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Impacts of U.S. Biofuels Mandates on Environmental and Energy Security

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IMPACTS OF U.S. BIOFUELS MANDATES

ON ENVIRONMENTAL AND ENERGY SECURITY

A Dissertation Presented

by

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ABSTRACT

ESTIMATING NITROGEN LOAD RESULTING FROM U.S. ETHANOL MANDATES

December 2013

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Congress passed the Energy Policy Act of 2005 and the Energy Independence and Security Act (EISA) of 2007 to reduce dependency on foreign oil by increasing the use of biofuels. EISA mandates 36 billion gallons of biofuel production in 2022, representing an increase of 118% in eight years. Existing and new farmland are expected to be employed to produce corn and other feedstock necessary to fulfill the biofuel mandate. There is little research on the potential environmental impact on water resources of meeting the ESIA biofuel mandates.

The objectives of this EISA study were twofold. First, the study measured the economic and environmental impact of nitrogen runoff on nation-wide water resources to from crop production to meet the EISA 2022 biofuel mandates. Second, the study evaluated the potential effectiveness of EISA 2022 mandates on energy security due to replacing oil with biofuels to meet the energy needs of the transportation sector.

The study used the SPARROW modeling method to estimate nitrogen fluxes in the Mississippi River basins based on production required to meet the EISA 2022 biofuels mandate. The scenarios results show that biofuel production can result an increase of nitrogen flux to the northern Gulf of Mexico from 270 to 1742 thousand metric tons, that is an increase from 21% to more than 100% from the total nitrogen flux estimated by the EPA, (2011). Using all cellulosic (hay) ethanol or biodiesel to meet the 2022 mandate is expected to significantly reduce nitrogen flux however it requires approximately 25% more land than the land needed in EISA specified 2022 scenario. The estimated environmental economic cost of producing 36 billion gallons of ethanol using corn feedstock to meet the EISA 2022 mandate is estimated to be $23 billion annually. Of that cost, more than 80% is due to the effect on human health.

One of the main objectives of EISA is to promote energy independence by mandating that U.S. transportation fuel contain domestically produced biofuels, mainly ethanol. However, basic supply and demand forces make the probability remote that the EISA 2022 mandated biofuel production of 36 billion can be used domestically. If the U.S. policy objective were to replace imported oil with domestic sources of energy promote the use of domestic fuel, then it would be more efficient to use locally produced fossil fuels such as natural gas to power motor vehicles. On the other hand, if the objective were to reduce the consumption of fossil fuels, then improving the efficiency of cars engines would yield better results. While is it unlikely that the U.S will stop producing ethanol (Ethanol is used as MTBE replacement), investing in higher ethanol blends does not seem to be an efficient solution to energy security.

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CHAPTER 1

INTRODUCTION

The growing concern over high crude oil prices and the U.S dependency on foreign oil led the U.S Congress to pass the Energy Policy Act of 2005 and Energy Independence and Security Act (EISA) of 2007. Both acts provide economic incentives to use renewable fuels, mainly ethanol, to meet the energy demands in the transportation sector and reduce dependency on foreign oil. The EISA set a mandatory renewable fuel standard requiring that at least 36 billion gallons of biofuels be used by 2022 as transportation fuel (Table 1). Since the majority of ethanol is produced from corn and other feedstock, it is expected that the agricultural sector will respond by increasing production (USDA, 2011).

Energy independence is one of the driving forces behind the U.S government support for the biofuels industry. The aim is to achieve a diverse energy portfolio that meets the county's energy demand with domestically produced renewable fuels. The Renewable Fuels Association (RFA) claims that in 2011, the ethanol industry produced 13.9 billion gallons of ethanol that replaced 485 million barrels of imported oil[1] (Renewable Fuels Association, 2012), however the RFA did not specify how only

[1] One gallon of ethanol contains 33% less energy than gasoline (EIA, 2013) and one barrel produces 19 gallons of motor gasoline.

imported oil was displaced rather than crude oil in general. In addition, the RFA claims that ethanol production supported more than 400,000 jobs, generated $42.4 billion dollars in GDP, and $29.9 billion in household income (Renewable Fuels Association, 2012). Since the majority of ethanol in the U.S. sold as gasoline additive, some argue that the lower price of blended fuels due to the subsidy will lower the price of gasoline thus increasing the demand for fuels which may offset the environmental (increased emissions) and fuel security benefits (increased fossil fuel demand) (Vedenov & Wetzstein, 2008).

Biofuels account for a small portion of the total U.S. energy sources but have significant environmental impact. Increased biofuels usage will increase nonpoint source pollution (e.g. fertilizer and pesticides) runoff to waterbodies as well as other less direct environmental impact (Dias De Oliveira *et al.*, 2005; Pimentel & Patzek, 2005; G. P. Robertson et al., 2011). The literature includes studies on the impact of biofuels on air quality (Gaffney et al., 2001; Graham et al., 2008; Poulopoulos et al., 2002); land use (Donner et al., 2004; Searchinger et al., 2008; Secchi & Babcock, 2007); and net energy value (Dias De Oliveira et al., 2005; Farrell et al., 2006; Groode, 2006; Lavigne & Powers, 2007; Pimentel & Patzek, 2005; Shapouri et al., 2008). However, but little attention has been paid to the impact of incremental biofuel use on water resources.

Reconciling the contradictory goals of energy independence and environmental sustainability is difficult due to the complex interactions among different goals. The purpose of EISA is to promote energy independence, improve environmental quality, and protect the consumer from the volatility of crude oil prices. Many studies (Luchansky & Monks, 2009; Rask, 1998; Solomon et al., 2007) have attempted to measure the impact of increased demand for ethanol on the environment, economy, and energy security; however, most studies only consider air quality as a measure of environmental quality and fail to assess the impacts of nitrogen runoff associated with the production of corn and other biomass. As the U.S is planning to increase ethanol production based on EISA mandates, it is important to understand the environmental implications behind such expansion to the agricultural sector. The biofuels mandates aim at reducing the greenhouse gases (GHGs) and increase energy security, but designing energy policies about only GHGs or energy security could lead to an increase nitrogen runoff. Currently the U.S is the leading producer of ethanol in the world, using corn for ethanol because of its low conversion cost, high yields, and availability. However, recognizing the economic and environmental impacts of producing ethanol from feedstock such as corn, EISA has capped the corn ethanol to 15 billion gallons by year 2015. EISA mandates that future ethanol must be produced from "advanced" or cellulosic sources such as switch grass and corn stover, and some biodiesel.

Table 1

Energy Security and Independence Act Ethanol Mandates *(bil. gallons)*

Year	Corn Ethanol	Cellulosic Ethanol	Other Sources i.e. biodiesel	Total
2011	12.6	0.25	1.10	13.95
2012	13.2	0.50	1.50	15.20
2013	13.8	1.00	1.75	16.55
2014	14.4	1.75	2.00	18.15
2015	15.0	3.00	2.50	20.50
2016	15.0	4.25	3.00	22.25
2017	15.0	5.50	3.50	24.00
2018	15.0	7.00	4.00	26.00

2019	15.0	8.50	4.50	28.00
2020	15.0	10.50	4.50	30.00
2021	15.0	13.50	4.50	33.00
2022	15.0	16.00	5.00	36.00

Source: (EISA, 2007).

Brief History of Biofuels

Biofuels are transportation fuels made from biomass; the most common classifications of liquid biofuels are biodiesel and ethanol (EIA, 2010). Biodiesel is essentially mono-alkyl ester-based oxygenated fuel made from renewable feedstock such as soybean oil, vegetable oils, and animal fats, or biomass such as microalgae (Petrou & Pappis, 2009; Puppan, 2002). Ethanol, on the other hand, is made by fermenting and distilling sugar-containing crops such as corn, wheat and sugar beets, sugar care, or lignocelluloses from corn or cotton stalks. Both biodiesel and ethanol can be used as a fuel additive or as a pure fuel. The blended gasoline E10 or “gasohol” (10% ethanol by volume) is sold in most gas stations in the U.S to improve fuel combustion and decrease carbon monoxide emission (EIA, 2007). Blends containing 75% to 85% ethanol by volume are referred to as E85 and these fuels have a higher octane than gasoline, but less energy per gallon (Puppan, 2002).

Ethanol was used as an automotive fuel in as early as the 1800s, when Henry Ford first introduced his quadricycle that ran on pure ethanol (EIA, 2008). The supply for ethanol decreased dramatically during WWI and WWII, however the interest in ethanol as an automotive fuel was diminished as leaded gasoline became the fuel of choice in the U.S due to low cost of production and new oil discoveries (Kovarik, 1998). Ethanol did not develop as a significant part of the U.S. energy portfolio until the 1970’s (Kovarik, 1998; EIA, 2008). Demand for petroleum peaked in 1970s largely as a result of the Arab oil embargo. Domestic production could not make up for the OPEC (Organization of Petroleum Exporting Countries) export embargo. Several federal and local policies were passed to provide incentives to develop ethanol fuels as an marginal substitute for imported oil as a means to enhance national energy and environmental security. In 1978, the U.S Congress passed the Energy Tax Act, the first major federal act supporting the ethanol industry (EIA, 2008). The Act exempted fuels blended with at least 10% ethanol from the $0.40 per gallon excise tax (EIA, 2008). The Energy Security Act of 1980 offered several incentives such as up to 1 million dollars in federal low interest loans to ethanol producers to cover construction cost, price guarantees, and purchase agreements (EIA, 2008). The Omnibus Recombination Act placed a tariff on imported ethanol to protect domestic producers from foreign competition (EIA, 2008). Ethanol producers received additional government support when Congress passed the Gasahol Competition Act banning retaliation against ethanol resellers by the major oil companies (EIA, 2008).

The demand for ethanol was further increased when the Clean Air Act of the 1990 mandated a seasonal use of oxygenated automotive fuels, such as ethanol, in some regions of the U.S and year round use in regions where the air quality was a matter of concern (EIA, 2008). States initially used Methyl *tert*-butyl ether (MTBE) as an oxygenate (organic compound added to gasoline to increase combustion efficiency) but rising concerns over the environmental and health impact of MTBE lead some states to ban its use and replaced it with ethanol (Vedenov & Wetzstein, 2008). Another notable piece of legislation in 1990 is the Energy Policy Act that extended the tax breaks of 1978 to blends containing less than 10% ethanol (EIA, 2008). In short, by 1990 the U.S. was supporting the ethanol industry by a broad-ranging combination of both monetary incentives and regulatory mandates.

Additional support came when the Energy Policy Act of 2005 mandated the use of four billion gallons of renewable fuels in 2006, raising the amount to 7.5 billion in 2012 (EPA, 2005). The policy also ensured that fuel blends get a tax credit of $0.51 per gallon of ethanol blended with gasoline. The most recent governmental support for biofuels was through the Energy Independence and Security Act of 2007. The purpose of this policy is to ensure greater energy security and independence through increasing the production of biofuels, increased efficiency of vehicles and buildings, and research on greenhouse gas capture and storage. The Renewable Fuel Program was expanded to require 36 billion gallons of renewable fuels by 2022 where 15 billion gallons must come from conventional biofuels as ethanol (See, Table 1) (EISA, 2007).

These incentives and mandates (and other market influences) have directly, and dramatically, changed the ethanol industry in the U.S. The amount of ethanol consumption in the U.S rose dramatically from an average of 1,387 million gallons in 1998 to around 12,946 million barrels by 2012 (EIA, 2013b). Several studies suggest that the major factors that caused such an increase were high crude oil prices, government incentives programs, and replacing MTBE with ethanol as a fuel additive (Luchansky & Monks, 2009; Rask, 1998; Solomon *et al.*, 2007). In general, government's support the private sector using incentive tools such as tax breaks (subsidies) to encourage the domestic industry and protect it from foreign competition. Subsidies reduce the marginal cost of production to the firm, which in return increase the supply and quantity demanded.

In summary, the use of biofuels in the United States has been justified through energy security and air quality acts. For the past 30 years, various bills and acts that support the corn and ethanol industries have been introduced and passed by the U.S. government. These industries first gained support in 1978 to use ethanol as an alternative fuel, followed by tax tariffs on imported oil in the 1980s. The justification to support the corn and ethanol industries was shifted from energy security to air quality in the 1990's through the passing of the Clean Air Act banned the use of MTBE and recommend ethanol as a replacement. The industries gained further support during the past decade by creating an ethanol mandates and providing economic incentives to ethanol producer. With a strong focus on the contribution of corn to energy production, the influence of these governmental energy policies should be assessed in terms of their impact on food commodity pricing.

Rising crude oil prices, pollution, and concerns over fossil fuels depletion made biofuels an attractive alternative to fossil fuels. Biofuels appeared as the perfect solution for the U.S' energy security, environmental goals, and support for the local economy. One of the reasons the concept of biofuels gained public and policy support because it seemed "intuitive". Homegrown plants can be converted into transportation fuels and should provide environmental gains (e.g., CO_2 reductions) when compared to fossil fuels. The energy and biofuels policies discussed above have pushed to support biofuels even though the actual energy, environmental, and economic benefits of biofuels remain debatable. The argument made was that imported oil made the U.S' energy security more vulnerable to imported crude oil price volatility and supply interruption, and that ethanol production locally would reduce the reliance on imported oil and thus improve energy security. While having a diverse energy portfolio is vital in reducing the risks of relaying on a single source of energy, the underlying assumption that biofuels are more "secure" than imported crude oil must be more systemically assessed.

Biofuels are promoted as a part of the solution to increased energy security, but there is little analysis on the reliability of their supply. There is even less discussion about what energy security is and how biofuels are to increase it. Moreover, the terms "security" and “independence” are often used interchangeably. Nevertheless, concerns about energy security fall into two categories: heavy reliance on imported crude oil; impacts of oil supply and price shocks on the economy. The assumption behind supporting biofuels is that they are more "secure" to use than imported oil because they are locally produced thus their supply is reliable, their prices are less volatile, and there are more environmentally friendly than fossil fuels (Kraemer, 2006).

Mandate Implementation

Congress established the Renewable Fuel Standard (RFS) with the enactment of the Energy Independence and Security Act of 2007. The RFSs are a mandatory minimum volume of biofuels to be used as a fuel additive or a substitute in the transportation fuels sold in the U.S. The biofuels industry is supported by government through providing a mandatory biofuel market, where fuel refineries and blenders must mix a minimum volume of biofuels in the gasoline they sell in the U.S. The RFS is administered by the EPA, which is tasked to ensure that the transportation fuels sold in the U.S contains the mandated volume of biofuels (EPA, 2013)

The EPA first estimates the expected volume of transportation fuel to be used in an upcoming year, and then determines the annual biofuel mandate as a percentage of total of the expected fuel consumptions. The EPA then obligates fuel suppliers to blend a specified volume of biofuels equals to a percentage of their annual fuel sales. Fuel suppliers must show that they have sold the quantity of biofuels assigned to them by acquiring Renewable Identification Numbers (RINs), and failure to meet the requirements is subject to civil penalties and fines (EPA, 2013). Suppliers who sell a surplus of biofuels can sell the extra RINs credits to another fuel suppliers or bank it for future use (EPA, 2013). As federal law requires refiners and blenders to increase the amounts of biofuels each year, however, refiners and blenders are close to facing what is referred to as a “blend wall”. The RFS requirements exceed the amount of biofuels that can be blended in the transportation fuel because the blending limit is 10% per volume for ethanol gallon of transportation fuel. As a result, in 2011 the EPA has approved blends as much as 15% to be sold in the US for vehicles made in 2001 and newer (EPA, 2013). Refiners and blenders can also sell E85, however, there is limited number flex-fuel vehicles (FFVs) and limited pumping stations.

EISA mandates are based on future expectations of motor gasoline demand; however, the actual demand for motor gasoline was less than expected. Thus, the volume of ethanol produced and consumed was less than the mandated ethanol (Figure 1) and as a result, refineries had to export the surplus ethanol. Based on the annual energy outlook 2013 projections (EIA, 2013), the demand for gasoline in expected to decrease over the next decade (Table 2). Based on these projections, the demand for ethanol in transportation was calculated assuming 10% and 15% by volume of motor gasoline. The results in Table 2 indicate that the increasing ethanol mandates will not be met by using ethanol as gasoline additive (E10 and E15), thus the surplus ethanol must be sold at E85 in order to meet the mandates. As mentioned above, however, the consumption of E85 remains a small fraction of the total gasoline consumption (Table 3).

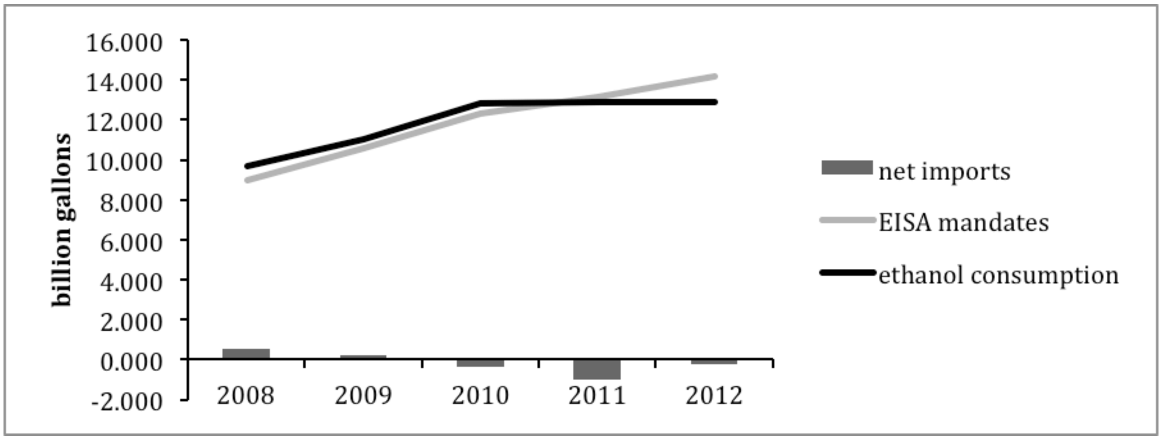


Figure 1: EISA 2007 mandates and actual consumption (EIA, 2012).

Table 2
Blending Wall Projections (Bil Gal.)

Year	EISA Ethanol Mandate	Gasoline Consumption *	Ethanol consumption (E10)	Ethanol consumption (E15)
2015	20.50	121.07	12.11	18.16
2016	22.25	120.34	12.03	18.05
2017	24.00	119.60	11.96	17.94
2018	26.00	118.82	11.88	17.82
2019	28.00	117.92	11.79	17.69
2020	30.00	117.05	11.71	17.56
2021	33.00	115.58	11.56	17.34
2022	36.00	113.92	11.39	17.09

* Conversation from EIA estimates is based on gasoline heat value 127,143 btu/gal of gal.

Table 3

Annual Consumption of E85 and Gasoline (thousand gallon of gasoline equivalent)

Year	E85	Finished Motor Gasoline
2000	12,388	3,272,563
2001	15,007	3,964,429
2002	18,250	4,821,139
2003	26,376	6,967,801
2004	31,581	8,342,816
2005	38,074	10,058,085
2006	44,041	11,634,399
2007	54,091	14,289,328
2008	62,464	16,501,240
2009	71,213	18,812,481
2010	90,323	23,860,808
2011	862,679	227,895,637

Objectives

The objective of Chapter 2, "Estimating Nitrogen Load Resulting from Biofuel Mandates", is to estimate nitrogen load resulting from the increased energy crops production used to meet the EISA mandates. The study utilizes SPARROW (Spatially Referenced Regressions On Watershed attributes) developed by the USGS for watershed modeling (Smith *et al.*, 1997). The approach is used to answer the two important questions; 1) What is the contribution of biofuels to nitrogen loading in the Mississippi River watershed?; and 2) Does switching to cellulosic ethanol drastically change the total nitrogen loading when compared to corn ethanol?

The objective of Chapter 3, "Estimating the Economic Cost of Nitrogen Fluxes from Energy Crop Production," is to estimate the total cost of nitrogen externalities (atmospheric and water) associated with meeting the EISA mandate and compare it hypothetical scenarios such as all corn, all cellulosic, or all biodiesel mandates using the cost-effectiveness approach.

The objective of Chapter 4, "Contribution of Biofuel Mandates to Energy Security," is to assess the effectiveness of EISA mandates in achieving energy security in the transportation sector by attempting to answer the following questions; (1) What is energy security and how is it measured?; and (2) How reliable is ethanol price and supply compared to imported oil?

CHAPTER 2

ESTIMATING NITROGEN LOAD RESULTING FROM BIOFUEL MANDATES

The Mississippi river basin drains approximately 40% of the United States, and is it the largest contributor of nutrients to the northern Gulf of Mexico (EPA, 2007; EPA, 2009; EPA, 2011). Nutrients can come from many sources such as discharge from sewage plants, atmospheric nitrogen deposition, agriculture, and urban development. However, excess fertilizers from agricultural fields are the largest source of nutrient runoff to the Mississippi river (Gowda *et al.*, 2007). Farmers use drainage tile systems to enhance crop yield by removing excess water from the rooting zone, the effluent carries large amounts of nitrate (fertilizer) from agricultural land to surrounding waterbodies (Gowda *et al.*, 2007). The average annual total nitrogen flux to the northern Gulf of Mexico from 2005 to 2009 was 1.26 million metric tons (EPA, 2011). The rate and the concentration of nitrate losses in drainage tiles vary with soil type, climate conditions, fertilizer application rate and timing, drain spacing, cover crops, crop yield and water table control practices (Kladivko *et al.*, 2004).

Agricultural water pollution is categorized as nonpoint source pollution because runoff is not generated from a single point. Runoff leaves agricultural fields in many places and it mixes with runoff from other fields, which makes monitoring difficult and expensive. Excess nutrient runoff can lead to eutrophication of water bodies and can cause hypoxic conditions (dissolved oxygen <2 mg/L). The hypoxia zone in the northern Gulf of Mexico occurs annually due to the overgrowth and decomposition of organic matter, affecting the natural functions of the ecosystem and threatening commercial and recreational gulf fisheries valued at $2.8 billion annually (EPA, 2011). The size of the hypoxia zone varies annually (long term average is 13,825 km^2), thus the Mississippi River/Gulf of Mexico Watershed Nutrient Task Force established a goal to reduce the size of the hypoxia zone to 5000 km^2 by 2015. However, there is little evidence of progress in reducing the size of the hypoxic zone as the current five-year average (2006-2010) is 17,300 km^2 (EPA, 2011).

One of the growing concerns of increased demand for biofuels from feedstock such as corn and soybeans is that it will result in an increase of nitrogen flux (amount of nitrogen leaving the edge of a field). The 2007 EISA mandates the production of 36 billion gallons of biofuels by 2022 of which 15 billion gallons is corn ethanol and 21 billion gallons is from sources other than corn, such as cellulosic sources and biodiesel. To meet the mandates, the agricultural sector is expected to increase corn production by increased chemical and fertilizer applications on the fields, and less crop rotation with soybeans (Simpson *et al.*, 2008). Shifting the crop rotation from the conventional corn-soybean to continuous corn would significantly increase nitrate-N runoff to surface water because continuous corn systems require a large fertilizer input (Thomas, 2009).

