EVALUATING EUTROPHICATION POTENTIAL OF BIOPRODUCTS USING LIFE CYCLE ASSESSMENT METHODS

by

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Submitted to the Graduate Faculty of

Swanson School of Engineering in partial fulfillment

of the requirements for the degree of

Doctor of Philosophy

University of Pittsburgh

2011

UNIVERSITY OF PITTSBURGH

SWANSON SCHOOL OF ENGINEERING

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March 17, 2011

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University of Pittsburgh, 2011

Reactive nutrients are accumulating rapidly in the environment due, in part, to increasing demand for food and energy products derived from agriculture. Recently, biobased fuels from renewable resources have gained high development priority due to national energy security policies and to their potential carbon emission reduction compared to their petroleum counterparts. However, biofuels from first generation feedstocks (e.g., corn and soybean) exhibit a significant environmental tradeoff in the form of increased water quality degradation (i.e. eutrophication and hypoxia). To mitigate eutrophication resulting from increased agricultural production, it is important 1) to identify eutrophication potential of the main bioproducts including biofuels and foods; and 2) to evaluate the effectiveness of possible mitigation strategies. Multiple strategies exist for reducing nutrient loading including optimizing farming practices and encouraging consumers to purchase low nutrient intensity bioproducts. This research quantified the life cycle nutrient flows and environmental impacts of foods and biofuels, and subsequently evaluated the mitigation potentials of management strategies.

Research results show that different food groups exhibit highly variable nitrogen-intensity, on average, red meat and dairy products require much more nitrogen than cereals/carbohydrates. The ranking of foods' nitrogen footprints is not consistent with their

carbon footprint. For example, dairy products and chicken/eggs have relatively high nitrogen footprint and low carbon footprints.

The life cycle assessment of biodiesels in Pennsylvania exhibits that fertilizer usage in the agricultural phase and fuel combustion in the use phase are main contributors to biodiesel's life cycle environmental impacts for all blends. Comparing biodiesels with conventional diesel, environmental tradeoffs exist between global warming potential and eutrophication potential. Local scouring of biodiesels has the lowest environmental impacts for B20 and B100.

Dietary shifts from dairy products and red meat to cereals can be an effective approach for lowering the personal nitrogen footprint. Altering farming practices (including shifting conventional tillage to no tillage, using manure, installing buffer strips surrounding farmlands etc) could reduce environmental impacts of bioproducts from life cycle perspectives too. The life cycle assessment analysis of bioproducts suggests environmentally benign farming practices and consumption shift to low nitrogen intensity foods to mitigate eutrophication issues.

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PREFACE

I am indebted to many people for the successful completion of this document. I am grateful for the generous support of my advisor, Dr. Amy Landis, who has been with me throughout the years as mentor and friend. I also extend special thanks to Dr. Melissa Bilec and Dr. Joe Marriot who have been great sources of advice. I thank the other members of my committee, Dr. Emily Elliott and Dr. Leonard Casson for their valuable input into this research.

The research has been funded in part by Mascaro Center for Sustainable Innovation at the University of Pittsburgh, the Pennsylvania Department of Transportation, and the National Science Foundation.

There are many other people without whom I would never have made it to the end of a successful graduate career. Thanks to my friends and colleagues in Sustainability and Green Design Group. They are Neethi Rajagopalan, Kullapa Sortatana, Bill Collinge, Scott Shrake, Can Aktas, Briana Niblick, Claire Antaya, Xi Zhao, and others. Thanks to my best friends, Jeremy Romeiko and Timothy Passaro, without whom I would never have reached the home stretch.

During my graduate student tenure, I have had many gifted advisors and mentors. For their friendship and guidance I would like to thank Dr. Radisav Vidic, Dr. Jason Monnell, Dr Vikas Khanna and other faculty members at Department of Civil and Environmental Engineering at University of Pittsburgh.

Finally, I owe my greatest debts to my family. I thank my parents for life and strength and determination to live it. Special thanks to my brother, Xiaozhen Xue, who has been there for me and supports me all the time.

1.0 MOTIVATION AND INTRODUCTION

Reactive nutrients are accumulating rapidly in the environment in response to population growth and associated activities, with agriculture and fossil-fuel combustion serving as the primary sources of increasing nutrient loads to watersheds. Nutrients are the major sources of water quality impairments in the U.S. According to the National Water Quality Inventory, EPA recognizes agricultural systems as the third largest source of impairment to surveyed estuaries, as a leading source of water quality degradation to surveyed rivers and lakes, and also as a major contributor to ground water contamination and wetlands degradation (EPA 2004). Nitrogen and phosphorus can accelerate the eutrophication process and result in hypoxia. Eutrophication is caused by the depletion of dissolved oxygen in water bodies and results in increased costs and difficulty in purifying drinking water as well as impairing the navigational and recreational use of waterbodies (Rabalais, Turner et al. 2001; Scavia and Bricker 2006).

Water bodies affected by nutrient pollution range from small lakes and reservoirs to bodies of national significance such as the Chesapeake Bay and Gulf of Mexico. Row crop production in the Corn Belt of the U.S. contributes the highest fluxes of nitrogen and phosphorus to the Mississippi River Basin and is considered as one of primary contributors to the growing hypoxic zone in the Gulf of Mexico. The size of the hypoxic zone across the northern Gulf of Mexico was estimated to be about 7903 square miles in 2007 by NOAA (NOAA research 2008; USGS 2008). The Mississippi River/Gulf of Mexico Watershed Nutrient Task Force (MR/GOM

WNTF) established a goal to reduce the hypoxic zone size to 5000 km^2 by 2015 as compared to the current five-year running average (2003-2007) of 14,600 km^2 (EPA 2008). To date there has been little evidence of progress toward this goal and there is concern that increased agricultural production may further hinder achievement of hypoxic zone reduction.

Agriculture is a major source of nutrient pollution. Monitoring in the National Water Quality Assessment Program found that the highest concentrations of nutrients in streams occur in agriculture basins. High concentrations of nitrogen inputs from fertilizers and manure used for crops and from livestock wastes were reported (USGS 1996). With the growth of biofuels and increasing demand for foods, eutrophication and hypoxia are likely to be exacerbated in the near future.

Fertilizer application, which is essential to ensure ample nutrition for crop production, results in significant water discharge of nitrogen and phosphorus from farmlands. Nutrient-enhanced primary production or eutrophication of estuarine and coastal waters is a key cause in the incidence of harmful algal blooms, oxygen depletion, and overall fisheries habitat decline. These impairments will increase in frequency and severity in receiving Atlantic and Gulf of Mexico coastal waters that are already stressed by the unwanted symptoms of nutrient over-enrichment. Over 60% of coastal rivers and bays in the US are moderately to severely degraded due to nutrient enrichment (EPA 2009). Although excessive nitrogen loading is the main culprit in estuarine and coastal eutrophication, phosphorus loading leads to severe degradation and impairment in freshwater lakes, rivers, and some estuarine and coastal waters, especially those also receiving high nitrogen loads. There is growing recognition, however, of a need for an expanded and complementary understanding of the sources and transport of both nitrogen and phosphorus and their complex interactions toward developing effective nutrient management

plans in coastal waters. Modeling nutrients flows in agriculture system provides basic information for applying pollution prevention technologies and making related policies for combating eutrophication problems.

Biofuels are gaining more attention due to relatively low global warming air emissions, increasing energy demand and national security concern. In the U.S., The Energy and Independence and Security Act of 2007 (EISA 2007) mandated the annual production of 56.8 billion L of ethanol (15 billion gallon/yr) from corn by 2015 and an additional 60.6 billion L (16 billion gallon/yr) of biofuels from cellulosic crops by 2022. The EISA requirements virtually guarantee a large increase in biofuels production. The continued growth could have far-reaching environmental and economic repercussions and it will likely highlight the independence and growing tension between energy and water security (Landis, Miller et al. 2007; Miller, Landis et al. 2007; Simpson, Sharpley et al. 2008).

Ethanol production in the U.S. is currently concentrated in the Midwestern states of Illinois, Nebraska, Iowa, Minnesota, South Dakota, Wisconsin, and Kansas. According to historical information and current policy, the Center for Agriculture and Rural Development projected large increases in corn prices and acreage and proportionally greater use of corn for ethanol production (Center for Agricultural and Rural development 2009). The rapid growth of grain-based ethanol production has major water quality implications for lakes, rivers, and coastal marine ecosystems in much of the USA, particularly along the Northern Gulf of Mexico and Atlantic Seaboard, including the two largest estuaries, the Chesapeake Bay and Albemarle-Pamlico Sounds. There is concern that the increase demand of biofuels and foods would worsen the eutrophication issues in water bodies in U.S.

Scientific strategies should be taken to reduce nutrient loads to surface waters from agricultural products. Multiple strategies could be used to mitigate eutrophication issues from life cycle perspectives. Emissions resulting in eutrophication potentials could be reduced for each life stage including agricultural production, foods/biofuels processing, transportation, consumption stage. These strategies include optimizing farming practices, improving processing technology, changing transportation modes, reducing transportation distances, installing buffer strips, altering food consumption patterns, using advanced water treatment facility, etc.

One such strategy is to induce changes in the way nutrients are managed in the field. A second is to intercept nutrient-laden runoff and filter out the nutrients before they reach surface waters. Indeed, integrating and optimizing farming practices will enable the mitigation of environmental degradation and reduce biobased products' environmental footprints (Barling and Moore 1994; B.-M. Vought, Pinay et al. 1995; Wood, Wood et al. 1999; Bundy, Andraski et al. 2001; Dinnes, Karlen et al. 2002; Gareau 2004; Gregory, Shea et al. 2005; Tong and Naramngam 2007; Triplett and Dick 2008). There are several well-known farming practices that have potential to alleviate bioproducts' environmental impacts resulting from agriculture; they include tillage practices, choice of fertilizer type and the use of buffer strips. Research results show that no tillage and appropriate use of fertilizer on the field can reduce nutrient runoffs to certain extents (Gascho, Davis et al. 1998; Wood, Wood et al. 1999). Buffer strips built surrounding farmlands have the capability to sequester nutrients and decrease nutrient delivery to waters (Barling and Moore 1994; Dosskey 2001; Turner and Rabalais 2003). Indentifying best farming practices in Midwestern states of U.S. is an important aspect to ensure sufficient biofuel feedstocks and minimum environmental burdens.

Changes to environmental impacts of bioproducts can be achieved by implementing best farming practicing within agriculture, as discussed previously, or by encouraging consumers to purchase and utilize products with low nutrient profiles. Food consumption patterns also significantly influence nutrient outputs. Vast quantities of food are demanded to satisfy basic human needs every day. Food production, processing, packaging and transportation activities have significant social, economic and environmental impacts. Farming systems, as a primary stage of food production, are widely recognized as an important contributor for water quality degradation. Nitrogen and phosphorus emissions in agriculture come from variable sources, and contribute to eutrophication potential to different extents (Miller, Landis et al. 2007), (Miller, Landis et al. 2006). Food processing industries generate large amounts of organic materials such as protein and lipids, emit high biochemical and chemical oxygen demands (BOD and COD), and are a source of considerable nitrogen emissions to both the air and water (Tusseau-Vuillemin 2001). Food choices offer a unique opportunity for consumers to lower their personal environmental footprints. Concerned consumers are calling for mitigating the environmental burden of food supply. Policymakers and producers therefore require scientifically defensible information about food products and production systems. This work quantifies nitrogen and phosphorus flows over the life cycle of the food production system, and analyzes possible solutions to manage consumption of nitrogen in order to mitigate associated environmental consequences.

This research evaluated the eutrophication potential of bioproducts using a life cycle approach. To this end, we 1) to quantify nutrient flows for the main bioproducts including biofuels and foods; 2) to identify eutrophication potential of bioproducts; and 3) to evaluate the effectiveness of possible mitigation strategies. This research quantified the life cycle nutrient

flows and environmental impacts of foods and biofuels. The effects of shifting food consumption patterns on nutrient discharge were investigated in this study. Finally, this research evaluates the mitigation potentials of different nutrient management strategies.

2.0 RESEARCH GOALS AND OBJECTIVES

To mitigate eutrophication potential resulting from intensified agricultural production system, it is important to quantify nutrient flows for bioproducts including biofuels and foods; to identify eutrophication potential of these bioproducts; and also to evaluate the efficiency of possible management strategies. One strategy, optimizing integrated farming practices will reduce nutrient emissions from supply's perspectives. A different strategy, encouraging consumers to purchase low nutrient intensity bioproducts, can reduce nutrient outputs from the demand side perspective. This research analyzed and evaluated mitigation potential of both types of strategies and suggested a portfolio of methods to improve nitrogen footprint of biofuels and foods. The specific objectives were:

1) To establish life cycle nutrient models for simulating and quantify nutrient runoff from agriculture activities aimed at biofuels and foods production,

2) To evaluate the eutrophication potential of food types and determine the reduction of eutrophication potential as result of food consumption shifts,

3) To examine the environmental impact resulting from agriculture for biofuels production in Pennsylvania, and

4) To identify and evaluate agricultural best management practices that reduce the life cycle environmental burden attributed to bioproducts.

3.0 BACKGROUND AND LITERATURE REVIEW

Increase demand for grain production led to nearly 7 million ha or a 15% increase in corn acreage in the U.S. from 2006 to 2007 (USDA-NRCS 2007). Corn planting projections for 2007 indicate much of this increase comes from continuous corn replacing soybeans, with additional acreage coming from land currently in the Conservation Reserve Program (CRP), hay and pasture. Wisner (Wisner 2007) projected that up to 2.9 million ha of CRP land may be converted to corn production. Elobeid et al. (Elobeid, Tokgoz et al. 2006) estimated an ethanol-related long-term increase of 7.3 million ha in corn acreage. Continuously expanding land use for biofuel production may increase N and P discharge to surface water and ground water (Costello, Griffin et al. 2009).

Corn is an inherently inefficient nitrogen user in that 40% to 60% is generally not taken up by the crop, and N loads to downstream aquatic ecosystems from corn-dominated landscapes are typically 20-40 kg N ha^{-1} yr^{-1}. Soybean averages 15-30 kg P ha^{-1} yr^{-1}. For phosphorus, average losses in runoff from corn (2-15 kg P ha^{-1} yr^{-1}) tend to be greater than from soybean (1-8 kg P ha^{-1} yr^{-1}) (Miller, Landis et al. 2006), (Powers 2007). The loss of phosphorus from perennial and hay crops is generally less than annuals due to decreased runoff volumes and lower crop phosphorus requirements contributing to smaller amounts of phosphorus (fertilizer or manure) being added. Water quality model simulations of converting CRP or perennial grasses to cropland confirm that delivered nitrogen and phosphorus loads increase by more than double the

percentage land area converted. Assuming fertilizer application rates remain the same, annual nutrient loads are estimated to increase by 117 million kg nitrogen (37% increase) and 9 million kg phosphorus (25% increase) (Simpson, Sharpley et al. 2008). Most of the change will occur in the Mississippi River Basin (MRB), and once fertilizer leaves fields in this basin, most of the nitrogen and phosphorus are delivered downstream to the Gulf of Mexico.

Simpson et al estimated 80% of increase in corn production will occur in the MRB and that in-river delivery of nitrogen and phosphorus is about 70% of edge of stream inputs from agricultural fields. Also, the US Geological Survey model shows very little retention in the Mississippi River, once nitrogen and phosphorus get into the river system. Based on a 5 year rolling average load of 813,000 million ton N yr^{-1} and 154,000 million ton P yr^{-1} and assuming that 80% of the projected 7.3 million ha of new corn is in the MRB, nutrient fluxes to the Gulf of Mexico compared with recent years would increase. Furthermore, if all other production decisions remain unchanged (which is unlikely), the conversion of 7.3 million ha of soybeans and perennial grass/meadows to corn will result in a major increase in N and P loads to ground and surface waters. The biofuel feedstock production would hinder the achievement national goals to reduce N and P loads from the MRB by 40% or more to reduce the size of bottom water hypoxia (<2 mg dissolved oxygen L-1), Or “dead zone,” in the northern Gulf of Mexico (Rabalais, Turner et al.; Rabalais, Turner et al. 2001; Rabalais 2002; Rabalais, Turner et al. 2002; Scavia, Rabalais et al. 2003; Scavia, Justic et al. 2004; Scavia and Bricker 2006; Scavia, Kelly et al. 2006; Donner and Scavia 2007; Scavia and Donnelly 2007).

3.1 NUTRIENT MODELS

Disturbances in the N and P cycles in particular suggest the need for critical examination of the environmental tradeoffs associated with increased bioproduction. Despite their importance, nonpoint sources of nutrients are generally not quantified in bioproduct studies because of data variability and uncertainty.

Currently, both process based models and statistical models have been used as tools to qualify nutrient outputs from agricultural systems. Process based models are generally mechanisms descriptive models and contain many variables such as: soil quality, weather conditions, farming practices, geographical information, etc. To reduce variability among parameters when process based models are used, researchers often prefer to limit the analysis to a relatively small geographic region, assuming that the relative uniformity of system variables, such as climate and soil type, will allow greater precision in inventory estimation. Process based models, such as SWAT, SPARROW, CENTURY, and EPIC etc. can be very effective to model nutrient fluxes in agricultural systems. However, the outputs data are only applicable to relatively small regions.

SWAT (USDA 2008) is the acronym for Soil and Water Assessment Tool. SWAT is a physical river basin model that was developed for the USDA Agricultural Research Service, by the Blackland Research Center in Texas. In the current modeling approach the catchment was divided into 43 hydrologic responses unites which consist of different combinations of the existing landcover and soil types. Nitrogen and phosphorus losses arising from these hydrological units were estimated for the period 1990-2001 through the simultaneous simulation of water and sediment processes that are closely linked to the nutrient processes. The model took into account soil temperature in order to quantify water and nutrient transport to deeper layers,

considering negligible downward movement when the soil temperature was under 0 °C. SWAT directly models the loading of water, sediment and nutrients from land areas in a watershed. However, nutrients load to the stream network from sources not associated with a land area for some watersheds.

EPIC (Environmental Policy Integrated Climate) is a continuous simulation model that has been used to examine long-term effects of various components of soil erosion on crop production. EPIC is a public domain model that has been used to examine the effects of soil erosion on crop production in over 60 different countries in Asia, South America, and Europe. The model is used to examine soil erosion, economic factors, hydrologic patterns, weather effects, nutrients, plant growth dynamics, and crop management. The major components in EPIC are weather simulation, hydrology, erosion-sedimentation, nutrient cycling, pesticide fate, plant growth, soil temperature, tillage, economics and plant environment control.

SPARROW (USGS 2009) is a watershed modeling technique for relating water-quality measurements made at a network of monitoring stations to attributes of the watersheds containing the stations. The cores of the model consist of a nonlinear regression equation describing the non-conservative transport of contaminants from point and diffuse sources on land to rivers and through the stream and river networks. The model predicts contaminant flux, concentration, and yield in streams and has been used to evaluate alternative hypotheses about the important contaminant sources and watershed properties that control transport over large spatial scales.

CENTURY (NREL 2009) is a model of terrestrial biogeochemistry based on relationships between climate, human management (e.g, fire, grazing), soil properties, plant productivity, and decomposition. This model simulates C, N, P, and S dynamics through an

annual cycle over time scales of centuries and millennia. The producer submodel may be a grassland/crop, forest or savanna system, with the flexibility of specifying potential primary production curves representing the site-specific plant community. CENTURY was especially developed to deal with a wide range of cropping system rotations and tillage practices for system analysis of the effects of management and global change on productivity and sustainability of agroecosystem.

