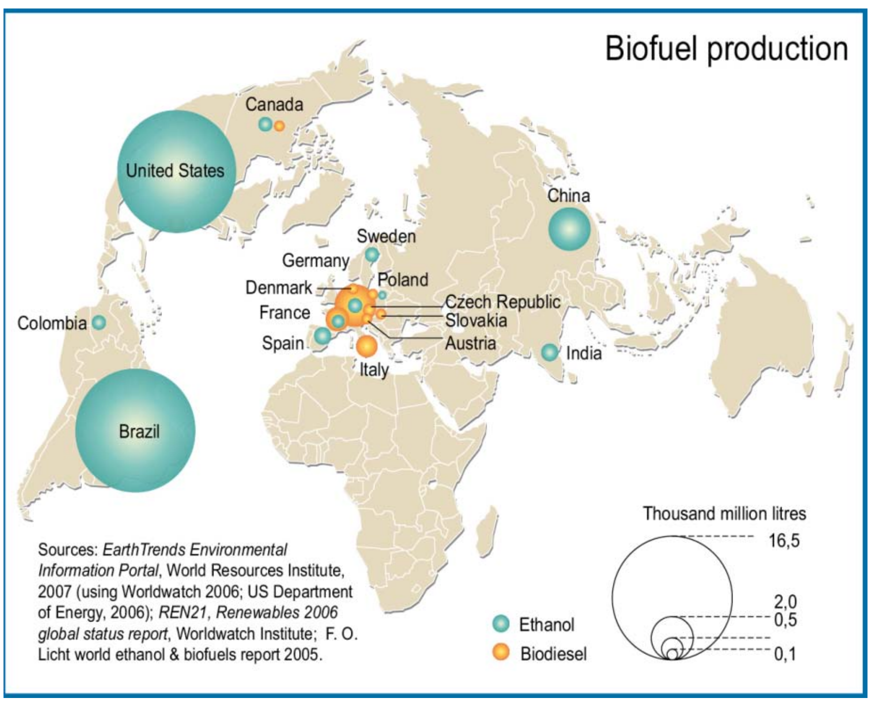
EPA's RFS2 RIA investigated the impacts of 20 volume percent biodiesel fuels on NO_x, PM, HC, and CO emissions from heavy-duty diesel vehicles, compared to using 100 percent petroleum-based diesel. Average NO_x emissions were found to increase 2.2 percent, while PM, HC, and CO were found to decrease 15.6 percent, 13.8 percent, and 14.1 percent, respectively, for all test cycles run on 20 volume percent soybean-based biodiesel fuel. Biodiesel results were included in the EPA analysis; however, the biodiesel contribution to overall emissions is quite small (U.S. EPA, 2010b).

5. INTERNATIONAL CONSIDERATIONS

5.1 Introduction

In the global context, biofuel demands from an increasing number of countries will have direct and indirect impacts, not only on countries that produce biofuels, but on countries that currently rely on imports of agricultural commodities from biofuel producers (Hertel et al., 2010; Pimentel et al., 2009; Zah and Ruddy, 2009). Section 204 of the Energy Independence and Security Act (EISA) calls for EPA to report to Congress the environmental impacts outside the United States caused by U.S. biofuel use. Thus, the following discussion focuses on potential impacts in foreign countries that may result from implementation of the RFS2 standards.

International trade is the primary mechanism through which foreign nations will be impacted by U.S. biofuel policy. Ethanol and, to a much smaller degree, biodiesel, have become global commodities. They are produced in many countries (Figure 5-1) and traded in international markets. Primary producers of ethanol are Brazil, the U.S., the European Union, India, and China. Brazil is the only significant exporter of ethanol (Fabiosa, 2010). Changes in U.S. production and consumption volumes, such as those in RFS2, will result in land allocation impacts that have global ramifications through international trade and market price. As a crop price rises, land will be reallocated to grow more of that crop in response to market price; conversely declining prices for a particular crop will tend to reallocate land away from that less profitable crop in favor of a more profitable one. Increased competition for arable land is expected to result in more land being allocated for crop production (Fabiosa, 2010).



Source: UNEP/GRID-Arendal, 2009.

Figure 5-1: Biofuel Production Map

Resulting environmental impacts, both positive and negative, include effects from land use change and effects on air quality, water quality, and biodiversity. From a U.S. perspective, the severity of these impacts will depend on the volume and location of future imports and exports, both of biofuel and displaced agricultural goods.

In 2008, the United States and Brazil together produced 89 percent of the world's fuel ethanol, with the U.S. producing around 9 billion gallons (see Table 5-1) (EIA, n.d. [c]). In 2009, U.S. ethanol production increased to 10.9 billion gallons, and similar increases occurred in most ethanol-producing nations as they attempted to increase the portion of biofuel in their energy mix (EIA, n.d. [c]). As a result, total world production has nearly doubled from 10.9 billion gallons in 2006 to 20.3 billion gallons in 2009. Figure 5-1 shows the geographical distribution of biofuel production. Patterns of ethanol consumption generally matched those of production, with the largest producers also being the largest consumers (EIA, n.d. [c]).

Table 5-1: Top Fuel Ethanol-Producing Countries from 2005 to 2009
(All figures are in millions of gallons)

Country/Region	2005	2006	2007	2008	2009
United States	3,904	4,884	6,521	9,283	10,938
Brazil	4,237	4,693	5,959	7,148	6,896
European Union	216	427	477	723	951
China	317	369	440	526	567
Canada	67	67	212	250	287
Jamaica	34	80	74	98	106
Thailand	18	34	46	87	106
India	57	63	69	71	89
Colombia	8	71	72	67	80
Australia	6	20	21	38	54
Other	93	216	276	393	274
Total World Production	**8,957**	**10,924**	**14,167**	**18,684**	**20,348**

Source: EIA, n.d. [c].

The market for the other major biofuel, biodiesel, is concentrated in Europe, which represented about 60 percent of world production as of 2009 (EIA, n.d. [b]). The other 40 percent of the market is largely made up by the United States, Brazil, Argentina, and Thailand, with U.S. production estimated at 505 million gallons for 2009, or about 10 percent of the world total (EIA, n.d. [b]). World biodiesel production has been rapidly increasing over the past decade, from 242 million gallons in 2000 to about 4.7 billion gallons in 2009 (EIA, n.d. [b]). These production increases have been driven by increased consumption targets. For instance, Brazil has planned to increase its biodiesel blend from 5 to 10 percent by 2015.

5.2 Import/Export Volumes

U.S. biofuel import volumes will depend largely on domestic production capacity, including the efficiency of the domestic ethanol-producing sector, the yields attained, and any excess demand left to be met via imported biofuel or biofuel feedstocks.

With respect to production capacity, as discussed in Chapter 2, the renewable fuel volume mandates under EISA require that U.S. biofuel consumption steadily increase to 36 billion gallons by 2022. This biofuel will be comprised of both conventional and advanced biofuel (including cellulosic ethanol, algal biodiesel, and other forms of advanced biofuel). Most of the 10.9 billion gallons of conventional ethanol that the U.S. produced in 2009 came from corn starch; by 2015, this figure is expected to increase to the targeted volume provided for in the revised Renewable Fuel Standards (RFS2) program (as required under EISA) of 15 billion gallons (GAO, 2007; U.S. EPA, 2010b). Future production volumes of advanced biofuel that have not yet been commercially developed are not quite as certain. In its RFS2 Regulatory Impact Analysis (RIA), EPA estimated that cellulosic technologies could combine to provide an additional 16 billion gallons of ethanol by 2022, with a substantial portion of this, 7.8 billion gallons worth, using corn stover as a cellulosic feedstock source (U.S. EPA, 2010b). In addition, U.S. biodiesel production is projected to increase to roughly 1.3 billion gallons by 2019 (FAPRI, 2010d). The RFS2 RIA also projected that the remaining 4 billion gallons needed to meet the EISA 2022 mandate would be comprised of a combination of imported sugarcane ethanol from Brazil as well as "other advanced biofuel," including cellulosic biofuel, biomass-based diesel, and co-processed renewable diesel (U.S. EPA, 2010b). Table 5-2 shows the projected import volumes forecasted in the RIA for each year from 2011 to 2022.

Table 5-2: RFS2 RIA Projected Imports and Corn Ethanol Production, 2011-2022

	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022
U.S. Corn Ethanol Production	12,070	12,830	13,420	14,090	14,790	15,000	15,000	15,000	15,000	15,000	15,000	15,000
Projected Imports	160	180	190	200	390	630	1,070	1,510	1,960	1,880	1,810	2,240

Source: U.S. EPA, 2010b, pp. 77-78.
Figures are in millions of gallons.

As Table 5-2 shows, import volumes will be at or below 200 million gallons in years preceding 2015, followed by a significant increase in import volumes between 2015 and 2022. This is in part because domestic corn starch ethanol production is expected to increase rapidly up until 2015 and then level off, and also because the RFS2 total renewable fuel requirements increase more rapidly in the later years. It should also be noted that 2010 import figures have been much lower than those expected when forecasts were made in 2009. Imports of fuel ethanol for the first three-quarters of 2010 (USDA, n.d.) have totaled 17 million gallons –well below EPA's 200 million gallons forecast (EPA, 2010b). Therefore, ethanol imports may be significantly lower than the projections in Table 5-2. U.S. biofuel imports and exports will also be influenced by trade policy, including tariffs and other incentives in the U.S. and in other countries. Even if the U.S. succeeds in meeting the RFS2 targets, the U.S. likely will continue to import and export biofuel as individual producers take advantage of international price differences. Over the past decade (2002 to 2009), U.S. ethanol import quantities varied considerably (see Table 5-3) , mostly due to volatility in the prices of related commodities such as corn, sugar, and other feedstocks, as well as prices of energy commodities such as oil.

Table 5-3: Historical U.S. Domestic Ethanol Production and Imports

	2002	2003	2004	2005	2006	2007	2008	2009
U.S. Production	2,140	2,804	3,395	3,904	4,884	6,521	9,283	10,938
Imports	13	12	149	136	731	439	530	198

Source: EIA, n.d. [c] for production figures; EIA, n.d. [d] for import figures.
Note: Figures are in millions of gallons.

The bulk of U.S. ethanol imports are sugarcane-based ethanol from Brazil. In 2008, the U.S. was the largest importer of Brazilian ethanol, followed by the Netherlands and a number of Caribbean countries (see Table 5-4). However, foreign-produced ethanol is also imported to the U.S. via these Caribbean countries where the Caribbean Basin Initiative (CBI), a regional trade agreement, enables up to 7 percent of the biofuel consumed in the U.S. to be imported from CBI member countries duty-free (Yacobucci, 2005; Farinelli et al., 2009). Therefore, most of the Brazilian exports shown as going to CBI member countries such as Costa Rica, Jamaica, El Salvador, Trinidad and Tobago, (see Table 5-4), is eventually re-exported to the United States. Looking closer at these Brazilian export figures in Table 5-4, it is evident that ethanol trade changed somewhat dramatically in 2009, with most destinations experiencing a significant decline in imports. A large part of this decline is due to the drop in U.S. imports caused by a change in energy prices, as well an increase in sugar prices that made imported Brazilian ethanol less competitive in the U.S. market (Lee and Sumner, 2010). These rising sugar prices, as

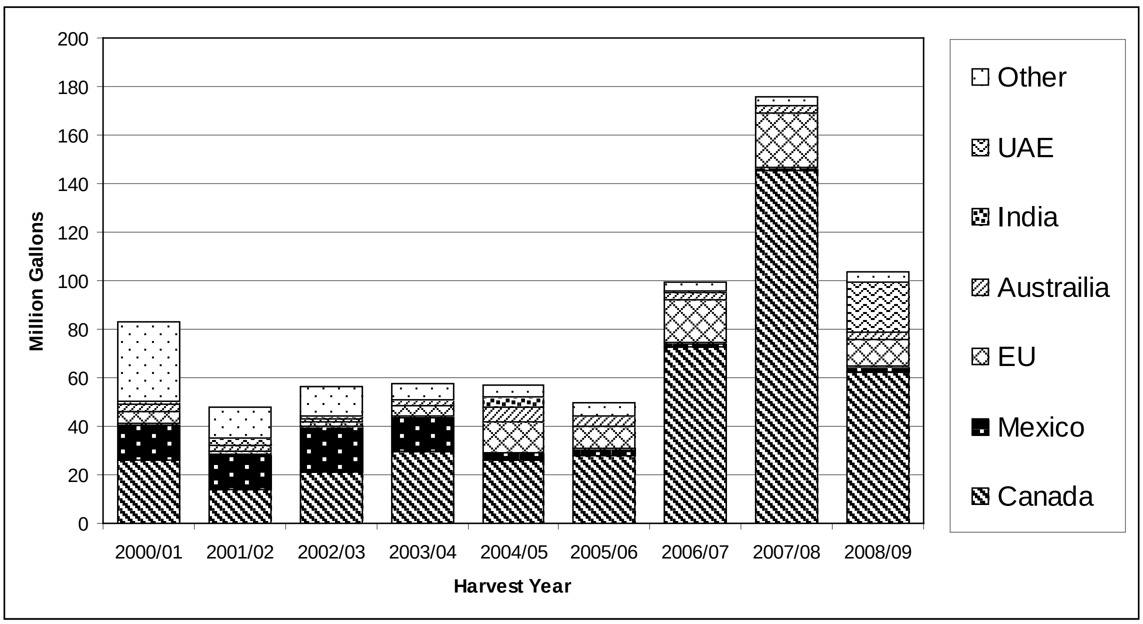
Table 5-4: 2008-2009 Brazilian Ethanol Exports by Country of Destination

Destination Country	Volume (million gallons)			
	2008	% of Total	2009	% of Total
Total	1,352.9	100%	870.8	100%
United States	401.6	29.7%	71.9	8.3%
Netherlands	351.9	26.0%	179.2	20.6%
Jamaica	115.3	8.5%	115.6	13.3%
El Salvador	94.1	7.0%	18.8	2.2%
Japan	69.6	5.1%	74.0	8.4%
Trinidad and Tobago	59.3	4.4%	37.0	4.2%
Virgin Islands (U.S)	49.7	3.7%	3.4	0.4%
Korea, Republic of (South Korea)	49.3	3.6%	82.9	9.5%
Costa Rica	28.9	2.1%	26.5	3.0%
Nigeria	25.9	1.9%	30.6	3.5%
United Kingdom	18.4	1.4%	42.7	4.9%

Source: SECEX, n.d.
Note: Percentages do not sum to 100 percent because some destinations are not listed. Original data were converted from liters to gallons.

well as the recent strengthening of Brazil's currency, could significantly hinder Brazil's ability to supply the U.S. market moving forward. Even if the 54-cent-per-gallon tariff on ethanol imports does expire as planned at the end of 2010, these factors may limit future imports (USDA, 2010d).

The U.S. also exports biofuel (including ethanol and biodiesel) to foreign countries. Canada has been the primary recipient of U.S. exports, with Europe becoming a more prevalent destination beginning in 2004 (see Figure 5-2) as its biofuel consumption has increased. U.S. ethanol exports have increased in recent years due to increased production. However, export levels, ranging from about 50 million to 175 million gallons, are no more than 1 percent of domestic production and are far outweighed by imports. Exports are likely to continue to lag behind imports in the near-term as consumption rises.



Source: ERS, 2010a.
Note: Original data were converted from liters to gallons, graph was created by ERG.

Figure 5-2: Historic U.S. Ethanol Export Volumes and Destinations

Table 5-5 shows the 2008 U.S. biodiesel trade balance. At that time, 46.8 percent of domestically produced biodiesel was exported. Biodiesel export volume has increased dramatically in recent years, from about 9 million gallons in 2005 to nearly 677 million in 2008 (EIA, n.d.[b]). In 2009, biodiesel export volume fell dramatically to only 266 million gallons (USDA, n.d.). Current projections have net U.S. biodiesel exports (i.e., exports minus imports) falling for the next few years and then rising back up to around 100 million gallons by the end of the decade (FAPRI, 2010d).

Table 5-5: 2008 U.S. Biodiesel Balance of Trade

Item	Quantity
U.S. Production	774 million gallons
U.S. Consumption	412 million gallons
Production – consumption =	362 million gallons
U.S. Imports	315 million gallons
U.S. Exports	677 million gallons
Exports – Imports =	362 million gallons

Source: EIA, 2009, n.d.[c].

5.3 Environmental Impacts of Direct and Indirect Land Use Changes

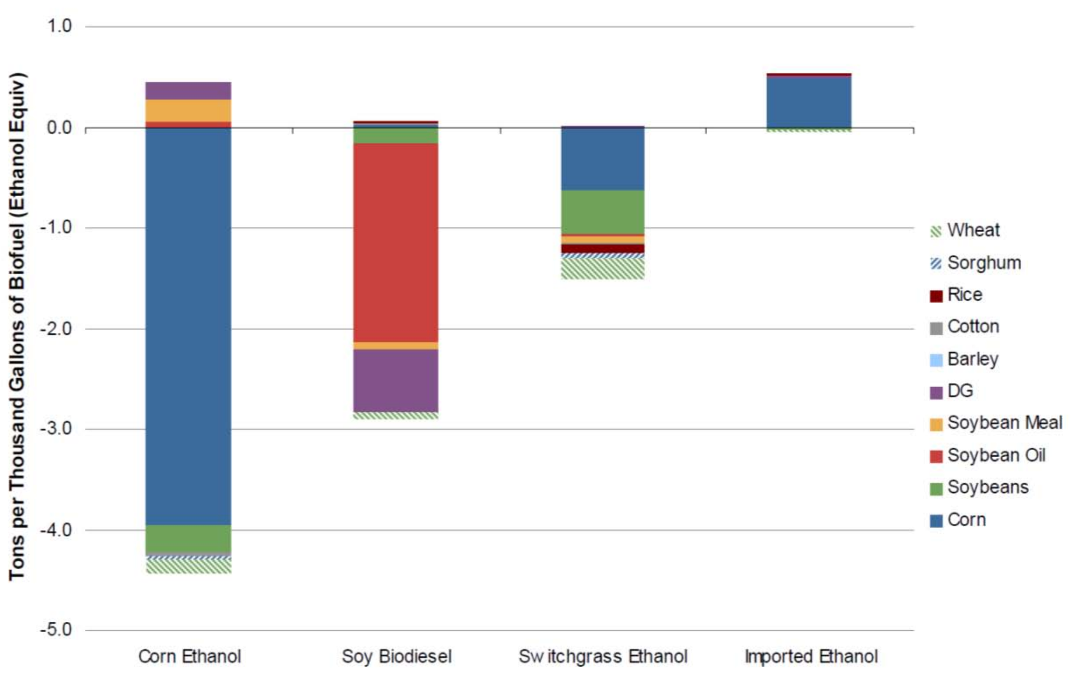
The issue of land use change inherently includes international considerations, as the demand for biofuel in the U.S. can influence the international availability of crops such as corn and soybeans for both biofuel and agricultural markets, which in turn can incentivize land use changes in other countries to meet that demand. In this report, *land use* is defined as “the human use of land involving the management and modification of natural environment or wilderness into built environment such as fields, pastures, and settlements”. Land use changes are considered either direct or indirect. In the context of biofuels, *direct land use change (DLUC)* refers to land conversion that is directly related to the biofuel supply chain. An example of direct land use change would be the planting of sugarcane on Brazilian land, which was previously native forest or used for another non-biofuel crop, for the purpose of increasing the supply of ethanol to export to the United States. *Indirect land use change (ILUC)* refers to land conversion that is a market-oriented response to changes in the supply and demand of goods that arise from increased production of biofuel feedstocks. An example of indirect land use change would be the clearing of foreign land to plant corn in response to reduced U.S. corn exports caused by increased U.S. corn ethanol production. Some have argued that these indirect impacts should not be counted as part of the biofuel carbon footprint because they are too difficult to relate back to biofuel production. However, EISA requires that “direct emissions and significant indirect emissions such as significant emissions from land use change” be considered as part of the analysis of environmental impacts stemming from domestic biofuel production and consumption.

In its RFS2 Regulatory Impact Analysis, EPA estimated greenhouse gas (GHG) impacts of direct and indirect land use change using the FAPRI-CARD model.[24] This model predicts world prices by equating excess supply and demand across countries. Changes in world prices determine changes in worldwide commodity production and trade. Under this model, two primary domestic effects directly affect a commodity’s worldwide use and trade: change in U.S. exports and changes in domestic U.S. prices (U.S. EPA, 2010b). Using this model, along with MODIS satellite data and other models, the RIA analysis compares 2022 crop area and production (by crop type and country) predicted to result with and without (i.e., “business as usual”) EISA requirements. The results of this analysis are shown in Figures 5-3, 5-4, 5-5, and

[24] FAPRI-CARD is a worldwide agricultural sector economic model. For the RIA, the model was run by the Center for Agricultural and Rural Development at Iowa State University on behalf of EPA.

5-6 and in Table 5-6. In Figures 5-3, 5-4, 5-5, and 5-6, each column shows the marginal impact of a scenario that focuses on that particular feedstock in isolation.

The model forecasts that, by 2022, for every increase of 1,000 gallons of corn starch ethanol production in the U.S., corn exports will have decreased by 4 tons. Similarly, for every increase of 1,000 gallons of soybean-based biodiesel produced domestically, soybean oil exports will have decreased by just over 2 tons (see Figure 5-3) (U.S. EPA, 2010b). Thus, as the U.S. increases domestic production of corn starch ethanol and soybean diesel, exports of corn and soybean for agricultural or other uses are expected to decline, which may result in indirect land use change in the form of land conversion to agriculture in other countries. This result is consistent with the results of a 2009 study, which predicted that due to production increases required by EISA, U.S. coarse grain exports will decrease to all destinations and this will cause dominant export competitors and trading partners, likely in Latin America, China, and the Pacific Rim, to convert more of their lands to make up the difference (Hertel et al., 2010; Keeney and Hertel, 2009). However, given that RFS2 limits the amount of corn starch ethanol that can be counted toward the mandated volume targets at 15 billion gallons─a level the U.S. is expected to reach by 2015 or sooner (GAO, 2007; U.S. EPA, 2010b), indirect land use change impacts resulting from changing trade patterns of corn and other grains may level off at that point. In fact, U.S. biofuel consumption could decrease pressure on conversion of land to agricultural use if agricultural yield improvements occur or if cellulosic technologies develop to replace conventional ethanol production.



Source: U.S. EPA, 2010b.

Figure 5-3: Change in U.S. Exports by Crop Anticipated to Result from EISA Requirements by 2022 (tons per 1,000 gallons of renewable fuel)

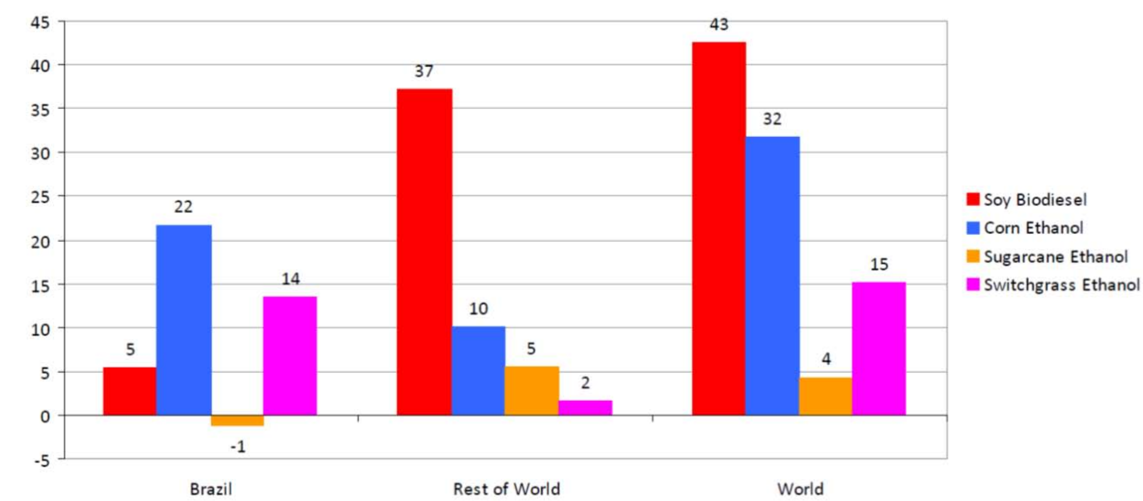
The model also predicts that the additional biofuel produced to meet the EISA mandates compared to “business as usual” (2.7 billion gallons of corn starch ethanol, 7.9 billion gallons of switchgrass cellulosic ethanol, 1.6 billion gallons of imported sugarcane ethanol, and 0.5 billion gallons of soybean biodiesel) will lead to the creation of 2 million acres, 3.4 million acres, 1.1 million acres, and 1.7 million acres, respectively, of additional international cropland (see Table 5-6) to supply U.S. biofuel imports and also to make up for the U.S. reductions in exports shown in Figure 5-3 (U.S. EPA, 2010b).

Table 5-6: Changes in International Crop Area Harvested by Renewable Fuel Anticipated to Result from EISA Requirements by 2022

Feedstock’s Marginal Effect Considered	International Crop Area Change (000s acres)	Normalized Crop Area Change (acre / billion BTU)
Corn Ethanol	1,950	9.74
Soy-Based Biodiesel	1,675	26.32
Sugarcane Ethanol	1,063	10.82
Switchgrass Ethanol	3,356	5.56

Source: U.S. EPA, 2010b.
Note: Figures converted from hectare to acre

Further, these direct and indirect land use changes will lead to significant GHG emissions according to the model (before accounting for GHG savings resulting from petroleum displaced as the biofuel is consumed). Figure 5-4 shows that, based on the model presented in the RIA, soy-based biodiesel causes the largest release of international land use change GHG emissions. The majority of international land use change emissions originate in Brazil in the corn ethanol and switchgrass ethanol scenarios. This is largely a consequence of projected pasture expansion in Brazil, and especially in the Amazon region where land clearing causes substantial

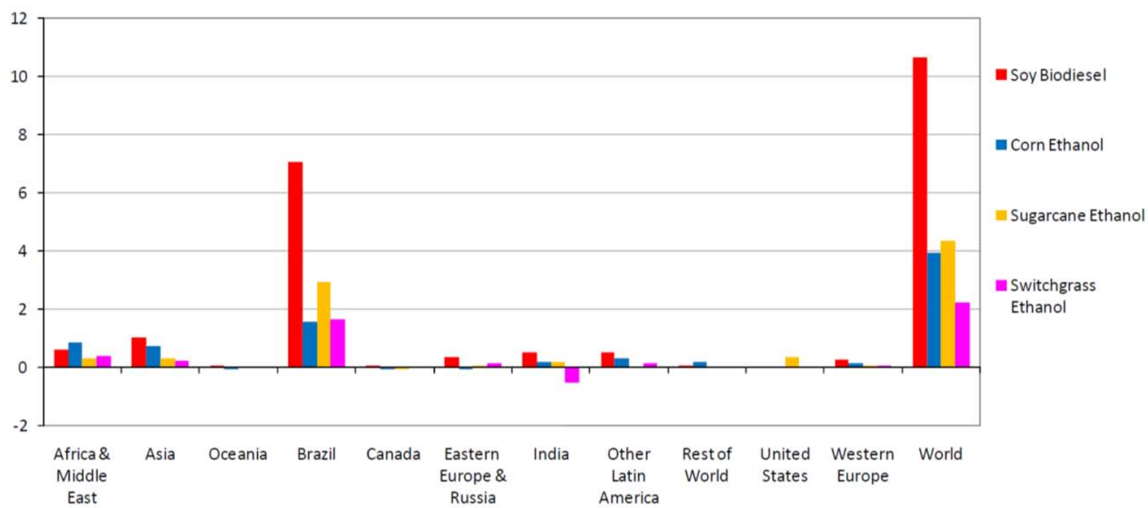


Source: U.S. EPA, 2010b.

Figure 5-4: International Land Use Change GHG Emissions by Renewable Fuel Anticipated to Result from EISA Requirements by 2022 (kgCO2e/mmBTU)

GHG emissions. Of the renewable fuels analyzed, the model found that sugarcane ethanol causes the least amount of land use change emissions. This was due largely to the EPA projection that sugarcane crops would expand onto grasslands in South and Southeast Brazil, which results in a net sequestration because sugarcane sequesters more biomass carbon than the grasslands it would replace.

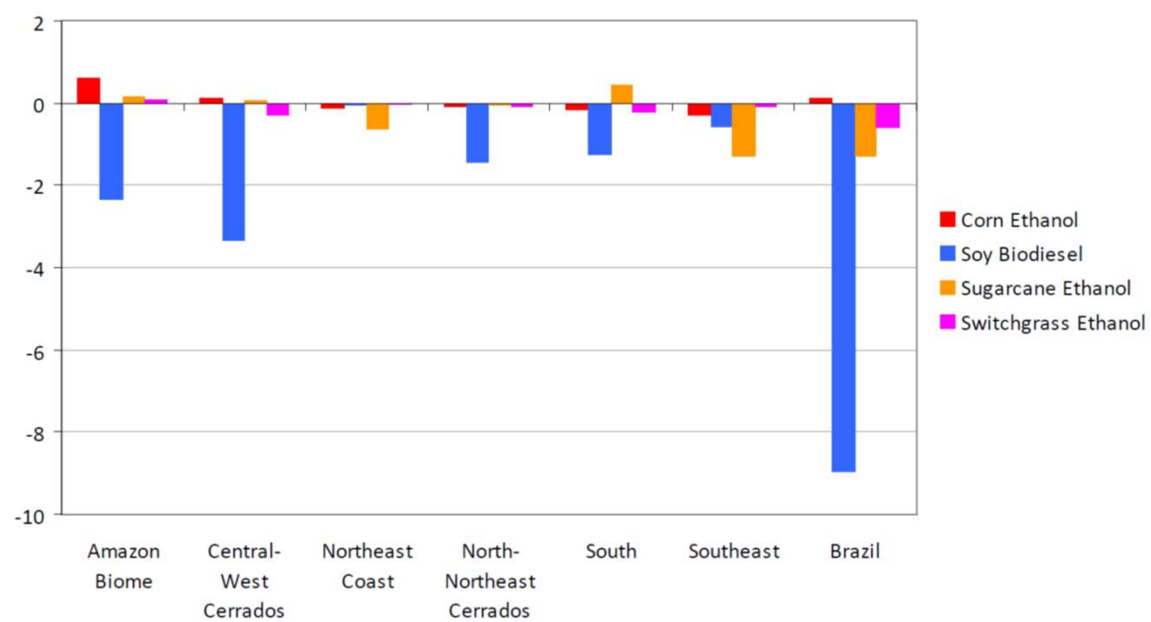
The GHG emissions shown above can be seen as an international “carbon debt” (Fargione et al., 2008). Clearing forested areas or pasture land for new cropland creates this carbon debt in which microbial decomposition of organic carbon stored in plant biomass (e.g. branches, leaves, and fine roots) and soils leads to a prolonged period of GHG emissions. As described in the RIA, the location of land use change is a critical factor determining the GHG impacts of land use change, because these impacts will vary substantially by region (U.S. EPA, 2010b). The conversion of higher carbon-storing types of land such as tropical rainforest will lead to more carbon emissions (U.S. EPA, 2010b). A 2008 study forecasted that land conversion of natural ecosystems to cropland would release an estimated 17 to 420 times as much carbon dioxide as the biofuels themselves can reduce per year by displacing petroleum fuel (Fargione et al., 2008). Therefore, biofuel consumption may take many years to “pay down” the carbon debt created from production through the GHG savings from displaced petroleum. On the other hand, biofuel made from more sustainable grasses or woody crops using higher-yield cellulosic technologies, or from waste biomass or biomass grown on degraded and abandoned agricultural lands results in much smaller carbon debts and is more likely to lead to overall GHG reductions (Fargione et al., 2008). Figure 5-5 shows forecasted crop area changes by region, with the heaviest impacts occurring in Brazil. It should be noted that the FAPRI-CARD model does not predict what type cropland will emerge in foreign countries if land use change does occur. This is an important source of uncertainty as GHG and other environmental impacts could vary significantly depending on what crops are grown to offset decreasing U.S. agricultural exports.



Source: U.S. EPA, 2010b.