A study by Costello *et al.*, (2009) to assess the impact of biofuel crop production on the formation of hypoxia in the Gulf of Mexico found that meeting the 2022 biofuel mandate will increase the average nitrate-N output by 300,000 to 750,000 metric tons depending on biomass sources used for fuel. This represents approximately 23.8% to

60% of the average annual total nitrogen flux to the northern Gulf of Mexico from 2005 to 2009. Even though scenarios where ethanol is produced using only cellulosic crops had lower average NO_3 fluxes than those of corn for production of ethanol, the range of nitrate fluxes for all scenarios highly overlap, which indicate the uncertainty in achieving any specific scenario (Costello *et al.*, 2009). Finally, the nitrate decrease in any modeled scenario was found to be insufficient to reduce the size of the hypoxia zone below the EPA's 5000 km^2 target (Costello *et al.*, 2009). Another study Donner & Kucharik (2008) show that the increase in corn production to meet the ethanol targets by 2022 would increase the annual flux of dissolved inorganic nitrogen to the Mississippi and Atchafalaya Rivers by 10–34%. It would also increase the likelihood that annual dissolved inorganic nitrogen fluxes exceeds the target for reducing hypoxia in the Gulf of Mexico to more than 95% (Donner & Kucharik, 2008).

Farmers can increase corn production by not only increasing fertilizer application, but also increase corn acreage. Secchi et al., (2011) showed that a 14.4% increase in corn acreage would result in a 5.4% increase in total nitrogen loads and in a 4.1% increase in total phosphorus. The authors also projected that at very high but realistic corn prices, farmers would increase acreage by 57% with a resulting 18.5% increase in nitrogen and 12% increase in phosphorus. Table 4 compares the crops used for biofuels production in terms of fertilization rate, yield, percent of nitrogen runoff, and nitrogen flux. Corn has the highest fertilization rate while soybean fertilization rate is the lowest because soybeans fix atmospheric nitrogen into the soil. Costello *et al.*, (2009) and Miller *et al.*, (2006) have estimated that 24% of nitrogen applied to corn and soybean fields leaves as runoff, while its 13% for switchgrass. Cellulosic feedstock, such as switchgrass, can be a potential replacement for corn because they do not compete with food crops and have less nitrogen runoff than monocrops (crops grown of the same land year after year). Switchgrass and other grasses are grown on non-tilled land and require less nitrogen input (Parrish & Fike, 2005). Another potential source of cellulosic ethanol is corn stover, however, the optimal removal rate is an issue of debate (Hoskinson *et al.*, 2007; Sheehan *et al.*, 2003). In addition, excess removal of corn stover from the fields cans adversely impact water and soil quality. While corn needs large amounts of fertilizer and has high fugitive nitrogen per acre compared to other crops, it is a high yielding feedstock which means more fuel can be produced per area than other crops and corn has a high conversion efficiency.

Table 4

A Summary of Nitrogen Inputs and Outputs for Various Crops

	Fertilizer Application Rate (kg-N/ha)	Yield (bu/acre)	Fertilizer as % Runoff	Nitrogen Flux (kg-N/ha)	Sources
Corn	160	136	24%	38.41	(Miller *et al.*, 2006)
	185	144	n/a	7.13	(Thomas, 2009)
	151	n/a	n/a	5.39	(Thomas, 2009)
	140	n/a	n/a	26	(Powers *et al.*, 2011)
	174	90	24%	41.5	(Costello *et al.*, 2009)
Soybean	6.3	40.5	24%	20.85	(Miller *et al.*, 2006)
	24.0	33.3	24%	5.76	(Pimentel & Patzek, 2005)
	11.9	41.2	24%	n/a	(Costello *et al.*, 2009)
Switchgrass	67.3	n/a	n/a	7 - 9.1	(Powers *et al.*, 2011)
	74	5,100 kg/ha	13%	9.62	(Costello *et al.*, 2009)
	50	10,000 kg/ha	n/a	n/a	(Pimentel & Patzek, 2005)
	74	7,100 kg/ha	n/a	n/a	(Schmer *et al.*, 2008)
	74	5,000 kg/ha	n/a	n/a	(Perrin *et al.*, 2008)
	100	12,000-16,000 kg/ha	n/a	48	(Bai *et al.*, 2010)
Stover	n/a	50% removal	n/a	18.5 - 28.6	(Powers *et al.*, 2011)
	(1)	0.49	24%	n/a	(Costello *et al.*, 2009)
Hay	28	5600 kg N/ha	n/a	n/a	(USDA, 2013)

Notes: (1) Farmers add fertilizer when Stover is removed based on the nitrogen content of Stover (0.58-0.8% Nitrogen per dry mass of residue).

Estimating Total Nitrogen Runoff

Modeling nonpoint source (NPS) pollution from agricultural processes is a difficult task due to the inherent variability and uncertainty in the system. Water pollution models range from simple statistical models (regression) to estimate the parameters of the system, to mechanistic models that require many parameters that are determined a priori. Purely statistical models are simple empirical representations of a physical system and do not required full understanding of the different processes that govern the system; they simply correlate observations watershed properties. The main advantage of statistical models is that they can be used to make predictions about large watersheds based upon little data and inputs. However, they assume uniformity though out the region and do not provide details on fate and transport of different nitrogen species in the system (soil, plant, water) (Schwarz *et al.*, 2009). On the other hand, mechanistic models are complex mass balance equations that simulate the various processes of a system (decay and/or transport of contaminant) based on a set of inputs (hydrology, soil, climate). The mathematical representation of these processes is usually based on a priori knowledge of the system. The complexity of mechanistic models requires intensive data and calibration. Thus, they are normally applied to small-scale regions where the parameters can be determined thought intense field studies prior to building the model. In many occasions data on a large scale is not available to accurately generate results, which limits the use of these models to small regions. Also, these models need calibration in order to fit observed field data, thus without field data the results can be uncertain.

Physical models such as GLEAMS (Knisel, 1993; Leonard *et al.*, 1987) and EPIC (Sharpley & Williams, 1990; Williams *et al.*, 1995) are used to study a specific geographic region, assuming uniformities of the system variables across the study area. These models are data intensive and often are not applicable to other regions. The GLEAMS model simulates the edge-of-field and bottom-of-root-zone loadings of water, sediment, pesticides, and plant nutrients (Leonard *et al.*, 1987). The model inputs are management practice, climatic data and soils. Users can control input variables such as crop rotation information, pesticide and fertilizer types and application rates to study their effects on the study area. GLEAM consists of the following sub-models: The hydrology sub-model calculates the water balance in the root zone. Some of the input parameters include soil water field capacity, wilting point, organic matter, porosity, curve number, leaf area index, monthly maximum and minimum temperatures, solar radiation and wind velocity (Knisel, 1993; Leonard *et al.*, 1987). The nutrient sub-model includes the nitrogen and phosphorus cycles (mineralization, immobilization, nitrification), inputs (mineral and organic fertilizers, N in rainfall and in irrigation water, symbiotic fixation, mineralization from soil organic matter and crop residue) and outputs (volatilization, denitrification, leaching, crop uptake, runoff) in the root zone (Knisel, 1993; Leonard *et al.*, 1987) .

Similar to GLEAMS, the EPIC model is a field scale model that is designed to simulate environmental outcomes in response to agricultural practices and management strategies such as crop rotations, tillage levels, soil, fertilizer applications and environmental conditions (Gassman *et al.*, 2004; Sharpley & Williams, 1990; Williams, 1990; Williams *et al.*, 1995) . The major assumption is that the characteristics of the drainage areas are homogenous, and by varying the parameters in the model the user can simulate field emissions under various farming and environmental conditions. EPIC is a continuous simulation model using a daily time step with nine major components: weather simulation, hydrology, erosion- sedimentation, nutrient cycling, pesticide fate, plant growth, soil temperature, tillage, and economics (Sharpley & Williams, 1990; Williams *et al.*, 1995) . The hydrology sub-models are: surface runoff, percolation, and lateral subsurface flow, evapotranspiration, and watertable dynamics, and Snowmelt. The nutrients component simulates nitrogen and phosphorus fertilization, transformations, uptake, and nutrient movement, which can be applied as mineral fertilizers, mixed with irrigation water, or as animal manure. Yoon *et al.*, (1997) compared the performance of GLEAMS against EPIC by simulating the effects of conservation and conventional tillage systems on runoff from a field-sized watershed in the Tennessee Valley region of Alabama. The study found that GLEAMS and EPIC under predicted nutrient losses in runoff for both tillage systems, but EPIC simulated the effect of tillage on soluble-P losses better than GLEAMS, however, its did poorly in predicting the losses of nutrients in sediments (Yoon *et al.*, 1997) .

Many of the model inputs, such as soil properties, land characteristics, and climate data due to their spatial variability, are stochastic in nature or not known with certainty. To account for the uncertainty in the data and to provide greater confidence in the expected results, some studies prefer using stochastic methods Howarth *et al.*, 2002; Lacroix *et al.*, 2005; Miller *et al.*, 2006; Yulianti *et al.*, 1999) . These methods examine the range of outputs for a range of random inputs with specified probability distributions based on measured data. Lacroix *et al.*, (2005) suggested modifying deterministic models to include variables associated with stochastic yield and runoff. The main advantage of their approach is that the probability functions of the random variables are a realistic representation of uncertainty because they are derived from large data sets. A study by Miller *et al.*, (2006) proposed the use of a probabilistic approach to model nitrogen fluxes from a large geographic agricultural area. The goal of their study was to determine the variability of the nitrogen fluxes in corn-soybean rotations for the U.S. Corn Belt, which authors defined as the nine states that produce 80% or more of the national corn and soybean production. The authors proposed using Monte Carlo analysis to simulate a probable range of outcomes given a set of variable inputs. This approach was adopted by Costello *et al.*, (2009) to assess the impact of biofuel crop production on the formation of hypoxia in the Gulf of Mexico. Nitrate loadings to the Mississippi river waters were stochastically estimated for crop production scenarios that meet the Energy Independence and Security Act of 2007's biofuel goals for 2015 and 2022. It was assumed that ethanol is produced from corn, corn stover, and switchgrass; all biodiesel was produced from soybeans (Costello *et al.*, 2009). The study also assessed the effect of nutrient management using vegetative buffer strips on nitrate loading. The study found that scenarios where ethanol is produced using only cellulosic residue had a lower NO_3 fluxes than those that included corn for production of ethanol (Costello *et al.*, 2009). Also, the range of nitrate fluxes (80% confidence interval) for all scenarios highly overlap which indicated that nitrate mass reductions may not be realized in any specific year (Costello *et al.*, 2009). Finally, nitrate decrease in any scenario is insufficient to reduce the size of the hypoxia zone below the EPA's 5000 km^2 target.

The SPAtially Referenced Regressions On Watershed attributes (SPARROW), by

Smith *et al.*, (1997) is a watershed modeling technique that uses a hybrid statistical/mechanistic approach to estimate pollution sources and transport of pollutants to surface waters. The model parameters are estimated by correlating stream water quality data with spatial data on pollutant sources, climatic and hydrologic properties that affect pollutants transport (Schwarz *et al.*, 2009). Estimating the parameters statistically provides a measure of the significance of the different water quality attributes in predicting nitrogen loading. SPARROW has been applied at both national and regional scales. On a national scale, SPARROW was applied to estimate national nitrogen and phosphorus models to predict the flux of nutrients to the Gulf of Mexico (Alexander *et al.*, 2007). On a regional scale, SPARROW was used to estimate nutrients in streams located in New England (Moore *et al.*, 2004). The system boundary is the whole Mississippi river watershed, which makes SPARROW advantageous for the study for several reasons. First, SPARROW is a hybrid model that combines statistical approaches to estimate parameters specific to a given region with a mechanistic functional form of the model (mass balance). Second, it utilizes GIS data allocated to individual stream catchment areas, thus SPARROW can be applied to a large region without assuming uniform properties for the whole region Third, SPARROW relies on a detailed stream reach network to which data on stream characteristics are spatially referenced.

Methodology

The objective of a SPARROW model is to establish a mathematical relation between water quality measurements and various attributes of the watersheds including sources of pollution. SPARROW utilizes statistical methods to explain the water quality measurements (constituent mass or load) taken at a given stream in relation to upstream sources and watershed properties such as precipitation, land use, soil properties that are thought to influence the transport of pollutants to streams (Schwarz *et al.*, 2009). The water quality data are used to estimate the mass annual load of pollutants that enter a stream at each monitoring station site. Geospatial data are then used to relate watershed attributes with load estimates. SPARROW is used in this study to estimate the spatial distributions of total nitrogen, sources of the nitrogen, and the delivery of nitrogen to Gulf of Mexico. The results of the model can be used to identify the variables that are significant predictors of nitrogen levels in streams, and evaluate the impacts of various nitrogen inputs on water quality. The major difference between this study and other SPARROW studies is that it estimates the contribution of individual crops (rather than aggregate) and various land uses to nitrogen loading. Also, the nitrogen model is estimated for the Mississippi River basin to include the major crops regions and assess the nitrogen flux to the northern Gulf of Mexico. The estimated nitrogen model is then used to predict total nitrogen loadings from energy crops production.

Crops considered in this study are corn/soybeans, hay (excluding alfalfa), and an aggregate of all other crops including alfalfa. The original approach was to model estimate runoff from corn and soybeans separately. However, the majority of corn and soybeans are grown on rotation fields and there is no data to describe which crop was grown on such fields at specific point in time (year), also more than 80% of corn and soybeans are produced on rotation fields (Wallander, 2013). The corn and soybeans variable explains the average runoff leaving corn/soybean rotation fields. Hay and other herbaceous plants are used here as a surrogate for cellulosic plants (switchgrass) for the following reasons: 1) there is not enough switchgrass production in the U.S to explain a correlation between nitrogen loading and production, and 2) there is no clear primary source of cellulosic ethanol. Hay's fertilization rate and yield are comparable to those of switchgrass (Table 4), hay is used here to provide an initial approximation of the nitrogen flux as a result of producing cellulosic ethanol.

The area for the SPARROW model includes the Upper and Lower Mississippi River, Ohio River, Missouri River, and Arkansas River Basins (Figure 2). For this study a regional total nitrogen model was calibrated for year 2002, the selected period coinciding with the latest data compiled by the USGS for SPARROW purposes (USGS, 2013). In order the assess the nitrogen flux from the EISA mandates the following scenarios are considered; EISA 2015 and 2022 mandates, hypothetical scenarios such as 2022 all corn ethanol, 2022 all cellulosic ethanol, and 2022 all soybeans biodiesel. The scenarios will be compared based on two criteria, total nitrogen flux and land requirements. These projections are based on 2002 land use, while is it impossible to determine where corn and soybeans will be grown in the future, it is assumed that the future demands will be met through increased production in the Mississippi river basin. Figure 3 shows that corn and soybeans area increased mainly in the Mississippi river region from 2002-2012.

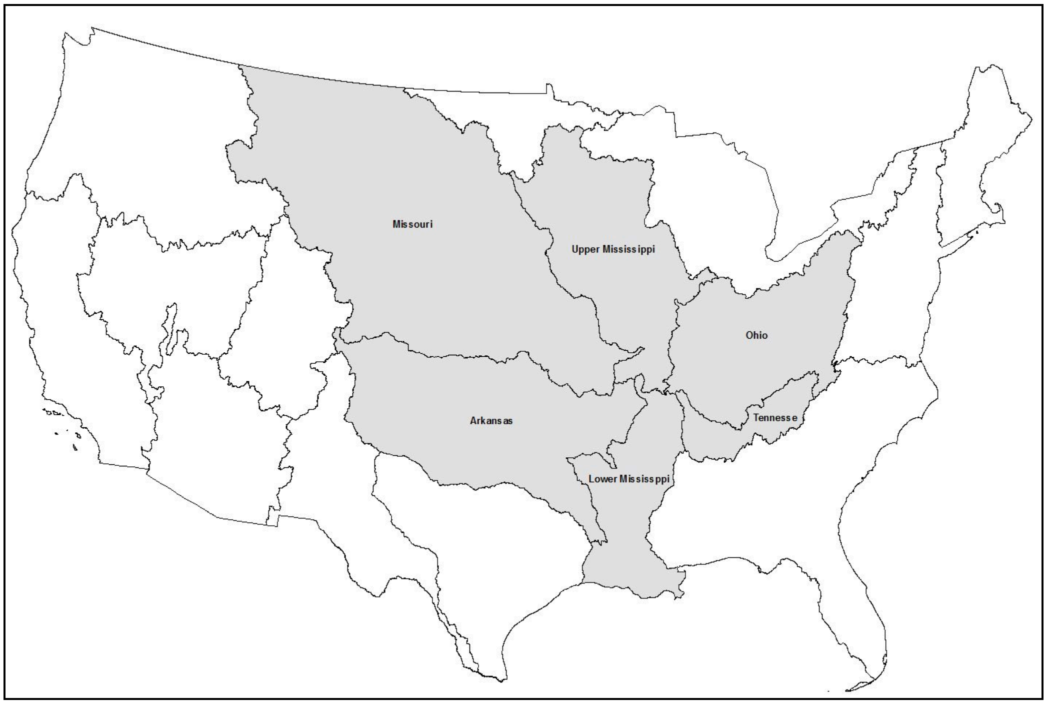


Figure 2: River basins used in SPARROW (USDA, 2012).

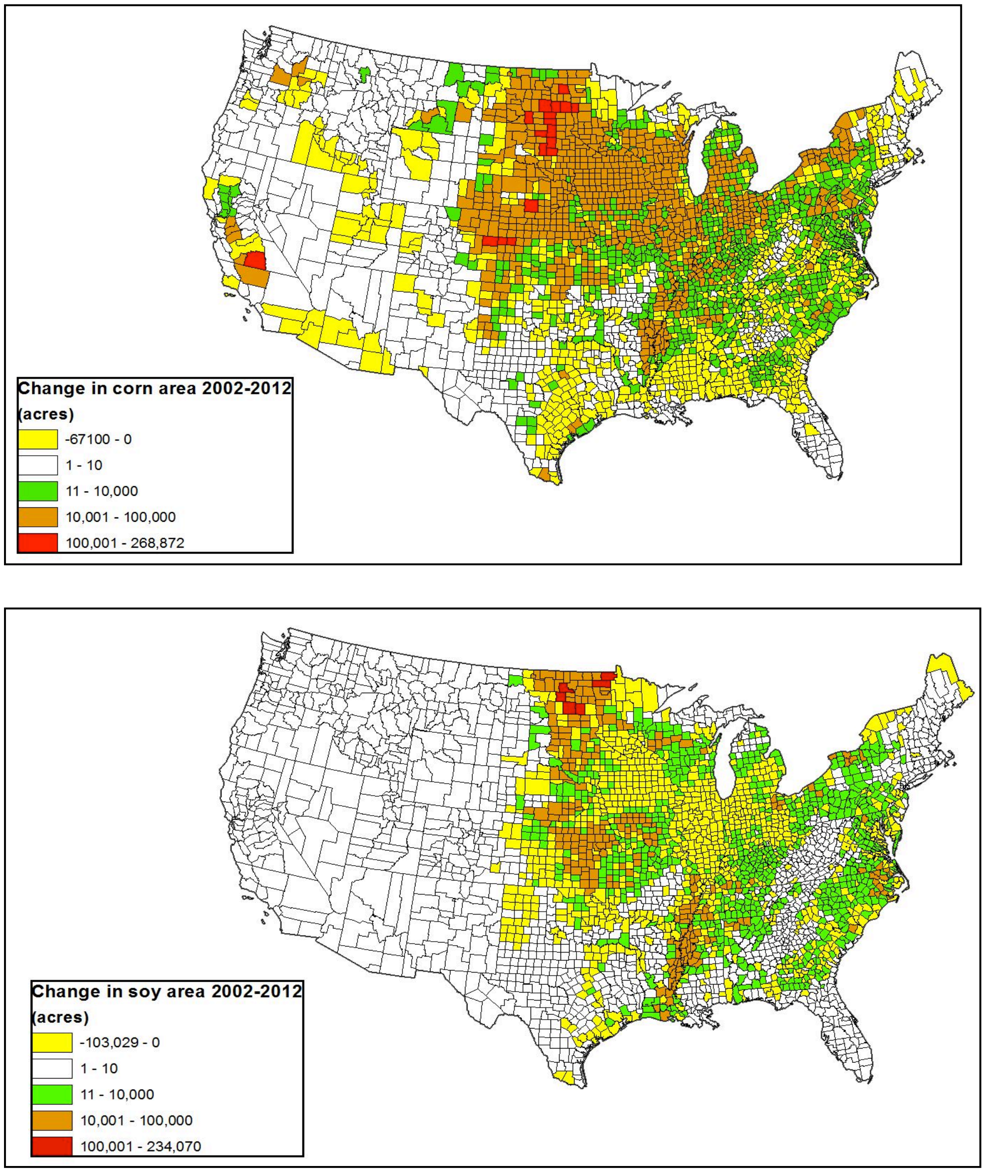


Figure 3: Change in corn (top) and soybeans (bottom) acres from 2002 -2011 (USDA, 2012).

The model uses a mass balance approach to quantify the long-term supply, transport, and fate of nutrients in streams and watersheds. In general, the load leaving the reach is the sum of (1) load generated within the upstream reaches and transported to the given reach and (2) the loads generated within the reach's own watershed (Schwarz *et al.*, 2009). The loads (dependent variables) are evaluated as a nonlinear function of the independent variables such as nitrogen sources (agriculture, point sources, atmospheric deposition), land to water delivery (precipitation, soil permeability, land cover), and decay processes (hydraulic retention time and settling). The functional relations that determine the nitrogen flux at each reach is by (eqn 1) described by Schwarz *et al.*, (2009) as:

$$F_i^* = \underbrace{\left(\sum_{j\in J(i)} F_j'\right)\delta_i A(Z_i^S, Z_i^R; \theta_S \theta_R)}_{\text{Nitrogen flux generated in upstream reaches (kg)}} + \underbrace{\left(\sum_{n=1}^{N_S} S_{n,i}\beta_n D_n(Z_i^D; \theta_D)\right) A'(Z_i^S, Z_i^R; \theta_S, \theta_R)}_{\text{Nitrogen flux generated within a reach (kg)}} \quad \text{(eqn 1)}$$

where;

F_i^* = Estimated nitrogen flux leaving reach *i*

$J(i)$ = The set of adjacent reaches upstream of reach *i*, plus additional flux that is generated within the incremental reach segment *i*

F_j' = flux leaving reaches upstream of reach *i*

Ns = number of nitrogen sources

Coefficients

δ_i = the fraction of upstream flux delivered to reach *i*

$A(.)$ = stream attenuation function acting on flux originating upstream of reach *i* and travels along the reach *i*

$\theta_{S,R}$ = steam and reservoir coefficients

θ^D = Delivery coefficients

Functions

$A(.)$ = stream attenuation function acting on flux originating upstream of reach *i* and travels along the reach *i*

$A'(.)$ = stream attenuation function acting on flux originating within reach *i* and travels along the reach *i*

$D_n(.)$ = land to water delivery function

Variables

$Z^{S,R}$ = stream (s) and reservoir (R) characteristics

S_n = source variable n

Z^D = Delivery variables

Equation 1 can be broken into two major elements. The element explains the nitrogen leaving upstream reach of reach *i* and attenuated in reach *i*. The second element explains the nitrogen leaving reach *i*. The first summation term $\left(\sum_{j \in J(i)} F_j^*\right)$ describes the nitrogen flux that leaves upstream reaches and delivered to reach *i*, $\delta_i A(Z_i^S, Z_i^R; \theta_S \theta_R)$ explains the attenuation processes acting on the nitrogen flux at it travel down the stream. The term $\left(\sum_{n=1}^{N_S} S_{n,i} \beta_n D_n(Z_i^D; \theta_D)\right)$ explains the nitrogen generated in the catchment area of reach *i* and delivered to reach *i*, $A'(Z_i^S, Z_i^R; \theta_S, \theta_R)$ explains the attenuation processes acting on the nitrogen flux at it travel down the stream. The model coefficients are estimated for each of the independent variables (sources, land to water, and decay) to evaluate the statistical significance of that variable in explaining the variation in stream nutrient loads (F_i^*). Source coefficients (β_n) determine the significance of different nitrogen sources (S_n) in explaining the nitrogen loads recorded at the monitoring stations.