Custom models, such as those based on emission factors and life cycle assessment, have also been developed. Miller et al. created a linear model coupled with Monte Carlo Analysis (MCA) to characterize nitrogen fluxes in corn-soybean agroecosystems (Miller, Landis et al. 2006). The model consists of 17 equations and 29 input parameters to determine exports from an agriculture system, using basic units of kg N/ha per year. For this model, the primary inputs of nitrogen into the system are defined as synthetic nitrogen fertilizer, biological nitrogen fixation, crop residues from previous years, and atmospheric nitrogen deposition. The exports included in the model are nitrogen in harvested grain, N_2O, NO_x (primarily as NO), NH_3, and NO_3^-. Nitrate emissions from corn fields are calculated using a fraction of applied fertilizer. To determine the nitrate load during soybean growing seasons, this model uses mineralized nitrogen multiplied by the nitrate conversion rate for fertilizer. However, this model cannot estimate nitrogen flows of manure fertilizers and phosphorus flows.

Powers also developed a linear tool to enable improved estimates of non-point source nutrient flows from row crop production that could be integrated into biodiesel for ethanol biofuels life cycle assessment studies (Powers 2007). Nutrient flows to the Mississippi River were calculated with a standard emission factor approach with the adaption that the emission factors varied as a function of rainfall. The general leaching models for total nitrogen and total

phosphorus are defined based on a fraction of the applied nutrient load. The nitrogen model is amended to account for the subsequent loss of nutrient via denitrificaiton in tile drains and local streams. The incorporation of annual variability in nutrient loads due to rainfall has not been used before with life cycle assessments for non-point source pollutants and provides an excellent approach for quantifying the true variability. But the specific linear model presented to estimate the fraction of fertilizer that leaches should not be applied directly to other locations.

Previous research and nutrient management practices mainly focused on nitrogen nutrients. Phosphorous may play a more significant role in the formation of hypoxia than previously thought. Understanding phosphorus delivery to ecosystem from farmlands is critical to establish management strategies to combat water quality degradation. Advanced statistical models which depict phosphorus flux in farming ecosystem will provide important scientific evidence to environmental management.

3.2 LIFE CYCLE ASSESSMENT

Life cycle assessment (LCA) is a systematic approach to analyze and assess the environmental impacts of a product or process over its entire life cycle. A typical LCA includes the major stages of a products life including raw material extraction, manufacturing, use and end-of-life. Guidelines for performing an LCA are delineated by the American National Standards Institute (ANSI) and International Organization of Standardizations' (ISO) 14040 series (ISO 1997). LCA is an iterative four-step process including 1) goal and scope definition, 2) life cycle inventory analysis, 3) life cycle impact assessment, and 4) interpretation.

The first step of an LCA, goal and scope definition, defines the extent of analysis and the system boundaries. The Inventory analysis documents material and energy flows which occur within the system boundaries (also called the life cycle inventory or LCI). Impact assessment characterizes and assesses the environmental effects using the data obtained from the inventory (also called the life cycle impact assessment or LCIA). The final stage, interpretation and improvement identifies opportunities to reduce the environmental burden throughout the products' life.

LCA can be used in product development and improvement, strategic planning, environmental performance indicator selection and marketing. Recently, the LCA method has evolved as an important tool in improving the environmental performance of food production systems. The general purpose of LCA in food products is basically to identify the problem areas and possible options for environmental improvement. Comparative LCA studies have been used to evaluate different product systems or choice of management strategies to identify the most environmentally-preferred system or option. LCA results have been used as basic information to support decision making.

3.3 FARMING PRACTICES

While the environmental benefits of renewable fuels derived from corn and soybeans have been well documented (i.e. exhibiting reduced global warming and fewer fossil fuel requirements), the environmental tradeoffs have only recently been identified and quantified, recently. The recent growth in demand for corn and soybean as raw materials for renewable fuels is coupled with

growth in the size of hypoxic zone in the Gulf of Mexico and causes concern about the viability and sustainability of these traditional US agricultural feedstocks.

Best management practices (BMPs) provide a valuable tool for sustainable agricultural management. For example, agricultural nitrogen mitigation strategies may ameliorate the adverse environmental impacts of bioproduction. BMPs will include nutrient management strategies, reduced and conventional tilling, as well as crop rotations and crop covers. Three specific approaches have been identified for the mitigation of nitrogen emissions: reduction of the conversion rate of non-reactive to reactive nitrogen, increased efficiency of nitrogen use and plant uptake, and increased denitrification of reactive nitrogen that is not recycled. Recycling nitrogen by using manure as a fertilizer and increasing biological nitrogen fixation by growing rotating legume crops could lessen the need for synthetic fertilizers. Precision agriculture, such as site specific application of fertilizer to account for soil variability of altering the time of delivery, has proven to greatly improve nitrogen through improved fertilizer use. Denitrification of excessive of N that is unable to be recycled might be facilitated through best management practices such as grass buffer strips, improved tillage practices and constructed wetlands.

Tillage is often practiced as a first step in the preparation for a soil bed to be made suitable for seed germination and seedling development. According to the amount of plow surfaces and crop residues, there are three typical tillage methods (EPA 2009): conventional tillage, reduced tillage, and conservation tillage. Conventional tillage involves plowing the entire soil surface and leaving less than 15% of crop residue to cover the soil surface after planting. Reduced tillage utilizes a chisel plow to mix soil and crop residue, leaving 15%-30% residue coverage on soil. Conservation tillage includes ridge tillage, no tillage etc. No tillage means no tilling is done at all. Plants are placed directly into the previous season’s crop residue. Compared

to conventional tillage, no tillage often has more environmental advantages including surface runoff reduction and soil erosion mitigation (Triplett and Dick 2008). Other possible environmental benefits include energy and emissions savings resulting from less fuel consumption for operating farming equipment and associated air emissions (Kim and Dale 2005; Lobb, Huffman et al. 2007).

The choice of fertilizer types can affect the energy profile, greenhouse gas emissions, and aqueous emissions attributed to the environmental footprint of agricultural products (Dinnes, Karlen et al. 2002; Tarkalson and Mikkelsen 2004). Both synthetic fertilizers and animal manure are used to enrich soil nutrition within Corn Belt agriculture (Gessel, Hansen et al. 2004). Compared to manures, most commercially synthetic fertilizers contain higher nutrient concentrations by weight, more appropriate nutrient ratios (i.e. N:P:K ratios), and are more readily available to crops when applied to the soil. High gaseous carbon and nitrogen fluxes that result from handling and applying manures are serious concerns for the use of manure as a source of fertilizer (Amon, Amon et al. 2001). However, manure has the potential to adjust the soil carbon cycle and maintain soil fertility. Reusing manures, which are usually waste products of dairy or poultry farms, is often explored as an economical and sustainable alternative to synthetic fertilizers (Sims 1987; Bundy, Andraski et al. 2001).

Establishment of buffer strips is used as an important component of integrated farming nutrient management plans. Installing riparian buffer zones is a recognized agroforestry practice that not only provides phytoremediation for nonpoint source pollutants but also increases biodiversity of terrestrial ecosystems (Turner and Rabalais 2003). It can also provide stream ban stabilization, moderate flooding damage, control nutrient leaching, sequester carbon, and recharge groundwater, and provide recreational opportunities to landowners. Riparian zones

generally consist of two types: grass strips and wooded buffers. Woody vegetated strips can include shrubs and have advantages in controlling band erosion and providing biological abundance, while grass strips might be more acceptable in keeping with the original character of the landscape. Both woody vegetation and grass strips can uptake of nutrients to improve water quality (Loyon, Guiziou et al. 2007). Vegetative buffers are one management technique shown to reduce nutrient, pesticide, and sediment loads to waterways. However, it is unclear how much of the runoff within the MARB can be effectively treated with vegetated buffer strips.

Alternative agricultural management strategies have been studied widely in the agricultural literature, primarily focusing on narrow aspects of agriculture management, such as their affects on yields, nutrient leaching etc. However, the overall environmental impacts from agriculture and related management strategies, and thus their implications on the environmental footprint of bioproducts have not been explored.

3.4 EUTROPHICATION POTENTIAL OF FOODS

Tilman et al. forecasted dependences of the global environmental impacts of agriculture on human population growth and found that consumption will continue to increase; 109 hectares of natural ecosystems would be converted to agriculture by 2050 to meet food demand. This would be accompanied by a 2.4- to 2.7-fold increase in nitrogen- and phosphorus-driven eutrophication of terrestrial, freshwater, and near-shore marine ecosystems, and comparable increases in pesticide use. This eutrophication and habitat destruction would cause unprecedented ecosystem simplification, loss of ecosystem services, and species extinctions. Significant scientific advances and regulatory, technological, and policy changes are needed to control the

environmental impacts of agricultural expansion (Tilman, Fargione et al. 2001; Tilman, Socolow et al. 2009).

Nitrogen and phosphorus, as main nutrition components to ensure plant growth, are two basic elements that cause eutrophication and hypoxia. Food supply activities are simplified into four iterative phases: farming stage, processing stage, packaging and transportation stage. Every phase generates considerable amounts of nitrogen and phosphorus emissions into surrounding atmospheric, aqueous and solid environments. Farming system, as a primary stage of food production, is widely recognized as an important contributor for water quality degradation (Landis, Miller et al. 2007; Miller, Landis et al. 2007), (Miller, Landis et al. 2006). Nitrogen and phosphorus emissions in agriculture come from variable sources, and contribute to eutrophication potential to different extents. The largest source of nitrogen emissions by mass is nonpoint-source nitrate emissions, which contribute to eutrophication and hypoxia. Most NO_3^- emissions are generated after the harvest when no crops are available to absorb the available inorganic nitrogen pool. N species often undergo chemical or biochemical reactions. Atmospheric nitrogen is converted to a reactive form by natural processes, such as biological nitrogen fixation (BNF) and lightning, and anthropogenically via manufacturing synthetic fertilizer. BNF occurs in the presence of select plant species that host microorganisms able to convert N_2 to a reactive form. Cultivated crops that fix nitrogen include soybeans, rice, alfalfa, and most legumes. Reactive nitrogen compounds are also released directly from agricultural systems via nitrification/denitrification (NO_x, N_2O) reactions occurring in soils. Nitrogen compounds (NO_x, N_2O) are also generated from combustion processes associated with agricultural operations and processing of food. Aqueous and atmospheric nitrogen can potentially result in eutrophication. Inputs of phosphorus are essential for profitable crop and

livestock agriculture. Phosphate (PO_4^{-3}) runoff and leaching from feedlots, cattle feedlots, hog farms, dairies, and barnyards are also reported as contributors to eutrophication (Basset-Mens and van der Werf 2005; USDA 2009).

Food processing industries generate large amounts of organic materials such as protein and lipids, high biochemical and chemical oxygen demands (BOD and COD), and considerable amount of nitrogen concentrations. Wastes originating from food processing industries are either collected at a municipal wastewater treatment plant or treated locally in the plants before being released to the environment, which can induce direct point source pollution or an indirect diffuse pollution after sludge disposal (Tusseau-Vuillemin 2001). Food packaging and distribution stages also generate certain amounts of N/P emissions. Previously, the evaluation of the environmental performance of packaging usually concentrates on a comparison of different packaging materials or types of packaging designs. Eutrophication potentials of glass bottle, plastic products and metal cans have been investigated (Yoshio and Haruo 2000).

Food has long held a prominent place in the life-cycle assessment literature due to its relative importance for many environmental problems. However, most analyses have limited to detailed case studies of either a single food or a limited set of items, though usually to higher level of detail than is possible for large groups of products (Andersson 2000; Schau and Fet 2008). A few studies exist which look at overall diet but these have been focused on the ecological relevance of carbon footprint and food consumption pattern (Weber and Matthews 2008). The study of nitrogen and phosphorus inventories over all food categories has not been found yet. The potential of reducing nitrification through shifting food consumption pattern have not been addressed.

3.5 ENVIRONMENTAL IMPACTS OF BIOFUELS

Biofuels produced from renewable resources have gained increased research and development priority due to issues of national security regarding U.S. fossil fuel consumption and concerns about burgeoning greenhouse gas emissions (GHGs). Biodiesel can be derived from various biological sources such as seed oils (e.g. soybeans, rapeseeds, sunflower seeds, palm oil, jatropha seeds, and waste cooking oil) and animal fats. In the U.S., a majority of biodiesel is produced from soybean oil. Biodiesel can be blended with conventional diesel fuel in any proportion and used in diesel engines without significant engine modifications (Humburg 2006). In recent years, the sales volume for biodiesel in United States has increased - from about 2 million gallons in 2000 to 250 million gallons in 2006 (National Biodiesel Board 2009). The expansion of biodiesel production is still expected due to policy guidance and economic incentives. A series of policies have been implemented to stimulate biofuel production. The Renewable Fuel Standard 2 (RFS2) requires the use of 500 million gallons of biodiesel in 2009, increasing gradually to 1 billion gallons in 2012 in US (USEPA 2010). In Pennsylvania, the Penn Security Fuels Initiative required increasing percentages of biodiesel in all diesel fuel sold in the state coupled to the expansion of in-state biodiesel production, which in turn is subsidized by State policy (report 2008). B2 (2% biodiesel by volume) requirement was effective on January 1, 2010, with higher blending levels required in the future if production thresholds are met.

Although these policies provide strong incentives for biodiesel production, the environmental impacts of biodiesel in the US are still under scrutiny. Life Cycle Assessments (LCAs) have been used to quantify environmental impacts of biofuels over their entire life time. The guidelines for performing LCA were outlined by International Organization for Standardization (ISO 2006). Environmental impacts of liquid biofuels are investigated widely.

Argonne National Laboratory reported that soybean based diesels can save large amounts of petroleum use and achieve a significant reduction in global warming potential emissions compared with petroleum-based fuels (Huo, Wang et al. 2008). Sheehan et al. (1998) performed a life cycle inventory of biodiesel and petroleum diesel and concluded that biodiesel from soybean could reduce consumption of petroleum and would also reduce life cycle carbon dioxide emissions (Sheehan, Camobreco et al. 1998). Besides energy and global warming potential, water quality impacts from soybean farming also have been investigated (Miller, Landis et al. 2006; Landis, Miller et al. 2007; Powers 2007). The use of fertilizers to ensure growth of plants results in excessive nutrient runoff, consequently causing eutrophication and hypoxia issues. Landis et al (2007) and Powers (2007) analyzed the water impacts of soybean farming using life cycle approaches (Landis, Miller et al. 2007; Powers 2007). Eutrophication potentials of biofuels are much higher than their counterpart products, while biofuels have lower global warming potentials. Most of these studies focused on environmental impacts of biodiesel derived from soybean in Corn Belt states, which produced more than 70% of US soybean in 2007 (USDA 2008). Environmental impacts of producing and using biodiesels in other regions have not been adequately addressed.

The environmental impacts of soybean oil are significantly variable due to regional agricultural and production practices. Kim et al (2009) investigated cradle-to-gate green house gas emissions of soybean oil in 40 counties in Corn Belt States (Kim and Dale 2009). They discovered that GHG emissions of soybean oil in different counties can vary by a factor of 5. Meanwhile, aqueous emissions also have high uncertainty due to locations and farming practices. Miller et al (2006) investigated nitrogen fluxes in corn-soybean ecosystem and found nitrate

emissions from soybean farmlands vary from 13 to 22 kg nitrogen/hectare soybean/year (Miller, Landis et al. 2006).

Effects of substituting biodiesel for petroleum diesel in environmental impact areas other than global warming potential and eutrophication are less frequently reported. Sheehan et al. (1998) reported that use of soybean biodiesel reduced carbon monoxide, particulate matter, and sulfur oxides emissions. However, biodiesel increased nitrogen oxides and hydrocarbon emissions (Sheehan, Camobreco et al. 1998). Argonne National Laboratories' Greenhouse Gases and Regulated Emissions in Transportation (GREET) model reports criteria air pollutants (CAPs) in addition to GHGs, but does not inventory toxic air pollutants or emissions to water and soil, and does not provide an impact assessment for any emissions beyond total CO_2-equivalent GHGs (Argonne National Laboratory 2006).

Like GREET, most LCA studies evaluate the environmental impacts of biofuels from a GHG and energy standpoint. As discussed at length in the introduction; serious tradeoffs in the form of water quality degradation exist for agriculturally derived products. Thus, it is important to quantify the gamut of environmental impacts of both biofuels and food products.

4.0 EUTROPHICATION POTENTIAL OF FOODS

4.1 INTRODUCTION

Food production, processing, distribution and consumption activities have significant social, economic, and environmental impacts. While vast quantities of foods are required to satisfy basic human needs every day, food production also results in natural resources depletion, water quality degradation, and climate change. Nutrient fluxes from food supply chains have resulted in water quality degradation in the form of hypoxia and eutrophication causing loss of ecosystem services and species extinctions.

Global warming potential of food production and transportation system is reported widely for assorted food types and different food supply systems. Previous studies show that livestock production systems have higher carbon footprints than crop and vegetable production systems. “Food miles” research focusing on carbon emissions during food delivery stage advocates localization of global supply network. Recent research reported the carbon emissions of food choices, and discovered eating less red meat and dairy could be a more effective way to lower an average U.S. household’s food related climate footprint than buying local food.

Life cycle assessments of foods have been utilized to evaluate and improve the environmental performance of food production systems. LCA results have been used in the development of eco-labeling criteria with the aim of informing consumers of the environmental

characteristics of products. However, most analyses are limited to case studies of either a single food or limited set of items. A few studies researched overall diet but these studies focused on consumption patterns. The study of nitrogen and phosphorus inventories over all food categories has not yet been performed. Additionally, the effects of reducing eutrophication potential through shifting food consumption pattern have not been addressed.

Food offers a unique opportunity for consumers to lower their personal carbon and nitrogen footprints. Concerned consumers are calling for mitigating environmental burden of food supply. Policymakers and producers therefore require scientifically defensible information about food products and production systems. This study identifies nitrogen/phosphorus flows over food production, processing, distribution stages, and analyzes possible solutions to control consumption of nitrogen in order to mitigate associated environmental consequences.

The main content of this chapter was published in Environmental Science & Technology (Xue and Landis, 2010).

4.2 MATERIALS AND METHODS

4.2.1 System boundary

The life cycle stages vary with different food products. Generally, food LCA stages include farming production, food processing, packaging, and delivery. Figure 1 shows the boundaries for the researched food groups.

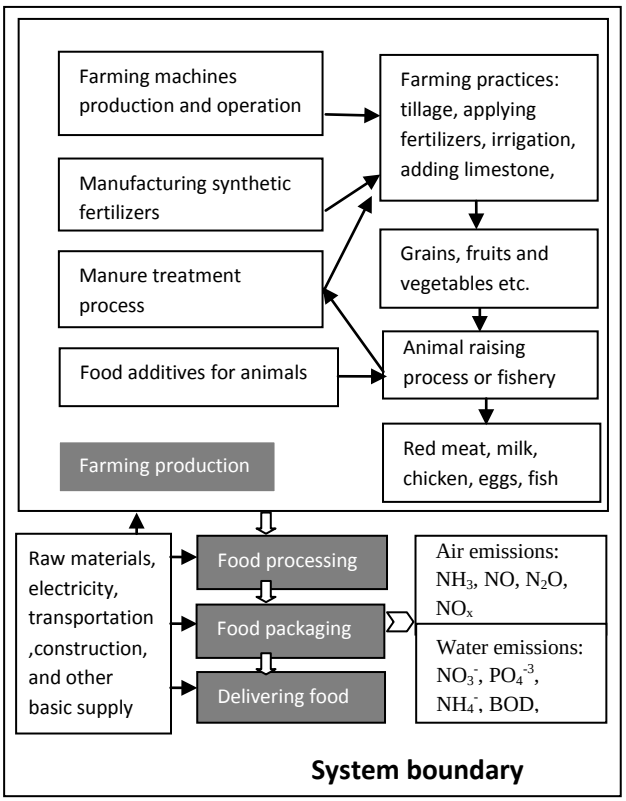

Figure 1. System boundaries of foods and the related nitrogen flows

4.2.2 Functional Units

The functional unit is the reference unit that forms the basis for comparison between different systems. Most published food LCA research uses mass or volume based functional units. In this article, kg food is defined as the functional unit to compare nitrogen profiles of different food groups. Normalized units (kcal food, $ food) are also employed to reflect the influences of economic value and energy content.