Figure 5-5: Harvested Crop Area Changes by Region Anticipated to Result from EISA Requirements by 2022, 2022 (ha / billion BTU)

Because Brazil will likely be a major supplier of U.S. ethanol, it is informative to consider land use changes there. Brazil faces challenges of multiple forms of land use change, both direct and indirect. Land use changes would occur as Brazil increases ethanol production by converting more land previously used to grow other agricultural goods or pasture lands to grow sugarcane. As pasture lands are converted to sugarcane production, ranchers are pressured to “intensify” livestock on smaller portions of land or clear more land (possibly Amazon rainforest or Cerrado woodland) (Bustamante et al., 2009). Figures 5-4 and 5-6 isolate the impacts on Brazil alone. The data presented in Figure 5-6 appear consistent with the prediction that pasture land will decrease in Brazil, while increasing in the rest of the world. However, it is unclear if this will result in rainforest loss or simply mean a greater number of livestock per acre. There are differing opinions on the result of this tradeoff and it is not possible at this time to predict with any certainty what type of land use change will result from increased U.S. demand for biofuel and what its environmental consequences will be (Fargione et al., 2008; Goldemberg et al., 2008; Searchinger et al., 2008). A recent study (Fabiosa et al., 2010) suggests that sugarcane-based ethanol production in Brazil has less significant impact on existing arable land allocation than corn-based ethanol expansion in the United States. Fabiosa also notes that increasing corn starch-based ethanol to 15 billion gallons in the U.S. would increase corn crop area in the United States by nearly 13 percent (corn crop area in Argentina and Brazil would increase by 9.5 and 4.5 percent, respectively).



Source: U.S. EPA, 2010b.

Figure 5-6: Pasture Area Changes in Brazil by Renewable Fuel Anticipated to Result from EISA Requirements by 2022 (ha / billion BTU)

5.4 <u>Other Environmental Impacts</u>

While production of biofuel feedstocks places only one of many demands on water, fertilizer, and other inputs, its impacts will increase as its production increases. It has been suggested that, because biofuel production requires approximately 70 to 400 times as much water as other energy sources such as fossil fuels, wind, and solar, an increase in biofuel crop production could further strain global supplies of water (Gerbens-Leenes et al., 2008). Studies have shown that water tables are already declining in the western United States, North India, Pakistan, North China, Mexico, and the Mediterranean (Shah et al., 2007). These trends indicate the vulnerability of various regions to water scarcity issues. The choice of feedstock, cultivation practices, and the location of cultivation will greatly influence how production of biofuel impacts water availability.

Water quality and flooding issues are also relevant. As described in Chapter 3, U.S. corn production has been a key driver of hypoxia in the Gulf of Mexico. Similar water quality issues could arise or be exacerbated in other countries if agricultural use from feedstock production expands. Conversion of land to feedstock production will have varying impacts, depending on prior ecological function of the converted land and the types of management practices employed. Impacts could include encroachment on wetlands and the discharge of excess nutrients to water resources. For example, Brazilian surface waters suffered from hypoxia during the early stages of their biofuel development when the vinasse, a by-product of the sugarcane-ethanol production process rich in nitrogen and potassium, was routinely discarded into rivers, lakes, and reservoirs, causing extensive eutrophication (Simpson et al., 2009). Brazilian federal law has prohibited the dumping of vinasse into any water body since 1978. The effluent is now returned to the field as fertilizer, and water quality has improved significantly. However, if other developing countries opt to produce biofuel and do not properly regulate water quality impacts, eutrophication could damage these nations' aquatic ecosystems. Also, if biofuel-related land use change does occur and if it results in deforestation and loss of wetlands, then increased flooding, sedimentation, and lower stream base flows are also likely to occur. Examples of this have already been seen around the world. For instance, in Southeast Asia, tropical peat swamps have been degraded because of loss of forest cover due to logging for timber and conversion of forests to oil-palm plantations for biofuel (Wösten et al., 2006). However, biofuel production was not the only cause of land conversion, and it is possible that food-related demands for palm oil would have caused similar deforestation.

Biofuel production also affects international air quality. While the displacement of petroleum fuels by biofuels does have a positive impact, the air quality issues associated with biofuel feedstock harvesting, refining, and transport could erode these savings if poor management practices are allowed to occur. For instance, the practice of burning sugarcane fields prior to harvesting is a serious air pollution issue in Brazil. This method has resulted in large aerosol and trace gas emissions, significant effects on the composition and acidity of rainwater over large areas of southern regions, and elevated ozone levels in those areas affected by the burning. However, harvest burning practices are being phased out in Brazil through state regulations. In 2007, state laws ensured that 40 percent of the sugarcane was harvested without burning in the state of Sao Paulo, and this is forecast to reach 50 percent by 2010 and about 90 percent by 2022 (Goldemberg et al., 2008; U.S. EPA, 2010b). Like many of the effects discussed so far, the severity of air emissions will be highly sensitive to the feedstock chosen, location of production, and management practices.

Finally, if increased biofuel consumption in the U.S. does lead to indirect land use changes and more natural habitat is cleared to create agricultural lands, a loss of biodiversity will occur. Many biofuel production regions coincide with areas with high biodiversity value. For example, Indonesia (palm oil), Malaysia (palm oil), and Brazil (sugar ethanol) all contain ecosystems with well above average biodiversity. Depending where biofuel feedstock production occurs, and the manner in which it occurs, impacts to biodiversity could be significant.

5.5 Concluding Remarks

Projections indicate that the EISA biofuel targets will likely alter U.S. and international trade patterns. How countries respond to U.S. market conditions could affect net GHG savings derived from biofuel consumption and the environmental impacts that result from biofuel production. As with biofuel production in the U.S., these impacts will depend largely on where the crops are grown and what agricultural practices are used to grow them. To the extent that local environmental impacts will have broader implications, such as contributing to global warming, global mitigation strategies will have to consider the international implications of biofuel production. Decisions made about what feedstocks to use, where to produce them, and what production methods to employ will have significant environmental and economic implications.

6. CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

A variety of factors make it difficult to draw conclusions about the potential environmental and resource conservation impacts of the increased biofuel production and use mandated by the Energy Independence and Security Act. Of the six feedstocks discussed in this Report, only corn starch and soybean have been implemented at commercial scale to produce ethanol and biodiesel, respectively. Production of biofuel from the other four feedstocks discussed in this report is in various stages of research and development. Even for corn starch and soybean, data needed to perform a thorough environmental life cycle assessment are incomplete and the relevant available data often have a high degree of uncertainty. Nevertheless, initial conclusions can be drawn about how increased biofuel production and use likely will affect (or is affecting, in the case of corn starch and soybean) water quality and quantity, soil and air quality, and ecosystems (biodiversity and invasive species) based on the data available as of July 2010. These conclusions are presented below for the full greenhouse gas (GHG biofuel life cycle (Section 6.1.1) and for stages in the life cycle: feedstock production (Section 6.1.2); biofuel production, transport, and storage (Section 6.1.3); and biofuel end use (Section 6.1.4). (See Figure 2-3 in Chapter 2 for life cycle description.) These conclusions do not account for existing or potential future mitigation measures or regulations.

6.1.1 Emissions Reduction

Fuel combustion at ethanol and biodiesel facilities releases GHGs. However, when the entire biofuel life cycle is considered (as described in Chapter 4, Section 4.3.2.3), the revisions to the Renewable Fuel Standard (RFS2) program mandated by the Energy Independence and Security Act (EISA) are expected to achieve a 138-million metric ton reduction in CO_2-equivalent emissions by 2022.

6.1.2 Feedstock Production

6.1.2.1 Overview

Figure 6-1 provides a qualitative overview, based on EPA’s best professional judgment, of the maximum potential range of domestic environmental and resource conservation impacts associated with per unit area production of the six feedstocks discussed in this Report. Qualitative assessment is grounded in information and data published in the peer-reviewed literature through July 2010, which are described in Chapter 3. Range extremes for each impact category were determined by considering plausible conditions under which a “most negative” and “most positive” environmental impact would likely arise. Key assumptions for these conditions appear in Figure 6-1; for full, detailed elaboration of the conditions, which encompass a variety of factors, including land use, feedstock production management choices, region, technology used, regulatory control, and mitigation measures, see Appendix C, Table C-1.

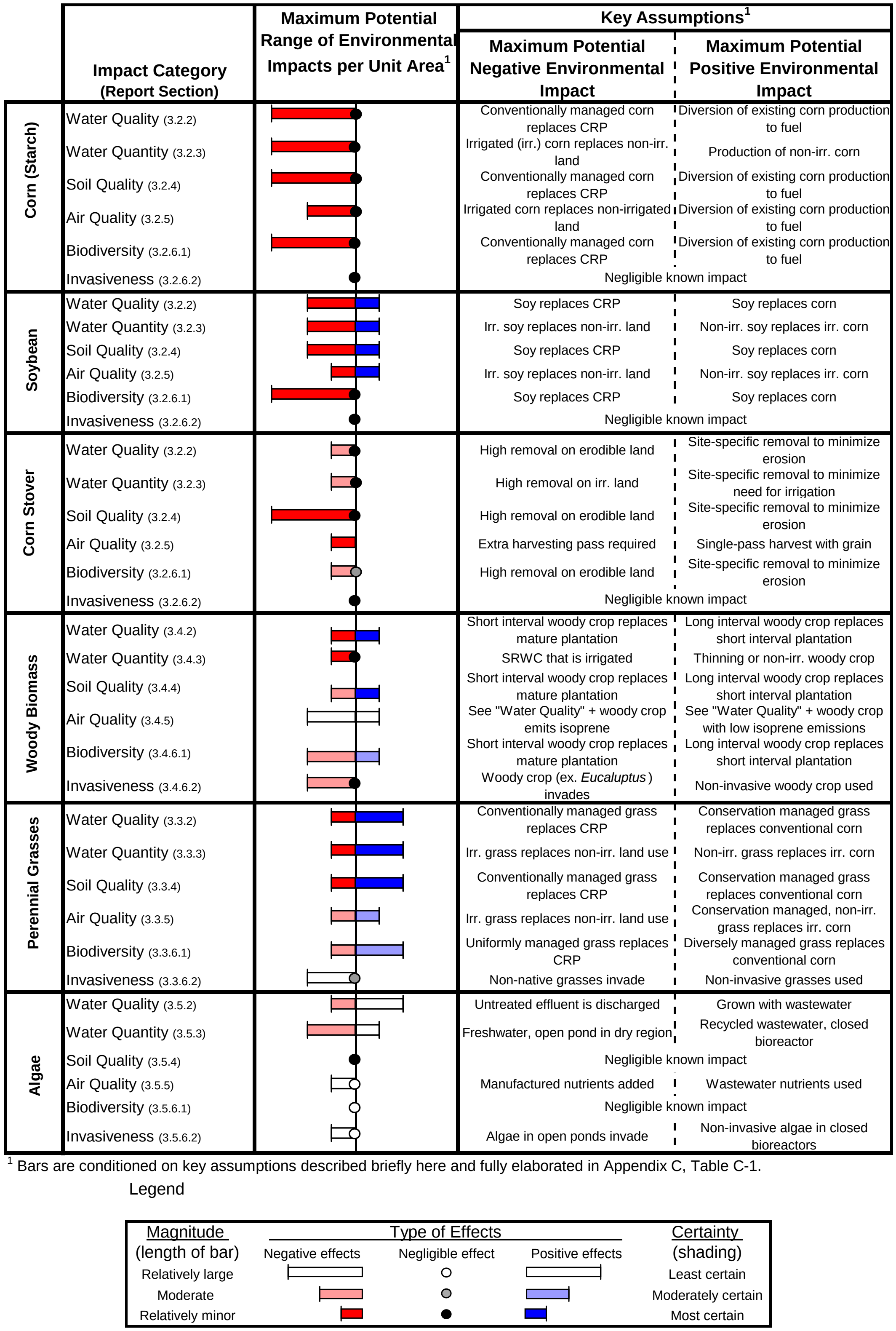


Figure 6-1: Maximum Potential Range of Environmental Impacts (on a Per Unit Area Basis) Resulting from Cultivation and Harvesting of the Six Biofuel Feedstocks Considered in This Report

Impacts shown in this figure are only relative to each other. No attempt has been made to compare impacts to those of petroleum production, nor do impacts represent possible environmental benefits gained by petroleum displacement. In addition, impacts are only relevant for those regions where each feedstock is likely to be grown (see Chapter 3). Impacts for corn stover do not include the impacts of corn production itself but rather impacts of stover removal above and beyond corn cultivation and harvest. Air quality impacts do not include changes in GHG emissions.

Bar direction signifies whether the effect is negative (left) or positive (right). *Bar length* indicates the anticipated magnitude of effect, and *shading density* depicts the associated degree of certainty. A *circle* signifies that no net effect is anticipated. Section numbers next to the impact category indicate where in this Report the information that provides the basis for the bars in this figure can be found.

When the potential range of production conditions is considered, four feedstocks (soybean, woody biomass, perennial grasses, and algae) are anticipated to have both negative and positive environmental impacts. The most positive environmental outcome for corn starch and corn stover is no net effect, achieved largely through minimization of land use change and through site-specific agricultural management, including comprehensive conservation practices. Most feedstocks (corn starch, soybean, corn stover, woody biomass, and perennial grasses) have the potential to have impacts in at least five of the six environmental categories shown in the figure; algae are anticipated to have impacts in only four of the categories (water quality, water quantity, air quality and invasiveness). A higher degree of certainty is associated with the two feedstocks that are already commercially produced (corn starch and soybean) than with those in development (corn stover, woody biomass, perennial grasses, and algae).

6.1.2.2 Conclusions

Key conclusions concerning environmental impacts of biofuel feedstock cultivation are as follows:

Water Quality. Increased cultivation of feedstocks for biofuel may affect water quality and hypoxia conditions in the Gulf of Mexico and other vulnerable water bodies through increased erosion and runoff and leaching of fertilizers and pesticides to ground and surface waters. Cellulosic feedstocks may have less water quality impact than corn starch and corn stover due to projected decreased fertilizer use and decreased soil erosion. Comprehensive management systems and practices are one tool that may mitigate some of these impacts if they are widely and effectively implemented. Compared to corn and soybeans, cultivation of some cellulosic feedstocks may provide benefits, including soil stabilization, reduced soil erosion and nutrient runoff, and increased nutrient filtration.

Water Quantity. Effects of feedstock production on water availability vary greatly by feedstock, processes used to produce the feedstock, and location. Corn and soybean cultivation for biofuel production have a greater water demand than perennial grasses, woody biomass, and algae. Regional differences are mostly due to precipitation which, when insufficient, necessitates supplemental irrigation, which can be a significant water use in the biofuel production process. In irrigated regions, the method and efficiency of irrigation can also affect the amount of water used. For both corn and soybeans, the source for irrigation water varies from region to region, potentially affecting water tables and/or surface waters. Removal of corn stover can reduce soil moisture, resulting in a need for increased irrigation. Water quantity effects may be mitigated by growing feedstocks in areas that do not require irrigation and by using efficient irrigation practices, such as reclaimed water use.

Soil Quality. Increased cultivation of corn, soybean, woody biomass, and perennial grasses will affect soil quality in various ways, depending on the feedstock. Effects include increased soil erosion, decreased soil organic matter content, increased soil GHG emissions, and increased nitrogen and phosphorus losses to ground and surface waters. Annual crops, such as soybean and corn, will have higher erosion rates than non-row crops, such as perennial grasses and woody biomass. However, cultivation of corn or soybean at higher rates (i.e., greater yield per acre) on existing corn or soybean acreage likely will not alter soil erosion rates significantly. Soil quality impacts from biofuel feedstocks may be ameliorated by the choice of feedstock and by the diligent use of generally accepted conservation practices.

Air Quality. Activities associated with growing biofuel feedstocks emit air pollutants, which affect air quality, with effects varying by region. Production of row crops will affect air quality more than non-row crops. Pollutants from row crops include farm equipment emissions and soil and related dust particles (e.g., fertilizer, pesticide, and manure) made airborne as a result of field tillage and fertilizer application, especially in drier areas of the country.

Biodiversity. Increased cultivation of corn and soy feedstocks could significantly affect biodiversity (1) through habitat alteration when uncultivated land is moved into production, and (2) from exposure of flora and fauna to high pesticides concentrations. Aquatic habitat may be impaired by soil erosion and nutrient runoff. Biodiversity impacts can be mitigated by choosing crop and cultivation methods that minimize habitat alteration and runoff.

Invasiveness. Corn and soybean pose little risk of becoming weedy or invasive in the U.S. In certain regions, some perennial grasses, short-rotation woody crops, and algae strains pose greater, though uncertain, risk of becoming an agricultural weed or invasive in natural areas. Transport of grass and short-rotation woody crop seeds and plant parts capable of vegetative reproduction from the field to biofuel production facilities may increase the opportunity for seeds and plant parts capable of vegetative reproduction to establish themselves in feral populations along transportation corridors. Algae produced in photo-bioreactors are less likely to become invasive than algae produced in open ponds.

6.1.3 Biofuel Production, Transport, and Storage

As described below, biofuel production, transport, and storage can impact water quality, water quantity, and air quality.

6.1.3.1 Overview

Figure 6-2 provides a qualitative overview, based on EPA's best professional judgment, of the maximum potential range of domestic environmental and resource conservation impacts associated with per unit volume production, transport, and storage of ethanol from corn and cellulosic feedstocks and biodiesel from soybean (though biodiesel from algae should not be appreciably different). Qualitative assessment is grounded in information and data published in the peer-reviewed literature through July 2010, which are described in Chapter 4. As with Figure 6-1, range extremes for each impact category were determined by considering plausible conditions under which a "most negative" and "most positive" environmental impact would likely arise. Key assumptions for these conditions appear in Figure 6-2; for full, detailed elaboration of the conditions, which encompass a variety of factors, including region, technology used, regulatory control, and mitigation measures, see Appendix C, Table C-2.

Impacts shown in this figure are only relative to each other. No attempt has been made to compare impacts to those of petroleum production, nor do impacts represent possible environmental benefits gained by petroleum displacement.

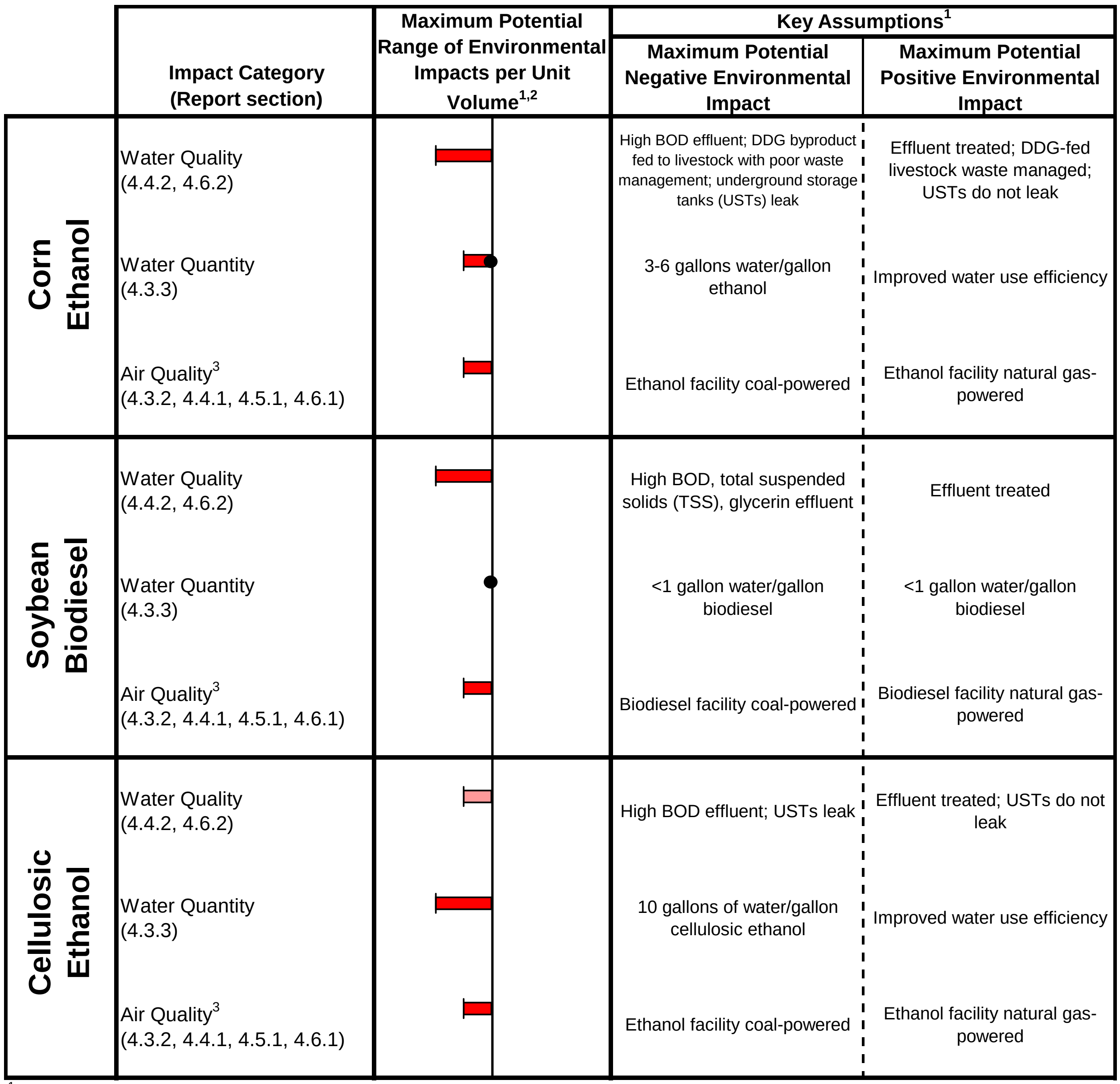
Bar conventions used in Figure 6-1 are the same as those used in Figure 6-2.

As Figure 6-2 illustrates, the environmental impacts of biofuel production, transport, and storage are expected to be largely negative (see Chapter 4 for more details). However, for all three fuel types, impacts can be minimized through appropriate facility siting, waste treatment, and improved, more efficient technology.

Water Quality. Pollutants in the wastewater discharged from biofuel production impact water quality. Biological oxygen demand (BOD), brine, ammonia-nitrogen, and phosphorus are primary pollutants of concern from ethanol facilities. BOD, total suspended solids, and glycerin pose the major water quality concerns in biodiesel facility effluent. Actual impacts depend on a range of factors, including the type of feedstock processed, biorefinery technology, effluent controls, and water re-use/recycling practices, as well as the facility location and source and receiving water.

Water Quantity. Biofuel production facilities draw on local water supplies to produce fuel, but the quantity of water used is modest compared to that required to produce biofuel feedstocks. Impacts will depend on the location of the facility in relation to water resources. Water availability issues can be mitigated by siting production facilities where water is abundant.

Air Quality. Emissions from biofuel production facilities are generated primarily by the stationary combustion equipment used for energy production. Compared to two scenarios ([1] the original renewable fuel standard of 7.5 billion gallons, and [2] a 2022 renewable fuel volume of 13.6 billion gallons projected by the Department of Energy's 2007Annual Energy Outlook), RFS2-mandated increased biofuel production will likely result in decreased emissions of carbon monoxide and benzene, and increased emissions of nitrogen oxides, volatile organic compounds, particulate matter, and several air toxics. Since biofuel production facilities are regulated under the Clean Air Act and subject to state/local permits, enforcement of existing regulations will mitigate air quality impacts. Emissions can be further reduced through use of cleaner fuels (e.g., natural gas instead of coal) and more efficient process and energy generation equipment.



[1] Bars are conditioned on key assumptions described briefly here and fully elaborated in Appendix C, Table C-2.

[2] Comparisons are made on the basis of equal volumes of the biofuels indicated.

[3] Impacts shown are immediate impacts from biofuel production to end use. No attempt is made in this table to represent air quality impacts based on displaced gasoline emissions. See Section 4.5 for more information.

Legend

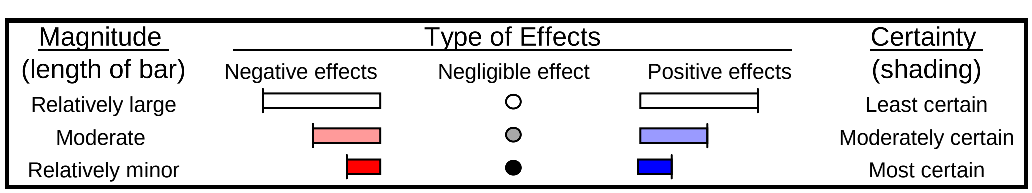


Figure 6-2: Maximum Potential Range of Environmental Impacts (on a Per Unit Volume Basis) Resulting from Ethanol and Biodiesel Production, Transport, and Storage

6.1.3.2 Biofuel Transport and Storage

Biofuel transport and storage may impact water and air quality.

Water Quality. Leaks and spills of biofuel from above-ground, underground, and transport tanks can potentially contaminate ground, surface, and drinking water. A leaking underground storage tank can also present other health and environmental risks, including the potential for fire and explosion. Enforcement of existing regulations concerning corrosion protection, leak detection, and spill and overfill prevention will minimize water contamination. Selection and use of appropriate materials and proper design in accordance with the applicable material standards will also prevent biofuel leaks.

Air Quality. Air quality will be affected by emissions from biofuel transport via rail, barge and tank truck and by evaporative, spillage, and permeation emissions from transfer and storage activities. However, the impacts are not expected to be significant.

6.1.4 Biofuel End Use

Air Quality. Evaporative and tailpipe emissions from biofuel combustion show great variability due to a range of factors, including the vehicle age, how the vehicle is operated, and ambient temperatures. Emissions in 2022 are expected to be higher for some pollutants and lower for others compared to two scenarios (described in 6.1.3.1). In general, biodiesel combustion has been shown to decrease particulate matter, carbon monoxide, and hydrocarbon emissions, increase nitrogen oxide emissions, and increase ozone-forming potential compared to fossil fuel diesel. Emissions from ethanol use are independent of feedstock; in contrast, emissions from biodiesel use differ according to the feedstock. Particulate matter, nitrous oxide, and carbon monoxide emissions are higher for plant-based biodiesel than for animal-based biodiesel.

6.1.5 International Considerations

Increases in U.S. biofuel production and consumption volumes will affect many different countries as trade patterns and prices adjust to equate global supply and demand. This will result in environmental impacts, both positive and negative, including effects from land use change and effects on air quality, water quality, and biodiversity. Direct and indirect land use changes will likely occur across the globe as the U.S. and other biofuel feedstock-producing countries alter their agricultural sectors to allow for greater biofuel production. Many locations where biofuel production is growing are areas of high biodiversity value. For example, Indonesia (palm oil), Malaysia (palm oil), and Brazil (sugar ethanol) all contain ecosystems with well-above-average biodiversity. Depending where biofuel feedstock production occurs, impacts to biodiversity could be significant. Particularly in Malaysia and Indonesia, which have already lost considerable forest cover due to their large timber industries, expansion of palm oil plantations for biodiesel could potentially compound impacts on natural resources. However, because corn ethanol, the biofuel with the greatest potential for international impact in terms of trade pattern changes, is limited by the RFS2 and is likely to reach this limit in the next few years, these international impacts could level off as corn starch ethanol production levels off or is replaced by more advanced technologies.

As with domestic production, the choice of feedstock, how and where it is grown, the resulting land-use changes, and how it is produced and transported will have a large effect on how biofuel production and use affects water quality and availability, air quality (e.g. due to emissions from burning crop residue), and biodiversity. The specific impacts will reflect a country's particular circumstances.

6.2 Recommendations

EISA Section 204 specifies that EPA include recommendations for actions to address any adverse impacts identified in this report. Responding specifically to this request requires a clear understanding of biofuel impacts and their causes. Impacts from corn starch and soybean production are relatively well understood, however, more information is needed about the adverse impacts associated with production of other feedstocks and with the production and use of advanced biofuel. This section presents four recommendations to address adverse impacts. Because biofuel impacts cross multiple topics and EPA responsibilities, EPA likely will address these recommendations through continued and strengthened cooperation with other federal agencies and international partners.

6.2.1 Comprehensive Environmental Assessment

The biofuel industry is poised for significant expansion in the next few years. A variety of new technologies likely will be implemented and old technologies modified to meet the demands of affordable and sustainable petroleum fuel alternatives. As emphasized by Congress in requiring triennial biofuel impact assessments, it is important to evaluate the environmental implications associated with the ongoing growth of the dynamic biofuel industry. However, as noted earlier, the inherent complexity and uncertainty of environmental impacts across the biofuel supply chain make it difficult to provide assessments that are sufficiently definitive to inform environmental decisions.

RECOMMENDATION: Develop and evaluate environmental life cycle assessments for biofuels. With this Report, EPA and the U.S. Departments of Agriculture and Energy (USDA and DOE) have begun to develop a framework and partnership that provide an important foundation for future assessments. Future assessments will address advanced biofuel production associated with specific feedstocks and associated by-products and provide a comparative context to fossil fuels. As described in Chapter 7, future assessments will be comprehensive and will address the major environmental parameters affected by increased biofuel production and use. These assessments will identify gaps and uncertainties in the knowledge base; inform the design and implementation of monitoring strategies and measures for evaluating impacts; provide comprehensive tools for comparing and evaluating development options; and provide the scientific bases for regulatory agencies and the biofuel industry to make environmentally conscious decisions.

6.2.2 Coordinated Research

The biofuel industry is expected to expand rapidly and broadly. This expansion will be shaped to a large degree by the research behind the technological developments that make biofuel production feasible. It will be important for the scientific infrastructure that supports policy and decision-making to keep pace with industry developments.

RECOMMENDATION: **Ensure the success of current and future environmental biofuel research through improved cooperation and sustained support.** The Biomass Research and Development Board, co-chaired by DOE and USDA, currently monitors interagency biofuel research cooperation. The Board recently proposed that an inventory be conducted of federal activities and jurisdictions relevant to environmental, health, and safety issues associated with biofuel production in order to identify issues of concern, research needs, and mitigation options. Efforts to adjust and expand existing research programs to conduct biofuel-relevant research have been initiated. Prioritization and collaboration by the research community will be critical to provide meaningful results in the near term and to meet the wide variety of research needs, including many that have already been identified, that will be important to the industry and to appropriate regulatory oversight.

6.2.3 Mitigation of Impacts from Feedstock Production

As the biofuel industry expands, it will be important to optimize benefits while minimizing adverse impacts. Since many of the known adverse impacts are due to feedstock production, this Report has described the potential for mitigation of those impacts through the adoption of conservation systems and practices on farms. USDA has a variety of programs that help agriculture producers implement these conservation systems. As USDA's Conservation Effects Assessment Project (CEAP) report on the Upper Mississippi River Basin demonstrates, much more needs to be done to control pollution from agriculture, especially from nitrogen. A collaborative effort is needed to develop and foster application of consistent and effective monitoring and mitigation procedures to protect the environment and conserve biodiversity and natural resources as biofuel production expands and advanced biofuels are commercially produced.

RECOMMENDATION: **Improve the ability of federal agencies (within their existing authorities) and industry to develop and implement best management and conservation practices and policies that will avoid or mitigate negative environmental effects from biofuel production and use.** These policies and practices should be aligned and assessed within the context of the environmental life cycle assessment and take a multi-factor and multi-scale view of biofuels and their potential environmental effects. Priority areas for development include (1) improved containment processes that minimize environmental exposure from air emissions and runoff into surface and ground water, and (2) methods to monitor, track, and report biofuel environmental impacts.

6.2.4 International Cooperation to Implement Sustainable Biofuel Practices

EISA specifically identifies "significant emissions from land use change" as a potential environmental impact stemming from domestic biofuel production and consumption. This concern is relevant to all countries engaged in biofuel production, but as the U.S. increases domestic production of corn starch ethanol and soybean diesel, exports of corn and soybean for agricultural or other uses are expected to decline, which may result in indirect land use change in the form of land conversion to agriculture in other countries. Additional biofuel produced to meet the EISA mandates will potentially lead to increases in acreages of international cropland, although these increases may level off after 2015 (see Section 5.2).

RECOMMENDATION: Engage the international community in cooperative efforts to identify and implement sustainable biofuel practices that minimize environmental impact. U.S. and international capacity to minimize the consequences of land use change will depend not only the willingness of governments and industry to make environmentally sound choices regarding biofuel production, processing, and use, but also on the availability of cost-effective mitigation strategies. The U.S. can significantly contribute to such an effort by actively engaging the scientific community and biofuel industry to collaboratively develop the body of knowledge needed to support sound environmental decision-making. This effort will be facilitated by a greater understanding and appreciation of how increased biofuel demand may impact the environment internationally, particularly in countries that are most active, or most likely to become active in biofuel production.