The land-to-water coefficients (θ^D) explain the effectiveness of different types of land attributes (Z^D) in increasing or decreasing the delivery of nitrogen to the stream reach. The loss coefficients (θ_S, θ_R) determine the significances of stream or reservoir characteristics ($Z^{S,R}$) such as depth, retention time, and settling rates in explaining nitrogen attenuation processes (denitrification, biological uptake, and settling). The coefficients are estimated a nonlinear weighted least squares method (NWLS), where the errors are assumed to be independent and have a mean of zero, the variance is observation specific, and the errors do not have a precise distribution. The NWLS method calibrates the SPARROW model by minimizing the error between the predicted and observed values of the monitored nitrogen loads (Schwarz *et al.*, 2009). The NWLS method pertains to large sample, however when applying NWLS, parameter estimates may be bias. SPARROW has an option to assign alternative parameters distribution by assessing the inputs empirical distribution through bootstrapping (Schwarz *et al.*, 2009). Bootstrapping estimates are used to ensure the accuracy of the NWLS estimates. The bootstrap analysis is based on resampling the data to generate 200 sets of model coefficients.

Model Inputs

Load Estimation. A SPARROW model requires estimates of long-term (more than 2 years) mean load from a spatially distributed set of monitoring stations. However, water quality is not measured with the frequency needed to estimate the long-term mean flux. Thus, nutrient loads are estimated indirectly by relating them to steam flow (Schwarz et al., 2009). Another problem arises in estimating nutrient loads arises from the inconsistencies between monitoring stations records and nutrient source records. Land use data is rarely available over time, and if it is, the time record rarely matches the period of flux information. For example, the National Land Cover Database (NLCD, 2012) reports land use data once every 6 years, but the water quality data available from monitoring stations are reported for few months in a given year. To address this problem, a load estimation model is de-trended to a base year where water quality and nutrient source data are assumed to reflect processes at a given point in time (Cohn et al., 1992; Schwarz et al., 2009). The de-trended model (eqn 2 and 3) is then used to estimate the daily average nitrogen loads given the daily steam flow. The daily loads are then summed to compute the annual nitrogen load. The procedure ensures that the monitoring station estimated record and the nitrogen sources data fall within the same time period. These estimates serve as the dependent variables in the SPARROW model.

The nitrogen load estimates (dependent variables) were obtained from a study by Saad et al., (2011) that estimated the long-term mean annual nitrogen loads (de-trended to 2002) for 2,739 monitoring stations throughout various Major River Basins (MRBs). The authors compiled water quality data from federal, state, and local agencies and from selected universities. Most water-quality data were obtained from the National Water Information System (NWIS) and the U.S. Environmental Protection Agency's STOrage and RETrieval (STORET) database. The authors estimated the detrended load estimates based on two models described by Cohn *et al.*, (1992): (1) a water-quality model that relates the logarithm of nitrogen concentration at time C_t to the logathirm of daily flow rate at time q_t, a decimal time term (years) to represent trend, T_t, sine and cosine functions of decimal time to account for seasonal variation, and a model residual, e_t (eqn 2),

$$\ln(C_t) = b_0 + \ln(b_q q_t) + \ln(b_T T_t) + \ln(b_s \sin 2\pi T_t) + \ln(b_c \cos 2\pi T_t) + e_t \text{ (eqn 2)}$$

The coefficients b_0, b_q, b_T, b_s, and b_c are estimated for each site using the ordinary least squares method, however, if some of the C_t measurements are censored, the coefficients are estimated by using adjusted maximum likelihood method. The residuals e_t are assumed to be independent and normally distributed with mean of zero. The second model is a flow model to remove trends in stream flow (eqn 3).

$$q_t = a_0 + a_T T_t + a_s \sin 2\pi T_t + a_c \cos 2\pi T_t + u_t \text{ (eqn 3)}$$

The coefficients a_0, a_q, a_T, a_s, and a_c are estimated using the maximum likelihood. The residuals u_t are model residuals that are assumed to be correlated across time according to a 30-day lag autoregressive model. The final de-trended flow q_t', is estimated using $q_t' = q_t + a_T (T_b - T_t)$ where T_b is decimal time 2002.5, to de-trend the estimated loads to 2002 since the SPARROW estimates for this study reflect 2002 land use.

Hydrologic Network. This study utilized the digital hydrologic streams network that developed by Nolan et al., (2002) to support SPARROW models within selected regions of the United States. This network is used by SPARROW to define surface-water flow paths that connect nitrogen sources and streams catchments with observations of water quality at downstream monitoring stations (Schwarz et al., 2009). It is a 1:500,000-scale Enhanced River Reach File 2.0 reach network for the conterminous U.S (MRB_E2RF1). The reach network includes characteristics that describe the morphology and hydraulic conditions of stream reaches such as mean discharge, mean velocity, reach length, and travel time. The network also includes reservoirs and lakes properties such as surface area and discharge. For this study, only the reaches located in Missouri, Mississippi, Ohio, and Arkansas River Basins were used. Incremental catchments for the 27,982 streams reaches basins mentioned above were delineated from 100-m digital elevation models (Nolan et al., 2002) . Catchments ranged in size from 0.01 to 6,365 km2 (reach catchment sizes: 5th percentile ~2.5 km2, 95th percentile ~373 km2, and median size ~72 km2).

Input Sources, Transport, and Decay Variables. The data used as independent variables are summarized in Table 5. Spatial data on nutrient sources and landscape characteristics were allocated to the incremental catchment of each stream reach using ArcGIS (ESRI, 2011). Nutrient sources considered in this study included point sources (sewerage, commercial, and industrial dischargers); land cover classes (such as developed, forested, and shrub lands); fertilizer applied to agricultural land; atmospheric deposition of nitrogen (natural and anthropogenic). Some variables such as developed lands may serve as a measure of various nonpoint urban sources that enter streams through runoff from surfaces and inflows from groundwater in urbanized catchments, for example, lawn fertilizers, and septic tanks.

Data regarding point sources were obtained from (Maupin & Ivahnenko, 2011), the results of their study were used as inputs for the SPARROW model in this study. The authors estimated the nutrients loads from point sources such as sewage treatment, commercial, and industrial plants. The authors used data from the USEPA Permit Compliance System (PCS) national data warehouse (USEPA, 2013) to calculate annual total nitrogen and total phosphorus loads to surface waters of the United States for 1992, 1997, and 2002. The Great Lakes, Ohio, Upper Mississippi, and Souris-Red-Rainey had the largest number of facilities (65,602) and the Southeast had the least number (7,913). The differences are attributed to the size of the watershed and population density. The Upper Mississippi region has around 48 facilities per 1000 km^2 compared to 10 facilities per 1000 km^2 in the Southeast (Maupin & Ivahnenko, 2011), which is consistent with population densities in various regions. Sewage systems account for 50% to 70% of the total number of facilities and account for 74% of the total nitrogen and 59% of total phosphorus loads for all regions.

The atmospheric nitrogen deposition data used as input to SPARROW are based on the use of wet deposition measurements at National Atmospheric Deposition Program (NADP). Nitrogen deposition data were obtained from Wieczorek & LaMotte (2010a). The authors estimated the long-term annual nitrogen deposition (detrended to base year 2002) and allocated it to MRB_E2RF1 catchments. These estimates were calculated from annual measurements (1990 to 2005) at 186 stations throughout the United States. The atmospheric nitrogen estimated by SPARROW would be expected to reflect regional nitrogen sources such as vehicle emissions and urban lands (Elliott *et al.*, 2010) .

Data on land cover areas throughout the conterminous United States used in the model were obtained from Wieczorek & LaMotte (2010e). The authors allocated the National Land Cover Database (NLCD) 2001 (Homer *et al.*, 2007) land use categories, which include water, developed, barred, forest, scrubland, herbaceous, planted/cultivated, and wetland, to MRB_E2RF1 catchments areas. The land use/land cover input areas used for this study are urban, barren, scrubland, and herbaceous and forested areas. Note that agricultural land use from NLCD were not used in this study, instead fertilizer inputs to crops were to used to describe agricultural nitrogen inputs (see section below). Urban areas are classified into open space, low, medium, and high intensity. For the purposes of this study the urban lands were aggregated into one class referred to hereafter as urban land. The nutrient sources from urban land area serve as a measure of different urban sources such as inputs from septic systems and surface runoff from fertilized land such as lawns and parks and other urban sources. Forested areas classified into deciduous, evergreen, and mixed forests, where were aggregated into one class called forest.

Data on nitrogen inputs to various crops used in this study were obtained from Wieczorek & LaMotte (2010f). Fertilizer and manure inputs were derived from 2002 sales and expenditures data from the Association of American Plant Food Control Officials and the U.S. Census of Agriculture (Ruddy *et al.*, 2006) and allocated to MRB_E2RF1 catchments by the fraction of the catchment's agricultural land (Wieczorek & LaMotte, 2010f). Farm fertilizer sales serve as a direct measure of commercial fertilizer use and intensity of farming practices. Manure data included inputs from confined (animal feeding operations for cattle, poultry, and dairy operations) and unconfined (farm, pasture and range livestock operations) sources.

Air temperature data for year 2002 averaged from minimum and maximum daily temperature values obtained from the Parameter-elevation Regressions on Independent Slopes Model (PRISM) digital data network (PRISM, 2012) and allocated to MRB_E2RF1 catchments by Wieczorek & LaMotte (2010b) and Wieczorek & LaMotte (2010c). The 2002 annual precipitation data were obtained from the Parameter-elevation

Regressions on Independent Slopes Model (PRISM) digital data network (PRISM, 2012) and allocated to MRB_E2RF1 catchments by Wieczorek & LaMotte (2010g). Mean soil permeability was obtained from the State Soil Geographic (STATSGO) Data Base digital data (Wolock, 1997) and allocated to MRB_E2RF1 catchments by Wieczorek & LaMotte (2010d). The data described above were used to calibrate the SPARROW model for this study.

Table 5

Descriptive Statistics for Data Used in SPARROW Allocated to Stream Catchments

Parameter	**Mean**	**Standard. Deviation**	**5th percentile**	**95th percentile**
Nitrogen Load (kg N/yr)	1.09E+07	4.38E+07	3.52E+07	5.13E+07
Point discharge (kg N/yr)	8.62E+03	1.44E+05	0.00	1.59E+04
Urban land (km^2)	6.18	13.80	0.00	23.53
Wetlands (km^2)	4.21	18.17	0.00	17.47
Forests (km^2)	24.90	48.64	0.00	109.99
Nitrogen applied to corn/soy (kg N/yr)	2.40E+05	6.66E+5	0.00	1.39E+06
Nitrogen applied to hay (kg N/yr)	5.85E+04	1.44E+05	4.51	2.71E+05
Nitrogen applied to other crops (kg N/yr)	1.18E+05	3.33E+05	9.84	5.27E+05
2002 Atmospheric deposition (kg N/yr)	4.68E+04	6.86E+04	6.92E+02	1.70E+05
Average Daily Temperature C, (2002)	10.45	4.20	3.68	16.72
Annual Total Precipitation cm, (2002)	81.81	46.56	23.77	1.60E+02
Reach water time of travel (days)	1.20	1.31	0.00	3.65
Reservoir residence time (days)	83.44	6.71E+03	0.00	2.52

Results

Calibration Results

Model coefficients (NWLS estimates) and nonparametric bootstrap mean estimate of the coefficients results for the SPARROW model developed for this study, including parameter estimates, standard errors, and *p*-values, are summarized in Table 6. The model was calibrated using 1003 observations from monitoring stations throughout the region. Since the input data is highly uncertain and the model describes a large watershed, a p-value of < 0.1 was used to make certain that the probability of obtaining an estimate was not too constricting. Also, many studies that used this p value (Alexander *et al.*, 2007; Robertson & Saad, 2011; Saad *et al.*, 2011). The total nitrogen model is as a function of seven nitrogen sources (nitrogen applied to corn/soybeans, hay, other crops, atmospheric deposition, point sources, urban land, and forests); two land-to-water delivery factors (precipitation and air temperature); decay in that is a function of the time of travel in streams and reservoirs.

The model results in Table 6 only include significant variables, other variables were removed from the model because they were highly insignificant, and removing them improved the model's fit. Most Coefficients (except other crops) were significant ($p < 0.1$) indicating that each of the nitrogen inputs, land-to-water, and decay variables are important in explaining the variation and the distribution in the measured loads at monitoring stations. Even though the variable "other crops" was insignificant, it was left in the model because it believe that other crops could be a significant source of nitrogen in sub regions not dominated by corn and soy. Most of the standard errors are small compared to magnitude of the coefficient, which indicates their resemblance of the overall population (except those for other crops and forests). Table 6 also shows that NWLS and bootstrap estimates are in close in value, which indicates stability of the model coefficients and confidence in the NWLS results, which rely of parametric estimation of the coefficients. Only the NWLS coefficients are used for further applications and analysis in this study.

As mentioned above, SPARROW correlates monitored nitrogen loads with watershed characteristics, thus the model's predictions are highly sensitive to the accurately of the monitored loads. The total nitrogen model explained approximately 93% (adjusted R^2) of the spatial variability in the log-transformed annual nitrogen load (Table 6). The R^2 value for SPARROW models is generally high because of the strong relation between drainage area and annual discharge (Schwarz *et al.*, 2009). After normalizing the annual nitrogen load for drainage area (referred to as yield hereafter), the model explained ~ 87% of the variability (Table 6). The root mean square error (RMSE), which describes the accuracy of the model predictions (nitrogen loads), was 0.57. The model was evaluated for evidence of regional prediction biases by visually inspecting each of the calibration sites studentized residual map (Figure 5) to identify the large over and under predictions. Studentized residual > 3.6 or < -3.6 are considered outliers and require more investigation (Schwarz *et al.*, 2009). The majority of residuals (95%) fall between -2.26 and 2.03; there are eight sites where the residuals values are considered outliers. The Shapiro–Wilk test (Table 6) indicates that the normalized residuals are weakly normal ($p < 0.1$), and according to Schwarz *et al.*, (2009), the model is robust enough to utilize weakly normal residuals.

The coefficients for the nutrient sources provide an estimate of the fraction or quantity of each input delivered to streams (Schwarz *et al.*, 2009). The point sources coefficient should be near 1 since they discharge directly to streams and are unaffected by the land-to-water variables. However, the SPARROW estimate for point sources in this study was approximately .72, meaning that the point sources were under estimated, a results similar to Robertson & Saad (2011) . Schwarz *et al.*, (2009) points out that estimates should approximate 1 if all sources are accounted and losses are accurately described by the land-to-water variables. Thus point sources estimate was less than 1 in this study could be a result of using land to water variables that are not true representation of the actual processes. As expected, nonpoint-sources estimates are substantially smaller than 1 (Table 6) because they have been subjected to natural processes (denitrification plant uptake, and/or remain in the soil). The source coefficients indicate that 19% of the nitrogen applied to corn/soy fields, and 8.5% of the applied nitrogen to hay fields runs off to nearby streams. Costello *et al.*, (2009) found that 13% (standard deviation of 10%) of the nitrogen applied to switchgrass leaves the fields as runoff, which indicates that hay runoff is comparable to that of switchgrass. The negative sign of the temperature coefficient indicates that there is less nitrogen reaching the streams in regions with high air temperature, which could be attributed to increased biological activities (denitrofication and plant uptake). Wetlands variable (size of wetland) is also negatively correlated with nitrogen load. Precipitation on the other hand increases the nitrogen delivered to streams due to the increase surface runoff. The aquatic loss coefficients are positive which means that longer residence times for both streams and reservoirs allow for longer in-stream nitrogen decay.

Table 6

Calibration Results

Parameter	Units	Coefficient Units	NWLS estimate	Confidence Interval Lower 90%	Confidence Interval Upper 90%	Standard Error	p-value	Bootstrap Estimate
Sources								
Nitrogen applied to corn/soy	kg N/yr	Fraction	0.195	0.178	0.216	0.011	0.000	0.194
Nitrogen applied to hay	kg N/yr	Fraction	0.086	0.068	0.110	0.013	0.000	0.086
Nitrogen applied to other crops	kg N/yr	Fraction	0.006	-0.005	0.012	0.008	0.452	0.006
Atmospheric Deposition	kg N/yr	Fraction	0.110	0.057	0.159	0.030	0.000	0.110
Point discharge	kg N/yr	Fraction	0.727	0.534	0.912	0.142	0.000	0.729
Urban Land	km^2	kg $N/km^2/yr$	1,210	863	1,606	224	0.000	1,207
Forests	km^2	kg $N/km^2/yr$	28.68	3.05	55.91	15.14	0.059	28.57
Land-to-water delivery								
Average Daily Temperature	Celsius	Celsius	-0.069	-0.087	-0.049	0.012	0.000	-0.069
Annual Total Precipitation	cm/yr	cm/yr	0.016	0.014	0.018	0.001	0.000	0.016
Wetlands	km^2	km^2	-0.007	-0.010	-0.003	0.002	0.003	-0.007
Aquatic loss								
Reach water time of travel	days	$days^{-1}$	0.051	0.038	0.067	0.009	0.000	0.050
Reservoir residence time	days	$days^{-1}$	0.002	0.002	0.003	0.001	0.000	0.002
Summary Statistics								
Number of sites		1,003						
RMSE		0.57						
Adjusted R-sq		0.92						
Yield R-sq		0.87						
Shapiro–Wilk		0.92						
P-Value		0.00						

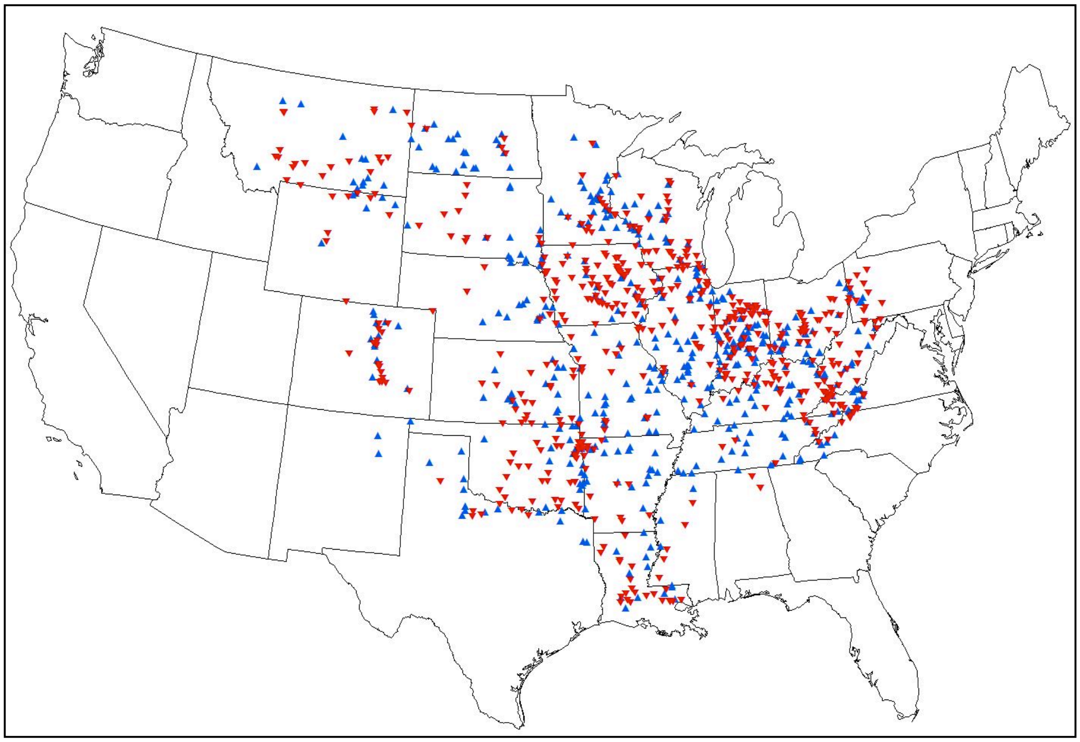


Figure 4: Studentized residuals.

Model Predictions

The distributions in total nitrogen flux and incremental nitrogen yields are shown in Figure 7 and Table 8. Figure 5 shows that the nitrogen flux and yield are highest in the Upper Mississippi River and parts of the Missouri and Ohio River. The mean upstream yield (total flux predicted to leave the reach divided by the total upstream area) was 7.13 kg/ha/yr, and the incremental yield (total flux originating within the reach's incremental watershed and delivered to the reach outlet divided by the area of the incremental watershed) was 8.69 kg/ha/yr (Table 7). The high standard deviations in the yields suggest that the spatial distribution of yields is skewed. Also, while the mean estimates of incremental and upstream yields are similar in value, the standard deviation of the incremental yield is higher than that of upstream yield, which could be attributed to the uneven spatial distribution of nitrogen inputs across the whole watershed. For example, some areas are dominated by one source; however, in other area these sources may be insignificant. On average, corn and soybeans fields runoff account for 30.27% of the nitrogen leaving incremental catchment areas followed by atmospheric deposition (25.45%) and urban land (19.47%) (Table 7). The predicted total nitrogen load in the whole Mississippi river basin was 2.67 million metric tons, where runoff from corn and soybeans fields account for 63.24% followed by urban runoff at 11.36% (Table 7). Robertson & Saad (2011) showed that percent contribution of nitrogen by source varies widely depending on the river basin. They modeled nitrogen flux in the Red river, Upper Mississippi river, and the Ohio river and found that agricultural nitrogen runoff accounts for 50% - 80%, point sources account for 3.3% - 13%, and atmospheric deposition account for 16% - 35.2%. The results suggest that the Upper Mississippi River has largest nitrogen flux, or 950 thousand metric tons of nitrogen where corn/soybean flux accounted for 77% of the total nitrogen flux, where the Tennessee River has the least nitrogen flux due to its small size compared to the other basins and corn/soybeans flux only accounts for 27% (Table 8).