4.2.3 Life cycle inventory and impact analysis

Several existing tools were used to compile the LCI, including SimaPro and GREET. Simapro software developed by Pre consultants is expandable and transparent software that integrates inventory data for a broad spectrum of industrial and economic sectors (Pre consultants 2009). Greenhouse gases, Regulated emissions, and Energy use in Transportation (GREET) model created by Argonne National Laboratories delineates life cycle energy use and emissions of criteria air pollutants based on EPA emission factors of transportation stages (Burnham, Wang et al. 2006) .

While SimaPro software and associated databases including Ecoinvent V2 (Frischknecht, Jungbluth et al. 2005), LCA food (Food database 2007), Industry data 2.0 (Pre consultants 2009), BUWAL250 (Pre consultants 2004), IDEMAT 2001 (Pre consultants 2004) are used to compile nutrient inventory for the food processing and packaging stages, GREET is employed to account for inventory in the transportation stage. Additionally, the LCI for the agricultural stage was created using a variety of data collected from published articles and SimaPro databases. The LCI data sources are outlined in Table 1, while detailed data sources are illustrated in supporting information. Total nutrient output is equal to the sum of the nutrient flows from every stage of the food production system including, agriculture, processing, packaging and transportation.

Table 1. Stages of food LCA and associated emissions with eutrophication potentials

Stages	Emissions of concern	Database
Farming	NH_3, NO, N_2O, NO_x, NO_3, PO_4^{-3}, NH_4^-,BOD, COD	Peer reviewed articles*, Ecoinvent V2

Table 1. (Continued)

Food processing	NH_3, NO, N_2O, NO_x, NO_3^-, PO_4^{-3}, NH_4^-, BOD, COD	Peer reviewed articles*, Ecoinvent V2, LCA food, Industry data 2.0, BUWAL250, IDEMAT 2001
Food packaging	NH_3, NO, N_2O, NO_x, NO_3^-, PO_4^{-3}, NH_4^-, BOD, COD	Ecoinvent V2, LCA food, Franklin US 98, Industry data 2.0, BUWAL250, IDEMAT 2001
Transportation	NO, N_2O, NO_x	GREET 1.8

*the use of databases are explained in Appendix A.

Packaging and packing containers can be divided into two groups: commercial packages and transportation containers. Goods are packed in commercial packages in the small quantities required by the direct consumers. These packages protect the product; guarantee its quantities and composition, mode of use and, occasionally, its price. Packages are produced from different materials in a variety of types, e.g. bags, cartons, glassware, cans etc. This article only considers commercial packages (USDA 2002; Robertson 2006). Detailed assumptions of packaging materials for foods are given in supporting information.

Distances and transportation modes for delivering food subgroups are obtained from publications (Weber and Matthews 2008). All GREET default assumptions are followed to calculate gas emissions during transportation stage (Burnham, Wang et al. 2006).

The LCIA was conducted utilizing TRACI (Tool for Reduction and Assessment of Chemical and Other Environmental Impacts), which was developed by the USEPA (Bare, Norris et al. 2003). TRACI is used to calculate eutrophication potential for the system. TRACI defines characterization factors (CFs) relating nitrogen and phosphorus species to eutrophication

potential, thus allowing the LCI data to be expressed in terms of the TRACI defined reference compound, N-equivalents.

Monte Carlo Analysis (MCA) is used to quantify variability and uncertainty of LCI (Landis, Miller et al. 2007), (Miller, Landis et al. 2006; Costello, Griffin et al. 2009). Any independent variable with a range of estimates or possible values can be assigned a probability distribution. If ample data sets are collected for the independent variable, best-fit probability distributions can be determined using Anderson-Darling tests. Independent nitrogen equivalent values are collected or calculated for each LCA stage of every food group. Crystal Ball 7 software was used to define probability distributions of nitrogen equivalent values for every stage and to conduct the MCA (Miller, Landis et al. 2006). The distribution of output variables, as a function of independent values, is generated through MCA, which repeatedly and randomly samples values from the probability distributions of independent values. The distribution ranges of total nitrogen equivalent are determined by distributions of each stage's nitrogen equivalent value through MCA method.

4.3 RESULTS

4.3.1 Contribution to eutrophication potential at each life cycle stage

Figure 2 shows nutrient outputs for food groups. Red meat has the highest eutrophication potential, followed by dairy products, chicken/eggs and fish. Cereal/carbs subgroup is identified to have the lowest nutrient footprint among all food subgroups. While producing, processing, transporting and packaging 1 kg red meat generates 56.5-428.0 g nitrogen - equivalent emissions,

around 1.7-4.0 g nitrogen- equivalent emissions are released to supply 1 kg cereal/carbs. Agriculture stage is the largest eutrophication emission sector, which shares more than 70% of total eutrophication potential. Both of plant production and animal raising systems are reported to be responsible for eutrophication issues of surrounding water bodies. Corn and soybean farming systems, providing feedstock for human diet and animal feed, emitted large amounts of NO_3^- and PO_4^{-3} into groundwater and surface water (Landis, Miller et al. 2007). Manures from animal raising system, contain high nutrient contents, and are highly volatile. Atmospheric NH3 and NOx, (evaporated from manures) and aqueous N, P species (transformed or dissolved from manures) can significantly influence nutrients inventory of food supply chains. Nutrient footprints of red meat and dairy products include direct nutrient emissions from animal raising system and upstream nutrient emissions from plant production, so red meat and dairy products have highest eutrophication potential from life cycle perspectives.

Eutrophication potential of food processing varies with processing techniques and distances. Processing dairy products and meat products also has important impacts on food's eutrophication potential. Industrial milk processing (including liquid milk, milk powder, cheese, butter etc.) generates distinct amounts of nutrient wastes. Eide et al reported that eutrophication potential of processing milk ranged from 6200 g O_2/1000 L milk to 8000 g O_2/1000 L milk (Cederberg and Mattsson 2000; Eide 2002; Cederberg and Stadig 2003). Slaughtering animals also influence nutrient inventory significantly. The major source of nitrogen and phosphorus is from the protein in that meat particles and blood in the wastewater from slaughter plants. Other sources of nitrogen are the manure and partially-digested feeds from stomachs and gizzards and intestines, as well as urine (Tusseau-Vuillemin 2001). Transportation stages and packaging stages have negligible influences on eutrophication profiles of food groups. Although

transportation distances are long, eutrophication potentials resulting from NH_3, NO_x depositions are relatively small. The usage of packing materials for supplying foods is also a minor consideration for food's eutrophication profiles.

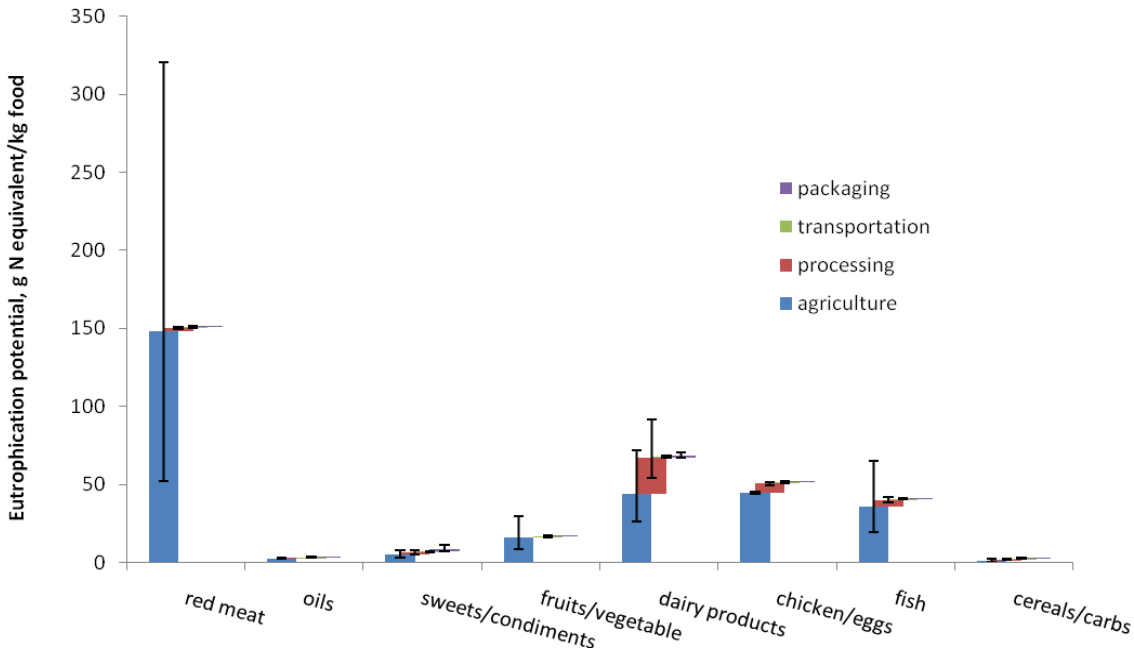

Figure 2. Eutrophication potential of researched food groups by life cycle stage

Comparative results of eutriphication potential among different food groups inform consumers the relevance between lowering eutrophication footprints and food consumption pattern. However, different foods groups have different prices, different nutrients, and of course are more or less pleasant to eat depending on consumers' taste. Figure 3 shows a comparison of total impacts with impacts normalized by expenditure, calories and protein mass. Prices, calorie and protein contents of different food groups are published on USDA websites (USDA 2009; USDA 2009). Results show when consumers spend 1 dollar on foods, the supply chains of foods generates 1 g N equivalent- 9 g N equivalent to environments. Similarly, the supply chains

produce 1 g N equivalent- 20 g N equivalent, 32 g N equivalent- 7307 g N equivalent, separately, when 1 Kcal energy or 1g protein is delivered to consumers' baskets. Cereals/carbs group has the lowest N emissions normalized by food price, calories, and protein mass. Compared with other food groups, cereals/carbs group is the most environmentally friendly choice for reducing nutrient emissions, when the same amount expenditure, or the same energy content, or the same protein content is considered.

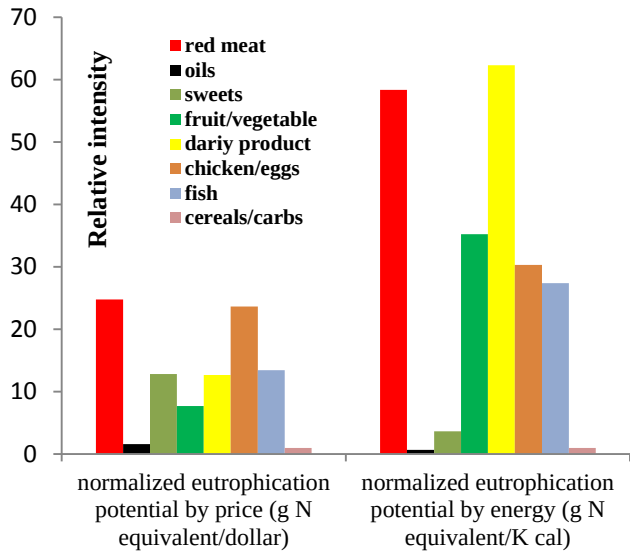

Figure 3. Comparison of normalization factors for eutrophication potential of food groups

4.4 DISCUSSION

4.4.1 Uncertainty in results

Uncertainty and variability of nutrient inventory among food groups was assessed using MCA. Results show the agricultural systems exhibit considerable variability and uncertainty in emission profiles because of differences in geography, climate, and agricultural practices. The uncertainty of meat production stage stems from different feed choices, animal raising practices, farms' locations and others. The choices of feed intake influence the amount of nitrogen excreted by animals and nitrogen emitted from feed production processes (Strid Eriksson, Elmquist et al. 2005). Besides feed choices, production modes also have an impact on eutrophication potential (Basset-Mens and van der Werf 2005). The uncertainty of eutrophication potential resulted from pork production was investigated through differentiating the production modes (conventional, quality label, organic) and farming practices (good agricultural practices versus over fertilized). The scenario analysis results exhibit uncertainty of good agricultural pracitices scenarios is around ±50% and mainly due to field emissions (around ± 35%) (Basset-Mens and van der Werf 2005). Additionally, the impact of farms' locations on eutrophication potential has been quantified by previous researchers. Kumm (2002) found that the nitrate leaching potential is related to the spatial location of farms in Sweden(Kumm 2003). Farms in central Sweden (lower precipitation, clay soils) had only one-third of the leaching level of farms in south-western Sweden (higher precipitation, sandy soils). Temperature, humidity and soil compositions can greatly influence denitrifaction/nitrification rates, and P adsorption capacity of soil particles, consequently influence the amounts of nutrients transported into water bodies. The use of average data to characterize agricultural system may not represent emissions occurring during

“extreme” years (such as rainy or drought years), and the subsequent environmental impacts (Miller, Landis et al. 2006). Since the agricultural sector is a dominant contributor for eutrophication potentials of foods, identifying and characterizing the uncertainty of nutrient flows in agricultural systems is important for future research. Transportation distances and choice of packing materials in assumptions only reflect the average level of food supply chains, however the impacts to eutrophication from transportation are negligible. When aggregate food groups are researched rather than specific food types (such as the difference between grass-fed versus grain-fed meat, organic farming vs. conventional farming, etc), uncertainty and variability are enhanced. Constrains of data availability limited the number of researched food types. However, the average values and variability evaluation of the nutrient inventory presented within this research among food groups are still meaningful to investigate eutrophication footprints of diet habits.

4.4.2 Comparing carbon footprints and nitrogen footprints

Figure 4 compares nutrient footprints and carbon footprints of different food groups. Food groups close to the origin have both low carbon footprint and low nitrogen footprint. For example, cereals/carbs and beverages are the most environmentally preferred food types from a carbon and nitrogen life cycle perspective. Oppositely, red meat, having the highest carbon footprint and the highest nitrogen footprint, is the least environmentally preferred food type. Dairy products, fish and chicken/eggs have relatively higher nitrogen footprint and lower carbon footprints. Conversely, sweets, oils, fruits and vegetables have relatively lower nitrogen footprints and higher carbon footprints. The inconsistency between carbon footprints and nitrogen footprints indicates tradeoffs of shifting food consumption habits and inherent

environmental complexities of food policy decisions. For example, solely minimizing consumers' C-footprint would suggest that one consumes cereals/carbs, dairy, chicken/eggs, and fish (Weber and Matthews 2008). However, if the N-footprint is also considered, as shown in Figure 4, dairy products are not necessarily ideal since they have the second highest eutrophication potential, while fruits and vegetables might be reconsidered since they have a minimal N-footprint.

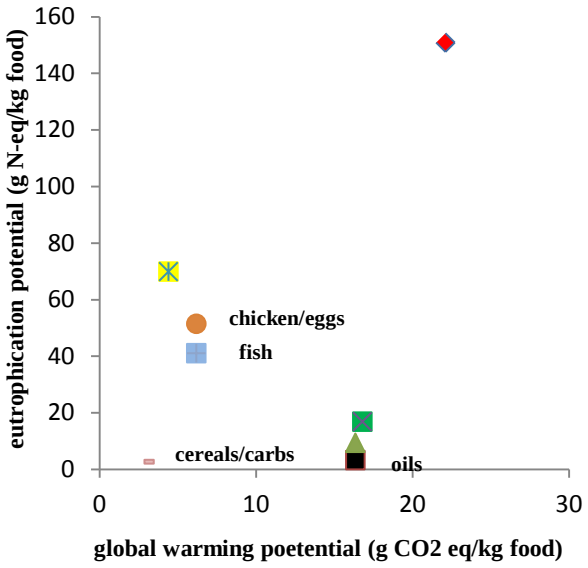

Figure 4. Comparison of carbon footprints and nitrogen footprints

4.4.3 Nitrogen output reduction due to consumption pattern shifts

The effect of food consumption pattern shifts on nitrogen equivalent emissions reduction is estimated assuming: 1) eutrophication profiles of foods are unchanged during the shift, 2) linear relationships exist between nutrient output and mass of consumed food, and 3) consumers maintain constant calorie consumption during food pattern shifts. In reality, the shift of dietary

habits is a relatively slow process (USDA 2002; Gehlhar and Coyle 2005). Technology improvement and policy incentives may reduce environmental footprints of food production and processing in relatively short period of time. Estimated results (Figure 5) show that food supply chains generate 40 kg nitrogen equivalent for meeting one person's food needs annually. Shifting red meat and dairy products to other low nitrogen intensive food groups may significantly reduce personal eutrophication potential. Among possible consumers' behaviors changes, shifting dairy products to cereals products is the most effective way to mitigate personal eutrophication potential from both a cost and nutrient emissions perspective (Figure 5 and Figure 6). Shifting 5% dairy products to cereals groups and maintaining the same calories can prevent 380 g Nitrogen equivalent emissions to environment. On the extreme, 7630 g nitrogen equivalent emissions can theoretically be avoided if 100% dairy products are replaced by cereals/carbs products.

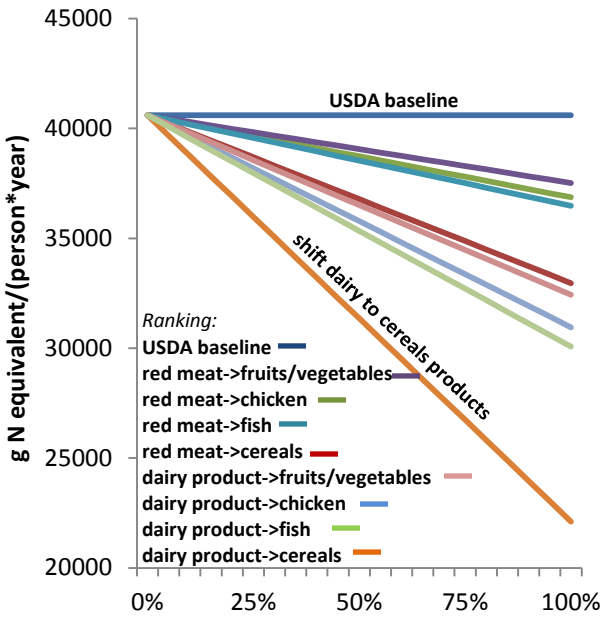

Figure 5. Eutrophication potential reductions due to food consumption shifts

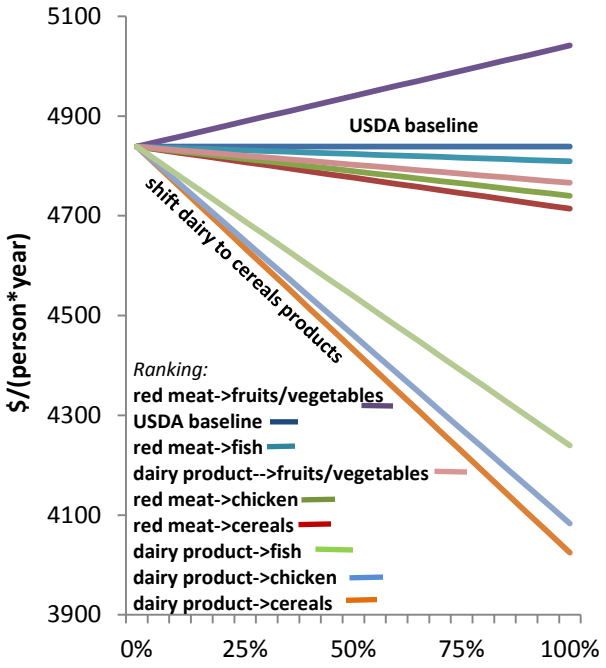

Figure 6. Cost reductions due to food consumption shifts

The fluctuation of food cost as result of food consumption pattern shifts is estimated based on the same set of assumptions as discussed previously in addition to the assumption that 4) price of food groups maintains the same as the 2007 baseline and 5) linear relationship exists between food cost and mass of bought food. Estimated results (Figure 6) show that food supply chains cost $4838 for meeting one person's food needs annually. This study does not account for cost of dining out. It is the most economically effective to change dairy product into cereals groups, while shifting red meat to vegetables may increase cost. Shifting 5% dairy products to cereals groups and maintaining the same calories can save $40 annually. In the extreme case, replacing 100% of dairy product with cereals/carbs products could save $810.