7. ASSESSING ENVIRONMENTAL IMPACTS FROM BIOFUELS: 2013 TO 2022

7.1 Introduction

In requiring EPA to report triennially under EISA Section 204, Congress recognized that the environmental and resource conservation impacts of increased biofuel production and use will be dynamic, changing in both nature and scope, based on the amount, type, and location of biofuels produced and used. This first triennial Report to Congress, which reflects the state of scientific knowledge as of July 2010, is a first step toward identifying information that supports future assessment of environmental impacts from increased biofuel production and use.

This chapter outlines an approach EPA will use for its future assessments, beginning with the 2013 Report to Congress. In developing future assessments, EPA will work closely with the U.S. Departments of Agriculture and Energy (USDA and DOE) and will seek extensive input from industry and other stakeholders and peer review from the scientific community to create substantive, science-based analyses that facilitate environmental decision-making. Future assessments will benefit from advances in the science of environmental assessment and increased availability of relevant research results on this important topic.

EPA anticipates that additional research and analyses will allow for more robust and quantitative assessments of biofuel environmental impacts than are reported here. For example, life cycle assessment (LCA) tools and approaches that are currently used for evaluating “cradle-to-grave” resource consumption and waste disposal for specific products can be integrated into risk assessment to form a powerful composite approach for assessing environmental impacts. An approach to more comprehensive environmental analyses that is consistent with the integration of LCA and risk assessment methods has been used in different assessments (Davis and Thomas, 2006; Davis 2007). This approach would necessitate extending consideration of factors across the entire biofuel life cycle, including current and future feedstock production and biofuel conversion, distribution, use. The Agency has already applied LCA to assess greenhouse gas (GHG) emissions as part of its revised Renewable Fuel Standard (RFS2) program (U.S. EPA, 2010b) and could adapt this approach to analyze other aspects of biofuel production and use, such as water consumption; evaluation of fossil fuels versus biofuels; net energy balance; production and use scenarios; and market impacts (economics).

7.2 Components of the 2013 Assessment

This section briefly describes key components that EPA plans to utilize in conducting its 2013 assessment. Comprehensive environmental assessment (CEA) would provide an organizing framework for evaluating and, where possible quantifying, risk and benefits of biofuel production and use. CEA would integrate LCA, described in Section 7.2.1, and environmental risk assessment, described in Section 7.2.2. The latter could be used to systematically assess environmental risks, both human health (see Section 7.2.3) and ecological, for each stage in the life cycle and potentially cumulative impacts. Conceptual models (Section 7.2.4) will illustrate the important factors being considered in each stage of the life cycle and indicate how these factors are interrelated. Where possible, environmental indicators and other metrics (Section 7.2.5) will be developed over the next several years to track the impacts of biofuel production and use throughout its life cycle and measure the effectiveness of regulatory and voluntary practices in ameliorating these impacts. A scenario-based approach (Section 7.2.6) is currently envisioned to provide a comparative basis for projecting and assessing how biofuel production and use will affect the environment in future years. Finally, the 2013 assessment will include other components, such as a comparison to fossil fuels, net energy balance, and analysis of market impacts (Section 7.2.7), that are important to evaluating biofuel impacts.

7.2.1 Life Cycle Assessments

LCAs have been widely used to assess the potential and pitfalls for bio-ethanol as a transportation fuel (von Blottnitz and Curran, 2007). The majority of such analyses have focused on particular components such as GHG emissions and energy balances (Hill, 2009), with varied results based on the assumptions and input parameters used to drive assessments. In some cases, the scientific community seems close to reconciling the various assumptions used by different investigators (Anex and Lifset, 2009). To better address the EISA reporting mandate, however, a broader profile of potential environmental impacts should be considered. This approach has been used in several studies (von Blottnitz and Curran, 2007) and applied to evaluating trade-offs for fuel options (Davis and Thomas, 2006). As part of the 2013 assessment, EPA anticipates utilizing LCA in a broad context, one that considers a full range of potential environmental effects and their magnitude. A variety of environmental LCA approaches have been developed that would prove useful for such an effort (Duncan et al., 2008; Ekvall, 2005; Hill et al., 2006; Puppan, 2002).

7.2.2 Environmental Risk Assessment

Environmental risk assessment will be fundamental for systematically evaluating the human and environmental impacts of the activities involved in biofuel production and use. Environmental risk assessment can be used to estimate the risks associated with each stage of the biofuel life cycle, from production of raw materials through transportation to waste products. Environmental risk assessment is initiated by clearly articulating the problem (i.e., problem formulation), describing the critical factor, pathways, and linkages among these factors, quantifying human/ecological exposure and effects, and subsequently characterizing and estimating the risks associated these effects. Environmental risk assessment will identify which stages in the biofuel life cycle contribute the greatest risk so that more informed risk management practices can be developed and implemented for these stages.

7.2.3 Human Health Assessment

Increasing biofuel use presents the potential for distinct health effects separate from the known impacts of fossil fuels. The fate and transport of these new fuel blends in the environment and the subsequent exposures and human health effects have not been fully studied. Drawing definitive conclusions on health impacts is not realistic at this time, given the unknowns surrounding the feedstocks, technologies, and fuel blends that will be used to meet target volumes, and the relatively limited availability of toxicological data to directly evaluate the potential health effects of the various emissions.

Health effects will be assessed in the 2013 report, provided adequate data are available. In examining the health risks and benefits of increased biofuel use, it will be important to understand the unique characteristics of the new fuel blends; how and when releases occur; the fate and transport of these releases; the relevant routes and duration of exposures to humans; and the toxic effects of those exposures. Both individual and population exposures will be important to consider. For example, populations in regions that both produce and use biofuel will experience different exposures than those in regions that only use the fuel. Individuals within the same region may experience different exposures (i.e., occupational, consumer, or public exposures), and vulnerable populations may be at greater risk of adverse effects, depending on their sensitivity.

7.2.4 Conceptual Models

A number of tools are available for use in problem formulation, including conceptual diagrams, which hypothesize relationships between activities and impacts. These diagrams can support multiple purposes, including defining system boundaries; enhancing understanding of the system being analyzed; and supporting communication among assessors, between assessors and stakeholders, and, ultimately, with risk managers.

The information provided in Chapters 3, 4, and 5 of this 2010 assessment lay a foundation for constructing initial conceptual models to show relationships among biofuel activities and impacts. Figures 7-1 and 7-2 present generalized conceptual models for feedstock and biofuel production, respectively. Appendix D provides detailed conceptual diagrams for each of the feedstocks and fuels considered in this Report. Based on the information gathered during this 2010 assessment, the diagrams show the activities (e.g., crop rotation, water use) associated with the model's domain area and how, through a series of relationships indicated with lines and arrows, these activities are associated with products and impacts. These diagrams are the first step in mathematically simulating the system and quantifying impacts. Diagrams such as these will be important tools for assessments in EPA's future Reports to Congress.

7.2.5 Monitoring, Measures, and Indicators

EPA's ability to accurately assess impacts attributable to biofuels production and use will depend on having timely, relevant, and accurate monitoring information that tracks potential impacts, and how effective regulatory and voluntary management practices, risk management practices, and other measures are in protecting the environment. While current environmental monitoring by various agencies can provide helpful information, targeted monitoring for potential biofuel impacts will be needed, requiring a collaborative effort across multiple agencies and other organizations. Indicators and measures will be important for a variety of environmental effects, including GHG emissions, human and ecological health indicators, eutrophication, and many others. These metrics will inform decisions at all levels along the biofuel supply chain and well beyond the scope of the individual decision.

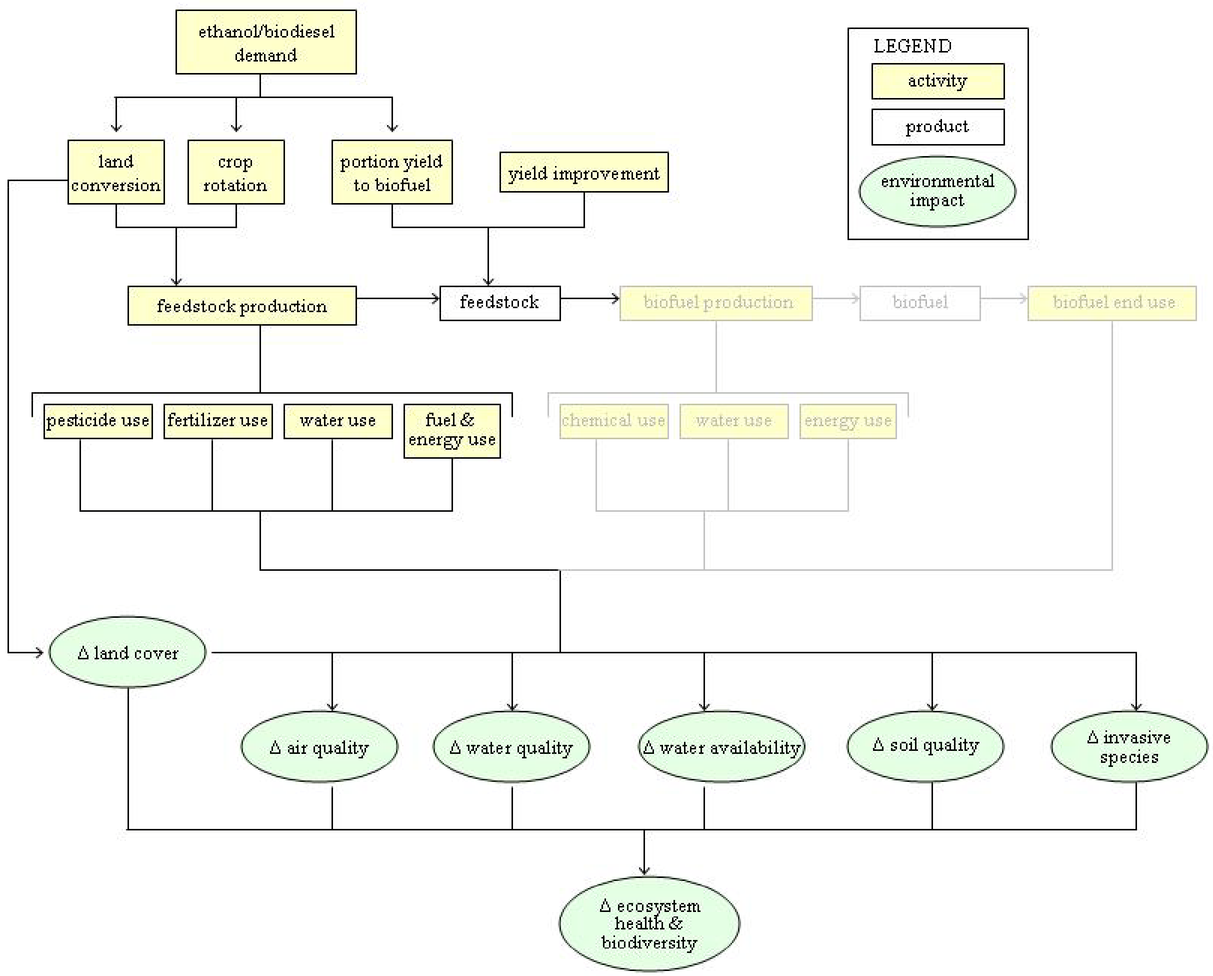
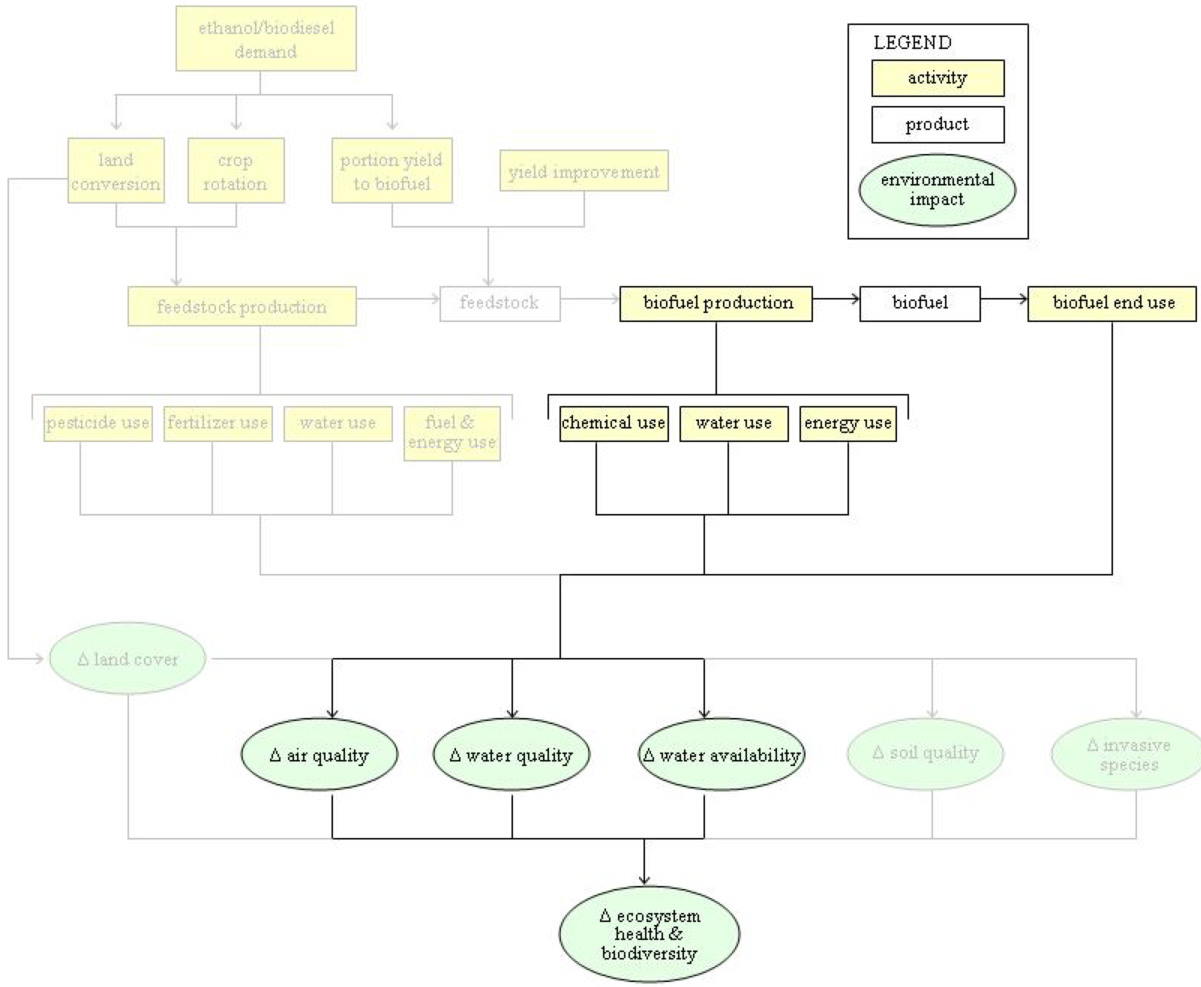


Figure 7-1: Conceptual Diagram of the Environmental Impacts of Biofuel Feedstock Production



120

121 **Figure 7-2: Conceptual Diagram of the Environmental Impacts of Biofuel**
122 **Production and Use**

7.2.6 Scenarios

EPA's 2013 Report to Congress will assess the environmental impacts of all five stages in the biofuel supply chain (Figure 2-3). One approach may be to create scenarios based on volumetric biofuel requirements for 2022 as presented in the RFS2 (see Table 2-1). For example:

- **Scenario A:** 2022 RFS2-projected feedstock mix produced with comprehensive conservation systems.

- **Scenario B:** 2022 RFS2-projected feedstock mix produced with existing levels of conservation practice implementation.

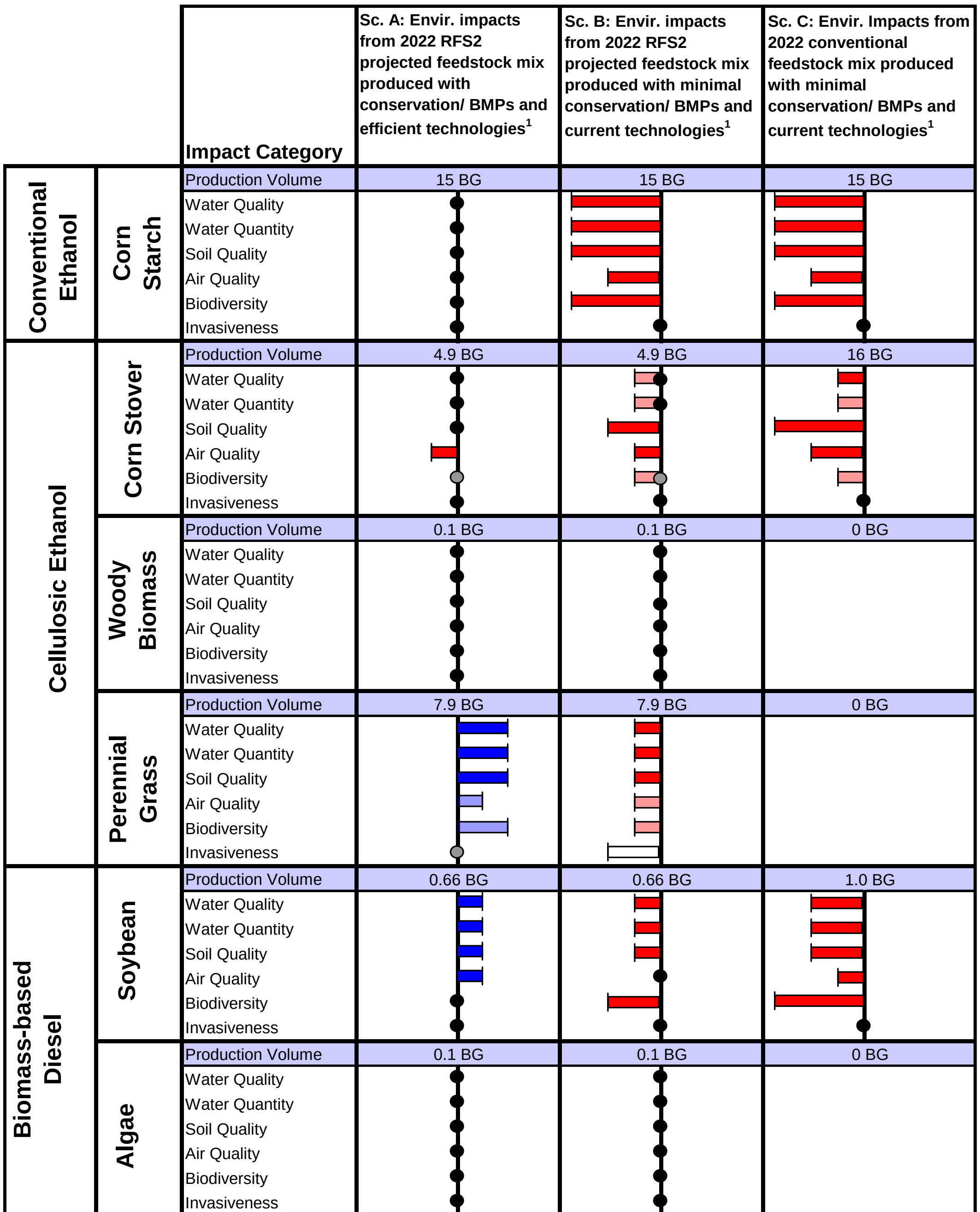
- **Scenario C:** 2022 conventional feedstock mix (corn starch, corn stover, and soybean) produced with existing levels of conservation practice implementation.

Figure 7-3 shows possible impacts in all six impact categories for these three scenarios based on the feedstocks and fuels discussed in this Report. Scenarios are for illustrative purposes only to show the potential rage of environmental impacts given assumptions about feedstock production locations and practices; fuel production, transport, storage, and use patterns and technologies; and target volumes (Appendix C, Table C-3). They do not necessarily represent the most likely future developments in biofuel production systems. The magnitude, direction, and certainty of bars (see figure legend) are based on expert interpretation of all available scientific, peer-reviewed literature as of July 2010. Bars are relative to one another and do not reflect a comparison with petroleum-based transportation fuel. Future versions of this Report to Congress will expand and update this assessment.

As noted earlier, the landscape of feedstock/biofuel production, conversion, and use is highly dynamic and constantly evolving. Which feedstocks and technologies are used and to what extent will be influenced by technological developments and market forces that are difficult to predict. Development of scenarios for future assessments will need to model or otherwise account for key factors that influence the biofuel market dynamics and associated environmental impacts. These factors include:

- *Regional considerations.* In general, biofuel conversion facilities will tend to be sited at reasonable distances from feedstock production areas, since cost considerations limit the distances over which biofuel feedstocks can be transported. Consequently, environmental impacts of both feedstock production and biofuel conversion will tend to be concentrated in particular regions.

- *Scale and volume of future commercial biofuel operations.* Future development and application of commercially viable biofuel technologies will change the nature of energy feedstocks and conversion processes in use, as well as the scale of their operation. While the continued use of corn starch for ethanol will likely not change, the future profile of feedstocks and biofuels could vary from those used in 2010, but which will actually be used and to what extent is highly uncertain.



[1] Bars represent total environmental impacts based on impacts from feedstock production on a per area basis; fuel production, distribution, storage and use on a per volume basis; total volume produced; and assumptions of each scenario fully described in Appendix C, Table C-3.

Legend

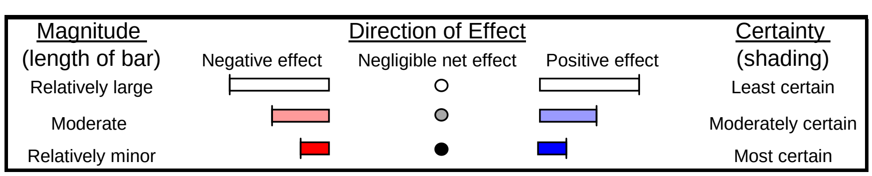


Figure 7-3: Cumulative Domestic Environmental Impacts of All Steps in the Biofuel Supply Chain System under Three Scenarios in 2022

- *Hybrid processes.* Biofuel conversion processes (e.g., biochemical and thermochemical processes) may evolve in the future to be hybrid processes that would produce not only biofuel but also synthetic chemicals and other industrial co-products. Integrated biorefineries may have the ability to make use of a biofuel-only or a hybrid conversion platform. Each new conversion option will present its own range of potential environmental impacts.

- *Changes in vehicle technologies.* Changes in vehicle technologies, patterns of vehicle sales, and fueling behavior will be needed to accommodate higher ethanol production volumes. Conversely, changes in vehicle technologies driven by other considerations, such as the development of plug-in hybrid electric or all-electric vehicles, could change the demand for liquid biofuels.

- *Changes in agricultural practices due to biofuel production and implications for environmental impacts.* Recent increases in ethanol production have expanded the market demand for corn grain, and farmers have responded to this increased demand by changing production practices from corn-soy rotations to corn-corn-soy or even continuous corn production. It is not clear what the effects of production shifts, agricultural residue use, and associated farm-level management practice changes will be in the short term.

7.2.7 Other Components

In addition to the above components, the 2013 assessment will include a several analyses that provide important perspective for understand and evaluating the impacts of biofuel production and use, as described below.

Comparison of Fossil Fuel to Biofuel. While this report provides a starting point for comparing the relative impacts associated with a range of different biofuel feedstock and production processes, it will also be useful to assess biofuel impacts in the larger context of the conventional petroleum fuels that are being displaced under the RFS2 mandates. Ideally, this comparison would cover the full life cycle for each fuel. Such an evaluation will facilitate comprehensive assessment of the relative costs and benefits of RFS2 beyond GHG impacts, and support identification of effective mitigation measures for key impacts. This type of evaluation has been recommended by the National Advisory Council for Environmental Policy and Technology as a means of conducting integrated environmental decision making (NACEPT 2008). Given the limitations of currently available information, a comparative assessment of petroleum fuel and biofuel impacts will be largely qualitative, with significant data gaps and uncertainties. Nevertheless, EPA anticipates that even a qualitative comparative analysis will be an important component of the 2013 assessment.

Net Energy Balance. Net energy balance (i.e., the amount of energy used to develop biofuels compared to the energy value derived from biofuels) is an important metric that will be addressed in the 2013 assessment. It enables comparison of biofuel produced from different feedstocks and via different conversion processes, as well as comparison between biofuel and gasoline. The net energy balance will include consideration of energy embedded in co-products of the fuel conversion process. For example, increases in corn ethanol production will increase the amount of co-products used in animal feed, which in turn displaces whole corn and soybean meal used for the same purpose; the "displaced" energy is credited to the ethanol system and offsets some of the energy required for production (Hammerschlag, 2006; Liska et al., 2008).

Market Impacts. Biofuels displace fossil energy resources, but also consume petroleum products, natural gas, electricity (much of which comes from nonrenewable energy sources), and even coal at different points along their supply chain. Consequently, changes in fossil fuel prices will impact the economics of biofuel production in unpredictable ways. The 2013 assessment will address market impacts and incorporate modeling of coupled energy systems and agricultural markets.

8. REFERENCES

Adegbidi, HG; Volk, TA; White, EH; Abrahamson, LP; Briggs, RD; Bickelhaupt, DH. 2001. Biomass and nutrient removal by willow clones in experimental bioenergy plantations in New York state. Biomass and Bioenergy 20(6): 399-411.

Advanced Biofuels Task Force. 2008. Advanced Biofuels Task Force report. Commonwealth of Massachusetts. Available at: http://www.mass.gov/Eoeea/docs/eea/biofuels/biofuels_complete.pdf.

Adviento-Borbe, MAA; Haddix, ML; Binder, DL; Walters, DT; Dobermann, A. 2007. Soil greenhouse gas fluxes and global warming potential in four high-yielding maize systems. Global Change Biology 13(9): 1972-1988.

Alexander, RB; Smith, RA; Schwarz, GE; Boyer, EW; Nolan, JV; Brakebill, JW. 2008. Differences in phosphorus and nitrogen delivery to the Gulf of Mexico from the Mississippi River Basin. Environmental Science and Technology 42(3): 822-830. Available at: http://pubs.acs.org/cgi-bin/abstract.cgi/esthag/2008/42/i03/abs/es0716103.html.

Alley, WM; Healy, RW; LaBaugh, JW; Reilly, TE. 2002. Flow and storage in ground water systems. Science 296: 1985-1990.

Alternative Fuels and Advanced Vehicles Data Center. 2010. E10 and other low-level ethanol blends. U.S. Department of Energy. Available at: http://www.afdc.energy.gov/afdc/ethanol/blends_e10.html.

Andow, DA; Zwalen, C. 2006. Assessing environmental risks of transgenic plants. Ecology Letters 9: 196-214.

Anex, R; Lifset, R. 2009. Post script to the corn ethanol debate: Reaching consensus. Journal of Industrial Ecology 13(6): 996-998.

Angermeier, PL; Karr, JR. 1984. Relationships between woody debris and fish habitat in a small warmwater stream. Transaction of the American Fisheries Society 113(6): 716-726.

Antizar-Ladislao, B; Turrion-Gomez, J. 2008. Second-generation biofuels and local bioenergy systems. Biofuels, Bioproducts and Biorefining 2: 455-469.

Aqua Terra. 2010. Phase III/IV modeling water quality impacts of corn production for ethanol in the upper Mississippi River basin. U.S. Environmental Protection Agency.

Aronsson, PG; Bergstrom, LF; Elowson, SNE. 2000. Long-term influence of intensively cultured short-rotation willow coppice on nitrogen concentrations in ground water. Journal of Environmental Management 58(2): 135-145.

ARS (United States Department of Agriculture, Agricultural Research Service). 2003. Agricultural phosphorus and eutrophication. Second edition. ARS-149. Available at: http://www.sera17.ext.vt.edu/Documents/AG_Phos_Eutro_2.pdf.

ARS (United States Department of Agriculture, Agricultural Research Service). 2006. Accomplishment report (2002–2006). National Program 207, Integrated Agricultural Systems. Available at: http://www.ars.usda.gov/research/programs/programs.htm?np_code=207&docid=17659.

ARS (United States Department of Agriculture, Agricultural Research Service). n.d. FY-2005 annual report Manure and Byproduct Utilization National Program 206. Available at: http://www.ars.usda.gov/research/programs/programs.htm?np_code=206&docid=13337.

Artuzi, JP; Contiero, RL. 2006. Herbicidas aplicados na soja e produtividade do milho em sucessão. Pesquisa Agropecuária Brasileira 41: 1119-1123.

Auer, C. 2008. Ecological risk assessment and regulation for genetically-modified ornamental plants. Critical Reviews in Plant Sciences 27: 255-271.

Aust, WM; Blinn, CR. 2004. Forestry Best Management Practices for timber harvesting and site preparation in the eastern United States: An overview of water quality and productivity research during the past 20 years (1982-2002). Water, Air, and Soil Pollution: Focus 4(1): 5-36.

Aust, WM; Lea, R; Gregory, JD. 1991. Removal of floodwater sediments by a clearcut tupelo-cypress wetland. Journal of the American Water Resources Association 27(1): 111-116.

Babcock, BA; McPhail, LL. 2009. Corn or soybean for 2009? Iowa Ag Review 15(1). Available at: http://www.card.iastate.edu/iowa_ag_review/winter_09/article4.aspx.

Baeumler, R; Zech, W. 1998. Soil solution chemistry and impact of forest thinning in mountain forests in the Bavarian Alps. Forest Ecology and Management 108(3): 231-238.

Baliga, R; Powers, S. 2010. Sustainable algae biodiesel production in cold climates. International Journal of Chemical Engineering. Available at: http://downloads.hindawi.com/journals/ijce/2010/102179.pdf.

Baker, A; Zahniser, S. 2006. Ethanol Reshapes the Corn Market. USDA ERS. Amber Waves. Available at: http://www.ers.usda.gov/AmberWaves/April06/Features/Ethanol.htm.

Barney, JN; DiTomaso, JM. 2008. Nonnative species and bioenergy: Are we cultivating the next invader? Bioscience 58(1): 1-7.

Barney, JN; Mann, JJ; Kyser, GB; Blumwald, E; Van Deynze, A; DiTomaso, JM. 2009. Tolerance of switchgrass to extreme soil moisture stress: Ecological implications. Plant Science 177: 724-732.

Bartolino, JR; Cunningham, WL. 2003. Ground-water depletion across the nation. USGS Fact Sheet 103-03. U.S. Geological Survey.

Bashkin, MA; Binkley, D. 1998. Changes in soil carbon following afforestation in Hawaii. Ecology 79(3): 828-833.

Beale, CV; Long, SP. 1997. Seasonal dynamics of nutrient accumulation and partitioning in the perennial C-4-grasses Miscanthus x giganteus and Spartina cynosuroides. Biomass and Bioenergy 12(6): 419-428.

Beale, CV; Morison, JIL; Long, SP. 1999. Water use efficiency of C4 perennial grasses in a temperate climate. Agricultural and Forest Meteorology 96: 103-115.

Beaty, ER; Engel, JL; Powell, JD. 1978. Tiller development and growth in switchgrass. Journal of Range Management 31(5): 361-365.

Bell, GP. 1997. Ecology and management of Arundo donax, and approaches to riparian habitat restoration in Southern California. In: Brock, JH; Wade, M; Pysek, P; Green, D (eds). Plant invasions: Studies from North America and Europe. Backhuys Publishers. pp. 103-113.

Bellamy, PE; Croxton, PJ; Heard, MS; Hinsley, SA; Hulmes, L; Hulmes, S; Nuttall, P; Pywell, RF; Rothery, P. 2009. The impact of growing Miscanthus for biomass on farmland bird populations. Biomass and Bioenergy 33: 191-199.

Berndes, G. 2002. Bioenergy and water—the implications of large-scale bioenergy production for water use and supply. Global Environmental Change 12: 253-271. Available at: http://www.unep.fr/energy/activities/water/pdf/Reading%20list%20materials_15April2010/Reading%20list%20materials_15April2010/Berndes%282002%29%20-%20Bioenergy%20and%20water_Largescale%20bioeenrgy%20production.pdf.

Binkley, D; Brown, TC. 1993. Forest practices as nonpoint sources of pollution in North America. Water Resources Bulletin 29(5): 729-740.

Blanco-Canqui, H; Lal, R. 2009a. Corn stover removal for expanded uses reduces soil fertility and structural stability. Soil Science Society of America Journal 73(2): 418-426.