Table 7

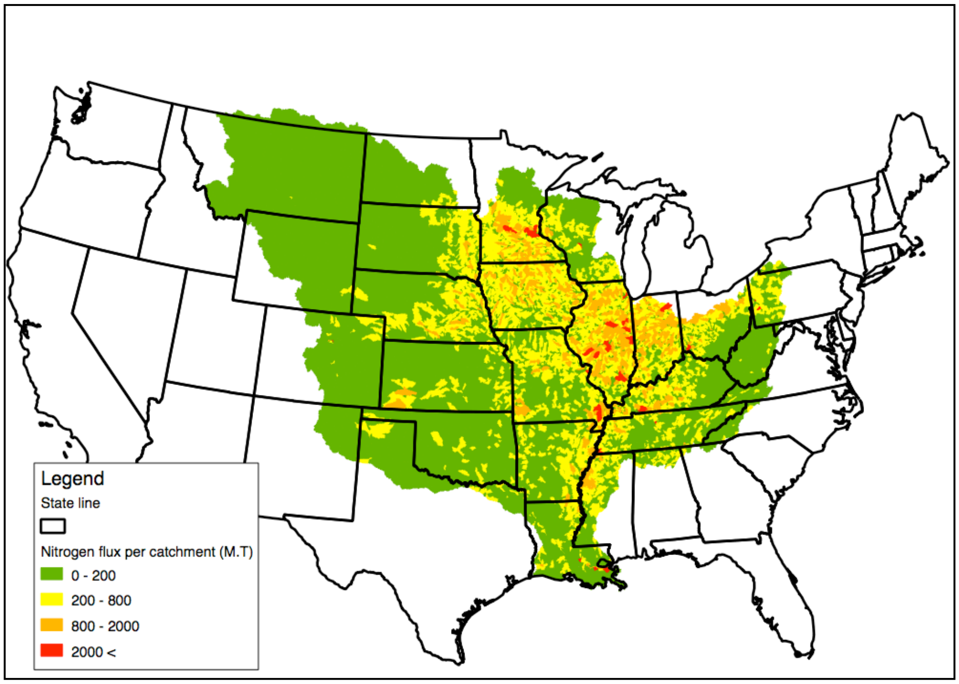
Distributions of Nitrogen Yields and Total Nitrogen Load

Variable	Mean Yield	SD	25th	50th	75th	90th	Nitrogen load (kg/yr)	Total Share
Upstream Yield (kg/ha/yr)	7.13	11.75	0.51	3.35	9.18	21.52		
Incremental Yield (kg/ha/yr)	8.69	40.22	0.65	3.83	10.85	22.52		
Incremental source share (%):								
Nitrogen applied to corn and soybeans	30.27	34.29	0.20	12.26	61.94	88.07	1.69E+09	63.24%
Nitrogen applied to hay	14.33	16.67	0.93	7.50	22.98	39.04	2.10E+08	7.85%
Nitrogen applied to other crops	2.25	3.86	0.15	0.57	2.43	7.05	2.14E+07	0.80%
Atmospheric Deposition	25.45	24.23	7.10	17.32	35.08	61.62	2.05E+08	7.66%
Point discharge	2.33	10.38	0.00	0.00	0.00	2.23	2.05E+08	7.67%
Urban Land	19.47	18.60	4.93	14.40	28.86	44.66	3.04E+08	11.36%
Forests	5.90	11.48	0.05	0.75	6.27	17.22	3.78E+07	1.41%
Total							2.67E+09	

Table 8

Nitrogen Flux by River Basin

River Basin	**Area km^2**	**Nitrogen Flux kg**	**Corn/ Soy**	**Hay**	**Other Crops**	**Point Source**	**Urban**	**Fores t**	**Atmospheric**
Tennessee	1.1.E+05	1.1.E+08	26.9%	22.0%	0.4%	7.0%	24.4%	4.8%	14.5%
Arkansas	6.4.E+05	2.0.E+08	25.0%	28.5%	1.8%	7.9%	19.4%	3.0%	14.4%
Lower Miss.	2.6.E+05	2.7.E+08	54.7%	6.0%	1.8%	10.3%	15.4%	1.5%	10.2%
Missouri	1.4.E+06	4.7.E+08	70.2%	8.5%	1.3%	4.0%	8.3%	0.9%	6.8%
Ohio	4.2.E+05	6.8.E+08	58.9%	8.5%	0.3%	9.1%	13.2%	2.0%	8.0%
Upper Miss.	4.9.E+05	9.5.E+08	77.7%	1.6%	0.4%	7.7%	7.2%	0.5%	4.9%



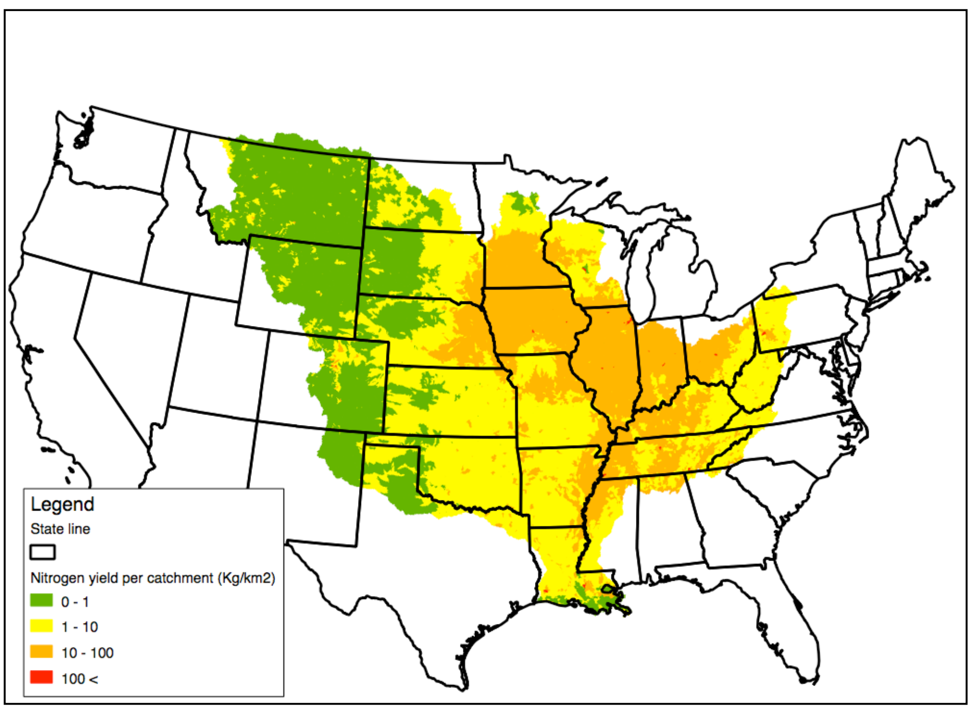


Figure 5: Visual representation of SPARROW predictions. Total nitrogen flux per catchment (top), Nitrogen yield per catchment (kg/km^2) (bottom).

The second objective of this study is to estimate the amount of nitrogen flux resulting from EISA mandates and compare them to other hypothetical scenarios. By year 2022, approximately 42% and 44% of the biofuels fuels will be produced from corn and cellulosic crops respectively, and the rest is biodiesel from various sources. While it is still unclear what type of cellulosic crop(s) will be used in the future, switchgrass seems to be the main focus of many studies (Bai *et al.*, 2010; Costello *et al.*, 2009; Nelson *et al.*, 2006; Parrish & Fike, 2005; Pimentel & Patzek, 2005; Schmer *et al.*, 2008). To estimate the nitrogen flux resulting from meeting the renewable fuels mandates, the biofuel volumes for the scenarios mentioned above were converted to nitrogen inputs using crops fertilization rates, crop yields, and crop-to-fuel conversion factors (Table 9). The nitrogen inputs were then used in SPARROW to predict the nitrogen runoff using the calibrated model. All cellulosic ethanol is assumed to be produced from hay, and since the mandates did not specify which crops are to be used for biodiesel production, for this study it is assumed that biodiesel is produced from soybeans, an assumption Costello *et al.*, (2009) used in their study. While this is a hypothetical scenario, the rationale for it is to assess the total nitrogen flux from using soybeans as a source of biodiesel.

Table 9

Inputs Used to Predict EISA Nitrogen Loads

Fuel	**Mandate (bil. Gal.)**		**Conversion factors (L/kg)**	**Yield** [3]	**Fertilization rate (kg N/ha)** [4]
	2015	**2022**			
Corn Ethanol	15.0	15.0	0.426 [1]	113.60 bu/acre	149
Cellulosic Ethanol	3.0	16.0	0.330 [1]	2.27 ton/acre	28
Soybean Biodiesel	2.5	5.0	0.2-1.4 [2]	37.26 bu/acre	27

1. (Costello et al., 2009).
2. (Hill et al., 2006; C. King et al., 2008)
3. (USDA, 2013)
4. (Alexander *et al.*, 2007)

Figure 5 shows the estimated annual total nitrogen flux resulting from meeting the different biofuels scenarios. As expected, meeting the 2022 ethanol mandate using only corn produced the largest nitrogen flux (1.7 million metric tons), whereas all cellulosic ethanol scenario produced 270 thousand metric tons of nitrogen. The total nitrogen flux from meeting the 2015 and 2022 EISA mandates is 765.3 and 892.3 thousand metric tons respectively. Costello *et al.*, (2009) have estimated nitrate-N for same years (2015 and 2022) to be 580 and 600 thousand metric tons, note that the authors only accounted for ethanol production and did not include biodiesel in their estimates. Meeting the mandates using only cellulosic ethanol or biodiesel requires large areas of land due to low hay and soybeans yields (mass/area) compare to that of corn. Figure 6 shows that loads from corn ethanol do not change due to capping corn ethanol at 15 billion gallon per year (712 thousand metric tons or approximately 80% of the total nitrogen load in 2022). The annual total nitrogen load from cellulosic ethanol in 2022 is significantly less than that of corn ethanol (117.5 and 712 thousand metric tons respectively) even though the volume of fuel mandated is roughly the same. This is due to the low fertilization rate compared to corn (Table 9). Nitrogen load from soybeans is very small because the biodiesel mandate is small relative to ethanol fuels. The 90% Confidence Intervals (CI) (error bars in Figure 7) indicate the reliability of the predictions and reflect their spatial variability. After investigating the 90%, it was concluded that the 2022 nitrogen loads from biofuels is not significantly different that of year 2015 and that is mainly due to the large variability of corn ethanol estimates. It is important to note that hay fertilization rate (28 kg N/ha) and yield (5.6 tons/ha) were used to estimate the nitrogen flux and land requirements for cellulosic ethanol, however, switchgrass is produced using higher fertilization and yield (Table 2). The nitrogen flux from switchgrass ethanol mandate (Figure 8) was estimated as follows: 1) the mass of swichgrass was calculated using data in Table 9, 2) the area for switchgrass was calculated by multiplying the mass by the yield (9.9 ton/ha) described by Duffy, (2008), 3) The area was multiplied by the fertilization rate (112 kg N/ha) to produce the total nitrogen required for switchgrass, then multiply it by hay's nitrogen runoff estimated by SPARROW. While using a higher fertilization rate results in more nitrogen flux than hay, the increased yield results in smaller area requirement for switchgrass. Normalizing the total nitrogen estimates for fuel volume, results indicate that the land requirement are 0.003, 0.005, 0.0047 acres/L for corn ethanol, hay ethanol, and biodiesel respectively (Figure 9).

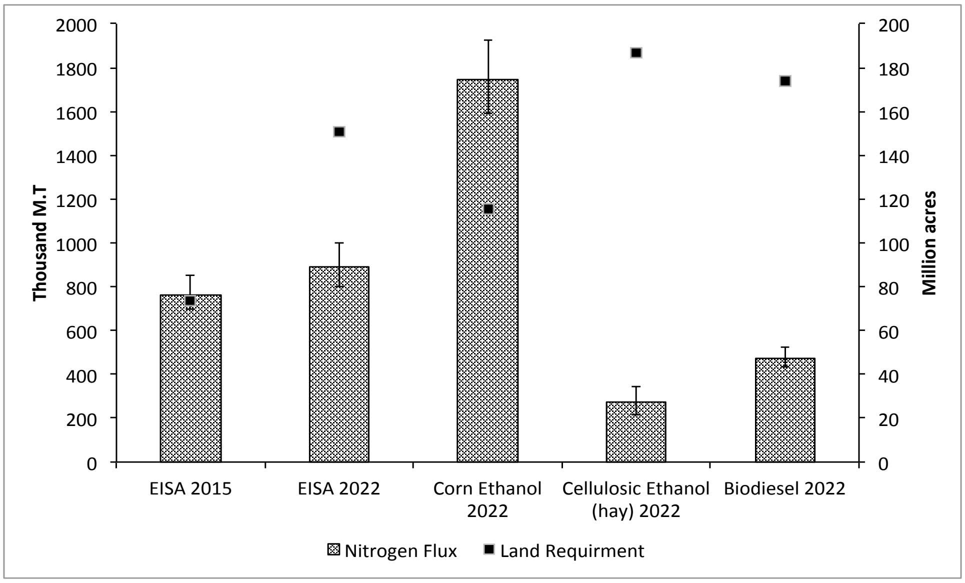


Figure 6: Total nitrogen flux and land requirements for different Biofuels scenarios.

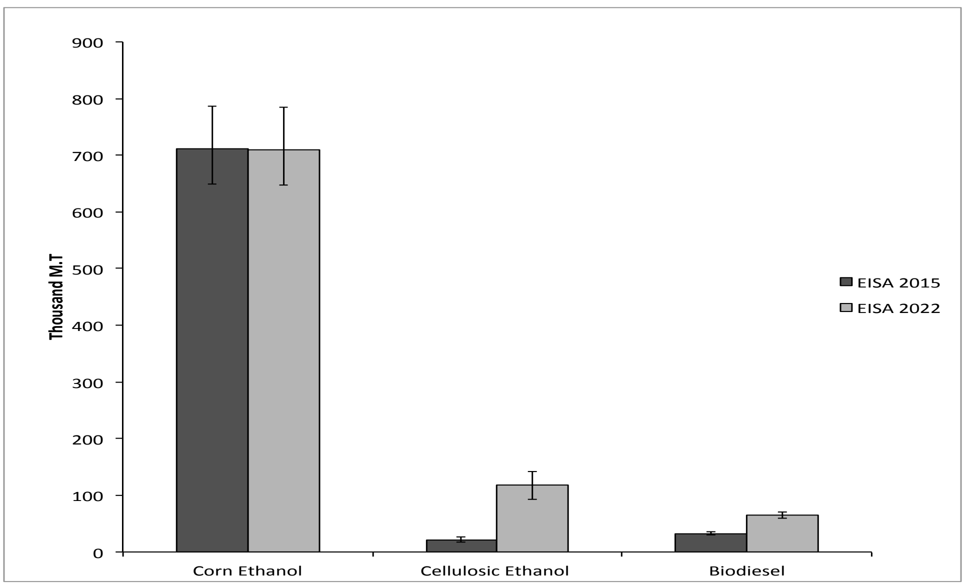


Figure 7: Nitrogen flux by fuel type for EISA mandates 2015 and 2022.

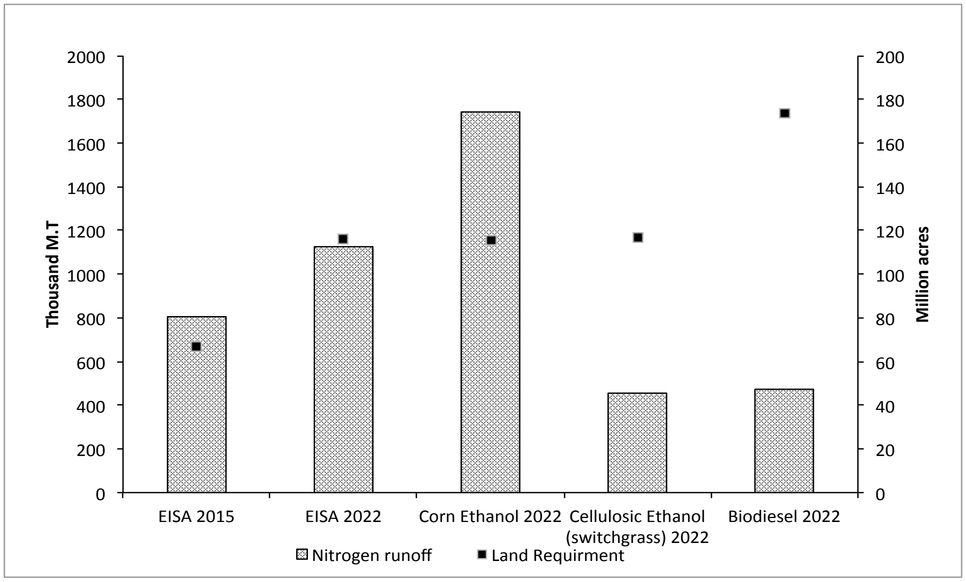


Figure 8: Total nitrogen flux and land requirements for different Biofuels scenarios using switchgrass.

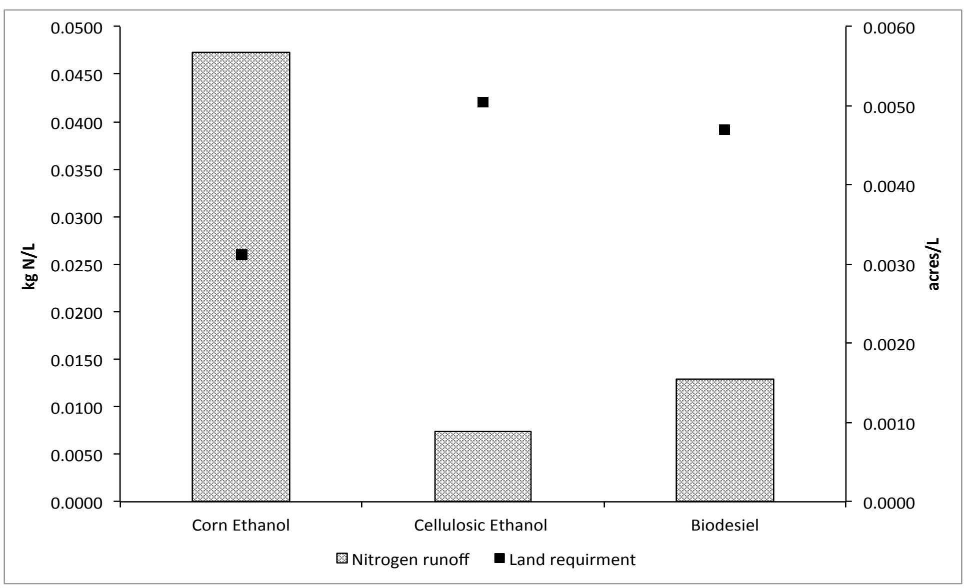


Figure 9: Total nitrogen flux and land requirements per volume of fuel.

Summary and Conclusions

The objective of this study was to use SPAtially Referenced Regressions On Watershed attributes (SPARROW) modeling method to predict nitrogen fluxes in the Mississippi River basins as a result of EISA biofuel mandates. The model was estimated to reflect 2002 land use, climatic conditions, and basin characteristics. Agricultural runoff was the largest contributor to the nitrogen yields and total flux in the study area, followed by urban runoff and point discharge. Spatial analysis of SPARROW predictions shows that the Upper Mississippi River and Ohio River contribute the largest share of nitrogen flux and have a higher nitrogen yield than the other regions mainly due to high agricultural nitrogen flux. The model results also show that nitrogen delivery to streams was affected by precipitation and air temperature, thus dryer and warmer years will result in relatively less nitrogen flux than wet and colder years. Nitrogen decay was affected by residence time in streams and reservoirs, streams have shown to be 15 times more effective in processing nitrogen than reservoirs.

The scenarios results show that biofuel production can result an increase of nitrogen flux to the northern Gulf of Mexico from 270 to 1742 thousand metric tons, that is an increase from 21% to more than 100% from the total nitrogen flux estimated by the EPA, (2011). Using all cellulosic (hay) ethanol or biodiesel to meet the 2022 mandate is expected to significantly reduce nitrogen flux however it requires approximately 25% more land than the land needed in EISA specified 2022 scenario. Producing ethanol from switchgrass rather than hay results in 3 times more nitrogen flux but requires 43% less land. The all corn ethanol for 2022 scenario mandates is expected to have double the nitrogen flux when compared to the EISA specified 2022 scenario, however, it will require less land area.

This study does not recommend one scenario over the others because evaluating the cost and benefits of each scenario is beyond the scope of this study. The results can be used by the policy makers to access the different alternative in meeting the EISA mandates and choosing the best scenario will depend on the value of the resources (land or water) used to produce the biofuels. One must note that these predictions are based on 2002 farmland spatial distribution and fertilization rates, which are not expected to change by 2015 or 2022. Its apparent from the results that shifting the U.S energy supply from foreign oil to the farms of Midwest cannot occur without economic (land and crop) and environmental (water quality) impacts, which could potentially lead to more eutrophication in streams and hypoxia in the Gulf of Mexico. As a result, the ecosystems and the fishing industries would be adversely affected. Sustainably biofuels production requires an assessment of crop selection and land management options. SPARROW maps can be used to identify areas where nitrogen flux is high and possibly implement land management practices in these areas.

CHAPTER 3

ESTIMATING THE ECONOMIC COST OF NITROGEN FLUXES FROM ENERGY CROP PRODUCTION

One of the growing concerns of increased demand for biofuels from feedstock such as corn and soybeans is that it will result in an increase of nitrogen flux (amount of nitrogen leaving the edge of a field). The 2007 EISA mandates the production of 36 billion gallons of biofuels by 2022 of which 15 billion gallons is corn ethanol and 21 billion gallons is from sources other than corn, such as cellulosic sources (switchgrass, corn stover) and biodiesel. To meet the mandates, the agricultural sector is expected to increase corn production by increased chemical and fertilizer applications on the fields, and less crop rotation with soybeans (Simpson *et al.*, 2008). Shifting the crop rotation from the conventional corn-soybean to continuous corn would significantly increase nitrate-N runoff to surface water because continuous corn systems require a large fertilizer input (Thomas, 2009).

The popularity of biofuels has also increased in the recent years as a result of growing concerns over the impacts of elevated CO_2 concentration in the atmosphere on global climate due to the burning of fossil fuels. Biofuels are attractive because they are believed to be carbon neutral, meaning that carbon is fixed in the biomass used to produce biofuels and then released back into the atmosphere during the burning of the biofuels. In contrast, burning fossil fuels releases carbon that was stored outside of the current carbon cycle for millions of years and not balanced by photosynthesis, which has led to a net increase in carbon stored in the atmosphere, in the ocean and on land.

Life cycle analysis (LCA) studies have shown, however, that not all biofuels are equally beneficial in reducing carbon and other greenhouse gases (GHG). The LCA accounts for emissions from burning the fuels directly plus emissions from production and transport of fertilizers and herbicides, land conversions, and transportation. The literature is rich with studies regarding LCA of biofuels, and the variability in the estimates is due to differences in the assumptions and system boundaries, (Bai *et al.*, 2010; Blottnitz & Curran, 2007; BRYAN *et al.*, 2010; Demirbas, 2009; EPA, 2010; Hill *et al.*, 2006; Silva Lora *et al.*, 2011). A study by the EPA, (2010) estimated the GHGs emissions of biofuels and compared it to fossil fuels, the study found that corn ethanol produced 79% of GHGs emissions when compared to conventional gasoline and switchgrass ethanol produced a -10% GHGs, whereas biodiesel released 43% of GHGs released from burning petroleum diesel. Others argue that biofuels are carbon positive (net increase in CO_2) when accounting for fossil fuels used in production and inputs such as fertilizers. Robertson *et al.*, (2008) provided an analysis of the environmental impacts of both grain-based and biomass/cellulosic-based ethanol production. They concluded that corn-based ethanol could increase the carbon debt if produced without proper management. The environmental benefits of biofuels are during the combustion phase, but they are carbon positive during the agricultural phase, and that the net environmental impact of biofuels depends on the agricultural and climate conditions (Puppan, 2002).