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5.0 ENVRIONMENTAL IMPACTS OF BIODIESESLS

5.1 INTRODUCTION

The University of Pittsburgh collaborated with the Pennsylvania Department of Transportation (PennDOT) to conduct a comprehensive feasibility study regarding the transition of PennDOT fleet vehicles from ultra low sulfur diesel (ULSD) to biodiesel. The feasibility study comprised three main phases: investigation of best practices, data collection and analysis for implementation, and economic and environmental analysis. In order to properly evaluate the environmental and economic performance of biodiesels, the research team conducted both life cycle costs and life cycle assessments. This chapter focuses on the portion of this project conducted by the thesis author, Xiaobo Xue; the life cycle assessments of biodiesels.

As described in Chapter 3, previous life cycle assessment studies of biofuels have not fully investigated agricultural or regional impacts for a wide range of environmental impact categories. This study adds to the available literature by providing a comparative LCIA of biodiesel across TRACI impact categories, using regional inventory data and LCIA characterization factors (CFs) when available. In cooperation with the Pennsylvania Department of Transportation (PennDOT), fuel usage and fleet data were obtained to support the LCA (Shrake et al, 2010). Life cycle environmental impacts of different biodiesel blend levels (B5, B20 and B100) were compared using various regional production scenarios. The regional and

comprehensive focus of this study is helpful to quantify the environmental impacts of biodiesel blends and aid state-level policy decision-making.

5.2 METHODS

A process life cycle assessment method was used to quantify the environmental impacts of different biodiesel blends. LCA consists of four steps: 1) scoping and defining system boundaries, 2) inventory analysis, 3) impact assessment, and 4) interpretation and improvement. Each step of the LCA conducted for this study is described in detail in subsequent sections. Different production scenarios and the related regional variations were evaluated at both the inventory and impact assessment steps.

5.2.1 System boundary

The scope of the analysis considers the major stages in the life cycle of fuels, including feedstock production, fuel processing and transportation, and combustion of the fuel for equipment operations. The researched fuels include B5, B20, B100 and conventional diesel.

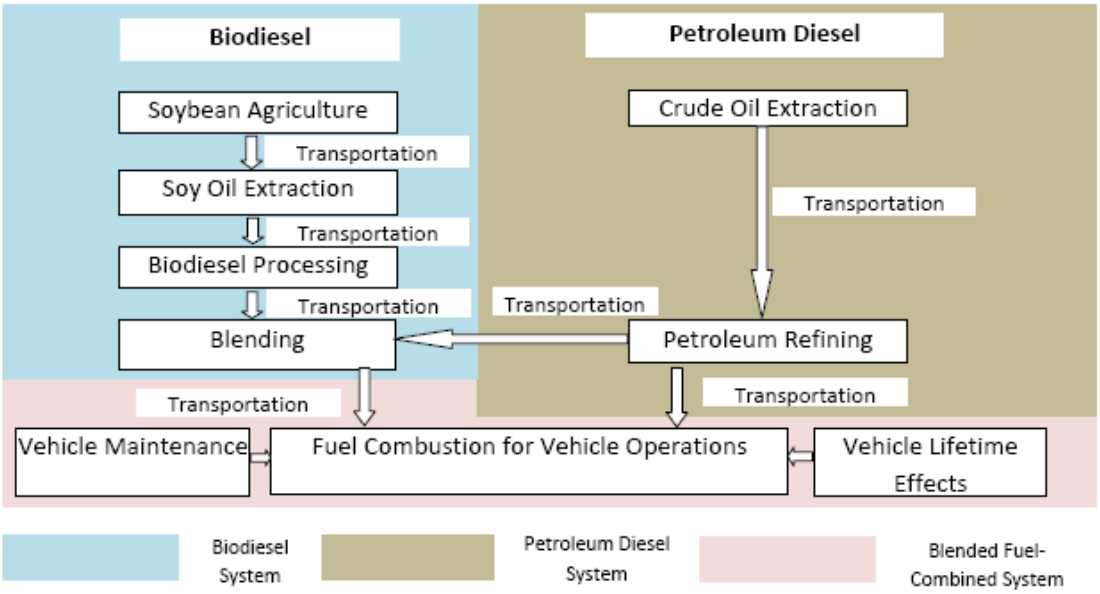

Figure 7. System boundary of life cycle biodiesel study

The system boundary shown in Figure 7 includes soybean farming activities such as planting seeds, tillage, fertilizing, applying pesticides and herbicides, harvesting, and storing soybean grains. Moreover, upstream activities, such as manufacturing farming equipment and chemicals, producing animal manure, etc. are also accounted for in the system boundary. The activities of processing and transportation are concentrated in three main contributors: transportation, soy oil processing, and transesterification. The processing phase was broken into two major parts of soy oil extraction and soy oil transesterification. Transportation includes three stages: soy from field to production facility; biodiesel from production facility to distribution facility; and biodiesel blends from distribution facility to PennDOT tanks. For the diesel use stage, the tailpipe emissions resulting from the combustion of the fuel in the vehicle's engine are included in system boundary. The life cycle of petroleum diesel includes extraction of the crude oil, whether using conventional drilling or newer enhanced recovery methods; transportation to the refinery; refining, and transportation to the point of use.

The environmental impacts of a hypothetical policy decision to stimulate Pennsylvania's economy by encouraging in-state production of biodiesel for use in PennDOT's fleet were examined through the development of several scenarios, identified in Table 2. Quantities of biodiesel required were estimated by considering the use of different blend levels by PennDOT. For B5, only in-state production consisting of both the agriculture and processing stages was considered, given the small demand relative to overall soybean and biodiesel production capacity in PA.

Table 2. Production scenarios for biodiesel blends

Blends	**Biodiesel Production Scenarios**	
B5	Full In-State Production	Soybeans grown, oil extracted and refined into biodiesel inside PA
B20	Full In-State Production	Soybeans grown, oil extracted and refined into biodiesel inside PA
	In-State Processing Only	Soybeans grown and oil extracted outside PA, with transport to a PA-based biodiesel refinery for processing
	National Average	Soybeans grown, oil extracted and refined into biodiesel outside PA
B100	Full In-State Production	Soybeans grown, oil extracted and refined into biodiesel inside PA
	In-State Processing Only	Soybeans grown and oil extracted outside PA, with transport to a PA-based biodiesel refinery for processing
	National Average	Soybeans grown, oil extracted and refined into biodiesel outside PA

5.2.2 Life cycle inventory

Various data sources were used to compile life cycle inventory in this study. Greenhouse gases, Regulated Emissions, and Energy use in Transportation (GREET) developed by Argonne National Laboratories was utilized to compile a LCI of energy use and criteria air pollutants and greenhouse gases (Argonne National Laboratory 2006). To augment GREET data with other emissions; other LCI databases were used to compile full inventory including air emissions, water emissions, toxic substances and waste generations etc. For this study, the Franklin 98, USLCI and the ecoinvent v2.0 databases were utilized (NREL 2004; consultants 2007), (center 2009).

Several customized models were also developed for this study. For on-field aqueous nutrient emissions, a fractional approach developed by Landis et al. was employed (Miller, Landis et al. 2006; Landis, Miller et al. 2007). Nutrient emissions are considered to respond to fertilizer application rates in the model. Combustion emissions during vehicle operations were calculated using emissions factors for different fuel types and different vehicle types extracted from EPA's NONROAD and MOVES models (Landis, Bilec et al. 2010). Emissions factors are included in the Appendix B.

Life cycle emissions during soybean farming were constructed using data identified in Table 3 and the Appendix B. Appropriate data were collected to model current farming practices in PA and Corn Belt states. A variety of data collected from government reports and peer reviewed literature was used to determine key parameters of life cycle inventory for soybean farming stage.

Table 3. Critical parameters, processes and data sources for LCI

#	**Parameter or process**	**Data source**	**References**
1	Area of soybean farmlands	USDA	(USDA 2008; USDA 2008)
2	Soybean yield rate	USDA	(USDA 2008; USDA 2008)
3	Fertilizer application rate	USDA	(USDA 2008; USDA 2009)
4	Herbicide application rate	Publications	(Colorado State University 2008)
5	Water usage for soybean farming	USDA and publication	(Dominguez-Faus, Powers et al. 2009; USDA 2009)
6	Tillage	Publications	(National Sustainable Agriculture Service 2008)
7	Farming equipment use	Publications	(Colorado State University 2008)
8	Nutrient emissions	Publications	(Miller, Landis et al. 2006; Landis, Miller et al. 2007; Miller, Landis et al. 2007)
9	Transportation to oil plant	Data provided by PennDOT and GREET	(Argonne National Lab 2006; Huo, Wang et al. 2008)
10	Emissions from soy oil production	GREET, ecoinvent v2 database	(Argonne National Lab 2006; Huo, Wang et al. 2008)
11	Transesterification	GREET, ecoinvent v2	(Ecoinvent Center 2009)
12	Biodiesel transportation	Data provided by PennDOT, Franklin, US LCI 98 databases	(Pre consultants 2007)

Table 3. (Continued)

13	Biodiesel combustion	Data provided by PennDOT, US EPA MOVES, NONROAD	(Landis, Bilec et al. 2010)

The emissions during transportation stages and soy oil production were compiled from GREET, and the Franklin, US LCI and ecoinvent v2 databases. The transportation distances for the three scenarios are reported in Table 4.

Table 4. Transportation distances for selected biodiesel production scenarios

Scenarios	**Transportation stages**		
	Soybeans from field to oil mill	Soy oil from oil mill to biodiesel refinery	Biodiesel from refinery to PennDOT tanks
Full in-state production	150 miles	75 miles	75 miles
In-state processing state	150 miles	600 miles	75 miles
National average	150 miles	75 miles	75 miles

The LCI for the vehicle operation and emissions was generated using the fuel consumption and vehicle type information obtained from PennDOT. The details of developing and verifying emission factors were discussed in previous research (Landis, Bilec et al. 2010). The emission factors employed in this study are shown in Table 5.

Table 5. Emission factors for fleetwide consumption

Emission type	**NO_x**	**PM**	**CO**	**HC**	**SO_2**	**CO_2**	**CH_4**	**N_2O**
Units	g/gal	g/gal	g/gal	g/gal	g/gal	g/gal	g/gal	g/gal
ULSD Emission Factor	54.91	2.39	21.12	4.63	0.095	10.29	0.21	0.30
B5 Emission Factor	55.19	2.30	20.53	4.46	0.09	10.29	0.21	0.30
B20 Emission Factor	56.12	2.02	18.15	3.99	0.08	10.29	0.21	0.30
B100 Emission Factor	60.41	1.24	10.98	1.53	0.09	10.29	0.21	0.30

The GREET model and USLCI databases were used to calculate the LCI for diesel fuel used in blending with biodiesel and in the baseline ULSD. Data from the GREET model includes energy consumption and combustion-related emissions from the "well to pump" portion of the life cycle for diesel fuel. GREET does not include the emissions from fuel combustion in heavy-duty trucks and off-road vehicles. Combustion emissions from off-road and heavy-duty vehicles representative of the PennDOT fleet were modeled using the emission factors described in Table 4. Emissions of pollutants other than the greenhouse gases and criteria air pollutants covered by GREET were modeled using the USLCI database. These emissions include hazardous air pollutants and water emissions which are released throughout the process of fuel extraction, refining and transportation.

5.2.3 Allocation and functional unit

For soybean agriculture, the market-value based allocation ratio used in GREET was used to assign impacts to soy oil or its co-product, soy meal (Huo, Wang et al. 2008). This allocation resulted in 42% of the impacts from the soybean system allocated to soy oil. The same allocation was performed for soybean transportation and soy oil extraction. One ULSD-equivalent gallon of fuel was defined as the functional unit in this study. To obtain ULSD equivalent volumes, the volume of any biodiesel blend is multiplied by the ratio of its energy content to the average energy content of the same volume of ULSD. Average energy content for the fuels was obtained from GREET. Although energy related functional units were used in previous biodiesel LCA studies, an analysis based on volume related functional units can be easily applied to fleets where fuel consumption is tracked on a volumetric basis.

5.2.4 Life cycle impact assessment

The TRACI (Tool for the Reduction and Assessment of Chemical and other environmental Impacts) model was used to perform life cycle impact analysis of biodiesels. TRACI presents LCIA factors for the following environmental impact categories: three categories of human health effects (criteria air pollutants, cancer, and other non-cancer), photochemical oxidation, ozone layer depletion, aquatic ecotoxicity, acidification, eutrophication and global warming.

The original TRACI method developed by the USEPA (Norris 2001; Bare, Norris et al. 2002) contains regional CFs for selected LCIA categories for which the effects are regional in nature and for which the location-dependent variation is hypothesized to be significant compared to other uncertainties. The categories of global warming potential and ozone depletion are global

in nature and thus do not have regional CFs. Human health cancer and noncancer effects and ecotoxicity categories do not have regional CFs because the uncertainty of the toxicity values is likely to be greater than the regional variation. The remaining four categories – human health due to criteria air pollutants, acidification, eutrophication, and photochemical oxidation – include regional CFs which were derived using various environmental fate, transport and exposure models described in the original TRACI literature.

To apply the regionalized CFs it is necessary to determine which processes are occurring in the region under study and which processes are associated with the supply chain and are not subject to regional constraints. Thus, top-level processes and upstream processes which are geographically tied to the production locations are assigned the regional CFs, whereas geographically dispersed upstream processes are assigned the national average CFs. A further step was taken to establish national average CFs specific to soybean production by multiplying the individual state CFs by their fraction of total US soybean production.

5.3 RESULTS

5.3.1 Life cycle stage contributions

The stage contribution for each environmental impact category varied for each fuel (see Figure 8). From a life cycle perspective, tailpipe emissions (e.g., CO_2, NO_X, SO_X) dominated in global warming potential, acidification, respiratory effects and photochemical oxidation impacts for ULSD, B5 and B20. USLD production was the biggest contributor to carcinogens, non carcinogens, and ecotoxicity categories for USLD, B5 and B20. The agricultural stage had a

significant contribution to biodiesel's impacts in most categories, such as acidification, carcinogens, non-carcinogens, eutrophication, respiratory impacts and photochemical oxidationphotochemical oxidation. For B100, the agriculture stage contributes more than 70% of the total impacts in the carcinogens, noncarcinogens, eutrophication and ecotoxicity categories. Large amounts of air and water emissions were generated during farming practices and their upstream activities. Oil processing and transportation stages had minimal contributions to all categories.

Environmental tradeoffs existed between global warming potential and eutrophication potential of biodiesel. The LCA results for Biodiesels and ULSD are shown in Figure 8 displayed in terms of the relative impacts in each category, where the impacts were normalized to the fuel with the highest impact. Global warming potential of B5 was 3% lower than ULSD. The use of biodiesel blends resulted in a 10% decrease in global warming potential for B20 and a 50% reduction for B100, comparing with ULSD. Eutrophication potential of B5 was more than twice than that of petroleum. Eutrophication potential of biodiesels spiked with the increase of biodiesel blending ratios. Soybean farming exhibited negative global warming potential, because soybean plants have the capability to sequester CO_2 from air and offset air emissions from other agricultural stages. Meanwhile, nutrient runoff (mainly, NO_3^- and P emissions) resulting from fertilizer application contributed to significant eutrophication impacts.

The trends of other environmental impacts varied when the biodiesel blending levels increased. The acidification potential did not change much for different blending levels. Increased biodiesel contents increased acidification impact from agricultural stage and reduced the acidification from USLD production; therefore the total acidification impacts maintained almost the same values for USLD, B5, B20 and B100. For the respiratory effects categories,

results were approximately the same for ULSD, B5 and B20 but at least 25% higher for B100. This was because the reduction in tailpipe emissions associated with the biodiesel blends balances the increased emissions associated with agriculture. A complex interaction of tailpipe emissions differences and agricultural emissions increases resulted in very little change in impacts for lower biodiesel blends, with a marked increase for B100. Human health carcinogens were dominated by both the agricultural stage and ULSD production stage. This impact increased roughly proportionately to the biodiesel content. Photochemical oxidation impact also increased roughly proportionately to biodiesel content, but it was driven primarily by the increased NOx emissions for biodiesel and to a lesser degree by agricultural NOx emissions. In the human health non-carcinogens and ecotoxicity categories (where similar substances are involved), the decreased impacts for increasing biodiesel blending levels were explained by decreased requirements of ULSD production.

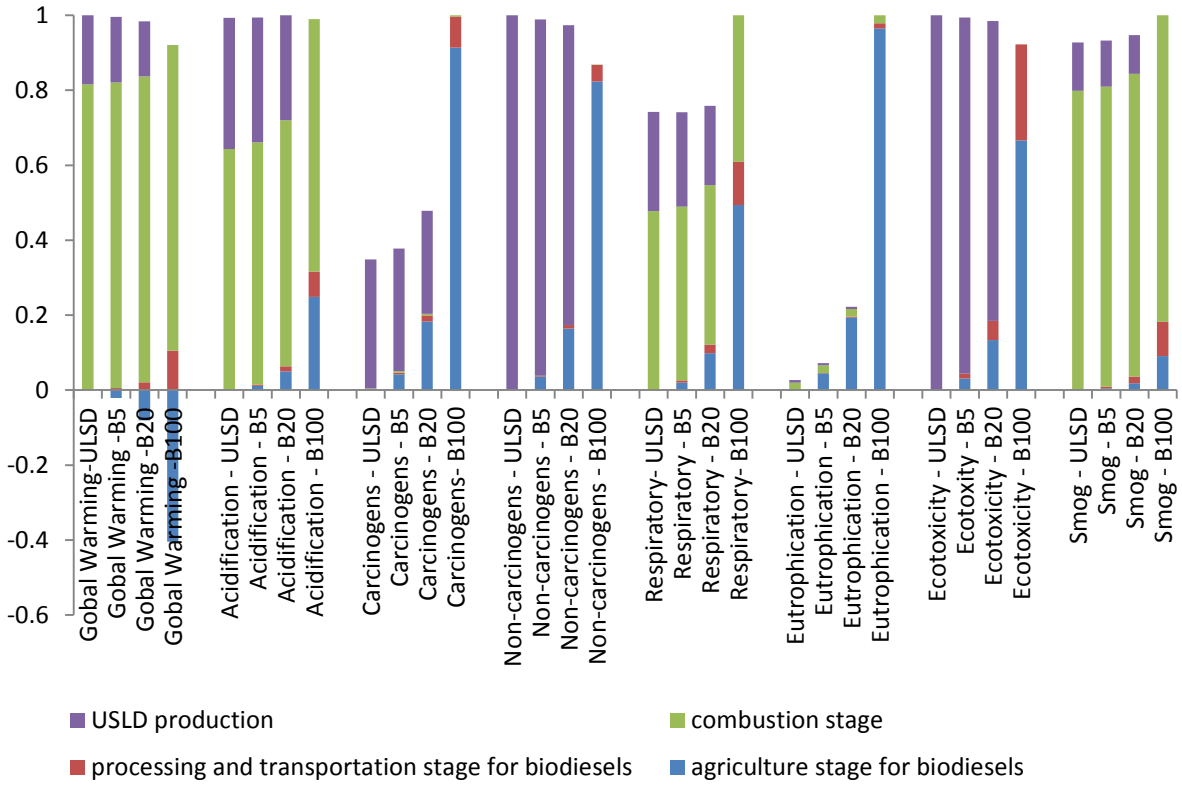

Figure 8. The life cycle environmental impacts for B5, B20, B100 and Ultra low sulfur diesel

5.3.2 Regional variation in life cycle impacts

Figure 9 shows the comparison across TRACI categories for the B100 used in the various blends for the full in-state production, in-state processing only and national average scenarios. Differences in the environmental impacts between the scenarios are a result of two factors: changes in the inventory due to increased or decreased input requirements, and changes in the LCIA CFs due to changing the location of production. For the acidification, eutrophication and photochemical oxidation categories, regional CFs are available in TRACI. In the case of Pennsylvania, CFs are lower than the national average CFs in all three categories. CFs for

nationwide soybean production in the national average and in-state processing only scenarios are higher than the national average CFs, due to the concentration of soybean production in Corn Belt states with higher CFs in these categories.