Blanco-Canqui, H; Lal, R. 2009b. Crop residue removal impacts on soil productivity and environmental quality. Critical Reviews in Plant Sciences 28(3): 139-163.

Blanco-Canqui, H; Gantzer, CJ; Anderson, SH; Alberts, EE; Thompson, AL. 2004. Grass barrier and vegetative filter strip effectiveness in reducing runoff, sediment, nitrogen, and phosphorus loss. Soil Science Society of America Journal 68(5): 1670-1678.

Bracmort, K; Gorte RW. 2010. Biomass: Comparison of definitions in legislation. Congressional Research Service, R40529.

Brady, NC; Weil, RR. 2000. Elements of the nature and properties of soils. 12th edition. Prentice-Hall.

Bransby, DI; McLaughlin, SB; Parrish, DJ. 1998. A review of carbon and nitrogen balances in switchgrass grown for energy. Biomass and Bioenergy 14(4): 379-384.

Brinmer, TA; Gallivan, GJ; Stephenson, GR. 2005. Influence of herbicide-resistant canola on the environmental impact of week management. Pest Management Science 61:47-52.

Brooke, R; Fogel, G; Glaser, A; Griffin, E; Johnson, K. 2009. Corn ethanol and wildlife: How increases in corn plantings are affecting habitat and wildlife in the Prairie Pothole region. National Wildlife Federation. Available at: http://online.nwf.org/site/DocServer/NWF7419_corn_ethanol_web.pdf?docID=12841&JServSessionIdr004=2v1jxhesi3.app39a.

Brookes, G; Barfoot P. 2006. Global impact of biotech crops: Socio-economic and environmental effects, 1996-2006. AgBioForum 11(1): 21-38.

Brookes, G; Barfoot P. 2008. Global impact of biotech crops: Socio-economic and environmental effects in the first ten years of commercial use. AgBioForum 9(3): 139-151.

Brookes, G; Barfoot, P. 2009. Global impact of biotech crops: Income and production effects, 1996-2007. AgBioForum 12(2): 184-208.

Brookes, G; Barfoot, P. 2010. Global impact of biotech crops: Environmental effects, 1996-2008. AgBioForum 13(1): 76-94.

Brooks, JP; Adeli, A; Read, JJ; McLaughlin, MR. 2009. Rainfall simulation in greenhouse microcosms to assess bacterial-associated runoff from land-applied poultry litter. Journal of Environmental Quality 38(1): 218-229.

Bustamante, MMC; Melillo, J; Connor, DJ; Hardy, Y; Lambin, E; Lotze-Campen, H; Ravindranath, NH; Searchinger, T; Tschirley, J; Watson, H. 2009. What are the final land limits. In: Howarth, RW; Bringezu, S (eds.). Biofuels: Environmental consequences and interactions with changing land use. Proceedings of the Scientific Committee on Problems of the Environment (SCOPE) International Biofuels Project Rapid Assessment. pp. 271-291.

Campbell, JE; Lobell, DB; Genova, RC; Field, CB. 2008. The global potential of bioenergy on abandoned agriculture lands. Environmental Science and Technology 42: 5791-5794.

Cardinale, BJ; Srivastava, DS; Duffy, JE; Wright, JP; Downing, AL; Sankaran, M; Jouseau, C. 2006. Effects of biodiversity on the functioning of trophic groups and ecosystems. Nature 443(7114): 989-992.

Cardinale, BJ; Wright, JP; Cadotte, MW; Carroll, IT; Hector, A; Srivastava, DS; Loreau, M; Weis, JJ. 2007. Impacts of plant diversity on biomass production increase through time because of species complementarity. Proceedings of the National Academy of Sciences of the United States of America 104: 18123-18128.

Cassel, DK; Raczkowski, CW; Denton, HP. 1995. Tillage effects on corn production and soil physical conditions. Soil Science Society of America Journal 59(5): 1436-1443.

CENR (Committee on Environment and Natural Resources). 2010. Scientific assessment of hypoxia in U.S. coastal waters. Interagency Working Group on Harmful Algal Blooms, Hypoxia, and Human Health of the Joint Subcommittee on Ocean Science and Technology. Available at: http://www.whitehouse.gov/sites/default/files/microsites/ostp/hypoxia-report.pdf.

Chapotin, SM; Wolt, JD. 2007. Genetically modified crops for the bioeconomy: meeting public and regulatory expectations. Transgenic Research 16: 675-688.

Chen, W; Wei, X. 2008. Assessing the relations between aquatic habitat indicators and forest harvesting at watershed scale in the interior of British Columbia. Forest Ecology and Management 256(1-2): 152-160.

Chester, M; Martin, E. 2009. Cellulosic ethanol from municipal solid waste: A case study of the economic, energy, and greenhouse gas impacts in California. Environmental Science and Technology 43(14): 5183-5189.

Chester, M; Plevin, R; Rajagopal, D; Kammen, D. 2007. Biopower and waste conversion technologies for Santa Barbara County, California. Report for the Community Environmental Council. Available at: http://www.ce.berkeley.edu/~chester/library/santa_barba_biopower_potential.pdf.

Chiu, Y-W; Walseth, B; Suh, S. 2009. Water embodied in bioethanol in the United States. Environmental Science and Technology 43(8): 2688-2692. Available at: http://pubs.acs.org/doi/suppl/10.1021/es8031067/suppl_file/es8031067_si_001.pdf.

Chisti, Y. 2007. Biodiesel from microalgae. Biotechnology Advances 25: 294-306.

Christian, DP; Hoffman, W; Hanowski, JM; Niemi, GJ; Beyea, J. 1998. Bird and mammal diversity on woody biomass plantations in North America. Biomass and Bioenergy 14: 395-402.

Clarens, AF; Resurreccion, EP; White, MA; Colosi, LM. 2010. Environmental life cycle comparison of algae to other bioenergy feedstocks. Environmental Science and Technology 44(5): 1813-1819.

Clifton-Brown, JC; Lewandowski, I; Andersson, B; Basch, G; Christian, DG; Kjeldsen, JB; Jorgensen, U; Mortensen, JV; Riche, AB; Schwarz, K-U; Tayebi, K; Teixeira, F. 2001. Performance of 15 Miscanthus genotypes at five sites in Europe. Agronomy Journal 93: 1013-1019.

Clifton-Brown, JC; Breuer, J; Jones, MB. 2007. Carbon mitigation by the energy crop, Miscanthus. Global Change Biology 13(11): 2296-2307.

Cole, GA. 1983. Textbook of limnology. Third edition. Waveland Press.

Conner, AJ; Glare, TR; Nap, JP. 2003. The release of genetically modified crops into the environment. Part II. Overview of ecological risk assessment. The Plant Journal (33): 19-46.

Corseuil, HX; Hunt, CS; Ferreira dos Santos, RC; Alvarez, PJJ. 1998. The influence of the gasoline oxygenate ethanol on aerobic and anaerobic BTX degradation. Water Resources Research 33(7): 2056–2072.

Craig, W; Tepfer, M; Degrassi, G; Ripandelli, D. 2008. An overview of general features of risk assessments of genetically modified crops. Euphytica 164(3): 853-880.

Cram, DS; Baker, TT; Fernald, AG; Madrid, A; Rummer, B. 2007. Mechanical thinning impacts on runoff, infiltration, and sediment yield following fuel reduction treatments in southwestern dry mixed conifer forest. Journal of Soil and Water Conservation 62(5): 359-366.

Cruse, RM; Herndl, CG. 2009. Balancing corn stover harvest for biofuels with soil and water conservation. Journal of Soil and Water Conservation 64: 286-291.

Dale, VH; Kline, KL; Wiens, J; Fargione, J. 2010. Biofuels: Implications for land use and biodiversity. Ecological Society of America. Available at: http://esa.org/biofuelsreports/files/ESA%20Biofuels%20Report_VH%20Dale%20et%20al.pdf.

Danalatos, NG; Archontoulis, SV; Mitsios, I. 2007. Potential growth and biomass productivity of Miscanthus x giganteus as affected by plant density and N-fertilization in central Greece. Biomass and Bioenergy 31(2-3): 145-152.

Darnall, DW; Greene, B; Hosea, H; McPherson, RA; Henzl, M. 1986. Recovery of heavy metals by immobilized algae. In: Thompson, R. (ed). Trace metal removal from aqueous solution. Special Publication No. 61. Royal Society of Chemistry. pp. 1-24.

Da Silva, MLB; Alvarez, PJJ. 2002. Effects of ethanol versus MTBE on benzene, toluene, ethylbenzene, and xylene natural attenuation in aquifer columns. Journal of Environmental Engineering 128: 862-867.

Davidson, EA; Ackerman, IL. 1993. Changes in soil carbon inventories following cultivation of previously untilled soils. Biogeochemistry 20(3): 161-193.

Davis, JM. 2007. How to assess the risks of nanotechnology: Learning from past experience. Journal of Nanoscience and Nanotechnology 7: 402-409.

Davis, JM; Thomas, VM. 2006. Systematic approach to evaluating trade-offs among fuel options: The lessons of MTBE. Annals of the New York Academy of Sciences 1076: 498-515.

Davis, SC; Parton, WJ; Dohleman, FG; Smith, CM; Del Grosso, S; Kent, AD; Delucia, EH. 2010. Comparative biogeochemical cycles of bioenergy crops reveal nitrogen-fixation and low greenhouse gas emissions in a Miscanthus x giganteus agro-ecosystem. Ecosystems 13(1): 144-156.

Day, JD; Edwards, AP; Rodgers, GA. 1991. Development of an industrial-scale process for the heterotrophic production of a micro-algal mollusc feed. Bioresource Technology 38(2-3): 245-249.

Dhondt, AA; Wrege, PH; Cerretani, J; Sydenstricker, KV. 2007. Avian species richness and reproduction in short rotation coppice habitats in central and western New York. Bird Study 54: 12-22.

Dinnes, DL; Karlen, DL; Jaynes, DB; Kaspar, TC; Hatfield, JL; Colvin, TS; Cambardella, CA. 2002. Nitrogen management strategies to reduce nitrate leaching in tile-drained midwestern soils. Agronomy Journal 94(1): 153-171.

DiTomaso, JM; Reaser, JK.; Dionigi, CP; Doering, OC; Chilton, E; Schardt, JD; Barney, JN. 2010. Biofuel vs. bioinvasion: Seeding policy priorities. Environmental Science and Technology 44: 6906-6910.

Dominguez-Faus, R; Powers, S; Burken, J; Alvarez, P. 2009. The water footprint of biofuels: A drink or drive issue? Environmental Science and Technology 43(9): 3005-3010.

Donner, SD; Kucharik, CJ; Foley, JA. 2004. Impact of changing land use practices on nitrate export by the Mississippi River. Global Biogeochemical Cycles 18: GB1028.

Downing, M; Walsh, M; McLaughlin, S. 1995. Perennial grasses for energy and conservation: Evaluating some ecological, agricultural, and economic issues. Center for Agriculture, Food and Environment, Tufts University.

Drinkwater, LE; Wagoner, P; Sarrantonio, M. 1998. Legume-based cropping systems have reduced carbon and nitrogen losses. Nature 396(6708): 262-265.

Duncan, M; Lippiatt, BC; Haq, Z; Wang, M; Conway, RK. 2008. Metrics to support informed decision making for consumers of biobased products. USDA/OCE AIB 803.

Edeso, JM; Merino, A; Gonzalez, MJ; Marauri, P. 1999. Soil erosion under different harvesting managements in steep forestlands from northern Spain. Land Degradation and Development 10(1): 79-88.

EIA (Energy Information Administration). 2007. Methodology for allocating municipal solid waste to biogenic and non-biogenic energy. Available at: http://www.eia.doe.gov/cneaf/solar.renewables/page/mswaste/msw.pdf.

EIA (Energy Information Administration). 2009. Short-term energy outlook supplement: Biodiesel supply and consumption in the short-term energy outlook. Available at: http://www.eia.doe.gov/emeu/steo/pub/special/2009_sp_01.pdf.

EIA (Energy Information Administration). 2010. Biodiesel monthly report: 2009 edition. Available at: http://www.eia.doe.gov/cneaf/solar.renewables/page/biodiesel/biodiesel.html.

EIA (Energy Information Administration). n.d.[a]. Total biofuels consumption (thousand barrels per day). International energy statistics. Available at: http://tonto.eia.gov/cfapps/ipdbproject/IEDIndex3.cfm?tid=79&pid=79&aid=2.

EIA (Energy Information Administration). n.d.[b]. Biodiesel production (thousand barrels per day). International energy statistics. Available at: http://tonto.eia.doe.gov/cfapps/ipdbproject/iedindex3.cfm?tid=79&pid=81&aid=1&cid=&syid=2004&eyid=2008&unit=TBPD.

EIA (Energy Information Administration). n.d.[c]. Total biofuels production (thousand barrels per day). International energy statistics. Available at: http://tonto.eia.gov/cfapps/ipdbproject/IEDIndex3.cfm?tid=79&pid=79&aid=1.

EIA (Energy Information Administration). n.d.[d]. U.S. imports of fuel ethanol (thousands of barrels per day). Petroleum Navigator. Available at: http://www.eia.doe.gov/dnav/pet/hist/LeafHandler.ashx?n=PET&s=MFEIMUS1&f=A.

Ekvall, T. 2005. SETAC summaries. Journal of Cleaner Production 13: 1351-1358.

ERS (United States Department of Agriculture, Economic Research Service). 2008. 2008 farm bill side-by-side. Available at: http://www.ers.usda.gov/FarmBill/2008/Titles/TitleIIConservation.htm#conservation.

ERS (United States Department of Agriculture, Economic Research Service). 2010a. NEW feed grains data: Yearbook tables: U.S. exports of ethyl alcohol by selected destinations. Available at: http://www.ers.usda.gov/Data/feedgrains/Table.asp?t=32.

ERS (United States Department of Agriculture, Economic Research Service). 2010b. Feed grains data yearbook tables: Table 1. Corn, sorghum, barley, and oats: Planted acreage, harvested acreage, production, yield, and farm price. Available at: http://www.ers.usda.gov/data/feedgrains/Table.asp?t=01.

ERS (United States Department of Agriculture, Economic Research Service). 2010c. Feed grains data yearbook tables: Table 31. Corn: Food, seed, and industrial use. Available at: http://www.ers.usda.gov/data/feedgrains/Table.asp?t=31.

Evans, JM; Cohen, MJ. 2009. Regional water resource implications of bioethanol production in the southeastern United States. Global Change Biology 15: 2261-2273.

Fabiosa, J; Beghin, J; Fengxia, D; Elobeid, A; Tokgoz, S; Yu, T. 2010. Land allocation effects of the global ethanol surge: Predictions from the International FAPRI model. Land Economics 86(4): 687-706.

FAPRI (Food and Agricultural Policy Research Institute). 2008. U.S. and world agricultural outlook. Available at: http://www.fapri.iastate.edu/outlook/2008/.

FAPRI (Food and Agricultural Policy Research Institute). 2010a. U.S. and world agricultural outlook. Available at: http://www.fapri.iastate.edu/outlook/2010/.

FAPRI (Food and Agricultural Policy Research Institute). 2010b. U.S. crops. In: U.S. and world agricultural outlook. Available at: http://www.fapri.iastate.edu/outlook/2010/text/6US_Crops.pdf.

FAPRI (Food and Agricultural Policy Research Institute). 2010c. U.S. biofuel baseline briefing book: Projections for agricultural and biofuel markets. FAPRI-MU Report #04-10. Available at: http://www.fapri.missouri.edu/outreach/publications/2010/FAPRI_MU_Report_04_10.pdf.

FAPRI (Food and Agricultural Policy Research Institute). 2010d. World biofuels. In: U.S. and world agricultural outlook. Available at: http://www.fapri.iastate.edu/outlook/2010/text/15Biofuels.pdf.

Fargione, J; Hill, J; Tilman, D; Polasky, S; Hawthorne, P. 2008. Land clearing and the biofuel carbon debt. Science 319: 1235-1238.

Fargione, JE; Cooper, TR; Flaspohler, DJ; Hill, J; Lehman, C; McCoy, T; McLeod, S; Nelson, EJ; Oberhauser, KS; Tilman, D. 2009. Bioenergy and wildlife: Threats and opportunities for grassland conservation. BioScience 59(9): 767-777.

Farinelli, B; Carter, CA; Lin, C-YC; Sumner, DA. 2009. Import demand for Brazilian ethanol: A cross-country analysis. Journal of Cleaner Production 17: S9-S17.

Federer, CA; Hornbeck, JW; Tritton, LM; Martin, CW; Pierce, RS; Smith, CT (1989). Long-term depletion of calcium and other nutrients in Eastern US forests. Environmental Management 13(5):593-601.

Fenn, ME; Poth, MA; Aber, JD; Baron, JS; Bormann, BT; Johnson, DW; Lemly, AD; McNulty, SG; Ryan, DE; Stottlemyer, R. 1998. Nitrogen excess in North American ecosystems: Predisposing factors, ecosystem responses, and management strategies. Ecological Applications 8(3): 706-733.

Firbank, L. 2007. Assessing the ecological impacts of bioenergy projects. Bioenergy Research doi: 10.1007/s12155-007-9000-8. Available at: https://files.pbworks.com/download/HjG9dZBkTr/np-net/12639072/Firbank%20%282008%29%20Assessing%20ecological%20impacts%20bioenergy%20projects.pdf.

Flynn, KJ; Greenwell, HC; Lovitt, RW; Shields, RJ. 2010. Selection for fitness at the individual or population level: Modeling effects of genetic modification in microalgae on productivity and environmental safety. Journal of Theoretical Biology 263(3): 269-280.

Follett, RF; Varvel, GE; Kimble, JM; Vogel, KP. 2009. No-till corn after bromegrass: Effect on soil carbon and soil aggregates. Agronomy Journal 101(2): 261-268.

Foust, TD; Aden, A; Dutta, A; Phillips, S. 2009. An economic and environmental comparison of a biochemical and a thermochemical lignocellulosic ethanol conversion processes. Cellulose 16(4): 547-565.

Franzluebbers, AJ; Wright, SF; Stuedemann, JA. 2000. Soil aggregation and glomalin under pastures in the Southern Piedmont USA. Soil Science Society of America Journal 64(3): 1018-1026.

FSA (United States Department of Agriculture, Farm Service Agency). 2008. Conservation Reserve Program: Summary and enrollment statistics, FY 2007. Available at: http://www.fsa.usda.gov/Internet/FSA_File/annual_consv_2007.pdf.

FSA (United States Department of Agriculture, Farm Service Agency). 2009. Conservation Reserve Program: Summary and enrollment statistics, FY 2008. Available at: http://www.fsa.usda.gov/Internet/FSA_File/annualsummary2008.pdf.

Fulcrum Bioenergy. 2009. Fulcrum Bioenergy announces next generation ethanol breakthrough. News release. Available at: http://fulcrum-bioenergy.com/documents/TurningPointPlant09-01-09.pdf.

Gaffney JS; Marley NA. 2009. The impacts of combustion emissions on air quality and climate—from coal to biofuels and beyond. Atmospheric Environment 43: 23-36.

GAO (Government Accountability Office). 2007. Biofuels: DOE lacks a strategic approach to coordinate increasing production with infrastructure development and vehicle needs. GAO-07-713. Available at: http://www.gao.gov/new.items/d07713.pdf.

Garten, CT. 2002. Soil carbon storage beneath recently established tree plantations in Tennessee and South Carolina, USA. Biomass and Bioenergy 23(2): 93-102.

Garten, CT; Wullschleger, SD. 2000. Soil carbon dynamics beneath switchgrass as indicated by stable isotope analysis. Journal of Environmental Quality 29(2): 645-653.

Gerbens-Leenes, PW; Hoekstra, A; van der Meer, T. 2008. Water footprint of bio-energy and other primary energy carriers. Value of Water Research Report Series No. 29. UNESCO-IHE. Available at: http://www.waterfootprint.org/Reports/Report29-WaterFootprintBioenergy.pdf.

Ginnebaugh, D; Liang, J; Jacobson, M. 2010. Examining the temperature dependence of ethanol (E85) versus gasoline emissions on air pollution with a largely-explicit chemical mechanism. Atmospheric Environment 44: 1192-1199.

Gleason, RA; Euliss Jr., NH; Hubbard, DE; Duffy, WG. 2003. Effects of sediment load on emergence of aquatic invertebrates and plants from wetland soil egg and seed banks. Wetlands 23(1): 26-34.

Goldemberg, J; Teixeira Coelho, S; Guardabassi, P. 2008. The sustainability of ethanol production from sugarcane. Energy Policy 36: 2086-2097.

Goodlass, G; Green, M; Hilton, B; McDonough, S. 2007. Nitrate leaching from short-rotation coppice. Soil Use and Management 23(2): 178-184.

Goolsby, DA; Battaglin, WA; Lawrence, GB; Artz, RS; Aulenbach, BT; Hooper, RP; Keeney, DR; Stensland, GJ. 1999. Flux and sources of nutrients in the Mississippi-Atchafalaya River Basin: Topic 3 report for the integrated assessment of hypoxia in the Gulf of Mexico. Decision Analysis Series No. 17. National Oceanic and Atmospheric Administration.

Gorte, RW. 2009. Wildfire fuels and fuel reduction. CRS Report R40811. Available at: http://ncseonline.org/NLE/CRSreports/09Sept/R40811.pdf.

Grady, KC; Hart, SC. 2006. Influences of thinning, prescribed burning, and wildfire on soil processes and properties in southwestern ponderosa pine forests: A retrospective study. Forest Ecology and Management 234(1-3): 123-135.

Graham, RL; Nelson, R; Sheehan, J; Perlack, RD; Wright, LL. 2007. Current and potential US corn stover supplies. Agronomy Journal 99(1): 1-11.

Graham, LA; Belisle, SL; Baas, C-L. 2008. Emissions from light duty gasoline vehicles operating on low blend ethanol gasoline and E85. Atmospheric Environment 42: 4498-4516.

Gressel, J. 2008. Transgenics are imperative for biofuel crops. Plant Science 174: 246-263.

Grigal, DF; Berguson, WE. 1998. Soil carbon changes associated with short-rotation systems. Biomass and Bioenergy 14(4): 371-377.

Groom, MJ; Gray, EM; Townsend, PA. 2008. Biofuels and biodiversity: Principles for creating better policies for biofuel production. Conservation Biology 22: 602-609.

Guan, TY; Holley, RA. 2003. Pathogen survival in swine manure environments and transmission of human enteric illness: A review. Journal of Environmental Quality 32(2): 383-392.

Hammerschlag, R. 2006. Ethanol's energy return on investment: A survey of the literature 1990-present. Environmental Science and Technology 40(6): 1744-1750.

Hansen, EA. 1988. Irrigating short rotation intensive culture hybrid poplars. Biomass 16: 237-250.

Heaton, E; Voigt, T; Long, SP. 2004. A quantitative review comparing the yields of two candidate C4 perennial biomass crops in relation to nitrogen, temperature and water. Biomass and Bioenergy 27: 21-30.

Heaton, EA; Dohleman, FG; Long, SP. 2008. Meeting US biofuel goals with less land: The potential of Miscanthus. Global Change Biology 14(9): 2000-2014.

Hector, A; Bagchi, R. 2007. Biodiversity and ecosystem multifunctionality. Nature 448: 188-U6.

Helou, AE; Tran, K; Buncio, C. 2010. Energy recovery from municipal solid waste in California: Needs and challenges. Proceedings of the 18th Annual North American Waste-to-Energy Conference. NAWTEC18-3568.

Hertel, TW; Tyner, WE; Birur, DK. 2010. The global impacts of biofuel mandates. The Energy Journal 31(1): 75-100.

Hess, P; Johnston, M; Brown-Steiner, B; Holloway, T; de Andrade, J; Artaxo, P. 2009. Chapter 10: Air quality issues associated with biofuel production and use. In: Howarth, RW; Bringezu, S (eds.). Biofuels: Environmental consequences and interactions with changing land use. Proceedings of the Scientific Committee on Problems of the Environment (SCOPE) International Biofuels Project Rapid Assessment. pp. 169-194. Available at: http://cip.cornell.edu/scope/1245782010.

Hill, J. 2007. Environmental costs and benefits of transportation biofuel production from food- and lignocelluloses-based energy crops, a review. Agronomy for Sustainable Development 27(1): 1-12.

Hill, J. 2009. Environmental costs and benefits of transportation biofuel production from food- and lignocellulose-based energy crops: A review. In: Lichtfouse, E; Navarrete, M; Debaeke, P; Souchère, V; Alberola, C (eds.). Sustainable Agriculture. Springer.

Hill, J; Nelson, E; Tilman, D; Polasky, S; Tiffany, D. 2006. Environmental, economic, and energetic costs and benefits of biodiesel and ethanol biofuels. Proceedings of the National Academy of Sciences 103(30): 11206-11210.

Hill, J; Polasky, S; Nelson, E; Tilman, D; Huo, H; Ludwig, L; Neumann, J; Zheng, H; Bonta, D. 2009. Climate change and health costs of air emissions from biofuels and gasoline. Proceedings of the National Academy of Sciences 106(6): 2077-2082. Available at: http://www.pnas.org/content/early/2009/02/02/0812835106.abstract.

Hoffmann, JP. 1998. Wastewater treatment with suspended and nonsuspended algae. Journal of Phycology 34(5): 757-763.

Hong, H; Wang, MYW. 2009. Total versus urban: Well-to-wheels-assessment of criteria pollutant emissions from various vehicle/fuel systems. Atmospheric Environment 43: 1796-1804.

Hooper, DU; Chapin, FS; Ewel, JJ; Hector, A; Inchausti, P; Lavorel, S; Lawton, JH; Lodge, DM; Loreau, M; Naeem, S; Schmid, B; Setälä, H; Symstad, AJ; Vandermeer, J; Wardle, DA. 2005. Effects of biodiversity on ecosystem functioning: A consensus of current knowledge. Ecological Monographs 75(1): 3-35.

Hunt, S; Stair, P. 2006. Biofuels hit a gusher. In: Vital Signs 2006-2007. Worldwatch Institute. pp. 40-41.

Huntington, TG. 1995. Carbon sequestration in an aggrading forest ecosystem in the southeastern USA. Soil Science Society of America Journal 59(5): 1459-1467.

Huo, H; Wang, M; Bloyd, C; Putsche, V. 2008. Life-cycle assessment of energy and greenhouse gas effects of soybean-derived biodiesel and renewable fuels. Available at: http://www.transportation.anl.gov/pdfs/AF/467.pdf.

Huo, H; Wang, M; Bloyd, C; Putsche, V. 2009. Life-cycle assessment of energy use and greenhouse gas emission of soybean-derived biodiesel and renewable fuels. Environmental Science and Technology 43: 750-756.

Iowa State University. 2008. General guide for crop nutrient and limestone recommendations in Iowa. Iowa State University Extension Publication PM 1688.

Iowa State University. 2009. Estimated costs of crop production in Iowa—2010. Iowa State University Extension Publication FM 1712. Available at: http://www.extension.iastate.edu/Publications/fm1712.pdf.

ISO (International Organization for Standardization). 2006. ISO 14040:2006: Environmental management—life cycle assessment—principles and framework.

Jackson, RB; Jobbagy, EG; Avissar, R; Roy, SB; Barrett, DJ; Cook, CW; Farley, KA; le Maitre, DC; McCarl, BA; Murray, BC. 2005. Trading water for carbon with biological carbon sequestration. Science 310: 1944-1947.

Jakob, K; Zhou, FS; Paterson, AH. 2009. Genetic improvement of C4 grasses as cellulosic biofuel feedstocks. In Vitro Cellular and Developmental Biology–Plant 45(3): 291-305.

Jandl, R; Lindner, M; Vesterdal, L; Bauwens, B; Baritz, R; Hagedorn, F; Johnson, DW; Minkkinen, K; Byrne, KA. 2007. How strongly can forest management influence soil carbon sequestration? Geoderma 137(3-4): 253-268.

Johnson, DW; Curtis, PS. 2001. Effects of forest management on soil C and N storage: Meta analysis. Forest Ecology and Management 140(2-3): 227-238.

Johnson, JMF; Allmaras, RR; Reicosky, DC. 2006. Estimating source carbon from crop residues, roots and rhizodeposits using the national grain-yield database. Agronomy Journal 98(3): 622-636.

Kahn, N; Warith, MA; Luk, G. 2007. A comparison of acute toxicity of biodiesel, biodiesel blends, and diesel on aquatic organisms. Journal of the Air and Waste Management Association 57(3): 286-296.

Kalogo, Y; Habibi, S; Maclean, HL; Joshi, SV. 2007. Environmental implications of municipal solid waste-derived ethanol. Environmental Science and Technology 41(1): 35-41.

Kantor, LS; Lipton, K; Manchester, A; Oliveira, V. 1997. Estimating and addressing America's food losses. Food Review January-April 2-12. Available at: http://www.ers.usda.gov/Publications/FoodReview/Jan1997/Jan97a.pdf.

Kanwar, RS; Colvin, TS; Karlen, DL. 1997. Ridge, moldboard, chisel, and no-till effects on tile water quality beneath two cropping systems. Journal of Production Agriculture 10: 227-234.

Karlen, DL; Lal, R; Follett, RF; Kimble, JM; Hatfield, JL; Miranowski, JM; Cambardella, CA; Manale, A; Anex, RP; Rice, CW. 2009. Crop residues: The rest of the story. Environmental Science and Technology 43(21): 8011-8015.

Kausch, AP; Hague, J; Oliver, M; Watrud, L; Mallory-Smith, C; Meier, V; Snow, A; Stewart, N. 2010a. Gene flow in genetically engineered perennial grasses: Lessons for modification of dedicated bioenergy crops. In: Mascia, PN; et al. (eds). Plant biotechnology for sustainable production of energy and co-products. Biotechnology in Agriculture and Forestry 66(3): 285-297.

Kausch, AP; Hague, J; Oliver, M; Li, Y; Daniell, H; Mascia, P; Watrud, LS; Stewart Jr., CN. 2010b. Transgenic perennial biofuel feedstocks and strategies for bioconfinement. Biofuels 1(1): 163-176.

Keeney, R; Hertel, TW. 2009. The indirect land use impacts of United States biofuel policies: The importance of acreage, yield, and bilateral trade responses. American Journal of Agricultural Economics 91(4): 895-909.

Kenny, JF; Barber, NL; Hutson, SS; Linsey, KS; Lovelace, JK; Maupin, MA. 2009. Estimated use of water in the United States in 2005. Circular 1344. U.S. Geological Survey.

Keoleian, GA; Volk, TA. 2005. Renewable energy from willow biomass crops: Life cycle energy, environmental and economic performance. Critical Reviews in Plant Sciences 24(5-6): 385-406.

Keshwani, D; Cheng, J. 2009. Switchgrass for bioethanol and other value-added applications: A review. Bioresource Technology 100(4): 1515-1523.

Kim, S; Dale, BE. 2005. Life cycle assessment of various cropping systems utilized for producing biofuels: Bioethanol and biodiesel. Biomass and Bioenergy 29(6): 426-439.

Kimble, J. n.d. Biofuels and emerging issues for emergency responders. U.S. Environmental Protection Agency. Available at: http://www.epa.gov/oem/docs/oil/fss/fss09/kimblebiofuels.pdf.

Kleter, GA; Bhula, R; Bodnaruk, K; Carazo, E; Felsot, AS; Harris, CA; Katayama, A; Kuiper, HA; Racke, KD; Rubin, B; Shevah, Y; Stephenson, GR; Tanaka, K; Unsworth, J; Wauchope, RD; Wong, SS. 2008. Altered pesticide use on transgenic crops and the associated general impact from an environmental perspective. Pest Management Science 63(11): 1107-1115.

Klocke, NL; Watts, DG; Schneekloth, JP; Davidson, DR; Todd, RW; Parkhurst, AM. 1999. Nitrate leaching in irrigated corn and soybean in a semi-arid climate. Transactions of the American Society of Agricultural Engineers 42(6): 1621-1630.

Knight, DH; Yavitt, JB; Joyce, GD. 1991. Water and nitrogen outflow from lodgepole pine forest after 2 levels of tree mortality. Forest Ecology and Management 46(3-4): 215-225.

Knox, OGG; Constable, GA; Pyke, B; Gupta, VVSR. 2006. Environmental impact of conventional and Bt insecticidal cotton expressing one and two Cry genes in Australia. Australian Journal of Agricultural Research 57(5): 501-509.