Impact of Biofuel Production on Nitrate Loading

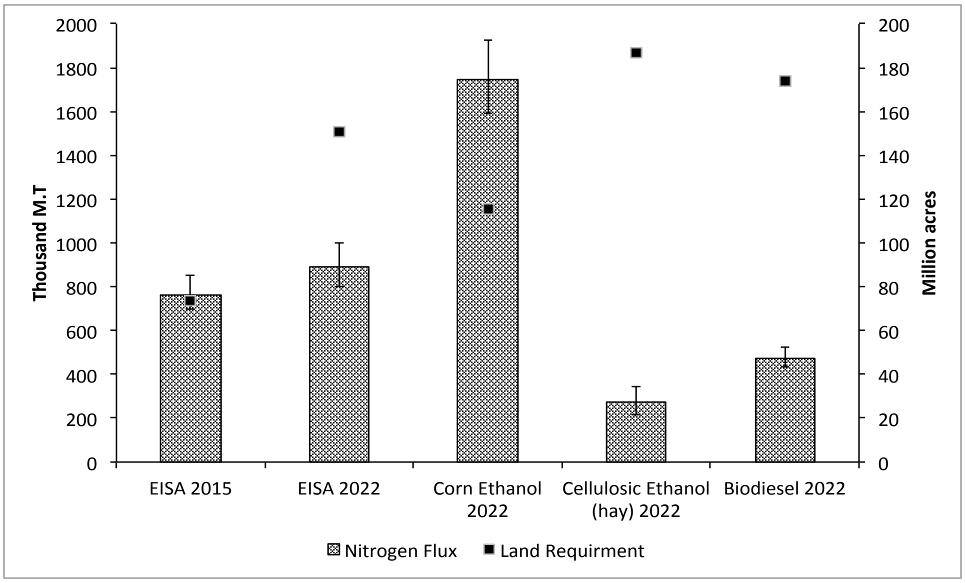
A study by Costello *et al.*, (2009) to assess the impact of energy crop production on the formation of hypoxia in the Gulf of Mexico found that meeting the 2022 biofuel mandate will increase the nitrate-N output by 300,000 to 750,000 metric tons depending on biomass sources used for fuel. That is approximately 23.8% to 60% the average annual total nitrogen flux to the northern Gulf of Mexico from 2005 to 2009. Donner & Kucharik, (2008) found that the increase in corn production to meet the ethanol targets by 2022 would increase the annual flux of dissolved inorganic nitrogen to the Mississippi and Atchafalaya Rivers by 10–34%. Chapter 2 estimated the nitrogen flux resulting from various biofuels mandates using SPARROW watershed modeling technique that uses a hybrid statistical/mechanistic approach to estimate pollution sources and transport of pollutants to surface waters. The model parameters are estimated by correlating stream-water quality data with spatial data on pollutant sources, climatic and hydrologic properties that affect pollutants transport (Schwarz *et al.*, 2009).

Crops considered in this study are corn/soybeans, hay (excluding alfalfa), and an aggregate of all other crops including alfalfa. The original approach was to model estimate runoff from corn and soybeans separately. However, the majority of corn and soybeans are grown on rotation fields and there is no data to describe which crop was grown on such fields at specific point in time (year), also more than 80% of corn and soybeans are produced on rotation fields (Wallander, 2013). The corn and soybeans variable explains the average runoff leaving corn/soybean rotation fields. Hay crops are grasses and other herbaceous plant that are grown for animal feed. Hay provides a proxy for cellulosic plants (switchgrass) for the following two reasons. First, there is not enough switchgrass production in the U.S to explain a correlation between nitrogen loading and production. Second, up to this point, this is no clear market trend as what type of grass will be used as the primary source of cellulosic ethanol. Hay's fertilizer loss rate are comparable to those of switchgrass; hay is used here to provide an initial approximation of the nitrogen flux as a result of producing cellulosic ethanol.

The area for the SPARROW model includes the Upper and Lower Mississippi River, Ohio River, Missouri River, and Arkansas River Basins. For this study a regional total nitrogen model was calibrated for year 2002, the selected period coinciding with the latest data compiled by the USGS for SPARROW purposes (USGS, 2013). In order the assess the nitrogen flux from the EISA mandates the following scenarios are considered; EISA 2015 and 2022 mandates, hypothetical scenarios such as 2022 all corn ethanol, 2022 all cellulosic ethanol, and 2022 all soybeans biodiesel. The scenarios will be compared based on two criteria, total nitrogen flux and land requirements. These projections are based on 2002 land use, while is it impossible to determine where corn and soybeans will be grown in the future, it is assumed that the future demands will be met through increased production in the Mississippi river basin.

The SPARROW calibration results show that 19% of the nitrogen applied to corn/soy fields, and 8.5% of the applied nitrogen to hay fields runs off to nearby streams. Costello et al., (2009) found that 13% (standard deviation of 10%) of the nitrogen applied to switchgrass (74 kg-N/ha) leaves the fields as runoff, which indicates that hay runoff is comparable to that of switchgrass. Figure (10) shows the estimated annual total nitrogen flux resulting from meeting the different biofuels scenarios. As expected, meeting the 2022 ethanol mandate using only corn produced the largest nitrogen flux (1.7 million metric tons), whereas all cellulosic ethanol scenario produced 270 thousand metric tons of nitrogen. The total nitrogen flux from meeting the 2015 and 2022 EISA mandates is 765.3 and 892.3 thousand metric tons respectively. Costello *et al.*, (2009) have estimated nitrate-N for same years (2015 and 2022) to be 580 and 600 thousand metric tons, note that the authors only accounted for ethanol production and did not include biodiesel in their estimates.

Meeting the mandates using only cellulosic ethanol or biodiesel requires large areas of land due to low hay and soybeans yields (mass/area) compared to that of corn. It is important to note that hay fertilization rate (28 kg N/ha) and yield (5.6 tons/ha) were used to estimate the nitrogen flux and land requirements for cellulosic ethanol, however, switchgrass is produced using higher fertilization and yield (Table 2). The nitrogen flux from switchgrass ethanol mandate (Figure 10) was estimated as follows: 1) the mass of swichgrass was calculated using data in Table 9, 2) the area for switchgrass was calculated by multiplying the mass by the yield (9.9 ton/ha) described by Duffy, (2008), 3) the area was multiplied by the fertilization rate (112 kg N/ha) to produce the total nitrogen required for switchgrass, then multiply it by hay's nitrogen runoff estimated by SPARROW. Table 10 shows the nitrogen flux broken down by river basin. The Upper Mississippi account for the most nitrogen flux in every scenario expect all cellulosic ethanol, mainly because of the intensity of the grain crops grown in that watershed and size of the watershed.



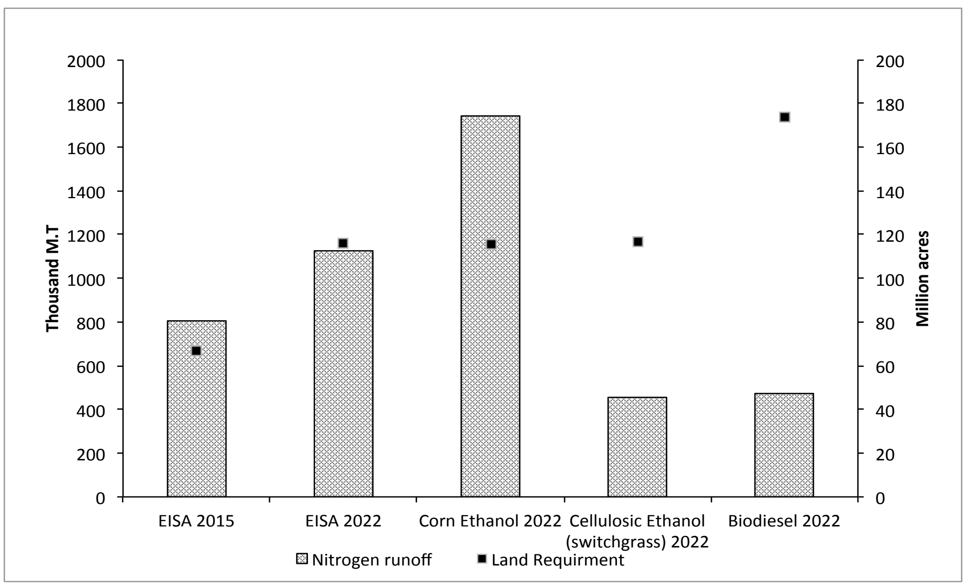


Figure 10: SPARROW predictions of nitrogen fluxes. Hay for cellulosic ethanol (top), Switchgrass for cellulosic ethanol (bottom).

Table 10

SPARROW Predictions by River Basin (1000 metric tons of nitrogen)

River Basin	EISA 2015	EISA 2022	All Corn Ethanol 2022	All Cellulosic Ethanol 2022	All Biodiesel 2022
Ohio	181.02	214.68	410.32	73.85*	111.55
Tennessee	15.24	27.18	30.08	30.87	8.18
Upper Miss.	324.84*	343.38*	756.89*	19.15	205.77*
Lower Miss.	66.76	76.91	152.45	21.09	41.45
Missouri	149.53	173.82	340.67	51.49	92.61
Arkansas	27.94	56.37	51.91	74.33*	14.11
Total	765.33	892.33	1742.33	270.77	473.67

Nitrogen Cycle and Human Impact

The atmosphere is the largest reservoir of nitrogen (mostly in the form of N_2). Bacteria in the soil coverts atmospheric nitrogen (N_2) to form inorganic nitrogen (ammonia and ammonium) through a process called biological nitrogen fixation (Kadlec, 1995). Under aerobic conditions (soil or water), ammonia and ammonium are converted to nitrate through nitrification by bacteria called Nitrobacter. Plants can assimilate all forms of inorganic nitrogen (ammonia, ammonium, and nitrate) to form organic nitrogen in the form of cells, tissues, and amino acids. When plants and animals generate waste or when they die, organic nitrogen is returned back to the ecosystem and converted back to ammonia (NH_3) through nitrogen mineralization. Finally, under anaerobic conditions inorganic nitrogen is reduced (denitrofication) to nitrogen gas, nitrous oxide (N_2O), or nitric oxide (NO) and released back in the atmosphere. The major anthropogenic sources of nitrogen are; (1) fossil fuel burning which adds nitric oxide to the atmosphere (which is eventually causes acid rain), (2) fertilizers in the form of nitrate, ammonia, or manure (organic nitrogen), and (3) human waste from septic tanks and waste water treatments plants, and urban runoff. In agriculture, fertilizers are applied in excess because farmers use a "recommended" application rate for their crops, which usually does not take into account the available nitrogen in the soil. The reason is economical as it is cheaper to apply uniform rate on soils than test each soil patch or region for nitrogen. The excess fertilizer eventually ends up in ground water, streams, and the atmosphere where is can adversely impact the ecosystem(s), which is defined as the community of the living plant, animal, and microorganisms interacting with the nonliving environment (Millennium Ecosystem Assessment, 2005).

Humans depend on the benefits that the ecosystem provides, such as clean water and water and food production, such benefits are called ecosystem services (Millennium Ecosystem Assessment, 2005). Streams have a natural capacity to handle some excess nitrogen though the process described above. However, when this natural capacity is exceeded, the stream become eutrophic (excess nutrients promote accelerated algae growth). As algae die, they decompose and deplete the oxygen available for other organisms in the streams. On a large scale, much of the excess nitrogen applied in the corn belt region gets carried by the Mississippi river and ends up in the Gulf of Mexico, which caused hypoxia (dissolved oxygen less than 2 mg/l) (Kadlec, 1995).

Nitrogen emissions can affect the ecosystem (atmospheric and aquatic) and human health in several ways. NO_x emission adversely affects the ecosystem through the formation of ozone in the lower atmosphere. These impacts include trees and plant growth impairment, reduction in habitat quality for wildlife, and respiratory problems (EPA, 2002). In addition, NO_x reacts with compounds found in the atmosphere to form the fine particulate matter (PM) causing respiratory damage (bronchitis or asthma) as they penetrate deep into lung tissue (EPA, 2002). Nitrates found in drinking water come primarily from agricultural sources, and can result in methemoglobinemia or Blue Baby Syndrome. Nitrates can also increase the acidity of the water and make toxic metals more soluble and end up in the food chain. However, the most commonly reported impacts of nitrogen are eutrophication of coastal water bodies (EPA, 2002). Impacted water bodies can suffer from declines in commercial and recreational fisheries. In addition, increased affects swimming and boating and usually produces foul odors.

Policy Scenarios Analysis

In the case of environmental goods such as water quality, prices of benefits and costs of are seldom known with certainty because environmental goods (e.g. water) are market goods and are not monetized. Also, the linkages between pollution levels and economic damages are difficult to determine. In order to come up with a holistic approach to environmental problem, one must first understand why polluters pollute the environment. The short answer is that they use the environment as a no-cost (to the polluter) waste repository; hence they externalize the cost of pollution to society. Polluters externalize the cost of pollution to society because they have no incentive to consider the costs of pollution. In the case of energy crops[2], pollution occurs at high levels because crops supply is based on meeting the EISA mandates rather an efficient supply of energy crops.

In order to compare different policy scenarios a conventional economic approach is to use benefit-cost analysis to compare policies. However, quantifying the total benefits and costs is difficult due to the reasons mentioned above. When benefit/cost analysis methods fail to achieve desirable outcomes, policies can be designed to achieve predetermined pollution targets at the least cost (Tietenberg & Lewis, 2009a). Known as cost-effectiveness analysis, the method involves an optimization problem of finding the lowest cost of accomplishing a policy target. This method does not produce an efficient outcome unless the specified policy targets are themselves efficient (Tietenberg & Lewis, 2009a), thus the effectiveness of this method will depend on choice of targets and the instrument used to achieve them (Ribaudo *et al.*, 1999).

Economic Costs of Nitrogen Externalities

The objective of environmental policy is to improve environmental quality by getting polluters to internalize the costs of pollution as a cost of production through economics incentives or standards and regulations (Doering *et al.*, 1999; Ribaudo *et al.*, 1999). Monetizing environmental externalities (agricultural impacts) is not an easy task due to the complex relationship between agricultural production and water pollution. According to Ribaudo *et al.*, (1999) there are five links that describe this relationship between agricultural pollution and water pollution. First, is the link between production of agricultural products and pollutant movement off the field, which is governed by

[2] Crops used to produce biofuels

variables such as soil characteristics, climate, land management. Second, is the transport of pollutant from the field. Third, is the discharge of pollutants into waterbodies where they may adversely impacts water quality. Fourth, is the link between water quality and ecosystem services, i.e. how water quality affect ecosystem services; Fifth, is the link between the ecosystem services and their economic value. The price (value) of any output is a function of demand. For example, the more demand for biofuels, the greater the economic cost associated with impaired waterbodies due to pollution from energy crops production. Runoff leaves agricultural fields in many places and is diffuses with runoff from other fields, which makes monitoring and quantifying (links 1-3) difficult and expensive. The difficulties in links 4 and 5 are mainly distinguishing between natural and anthropologic levels of pollution and identifying what ecosystem functions are impacted.

Externalities and Ecosystem Services

The externalities associated with biofuels production and agricultural runoff vary widely due to differences in the ecosystem services being evaluated, location of the ecosystem, economic valuation methods, and physical models used to describe water quality. This section provides a literature review on the studies assessing the cost of externalities. The costs are summarized in Table 11. Pimentel *et al.*, (1988) and Pimentel (2003) estimated the external environmental cost of corn ethanol (Table 11) on a national scale to be \$0.06/L in 2003\$, without specifying the environmental impact, he considered in the analysis. Hill *et al.*, (2009) estimated the social cost (using mean carbon mitigation costs and mean published PM costs) of greenhouse gases from production and consumption of gasoline (\$0.10/L), corn ethanol (\$0.08–\$0.14/L), and cellulosic ethanol ($0.01–$0.02/L). Kusiima & Powers, (2010) estimated the most probable externalities costs associated with biofuels based on published environmental costs of inputs used in the production, and their conclusion was similar to Hill et al., (2009). The environmental cost of corn ethanol ($0.57/L in 2008$) is significantly higher than cellulosic ethanol made from switchgrass and forest residue ($0.098/L and $0.07/L in 2008$). They also found that the costs associated with the manufacture of nitrogen fertilizer, the release of NH_3 to the atmosphere, and nitrogen to surface waters account for 46% of corn ethanol costs and 44% of cellulosic (stover) ethanol.

A study by Compton *et al.*, (2011) summarized the literature studies regarding the economic costs associated with nitrogen externalities categorized by environmental impact (Table 12). Birch *et al.*, (2010) estimated the economic cost of damages that result from the release of nitrogen into waterbodies from agricultural runoff, livestock, and sewage treatment power plants in the Chesapeake Bay. They found that the total nitrogen flux into the atmosphere (280,000 metric tons of reactive N per year) is less that the direct releases of nitrogen into terrestrial and aquatic systems (456,000 metric tons of reactive N per year). However, the total economic damage to the atmosphere is larger because of the "cascading" effect of nitrogen from the atmosphere to terrestrial and aquatic systems and the human health impacts associated with air pollution.

Agricultural production released 370,000 metric tons of reactive nitrogen per year and caused $1.7 billion of damages, however these damages only included damages caused by the emissions of terrestrial nitrogen to the atmosphere (Birch *et al.*, 2010). The authors did not monetize damages to terrestrial and freshwater systems caused by nitrogen runoff due to difficulties quantifying the impacts associated with nitrogen.

Instead, they listed the abatement (treatment or reduction) costs estimated by (Chesapeake Bay Program, 2003), \$10000 per metric ton of nitrate from agricultural fields and \$18000 per metric tons of nitrate from point sources (2000\$). Dodds *et al.*, (2008) monetize the average costs of freshwater eutrophication, including recreation, waterfront real estate, and recovery of endangered species at \$2.2 billion per year, or less than \$0.01/kg N (Compton *et al.*, 2011). A study by the EPA (2011) estimated the average annual total nitrogen flux to the northern Gulf of Mexico from 2005 to 2009 at 1.26 million metric tons, costing \$2.8 billion of damage to commercial and recreational gulf fisheries, or \$2.2/kg N. That result is very low compared to the cost reported by Compton et al., (2011) (\$56.00/kg N) summarized in Table 12. Compton et al., (2011) calculated the economic cost of nitrogen damages on fisheries in the Gulf of Mexico using the results of studies by Jordan et al., (2012) and Latimer & Rego, (2010). Using the empirical relationship between nitrogen loading and submerged aquatic vegetation (SAV) reduction in shallow New England estuaries estimated by Latimer & Rego, (2010), and Jordan et al., (2012) estimate of a 20% loss of submerged aquatic vegetation (SAV) damage cost on shrimp and crab fisheries is \$764 ha^{-1} $year^{-1}$in 2008 dollars.

Other studies estimated the cost of nitrogen abatement, (Doering *et al.*, 1999; M. O. Ribaudo *et al.*, 2001) and removal found the cost was \$4,440 - \$5,620 in 2008 dollars per metric ton. Ribaudo *et al.*, (2005) estimated the abatement cost using a farmer and point sources polluters trading system and found the cost to be \$45,822 – \$143,644 (2005\$) [\$51,837-\$16,2501 (2011\$)] per metric ton removed. Secchi *et al.*, (2007) estimated the total cost of mix of management options (CRP lands, buffer zones, nutrient management) to be \$6,998 - \$7,500 in 2007\$ per metric of total nitrogen removed.

Table 11

Externalities Associated with Ethanol

Ethanol Type	Cost*	Impact	Source
Corn	$0.06/L	Unspecified impact	(Pimentel, 2003)
	$0.08-$0.14/L	Climate impact	(Hill *et al.*, 2009)
	0.571/L	Total impact	(Kusiima & Powers, 2010)
Cellulosic	$0.01-$0.02/L	Climate impact	(Hill *et al.*, 2009)
Stover	$0.202/L	Total impact	(Kusiima & Powers, 2010)
Switchgrass	$0.098/L	Total impact	(Kusiima & Powers, 2010)
Forest Residue	$0.070/L	Total impact	(Kusiima & Powers, 2010)

* All values in 2008 Dollars.

Table 12

Nitrogen Externalities

Pollutant	Effect on services	$/kg of N		Source
NO_x	Reduced visibility	$ 0.31		(Birch *et al.*, 2010)
	Human health cost	$14.93		(Birch *et al.*, 2010)
	Crop declines from ozone	$ 1.51		(Birch *et al.*, 2010)
	Forest declines from ozone	$ 0.89		(Birch *et al.*, 2010)
	Damage to buildings	$ 0.09		(Birch *et al.*, 2010)
	Total NO_x cost	**$ 17.73**		
N_2O	UV damage	$ 1.33		(Compton *et al.*, 2011)
	Greenhouse gas effect	$13.98		(Timmons, 2013)
	Total N_2O cost	**$15.31**		
NH_3	Human health cost	**$15.48**		(Birch *et al.*, 2010)
		Low	High	
Freshwater eutrophication	Reduced lake waterfront property values	< $0.01	< $0.01	(Dodds *et al.*, 2008)
	Costs to recreational freshwater use	< $0.01	< $0.01	(Dodds *et al.*, 2008)
	Costs related to freshwater endangered species	< $0.01	< $0.01	(Dodds *et al.*, 2008)
Drinking water contamination	Purchases of bottled water because of eutrophication (odor and taste issues)	< $0.01	< $0.01	(Dodds *et al.*, 2008)
	Treatment for nitrate in drinking water wells	$0.16	$ 0.16	(Compton *et al.*, 2011)
	Health costs of nitrate in drinking water – colon cancer	$0.14	$ 3.38	(Compton *et al.*, 2011)
Coastal eutrophication	Recreational use of estuary	$6.38	$ 6.38	(Birch *et al.*, 2010)
	Fisheries decline in Gulf of Mexico related to SAV loss from N loading and eutrophication		$56.00	(Compton *et al.*, 2011)
	Damages to commercial and recreational gulf fisheries	$2.20		EPA (2011)
	Total water damages cost	**$8.88**	**$65.62**	

Source: Modified from Compton et al., (2011).

Methodology

The objective of this study is to estimate the total private (cost of production) and external (environmental damages) cost of nitrogen (atmospheric and water) associated with meeting the EISA mandate and compare it hypothetical scenarios such as all corn, all cellulosic, or all biodiesel mandates. From Table 13, in year 2015, 15 billion gallons of ethanol will be produced from corn, 3 billion gallons of ethanol will be produced from various cellulosic sources, and 2.5 billion gallons of biodiesel. In 2022, 15 billion gallons of ethanol will be produced from corn; 16 billion gallons of ethanol will be produced from various cellulosic sources, and 5 billion gallons of biodiesel. The total cost of the these mandates is compare to hypothetical 2022 mandate where the biofuel are assumed to be all corn ethanol, all cellulosic ethanol, or all biodiesel (soybeans).

The nitrogen fluxes to waterbodies associated with the above scenarios were estimated using SPARROW Table 15. The atmospheric nitrogen (N) consists of fluxes in the form of NO_x, N_2O, and NH_3 (in N equivalent) released from the fertilizers applied to energy crops. These emissions were calculated for each scenario using nitrogen inputs and land requirements inputs and displayed in Table 15. The total externality is calculated using the benefit-transfer approach (Tietenberg & Lewis, 2009b). The advantage of this method is the quick and direct use of estimates, however the accuracy of the estimates degrades the further the new location and time deviates from the original estimates.

It is important to note that the externality estimate in Table 12 is a point estimate on the marginal cost curve of nitrogen damage (except the $2.2/kg-N is the average cost reported by the EPA). Multiplying a single marginal cost estimate by the total quantity of nitrogen may not result in an accurate estimation of the total cost of nitrogen externality, if marginal costs are not constant. Due to the lack of estimates of the marginal costs of nitrogen damage in the literature, a range of total nitrogen externality costs was estimated given the range of reported nitrogen externality estimates in the literature (Table 12) to determine the possible impacts of nitrogen damage costs on policy choices.