Differences in the remaining categories result from differences in the inventory alone, since only national CFs are available in TRACI. Pennsylvania soybean farming resulted in reduced environmental impacts compared to Corn Belt agriculture. The N fertilizer input for Corn Belt is 18.7% higher than the input for Pennsylvania; meanwhile the P fertilizer input for Corn Belt is 11.2% higher than the value for PA. The higher chemical input (such as fertilizers, herbicides etc.) in Corn Belt states resulted in a higher global warming potential for soybean farming due to greater energy requirements in the manufacturing process. The higher fertilizer application rate also causes a relatively higher eutrophication potential generated from nutrient runoff and leaching in Corn Belt states. In addition, the herbicide application rate in PA is 3% lower than the rate in Corn Belt states, so environmental impacts associated with herbicides (such as human and ecological toxicity) are slightly lower than Corn Belt states.

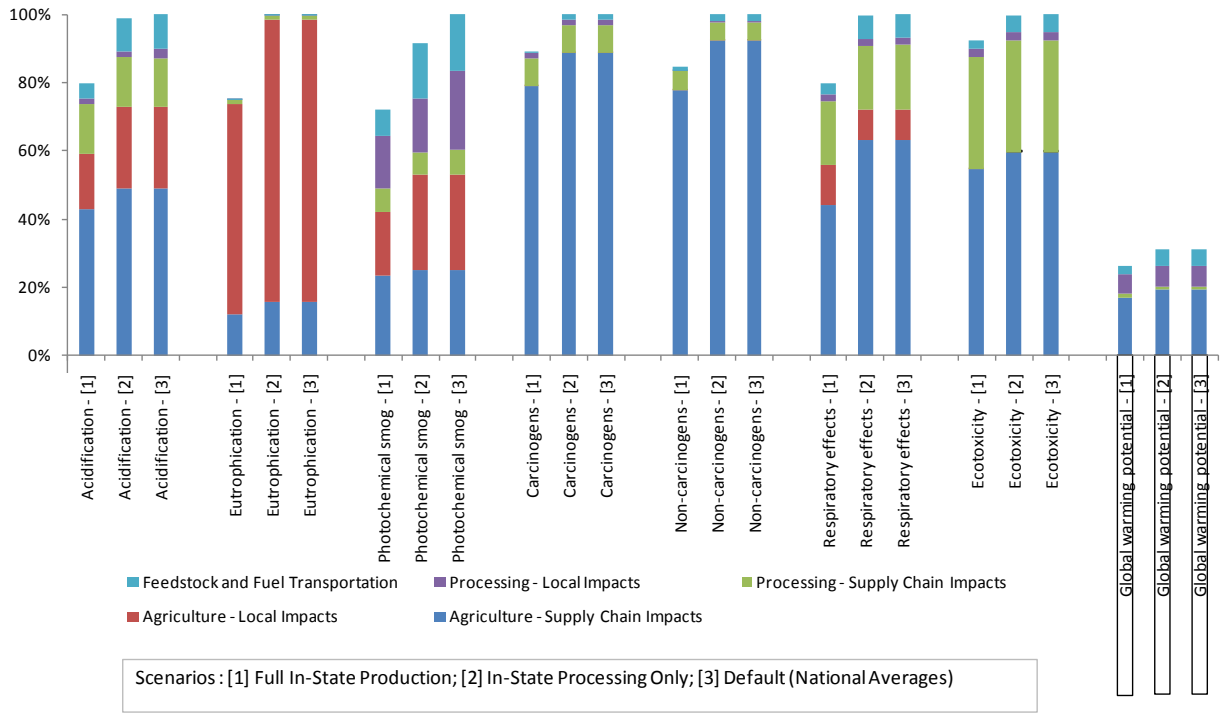

Figure 9. Life cycle impacts for different regional production sceanrios

5.4 DISCUSSION

5.4.1 Regionalized stage contributions to LCIA categories for B100

Photochemical oxidation potential is affected significantly by each modeled stage, including both regional and supply chain effects as well as feedstock and fuel transportation. Thus, results in this category are different for the three scenarios. The full in-state production scenario had the lowest normalized total result (72%), followed by in-state production only (91%) and the national average scenario (100%). For the agriculture stage, local impacts are dominated by NOx emissions from on-field volatilization and fuel combustion in farming equipment, which do

not change in the model due to a lack of location-based data in the literature. However, the CF for NOx in PA is 0.67, compared with 0.98 for national average soybean production, leading to a lower result in this category for PA agriculture. The upstream impacts of agriculture are also less for PA agriculture due to reduced fertilizer usage and thus lower associated combustion emissions from fertilizer manufacturing. The local emissions for processing are unchanged between the scenarios, but the lower CFs for PA lead to a lower result in this category, while upstream impacts remain the same. Over 90% of the localized impacts in the processing stage for this category are attributed to hexane released during the soy oil extraction process. Since truck transportation is a significant contributor to NOx emissions, the in-state production result for this stage is lower due to the reduced transportation distances.

Results for the acidification category follow a similar pattern to photochemical oxidation (80% for full in-state production, 99% for in-state processing only, and 100% for the national average scenario), due to the existence of state-level CFs and the lower fertilizer intensity of PA agriculture, and because acidifying emissions are generally related to combustion of fossil fuels for energy. However, the agricultural supply chain generates up to 49% of the impacts in this category, much higher than that for photochemical oxidation. Fertilizer production generates up to 50% of the total agricultural impacts, and almost twice as much as the regional total. In turn, 75% of the acidification potential of fertilizer production is due to SO_2 emissions from phosphate fertilizer production.

Normalized eutrophication results are 71% for full in-state production and 100% for the other two scenarios, due mainly to the lower usage of fertilizer in PA. While state-level CFs for eutrophication exist for both air and water emissions, their magnitudes for PA are within 10% of the national averages. Water emissions of N (as NO_3^-) and P from farm fields contribute up to

83% of the results in this category; however, the agricultural supply chain contribution is also significant. Similar to the acidification category, fertilizer production generates the majority of the supply chain nutrient emissions – up to 16% of the total LCIA result for eutrophication – the vast majority of which (>95%) are water emissions of phosphate from phosphate fertilizer production. Processing and transportation stages together contribute less than 2% of the LCIA result.

Normalized results are 89%, 85% and 92% for full in-state production for human health - cancer, human health - non-cancer and ecotoxicity categories respectively, and 100% in all three categories for the other two scenarios. The reduced fertilizer usage for PA soybean farming is responsible for this difference; state-level CFs do not exist for these categories. The agricultural supply chain contributes up to 89%, 92%, and 60%, respectively of the total LCIA results for these categories, with the processing and transportation supply chain contributing 33% of the ecotoxicity result. As in the acidification category, a large percentage of the total LCIA results (42%, 60%, and 21%, respectively) stem from fertilizer manufacturing, particularly phosphate fertilizer.

Uniquely in this analysis, the reduced impacts from the PA production inventory are countered by higher CFs for PA in this category, for some pollutants. CFs for particulate matter are higher for PA than for the national average by up to 75%, whereas CFs representing national soybean production are up to 6% higher than the national average. CFs for NOx and SO_2, important PM precursors, are slightly lower than the national average for both regions. The difference in CFs results in regionalized impacts for PA agriculture being higher than average soybean production (12% and 9% of the normalized results, respectively). However, the total results are still lowest for full in-state production (80%) compared to the other two scenarios

(100%), reflecting the lower agricultural supply chain requirements and shorter transportation distances.

Global warming potentials for all three scenarios are negative, reflecting the dominance of the carbon sequestration in the soybean crops compared to the greenhouse gas emissions of the production processes. The regionalized agriculture stage consists mainly of carbon sequestration by the soybeans, N_2O emissions from soil nitrogen processes, and combustion of petroleum fuels in on-farm operations. Regionalized agricultural processes are estimated to sequester greenhouse gases at a rate 3.5 times their emissions. The GHG benefit is countered somewhat by energy used in the remaining stages, with the largest contribution (up to 37% of the remaining total) from fertilizer manufacturing. Total global warming potential is lowest for the full in-state production scenario, resulting from reduced fertilizer usage and lower transportation distances.

For several categories – human health impacts due to criteria air pollutants and photochemical oxidation – combining the regionalized inventory with regional CFs more than halves the percentage increase due to switching from ULSD to B100 (from 32% to 13% and from 8% to 2%, respectively). For acidification, regionalizing production results in a 7% decrease from ULSD to B100, where as the national average scenario shows less than a 1% reduction. In the other categories, effects of regionalization are small in comparison to increases or decreases due to the shift in fuels. For all categories, the effects for B20 are slightly greater in magnitude than a simple application of the volume percentage times the change for B100, because tailpipe emissions of most pollutants decline somewhat more rapidly at lower biodiesel blend percentages (USEPA 2002).

The regional analysis described herein is limited to parameters for which data were readily available. At the inventory stage, differences in agricultural chemical usage and transportation distances varied between scenarios. Insufficient data was available to model differences in on-field air and water emissions rates; the linear model used to predict emissions of N and P compounds depends on fertilizer usage rates only. Both air and water nutrient emissions may also vary based on soil characteristics, precipitation, temperature and other environmental factors. These parameters may vary from field to field, but may also display distinct regional trends. Future research is needed to quantify these effects to ascertain if additional regional differences should be incorporated in the life cycle assessment of biofuel crops. In contrast to farming, soy oil milling and biodiesel production activities are concentrated at individual plants which may be considered point sources of emissions. These emissions may vary regionally if one or more states features newer plants due to economic or policy shifts. Frey et al. (Pang, Frey et al. 2009) have shown that soy oil plants conforming to EPA new source performance standards (NSPS) lower life cycle energy consumption and emissions of CO and hydrocarbons.

With respect to LCIA CFs, only one peer-reviewed method (TRACI) was available for the US. For an extended discussion of the uncertainties associated with TRACI as identified by the authors, the reader is referred to the articles by Bare and Norris (Norris 2001; Bare, Norris et al. 2002). As environmental fate and transport models become more comprehensive and additional research is conducted into the risks of pollutant exposure, regional CFs may be revisited. For example, Shah and Ries (Shah and Ries 2009) have proposed an alternate method for photochemical oxidation characterization, in which the spatial variability of CFs for NOx is different than that for VOCs, whereas they are considered the same in TRACI. Recent research

by the US Geological Survey (Alexander, Smith et al. 2007) has shown that eutrophication potential from agricultural activities in the Mississippi River/Gulf of Mexico basin varies significantly between tributary watersheds.

Finally, this analysis does not consider the environmental impacts driven by land use change and establishing new infrastructure for biodiesel. However, increasing demand for biodiesel may significantly change land use patterns and stimulate infrastructure construction. If PennDOT implemented statewide B20 usage with a preferential policy for full in-state production, approximately 10% of the current PA soybean crop would be required to meet the demand. It is not known whether this amount would result in significant environmental effects due to shifts in land usage.

6.0 ENVIONMENTAL IMPACTS OF FARMING PRACTICES

6.1 INTRODUCTION

Optimizing farming practices may play an important role in reducing overall environmental impacts of bioproducts caused by agriculture. Alternative agricultural management strategies have been studied widely in the agricultural literature, primarily focusing on narrow aspects of agriculture management, such as their effects on yields, nutrient leaching etc. However, the overall environmental impacts from agriculture and related management strategies, and thus their implications on the environmental footprint of bioproducts have not been explored adequately. With the growth of biofuels and biobased products, Life Cycle Assessments (LCAs) can be a useful method to quantify environmental impacts of agricultural practices. Comparative LCAs among possible products or processes can help to determine the environmentally preferable alternative. Currently, agricultural LCAs have been mainly carried out for single crops or production processes (Landis, Miller et al. 2007; Pelletier, Arsenault et al. 2008). The comparative analysis of farming practices, which is important to identify environmentally preferred practices and reduce negative environmental impacts resulted from corn farming, is still lacking from life cycle perspectives.

This chapter aims to quantify energy consumption, air emissions, and aqueous nutrient emissions under different corn farming management scenarios. The inventory includes global

warming emissions, aqueous nutrients (N, P) and energy usage. Comparative LCA results of the three farming practices including tillage practices, choice of fertilizer types, and installing buffer strips are presented. Moreover, this chapter provided detailed and comparative description of environmental impacts to identify farming practices that improve environmental performances of crop agriculture.

6.2 METHODS

6.2.1 System boundary

The agricultural system boundary, material and energy flows accounted for in the LCA are depicted in Figure 10. The system boundaries include on-field production practices (tillage practices and fertilizer application), integrated farming practices (buffer strips), associated equipment and chemical manufacturing, transportation processes, and also power generation. Energy usage, atmospheric and aqueous emissions are calculated during every stage.

Geographically, this system reflects the farming scenarios in US Corn Belt states, which produced more than 75% of total U.S. corn in 2008. US Corn Belt states include Iowa, Illinois, Nebraska, Minnesota, Indiana, Ohio, South Dakota, Wisconsin, and Missouri. Data was collected from literature that included experimental data, on-field survey data, and geological modeling estimation results between 1990 and 2007.

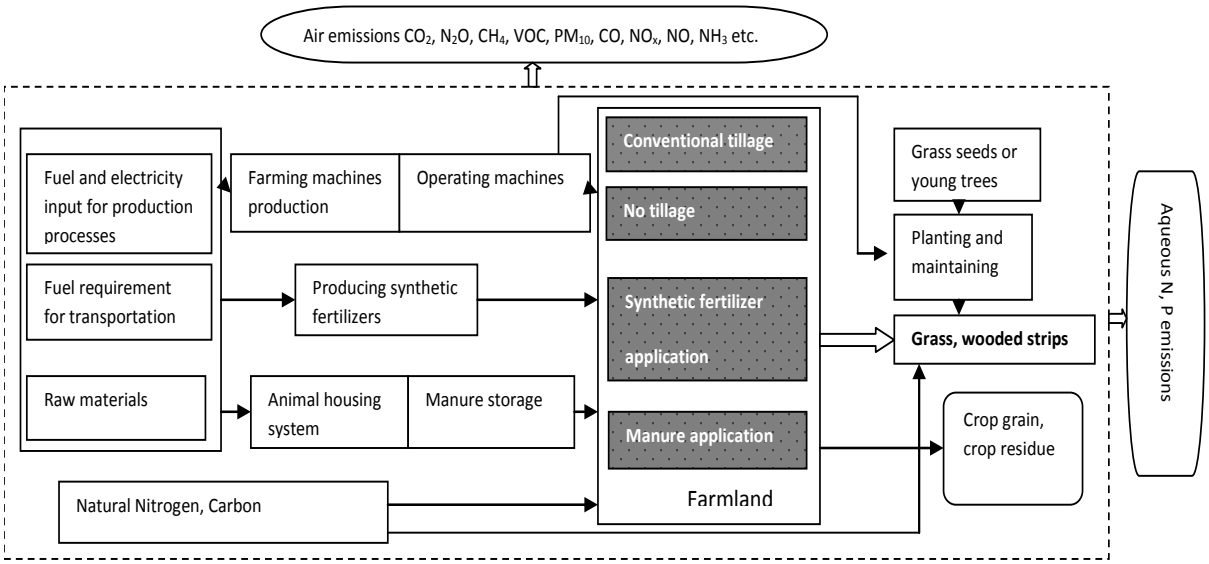

Figure 10. System boundary for the agriculture life cycle inventory

6.2.2 Allocation and functional units

Functional units aim to provide a reference level for comparison. Per kg corn was used as the functional unit to compare the effects of farming practices on corn production in this study. Allocation within LCA allows for products from corn farming to be attributed an appropriate percentage of environmental impact from the corn farming process. Allocation was conducted on a mass basis to corn crops normalized per year. All energy use and associated emissions are allocated 100% to the corn grain on a mass basis, because the boundaries of LCI end before the milling phase. Environmental impacts from manure practices were allocated in two manners for comparison: Firstly, manure was treated as a waste, thus emissions associated with animal feeding operations were not allocated to manure as fertilizer. And secondly, manure was treated as a co-product of animal husbandry systems. Energy usage of producing manures was determined according to total energy consumption in animal raising systems and the energy

based ratio of animal excretion/animal feed. 30% of energy consumed by animal husbandry system and associated emissions were allocated to manures. This ratio is estimated based on energy flows on cattle and dairy farms in Midwest states (Jewell 1975).

This research conducted nutrient flow analysis of corn production. Actually, corn is usually rotated with soybeans or other crops. When fertilizers as major N and P sources are applied in farmlands, these nutrients will be shared between corn and other rotation crops. The total nitrogen/total phosphorus leaching share of corn fields was determined by area weighed average values (Powers 2007). Thus, 51% of TN and 61% of TP was allocated to corn.

6.2.3 Life cycle inventory and impact analysis

A description of the models used to develop the LCI is discussed in this section. Energy flows and associated EPA criteria air emissions are calculated with GREET1.8 (Greenhouse gases, Regulated Emissions, and Energy use in Transportation) model while nutrient outputs are estimated through linear fractional models. Variability of agricultural systems is accounted for using statistical analyses and Monte Carlo Analysis (MCA).

As explained before, GREET 1.8 was developed by Argonne National Laboratories and was utilized within this study to compile a LCI of on-farm and upstream energy use and air emissions. GREET lacks description on handling manures as fertilizers, on-farm application of agricultural machines, and installing and maintaining buffer strips. Therefore, these aspects of agricultural management strategies were added into the study to complement data available from GREET.

Three major modifications were made to GREET to enhance the level of detail of fertilizer types and farming equipment usage; (a) addition of energy consumption and emissions

for producing, handling, and transporting manure, (b) detailed description of farming equipment used during different tillage practices, and (c) installing and managing buffer strips. These three agricultural management strategies were modeled independently, as described below, while the energy and fuel inputs for the additional activities and associated emissions were estimated by GREET.

To model manure, energy consumption and GHG emissions related to manure production, handling, and application were added to GREET. IPCC equations and suggested input values for North American region were used to estimate GHG emissions in the animal raising and manure management systems (IPCC 1996). GHG emissions after manure application on corn farm lands were determined according to publications (Comfort, Kelling et al. 1990; Mosier 1994; Ginting, Kessavalou et al. 2003; Hernandez-Ramirez, Brouder et al. 2009). Detailed datasources are reported in supporting information. The distance of transporting manure from storage room to farmland was assumed to 50 miles by light trucks based on publications (Araji, Abdo et al. 2001; MacDonald, Ribaudo et al. 2009).

To model the effects of different types of tillage, detailed inventories of energy consumption and air emissions for operating farming equipment were incorporated. Energy consumption for operating farming equipment (including tractors, field cultivators, plowers, combines, irrigators, and sprayers etc.) was quantified based on publications (Uri 1998). Air emissions generated by operating farming equipment were estimated by GREET model and energy consumption.

Most current results show crops rotation, soil properties, and weather conditions can complicate the effects of tillage practices on corn yield rate. The differences in corn yields can be neglected after long-term application of tillage practices (Triplett and Dick 2008). In Corn Belt

states, corn is usually rotated with soybeans. This study mainly focused on corn rotated with soybean and assumed that the corn yield rate under conventional tillage was as the same as no tillage.

Buffer strips are usually installed at the edge of farmlands. According to government guiding documents (Schultz, Collettil et al. 1995; Fischer and Fishchenich 2000), buffer strips' width was modeled as 30 meters to calculate pertaining energy consumption and coupled air emissions. Farming equipment used during installation and maintenance (i.e. tractors, mowers, cultivators) was assumed to be the same as those used within the tillage practice.

In addition, fertilizer application rates, average farmland acreage and corn yield rates were adjusted within GREET to match recent US Corn Belt levels. A detailed description is presented in Appendix C.