Koh, LP; Ghazoul, J. 2008. Biofuels, biodiversity, and people: Understanding the conflicts and finding opportunities. Biological Conservation 141: 2450-2460.

Kreutzweiser, DP; Hazlett, PW; Gunn, JM. 2008. Logging impacts on the biogeochemistry of boreal forest soils and nutrient export to aquatic systems: A review. Environmental Reviews 16: 157-179.

Lal, R. 1997. Residue management, conservation tillage and soil restoration for mitigating greenhouse effect by CO2-enrichment. Soil & Tillage Research 43(1-2): 81-107.

Lal, R. 2003. Global potential of soil carbon sequestration to mitigate the greenhouse effect. Critical Reviews in Plant Sciences 22(2): 151-184.

Lal, R. 2004. Soil carbon sequestration impacts on global climate change and food security. Science 304(5677): 1623-1627.

Lal, R. 2009. Soil quality impacts of residue removal for bioethanol production. Soil and Tillage Research 102(2): 233-241.

Landis, DA; Gardiner, MM; van der Werf, W; Swinton, SM. 2008. Increasing corn for biofuel production reduces biocontrol services in agricultural landscapes. Proceedings of the National Academy of Sciences 105: 20552-20557. Available at: http://www.pnas.org/content/105/51/20552.full.pdf+html.

Lapola, D; Schaldach, R; Alcamo, J; Bondeau, A; Koch, J; Koelking, C; Priess, J. 2010. Indirect land-use changes can overcome carbon savings from biofuels in Brazil. Proceedings of the National Academy of Sciences of the United States of America. Available at: http://www.pnas.org/content/early/2010/02/02/0907318107.full.pdf+html.

Lapuerta, M; Armas, O; Rodriguez-Fernandez, J. 2008. Effect of biodiesel fuels on diesel engine emissions. Progress in Energy and Combustion Science 34(2): 198-223.

Lavergne, S; Molofsky, J. 2004. Reed canary grass (Phalaris arundinacea) as a biological model in the study of plant invasions. Critical Reviews in Plant Sciences 23(5): 415-429.

Lee, D; Nair, R; Chen, A. 2009. Regulatory hurdles for transgenic biofuel crops. Biofuels Bioprod Refin 3:468-480.

Lee, LS; Carmosini, N; Sassman, SA; Dion, HM; Sepulveda, MS. 2007a. Agricultural contributions of antimicrobials and hormones on soil and water quality. Advances in Agronomy, vol. 93. Elsevier Academic Press Inc. pp. 1-68.

Lee, DK; Owens, VN; Doolittle, JJ. 2007b. Switchgrass and soil carbon sequestration response to ammonium nitrate, manure, and harvest frequency on conservation reserve program land. Agronomy Journal 99(2): 462-468.

Lee, H; Sumner, DA. 2010. International trade patterns and policy for ethanol in the United States. Handbook of Bioenergy Policy and Economics: Natural Resource Management and Policy 33(5): 327-345.

Levin, RB; Epstein, PR; Ford, TE; Harrington, W; Olson, E; Reichard, EG. 2002. U.S. drinking water challenges in the twenty-first century. Environmental Health Perspectives 110(1): 43-52.

Lewandowski, I; Clifton-Brown, JC; Scurlock, JMO; Huisman, W. 2000. Miscanthus: European experience with a novel energy crop. Biomass and Bioenergy 19: 209-227.

Lewandowski, I; Clifton-Brown, JC; Andersson, B; Basch, G; Christian, DG; Jorgensen, U; Jones, MB; Riche, AB; Schwarz, KU; Tayebi, K; Teixeira, F. 2003. Environment and harvest time affects the combustion qualities of Miscanthus genotypes. Agronomy Journal 95: 1274-1280.

Licht, FO. 2007. World—biodiesel production (tonnes). F.O. Licht’s World Ethanol and Biofuels Report 5(14): 291.

Liska, AJ, Yang, HS; Bremer, VR; Klopfenstein, TJ; Walters, DT; Erickson, GE; Cassman, KG. 2008. Improvements in life cycle energy efficiency and greenhouse gas emissions of corn-ethanol. Journal of Industrial Ecology 13(1): 58-74.

Loreau, M; Naeem, S; Inchausti, P (eds.). 2002. Biodiversity and ecosystem functioning: Synthesis and perspectives. Oxford University Press.

Luiro, J; Kukkola, M; Saarsalmi, A; Tamminen, P; Helmisaari, HS. 2010. Logging residue removal after thinning in boreal forests: Long-term impact on the nutrient status of Norway spruce and Scots pine needles. Tree Physiology 30(1): 78-88.

Lundborg, A. 1997. Reducing nitrogen load: Whole tree harvesting, a literature review. Ambio 26(6): 387-393.

MacDonald, JM; Ribaudo, MO; Livingston, MJ; Beckman, J; Huang, W. 2009. Manure use for fertilizer and energy: Report to congress. Administrative Publication No. AP-037. U.S. Department of Agriculture, Economic Research Service.

Mackay, DM; de Sieyes, NR; Einarson, MD; Feris, KP; Pappas, AA; Wood, IA; Jacobson, L; Justice, LG; Noske, MN; Scow, KM; Wilson, JT. 2006. Impact of ethanol on the natural attenuation of benzene, toluene, and o-xylene in a normally sulfate-reducing aquifer. Environmental Science and Technology 40: 6123-6130.

Malcolm, SA; Aillery, M; Weinberg, M. 2009. Ethanol and a changing agricultural landscape. Economic Research Report 86. U.S. Department of Agriculture, Economic Research Service.

Malik, YS; Randall, GW; Goyal, SM. 2004. Fate of salmonella following application of swine manure to tile-drained clay loam soil. Journal of Water and Health 2(2): 97-101.

McBroom, MW; Beasley, RS; Chang, MT; Ice, GG. 2008. Storm runoff and sediment losses from forest clearcutting and stand re-establishment with best management practices in East Texas, USA. Hydrological Processes 22(10): 1509-1522.

McLaughlin, SB; Kszos, LA. 2005. Development of switchgrass (Panicum virgatum) as a bioenergy feedstock in the United States. Biomass and Bioenergy 28(6): 515-535.

McLaughlin, JW; Phillips, SA. 2006. Soil carbon, nitrogen, and base cation cycling 17 years after whole-tree harvesting in a low-elevation red spruce (Picea rubens)–balsam fir (Abies balsainea) forested watershed in central Maine, USA. Forest Ecology and Management 222(1-3): 234-253.

McLaughlin, SB; Walsh, ME. 1998. Evaluating environmental consequences of producing herbaceous crops for bioenergy. Biomass and Bioenergy 14(4): 317-324.

Mercer, KL; Wainwright, JD. 2008. Gene flow from transgenic maize to landraces in Mexico: An analysis. Agriculture, Ecosystems and Environment 123(2008): 109-115.

Milbrandt, A. 2005. A geographic perspective on the current biomass resource availability in the United States. Technical Report NREL/TP-560-39181. U.S. Department of Energy.

Miller, EL; Beasley, RS, Lawson, ER. 1988. Forest harvest and site preparation effects on erosion and sedimentation in the Ouachita Mountains. Journal of Environmental Quality 17: 219-225.

Minnesota Department of Agriculture. 2010. Conservation practices: Minnesota conservation funding guide. Available at: http://www.mda.state.mn.us/protecting/conservation/practices/nutrientmgmt.aspx.

Missouri Department of Natural Resources. 2008. Compliance assistance for biodiesel production plants. Natural Resources Fact Sheet. PUB002230. Available at: http://www.dnr.mo.gov/pubs/pub2230.pdf.

Mitchell, D. 2008. A note on rising food prices. Policy Research Working Paper 4682. The World Bank.

mongabay.com. n.d.[a]. Indonesia. Available at: http://rainforests.mongabay.com/deforestation/2000/Indonesia.htm.

mongabay.com. n.d.[b]. Malaysia. Available at: http://rainforests.mongabay.com/deforestation/2000/Malaysia.htm.

Moore, PA; Daniel, TC; Sharpley, AN; Wood, CW. 1995. Poultry manure management—environmentally sound options. Journal of Soil and Water Conservation 50(3): 321-327.

Muhs, J; Viamajala, S; Heydorn, B; Edwards, M; Hu, Q; Hobbs, R; Allen, M; Smith, DB; Fenk, T; Bayless, D; Cooksey, K; Kuritz, T; Crocker, M; Morton, S; Sears, J; Daggett, D; Hazlebeck, D; Hassenia, J. 2009. Algae biofuels and carbon recycling: A summary of opportunities, challenges, and research needs. Utah State University. Available at: www.utah.gov/ustar/documents/63.pdf.

Munoz, R., and B. Guieysse. 2006. Algal-bacterial processes for the treatment of hazardous contaminants: A review. Water Research 40: 2799-2815.

Murray, LD; Best, LB. 2003. Short-term bird response to harvesting switchgrass for biomass in Iowa. Journal of Wildlife Management 67: 611-621.

Murray, LD; Best, LB; Jacobsen, TJ; Braster, ML. 2003. Potential effects on grassland birds of converting marginal cropland to switchgrass biomass production. Biomass and Bioenergy 25: 167-175.

NASS (United States Department of Agriculture, National Agricultural Statistics Service). 2006. Agricultural chemical usage 2005 field crops summary. Ag Ch 1 (06). Available at: http://usda.mannlib.cornell.edu/usda/nass/AgriChemUsFC//2000s/2006/AgriChemUsFC-05-17-2006.pdf.

NASS (United States Department of Agriculture, National Agricultural Statistics Service). 2007a. Prospective plantings. March 30, 2007. Available at: http://usda.mannlib.cornell.edu/usda/nass/ProsPlan//2000s/2007/ProsPlan-03-30-2007.pdf.

NASS (United States Department of Agriculture, National Agricultural Statistics Service). 2007b. Agricultural chemical usage 2006 field crops summary. Ag Ch 1 (07)a. Available at: http://usda.mannlib.cornell.edu/usda/nass/AgriChemUsFC//2000s/2007/AgriChemUsFC-05-16-2007_revision.pdf.

NASS (U.S. Department of Agriculture, National Agricultural Statistics Service). 2009. 2007 census of agriculture. Available at: http://www.agcensus.usda.gov.

NASS (United States Department of Agriculture, National Agricultural Statistics Service). 2010a. Quick stats. Available at: http://www.nass.usda.gov/Data_and_Statistics/Quick_Stats.

NASS (United States Department of Agriculture, National Agricultural Statistics Service). 2010b. County maps. Available at: http://www.nass.usda.gov/Charts_and_Maps/Crops_County/index.asp.

NASS (United States Department of Agriculture, National Agricultural Statistics Service). 2010c. National statistics for corn—corn grain yield. Available at: http://www.nass.usda.gov/Statistics_by_Subject/result.php?AFBDFE1E-1AFC-35DE-8A93-7FB72F0DA089&sector=CROPS&group=FIELD%20CROPS&comm=CORN.

Nebraska Department of Environmental Quality. 2009. Waste tips for biodiesel fuel plants. Available at: http://www.deq.state.ne.us/Publica.nsf/a9f87abbcc29fa1f8625687700625436/a85102870a5b10bf862575d90051c55f?OpenDocument.

Neary, DG; Ice, GG; Jackson, CR. 2009. Linkages between forest soils and water quality and quantity. Forest Ecology and Management 258(10): 2269-2281.

Nelson, RG; Ascough, JC; Langemeier, MR. 2006. Environmental and economic analysis of switchgrass production for water quality improvement in northeast Kansas. Journal of Environmental Management 79(4): 336-348.

Nickson, T. 2008. Planning environmental risk assessment for genetically modified crops: Problem formulation for stress-tolerant crops. Plant Physiology 147: 494-502.

NRC (National Research Council). 2000. Genetically modified pest-protected plants: Science and regulation. National Academies Press.

NRC (National Research Council). 2001. Ecological monitoring of genetically modified crops. A workshop summary. National Academies Press.

NRC (National Research Council). 2002. Environmental effects of transgenic plants: The scope and adequacy of regulation. National Academies Press.

NRC (National Research Council) 2008. Water implications of biofuels production in the United States. National Academies Press. Available at: http://www.nap.edu/catalog/12039.html.

NRC (National Research Council). 2010. The impact of genetically engineered crops on farm sustainability in the United States. National Academies Press.

NRCS (United States Department of Agriculture, National Resources Conservation Service). 2009. NI_190_304, comprehensive nutrient management plan technical criteria. Available at: http://directives.sc.egov.usda.gov/viewerFS.aspx?hid=25686.

NRCS (United States Department of Agriculture, National Resources Conservation Service). 2010. Assessment of the effects of conservation practices on cultivated cropland in the Upper Mississippi River Basin. Available at: http://www.nrcs.usda.gov/technical/NRI/ceap/umrb/index.html.

NRDC (Natural Resources Defense Council). 2009. Cultivating clean energy: The promise of algae biofuels.

Nyakatawa, EZ; Mays, DA; Tolbert, VR; Green, TH; Bingham, L. 2006. Runoff, sediment, nitrogen, and phosphorus losses from agricultural land converted to sweetgum and switchgrass bioenergy feedstock production in north Alabama. Biomass and Bioenergy 30(7): 655-664.

Ochoa-Acuna, HG; Bialkowski, W; Yale, G; Hahn, L. 2009. Toxicity of soybean rust fungicides to freshwater algae and Daphnia magna. Ecotoxicology 18(4): 440-446.

OSTP (Office of Science and Technology Policy). 1986. Coordinated framework for regulation of biotechnology. Federal Register 51: 23302.

Owen, MDK. 2005. Maize and soybeans—controllable volunteerism without ferality? In: Gressel, J (ed.). Crop ferality and volunteerism. CRC Press. pp. 149-165.

Oyediran, IO; Hibbard, BE; Clark, TL. 2004. Prairie grasses as hosts of the western corn rootworm (Coleoptera: Chrysomelidae). Environmental Entomology 33: 740-747.

Oyediran, IO; Hibbard, BE; Clark, TL. 2005. Western corn rootworm (Coleoptera: Chrysomelidae) beetle emergence from weedy Cry3Bb1 rootworm-resistant transgenic corn. Journal of Economic Entomology 98(5): 1679-1684.

Paerl, HW; Robin, D; Whitall, DR. 2002. Atmospheric deposition of nitrogen: Implications for nutrient over-enrichment of coastal waters. Estuaries 25(4B): 677-693.

Parrish, DJ; Fike, JH. 2005. The biology and agronomy of switchgrass for biofuels. Critical Reviews in Plant Sciences 24: 423-459.

Pate, R, Hightower, M; Cameron, C; Einfeld, W. 2007. Overview of energy-water interdependencies and the emerging energy demands on water resources. Report SAND 2007-1349C. Sandia National Laboratories.

Paul, KI; Polglase, PJ; Nyakuengama, JG; Khanna, PK. 2002. Change in soil carbon following afforestation. Forest Ecology and Management 168(1-3): 241-257.

Perkins, JH. 2009. Integrated pest management, biofuels, and a new green revolution: A case study of the American Midwest. In: Peshin, R; Dhawan, AK (eds.). Integrated pest management: Dissemination and impact. Springer Netherlands. pp. 581-607.

Perlack, R; Wright, L; Turhollow, A; Graham, R; Stokes, B; Erbach, D. 2005. Biomass as feedstock for a bioenergy and bioproducts industry: The technical feasibility of a billion-ton annual supply. U.S. Department of Agriculture and U.S. Department of Energy. Oak Ridge National Laboratory. ORNL/TM-2005/66. Available at: http://feedstockreview.ornl.gov/pdf/billion_ton_vision.pdf.

Perry, CH; Brooks, KN; Grigal, DF; Isebrands, JG; Tolbert, VR. 1998. A comparison of nutrient export from short-rotation hybrid poplar plantations and natural forest stands. Oak Ridge National Laboratory.

Perry, CH; Miller, RC; Brooks, KN. 2001. Impacts of short-rotation hybrid poplar plantations on regional water yields. Forest Ecology and Management 143: 143-151.

Perttu, KL. 1995. Ecological, biological balances and conservation. Biomass and Bioenergy 9: 107-116.

Phillips, MJ; Swift Jr., LW; Blinn, CR. 2000. Best management practices for riparian areas. In: Verry, ES; Hornbeck, JW; Dolloff, CA (eds). Riparian management in forests of the continental Eastern United States. CRC Press. pp. 273-286.

Pimentel, D. 2006. Soil erosion: A food and environmental threat. Environment, Development and Sustainability 8(1): 119-137.

Pimentel, D; Patzek, TW. 2005. Ethanol production using corn, switchgrass, and wood; biodiesel production using soybean and sunflower. Natural Resources Research 14(1): 65-76.

Pimentel, D; Pimentel, M. 2008. Food, energy, and society. CRC Press.

Pimentel, D; Marklein, A; Toth, MA; Karpoff, MN; Paul, GS; McCormack, R; Kyriazis, J; Krueger, T. 2009. Food versus biofuels: Environmental and economic costs. Human Ecology 37:1-12.

Poff, NL; Zimmerman, JKH. 2010. Ecological responses to altered flow regimes: A literature review to inform the science and management of environmental flows. Freshwater Biology 55(1): 194-205.

Pollard, SJT; Kemp, RV; Crawford, M; Duarte-Davidson, R; Irwin, JG; Yearsley, R. 2004. Characterizing environmental harm: Developments in an approach to strategic risk assessment and risk management. Risk Analysis 24: 1551-1560.

Powers, SE. 2005. Quantifying cradle-to-farm gate life-cycle impacts associated with fertilizer used for corn, soybean, and stover production. NREL/TP-510-27500. National Renewable Energy Laboratory. Available at: http://www1.eere.energy.gov/biomass/pdfs/37500.pdf.

Powers, SE. 2007. Nutrient loads to surface water from row crop production. International Journal of Life Cycle Assessment 12(6): 399-407.

Powers, SE; Hunt, CS; Heermann, SE; Corseuil, HX; Rice, D; Alvarez, PJJ. 2001. The transport and fate of ethanol and BTEX in groundwater contaminated by gasohol. Critical Reviews in Environmental Science and Technology 31(1): 79-123.

Puppan, D. 2002. Environmental evaluation of biofuels. Periodica Polytechnica Ser. Soc. Man. Sci. 10(1): 95-116.

Pyter, R; Heaton, E; Dohleman, F; Voigt, T; Long, S. 2009. Agronomic experiences with Miscanthus x giganteus in Illinois, USA. Methods in Molecular Biology 581: 41-52. Available at: http://www.springerlink.com/content/r5343757p6367878/.

Quinn, LD; Allen, DJ; Stewart, JR. 2010. Invasiveness potential of Miscanthus sinensis: implications for bioenergy production in the U.S. GCB Bioenergy, no. doi: 10.1111/j.1757-1707.2010.01062.x.

Quinn, LD; Stewart, JR. 2010. Assessing the invasiveness of Miscanthus sinensis, a potential bioenergy crop. OOS 38—Ecological Dimensions of Biofuel Production. Available at: http://eco.confex.com/eco/2010/techprogram/P22893.htm.

Ragauskas, AJ; Williams, CK; Davison, BH; Britovsek, G; Cairney, J; Eckert, CA; Frederick, WJ Jr.; Hallett, JP; Leak DJ; Liotta, CL; Mielenz, JR; Murphy, R; Templer, R; Tschaplinski, T. 2006. The path forward for biofuels and biomaterials. Science 311: 484-489.

Raghu, S; Anderson, RC; Daehler, CC; Davis, AS; Wiedenmann, RN; Simberloff, D; Mack, RN. 2006. Adding biofuels to the invasive species fire? Science 313: 1742. http://energyandenvironmentblog.dallasnews.com/invasive%20species%20and%20biofuels.pdf.

Randall, GW; Huggins, DR; Russelle, MP; Fuchs, DJ; Nelson, WW; Anderson, JL. 1997. Nitrate losses through subsurface tile drainage in conservation reserve program, alfalfa, and row crop systems. Journal of Environmental Quality 26(5): 1240-1247.

Ranney, JW; Mann, LK. 1994. Environmental considerations in energy crop production. Biomass Bioenergy 6(3): 211.

Raybould, A. 2007. Ecological versus ecotoxicological methods for assessing the environmental risks of transgenic crops. Plant Science 173: 589-602.

Regassa, T; Koelsch, R; Erickson, G. 2008. Impact of feeding distillers grains on nutrient planning for beef cattle systems. University of Nebraska-Lincoln Extension—RP190. Available at: http://www.ianrpubs.unl.edu/epublic/live/rp190/build/rp190.pdf.

Reicosky, DC; Kemper, WD; Langdale, GW; Douglas, CL; Rasmussen, PE. 1995. Soil organic-matter changes resulting from tillage and biomass production. Journal of Soil and Water Conservation 50(3): 253-261.

Reilly, TE; Dennehy, KF; Alley, WM; Cunningham, WL. 2008. Ground-water availability in the United States. Circular 1323. U.S. Geological Survey.

Reiss, J; Bridle, JR; Montoya, JM; Woodward, G. 2009. Emerging horizons in biodiversity and ecosystem functioning research. Trends in Ecology and Evolution 24(9): 505-514.

RFA (Renewable Fuels Association). 2010. 2010 ethanol industry outlook: Climate of opportunity. Available at: http://ethanolrfa.org/page/-/objects/pdf/outlook/RFAoutlook2010_fin.pdf?nocdn=1.

Rice, ME. 2003. Transgenic rootworm corn: Assessing potential agronomic, economic, and environmental benefits. Online. Plant Health Progress doi: 10.1094/PHP-2004-0301-01-RV.

Richardson, DM. 1998. Forestry trees as invasive aliens. Conservation Biology 12(1): 18-26.

Richter, DD; Markewitz, D; Trumbore, SE; Wells, CG. 1999. Rapid accumulation and turnover of soil carbon in a re-establishing forest. Nature 400(6739): 56-58.

Richter, GM; Riche, AB; Dailey, AG; Gezan, SA; Powlson, DS. 2008. Is UK biofuel supply from Miscanthus water-limited? Soil Use and Management 24: 235-245.

Rittmann, BE. 2008. Opportunities for renewable bioenergy using microorganisms. Biotechnology and Bioengineering 100: 203-212.

Rockwood, DL; Rudie, AW; Ralph, SA; Zhu, JY; Winandy, JE. 2008). Energy product options for Eucalyptus species grown as short rotation woody crops. International Journal of Molecular Sciences 9: 1361-1378.

Romeis, J; Bartsch, D; Bigler, F; Candolfi, MP; Gielkens, MMC; Hartley, SE; Hellmich, RL; Huesing, JE; Jepson, PC; Layton, R; Quemada, H; Raybould, A; Rose, RI; Schiemann, J; Sears, MK; Shelton, AM; Sweet, J; Vaituzis, Z; Wolt, JD. 2008. Assessment of risk of insect-resistant transgenic crops to nontarget arthropods. Nature Biotechnology 26: 203-208.

Rost, S; Gerten, D; Hoff, H; Lucht, W; Falkenmark, M; Rockstrom, J. 2009. Global potential to increase crop production through water management in rainfed agriculture. Environmental Research Letters 4. Available at: http://iopscience.iop.org/1748-9326/4/4/044002/pdf/1748-9326_4_4_044002.pdf.

Roth, AM; Sample, DW; Ribic, CA; Paine, L; Undersander, DJ; Bartelt, GA. 2005. Grassland bird response to harvesting switchgrass as a biomass energy crop. Biomass and Bioenergy 28: 490-498.

Ruiz-Aguilar, GML; Fernandez-Sanchez, JM; Kane, SR; Kim, D; Alvarez, PJJ. 2002. Effect of ethanol and methyl-tert-butyl ether on monoaromatic hydrocarbon biodegradation: Response variability for aquifer materials under various electron-accepting conditions. Environmental Toxicology and Chemistry 21(12): 2631-2639.

Rupert, MG. 2008. Decadal-scale changes in nitrate in ground water of the United States, 1988-2004. Journal of Environmental Quality 37: S-240-S-248.

SAB (United States Environmental Protection Agency, Science Advisory Board). 2007. Hypoxia in the northern Gulf of Mexico, an update by the EPA Science Advisory Board. EPA-SAB-08-003.

Sartori, F; Lal, R; Ebinger, MH; Parrish, DJ. 2006. Potential soil carbon sequestration and CO2 offset by dedicated energy crops in the USA. Critical Reviews in Plant Sciences 25(5): 441-472.

Sawyer, J; Mallarino; A. 2007. Carbon and nitrogen cycling with corn biomass harvest. Integrated Crop Management IC-498(22): 250. Available at: http://www.ipm.iastate.edu/ipm/icm/2007/8-6/cn.html.

Schiffman, PM; Johnson, WC. 1989. Phytomass and detrital carbon storage during forest regrowth in the southeastern United-States Piedmont. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 19(1): 69-78.

Schmer, MR; Vogel, KP; Mitchell, RB; Perrin, RK. 2008. Net energy of cellulosic ethanol from switchgrass. Proceedings of the National Academy of Sciences 105(2): 464-469.

Schneckenberger, K; Kuzyakov, Y. 2007. Carbon sequestration under Miscanthus in sandy and loamy soils estimated by natural C-13 abundance. Journal of Plant Nutrition and Soil Science–Zeitschrift Fur Pflanzenernahrung Und Bodenkunde 170(4): 538-542.

Schwer, CB; Clausen, JC. 1989. Vegetative filter treatment of diary milkhouse wastewater. Journal of Environmental Quality 18: 446-451. See also: http://cfpub.epa.gov/npdes/stormwater/menuofbmps/index.cfm?action=factsheet_results&view=specific&bmp=82.

Searchinger, T; Heimlich, R; Houghton, RA; Dong, FX; Elobeid, A; Fabiosa, J; Tokgoz, S; Hayes, D; Yu, TH. 2008. Use of US croplands for biofuels increases greenhouse gases through emissions from land use change. Science 319(5867): 1238-1240.

Secchi, S; Babcock, BA. 2007. Figure 9. Acreage out of CRP as a function of corn prices and CRP payments. In: Impact of high crop prices on environmental quality: A case of Iowa and the Conservation Reserve Program. Report No. 07-WP-447. Iowa State University, Center for Agricultural and Rural Development.

Secchi, S; Gassman, PW; Williams, JR; Babcock, BA. 2009. Corn-based ethanol production and environmental quality: A case of Iowa and the conservation reserve program. Environmental Management 44(4): 732-744.

SECEX (Brazilian Ministry of Development, Industry and Trade). n.d. Table: Destino das exportações brasileiras de álcool etílico. Available at: http://www.desenvolvimento.gov.br/sitio/interna/interna.php?area=2&menu=999.

Semere, T; Slater, FM. 2006. Ground flora, small mammal and bird species diversity in miscanthus and reed canary-grass fields. Biomass and Bioenergy 31(1): 20-29.

Sexton, S; Zilberman, D; Rajagopal, D; Hochman, G. 2009. The role of biotechnology in a sustainable biofuel future. AgBioForum 12(1): 130-140.

Shah, T; Burke, J; Villholth, K. 2007. Groundwater: A global assessment of scale and significance. Chapter 10. In: Molden, D. (ed.). Water for food, water for life: A comprehensive assessment of water management in agriculture. Earthscan.

Sheehan, J; Camobreco, V; Duffield, J; Graboski, M; Shapouri, H. 1998a. Life cycle inventory of biodiesel and petroleum diesel for use in an urban bus. NREL/SR-580-24089. National Renewable Energy Laboratory.

Sheehan, J; Camboreco, V; Duffield, J; Graboski, M; Shapouri, H. 1998b. An overview of biodiesel and petroleum diesel life cycles. National Renewable Energy Laboratory.

Sheehan, J; Aden, A; Paustian, K; Killian, K; Brenner, J; Walsh, M; Nelson, R. 2004. Energy and environmental aspects of using corn stover for fuel ethanol. Journal of Industrial Ecology 7(3-4): 117-146.

Shepard, JP. 2006. Water quality protection in bioenergy production: the US system of forestry Best Management Practices. Biomass and Bioenergy 30(4): 378-384.

Shinners, KJ; Boettcher, GC; Hoffman, DS; Munk, JT; Muck, RE; Weimer, PJ. 2009. Single-pass harvest of corn grain and stover: Performance of three harvester configurations. Transactions of the ASABE 52(1): 51-60.

Shipitalo, MJ; Edwards, WM. 1998. Runoff and erosion control with conservation tillage and reduced-input practices on cropped watersheds. Soil & Tillage Research 46(1-2): 1-12.

Simpson, TW; Sharpley, AN; Howarth, RW; Paerl, HW; Mankin, KR. 2008. The new gold rush: Fueling ethanol production while protecting water quality. Journal of Environmental Quality 37(2): 318-324.

Simpson, TW; Martinelli, LA; Sharpley, AN; Howarth, RW. 2009. Impact of ethanol production on nutrient cycles and water quality: The United States and Brazil as case studies. In: Howarth, RW; Bringezu, S (eds). Biofuels: Environmental consequences and interactions with changing land use. Proceeding of the Scientific Committee on Problems of the Environment (SCOPE) International Biofuels Project Rapid Assessment. pp. 153-167.

South Dakota State University. 2009. The cost of wet corn at harvest. Extension Extra. ExEx5056. Available at: http://agbiopubs.sdstate.edu/articles/ExEx5056.pdf.

Spencer, JL; Raghu, S. 2009. Refuge or reservoir? The potential impacts of the biofuel crop Miscanthus x giganteus on a major pest of maize. PLoS One 4(12): e8336.

Stanton, B; Eaton, J; Johnson, J; Rice, D; Schuette, B; Moser, B. 2002. Hybrid poplar in the Pacific Northwest—the effects of market-driven management. Journal of Forestry 100(4): 28-33.

Sticklen, MB. 2007. Feedstock crop genetic engineering for alcohol fuels. Crop Science 47: 2238-2248.

Sticklen, MB. 2009. Expediting the biofuels agenda via genetic manipulations of cellulosic bioenergy crops. Biofuels, Bioproducts and Biorefining 3: 448-455.

Stout, BM; Benfield, EF; Webster, JR. 1993. Effects of forest disturbance on shredder production in Southern Appalachian headwater streams. Freshwater Biology 29(1): 56-69.

Suter, GW. 2007. Ecological risk assessment. CRC Press.

Taylor, RL; Maxwell, BD; Boik, RJ. 2006. Indirect effects of herbicides on bird food resources and beneficial arthropods. Agriculture, Ecosystems and Environment 116(3-4): 157-164.

Teasdale, JR; Brandsaeter, LO; Calegari, A; Skora Neto, F. 2007. Cover crops and weed management. Non-chemical weed management. Available at: http://www.ars.usda.gov/sp2UserFiles/Place/12650400/Non-ChemWeedMgmt.pdf.

Thiffault, E; Pare, D; Belanger, N; Munson, A; Marquis, F. 2006. Harvesting intensity at clear-felling in the boreal forest: Impact on soil and foliar nutrient status. Soil Science Society of America Journal 70(2): 691-701.

Thornton, KW; Holbrook, SP; Stolte, KL; Landy, RB. 2000. Effects of forest management practices on Mid-Atlantic streams. Environmental Monitoring and Assessment 63: 31-41.

Thurmond, W. 2008. Biodiesel 2020: A global market survey. Available at: http://www.emerging-markets.com/biodiesel.

Tilman, D; Lehman, C. 2006. Carbon-negative biofuels from low-input high diversity grassland biomass. Science 314: 1598-1600.

Tilman, D; Hill, J; Lehman, C. 2006. Carbon-negative biofuels from low-impact high-diversity grassland biomass. Science 314 (5805): 1598-1600.

Tilman, D; Socolow, R; Foley, JA; Hill, J; Larson, E; Lynd, L; Pacala, S; Reilly, J; Searchinger, T; Somerville, C; Williams, R. 2009. Beneficial biofuels—the food, energy, and environment trilemma. Science 325: 270-271.

Timmons, D; Allen, G; Damery, D. 2008. Biomass energy crops: Massachusetts' potential. http://www.mass.gov/Eoeea/docs/doer/renewables/biomass/bio-ma-potential-crop.pdf.

Titus, BD; Roberts, BA; Deering, KW. 1997. Soil solution concentrations on three white birch sites in central Newfoundland following different harvesting intensities. Biomass and Bioenergy 13(4-5): 313-330.

Unc, A; Goss, MJ. 2004. Transport of bacteria from manure and protection of water resources. Applied Soil Ecology 25(1): 1-18.