Table 13

Nitrogen Inputs and Land Requirements for Mandates

Scenario	**Volume (bil. Gal.)**	**Nitrogen Inputs (kg)**	**Land (acres)**
EISA 2015	20.5	2.21 E+09	7.37 E+07
EISA 2022	36.0	3.09 E+09	1.51 E+08
Corn Ethanol 2022	36.0	6.97 E+09	1.16 E+08
Cellulosic Ethanol 2022	36.0	2.12 E+09	1.87 E+08
Biodiesel 2022	36.0	1.90 E+09	1.74 E+08

Table 14

Atmospheric Emissions Factor

	N_2O (kg N/ha)	**NO (kg N/ha)**	**NH_3**	**Source**
Corn	4.19	5.39 kg N/ha	2.0% of N	(Miller *et al.*, 2006)
Soybeans	1.38	1.43 kg N/ha	2.0% of N	(Miller *et al.*, 2006)
Hay	2.3% of N [a]	0.5% of app. N [b]	2.4% of app. N [c]	a (IPCC, 2006) b (Mikkelsen, 2009) c (Veldkamp & Keller, 1997)

The private cost for the scenarios is estimated using the average price (Free on Board)[3] of corn ethanol and biodiesel in 2011 reported by (AgMRC, 2013). It is assumed that the prices represent the marginal cost of production assuming that the biofuels market is a completive one. The average cost of corn ethanol is $2.56/gallon while the average cost of biodiesel is $5.17/gallons (AgMRC, 2013). Since corn ethanol has the largest market share, the ethanol prices reported by the various agencies reflect price of ethanol made from corn. A survey conducted by Bloomberg new energy finance (Bloomberg, 2013) showed that in 2012 the cost of cellulosic ethanol is 40% higher than corn ethanol ($3.55/gallon), but expected that cost would become competitive with corn ethanol by 2016 due to improvements in production technologies. This study compares the total private costs using the current costs for ethanol and biodiesel and the current and the possible future cost of cellulosic ethanol.

Results and Discussion

Table 15 shows the total nitrogen emissions to the atmosphere and water and the externality price associated with each pollutant. The nitrogen flux to waterbodies represents the largest share of the total nitrogen flux. As expected, using corn for ethanol releases the most nitrogen to the environment, mainly due to high fertilization rate and high runoff rates compare to other crops . The total externality (atmospheric and water damages) costs for each scenario are shown in Figure 11. The results show that the costs are significantly different (six fold) between the two cases mainly due to the wide range of expected cost of nitrogen damages. However, water externalities account for the

[3] The price agreed upon between the seller and buyer, which does not include transportation and insurance.

majority of the cost in both cases. Producing ethanol from cellulosic courses is expected to minimize negative externalities while corn ethanol is expected to have the largest externality cost.

Table 15

Total Nitrogen Fluxes to the Atmosphere and Waterbodies

					Flux to	Waterbodies	Total	Nitrogen
	Energy crop	N_2O-N	NO-N	NH_3-N	Minimum	Maximum	Minimum	Maximum
EISA 2015	Corn	79.30	102.00	0.38	649.00	787.21	830.68	968.90
	Soybeans	6.55	6.78	0.03	29.46	35.74	42.82	49.10
	Hay	3.95	0.01	0.04	16.58	26.65	20.58	30.65
EISA 2022	Corn	79.30	102.00	0.38	647.43	785.31	829.11	967.00
	Soybeans	13.10	13.60	0.05	58.79	71.31	85.54	98.06
	Hay	21.10	0.05	0.22	92.86	142.23	114.23	163.59
Corn Ethanol 2022	Corn	196.00	252.00	0.94	1588.22	1,926.46	2,037.16	2,375.40
Cellulosic Ethanol 2022	Hay	48.70	0.11	0.51	213.91	343.79	263.23	393.10
Biodiesel 2022	Soybeans	96.90	100.00	0.38	431.75	523.70	629.03	720.98
Externality ($/kg N)		$15.31	$17.73	$15.48	$8.88	$65.92		

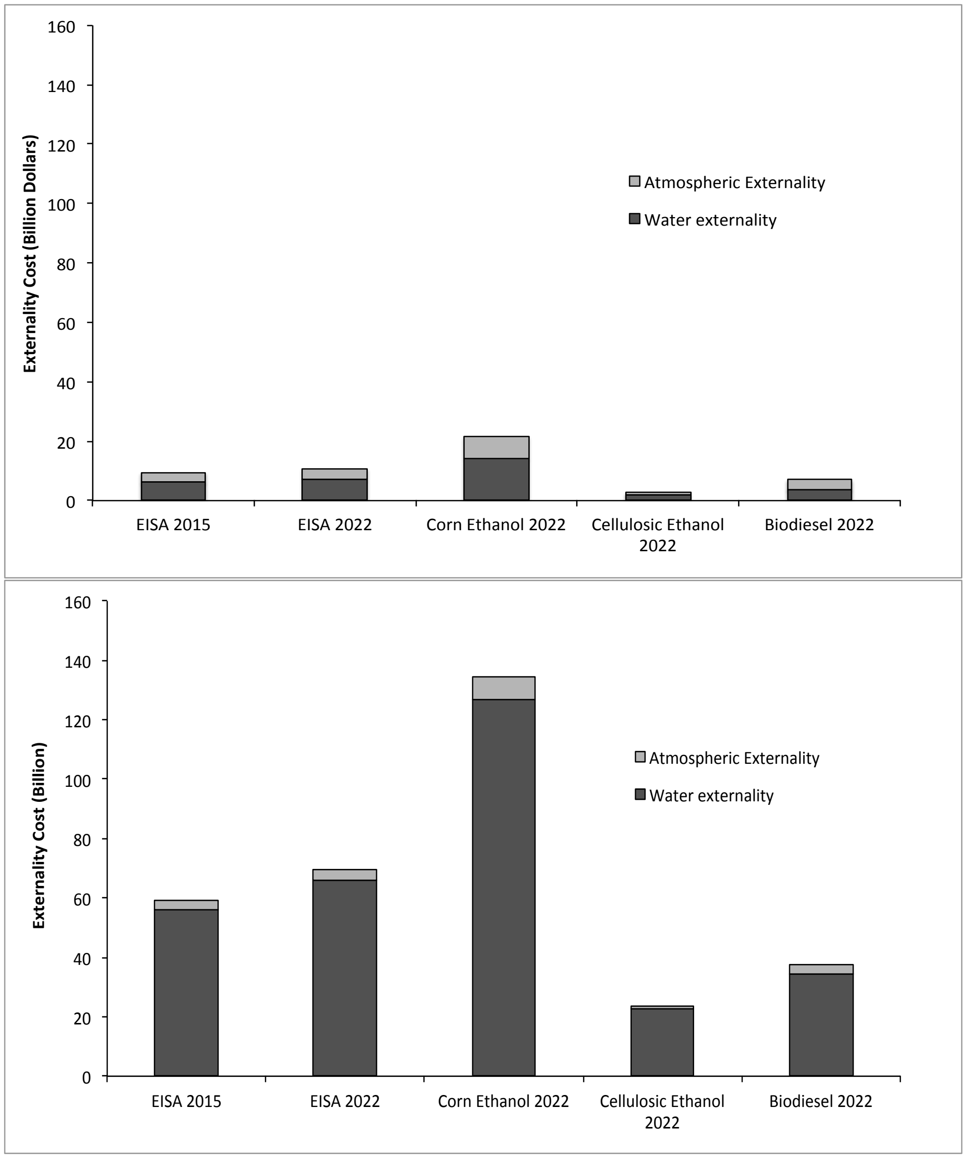


Figure 11: Total externality cost in billions.
Note: Assuming water damages cost of \$8.88/kg- N and minimum expected runoff (top), Assuming water damages cost of \$65.92/kg-N and maximum expected runoff (bottom).

Figure 12 shows the total cost (private and externality) of meeting the mandates at different price points. As mentioned above the cost of cellulosic ethanol varies from \$2.56/gallon (future cost) to \$3.55/gallon (current cost), biodiesel cost is \$5.17/gallon, and the ethanol cost is \$2.56/gallon. Also the impact of nitrogen on water varies from

$8.88 to $65.62 per kg-N. The objective of comparing the different scenarios is to assess the impact the uncertainty of the different price points on policy choices. The important policy question is whether to maintain the current EISA 2022 mandate or use other scenario alternatives. The corn ethanol scenario may be unrealistic due to its impact on food prices (see chapter 4) and it is only cheaper than the other scenarios ($113 billion dollars) in the case where case where external costs are low and production costs of cellulosic ethanol are high (Figure 12). Biodiesel is more expensive than EISA 2022 in all cases. Cellulosic ethanol is cheaper than EISA 2022 in two cases; low external cost and low production cost ($95 billion dollars), and high external cost and high production cost ($151 billion dollars). However, cellulosic ethanol cost is no worse that EISA 2022 ($130 billions dollars) in the case where external cost is low and production costs are high. This case may be the most realistic because the cost of cellulosic ethanol may not be competitive with corn ethanol in the near future and using $65.62 per kg-N as a price point for nitrogen damages may over estimate the total cost of externalities.

Further investigation (Figure 13) shows that the private costs account for a large portion of the total cost, especially with producing cellulosic ethanol or biodiesel. Given the high uncertainty in the given data inputs (prices and nitrogen estimates), it is not apparent whether EISA 2022, Corn ethanol 2022, or Cellulosic ethanol 2022 is the best option to meet the mandate (Figure 13a and 13b). Figure 13c shows that at the price point of $65.62 per kg-N the externality cost accounts for the majority of the total cost of corn ethanol. It is apparent the price point of nitrogen externality changes the optimum policy choice, and thus one must use caution when using such data.

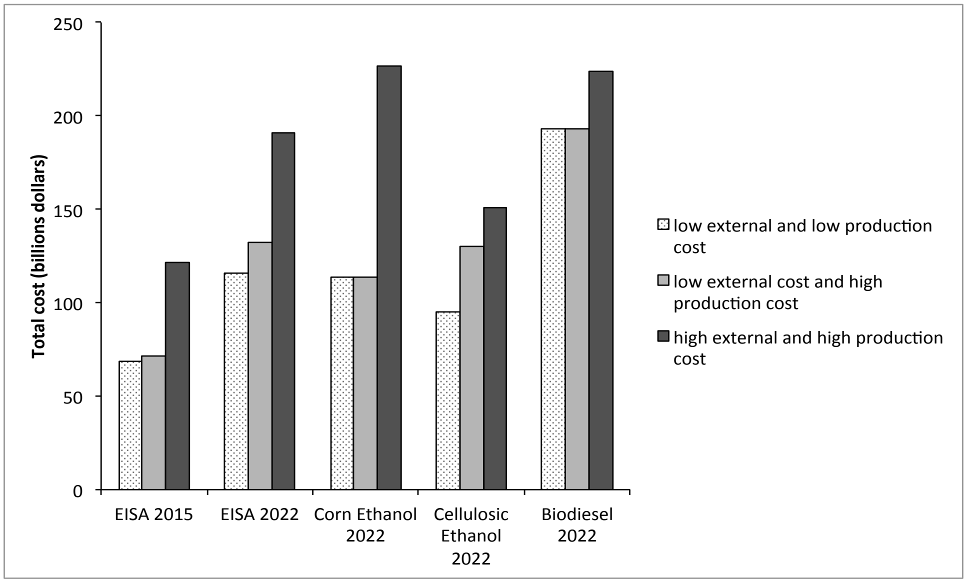
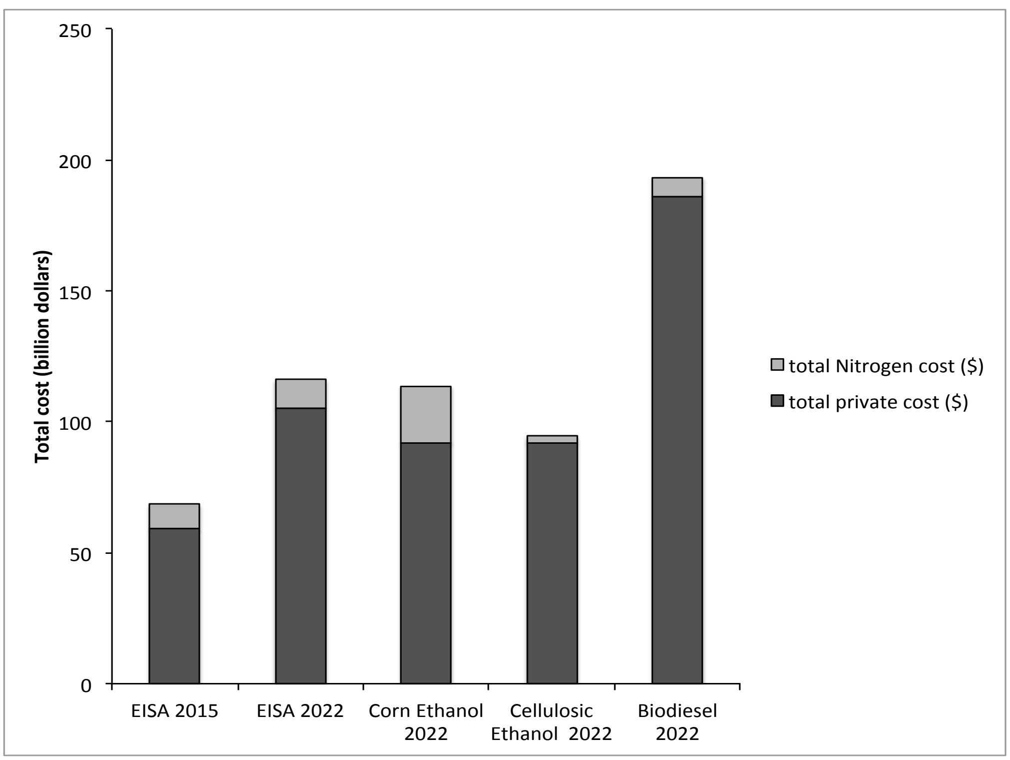
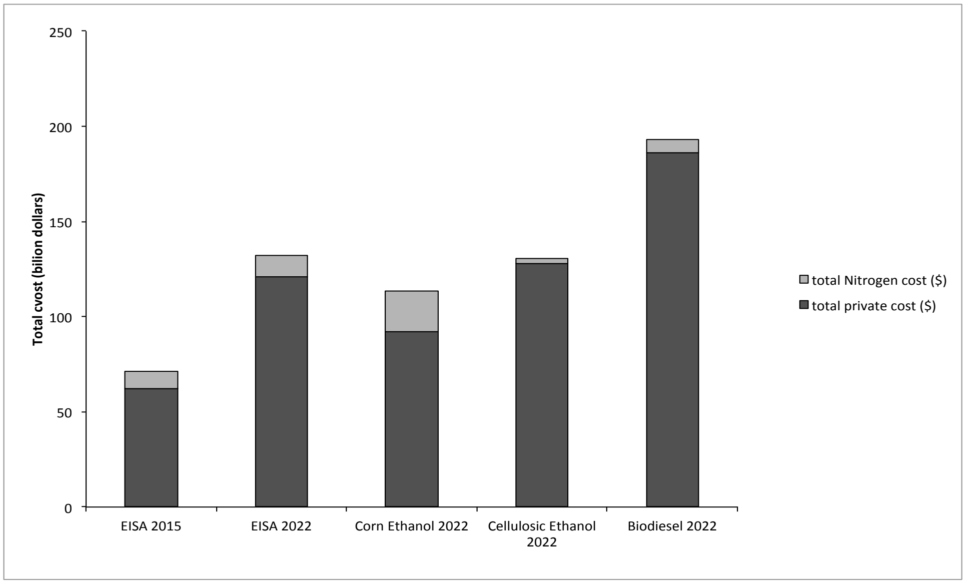


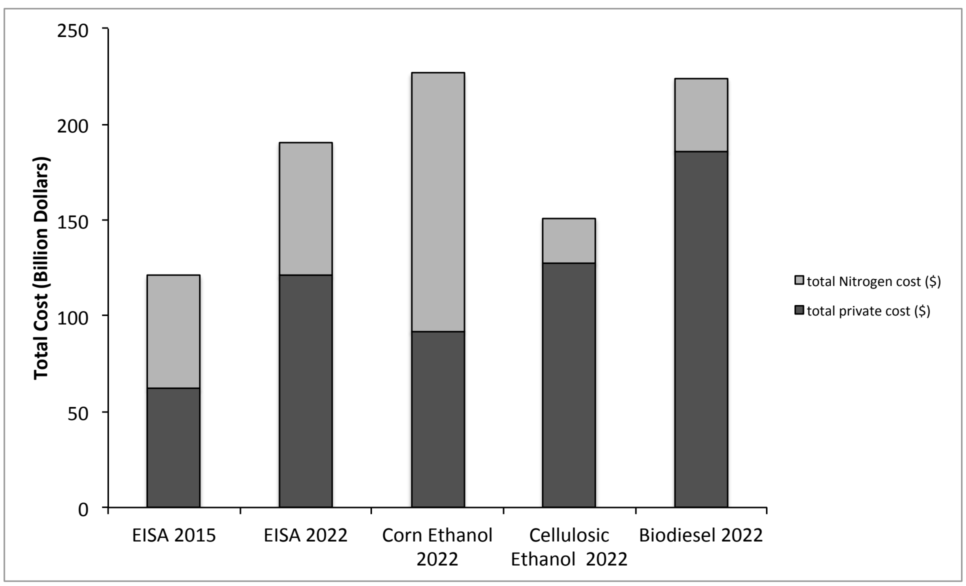
Figure 12: Minimum and maximum expected total cost of mandates (bil. of dollars).



(a)



(b)



(c)

Figure 13: Comparison between private and external costs. (a) low externality cost and low production cost, (b) low externality cost and high production cost, (c) high externality cost and high production cost.

Summary, Caveats and Recommendations

The total costs (private and external) of meeting the mandates were calculated by first estimating the nitrogen fluxes associated with the EISA mandates. The nitrogen fluxes to waterbodies associated with the above scenarios were estimated using SPARROW and the atmospheric nitrogen (N) consists of fluxes in the form of NO_x, N_2O, and NH_3 (in N equivalent) released from the fertilizers applied to energy crops. The results show that from an environmental prospective, cellulosic ethanol and biodiesel scenarios are the most effective in meeting the 2022 mandates while minimizing the impacts on the environment. However, due to the high cost of biodiesel production, the biodiesel scenario is the most costly. The total cost of cellulosic ethanol scenario is less than EISA 2022 scenario, which indicates that there is no economic or environmental incentive to produce corn ethanol by 2022, assuming that prices used in the estimate account for subsidies. The total cost results indicate that at the worst case (high externality cost and high private cost) it is clear that the cellulosic ethanol is the best option to meet option to meet the mandates. At low externality cost and low cost of production, the cellulosic ethanol scenario remains the best option to meet the mandates but with less certainty. When using the most realistic case (high production cost and low externality cost) corn ethanol is the least costly option to meet the 2022 mandate. This study finds that the price point of externality is a determining factor in policy outcome, thus is it important to investigate the economic cost of nitrogen damages with further details to narrow the range of uncertainty. It also highlights the importance of research to reduce the cost of cellulosic ethanol production.

The main finding of this analysis suggests that cellulosic ethanol would be the best approach for reducing both private costs and environmental impacts of biofuel production. However, cellulosic ethanol is not economically competitive yet. Until it is competitive, we recommend that in order to meet the mandates and minimize their total cost the U.S take actions in reducing environmental externality associated with corn ethanol. Since the market fails to account for these externalities, the government can correct for the market inefficiencies by providing economic incentives such as pollution fees or abetment subsidies. Profit maximizing polluters respond by considering these incentives when making decisions about production. From an economic efficiency perspective, there is no difference between fees (taxes) and subsidies. In reality, however, taxes reduce polluters' profits, while subsidies impose costs on the management agency. In addition, subsidies are considered inefficient in the sense that they may allow too many polluters to operate that would not do so in the case of taxes (Kolstad, 2000). Ideally, society should be compensated by an amount equal to the cost of damages caused from pollution, however, the exact amount of damages from pollution and their cost is never know with certainty.

Incentives can be based on how much runoff is generated at a field so that the external cost of pollution is considered by producers, however, the cost of monitoring runoff is high Ribaudo *et al.*, 1999). Design-based (expected runoff) incentives may be observed by a resource management agency through runoff simulations that are based on input and technology. The optimal incentive would equal the marginal increase in runoff at each field; however, it is not realistic and very costly to implement site-specific incentives (Ribaudo *et al.*, 1999). A study by (Huang & LeBlanc, 1994) examined the effects of residual nitrogen tax as a non-regulatory method to encourage farmers to limit nitrogen use. A tax scheme was proposed to penalize farmers for the potential leaching of residual nitrogen into groundwater and are subsidized for growing crops that capture and utilize residual soil nitrogen. Wu & Tanaka, (2005) used economic and physical models to estimate the social cost of reducing nitrogen runoff from the upper Mississippi river basin to the Gulf of Mexico under conservation subsidy policies and a fertilizer use tax. They found that the fertilizer-use tax is more cost effective than the other policies, however, more difficult politically to institute.

These policies incentivize farmers to reduce nitrogen runoff of their fields by using land management practices. For example, conservation buffers can help control pollutants, manage other environmental problems, and also provide other ecosystem services such as enhance fish and wildlife habitat, enhance aesthetics and recreation opportunities, and create sustainable landscapes (Bentrup, 2008). Studies have shown that conservations buffers can remove nutrients and effectively limit sediment runoff from fields (Doering *et al.*, 1999; DOSSKEY & MICHAEL G., 2001; Lee *et al.*, 2003; Mayer *et al.*, 2007; Ribaudo *et al.*, 2005; M. O. Ribaudo *et al.*, 2001; Secchi *et al.*, 2007). A meta-data study by (Mayer *et al.*, 2007) found that buffers have a mean nitrogen removal effectiveness of 67.5%. They also found that the removal effectiveness varies nonlinearly with width, but no significant removal differences were found among vegetation type. Lee *et al.*, (2003) measured the effectiveness of using switchgrass strips as buffer zones and found that total nitrogen was reduced by 80%. Thus, Farmers can profit from using buffer zones by growing cellulosic crops that can be used to produce biofuels.

CHAPTER 4
CONTRIBUTION OF ETHANOL MANDATES
TO ENERGY SECURITY

Congress passed the Energy Policy Act of 2005 and the Energy Independence and Security Act (EISA) of 2007 to reduce dependency on foreign oil by increasing the use of biofuels. The acts employ a combination of economic incentives and mandates to promote the use of biofuels, primarily ethanol, for transportation energy consumption. EISA created mandatory fuel standards requiring least 36 billion gallons of biofuels to be used by 2022 as transportation fuel (Table 1). Since the majority of ethanol is produced from corn and other feedstock, it was anticipated that agribusiness would respond by increasing production corn and other usable feedstock though increase in production (USDA, 2011b).

Energy Security

Few attempts have been made to define and measure energy security in order to meet energy polices. Energy security is a broad term that covers many different definitions and covers concerns about energy, economic growth, and political power (Westminster Energy Forum, 2006). It is also defined as having a reliable energy source(s) at a reasonable cost (Bielecki, 2002), or as securing adequate energy supplies at reasonable and prices in order to sustain economic growth. Ciuta (2010) argues that the definitions of energy security found in the literature is ambiguous, and security means different things to different authors, and that energy insecurity is the results of the increasing energy consumption and decreasing energy reserves. Based on that definition, the European Commission (Commission of the European Communities, 2000) assigned four attributes to energy security: *physical, economic, social, and environmental disrup*tions. A *physical disruption* refers to the temporary or permanent change to the supply of energy, thus this attribute measures the stability of an energy source in meeting energy demands (Commission of the European Communities, 2000). This approach was used by Eaves & Eaves (2007) to estimate the disruptions of ethanol supply from 1960 to 2005 to include major oil shock events. Using a logistic distribution fit to data, they found that the average distribution for corn yield was 3% (90% CI of -31% to 38%) while oil imports had an average distribution of 5% (90% CI of -15% to 25%), indicating that corn yield is less reliable than oil imports based on the lower end of the CI. *Economic disruptions* refer to the price volatility of energy produces, which in return reflect the stability of energy supplies (Commission of the European Communities, 2000). In general, energy prices sharply increase in response to market speculations regarding the future stability of an energy sources. *Social and environmental disruptions* refer to the ability of an energy source to create social conflict and cause damages to the environment.