Nutrient outputs were calculated via a linear fraction model, which is similar to an emission factor approach and was described for use in agricultural LCAs in previous publications (Miller, Landis et al. 2006; Powers 2007). The general leaching models for total nitrogen (TN) and total phosphorus (TP) are defined based on a fraction of the applied nutrient load. Fertilizer input amounts are normalized by corn yield rates. Equation 2 and 3 describe the amount of TN and TP lost from fields in runoff L^{runoff}_{N}, L^{runoff}_{P}, which are calculated as the fraction of chemical lost with respect to the total mass of chemical applied to a crop. The nitrogen model is amended to account for the subsequent loss of nitrate via the denitrification process. Actually there are complex interactions among N/P species, plants and surrounding environments. This model does not attempt to model such interactions but rather to utilize LCI in conjunction with MCA to represent the possible range of N/P in runoff. It was assumed no soil P

was available from the previous year as an input. Phosphorus inputs from the air (deposition), rain, and wind erosion were not accounted for.

Agricultural processes contain high degree of system variability which depends on geography, weather patterns and soil type etc. The use of MCA has been incorporated into LCA in previous studies and has been shown to provide an appropriate representation of agricultural variability (Miller, Landis et al. 2006; Landis, Miller et al. 2007).

Statistical software (Crystal Ball7.0 and Minitab 11.5) are used to model the variability and calculate the uncertainty of the linear fraction model. Chi-squared and Anderson-Darling tests were used to determine the best fit distribution for the models' input parameters and then MCA was employed to develop cumulative probability curves of parameters at confidence level of 90%. Crystal Ball and Minitab complemented each other's capabilities to provide optimized distribution of nutrient runoff. Independent observation values were collected to calibrate equations' parameters and verify modeling results. Detailed procedures and datasources are explained in Appendix C.

Life cycle impact assessment (LCIA) aims at describing the environmental consequences of the environmental emissions quantified in the inventory analysis. The impact assessment is achieved by translating the environmental loads from the inventory results into environmental impact using LCIA tools. TRACI was used to facilitate the characterization of stressors that may have global warming potential and eutrophication potential in this study. Global warming and eutrophication potential categories were calculated by multiplying TRACI's characterization factor (CF) corresponding to each LCI emission to the LCI output emission value. The CF represents the equivalent effect of individual compounds with respect to a reference substance. CO_2 equivalent and nitrogen equivalent were used as common units to aggregate and compare

environmental impacts of LCI emissions. The global warming potentials of CH_4 and N_2O are, respectively, 23 and 296 times as that of CO_2. The eutrophication potential of phosphorus (P) is 7.4 times as that of nitrogen (N).

6.3 RESULTS

6.3.1 Comparing conventional tillage with no tillage practices

As Figure 11 shows, the use of no tillage practices could result in a large reduction of fuel consumption compared to conventional tillage. No tillage practices require much less energy for soil preparation and cultivation processes. There are no distinct differences in energy consumption resulting from pesticide application, irrigation, storage and transportation between conventional tillage practices and no tillage practices. Theoretically, conventional tillage practices require lower amounts of pesticides because moldboard or field cultivator serves the function of killing and removing weeds. However, no significant differences are discovered within this study, which is possibly due to the system boundary definition which includes energy consumption of spraying pesticides and excludes manufacturing pesticides.

Other published reports show that more than 40% energy savings from reduced agricultural machinery operation can be obtained, when conventional tillage practices are switched into no tillage practices (Kim and Dale 2005). The results of this study are lower than previously reported values because broader farming categories were incorporated into farming inventories, including planting, spraying, harvesting, storing and transportation.

Accordingly, the amounts of air emissions are higher in conventional tillage practices than no tillage practices. Air emissions are released through manufacturing and operating farming machines. The main species quantified through GREET are CO_2, CO, CH_4, SO_X, and NOx. These emissions in part result from steel production and machine assembly processes, from vehicle usage, and from transportation processes. CO, SO_x, and CH_4 emissions are generated from incompletely oxygenated combustion during producing steel.

Furthermore, no tillage practices contribute considerably to sequestration of soil organic carbon (SOC), which can offset carbon emissions from agricultural inputs and machinery (Lal and Kimble 1997; West and Marland 2002; West and Post 2002). Tillage practices expose soil organic carbon in the soil surface, increasing the rate of biomass decomposition and mineralization, and thus increase the release of carbon dioxide into atmosphere. The Center for Research on Enhancing Carbon Sequestration in Terrestrial Ecosystems within the US Department of Energy estimates that changing from conventional tillage to no tillage will result in sequestration of 31 kg C/(ha*year) in agricultural soils, to a depth of 30cm. Although SOC is expected to change in response to a change in management practices, the change will be finite and the concentration of SOC will approach a new steady state after 10-20 years (West and Post 2002).

Tillage practices also influence aqueous nitrogen and phosphorus emissions from corn farmlands (Dinnes, Karlen et al. 2002). A slight increase in nutrient discharge was observed when conventional tillage is compared with no tillage practices. However, according to the distribution and range of probable values shown in Figure 4 in SI, no tillage nutrient leaching is highly variable, thus the advantages to conventional tillage with respect to nutrient leaching are

unclear. Tillage activities, by loosening soil structure and enhancing nutrient uptake rate, may result in lower nutrient leaching.

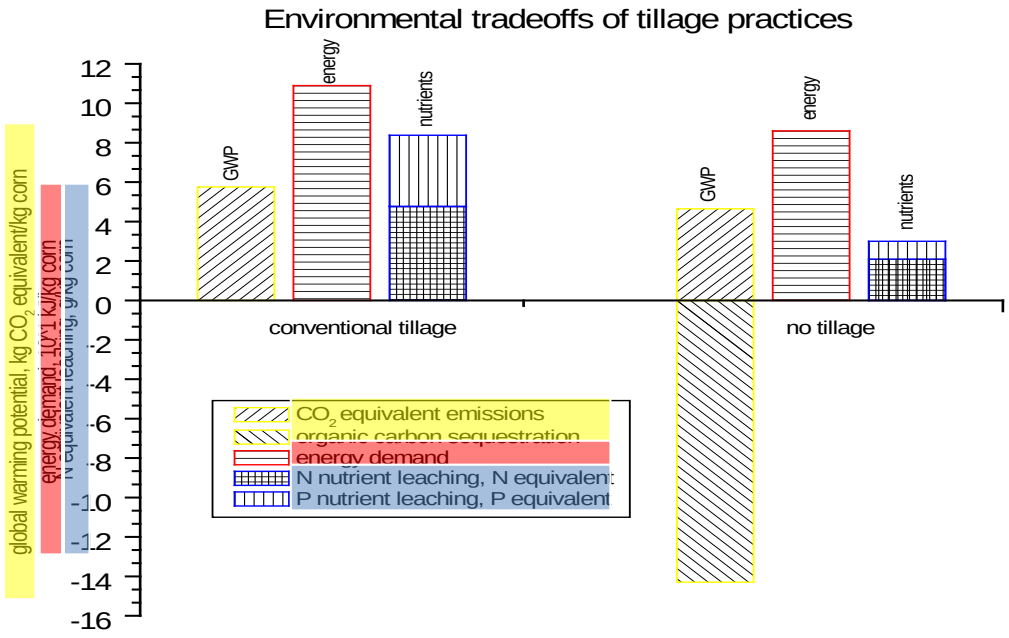

Figure 11. Environmental impacts of tillage practices

6.3.2 Comparing fertilizer types

Figure 12 presents the environmental impacts of alternative fertilizers. Energy consumption during fertilizer manufacturing, processing and transportation was determined using average values of ammonium nitrate, urea and ammonium. From a life cycle perspective, ammonium nitrate is the most energy demanding product among the four types of fertilizers evaluated (2.1 MJ/kg corn), followed by urea (1.7 MJ/kg corn), ammonium (1.4 MJ/kg corn), and finally manure (as coproduct in animal raising system, 0.2 MJ/kg corn), shown in detail in supporting

information. If farmers choose synthetic fertilizers as nutrient sources to produce 1 kg corn, around 1.7 MJ energy is required to transport raw materials, produce fertilizers, deliver final products to farmlands, and support all upstream activities as well. Compared with synthetic fertilizers, the manure requires much less energy than synthetic fertilizer over its life cycle. If manures are considered as coproducts of animal raising systems, energy embedded in manure practices (including building and operating animal raising systems, transporting manures) is around 0.2 MJ energy/kg corn. The energy used to haul manure to farmlands is minimal compared with energy allocated from the animal husbandry system, thus energy consumption for manure as waste is negligible.

These results are consistent with previous research about fertilizers. The synthesis of inorganic nitrogen fertilizer was reported as a very energy demanding process, typically consuming around 25-35 GJ to produce 1 ton ammonia through the steam reforming process (Kongshaug 1998). Natural gas, as both raw materials and possible energy source, is the dominant source of energy to produce fertilizers. Energy consumption of corn farming was investigated using life cycle assessment methods in previous studies (Kim and Dale 2005; Landis, Miller et al. 2007). To estimate total energy input for corn farming, these studies accounted for energy for planting, fertilizing, spraying, harvesting, as well as the energy required to manufacture synthetic fertilizers, pesticides, and limes. These studies show the total energy use varies from 1.5 to 3.4 MJ/kg corn depending on boundaries and allocation methods; Synthetic fertilizers accounts for a large portion of the energy consumption.

Figure 12 also shows greenhouse gas emissions, while Appendix C provides resultant life cycle air emissions attributed to synthetic fertilizer and manure practices. CO_2, CH_4, and N_2O gases are the dominant emission species for both synthetic fertilizers and organic manures.

Results show organic manure practices emit more CH_4 and N_2O. Higher amounts of N_2O emit from manure storage rooms and handling processes. The high release rate of CH_4 is due to biological degradation reaction during storing and handling manure (Amon, Amon et al. 2001; Sommer, Petersen et al. 2004; Loyon, Guiziou et al. 2007). In addition, significant amount of on-farm N_2O emission after applying manures also increases the global warming potential of manures.

Statistical analysis shows the nutrient discharge capabilities of fertilizer practices are different. The nitrogen amounts mainly vary from 4 to 10g N/kg corn, from 0.01 to 10g N/kg corn for synthetic fertilizers and manure respectively. Probability distributions of aqueous nitrogen and total phosphorus runoff values are illustrated in SI. The distribution curves representing cumulative percentages of nutrient runoff were estimated via MCA. The possible runoff concentrations of manure practices have a broader range and stronger tendency to approach high concentrations than synthetic fertilizers. While discharge concentrations lower than 4 g N/kg corn and higher than 10 g N/kg unlikely exist for synthetic fertilizers, those concentrations happen frequently for manure fertilizers. High uncertainty of nutrient contents and nutrient availability rates results in variability of nitrogen leaching for manure practices (Wood, Wood et al. 1999; Zhao, Gupta et al. 2001). For phosphorus discharge, there is no remarkable difference of distribution ranges between synthetic fertilizers and manure. However, organic manure has slightly higher possibility to leach more phosphorus. This phenomenon was also observed in previous studies (Sharpley, McDowell et al. 2001; Hart, Quin et al. 2004; Alexander, Smith et al. 2007). Because of the imbalance of N and P ratios of manure, the P fraction is usually overloaded to farmlands in order to ensure crops' N nutrient requirements, consequently generating higher P leaching potential for manure practices.

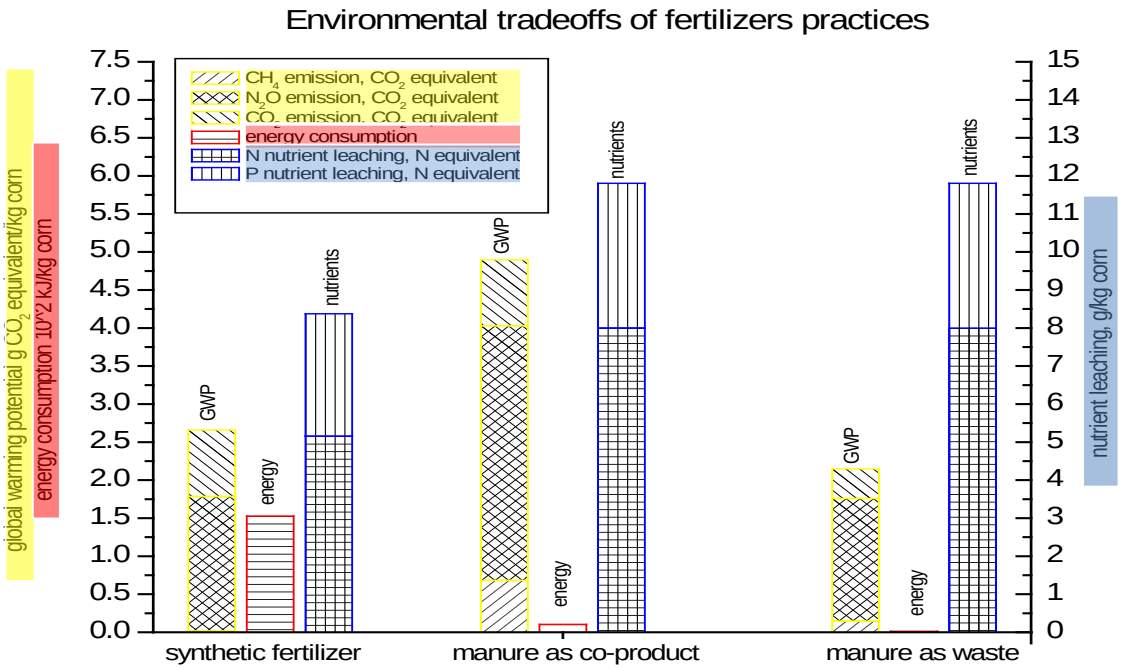

Figure 12. Environmental tradeoffs of fertilizer practices

6.3.3 Effects of buffer strips

Life cycle energy consumption and air emissions of building and maintaining the buffer strip are low compared to the entire life cycle impacts of corn farming. Planting, cultivation and mowing practices were incorporated to calculate energy flows of buffer strips. Generally, farmlands' area is much bigger than surrounding buffer strips' area, so the values of energy demand and associated air emissions resulting from buffer strips are very low when corn product related units are used to represent energy consumption and associated air emissions of managing buffer strips. However, the nutrient removal efficiency of buffer strips is significant. As Figure 13 shows, the nitrogen removal rate of buffers is estimated to be 75% while the phosphorus removal rate is

67%. Therefore, buffer strips are effective management tools for controlling nutrient leaching when GWP, energy, and eutrophication impacts are of concern for agricultural feedstocks.

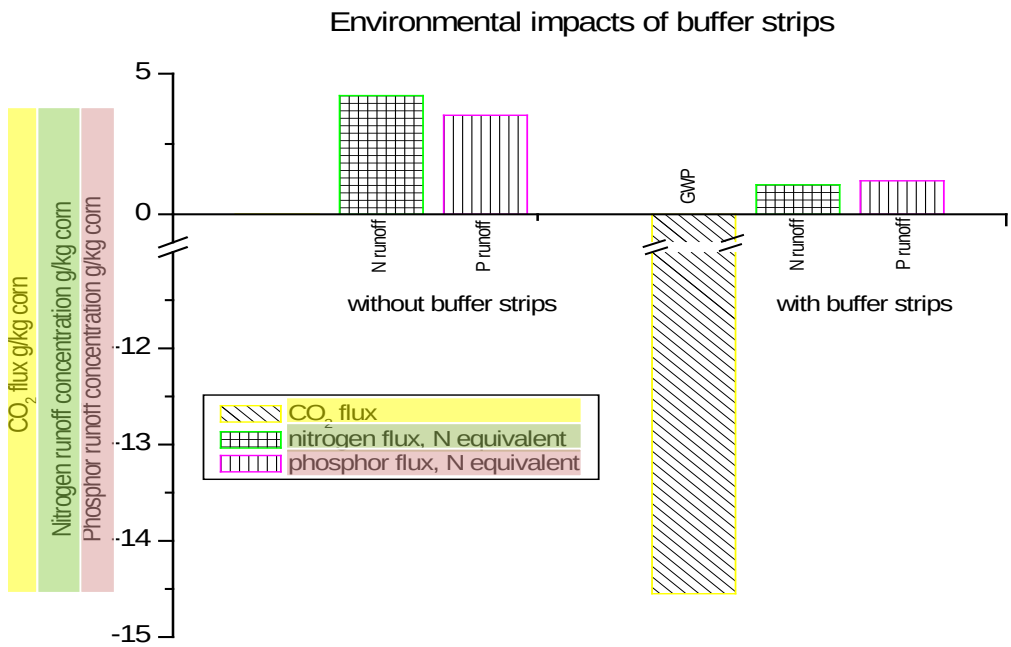

Figure 13. Environmental impacts of buffer strips

6.4 DISSCUSSION

6.4.1 Improving environmental performances of corn by altering farming practices

As discussed above, optimizing farming practices has the potential to reduce global warming potential and eutrophication potential resulting from corn farming in Corn Belt states. Appropriately altering farming practices (such as adopting no tillage and installing buffer strips)

can systematically improve environmental performances of corn farming. The effects of shifting farming practices on environmental impacts of corn are shown in Figure 14. The baseline of environmental impacts was determined using corn production reported by USDA and publications (Landis, Miller et al. 2007). We assume marginal shifts of farming practices do not change environmental impacts of each farming practice. Results show that installing buffer strips is the most effective way to reduce eutrophication potential compared to the use of manure and no till. Shifting synthetic fertilizers to manures has the greatest potential to reduce global warming potential among the three farming practices. Use of manure may increase eutrophication potential of corn farming. However, there are practical difficulties to apply these environmentally preferred practices. Corn farmlands are tiled in the Midwest, and buffers tend to be ineffective in removing nutrients in these areas. Potential economic loss resulted from retiring farmlands and installing buffer strips may discourage farmers to use buffer strips. Although the price of manure is relatively cheap, transporting manure from storage room to farmlands is relatively expensive. Besides expensive transportation, difficulties related to manure handling are also reported as barriers of manure application. In addition, this article only investigated the influences of three types of farming practices (tillage types, fertilizer types, the use of buffer strips) on environmental impacts of corn farming. The impacts of other important farming practices (such as crop rotation, fertilizer application techniques, etc.) should be researched to aid policy decision in the future.

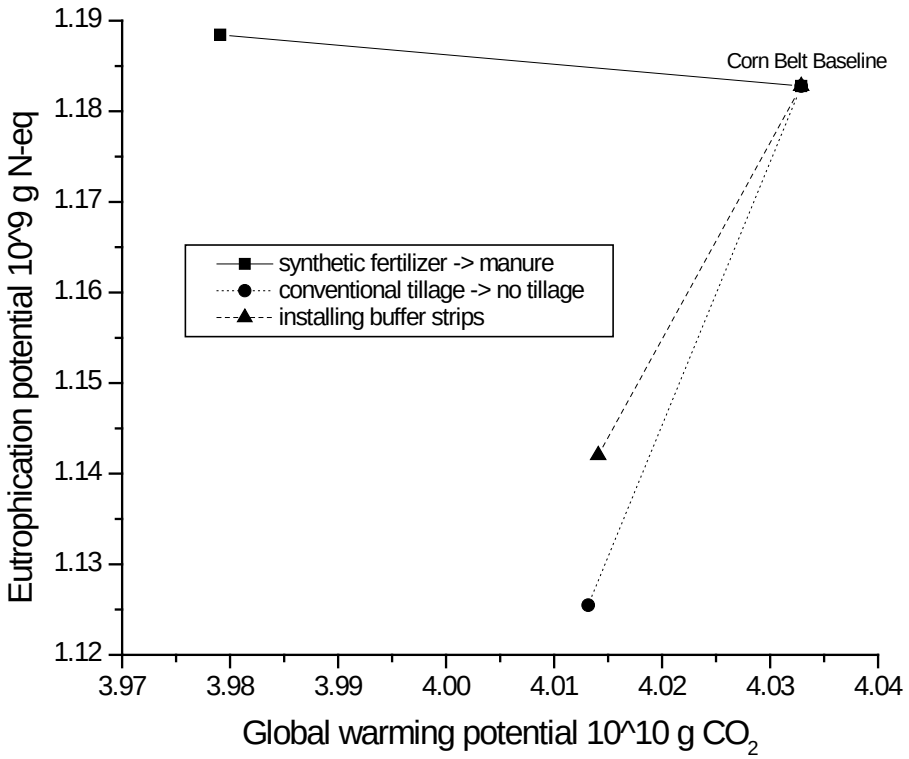

Figure 14. The potential of improving environmental impacts of corn farming by shifting farming practices in Corn Belt states

7.0 SUMMRAY AND RECOMMENDATIONS FOR FUTURE WORK

7.1 SUMMARY

Because of the demand for food and fuels to support the expanding world population and economic development, the application of fertilizers is predicted to continually increase. This will likely worsen coastal eutrophication and hypoxia. Effective and efficient solutions should be employed to reduce nitrogen applications and resultant nitrogen loads, thus eventually mitigating eutrophication issues.