UNEP/GRID-Arendal. 2009. Kick the habit: A UN guide to climate neutrality. Available at: http://maps.grida.no/go/graphic/biofuel-production-map.

Updegraff, K; Baughman, MJ; Taff, SJ. 2004. Environmental benefits of cropland conversion to hybrid poplar: economic and policy considerations. Biomass and Bioenergy 27: 411-428.

USDA (United States Department of Agriculture). 2010a. USDA long-term agricultural projection tables. Table 18. U.S. corn. Available at: http://usda.mannlib.cornell.edu/MannUsda/viewStaticPage.do?url=http://usda.mannlib.cornell.edu/usda/ers/94005/./2010/index.html.

USDA (United States Department of Agriculture). 2010b. USDA long-term agricultural projection tables. Table 23. Soybeans and products. Available at: http://usda.mannlib.cornell.edu/MannUsda/viewStaticPage.do?url=http://usda.mannlib.cornell.edu/usda/ers/94005/./2010/index.html.

USDA (United States Department of Agriculture). 2010c. USDA long-term agricultural projection tables. Table 17. Acreage for major field crops and Conservation Reserve Program (CRP) assumptions, long-term projections. Available at: http://usda.mannlib.cornell.edu/MannUsda/viewStaticPage.do?url=http://usda.mannlib.cornell.edu/usda/ers/94005/./2010/index.html.

USDA (United States Department of Agriculture). 2010d. Sugar and sweeteners outlook. Effects of global sugar markets on U.S. ethanol. Economic Research Service. Available at: http://www.ers.usda.gov/publications/sss/2010/Nov/SSSM267.pdf.

USDA (United States Department of Agriculture) n.d. Global Agricultural Trade System (GATS) online. Foreign Agriculture Service. Available at: http://www.fas.usda.gov/gats/ExpressQuery1.aspx.

U.S. DOE (United States Department of Energy). 2006. Energy demands on water resources: Report to Congress on the interdependency of energy and water. Available at: http://www.rivernetwork.org/sites/default/files/EnergyDemands_0.pdf.

U.S. DOE (United States Department of Energy). 2008. World biofuels production potential: Understanding the challenges to meeting the U.S. Renewable Fuel Standard. Available at: http://www.pi.energy.gov/documents/20080915_WorldBiofuelsProductionPotential.pdf.

U.S. DOE (United States Department of Energy). 2010. National algal biofuels technology roadmap. Available at: http://www1.eere.energy.gov/biomass/pdfs/algal_biofuels_roadmap.pdf.

U.S. EPA (United States Environmental Protection Agency). 1999. 5. Livestock manure management. In: U.S. methane emissions 1990-2020: Inventories, projections, and opportunities for reductions. Available at: http://epa.gov/methane/reports/05-manure.pdf.

U.S. EPA (United States Environmental Protection Agency). 2000. National water quality inventory 2000 report. Available at: http://www.epa.gov/305b/2000report.

U.S. EPA (United States Environmental Protection Agency). 2001. Control of air pollution from new motor vehicles: Heavy-duty engine and vehicle standards and highway diesel fuel sulfur control requirements. 40 CFR Parts 69, 80, and 86. Available at: http://www.epa.gov/otaq/highway-diesel/regs/2007-heavy-duty-highway.htm.

U.S. EPA (United States Environmental Protection Agency). 2002. A comprehensive analysis of biodiesel impacts on exhaust emissions. EPA-420-P-02-001. Available at: http://www.epa.gov/otaq/models/analysis/biodsl/p02001.pdf.

U.S. EPA (United States Environmental Protection Agency). 2006a. Framework for developing suspended and bedded sediments (SABS) water quality criteria. EPA-822-R-06-001. Available at: http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=164423.

U.S. EPA (United States Environmental Protection Agency). 2006b. Air quality criteria for ozone and related photochemical oxidants (2006 final). EPA-600-R-05-004aF-cF.

U.S. EPA (United States Environmental Protection Agency). 2006c. Wadeable Streams Assessment: A collaborative survey of the nation's streams. EPA-841-B-06-002. Available at: http://www.epa.gov/owow/streamsurvey.

U.S. EPA (United States Environmental Protection Agency). 2007a. Environmental law applicable to construction and operation of ethanol plants. EPA-907-B-07-001. Available at: http://www.epa.gov/region07/priorities/agriculture/pdf/ethanol_plants_manual.pdf.

U.S. EPA (United States Environmental Protection Agency). 2008a. Environmental law applicable to construction and operation of biodiesel production facilities. EPA-907-B-08-001. Available at: http://www.epa.gov/Region7/priorities/agriculture/pdf/biodiesel_manual.pdf.

U.S. EPA (United States Environmental Protection Agency). 2008b. National coastal condition report—III. EPA-842-R-08-002. Available at: http://www.epa.gov/nccr.

U.S. EPA (United States Environmental Protection Agency). 2008c. Applicability of effluent guidelines and categorical pretreatment standards to biodiesel manufacturing. Available at: http://www.epa.gov/npdes/pubs/memo_biodieselpretreatment_aug08.pdf.

U.S. EPA (United States Environmental Protection Agency). 2009a. National water quality inventory: Report to Congress: 2004 reporting cycle. EPA-841-R-08-001. Available at: http://www.epa.gov/305b.

U.S. EPA (United States Environmental Protection Agency). 2009b. Biofuels compendium. Available at: http://www.epa.gov/oust/altfuels/bfcompend.htm.

U.S. EPA (United States Environmental Protection Agency). 2009c. Biofuels compendium—ethanol—equipment compatibility. Available at: http://www.epa.gov/oust/altfuels/ethcompat.htm.

U.S. EPA (United States Environmental Protection Agency). 2009d. Report to Congress on public health, air quality, and water resource impacts of fuel additive substitutes for MTBE, as required by Section 1505 of the Energy Policy Act 2005.

U.S. EPA (United States Environmental Protection Agency). 2010a. Regulation of fuels and fuel additives: Changes to Renewable Fuel Standard Program: Final rule. Available at: http://www.regulations.gov/search/Regs/contentStreamer?objectId=0900006480ac93f2&disposition=attachment&contentType=pdf.

U.S. EPA (United States Environmental Protection Agency). 2010b. Renewable fuel standard program (RFS2) regulatory impact analysis. EPA-420-R-10-006. Available at: http://www.epa.gov/otaq/renewablefuels/420r10006.pdf.

U.S. EPA (United States Environmental Protection Agency). 2010c. Water quality assessment and total maximum daily loads information (ATTAINS). Available at: http://www.epa.gov/waters/ir/index.html.

U.S. EPA (United States Environmental Protection Agency). 2010d. National causes of impairment. In: National summary of state information. Available at: http://iaspub.epa.gov/waters10/attains_nation_cy.control#causes.

U.S. EPA (United States Environmental Protection Agency). 2010e. Acetochlor (CASRN 34256-82-1). In: Integrated Risk Information System. Available at: http://www.epa.gov/IRIS/subset/0521.htm.

U.S. EPA (United States Environmental Protection Agency). 2010f. Alachlor (CASRN 15972-60-8). In: Integrated Risk Information System. Available at: http://www.epa.gov/IRIS/subst/0129.htm.

U.S. EPA (United States Environmental Protection Agency). 2010g. Carbaryl (CASRN 63-25-2). In: Integrated Risk Information System. Available at: http://www.epa.gov/IRIS/subst/0019.htm.

U.S. EPA (United States Environmental Protection Agency). 2010h. Regulation of fuels and fuel additives: Changes to renewable fuel standard program. EPA-HQ-OAR-2005-0161.

U.S. EPA (United States Environmental Protection Agency). 2010i. Definition of solid waste for RCRA Subtitle C hazardous waste. Available at: http://www.epa.gov/osw/hazard/dsw/.

U.S. Forest Service. 2005. Evaluating sedimentation risks associated with fuel management. RMRS-RN-23-8-WWW.

U.S. Forest Service. 2010. Draft—National Report on Sustainable Forests—2010. Available at: http://www.fs.fed.us/research/sustain/2010SustainabilityReport/documents/draft2010sustainabilityreport.pdf.

U.S. ITC (United States International Trade Commission). 2010. Interactive Tariff and Trade DataWeb. U.S. imports for consumption. Report: HTS-2207: Ethyl alcohol, undenatured of an alcoholic strength by volume of 80% vol. or higher; ethyl alcohol and other spirits, denatured, of any strength first unit of quantity by first unit of quantity.

Van der Steen, NP; Nakiboneka, P; Mangalika, L; Ferrer, AVM; Gijzen, HJ. 2003. Effect of duckweed cover on greenhouse gas emissions and odor release from waste stabilization ponds. Water Science and Technology 48(2): 341-348.

Vogel, KP; Masters, RA. 1998. Developing switchgrass into a biomass fuel crop for the midwestern USA. Paper presented at BioEnergy '98: Expanding Bioenergy Partnerships, Madison, Wisconsin, 4–8 October 1998.

Volk, TA; Abrahamson, LP; Nowak, CA; Smart, LB; Tharakan, PJ; White, EH. 2006. The development of short-rotation willow in the northeastern United States for bioenergy and bioproducts, agroforestry and phytoremediation. Biomass and Bioenergy 30(8-9): 715-727.

Von Blottnitz, H; Curran, MA. 2007. A review of assessments conducted on bio-ethanol as a transportation fuel from a net energy, greenhouse gas, and environmental life cycle perspective. Journal of Cleaner Production 15: 607-619.

Walsh, ME; de la Torre Ugarte, DG; Shapouri, H; Slinsky, SP. 2003. Bioenergy crop production in the United States: Potential quantities, land use changes, and economic impacts on the agricultural sector. Environmental and Resource Economics 24(4): 313-333.

Walter, WD; Vercauteren, KC; Gilsdorf, JM; Hygnstrom, SE. 2009. Crop, native vegetation, and biofuels: Response of white-tailed deer to changing management priorities. Journal of Wildlife Management 73: 339-344.

Wang, L; Robertson, DM; Garrison, PJ. 2007a. Linkages between nutrients and assemblages of macroinvertebrates and fish in wadeable streams. Environmental Management 39: 194-212. Available at : http://www.springerlink.com/content/4k441384336k1x28/fulltext.pdf.

Wang, M; Wu, M; Huo, H. 2007b. Life-cycle energy and greenhouse gas emission impacts of different corn ethanol plant types. Environmental Research Letters 2: 1-13.

Weed, DAJ; and Kanwar, RS. 1996. Nitrate and water present in and flowing from root-zone soil. Journal of Environmental Quality 25: 709-719.

Weibe, K; Gollehon, N. 2006. Agricultural resources and environmental indicators, 2006 edition. Economic Information Bulletin No. (EIB-16). U.S. Department of Agriculture, Economic Research Service. Available at: http://www.ers.usda.gov/publications/arei/eib16/.

Weigelt, A; Weisser, WW; Buchmann, N; Scherer-Lorenzen, M. 2009. Biodiversity for multifunctional grasslands: Equal productivity in high-diversity low-input and low-diversity high-input systems. Biogeosciences 6: 1695-1706.

Whalen, J; Cissel, B. 2009. Soil insect management in field corn. University of Delaware Cooperative Extension.

Whicker, JJ; Pinder, JE; Breshears, DD. 2008. Thinning semiarid forests amplifies wind erosion comparably to wildfire: Implications for restoration and soil stability. Journal of Arid Environments 72(4): 494-508.

White, EM. 2010. Woody biomass for bioenergy and biofuels in the United States—a briefing paper. Gen. Tech. Rep. PNW-GTR-825. U.S. Department of Agriculture.

Wilkinson, M; Tepfer, M. 2009. Fitness and beyond: Preparing for the arrival of GM crops with ecologically important novel characters. Environmental Biosafety Research 8: 1-14.

Williams, PR; Inman, DD; Aden, A; Heath, GA. 2009. Environmental and sustainability factors associated with next-generation biofuels in the U.S.: What do we really know? Environmental Science and Technology 43: 4763-4775.

Wiltsee, G. 1998. Urban waste grease resource assessment. NREL/SR-570-26141. National Renewable Energy Laboratory. Available at: http://www.biodiesel.org/resources/sustainability/pdfs/NREL%20Urban%20Waste%20Grease%20Resources%20Assessment.pdf.

Winebrake, J; Wang, M; He, D. 2001. Toxic emissions from mobile sources: A total fuel-cycle analysis for conventional and alternative fuel vehicles. Journal of the Air and Waste Management Association 51: 1073-1086.

Wolt, JD. 2009. Advancing environmental risk assessment for transgenic biofeedstock crops. Biotechnology for Biofuels. 2: 27 doi: 10.1186/1754-6834-2-27.

Wösten, JHM; van den Berg, J; van Eijk, P; Gevers, GHM; Giesen, WBJT; Hooijer, A; Idris, A; Leenman, PH; Rais, DS; Siderius, C; Silvius, MJ; Suryadiputra, N; Wibisono, IT. 2006. Interrelationships between hydrology and ecology in fire degraded tropical peat swamp forests. International Journal of Water Resources Development 22(1): 157-174.

Wright LL. 1994. Production technology status of woody and herbaceous crops. Biomass and Bioenergy 6(3): 191-210.

Wu, M. 2008. Analysis of the efficiency of the U.S. ethanol industry. Center for Transportation Research, Argonne National Laboratory.

Wu, M; Wang, M. 2006. Energy and emission benefits of alternative transportation liquid fuel derived from switchgrass: A fuel life-cycle assessment. Biotechnology Progress 22(4): 1012-1024.

Wu, M; Mintz, M; Wang, M; Arora, S. 2009. Water consumption in the production of ethanol and petroleum gasoline. Environmental Management 44: 981-997.

Yacobucci, B. 2005. CRS report to Congress: Ethanol imports and the Caribbean Basin Initiative. Congressional Research Service. Order Code RS21930. Available at: http://www.policyarchive.org/handle/10207/bitstreams/3978.pdf.

Yanowitz, J; McCormick, R. 2009. Effect of E85 on tailpipe emissions from light-duty vehicles. Journal of the Air and Waste Management Association 59: 172–182.

Zah, R; Ruddy, TF. 2009. International trade in biofuels: An introduction to the special issue. Journal of Cleaner Production 17: S1-S3.

Zavaleta, ES; Pasari, JR; Hulvey, KB; Tilman, GD. 2010. Sustaining multiple ecosystem functions in grassland communities requires higher biodiversity. Proceedings of the National Academy of Sciences of the United States of America 107: 1443-1446.

Zhu, Y; Fox, RH. 2003. Corn-soybean rotation effects on nitrate leaching. Agronomy Journal 95: 1028-1033.

Appendix A

Glossary and Acronyms

advanced biofuel: A renewable fuel, other than ethanol derived from corn starch that has life cycle greenhouse gas (GHG) emissions that are at least 50 percent less than life cycle GHG emissions from petroleum fuel. A 60-percent reduction in GHG is required from cellulosic biofuels to get credit for being an “advanced” biofuel.

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: Any plant-like organisms that are usually photosynthetic and aquatic, but do not have true roots, stems, leaves, or vascular tissue, and that have simple reproductive structures. Algae are distributed worldwide in the sea, in fresh water, and in wastewater. Most are microscopic, but some are quite large (e.g., some marine seaweeds that can exceed 50 meters in length).

B100: Pure (i.e., 100 percent) biodiesel, also known as “neat biodiesel.”

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.

Best Management Practices (BMPs): Best management practices are the techniques, methods, processes, and activities commonly accepted and used to facilitate compliance with applicable requirements, and that provide an effective and practicable means of avoiding or reducing the potential environmental impacts.

biodiesel (also known as “biomass-based diesel”): A renewable fuel produced through transesterification of organically derived oils and fats. May be used as a replacement for or component of diesel fuel.

biodiversity: The variety and variability among living organisms and the ecological complexes in which they occur. Biodiversity can be defined as the number and relative frequency of different items, from complete ecosystems to the biochemical structures that are the molecular basis of heredity. Thus, the term encompasses ecosystems, species, and genes.

biofuel: Any fuel made from organic materials or their processing and conversion derivatives.

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.

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.

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 consumer-ready biofuel.

biofuel supply chain: The five main stages involved in the life cycle of a biofuel: feedstock production, feedstock logistics, fuel production, fuel distribution, and fuel use.

biogenic: Produced by living organisms or a biological process.

biomass: Any plant-derived organic matter (e.g., agricultural crops and crop wastes; wood and wood wastes and residues; aquatic plants; perennial grasses).

biomass-based diesel: See "biodiesel" above. Biomass-based diesel includes non-co-processed renewable diesel, which does not use the transesterification technology.

cellulosic biofuel: A renewable fuel derived from lignocellulose (i.e., plant biomass comprised of cellulose, hemicellulose, and lignin that is a main component of nearly every plant, tree, and bush in meadows, forests, and fields). Lignocellulose is converted to cellulosic biofuel by separating the sugars from the residual material, mostly lignin, and then fermenting, distilling, and dehydrating this sugar solution.

Conservation Reserve Program (CRP): A U.S. Department of Agriculture program that provides technical and financial assistance to eligible farmers and ranchers to address soil, water, and related natural resource concerns on their lands in an environmentally beneficial and cost-effective manner. It encourages farmers to convert highly erodible cropland or other environmentally sensitive acreage to vegetative cover, such as tame or native grasses, wildlife plantings, trees, filter strips, or riparian buffers. Farmers receive an annual rental payment for the term of the multi-year contract.

conservation tillage: Any cultivation system that leaves at least one third of the land surface covered with residue after planting in order to reduce soil erosion and conserve soil productivity. One example would be "no-till," where fields are not tilled at all and crops are planted directly into the existing residue. Other variations include "strip-till" or "ridge-till," which remove some, but not all, of the residue from the harvested area.

conventional biofuel: In the context of this report, "conventional biofuel" refers to ethanol derived from corn starch that does *not* lead to at least a 50 percent reduction in greenhouse gas emissions compared to petroleum.

corn stover: The stalks, leaves, husks, and cobs that are *not* removed from the fields when the corn is harvested.

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 corn or soybean bushels per acre.)

direct land use change: In the context of biofuel, "direct land use change" refers to land conversion that is directly related and easily attributable to the biofuel supply chain. For example, a U.S. farmer deciding to take land out of the Conservation Reserve Program in order to grow more corn for ethanol would be considered a direct land use change.

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.

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 comes from dry milling.

E10: A fuel mixture of 10 percent ethanol and 90 percent gasoline based on volume.

E85: A fuel mixture of 85 percent ethanol and 15 percent gasoline based on volume.

ecosystem health: The ability of an ecosystem to maintain its metabolic activity level and internal structure and organization, and to resist external stress over time and space scales relevant to the ecosystem.

effluent: Liquid or gas discharged in the course of industrial processing activities, usually containing residues from those processes.

Energy Independence and Security Act (Public Law 110-140) (EISA): Signed into law on December 19, 2007, this legislation established energy management goals and requirements while also amending portions of the National Energy Conservation Policy Act. EISA's stated goals are to move the U.S. toward greater energy independence and security; increase production of clean renewable fuels; protect consumers; increase the efficiency of products, buildings, and vehicles; promote research on and deploy greenhouse gas capture and storage options; and improve the energy performance of the federal government.

environmental life cycle assessment: In the context of this report, an environmental life cycle assessment is an assessment in which the LCA methodology (see "life cycle assessment") is applied to address the full range of potential environmental impacts over all environmental media.

ethanol (also known as "bioethanol"): A colorless, flammable liquid produced by fermentation of sugars. Ethanol is used directly as a fuel and fuel oxygenate.

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.

feedstock: In the context of biofuel, "feedstock" refers to a biomass-based material that is converted for use as a fuel or energy product.

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.

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.

forest residue: Includes 1) tops, limbs, and other woody material *not* removed in forest harvesting operations in commercial hardwood and softwood stands; and 2) woody material resulting from forest management operations such as pre-commercial thinning and removal of dead and dying trees.

forest thinning: Removal of residues from overgrown forests to reduce forest fire risk or increase forest productivity. Residues are typically too small or damaged to be sold as round wood but can be used as biofuel feedstock.

greenhouse gases: 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.

harvesting forest residue: See “forest thinning” above.

hemicellulose: any of various plant polysaccharides less complex than cellulose and easily hydrolysable to monosaccharides (simple sugars) and other products.

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.

hypoxia: The state of an aquatic ecosystem characterized by low dissolved oxygen levels (less than 2 to 3 parts per million) due to accelerated algal growth and reduced light penetration because of excessive nutrient levels (eutrophication). Low dissolved oxygen can reduce fish populations and species diversity in the affected area.

indirect land use change: In the context of biofuel, **“**indirect land use change**”** refers to land conversion that occurs as a market response to changes in the supply and demand of *goods other than biofuel* (e.g., food commodities) that result from changes in biofuel demand. For example, clearing of foreign land to plant corn as a food crop in response to reduced U.S. corn exports caused by increased use of U.S. corn to produce ethanol is considered to be an indirect land use change.

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.

invasive plants (also called invasives or noxious plants): An alien species whose introduction does or is likely to cause economic or environmental harm or harm to human health.

land cover: Vegetation, habitat, or other material covering a land surface.

land use: The human use of land involving the management and modification of natural environment or wilderness into built environment such as fields, pastures, and settlements.

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 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 above for definition of "biofuel life cycle.")

low-till: See "conservation tillage."

milling residues (primary and secondary): Wood and bark residues produced in processing (or milling) logs into lumber, plywood, and paper.

mitigation: In the context of the environment, action to reduce adverse environmental impacts.

neat biofuel: See "B100."

net energy balance: In the context of biofuel, refers to the energy content in the resulting biofuel less the total amount of energy used over the production and distribution process.

nitrogen fixation: The transformation of atmospheric nitrogen into nitrogen compounds that can be used by growing plants. Nitrogen-fixing species, such as soybeans, can accomplish this process directly.

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.

oxygenated fuels: Fuels, typically gasoline, that have been blended with alcohols or ethers that contain oxygen in order to reduce carbon monoxide and other emissions.

ozone: 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.

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.

renewable biomass: As defined by the 2007 Energy Independence and Security Act, renewable biomass means each of the following:

- Planted crops and crop residue from agricultural land cleared prior to December 19, 2007, and actively managed or fallow on that date.
- Planted trees and tree residue from tree plantations cleared prior to December 19, 2007, and actively managed on that date.
- Animal waste material and byproducts.
- Slash and pre-commercial thinnings from non-federal forestlands that are neither old-growth nor listed as critically imperiled or rare by a State Natural Heritage program.
- Biomass cleared from the vicinity of buildings and other areas at risk of wildfire.
- Algae.
- Separated yard waste and food waste.

renewable fuel: A fuel produced from renewable biomass that is used to replace or reduce the use of fossil fuel.

Renewable Fuels Standard (RFS) program: An EPA program created under the Energy Policy Act (EPAct) of 2005 that established the first renewable fuel volume mandate in the United States. The original RFS program (RFS1) required 7.5 billion gallons of renewable fuel to be blended into gasoline by 2012. (See below for RFS2.)

RFS2: The Renewable Fuels Standard 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.

RFS2 Regulatory Impact Analysis (RIA): EPA’s analysis of the impacts of the increase in production, distribution, and use of the renewable fuels need to meet the RFS2 volumes established by Congress in the 2007 Energy Independence and Security Act (EISA).

riparian forest buffer: An area of trees and shrubs located adjacent to streams, lakes, ponds, and wetlands that may reduce nutrient loading, soil erosion, and pesticide contamination by removing soil particles and contaminants from overland water flow.

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.

sedimentation: The process of solids settling out of water due to gravity.

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 3 to 15 years. Examples include: hybrid poplars (*Populus* spp.), willow (*Salix* spp.), Loblolly pine (*Pinus taeda*), and Eucalyptus.

soil erosion: The wearing away of land by the action of wind, water, gravity, or a combination thereof.

soil organic matter: Decomposing plant and animal material in soil.

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.

sugarcane bagasse: The fibrous material that remains after sugar is pressed from sugarcane.

sweet sorghum pulp: The bagasse or dry refuse left after the juice is extracted from sweet sorghum stalks during the production of ethanol and other sweet sorghum products. The pulp is usually treated as farm waste in plantations that grow sweet sorghum for biofuel production.

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.

turbidity: A cloudy condition in water due to suspended silt or organic matter.

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.

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 ground water sources (e.g., aquifers) for consumptive uses.

water quality: Water quality is a measure of the suitability of water for a particular use based on selected physical, chemical, and biological characteristics. It 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

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.

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 by-products of forest management activities.

APPENDIX B:

SUMMARY OF SELECTED STATUTORY AUTHORITIES HAVING POTENTIAL IMPACT ON THE PRODUCTION AND USE OF BIOFUELS

Table B-1: Summary of Selected Statutory Authorities Having Potential Impact on the Production and Use of Biofuels

Summary of Statute/Program	Stage of Lifecycle: Feedstock Production and Transport	Stage of Lifecycle: Biofuel Production, Transport, and Storage	Stage of Lifecycle: Use of Biofuel
Clean Air Act (CAA) (http://www.epa.gov/air/caa/)			
The CAA defines EPA's responsibilities for protecting and improving air quality and stratospheric ozone. It requires EPA to set national ambient air quality standards (NAAQS) for widespread pollutants from numerous and diverse sources considered harmful to public health and the environment. EPA and states must develop regulations to achieve and maintain the NAAQS and to control other pollutants.	Vehicles used for the transportation of feedstock may be subject to an inspection and maintenance program for tailpipe emissions and vehicle emission standards for air quality.	• A biofuel plant will need to obtain an air operating permit for day-to-day facility operations. Based on potential-to-emit, a facility may be required to obtain a Title V Air Operating Permit. Operating permits will be issued containing emission limits, monitoring, and record keeping requirements. • Pre-construction permits will be required for initial construction and for changes made to the plant. There are two types of major pre-construction permits under the New Source Review (NSR) Program: Prevention of Significant Deterioration permits, and Nonattainment NSR permits. A minor pre-construction permit would be required if major NSR is not required. • A vehicle used for the transportation of biofuels may be subject to an inspection and maintenance program.	The CAA regulates the amount of ethanol mixed in gasoline as part of the reformulated gasoline program.

Table B-1: Summary of Selected Statutory Authorities Having Potential Impact on the Production and Use of Biofuels

Summary of Statute/Program	Stage of Lifecycle: Feedstock Production and Transport	Stage of Lifecycle: Biofuel Production, Transport, and Storage	Stage of Lifecycle: Use of Biofuel
Clean Water Act (CWA) (http://www.epa.gov/watertrain/cwa/)			
The goal of the CWA is to restore and maintain the chemical, physical, and biological integrity of the nation's waters. Entities that discharge to waters of the U.S. through point sources (i.e., pipes, ditches, concentrated animal feeding operations), must obtain a National Pollutant Discharge Elimination System (NPDES) permit. These entities include many municipal, industrial, and construction-related sources of stormwater. States develop water quality standards (WQS) that define the goals for a water body by designating its uses, setting criteria to protect those uses, and establishing provisions to protect that water body. The CWA requires states to identify waters not meeting WQS and to develop Total Maximum Daily Loads (TMDLs) for those waters. TMDLs identify point and nonpoint source loads that can be discharged to a water body and still meet WQS.	Agricultural storm water and irrigation returns flows are exempted from NPDES permit requirements. Under Section 319, EPA provides grants to states to address non-point sources of pollution.	A biofuel production facility typically uses water for cooling and also for washing the biofuel product to remove impurities. The wastewater is discharged either directly to a water body or indirectly to a municipal wastewater treatment plant. Both are point source discharges, regardless whether the facility uses a septic tank or treatment prior to discharge. Any discharge into a water body by a point source must have an NPDES permit prior to discharge. Permits may be required for discharge to a municipal wastewater treatment system, which could include pre-treatment requirements. Land application of wastewater may be covered by an NPDES permit if it is determined that pollutants run off the application site to a waterway in a discernable channel or pipe. To minimize the impact of site runoff on water quality, a NPDES stormwater permit must be obtained for discharges to waters of the U.S. from any construction activity that disturbs 1 acre or more of land (including smaller sites that are part of a larger common plan of development).	Management of emergency response oil discharges must be reported to the National Response Center if they are in a quantity that "may be harmful."

Table B-1: Summary of Selected Statutory Authorities Having Potential Impact on the Production and Use of Biofuels

Summary of Statute/Program	Stage of Lifecycle: Feedstock Production and Transport	Stage of Lifecycle: Biofuel Production, Transport, and Storage	Stage of Lifecycle: Use of Biofuel
CWA: Section 404 Wetlands Program (www.epa.gov/owow/wetlands/laws/)			
Section 404 addresses the discharges of dredged or fill material into waters of the United States, including wetlands. Permits are required for activities such as expanded water resource projects (including dams, impoundments, and levees) and altering or dredging a water of the United States.	Most ongoing agricultural maintenance practices are exempt from Section 404.	Generally, Section 404 requires a permit before these materials may be placed in a U.S. water, such as a wetland, stream, river, slough, lake, bay, etc., during construction activities.	
Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) (http://www.epa.gov/lawsregs/laws/cercla.html)			
CERCLA provides a federal "Superfund" to clean up uncontrolled or abandoned hazardous-waste sites as well as accidents, spills, and other emergency releases of pollutants and contaminants into the environment. Through CERCLA, EPA was given authority to assure responsible parties' cooperation in site cleanup. CERCLA also regulates the property transfer of these sites.		Requirements under CERCLA that may apply include: • Reporting requirements for hazardous substances. • Implementation and periodic revision of the National Contingency Plan. • Management by emergency response authorities and responses to discharges of biofuels.	

Table B-1: Summary of Selected Statutory Authorities Having Potential Impact on the Production and Use of Biofuels

Summary of Statute/Program	Stage of Lifecycle		
	Feedstock Production and Transport	**Biofuel Production, Transport, and Storage**	**Use of Biofuel**
Emergency Planning and Community Right Know ACT (EPCRA) (http://www.epa.gov/oecaagct/lcra.html)			
The objective of the EPCRA is to: (1) allow state and local planning for chemical emergencies, (2) provide for notification of emergency releases of chemicals, and (3) address communities' right-to-know about toxic and hazardous chemicals.		Section 302 requires facilities with regulated chemicals (extremely hazardous substances) above threshold planning quantities to notify the state emergency response commission (SERC) and the local emergency planning committee (LEPC). Section 304 requires facilities to report a release of an extremely hazardous substance. Section 311 requires the facility to have material safety data sheets (MSDSs) on site for hazardous chemicals, as defined by the Occupational Safety and Health Act, that exceed certain quantities and to submit copies to their SERC, LEPC, and local fire department. Section 312 establishes reporting for any hazardous chemical or extremely hazardous chemical that is stored at a facility in excess of the designated threshold planning quantity. These reports are also known as the Tier II hazardous chemical inventory form. Section 313 (Toxics Release Inventory) requires owners or operators of certain facilities that manufacture, process or otherwise use any listed toxic chemicals, or chemical categories, in excess of threshold quantities to report annually to the EPA and to the state in which such facilities are located.	Electric utilities are subject to EPCRA Section 313 – Toxic Release Inventory Reporting.

Table B-1: Summary of Selected Statutory Authorities Having Potential Impact on the Production and Use of Biofuels

Summary of Statute/Program	Stage of Lifecycle: Feedstock Production and Transport	Stage of Lifecycle: Biofuel Production, Transport, and Storage	Stage of Lifecycle: Use of Biofuel
Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) (http://www.epa.gov/oecaagct/lfra.html)			
The objective of FIFRA is to provide federal control of pesticide distribution, sale, and use.	EPA reviews and registers pesticides for specified uses and can cancel the registration if information shows continued use would pose unreasonable risk. Consideration is given to worker exposure ecological exposure and food-chain imports.		
Hazardous Materials Transportation Act (Regulations codified 49 CFR) (http://www.phmsa.dot.gov/hazmat/regs and http://www.fmcsa.dot.gov/safety-security/hazmat/security-plan-guide.htm)			
The Department of Transportation regulations require procedures to be put in place ensuring the safe transport of hazardous materials. Also, regulation HM-232 requires companies to complete a written security assessment and to develop a security plan that is based on the assessment.		Requirements are in place for shippers and carriers of hazardous materials to prepare shipments for transport, placard containers for easy identification of hazards, and ensure the safe loading, unloading, and transport of materials. HM-232 requires companies to complete a written security assessment and to develop a security plan that is based on the assessment.	