This study aims to assess the ability of ethanol mandates to improve the energy security of the U.S based on the physical and economical attributes mentioned above. The physical disruption can be measured by comparing the reliability of biofuels supply compared to imported oil supply in order to determine the risk of supply disruption between the two. If the supply of ethanol is less likely to be disrupted when compared to imported oil, then the supply of ethanol is more reliable than imported oil. The economic stability of ethanol is quantified by comparing the price volatility of ethanol to the price volatility of crude oil in order to determine whether the price of locally produced fuels is less volatile than the price of imported oil. This analysis is based on the current political and economic landscape, which includes the subsidies for ethanol blending in the U.S. and EISA mandates.

This study does not attempt to distinguish between and different types of bioethanol and their effectiveness as biofuels, nor draw broad conclusions regarding biofuels vs. conventional fuels. In addition, no assumptions were made about the gasoline-ethanol blending ratios. However, one must note that 95% of the gasoline sold in the U.S contains up to 10% ethanol (E10) to meet air quality standards and does not qualify as alternative fuel (EPA, 2012). Higher ethanol blends require significant investments in the ethanol and gasoline infrastructure that are not considered in this study.

Energy Production

To achieve energy security it not only important to increase domestically produced fuels, but also to have a positive net energy value (NEV). NEV is used as a metric to assess the ability of various biomass sources in achieving energy independence. The NEV of a fuel is the difference between the energy content of the fuel and the energy used to produce and distribute it, usually represented as unit of energy per volume of fuel. To estimate the NEVs, the complete life cycle (production to use) of the fuel must be considered, including energy used in the production of crops, fertilizers, fuel production, transportation of feedstock and fuels, and energy use by vehicles.

The literature on NEVs of biofuels contains a wide range of estimates due to differing system boundaries and assumptions. Based on EISA mandates, the next generation of biofuels will be produced from cellulosic sources and oil seeds. While there are many sources of cellulosic ethanol and biodiesel, this study will look at corn stover, switchgrass, and soybeans due to their near future feasibility and quantities available. Several studies on life cycle assessment of biofuels have been conducted to determine the NEV of producing biofuels (Table 16). Corn ethanol has the lowest NEV and in some cases negative NEVs, however, when combined with corn stover and over byproducts such as distiller's dried grains and soluble (used as animal feed) the NEV for both can be as high as 32 MJ/L. Switchgrass ethanol and soybean biodiesel have high NEVs because of the low energy inputs for the case of switchgrass and high-energy by-products for soybeans. Even though cellulosic ethanol and biodiesel have high NEVs, given the current low U.S production capacity for cellulosic ethanol and the high costs associated with converting cellulose and soybeans, most of the current biofuel is produced from corn.

Table 16

Net Energy Value (NEV) for Corn, Soybean, Switchgrass, and Stover

Biomass	NEV (MJ/L)	Sources
Corn ethanol	-5.7	(Patzeka, 2004)
	-5.8	(Pimentel & Patzek, 2005)
	0.7	(Shapouri & McAloon, 2002)
	-1.0	(Graboski, 2002)
	0.8	(Dias De Oliveira *et al.*, 2005)
	2.1	(Wang, 2001)
	0.7	(Hill *et al.*, 2006)
	3.9	(ROBERTSON et al., 2011)
Corn + Stover ethanol	22.0	(Patzeka, 2004)
	32.0	(Pimentel & Patzek, 2005)
	27.7	(Shapouri & McAloon, 2002)
	29.3	(Graboski, 2002)
	30.7	(Dias De Oliveira *et al.*, 2005)
	24.3	(Wang, 2001)
Switchgrass ethanol	21.5	(Schmer et al., 2008)
	32.6	(Pimentel & Patzek, 2005)
Soybean biodiesel	25.1	(Hill *et al.*, 2006)
Corn Stover	26.7	(Fore *et al.*, 2011)
	17.1	(Blottnitz & Curran, 2007)
	11.2-16.8	(Luo *et al.*, 2009)

Source: Adopted and modified from Luo *et al.*, (2009).

Energy Sector Analysis

U.S total energy production increased more than 6% between 2008 and 2011. This increase occurred after a long period of small decline or no change in overall production. From 73,000,000 billion BTUs in 2008, U.S. energy production increased to 78,000,000 billion BTU in 2011 (Figure 14). This increase c\

n be attributed mainly to changes in fossil fuel production (62%) followed by changes in renewables (40%), where biomass account for only 12% of the renewables (EIA, 2012). Part of this change is attributable to improvements in crude oil recovery technologies and policies promoting renewable energy. The import share of petroleum increased steadily until 2006, but decreased every year since then. Approximately 18% of the total energy consumed in 2011 comes from imported petroleum products (EIA, 2012). Energy consumption decreased following the stock market crash in 2008 accounting for initial decline (Figure 14). Fossil energy represents the majority of total energy use. In 2011, the consumption of renewable (biomass; geothermal; wind; solar and conventional hydropower) energy was approximately 9,100 trillion BTUs, accounting for 11% of the total energy consumption (Figure 14). Of the major renewable fuel categories, biomass accounted for 48% of total renewable energy consumption in 2011, while hydroelectric power accounted 35%. Wind had a 13%, and solar and geothermal combined contributed 4.2% of total renewable energy consumption. Total renewable energy consumption increased by 1000 trillion BTUs in from 2010 to 2011. Each of the renewable fuels contributed to this increase. Hydroelectric power consumption increased by 630 trillion Btu. Wind energy consumption had the second largest growth, increased by 245 trillion Btu. Biomass consumption increased by 110 trillion Btu mainly due to increased biofuels consumption. The consumption of solar and geothermal energies came in last accounting to 32 trillion Btu and 18 trillion Btu, respectively (EIA, 2012). *Based on the most recent energy data from DOE's Energy Information Administration (EIA), ethanol accounts for 12% of the total renewables consumption (EIA, 2012).*

However, the overall change in US energy production and import share is, at its core, due to new and novel attributes. For example, the World Energy Outlook 2012 (produced by the UN International Energy Agency) (IEA, 2012) report predicts that the U.S. will become the largest global oil producer by 2020 as a result of advances in oil and shale gas recovery technologies, and fuel efficiency measures in transportation (IEA, 2012). The report also predicts that the U.S will become a net oil exporter by 2030, which will meet the energy independence goals but will not make the immune from the global energy markets

The transportation sector consumes the most energy among other sectors second to the industrial sector (Figure 15). The total energy consumed by the transportation sector in 2011 was 27,080 trillion BTUs. The sector relies heavily on liquid petroleum products to meet the demand (Figure 16) and around 44% of the petroleum and other liquid fuels consumed in the U.S are imported (EIA, 2012), The use of biofuels in the transportation sector is a very small but growing fraction of the total fuel consumption, approximately 4.5% in 2011 (EIA, 2012). This small percentage consisted of 13 billion gallons of corn ethanol that were produced using 41% of the corn production (USDA, 2011a). The use of biofuels is expected to increase driven largely by the biofuels policies, mandates, and incentives mentioned above.

In summary, the energy used in transportation sector is still dominated by fossils fuels. Recognizing the environmental and energy security impacts of fossils fuels, the U.S' government has taken measures to reduce these impacts. In addition to the EISA 2007 mandates to increase energy supply, the U.S Congress also passed the Corporate Average Fuel Economy (CAFE) (first enacted in 1975), which require vehicle manufacturers to comply with fuel economy standards set by the Department of Transportation.

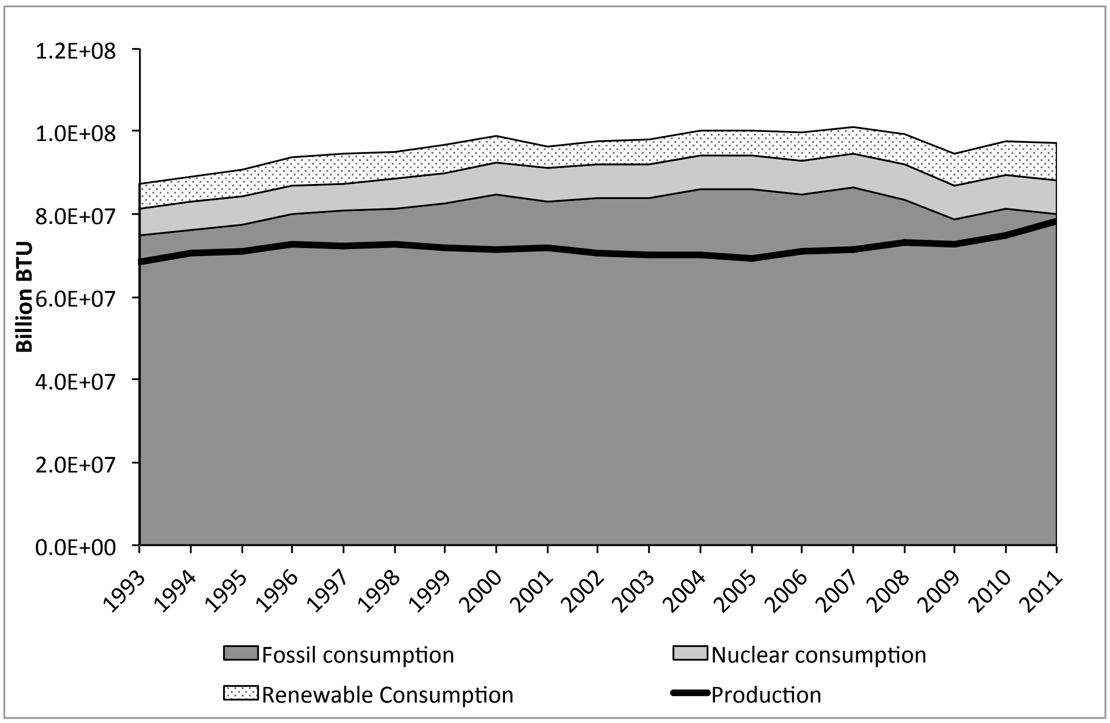


Figure 14: Historical energy production and consumption (EIA, 2012).

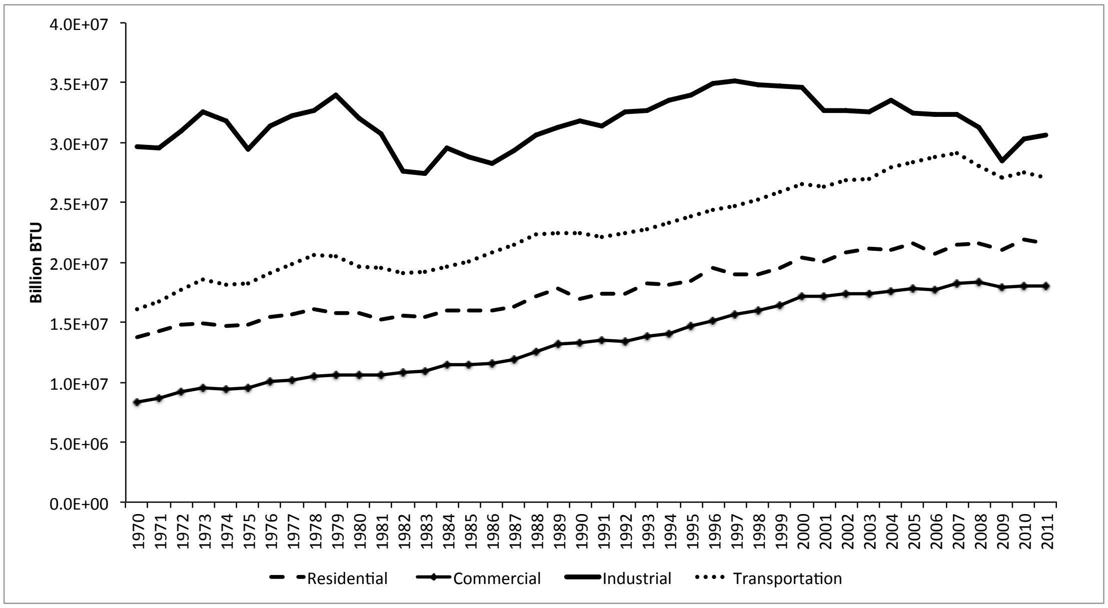


Figure 15: Energy consumption by sector (EIA, 2012).

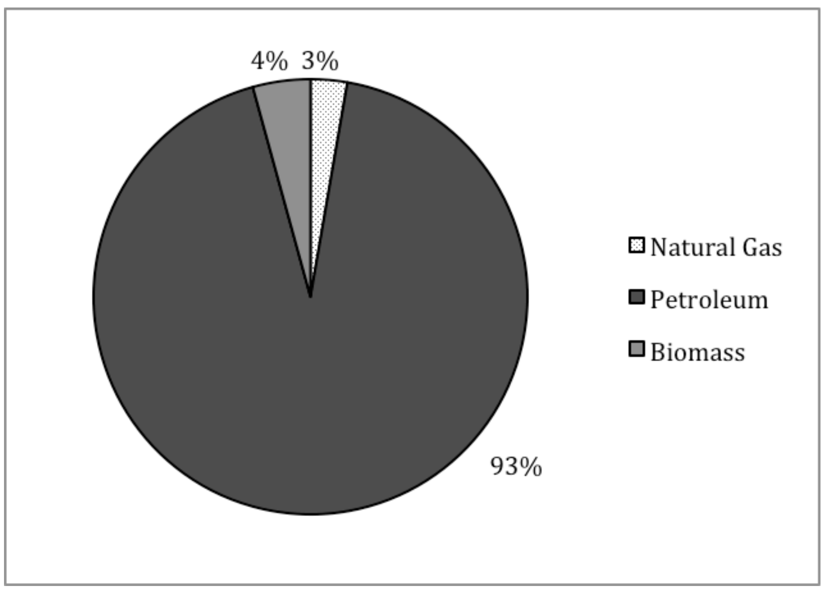


Figure 16: Transport energy consumption by type, 2011 (EIA, 2012).

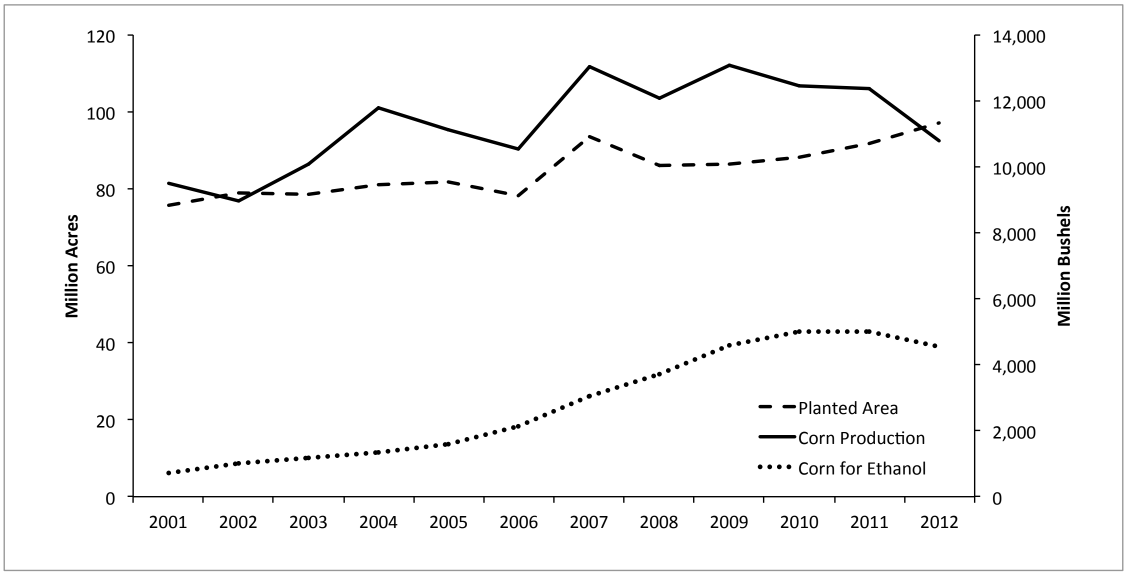
Impact on Food Prices

There are concerns over the impacts of ethanol production on food prices.

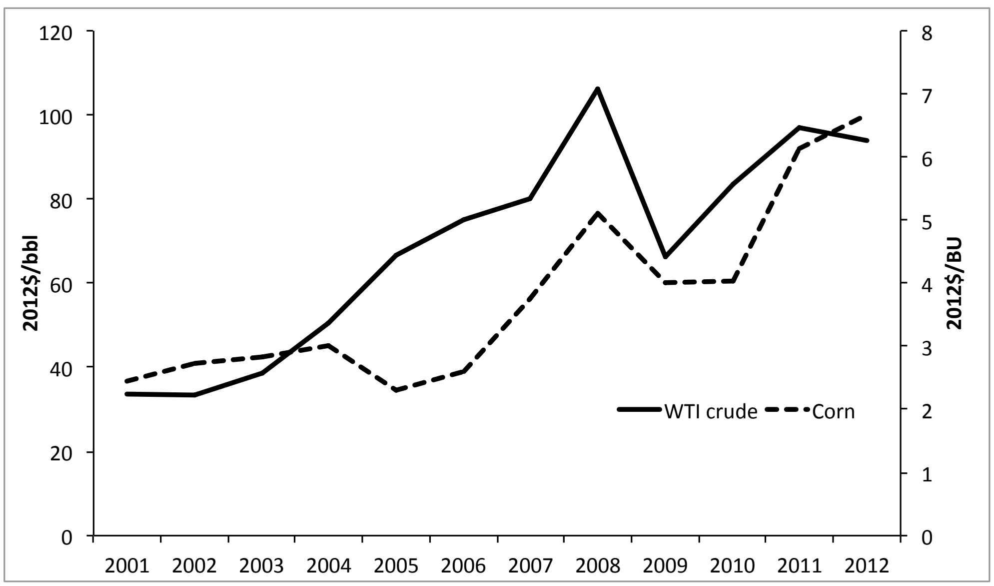
Ethanol production may impact food prices in two ways; either by allocating more food (e.g., corn) to produce ethanol, or by changing agricultural land to more energy crops than food crops. Figure 17a shows annual U.S. corn production and planted area from 2001 to 2012. Figure 14a also shows the quantity of corn used for ethanol production. Figure 17b shows the correlation between corn prices and energy prices, West Texas Intermediate (WTI) is used as an indicator of energy prices. Corn production (total commodity supply available as food and ethanol and total growing area dedicated to food/energy supply) have been increasing over the past 10 years, except for year 2012 where the number of corn bushels decreased compared to the previous years but the area plated remained high (Figure 17a). The trend in ethanol production could mean that farmers are meeting increased corn demand by increasing corn area rather than increasing yields. Also, the portion of corn devoted for ethanol production has *drastically increased over the past 10 year reaching around 40% of the total corn production in year 2012* (Figure 17a). In theory, the increase in demand for ethanol results in higher prices for corn, which creates incentives for farmers to increase acreage. As more lands are used in corn production, fewer lands are available for other crops.

Increased demand for ethanol results in increasing prices for crops that compete with corn for the same land. While the increase in demand does have an effect on the price of corn (Figure 14b), is it not the only factor. A study by Monteiro *et al.*, (2012) examined the influence of the production of ethanol in Brazil and the U.S on relative food prices; which is the ratio between food price index (US corn index or Brazilian sugar index). The authors argue that the increase in global agricultural commodities can be attributed to many factors such as accelerated economic growth in developing countries and associated increases in income, leading to a higher demand for food and a shift in consumer preferences (e.g, more dairy, and meat). This leads to higher commodity pricing (Monteiro *et al.*, 2012).

The depreciation in the exchange rate between the U.S dollar and other currencies can cause the overall relative food prices to increase due to a cheaper dollar. High crude oil prices (Figure 17b) have also raised the agricultural costs such as chemicals, fertilizers, and transportations contributing to higher food prices (Monteiro *et al.*, 2012). They found that the Brazilian market share of ethanol has a positive and significant effect on relative food prices, while areas allocated for sugar cane has a negative and significant effect on prices due to expanding sugar cane supply (high correlation between ethanol and sugar cane production). The authors concluded that exchange rates and oil prices displayed an expected effect on corn commodity prices where exchange rate has a negative effect and oil price has a positive affect both statistically significant. On the other hand, the area allocated for corn has a positive but insignificant effect on food prices.



(a)



(b)

Figure 17: (a) Corn production; (b) WTI crude and corn prices. (EIA, 2012; USDA, 2013).

Methodology

The objective of this study is to assess the assumptions behind the biofuels polices mentioned above, mainly market price and reliability of supply of biofuels (ethanol) compared to fossil fuels and/or imported oil. The monthly (2006-2012) U.S gasoline and ethanol[4] prices were obtained from AgMRC (2013), and converted to December 2012 prices. A trend analysis is used to determine if the prices of gasoline and ethanol are changing over time. The literature is rich with studies that discuss the volatility of oil prices (Gardebroek & Hernandez, 2012; Narayan & Narayan, 2007; Park & Ratti, 2008; Plourde & Watkins, 1998; Regnier, 2006; Weron, 2000) just to name a few. A study by Plourde & Watkins (1998) compared the volatility of oil prices to other commodities. Gardebroek & Hernandez (2012) examined the volatility spill over between the ethanol market and the corn market. The authors did not find evidence to support the claims that ethanol price volatility can lead to volatility in the agricultural markets; however, they found that volatility in the agricultural markets could lead to ethanol price volatility. All of the studies mentioned above utilize the use of price returns (the change in price over two periods of time) and estimate the standard deviation as a measure of volatility. Modeling volatility is beyond the scope of this study, the objective is to assess historic price volatilities of ethanol, imported oil, gasoline, and West Texas Intermediate (WTI) crude, which is used here as an indication of crude oil prices traded in the U.S. The rate of return is described as the following:

$$R = lnP_t - lnP_{t-1} \qquad \text{(eqn 2)}$$

4 Price of ethanol is in gasoline gallon equivalent (GGE) units to correct for the energy content per volume between gasoline and ethanol.

Where P_t is the price of at time t. The benefit of using returns is the normalization of prices especially when more than one commodity is being compared. The studies mentioned above used log returns are used because prices usually show a log-normal behavior rather than normal. It is assumed that ethanol is made from corn or hay, hay being a surrogate for cellulosic crops. In this study the prices of ethanol and gasoline are compared with each other to determine their price volatility. Statistical tests are then used to determine if the volatilities are statistically different among the prices. The annual prices of ethanol ($/gallon) were obtained from Nebraska's energy office for years 1982-2012 (Nebraska Energy Office, 2013). Imported crude oil (U.S F.O.B) prices and gasoline (all grades) were obtains from EIA for the same period (EIA, 2012). All prices were converted to 2012 dollars.

The reliability of biofuels supply is measured using annual variability of using energy crops to produce ethanol. Corn yield is used as a measure of the reliability of corn ethanol. Since the production of cellulosic ethanol is still small compared to corn ethanol, also up to this point, this is no clear market trend as what type of grass will be used as the primary source of cellulosic ethanol. Thus, the reliability of cellulosic ethanol is measured thought hay yields in the U.S. Hay crops are grasses and other herbaceous plant that is grown for animal feed, hay is used in this study as a surrogate for cellulosic plants. In this study, the period of analysis is from 1980 to 2011 to include the earliest record of ethanol production to assess whether EISA 2007 mandates have affected the reliability of corn production; this study also compares the reliability of hay production.

A distribution curve was fitted to the data and a probability density function was estimated for each of the corn, hay, and oil supply disruptions.

Results and Discussion

Figure 18 shows gasoline and ethanol prices (2012 dollars). The average gasoline price over the entire period is slightly higher than the price of ethanol, \$2.50 per gallon and \$3.44 per gallon respectively. The ethanol price trend indicates the price has been decreasing over time, whoever gasoline still remains a cheaper alternative. The negative trend in the price of ethanol can be attributed to the increased supply (distillers) and decreased gasoline demand.