Generally, there are three approaches to mitigate environmental nitrogen output: economical-wide shift due to changing purchasing behaviors; top-down control of pollution; and bottom-up run-off prevention strategies. Changing food purchase behaviors is the most effective method to reduce nitrogen impacts. If consumers reduce their demand for red meat, the nitrogen use for food production will decrease dramatically since meat production is a major contributor to eutrophication. However, this is a difficult strategy to employ in practice. Humans need protein from red meat for dietary and health reasons. In addition, relatively cheap prices and tastes stimulate people to consume red meat. Shifting consumption patterns and life styles is a rather slow process. Education and price regulation will encourage people to purchase foods with better nitrogen profiles. Top-down control of pollution focuses on optimizing farming practices during producing stage, for example, applying synthetic fertilizer precisely, advanced application

techniques, effectively use manure as nitrogen source, coordinating with other farming practices (e.g. irrigation, tillage). This method is relatively easy to practice, but is not so effective and efficient. Choosing farming practices highly depends on soil, climate, plant type, local infrastructures, material supply, financial capability and farmer's conceptions. Previous studies show that efficient utilization of fertilizer and other farming technologies will reduce 10-20% of nitrogen runoff from farmlands (Zhao, Gupta et al. 2001). Reusing manure (or waste from meat production) for nourishing plants will reduce nitrogen input for integrated food supplying, but handling and transporting manures will result in considerable air emissions and other environmental issues (Amon, Amon et al. 2001). Additionally, improving food process techniques and reducing food lost may also control final nitrogen runoff from life cycle perspectives. Bottom-up run-off prevention strategies include installing buffer strips at the end edge of farmlands, constructing wetlands near pasture and crop farmlands, using other water treatment facilities to remediate nutrient runoff (such as, chemical and biological treatment tanks for animal raising systems). This is an effective solution from a water quality protection perspective. Generally buffer strips can remove 60% of nitrogen, and constructed wetland can move 65% of nitrogen (Bundy, Andraski et al. 2001). However, farmers may not want to install buffer strips because of reduced farmlands' area and potential decreased grains yield. The economic feasibility of investing and constructing wetlands/other water treatment tanks is dubious. Water treatment facilities also have environmental impacts during their life cycle, and influence land use and community development. The life cycle environmental impacts and social efficiency of runoff prevention strategies is arguable. The agricultural sector is very important from environmental, economic, and social perspectives. A portfolio of solutions should be

suggested to meet the requirements of abundant food supply, acceptable environmental impacts, and socially sustainable development.

7.2 ROCOMMENDATIONS FOR FUTURE WORK

Although life cycle assessment methods have been utilized to quantify eutrophication potentials of products and processes recently, a lack of nutrient inventories and appropriate characterization factors (CF) is a major obstacle for LCAs to accurately estimate eutrophication potential.

Most of the available eutrophication CFs do not consider temporal and spatial differentiation of N/P emissions and their consequent environmental impacts. The CF represents the equivalent effect of individual compounds with respect to a reference substance. Eutrophication potential is calculated by multiplying TRACI's CF corresponding to each LCI emission to the LCI output emission value. Generally, nitrogen equivalent is used as a common unit to aggregate and compare eutrophication impacts of LCI emissions. TRACI has site-independent characterization factors for the whole US and limited CFs for several states. Research has demonstrated that eutrophication impacts are site dependent. The fate processes rely on variable characteristics of the emitting source, environmental media, and receiving environments. The impacts depend on background loads and different sensitivities of different ecosystems. CFs reflecting regional characteristics can help to quantify eutrophication potential and form effective strategies to combat eutrophication issues. Spatial differentiation has not been addressed adequately, because it is difficult to determine the environmental transport pathways and ultimate fates of emissions in complex ecosystems. Furthermore, the difficulties to tailor large scale emission transport models to reflect the specific region also inhibits accurate

calculation of regional CFs. To tackle this problem, interdisciplinary cooperation is urgently needed to propose regional eutrophication CFs and improve current impact assessment methods.

Current life cycle impact assessment tools do not account for nitrogen- vs. phosphorus-limited nature of eutrophied water bodies. Adding eutrophication potentials of all nitrogen and phosphorus emissions together may overestimate regional eutrophication potential. To reflect actual eutrophication potentials, it is important to determine limiting nutrients in eutrophied ecosystems. If nitrogen is the limiting species (for marine waters, mostly), eutrophication potential of phosphorus is zero. If phosphorus is the limiting form (for freshwaters, mostly), eutrophication potential of nitrogen is zero. If both of nitrogen and phosphorus forms are controlling elements, then the contributions from both nutrients should be accounted for eutrophication potential impact category. LCA coupled with field and modeling data should be developed to identify limiting nutrients and estimate the actual eutrophication potential of water bodies.

Existing life cycle assessment tools combine hypoxia and eutrophication potential as a single impact category. Although overload of Nitrogen/Phosphorus emission could result in hypoxia issues, the definition of hypoxia is distinct from eutrophication potential. The oxygen depletion model incorporated with nutrients inventory should be utilized to reflect hypoxia extent. Furthermore, life cycle impact assessment method is still lacking a procedure that relates nutrient emissions to ecological damage in ecosystems. Most of impact assessment tools (including TRACI, CML, IMPACT2002+ et al) only provide eutrophication CFs at mid-point level. Damage factors based on a concentration-response relationship between the concentration of nutrients and occurrences of illness should be determined to access possible ecological damage due to excessive nutrient inputs into ecosystems.

Moreover, to aid policy decisions for mitigating eutrophication potential, the analysis of variability and uncertainty is necessary to understand the possible impacts of policies and technologies. Typically, LCA studies use average values for inventory data to describe the components of a system. While this practice may be acceptable in industrial systems where variability is limited and uncertainties may be characterized, caution must be used in systems where average data do not depict the range of probable scenarios. Not only will the inventory outcomes vary considerably, but the potential impacts may be highly dependent on this variability.

It is important to draw the distinction between natural variability and uncertainty associated with data collection. Variability pertains to naturally occurring fluctuations which may include differences in geographic and climatic factors, or changes in agricultural practices. Uncertainty is characterized by the lack of confidence in a given parameter. Parameter distributions in the work are a measure of both naturally occurring variability and uncertainty in the distributions. The distributions show the range of emissions from the system, and how the emissions can change depending on a variety of factors. We suggest policy makers consider variability and uncertainty associated with each mitigation strategy.

Besides improving life cycle assessment tools, many questions about biofuel sustainability remain unanswered. Getting to a sustainable biofuel economy will also require a more comprehensive and collaborative research agenda than what has been undertaken to date. In particular, there is an urgent need for research that emphasizes a system approach to assess the energy yield, carbon implications, and the full impact of biofuel production on downstream and downwind ecosystem; a focus on ecosystem services-including those that are biodiversity based-to provide the information necessary for the development and implementation of land

management approaches that meet multiple needs; and an understanding of the implications of policy and management practices at different spatial scales-from farm and forest to landscapes, watersheds, food-sheds, and the globe- and an assessment of alternative cost-effective policies designed to meet sustainability goals.

APPENDIX A

SUPPORTING INFORMATION FOR EUTROPHICATON POTENTIAL OF FOODS

Appendix A presents assumption of food packaging, LCI data sources, and values of nitrogen intensity by different functional units.

A.1 ASSUMPTIONS OF FOOD PACKAING

Table 1. Assumption of food packaging

Food type	Packaging type	assumptions
Bread	Paper, plastic bag	0.5g PVC/500g bread
Cereals	Paper box, plastic bag	20g paper/200g cereal, 3g PVC/200g cereal
Apple	Plastic bag	2g PVC/2 Kg apple
Vegetables	Plastic bag	2g PVC/2 Kg vegetable
Tomatoes	Plastic box, plastic bag	2g PVC/ 500g tomatoes
Oil	Glass bottle, plastic bottle	70g glass/560 g oil, 20g PET/560g oil
Cheese	Plastic bag	3g PET/500g cheese

Table 1. (Continued)

Fish	Plastic bag, wrap	2g PET/500g fish
Shrimp	Plastic bag, wrap	2 g PS/ 500g pork, 1g PVC/500g pork
Trout	Plastic bag, wrap	2 g PS/ 500g pork, 1g PVC/500g pork
Chicken	Plastic bag, wrap	2 g PS/ 500g pork, 1g PVC/500g pork
Beef	Plastic bag, wrap	2 g PS/ 500g pork, 1g PVC/500g pork
Pork	Plastic bag, wrap	2 g PS/ 500g pork, 1g PVC/500g pork
Beverage	Plastic bottle, metal can	15g PEPT/500g liquid, 13 g aluminum can/300 liquid
ketchup	Plastic bottle, metal can	10g PEPT/300g ketchup

A.2 LIFE CYCLE ACTIVITES FOR RESEARCHED FOOD GROUPS

Table 2. Main products and activities considered in LCI for researched food groups

Food groups	Main products	Main activities
Cereals/carbs	White and whole wheat, durum, rye flour, rice, oat products, barley products, flour and meal, starch	Grain farming, flour/rice/wet corn milling, no frozen bread/bakery products, cookie and cracker manufacturing, mixes/dough from purchased flour, dry pasta manufacturing, tortilla manufacturing, breakfast cereal manufacturing
Fruit/vegetable	Apple, carrot, tomato, onion, peas, potato	Vegetable and melon farming, fruit farming, fruit and vegetable canning/drying

Table 2. (Continued)

Chicken/eggs	Chicken, turkey, eggs with shell, processed eggs	Tree nut farming, all other crop farming, poultry and egg production, poultry processing
Fish	Fish, shellfish, tuna(canned), salmon(canned), sardines(canned), other canned fish and cured fish	Fish feed processing, fishing, seafood preparation and packaging
Red meat	Beef, lamb and mutton, pork	Cattle ranching and farming, manure management, hunting and trapping, slaughtering, meat processed from carcasses
Oils	Butter, margarine, cooking oils, edible beef tallow, other edible fat and oils	Soybean processing, fats and oils refining and blending
Sweets/condiments	Cane and beet sugar, edible syrups, honey	sugar manufacturing, confectionery from cacao beans, mayonnaise/dressing/sauces, confectionery from purchase chocolate
Dairy products	Whole milk, skim milk, cream, cheese, ice cream, yogurt	Fluid milk manufacturing, creamery butter manufacturing, cheese manufacturing, dry/condensed/evaporated dairy, ice cream and frozen desserts

A.3 DATASOURCES

Table 3. Detailed datasources

Food subgroup	LCA stage	Datasources
Red meat	Farming	Journal articles((Cederberg and Stadig 2003; Basset-Mens and van der Werf 2005)), Ecoinvent
	Processing	Ecoinvent, LCA food DK, Industry data 2.0, BUWAL250
	Transportation	GREET
	Packaging	Ecoinvent, Franklin USA98, IDEMAT 2001, Industry data 2.0
oils	Farming	Journal articles((Brentrup 2003)), Ecoinvent
	Processing	Ecoinvent, LCA food DK, Industry data 2.0, BUWAL250
	Transportation	GREET
	Packaging	Ecoinvent, Franklin USA98, IDEMAT 2001, Industry data 2.0
Sweets and condiments	Farming	Journal articles((Andersson, Ohlsson et al. 1998; Brentrup, Kuters et al. 2001; Ramjeawon 2004)), Ecoinvent
	Processing	Ecoinvent, LCA food DK, Industry data 2.0, BUWAL250
	Transportation	GREET
	Packaging	Ecoinvent, Franklin USA98, IDEMAT 2001, Industry data 2.0

Table 3. (Continued)

Fruits/vegetables	Farming	Journal articles((Jones 2002; Schau and Fet 2008)), Ecoinvent
	Processing	Ecoinvent, LCA food DK, Industry data 2.0, BUWAL250
	Transportation	GREET1.8
	Packaging	Ecoinvent, Franklin USA98, IDEMAT 2001, Industry data 2.0
Dairy products	Farming	Journal articles((Cederberg and Mattsson 2000; Berlin 2002; Eide 2002; Cederberg and Stadig 2003; de Boer 2003; Hospido, Moreira et al. 2003; Thomassen, Dalgaard et al. 2008; Thomassen, van Calker et al. 2008)), Ecoinvent
	Processing	Journal article((Danalewich, Papagiannis et al. 1998)), Ecoinvent, LCA food DK, Industry data 2.0, BUWAL250
	Transportation	GREET
	Packaging	Ecoinvent, Franklin USA98, IDEMAT 2001, Industry data 2.0
Chicken/eggs	Farming	Ecoinvent
	Processing	Ecoinvent, LCA food DK, Industry data 2.0, BUWAL250
	Transportation	GREET
	Packaging	Ecoinvent, Franklin USA98, IDEMAT 2001, Industry data 2.0

Table 3. (Continued)

fish	Farming	Journal articles((Ziegler, Nilsson et al. 2003; Hospido and Tyedmers 2005; Hospido, Vazquez et al. 2006; Thrane 2006; Maz, Piedecausa et al. 2007; Pelletier, Ayer et al. 2007; Zufia and Arana 2008; Thrane, Ziegler et al. 2009)), Ecoinvent
	Processing	Ecoinvent, LCA food DK, Industry data 2.0
	Transportation	GREET
	Packaging	Ecoinvent, Franklin USA98, IDEMAT 2001, Industry data 2.0
Cereals/carbs	Farming	Journal articles((Andersson and Ohlsson 1999; Brentrup, Küsters et al. 2000; Brentrup 2003)), Ecoinvent
	Processing	Ecoinvent, LCA food DK, Industry data 2.0
	Transportation	GREET
	Packaging	Ecoinvent, Franklin USA98, IDEMAT 2001, Industry data 2.0

APPENDIX B

SUPPORTING INFORMAITON FOR ENVIRONMENTAL IMPACTS OF BIODIESELS

Appendix B explains system boundaries for each stage, transportation distances estimation and comparison to other studies.

B.1 DETAILED SYSTEM BOUNDARY

B.1.1 System boundary for soybean agriculture

The system boundary includes downstream on-field soybean farming production and upstream material and energy flows. Soybean farming activities including planting seeds, tillage, fertilizing, applying pesticides and herbicides, harvesting, and storing soybean grains are calculated. Moreover, upstream activities, such as construction, transportation, energy generation and power supply, manufacturing farming equipment and chemicals, producing animal manure, etc. are also included. Air emissions, water emissions, solid emissions during all mentioned processes are estimated.

B.1.2 System boundary for soy oil extraction, processing and transportation

The system boundaries of processing and transportation are concentrated in three main contributors, transportation, soy oil processing, and transesterification. The system as a whole is depicted in Figure. The processing phase was broken into two major parts, soy oil extraction and soy oil transesterification. Those two major categories contain the smaller stages such as wastewater treatment and minor onsite transportation. The second boundaries were mainly geographical and were ones that were established by location of PennDOT's fuel storage tanks as well as the location of the distribution centers that serviced those tanks. Since this task is evaluating only the PennDOT district 8 biodiesel use it focused strictly on those areas that would service the fuel tanks in District 8. In order to obtain the location of the biodiesel distribution locations, the fuel provider was contacted and locations were given for distribution facilities. Then, since PennDOT uses primarily soy based biodiesel, the system boundaries were set to include production facilities only within Pennsylvania and the states directly surrounding Pennsylvania that used soy beans as one of its primary feedstock.

B.1.3 System boundary for vehicle and fuel combustion

The primary source of environmental impacts for the use of a fuel is the exhaust gas resulting from the combustion of the fuel in the vehicle's engine. To the extent that different fuels may have different power outputs (as evidenced by fuel economy), may lead to increased or decreased maintenance, or affect the overall life of the equipment, these differences should be included in a comparative LCA of these fuels. However, there was no discernible effect on either

fuel economy or maintenance cost between ULSD and B5. Due to time frame of this study, data related to changes in vehicle lifetimes was not available for analysis; it was further assumed that because no significant effects were found for fuel economy and maintenance, that the comparative effect on vehicle lifetime is similarly negligible. Because PennDOT and other agencies have implemented aggressive preventative maintenance (pm) programs, it is likely that any issues which would affect vehicle lifetime would manifest themselves during pm inspections and testing. Additionally, no literature references were found that specifically addressed biodiesel use and equipment or engine lifetime. Thus, combustion emissions represent the only relevant component of the use phase LCI for this comparative assessment.

The life cycle of petroleum diesel includes extraction of the crude oil, whether using conventional drilling or newer enhanced recovery methods; transportation to the refinery; refining, and transportation to the point of use. Both oil extraction and oil refining have significant environmental impacts which must be included in the LCA. Unlike biodiesel, both crude petroleum and finished fuel products, such as diesel, are transported long distances via pipeline, which is somewhat more energy-efficient than truck or even train for transportation. These differences require attention and merit a separate LCI section for petroleum diesel from extraction to the point of use.

B.2 TRANSPORTATION DISTANCE ESTIMATION

The model for the transportation phases was developed in two stages: transportation of soy from field to production facility and transportation of biodiesel from production facility to distribution

facility. Because supplier and purchasing information is proprietary there was no directly accessible information specific to PennDOT as to the origin of the raw soybean. Thus the distance from the field to the processing facility was assumed to be 450 miles, which is the distance used in GREET. This distance was deemed reasonable since that is relatively the distance from many of the processing facilities to the Corn Belt in the Midwest.

The second portion of the transportation model was developed using information collected from PennDOT as well as its biodiesel supplier, Petroleum Products Corp (PPC). The locations of PennDOT's fuel storage tanks in District 8 as well as the location of PPC's distribution facilities were collected. The distribution network was then analyzed to find appropriate transportation distances between the distribution facilities and the storage tanks. The calculated distance of transportation from the distribution facilities to the tanks was about 17.5 miles with a standard deviation of 12.5 miles. A 95% confidence interval puts the distance between 5.5 and 29.5 miles for the final distribution.

The distance from the production facilities to the distribution facilities was determined in a similar manner. The difference was that the exact supplier was unknown and therefore a network was developed using a few assumptions. Since it is specified that PennDOT will only use plant based biofuels, all production facilities in a one state radius that used primarily soybean or rapeseed based feedstocks were plotted on the same map. Then distances from each production facility to each of PPC's distribution facilities were calculated. As shown in Table 1 the network was then analyzed to determine the average distance to the eight distribution facilities from the production plant. These values were then averaged to come up with an average transportation value of 221 miles. Based on the 95% confidence interval the transportation distance is between 148 and 295 miles.

B.3 FEEDSTOCK SUPLLY ESTIMATION

According to USDA data, 17.63 million bushels of soybeans were produced on 430,000 acres of farmland in Pennsylvania in 2007. In the same year, at least 9.2% of total soybeans produced in the US went into biodiesel refineries to meet national biodiesel consumption. If we follow this conservative percentage, Pennsylvania soybeans can potentially provide at least 2.11 million gallons of biodiesel. By contrast, total PennDOT diesel fuel consumption in 2007 was 13.2 million gallons. Statewide B5 implementation would thus require 5% of that total or 0.662 million gallons of biodiesel. Meanwhile, biotechnology development and economic incentives may increase biodiesel production to ensure B5's feedstock supply in near future. Basing on above estimation, soybean production in PA has enough production to satisfy state wide implementation of B5. However, it is evident that limited arable farmland areas in Pennsylvania may not be able to supply sufficient feedstock to meet demand for higher biodiesel blend levels.