Table B-1: Summary of Selected Statutory Authorities Having Potential Impact on the Production and Use of Biofuels

Summary of Statute/Program	Stage of Lifecycle: Feedstock Production and Transport	Stage of Lifecycle: Biofuel Production, Transport, and Storage	Stage of Lifecycle: Use of Biofuel
National Environmental Policy Act (NEPA) (http://www.epa.gov/compliance/nepa/)			
NEPA requires federal agencies to integrate environmental values into their decision making processes by considering the environmental impacts of their proposed actions and reasonable alternatives to those actions. To meet NEPA requirements in certain circumstances federal agencies prepare a detailed statement known as an Environmental Impact Statement (EIS).		If federal money is being used to partially or entirely finance the construction of a biofuel plant or any associated facility, such as an access road or water supply facility, then construction of the plant may be subject to NEPA. NEPA requires federal agencies to incorporate environmental considerations in their planning and decision-making and to prepare a detailed statement assessing the environmental impact of activities and alternatives that significantly affect the environment.	
Oil Pollution Act (OPA) of 1990 (http://www.epa.gov/lawsregs/laws/opa.html)			
The OPA of 1990 streamlined and strengthened EPA's ability to prevent and respond to catastrophic oil spills. A trust fund financed by a tax on oil is available to clean up spills when the responsible party is incapable or unwilling to do so. The OPA requires oil storage facilities and vessels to submit to the Federal government plans detailing how they will respond to large discharges.		Provides that the responsible party for a vessel or facility from which oil is discharged, or which poses a substantial threat of a discharge, is liable for: (1) certain specified damages resulting from the discharged oil; and (2) removal costs incurred in a manner consistent with the National Contingency Plan. Provides for spill contingency plans and mandates development of response plans for worst case discharge; and provides for requirements for spill removal equipment. Oil Spill Plans must be in place prior to operation, at facilities that have the potential to spill oil to navigable waters.	

Table B-1: Summary of Selected Statutory Authorities Having Potential Impact on the Production and Use of Biofuels

Summary of Statute/Program	Stage of Lifecycle		
	Feedstock Production and Transport	Biofuel Production, Transport, and Storage	Use of Biofuel
Renewable Fuel Standard (RFS) (http://www.epa.gov/otaq/fuels/renewablefuels/index.htm)			
The RFS program was created under the Energy Policy Act (EPAct) of 2005, and established the first renewable fuel volume mandate in the United States. As required under EPAct, the original RFS program (RFS1) required 7.5 billion gallons of renewable fuel to be blended into gasoline by 2012. Under the Energy Independence and Security Act (EISA) of 2007, the RFS program was expanded. EISA also required EPA to apply lifecycle greenhouse gas (GHG) performance threshold standards. The GHG requirement is that the lifecycle GHG emissions of a qualifying renewable fuel must be less than the lifecycle GHG emissions of the 2005 baseline average gasoline or diesel fuel that it replaces. Four different levels of reductions are required for the four different renewable fuel standards: Renewable Fuel (20%); Advanced Biofuel (50%); Biomass-based Diesel (50%); and Cellulosic Biofuel (60%).		If a facility produces 10,000 gallons or more of renewable fuel per year, then it may participate in the RFS program, though it is not required to do so. If a facility chooses to participate in the RFS program, it must satisfy the following criteria: • Register • Generate renewable identification • Transfer RINs with fuel • Provide product transfer documents • Follow blending requirements • Follow exporting requirements • Follow non-road use of fuel • Attest engagements • Keep records for 5 years • Report quarterly	

Table B-1: Summary of Selected Statutory Authorities Having Potential Impact on the Production and Use of Biofuels

<table>
<tr><th rowspan="2">Summary of Statute/Program</th><th colspan="3">Stage of Lifecycle</th></tr>
<tr><th>Feedstock Production and Transport</th><th>Biofuel Production, Transport, and Storage</th><th>Use of Biofuel</th></tr>
<tr><th colspan="4">Resource Conservation and Recovery Act (RCRA) (http://www.epa.gov/lawsregs/laws/rcra.html)</th></tr>
<tr><td>RCRA gives EPA the authority to control hazardous waste generation, transportation, treatment, storage, and disposal of hazardous waste. Facilities that handle hazardous waste are required to obtain an operating permit from the state agency or EPA. RCRA regulates USTs.</td><td></td><td>Regulatory issues related to waste generated by biofuel production - solid and hazardous waste include:
• New regulations on storage and transport of fuel related to expanded use of biofuels.
• New concerns related to assessing compatibility of fuel storage systems, managing water in storage tanks, protecting against corrosiveness and conductivity, managing methane formation, and detecting, preventing and responding to storage tank and pipe leaks and spills.
• Management of emergency response authorities and responses to biofuels spills.</td><td>UST leak detection and prevention are required.</td></tr>
<tr><th colspan="4">Safe Drinking Water Act (SDWA) (http://www.epa.gov/ogwdw/sdwa/)</th></tr>
<tr><td>The SDWA is the federal law that protects the safety of water distributed by public water systems. Under SDWA, EPA has National Primary Drinking Water Regulations for more than 90 contaminants and rules regarding monitoring of treated drinking water as well as reporting and public notification.</td><td>There are a number of threats to drinking water: anthropogenic chemicals including pesticides and improperly disposed chemicals; animal wastes; and naturally occurring substances. A primary impact to drinking water is nitrate pollution from row crops.</td><td>Wastewater from biofuel production facilities or corn starch ethanol facilities and leaking biofuel storage tanks can contaminate surface and ground drinking water resources, requiring treatment under SDWA.</td><td></td></tr>
</table>

Table B-1: Summary of Selected Statutory Authorities Having Potential Impact on the Production and Use of Biofuels

Summary of Statute/Program	Stage of Lifecycle Feedstock Production and Transport	 Biofuel Production, Transport, and Storage	 Use of Biofuel
Safe Drinking Water Act: Underground Injection Control (UIC) Program (http://www.epa.gov/safewater/uic/)			
The UIC program protects underground sources of drinking water by regulating the construction, operation, permitting, and closure of injection wells that place fluids underground for storage or disposal.	Agriculture drainage wells are Class V UIC wells. They are primarily regulated under state law.	A biofuels plant is subject to the requirements of the UIC Program if any of the following apply: • It is disposing of storm water, cooling water, industrial or other fluids into the subsurface via an injection well; • It has an on-site sanitary waste disposal system (e.g., septic system) that serves or has the capacity to serve 20 or more persons; • It has an on-site sanitary waste disposal system that is receiving other than a solely sanitary waste stream regardless of its capacity; or • It is undergoing a remediation process where fluids are being introduced into the subsurface via an injection well to facilitate or enhance the cleanup.	
Spill Prevention, Control and Countermeasure (SPCC) and Facility Response Plans (FRP) (http://www.epa.gov/oem/content/spcc/index.htm)			
The SPCC rule includes requirements for oil spill prevention, preparedness, and response to prevent oil discharges to navigable waters and adjoining shorelines. The rule requires specific facilities to prepare, amend, and implement SPCC Plans. The SPCC rule is part of the Oil Pollution Prevention regulation, which also includes the Facility Response Plan (FRP) rule.	The SPCC program requires certain farms (e.g., those that store oil and could reasonably be expected to discharge oil to waters of the US) to prepare and implement an SPCC Plan.	A biofuel facility is subject to this regulation if the following apply: • It is non-transportation related. • It has a total above-ground oil storage capacity greater than 1,320 gallons or a completely buried oil storage capacity greater than 42,000 gallons. • There is a reasonable expectation of an oil discharge into or upon navigable waters of the U.S. or adjoining shorelines. • Secondary containment cannot be provided for all regulated oil storage tanks.	

Table B-1: Summary of Selected Statutory Authorities Having Potential Impact on the Production and Use of Biofuels

Summary of Statute/Program	Stage of Lifecycle: Feedstock Production and Transport	Stage of Lifecycle: Biofuel Production, Transport, and Storage	Stage of Lifecycle: Use of Biofuel
Toxic Substances Control Act (TSCA) (http://www.epa.gov/lawsregs/laws/tsca.html)			
TSCA gives EPA broad authority to identify and control chemical substances that may pose a threat to human health or the environment. EPA's Office of Pollution Prevention and Toxics operates both the New Chemicals Program and the Biotechnology Program under Section 5 of TSCA. Both programs were established to help manage the potential risk from chemical substances and genetically-engineered (intergeneric) microorganisms new to the marketplace or applied in significant new uses. Additional sections of TSCA give EPA the broad authority to issue toxicity testing orders or to regulate the use of any existing chemicals that pose unreasonable risk.	Notification and review of new intergeneric genetically engineered microbes (e.g. bacteria, fungi and algae) used to produce biofuels feedstocks.	Mandatory notification and approval for new chemicals and new biological products, prior to manufacture and commercial use. New uses of chemicals are subject to review for potential environmental hazards under the Significant New Use Notification process. As a result of the review process, health and environmental effects testing of existing or new chemicals that pose unreasonable risk may be required. EPA may also restrict use and handling of chemicals or biological products as a result of their review.	

Appendix C

Basis for Figures 6-1, 6-2, and 7-3

This appendix presents three tables, Tables C-1, C-2, and C-3, which summarize the information providing the basis for Figures 6-1, 6-2, and 7-3, respectively. For each of the six feedstocks included in this report, Tables C-1 and C-2 briefly describe the current production status (Background), as well as the conditions anticipated to result in the most negative environmental effect (Most Negative Future Scenario) and the most positive environmental effect (Most Positive Future Scenario) in each of the environmental media considered. Table C-3 describes the basis for the three scenarios included in Figure 7-3.

Table C-1: Basis for Figure 6-1 (Maximum Potential Range of Environmental Impacts [on a Per Unit Area Basis] Resulting from Cultivation and Harvesting of the Six Biofuel Feedstocks Considered in this Report)

	Background	Impact Category	Conditions for Maximum Potential Negative Environmental Impact	Conditions for Maximum Potential Positive Environmental Impact
Corn Starch	Most current corn feedstock cultivation for biofuel production is a result of either 1) displacing soy production, 2) diverting existing corn grain to processing for fuel, or 3) placing former agricultural land back into production.	Water Quality	Corn grown with conventional tillage and high chemical inputs replaces lands in the Conservation Reserve Program (CRP).	Corn grown with comprehensive conservation practices replaces corn grown with existing production systems.
		Water Quantity	Irrigated corn replaces non-irrigated land use in drier area.	Production of non-irrigated corn.
		Soil Quality	Corn grown with conventional tillage and high chemical inputs replaces lands in the Conservation Reserve Program (CRP).	Corn grown with comprehensive conservation practices replaces corn grown with existing production systems.
		Air Quality	Irrigated corn grown with conventional tillage and high chemical inputs replaces lands in the Conservation Reserve Program (CRP).	Corn grown with comprehensive conservation practices replaces corn grown with existing production systems.
		Biodiversity	Corn grown with conventional tillage and high chemical inputs replaces lands in the Conservation Reserve Program (CRP).	Corn grown with comprehensive conservation practices replaces corn grown with existing production systems.
		Invasiveness	Negligible known effect.	Negligible known effect.

Table C-1: Basis for Figure 6-1 (Maximum Potential Range of Environmental Impacts [on a Per Unit Area Basis] Resulting from Cultivation and Harvesting of the Six Biofuel Feedstocks Considered in this Report)

	Background	Impact Category	Conditions for Maximum Potential Negative Environmental Impact	Conditions for Maximum Potential Positive Environmental Impact
Soybean	Most current soybean biofuel production comes from increased allocation of existing harvest to biodiesel.	Water Quality	Soy replaces lands in the Conservation Reserve Program (CRP).	Soy grown with comprehensive conservation practices replaces corn grown with conventional tillage and high chemical inputs.
		Water Quantity	Irrigated soy replaces non-irrigated land use in drier area.	Non-irrigated soy replaces irrigated corn.
		Soil Quality	Soy replaces lands in the Conservation Reserve Program (CRP).	Soy grown with comprehensive conservation practices replaces corn grown with conventional tillage and high chemical inputs.
		Air Quality	Irrigated soy replaces non-irrigated land use in drier area.	Non-irrigated soy grown with comprehensive conservation practices replaces corn grown with conventional tillage and high chemical inputs.
		Biodiversity	Soy replaces lands in the Conservation Reserve Program (CRP).	Soy grown with comprehensive conservation practices is diverted from existing production systems.
		Invasiveness	Negligible known effect.	Negligible known effect.

Table C-1: Basis for Figure 6-1 (Maximum Potential Range of Environmental Impacts [on a Per Unit Area Basis] Resulting from Cultivation and Harvesting of the Six Biofuel Feedstocks Considered in this Report)

	Background	Impact Category	Conditions for Maximum Potential Negative Environmental Impact	Conditions for Maximum Potential Positive Environmental Impact
Corn Stover	Not currently produced commercially for biofuel feedstock.	Water Quality	High rate of stover removal on highly erodible land.	Appropriate rate of stover removal to minimize erosion given site-specific characteristics and management practices.
		Water Quantity	High rate of stover removal on irrigated land in drier areas.	Appropriate rate of stover removal to minimize additional irrigation given site-specific characteristics and management practices.
		Soil Quality	High rate of stover removal on highly erodible land with low organic matter soil.	Appropriate rate of stover removal to minimize erosion given site-specific characteristics and management practices.
		Air Quality	High stover removal requires additional harvesting pass and increased subsequent fertilizer applications.	Appropriate rate of stover removal to minimize subsequent fertilizer applications; stover removed with corn in a single harvesting pass.
		Biodiversity	High rate of stover removal on highly erodible land that results in sedimentation to aquatic systems.	Appropriate rate of stover removal to minimize erosion given site-specific characteristics and management practices.
		Invasiveness	Negligible known effect.	Negligible known effect.

Table C-1: Basis for Figure 6-1 (Maximum Potential Range of Environmental Impacts [on a Per Unit Area Basis] Resulting from Cultivation and Harvesting of the Six Biofuel Feedstocks Considered in this Report)

	Background	Impact Category	Conditions for Maximum Potential Negative Environmental Impact	Conditions for Maximum Potential Positive Environmental Impact
Woody Biomass	Not currently produced commercially for biofuel feedstock.	Water Quality	Short-rotation woody crops (SRWC) with short replanting intervals and high chemical inputs, and without coppicing replace mature, managed tree plantations.	Short-rotation, coppiced woody crops with long replanting intervals, and low chemical inputs replace non-coppiced, managed forests with short replanting intervals, and high chemical inputs.
		Water Quantity	Irrigated SRWCs are grown in drier regions.	Production of non-irrigated SRWC in wetter regions. or Low to moderate rate of forest residue removal or thinning.
		Soil Quality	SRWC with short replanting intervals and without coppicing replace mature, managed tree plantations.	Short-rotation, coppiced woody crops with long replanting intervals, and low chemical inputs replace non-coppiced, managed forests with short replanting intervals, and high chemical inputs.
		Air Quality	SRWC with short replanting intervals, high chemical inputs, high isoprene emissions, and without coppicing replace mature, managed, low-isoprene emitting tree plantations.	Short-rotation, coppiced woody crops with long replanting intervals, low chemical inputs and low isoprene emissions replace non-coppiced, managed forests with short replanting intervals, high chemical inputs, and high isoprene emissions.
		Biodiversity	SRWC with short replanting intervals and high chemical inputs, and without coppicing replace mature, managed tree plantations.	Long rotation woody crop stands replace SRWC with short replanting intervals and high chemical inputs.
		Invasiveness	Woody species (e.g., *E. grandis*) are grown and become invasive.	Production and harvesting of non-invasive woody species.

Table C-1: Basis for Figure 6-1 (Maximum Potential Range of Environmental Impacts [on a Per Unit Area Basis] Resulting from Cultivation and Harvesting of the Six Biofuel Feedstocks Considered in this Report)

	Background	Impact Category	Conditions for Maximum Potential Negative Environmental Impact	Conditions for Maximum Potential Positive Environmental Impact
Perennial Grass	Not currently produced commercially for biofuel feedstock.	Water Quality	Perennial grasses established with conventional tillage and grown with a short planting interval and chemical inputs replace land in the CRP.	Perennial grasses established with no till grown with low chemical inputs and a long replanting interval replace corn grown with conventional tillage and high chemical inputs.
		Water Quantity	Irrigated perennial grasses replace non-irrigated land use in drier regions.	Non-irrigated perennial grasses replace irrigated corn.
		Soil Quality	Perennial grasses established with conventional tillage and grown with a short planting interval and chemical inputs replace land in the CRP.	Perennial grasses established with no till with a long replanting interval replace conventionally tilled row crops.
		Air Quality	Irrigated perennial grasses established with conventional tillage and grown with a short planting interval and chemical inputs replace land in the CRP.	Non-irrigated perennial grasses established with no till and grown with low chemical inputs and a long replanting interval replace irrigated corn grown with conventional tillage and high chemical inputs.
		Biodiversity	Uniformly-managed perennial grasses established with conventional tillage and grown with a short planting interval and chemical inputs replace land in the CRP.	Perennial grasses grown with low chemical inputs replace corn grown with conventional tillage and high chemical inputs.
		Invasiveness	Switchgrass (west of the Rockies) and *Miscanthus* are grown and become invasive or weedy.	Production of non-invasive native grasses or grass varieties bred for decreased invasiveness or weediness (e.g., sterile varieties).

Table C-1: Basis for Figure 6-1 (Maximum Potential Range of Environmental Impacts [on a Per Unit Area Basis] Resulting from Cultivation and Harvesting of the Six Biofuel Feedstocks Considered in this Report)

	Background	Impact Category	Conditions for Maximum Potential Negative Environmental Impact	Conditions for Maximum Potential Positive Environmental Impact
Algae	Not currently produced commercially for biofuel feedstock.	Water Quality	Untreated effluent from growing algae is discharged to the environment.	Algae are grown with wastewater; treated effluent is recycled for further use.
		Water Quantity	Algae are grown in drier regions (e.g., Southwest) with freshwater in open ponds.	Algae are grown with wastewater in closed bioreactors; treated effluent is recycled for further use.
		Soil Quality	Negligible known effect.	Negligible known effect
		Air Quality	Algae grown with added nutrients.	Algae grown with nutrients in wastewater.
		Biodiversity	Negligible known effect.	Negligible known effect.
		Invasiveness	Invasive algae species grown in open ponds escape and proliferate.	Production of non-invasive algae species in closed bioreactors.

Table C-2: Basis for Figure 6-2 (Maximum Potential Range of Environmental Impacts [on a Per Unit Volume Basis] Resulting from Ethanol and Biodiesel Production, Transport, Storage and Use)

	Background	Impact Category	Conditions for Maximum Potential Negative Environmental Impact	Conditions for Maximum Potential Positive Environmental Impact
Corn Ethanol	As of 2009, 180 corn ethanol facilities were operating in the U.S., mostly in the Midwest. Future corn ethanol production is expected to expand in the same region.	Water Quality	Effluent with high biological oxygen demand (BOD); Dried Distillers Grain (DDG) byproduct fed to livestock with inadequate waste management practices; under-ground storage tanks (UST) leak.	Effluent effectively treated for BOD; DDG-fed livestock waste incorporated into comprehensive nutrient management plan; USTs do not leak.
		Water Quantity	3-6 gallons of water required per gallon of ethanol produced.	Improvement in water use efficiency and recycling.
		Air Quality	Ethanol facility powered by coal.	Ethanol facility powered by natural gas.
Soybean Biodiesel	In 2009, 191 biodiesel facilities were operating in the U.S., many producing under their capacity.	Water Quality	Effluent with high BOD, total suspended solids (TSS) and glycerin content.	Effluent effectively treated for BOD, TSS and glycerin.
		Water Quantity	<1 gallon of water required per gallon of biodiesel produced.	<1 gallon of water required per gallon of biodiesel produced.
		Air Quality	Biodiesel facility powered by coal.	Biodiesel facility power by natural gas.

Table C-2: Basis for Figure 6-2 (Maximum Potential Range of Environmental Impacts [on a Per Unit Volume Basis] Resulting from Ethanol and Biodiesel Production, Transport, Storage and Use)

	Background	Impact Category	Conditions for Maximum Potential Negative Environmental Impact	Conditions for Maximum Potential Positive Environmental Impact
Cellulosic Ethanol	As of 2009, there were no commercially operating cellulosic ethanol facilities in the U.S. There is uncertainty about when and where the first facilities will start producing.	Water Quality	Effluent possibly with high BOD; USTs leak.	Effluent effectively treated for BOD; USTs do not leak.
		Water Quantity	10 gallons of water required per gallon of cellulosic ethanol.	Improvement in water use efficiency and recycling.
		Air Quality	Ethanol facility powered by coal.	Ethanol facility powered by natural gas.

11

Table C-3: Description of Scenarios in Figure 7-3 (Cumulative Domestic Environmental Impacts of All Steps in the Biofuel Supply Chain System under Three Scenarios in 2022)

Feedstock	**Scenario A** 2022 RFS2-projected feedstock mix produced with conservation/best management practices (BMPs) and efficient technologies	**Scenario B** 2022 RFS2-projected feedstock mix produced with minimal conservation/BMPs and current technologies	**Scenario C** 2022 conventional feedstock mix (corn starch, corn stover, and soybean) produced with minimal conservation/BMPs and current technologies
Conventional Ethanol			
	15 BG		
Corn Starch	• No decrease in crop rotation with soybeans.	• Increased use of continuous corn production and reduction in crop rotation.	
	• Increases in grain yield primarily due to breeding new varieties that also require fewer production inputs, including fertilizer, pesticides, and irrigation.	• Increased grain yields using greater production inputs, including fertilizer, pesticides, and irrigation.	
	• No conversion of marginal lands to corn production.	• Conversion of marginal land to corn production, including in areas that require irrigation.	
	• Increased use of conservation practices, including conservation tillage, nutrient management, and efficient irrigation delivery.	• No increases in conservation practices.	
	• Increased fuel conversion efficiency and reductions in fuel production inputs, including fresh water.	• No increases in fuel conversion efficiency nor reductions in fuel production inputs.	

Table C-3: Description of Scenarios in Figure 7-3 (Cumulative Domestic Environmental Impacts of All Steps in the Biofuel Supply Chain System under Three Scenarios in 2022)

<table>
<tr><th>Feedstock</th><th>Scenario A
2022 RFS2-projected feedstock mix produced with conservation/best management practices (BMPs) and efficient technologies</th><th>Scenario B
2022 RFS2-projected feedstock mix produced with minimal conservation/BMPs and current technologies</th><th>Scenario C
2022 conventional feedstock mix (corn starch, corn stover, and soybean) produced with minimal conservation/BMPs and current technologies</th></tr>
<tr><td colspan="4">Cellulosic Ethanol</td></tr>
<tr><td rowspan="3">Corn Stover</td><td colspan="2">4.9 BG</td><td>16 BG</td></tr>
<tr><td rowspan="2">• Stover harvest limited to acreage with low erosion potential, or erosion is mitigated with conservation tillage.
• Stover harvested with single-pass harvester.
• Increased use of conservation practices.</td><td>• Stover harvested on acreage regardless of erosion potential.</td><td></td></tr>
<tr><td colspan="2">• Stover harvested with multi-pass harvester.
• No increase in use of conservation practices.</td></tr>
<tr><td rowspan="2">Woody Biomass</td><td colspan="2">0.1 BG</td><td>0 BG</td></tr>
<tr><td>• Biomass produced via light-to-moderate forest thinning or from short-rotation woody crops cultivated with long planting intervals on non-federal land currently in managed forest with short planting intervals.</td><td>• Short-rotation woody crops with short planting intervals and high chemical inputs cultivated on non-federal land currently in mature, managed forest plantations.</td><td></td></tr>
<tr><td rowspan="2">Perennial Grass</td><td colspan="2">7.9 BG from dedicated energy crops</td><td>0 BG</td></tr>
<tr><td>• Switchgrass production area limited to east of Rocky Mountains.
• Conversion of land currently in row crop production to switchgrass production.
• Conversion of low diversity, marginal land to conservation managed switchgrass production.
• Increased fuel conversion efficiency and reductions in fuel production inputs, including fresh water.</td><td>• Cellulosic feedstock (switchgrass or Miscanthus) produced on marginal land requiring high production inputs, including fertilizer, pesticides, and irrigation.
• No increases in fuel conversion efficiency or reductions in fuel production inputs, including fresh water.</td><td></td></tr>
</table>

Table C-3: Description of Scenarios in Figure 7-3 (Cumulative Domestic Environmental Impacts of All Steps in the Biofuel Supply Chain System under Three Scenarios in 2022)

Feedstock	**Scenario A** 2022 RFS2-projected feedstock mix produced with conservation/best management practices (BMPs) and efficient technologies	**Scenario B** 2022 RFS2-projected feedstock mix produced with minimal conservation/BMPs and current technologies	**Scenario C** 2022 conventional feedstock mix (corn starch, corn stover, and soybean) produced with minimal conservation/BMPs and current technologies
Biomass-based Diesel			
Soybean	**0.66 BG**		**1 BG**
	• Increases in yield primarily due to breeding new varieties that also require fewer production inputs, including fertilizer, pesticides, and irrigation. • No conversion of marginal lands to soybean production. • Increased use of conservation practices, including conservation tillage, nutrient management, and efficient irrigation delivery.	• Increases in yield primarily due to higher production inputs, including fertilizer, pesticides, and irrigation. • Conversion of marginal lands to soybean production, including areas requiring irrigation. • No increase in use of conservation practices (i.e., conservation tillage, nutrient management, and efficient irrigation delivery).	
Algae	**0.1 BG**		**0 BG**
	• Algae production co-located with stationary carbon dioxide source on marginal land. • Nutrient inputs from wastewater or other waste sources. • Closed bioreactors co-located with publicly owned treatment works and other wastewater treatment facilities.	• Algae produced on land converted from natural cover. • Large nutrient inputs from non-waste sources. • Open pond production using fresh water.	

APPENDIX D:

CONCEPTUAL MODELS

As described in this report, the activities associated with cultivation of biofuel feedstocks and their conversion to fuel result in a complex set of inter-related environmental impacts. Conceptual models provide a useful tool to describe, understand, and communicate the complex pathways by which these activities lead to impacts. As noted in Chapter 7, EPA anticipates developing and using conceptual models as an important tool for the assessment in its 2013 Report to Congress. The conceptual models presented in this appendix lay a foundation for this future effort. Figures D-1 to D-7 present conceptual models for feedstock cultivation and harvest. Figures D-8 and D-9 present models for biofuel production and distribution. (Note that models are not included for end use of biofuel.) These early renditions graphically present the environmental effects most commonly identified in current peer-reviewed literature and, while comprehensive, do not attempt to include all possible effects.

Terms and Abbreviations Used in the Conceptual Models

From the Legend

- biotic response- Response of living parts of terrestrial or aquatic ecosystems, either in terms of number of species or numbers of individuals of a particular species
- ecosystem service- Direct or indirect contribution of the environment to human well-being
- environmental parameter- A measureable attribute of the environment

From the Diagrams

- aquatic life use support- A beneficial use designation in which the water body provides suitable habitat for survival and reproduction of desirable fish, shellfish, and other aquatic organisms (this is a synthetic quality made up of many different environmental parameters)
- BOD- biological oxygen demand
- contamination- Release of nutrients or pesticides used in feedstock production to waterways or bodies of water
- PM – particulate matter
- T & E species- threatened and endangered species
- VOC – volatile organic compound

Feedstock Production

Figures D-1 to D-7 present seven models for the six feedstocks covered in this report: corn starch; soybean; corn stover; perennial grass; woody biomass (short-rotation woody crops and forest thinning/residue removal); and algae production.

Different pathways are introduced at the top of several of these feedstocks models. These pathways were selected because: (1) they will likely be pursued in combination in order to grow enough feedstock to meet RFS2 2022 biofuel requirements (see Chapter 2 for a description of requirements), and (2) they result in different environmental impacts.

Arrows in the impact boxes (below the initial row of activities) depict whether the impacts are negative or positive. The number(s) by each arrow designate the pathway to which the arrows refer. A few pathways can have both negative and positive impacts (e.g., corn starch cultivation could result in increased or decreased use of ground and surface water). Dotted borders denote impacts that have a relatively large degree of uncertainty due to a lack of information. Dotted boxes without arrows depict highly uncertain impacts that nonetheless are described in the literature.

Fuel Production and Distribution

Figures D-8 and D-9 present conceptual models for production and distribution of the two biofuels covered in this report: ethanol and biodiesel.

Ethanol Production

Figure D-8 shows the activities and impacts associated with production and distribution of ethanol from both starch (i.e., corn grain) and cellulosic feedstocks, including corn stover, perennial grasses, and woody biomass. A single model is provided for these four types of feedstocks because their impacts and associated uncertainty are largely similar, with a few exceptions (e.g., water use will likely be slightly higher for cellulose conversion).

As depicted in the upper left of the Figure D-8, conversion of starch to ethanol consists of several sequential steps, including milling, hydrolysis, and fermentation. There currently are two distinct alternatives for converting cellulosic feedstock into ethanol: (1) biochemical conversion (which is preceded by a catalysis step to separate cellulose and hemicellulose from their tightly bound state with lignin), and (2) thermochemical conversion. These alternatives vary slightly in terms of their chemical processes and by-products. As with Figures D-1 to D-7, a dotted border is used to denote impacts with relatively large uncertainty due to a lack of information.

Biodiesel Production

Figure D-9 shows the activities and impacts associated with production of biodiesel from soybeans and algae. Several techniques may be used to convert plant oils into biodiesel, including hydrogenation, catalytic cracking, and transesterification. All these processes produce biodiesel, with glycerin as a by-product.