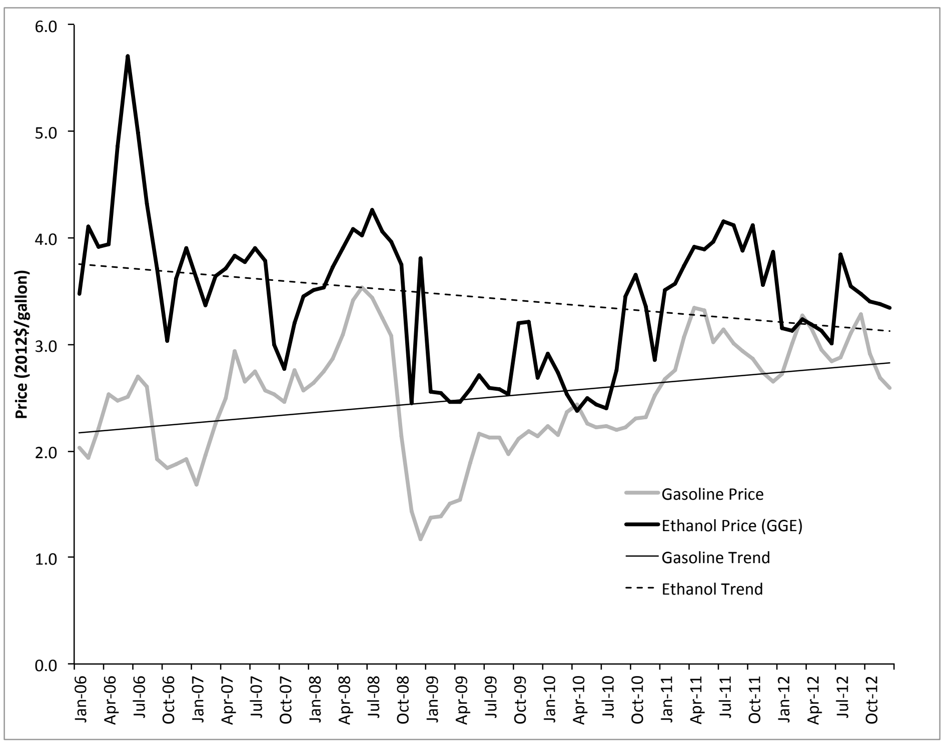


Figure 18: Gasoline and ethanol prices. (Ethanol* (\$/GGE). *Source:* (AgMRC, 2013).

Normality tests were performed on the price volatility dataset in order to ensure that the data is normally distributed since statistical tests for the equality of variance for two or more populations (e.q. ANOVA) are based on the assumption that the data is normally distributed. Levene's test shows that the data violates the classic ANOVA assumptions of equal variances. Thus, the non-parametric Kruskal-Wallis test is used. The results show that there is no significant difference (p value = 0.54) between the price volatility of ethanol, gasoline, imported oil, and WTI. Figure 19 shows the annual price rate of return of WTI, ethanol, gasoline, and imported oil with a mean of 0.042, -0.018, 0.006, and 0.009 respectively, and a standard deviation (volatility) of 0.214, 0.175, 0.122, and 0.271 respectively (Table 17). The results indicate that imported oil price has the highest average volatility among the other fuels. It was the highest during the period of 1980-1990, but became more stable over the last 12 years. Gasoline price is the least volatile throughout the period reaching a high of 0.143 in 2001-2012. From Table 17 the volatility of ethanol over the past 12 years is comparable to that of WTI and imported oil. While investigating the causes of these changes is beyond the scope of this study, Du *et al.*, (2011) and Wu *et al.* (2011) found evidence of spillover of crude oil volatility to the corn market in the recent years, especially after the introduction of the Energy Policy Act of 2005.

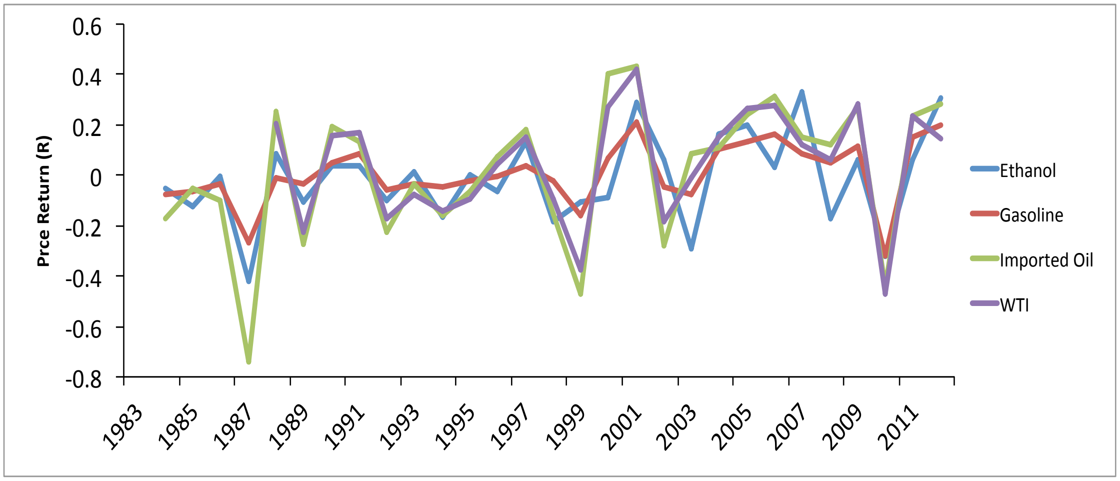


Figure 19: Price volatility, 2012$ (Nebraska Energy Office, 2013; EIA, 2012).

Table 17
*Volatility Estimates**

	WTI	Ethanol	Gasoline	Imported
Mean Return	0.042	-0.018	0.006	0.009
Standard Dev. (SD)	0.214	0.175	0.122	0.271
SD 1982-1990	0.176	0.150	0.099	0.299
SD 1991-2000	0.220	0.137	0.093	0.268
SD 2001-2012	0.212	0.207	0.143	0.224

Source: (Nebraska Energy Office, 2013; EIA, 2012).
*2012$.

The change in annual supply of corn yield and imported oil were found to be normally distributed using Kolmogorov-Smirnov test (p value = .2), while hay yield distribution was found to be weekly normal (p value =.038). The normal supply distribution statistics are shown in Table 18 and Figure 20. Other the past 30 years (1982-2012), the average percent change in annual corn yield is 1.6%, the high standard deviation is due to the large changes in annual yields from 1982-1996. The average percent change in hay production is -0.17%, and 2.1% for imported oil (Table 18). Assuming that historical data can be used to provide an indication of reliability, the lower limit of the confidence intervals (CI) can be used as worst-case scenario. The lower CI indicates that the percent change in annual corn yield is -4.2% whereas the percent change in annual imports is -0.5%. The probability of these changes is 2.31% for corn yield and 5.15% for imported oil. The results suggest that oil imports are more reliable than feedstock supply over the past 30 years. However, the reliability of corn has been improving since the late nineties (Figure 21).

Table 18

Fuel Supply Distribution

	Average % change	**Standard Deviation**	**95% CI**
Corn Yield	1.6%	16.1%	-4.2% – 7.5%
Hay Yield	-0.2%	6.2%	-2.4% – 2.1%
Imported OPEC Oil	2.1%	7.2%	-0.5% – 4.7%

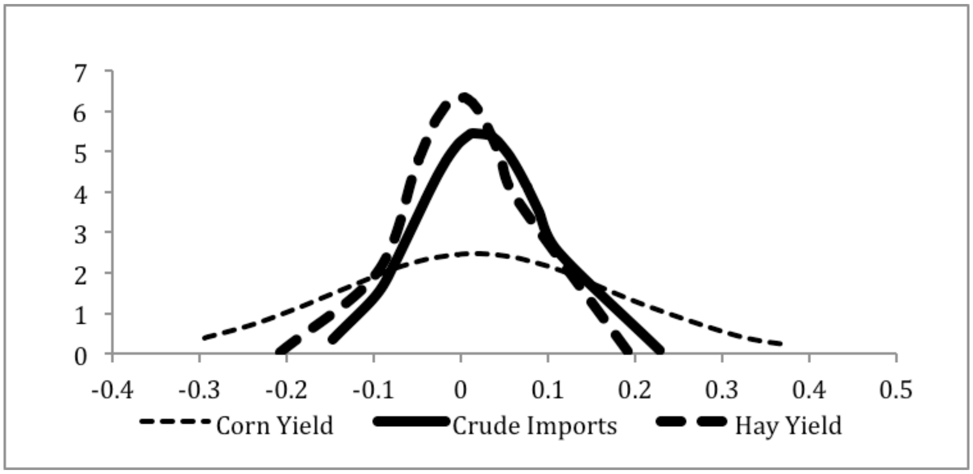


Figure 20: Probability density functions of quantity supplied

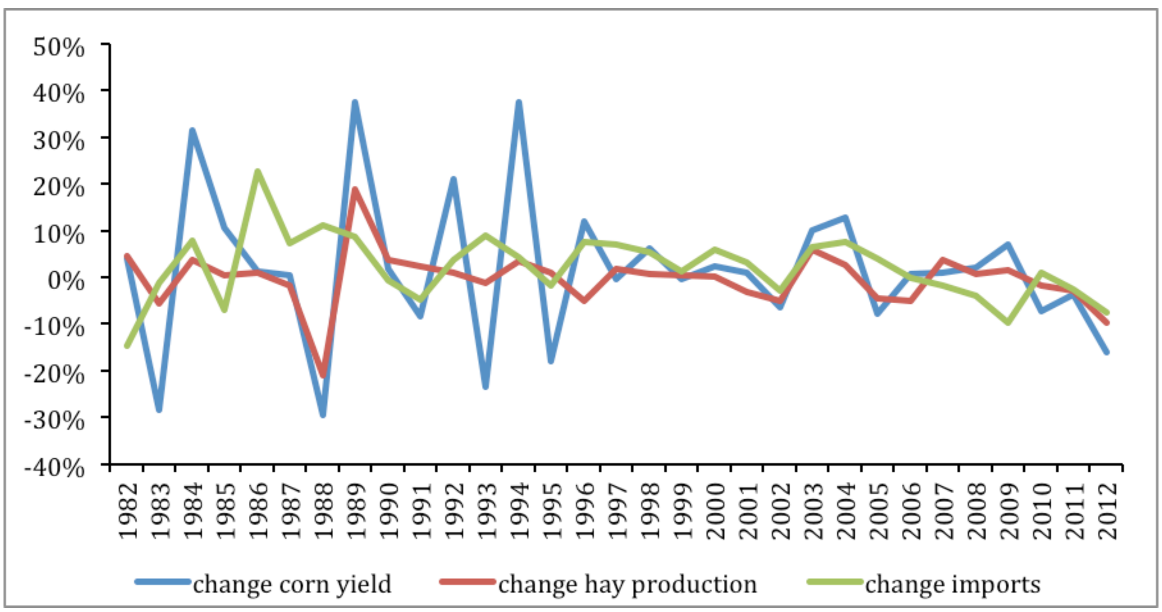


Figure 21: Annual supply disruption (USDA, 2012, EIA, 2012).

Conclusion

One of the main objectives of the Energy Independence and Security Act (EISA) is to promote energy independence through mandating that U.S. transportation fuel contains domestically produced biofuels, mainly ethanol. The majority of the energy consumed in the U.S comes from fossil sources; only 11% of the total energy consumption in 2011 is classified as renewable energy (biomass, geothermal, wind, solar and conventional hydropower). Ethanol accounts for 4.5% of the energy consumed in the transportation sector and it is not expected that ethanol share in the transportation sector will significantly increase by 2022 since 95% of gasoline sold in the U.S is E10 (90% gasoline 10% ethanol by volume).

Ethanol is used as gasoline additive rather than a substitute since most U.S. vehicles are not capable of using fuel blends higher than E10. Ethanol mandates are slated to increase 118% in the next 10 years (mostly from cellulosic sources), however it is still unclear how future supply of ethanol will be used, especially with measures such as CAFEs to increase efficiency and reduce gasoline demand. Additionally, advances in oil and shale-gas recovery technologies are expected to satisfy the domestic energy demands by 2030, thus raining the question whether biofuels will have a significant contribution in the U.S energy supply beyond 2020.

Based on the results mentioned above, ethanol price has been decreasing over the past six years, but remains more expensive than gasoline. The volatility analysis suggests that while the volatility of ethanol over the past 30 years is less than that of crude oil, the ethanol's volatility increased in the past 12 years and now it is comparable to crude oil. The literature studies suggest that crude oil volatility stilled over corn prices, which could explain the increased volatility in ethanol prices. The results also showed that there is no significant difference between the price volatility of ethanol, gasoline, and imported oil. These results were expected because fuel price are highly correlated with crude oil price, also because conventional fuels are used in the making of ethanol. Thus, ethanol does not impact gasoline price volatility unless it's made from energy sources that are uncorrelated with crude oil. The historical corn and hay supply is shown to be less reliable than imported oil, but it has been improving over time.

Ethanol is more expensive and less reliable than fossil fuels. Nevertheless, using ethanol in the transportation fuel supply has some benefits such as replacing MBTE and displaces a small fraction of the transportation fuel supply with renewable biofuels. However, the risk of energy (ethanol) supply disruption is now influenced by environmental factors rather than international political conflicts. Climate variability and extreme weather, such as droughts and floods, can highly affect agriculture thus making biofuels supply vulnerable to climatic events. A report by the IPCC concluded that on a global level, temperature increases could adversely affect crops in dry regions by increasing soil evapotranspiration rates and increase the chances of droughts (IPCC, 2007). In addition, the number of extreme precipitation events is expected to increase as a result of change in rainfall patterns. On a local level, the life cycle of crops would increase rapidly with increased CO_2 and temperature; however, more crops will begin to experience failure as temperature rises (USGCRP, 2008). More droughts and intense storms will increase erosion in arid lands. Finally, climate changes will likely decrease the vegetation cover that protects the soil from erosion (USGCRP, 2008). On the positive side, higher crop yields are expected over the first decades of the century due to increase in precipitation, CO_2, and temperature.

For an alternative energy source to contribute to energy security in the transportation, certain assumptions must be met. First, the fuel must be renewable and reliable. The characterization of ethanol as renewable is questionable, since it relies on a finite resource, farmland. Second, it must be economically feasible and decoupled from crude oil price volatility. Third, it must be an easy substitute for gasoline; it should not require massive changes to the installed base of internal combustion engines. The following analysis of the effect of EISA on energy security is framed by the foregoing three assumptions

One of the stated objectives of the Energy Independence and Security Act is to promote energy independence by mandating the use of domestic fuel sources, specifically biofuels. Based on the findings of the study, then it is more efficient to utilize abundant domestic natural gas to power motor vehicles. On the other hand, if the objective were to reduce the consumption of fossil fuels, then improving the efficiency of cars engines would yield better results. While is it unlikely that the U.S will stop producing ethanol (Ethanol is used as MTBE replacement), there is little support for making incremental investments in biofuels as for solution for energy security.

CHAPTER 5

SUMMARY AND CONCLUSION

Persistently high crude oil prices and a desire for the U.S to become energy independent led to the passage of Energy Policy Act of 2005 and Energy Independence and the Security Act (EISA) of 2007. These acts created a series of financial incentives and mandates to promote the use of renewable fuels, primarily ethanol, to meet transportation energy demand. The EISA mandates that at least 36 billion gallons of biofuels, mostly ethanol, be used by in the transportation industry by 2022. Existing and new farmland are being used to produce the corn and other feedstock necessary to create mandated biofuel. While biofuels currently account for a small portion of total U.S. energy sources, meeting the ESIA biofuel mandates will represent a massive increase in nonpoint source pollution (e.g. fertilizer and pesticides) runoff to waterbodies. There is little research on the potential environmental impact on water resources of meeting the ESIA biofuel mandates.

The purpose of EISA is to promote energy independence, improve environmental quality, and protect the consumer from the volatility of crude oil prices. However, the simultaneous policy imperatives of energy independence and environmental sustainability create complex and interacting environmental, economic, and political trade-offs. Many studies (Luchansky & Monks, 2009; Rask, 1998; Solomon *et al.*, 2007) have attempted to measure the impact of increased production of ethanol on the environment, economy, and energy security; however, few studies measure the impact environmental impact and costs of nitrogen runoff associated with the production of corn and other biomass. One of the EISA biofuels goals is to reduce greenhouse gases (GHGs) but the implementation of the EISA mandates is projected to create massive increases of nitrogen runoff. The potential economic and environmental impacts of producing ethanol from feedstock such as corn, EISA capped corn ethanol at 15 billion gallons by 2015. EISA now mandates that future ethanol be produced from "advanced" or cellulosic sources such as switch grass and corn stover, and some biodiesel.

The objective of this EISA study was twofold. First, the study measured the economic and environmental impact of nitrogen runoff on nation-wide water resources to from crop production to meet the EISA 2022 biofuel mandates. Second, the study evaluated the potential effectiveness of EISA 2022 mandates on energy security due to replacing oil with biofuels to meet the energy needs of the transportation sector. The results of these studies are summarized below.

Nitrogen Loads Resulting from Biofuels Mandates

One objective of this study was to use the SPARROW modeling method to predict nitrogen fluxes in the Mississippi River basins based on production to meet the EISA 2022 biofuels mandate. SPARROW model inputs included 2002 land use, climatic conditions, and basin characteristics. Using the all-corn scenario, the model predicted that biofuel production would increase nitrogen flux in the northern Gulf of Mexico from 270 to 1,742 thousand metric tons, that is an increase from 21% to more than 100%

(2011). If corn feedstock were replaced with cellulosic (hay) ethanol or biodiesel to meet the 2022 mandate, nitrogen flux would be substantially reduced, however 25% more land is necessary than estimated by the 2022 EISA scenario. Using 100% switchgrass instead results in 3 times more nitrogen flux but requires 43% less land. The all corn ethanol for 2022 scenario mandates is expected to have double the nitrogen flux when compared to the EISA specified 2022 scenario. The feedstock used to meet the EISA biofuel standard creates massive tread-offs between land use and nitrogen runoff.

This study does not recommend any one scenario, but rather creates a cost-benefit matrix for use by policy makers implementing EISA. The optimal strategy is a function of the value of the resources impacted (land and water) used to produce the biofuels. It is clear from the study that replacing oil with biofuels using Midwest farmland has substantial and measurable economic costs (land and crop) and environmental (water quality) impacts. Further, it would likely lead to more eutrophication in streams and hypoxia in the Gulf of Mexico and adversely affecting Mississippi River Basin ecosystems and the fishing industries.

Estimating the Economic Cost of Nitrogen Fluxes

from Energy Crops Production

One objective of this study was to estimate the total private (cost of production) and external (environmental damages) cost of nitrogen (atmospheric and water) associated with meeting the EISA 2022 mandate using hypothetical scenarios, such as all corn, all cellulosic, or all biodiesel mandates. The "all corn" scenario releases the most nitrogen to the environment, mainly due to high fertilization rate and high runoff rates relative to other crops. The total externality (atmospheric and water damages) costs for each scenario are shown in Figure 11. The results show that the costs are significantly different (six folds) between the two cases mainly due to the wide range of expected cost of nitrogen damages. However, water externalities account for the majority of the cost in both cases. Producing ethanol from cellulosic courses is expected to minimize negative externality while corn ethanol is expected to have the largest externality.

The study suggests that cellulosic ethanol would be the best approach for reducing both private costs and environmental impacts of biofuel production. However, cellulosic ethanol is not economically competitive yet. Based on currently available feedstock and production methods, the most realistic case (high production cost and low externality cost) is the use of corn ethanol to meet the EISA 2022 biofuel mandates. Until cellulosic ethanol is competitive, we recommend the creation of incentives to reduce environmental externalities (nitrogen costs) associated with corn ethanol. While the markets are incapable of accounting for environmental these externalities, the government can provide economic incentives to shape behavior.

Ideally, incentives to reduce nitrogen runoff would be implemented at the field level, however the administrative costs of measuring monitoring outputs are prohibitive (Ribaudo *et al.*, 1999). Design-based (expected runoff) incentives may be observed by a resource management agency through runoff simulations that are based on input and technology. The optimal incentive would equal the marginal increase in runoff at each field; however, it is not realistic and very costly to implement site-specific incentives (Ribaudo *et al.*, 1999). Nitrogen use could be discouraged using a non-regulatory tax scheme that penalizes residual nitrogen leaching into groundwater and subsidizes crops that capture and utilize residual soil nitrogen (Huang & LeBlanc, 1994). Wu and Tanaka

(2005) evaluated several methods for discouraging nitrogen use and found that the fertilizer-use tax is the most cost effective policy, but very difficult politically to institute.

Contribution of Biofuels Mandates to Energy Security

One of the main objectives of EISA is to promote energy independence by mandating that U.S. transportation fuel contain domestically produced biofuels, mainly ethanol. However, some basic supply and demand forces makes the probability remote that the EISA 2022 mandated biofuel volume increase of 118% between 2013 and 2022 will occur. Ethanol accounts for 4.5% of the energy consumed in the transportation sector and it is not expected to significantly increase by 2022 since 95% of all gasoline sold in the U.S is already E10 (90% gasoline 10% ethanol by volume). Most vehicles sold in the U.S cannot utilize fuel blends higher than E10. Ethanol mandates are expected double in the next 10 years (mostly from cellulosic sources), however it is still unclear how future supply of ethanol will be used, especially with measures such as CAFEs to increase efficiency and reduce gasoline demand. Additionally, advances in oil and shale gas-recovery technologies are expected to satisfy the domestic energy demands by 2030, thus raining the question whether biofuels will have a significant contribution in the U.S energy supply beyond 2020.

In terms of energy security, ethanol has a different risk profile than imported oil. Supply disruption is by environmental factors rather than international political conflicts. Climate variability and extreme weather, such as droughts and floods, provide the primary risks for supply disruption. A report by the IPCC concluded that on a global level, temperature increases could adversely affect crops in dry regions by increasing soil evapotranspiration rates and increase the chances of droughts (IPCC, 2007). In addition, climate change could change in rainfall patterns, rapidly increase CO_2 and temperature, create more droughts and intense storms, increase erosion, and decrease the vegetation cover (USGCRP, 2008). On the positive side, higher crop yields are expected due to increase in precipitation, CO_2, and temperature.

In order for ethanol to contribute to energy security in the transportation sector, it should have a number of attributes. First, it must be renewable and reliable, even though the "renewability" of biofuels is questionable since many of them require the utilization of agricultural soil, which is a finite resource. Second, it must be economically feasible and decoupled from crude oil price volatility. Third, it must be used as substitute for gasoline, which requires modifications to the internal combustion engines and major land use change to meet the demand. Thus, it is important to clarify the purpose of any energy policy before analyzing the potential sources of fuels. In order for the U.S. to meet its goals in energy security and environmental conservation, a clear policy will ensure that biofuel production maximizes the benefits of energy and environmental gains cost effectively.

If the U.S. policy objective were to replace imported oil with domestic sources of energy promote the use of domestic fuel, then it would be more efficient to use locally produced fossil fuels such as natural gas to power motor vehicles. On the other hand, if the objective were to reduce the consumption of fossil fuels, then improving the efficiency of cars engines would yield better results. While is it unlikely that the U.S will stop producing ethanol (Ethanol is used as MTBE replacement), investing in higher ethanol blends does not seem to be an efficient solution to energy security. With new oil discoveries and innovative technologies, the U.S can reducing its dependence on foreign fuel, however, that does not necessarily translate into cheaper energy prices because the U.S energy market is not isolated from global energy markets.

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