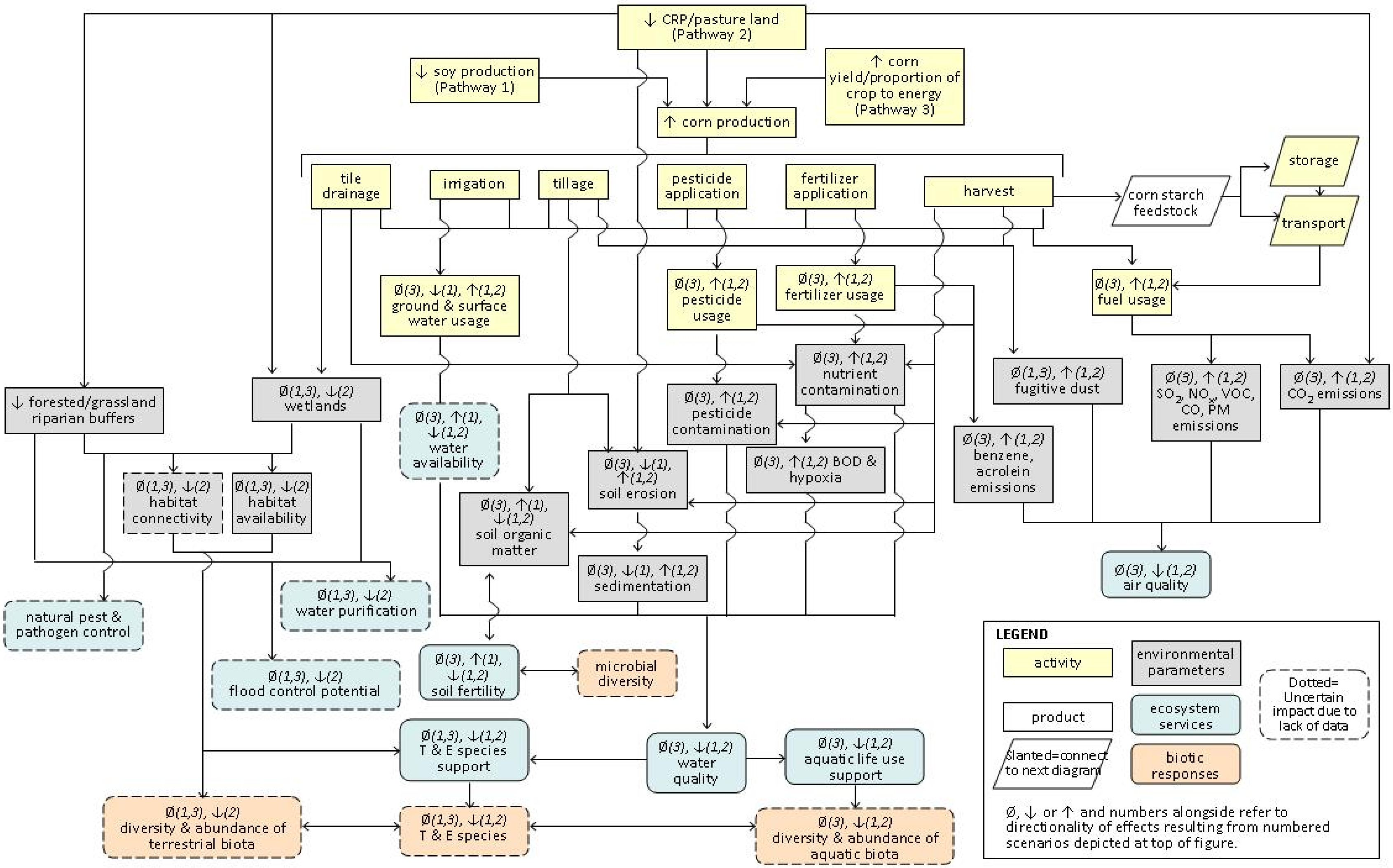


Figure D-1: Pathways for Potential Environmental Impacts of Corn Starch Feedstock Cultivation

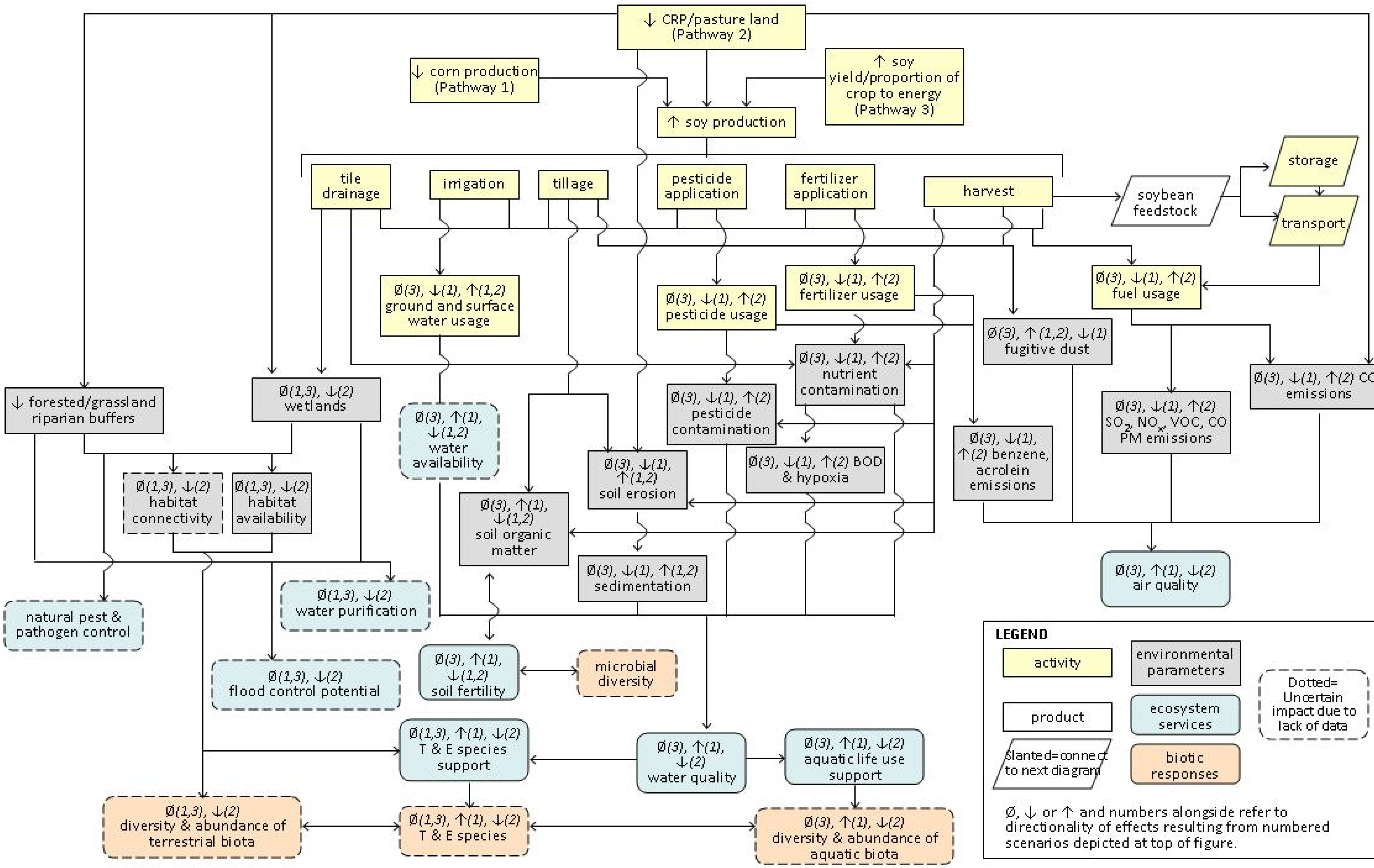


Figure D-2: Pathways for Potential Environmental Impacts of Soybean Feedstock Cultivation

.

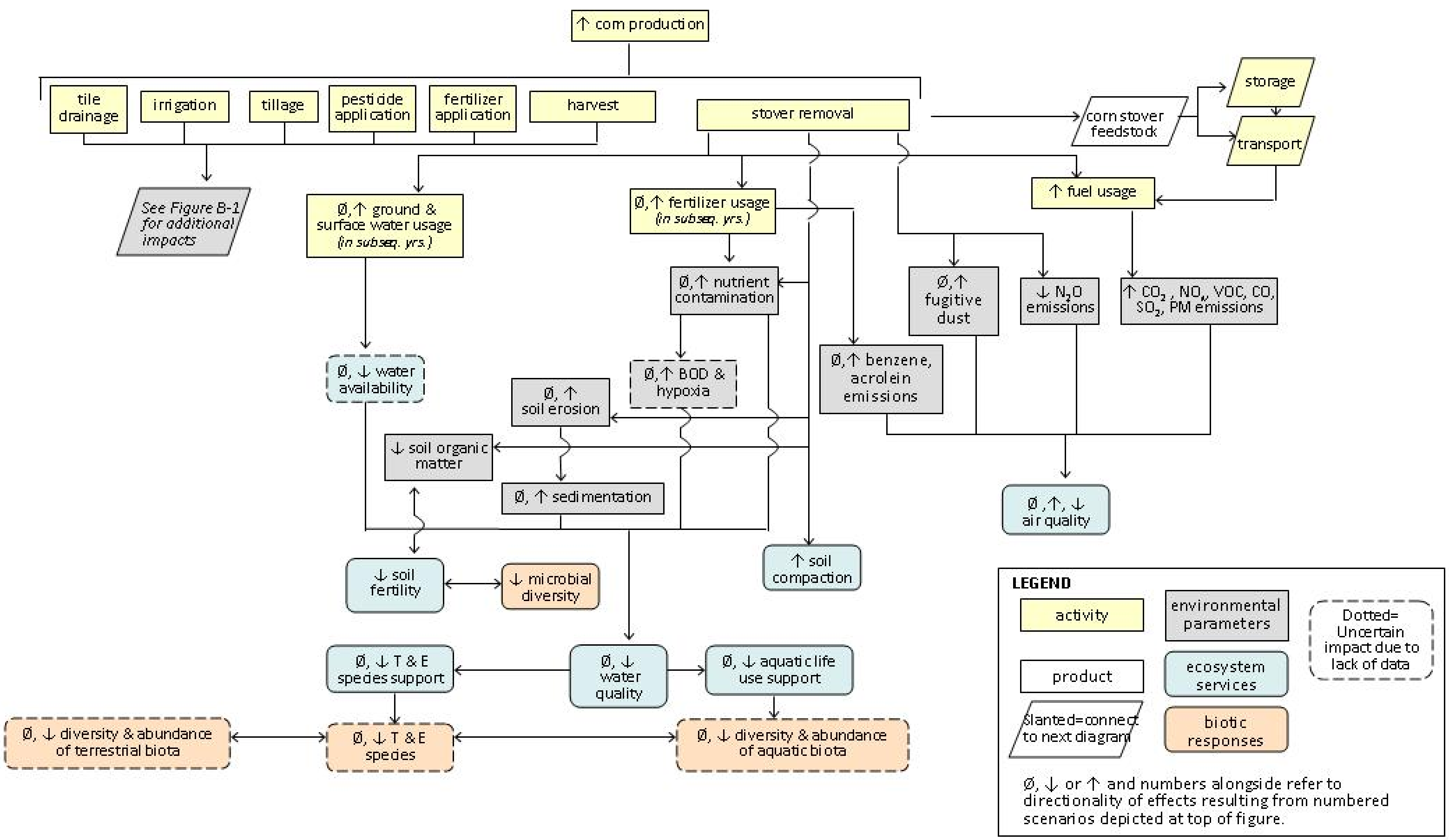


Figure D-3: Pathways for Potential Environmental Impacts of Corn Stover Feedstock Cultivation*

***Corn stover is a waste product of corn starch cultivation. The impacts of corn cultivation are shown in Figure D-1. Figure D-3 highlights the environmental impacts of stover removal *above and beyond* those impacts attributable to corn grain production.**

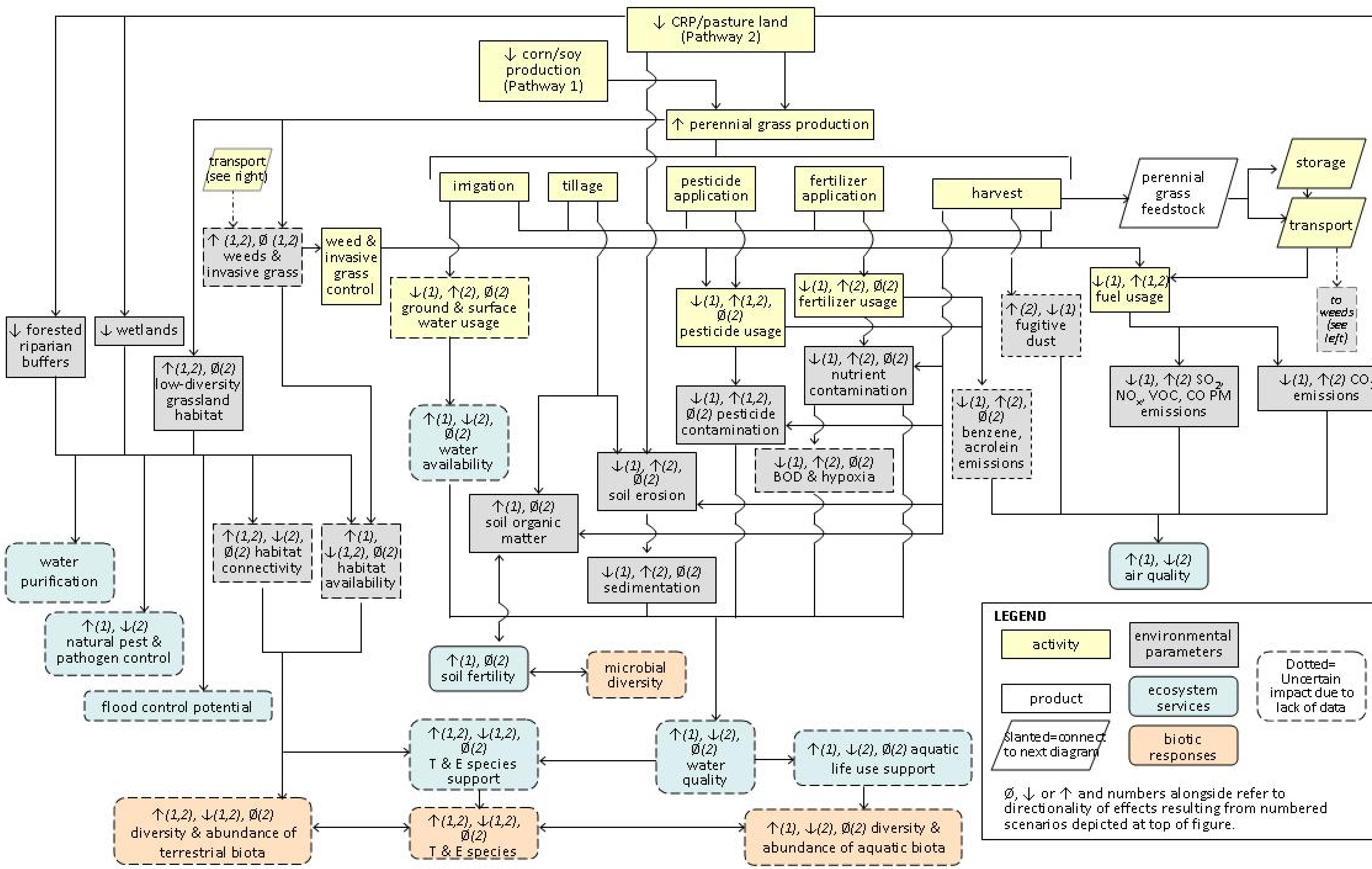


Figure D-4: Pathways for Potential Environmental Impacts of Perennial Grass Feedstock Cultivation

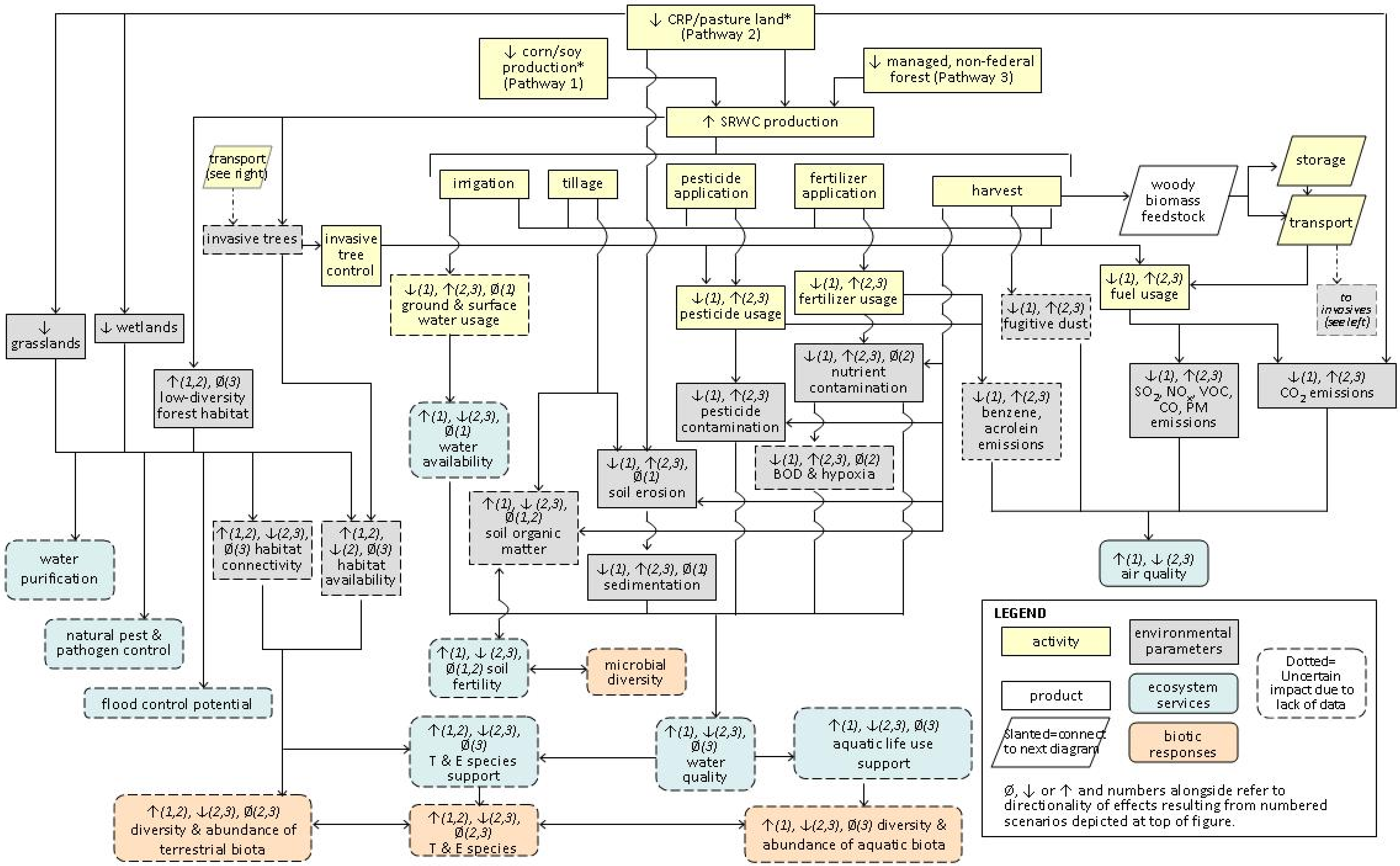
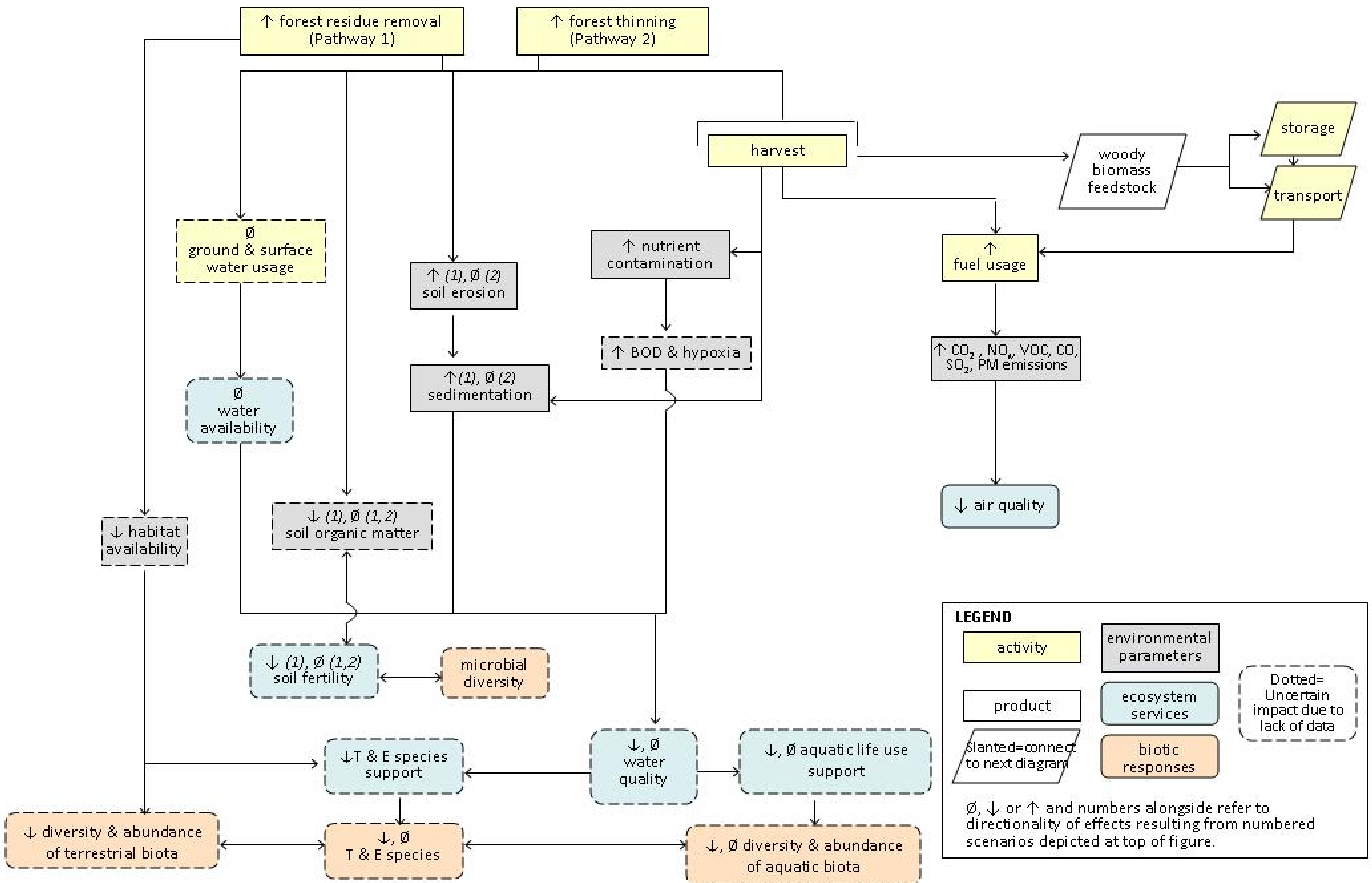
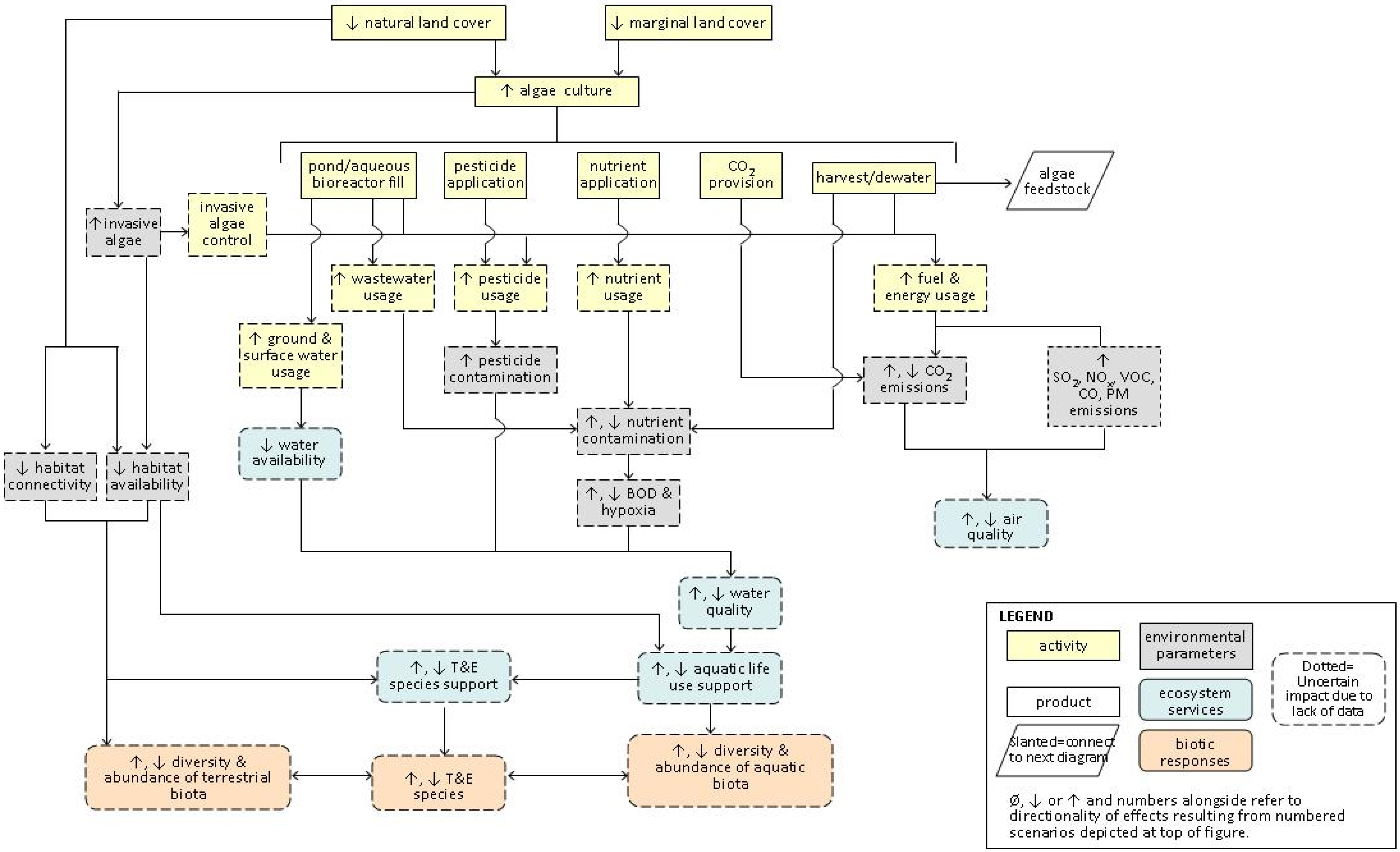


Figure D-5: Pathways for Potential Environmental Impacts of Short-Rotation Woody Crop Feedstock Cultivation

***These particular land use changes may not currently be allowable under RFS2.**



77
78
79 **Figure D-6: Pathways for Potential Environmental Impacts of Forest Thinning and Residue Removal**



80
81
82 **Figure D-7: Potential Environmental Impacts of Algae Feedstock Production**

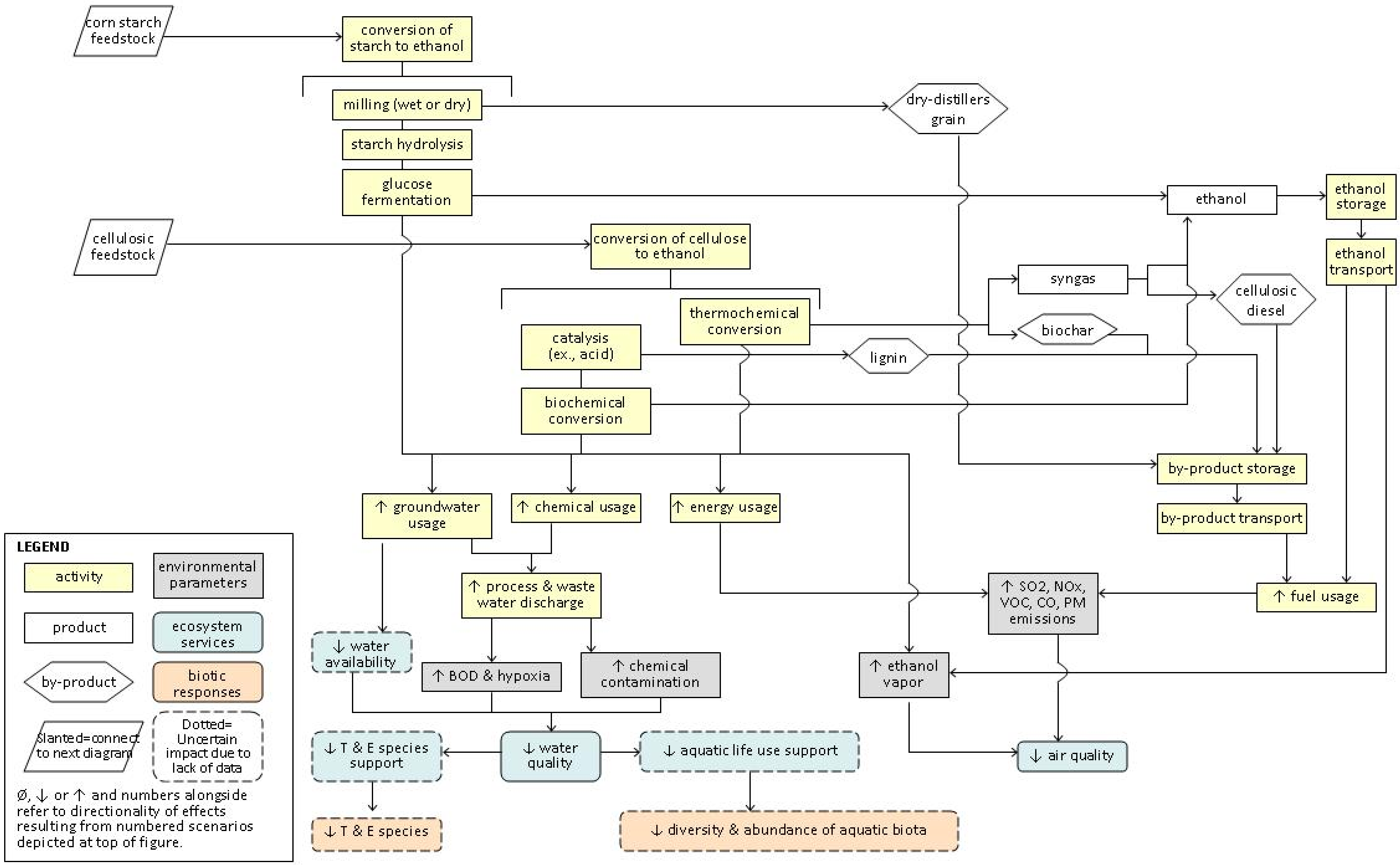


Figure D-8: Potential Environmental Impacts of Producing and Distributing Conventional and Cellulosic Ethanol (Impacts of Fuel Use Not Included)

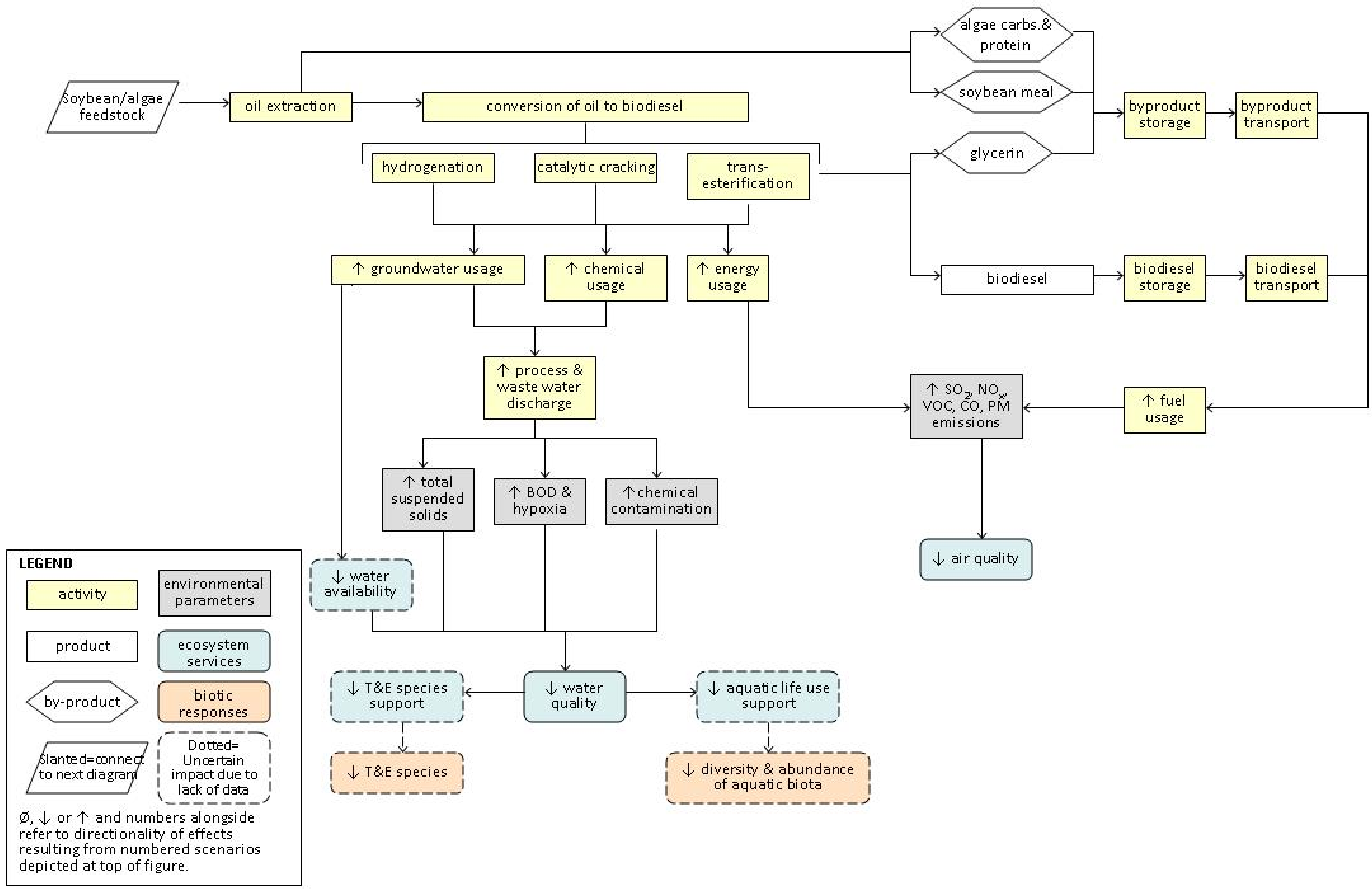


Figure D-9: Potential Environmental Impacts of Producing and Distributing Biodiesel (Impacts of Fuel Use Not Included)