B.4 COMPARING SOYBEAN FARMING IN CORN BELT WITH SOYBEAN FARMING IN PENNSYLVANIA

The comparative effects of soybean farming in Pennsylvania and Corn Belt states were estimated using TRACI. In general, Pennsylvania soybean farming resulted in reduced environmental impacts compared to Corn Belt agriculture. The higher chemical input (such as fertilizers, herbicides etc.) in Corn Belt states resulted in a higher global warming potential for soybean farming in Corn Belt. Manufacturing synthetic fertilizer is an energy-intensive process and generates a significant amount of CO2 due to fuel combustion. The high fertilizer application

rate also causes a relatively higher eutrophication potential generated from nutrient runoff and leaching in Corn Belt states. In addition, the herbicide application rate in PA is slightly lower than the rate in Corn Belt states, so environmental impacts associated with herbicides (such as human and ecological toxicity) are slightly lower than Corn Belt states. Figure provides a comparison of Pennsylvania and Corn Belt soybean agriculture across the TRACI impact categories, using Corn Belt agriculture as the baseline

B.5 COMPARISON TO OTHER STUDIES

Numerous studies have investigated environmental impacts of biofuels, with a primary focus on global warming potential and net energy value. In this study, we evaluated environmental impacts of biofuels in eight environmental impacts categories and compared our results to other published results in global warming potential category. As the Figure 4 shown, the estimated global warming potentials of biodiesel are different due to different geographical location, different system boundary, different allocation methods and distinct datasources etc. Kim et al investigated cradle-to-grate GHGs emissions of soybean oil in 40 counties in Corn Belt States. GHG emissions of soybean oil are 0.4-2.5 kg of CO2 equivalent per kilogram of soybean oil (Kim and Dale 2009). GREET model also predicts that the cradle-to-gate GHG of soybean oil is about 1.2 kg of CO2 equivalent per kilogram of soybean oil. Sheehan et al estimated the GHGs emission of biodiesel is around 136.45 gCO2 per horse power per hour (Sheehan, Camobreco et al. 1998). Our result is close to Sheehan's result and falls in the range of Kim's estimation. We adjusted GREET inputs by the latest chemical application rates and amended emission factors for fuels. These changes resulted in the discrepancy between our results and GREET results.

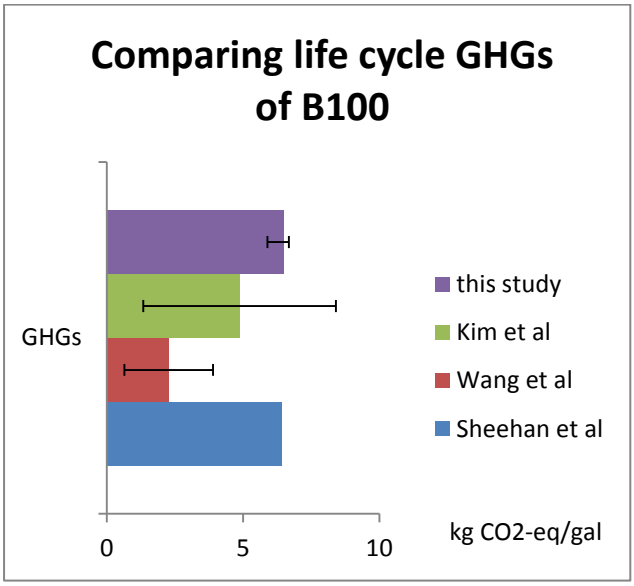

Figure 1. Comparison of global warming potential for different biodiesel studies

APPENDIX C

SUPPORTING INFORMAITON FOR ENVIRONMENTAL IMPACTS OF FARMING PRACTICES

Appendix C contains eight figures and nine tables to describe model development and use of datasources.

C.1 MODELS USED TO CREAT LIFE CYCLE INVENTORY

Table 4. Models used to create LCI

Models	Farming practices				
	Synthetic fertilizer	Manure	Conventional tillage	No tillage	Buffer strips
Default GREET 1.8 (energy and air emissions)	N(average of NH3, urea, and NH4NO)	Energy required for transportation and application	Energy required for manufacturing farming equipment	Energy required for manufacturing farming equipment	Energy required for manufacturing farming equipment

Table 4. (Continued)

Modified GREET 1.8 (energy and air emissions)	N application rates modified to reflect current situation of Corn belt states	Data collected from peer reviewed articles and governmental reports are used to estimate energy required for production and associated air emissions	Energy required for operating tractors and other farming equipment are calculated based on literatures.	Energy required for operating tractors and other farming equipment are calculated based on literatures. Carbon sequestration capability was obtained by publication.	Energy required for installing and maintaining buffer strips are estimated based on literatures. Carbon sequestration capability was estimated according to IPCC factors.
Fractional model (water emissions)	N and P runoff: loss from fields as a function of application rate, details are shown in table 1 in manuscript				N and P removal rate are estimated according to literature.

The GREET 1.8 model estimates the upstream energy required to produce basic fuel and subsequently aggregate these energy uses to an estimate of total energy. GREET 1.8 also calculates air emissions based on EPA emission factors. The water emissions were estimated by fractional model explained in Manuscript and Section 4 of Appendix C.

C.2 ENERGY CONSUMPTION AND GHGS EMISSIONS OF MANURE PRACTICES

C.2.1 Calculation logic

This Manure can be considered as waste product of animal raising system or co-product. Emissions from both scenarios are calculated.

If organic manures are considered as co-products of dairy, swine, or poultry raising systems, energy shares of producing manures are calculated through allocating total energy consumption between main products (meat or milk) and co-products (manures).

Energy consumption of producing organic manures = total energy consumption of animal raising system *allocation ratio

Total energy consumption for animal raising system = energy requirement of operating and maintaining animal farming (food intake, operating machines etc) + energy for transportation+ upstream activities

Total emission of organic manure = emissions from animal raising system*allocation ratio++ field emissions

Emissions from animal raising system =emission in animal house + emission in storage room + Emission from upstream activities

Energy allocation ratio (0.315) was used when manure is treated as co-product from animal raising system. This ratio is estimated based on energy flows on dairy farms in Midwest states (Jewell 1975). If manure is considered as waste, the allocation ratio is 0.

C.2.2 Datasources

Energy consumption values during every stage are collected from publications (Jewell 1975). IPCC equations and suggested values were used to calculate GHGs generated in animal house and storage room (IPCC 1996). GHGs emitted from corn farmland after manure application were estimated according to publications shown below.

C.3 ENERGY CONSUMPTION AND GHGS EMISSIONS OF TILLAGE PRACITCES

Energy consumption and related air emissions for applying and producing different farming equipments for soil preparation, planting seeds, cultivating soil bed, harvesting corn grains, irrigating corn farmlands, transporting corn grains to storage rooms are estimated. The values for producing stages are taken from GREET2.8. Energy consumption for applying different farming equipment was collected from publications (Uri 1998). Air emissions for application stages are calculated by energy consumption and GREET 1.8.

C.4 LINEAR NUTRIENT MODELS

C.4.1 Model description and datasources

The used models and datasources are explained below. Multiple sources including USDA, IPCC, and peer reviewed articles were collected for the farming practices study.

Table 5. Description of equations and parameters in linear nutrient models and their datasources

#	Variable	Description	Datasources
1	R	rate of application(kg x synthetic fertilizers or manure per ha crop)	USDA website, fertilizer application rate (USDA 2008)
2	Y	yield (kg crop grain per ha)	USDA website, corn yield rate (USDA 2010)
3	$f_{em,N}$	N runoff coefficient, this coefficient reflects the nutrient discharge potential of fertilizers and tillage practices.	Peer reviewed publication(Lucey and Goolsby 1993; Bjorneberg DL 1996; IPCC 1996; Weed and Kanwar 1996; David, Gentry et al. 1997; Jaynes, Hatfield et al. 1999; Vanni, Renwick et al. 2001; Bakhsh A 2002; Tomer, Meek et al. 2003)
4	$f_{em,P}$	P runoff coefficient, this coefficient reflects the nutrient discharge potential of fertilizers and tillage practices.	Peer reviewed publication(Gaynor and Findlay 1995; Gascho, Davis et al. 1998; McDowell, Sharpley et al. 2001; Cooperband, Bollero et al. 2002; Daverede, Kravchenko et al. 2003; Udawatta, Motavalli et al. 2004)
5	$f_{TN,NO3^-}$	ratio of nitrate to total nitrogen.	Peer reviewed publication (Powers 2007)
6	f_{de}	fraction of denifrication	Peer reviewed publication (Miller, Landis et al. 2006; Powers 2007)

C.4.2 Results and model validation

Linear models are used in agricultural life cycle assessment to estimate N, P concentrations of farmlands runoff. Probability distribution curves of nutrient runoffs are modeled through Minitab 15. In graphs, the three lines are cumulative probabilities via our models, upper lines are estimated values at 90% confident level, middle lines are median values; lower lines are estimated values at 10% confident level. Discrete dots are previously reported data for verifying models.

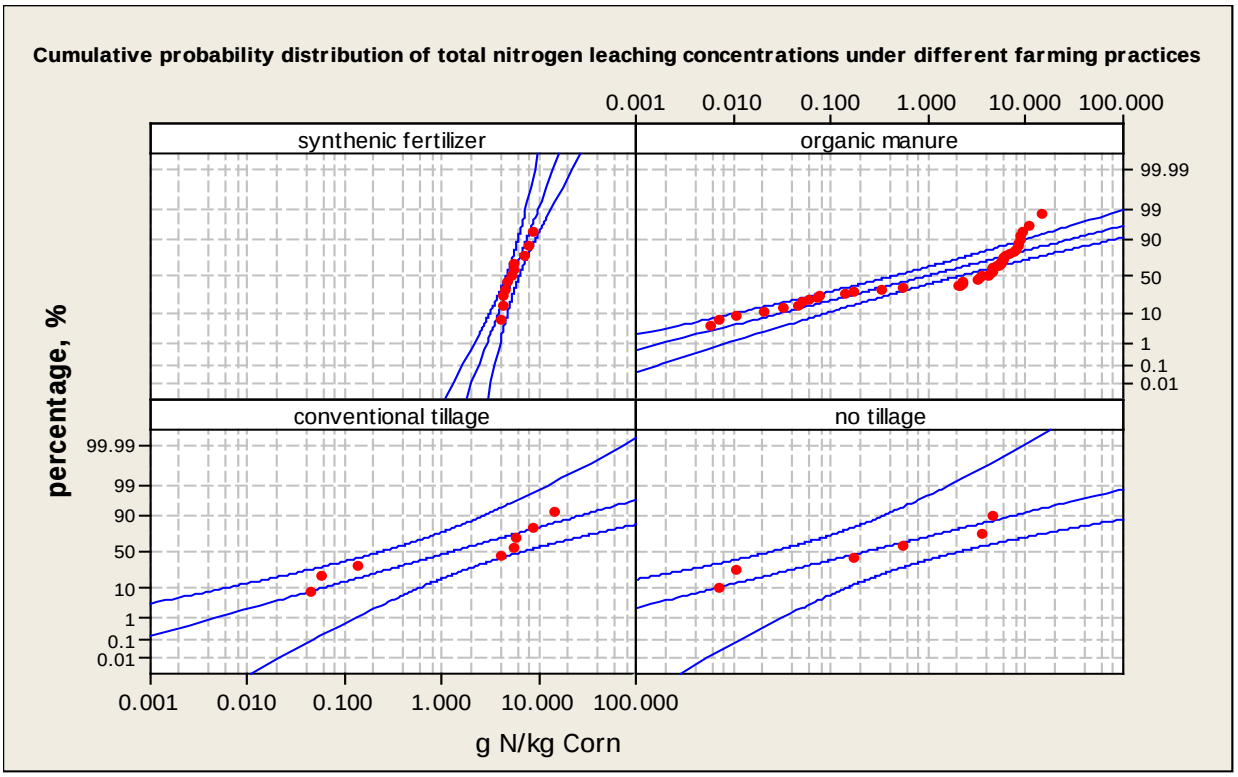

Figure 2. Probability distribution of total nitrogen leaching under different farming practices

In Figure 1, the three lines are cumulative probabilities of total nitrogen via our models, upper lines are estimated values at 90% confident level, middle lines are median values; lower lines are

estimated values at 10% confident level. Discrete dots are nutrient runoff values collected from publications. These values are used to validate model inputs and reported below.

Table 6. Independent observations used to validate the model

Items		Resources used for model validation
N runoff from different farming practices	Synthetic fertilizer	(Franklin, Cabrera et al. 2005; Miller, Landis et al. 2006; Landis, Miller et al. 2007)
	Manure	(Cooperband, Bollero et al. 2002; Franklin, Cabrera et al. 2005)
	Conventional tillage	(Angle, Mc Clung et al. 1984; Mcdowell 1984; Weed and Kanwar 1996; Miller, Landis et al. 2006; Landis, Miller et al. 2007)
	No tillage	(Angle, Mc Clung et al. 1984; Mcdowell 1984; Weed and Kanwar 1996; Miller, Landis et al. 2006; Landis, Miller et al. 2007)

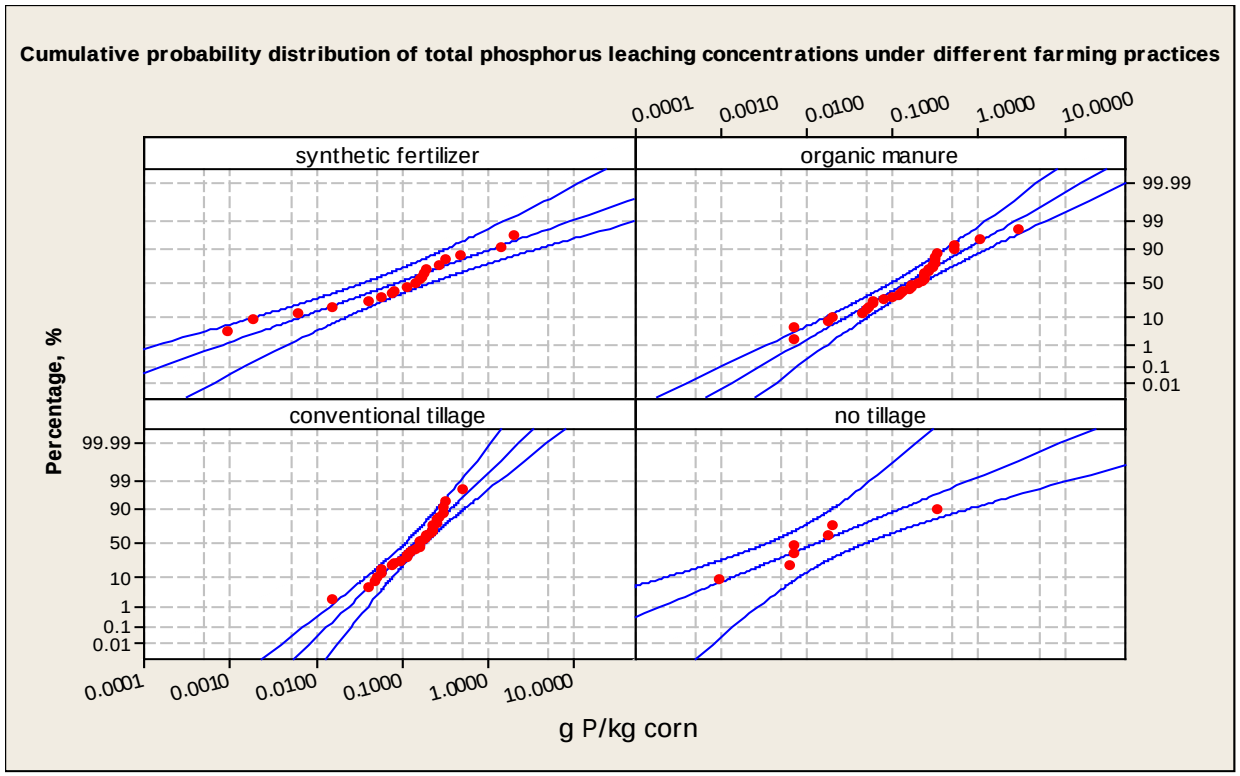

Figure 3. Probability distribution of total phosphorus leaching under different farming practices

In Figure 3, the three lines are cumulative probabilities of total nitrogen via our models, upper lines are estimated values at 90% confident level, middle lines are median values; lower lines are estimated values at 10% confident level. Discrete dots are nutrient runoff values collected from publications. These values are used to validate model inputs and reported below.

Model validation is an important part to test and verify the model results. We used multiple sources to verify model results including more than 20 independent studies conducted in Corn Belt states during last 10 years.

Table 7. Independent observations used to validate the model

Items		Resources used for model validation
N runoff from different farming practices	Synthetic fertilizer	(Andraski and Bundy 2003; Tabbara 2003; Franklin, Cabrera et al. 2005; Landis, Miller et al. 2007; Vadas, Good et al. 2009)
	Manure	(Bundy, Andraski et al. 2001; Kleinman, Sharpley et al. 2002; Andraski and Bundy 2003; Tabbara 2003; Gessel, Hansen et al. 2004; Franklin, Cabrera et al. 2005; David D. Tarkalson 2006; Vadas, Good et al. 2009)
	Conventional tillage	(Angle, Mc Clung et al. 1984; Mcdowell 1984; Bundy, Andraski et al. 2001; Andraski and Bundy 2003; Klatt, Mallarino et al. 2003; Landis, Miller et al. 2007)
	No tillage	(Angle, Mc Clung et al. 1984; Mcdowell 1984; Bundy, Andraski et al. 2001; Andraski and Bundy 2003; Klatt, Mallarino et al. 2003; Landis, Miller et al. 2007)

C.5 CARBON STORAGE OF NO TILLAGE PRACTICES AND BUFFER STRIPS

Basing on published articles, we calculated carbon sequestration of tillage practices and converting farmlands to buffer strips. Organic carbon sequestration of no tillage is based on previously research. Captured carbon of buffer strips is determined by IPCC equations.

Table 8. Datasources for carbon storage capability of no tillage and buffer strip

Items		Resources used for model validation
Carbon storage capability	No tillage	(Odell, Melsted et al. 1984; Kladivko 1986; Mielke 1986; Kitur, Olson et al. 1993; Robinson 1996; Hussain, Olson et al. 1998; Hussain, Olson et al. 1999; Lal, Follett et al. 1999; Yang and Wander 1999)
	Buffer strip	(IPCC 1996)

C.6 NUTRIENT REMOVAL RATES OF BUFFER STRIPS

C.6.1 Model description and datasources

Depending on buffer strips' areas, energy consumption for installing and maintain grass/wooded buffer strips is calculated. Farmlands are assumed to be rectangular. Buffer strips are assumed to be installed at the ending edge of farmlands.

$A_{bufferstrips} = (A_{farmlands})^{1/2} * 30$

Where, $A_{farmlands}$ is the area of the farmland; $A_{bufferstrips}$ is the area of the buffer strip.

Nutrient remediation capabilities of buffer strips are evaluated according to published data. Vegetation types, buffer strips' width, soil condition etc. influence the removal rates. This study collects published results from previous research and statistically describes the situations of Corn Belt.

Nutrient removal capabilities are evaluated by linear models:

$L_{buffers-runoff,N} = L_{farm-runoff,N} * (1-R_N)$

$L_{buffers-runoff,P} = L_{farm-runoff,P} * (1-R_P)$

Where, $L_{farm-runoff}$ is nutrient runoff from farmland lands; $L_{buffers-runoff}$ is nutrient runoff treated by buffer strips; R_N is nutrient removal rate; R_P is nutrient removal rate.

Table 9. Description of parameters and their datasources

#	Variable	Description	Datasources
1	R_N	R_N is nutrient removal rate	Peer reviewed publication (Peterjohn and Correll 1984; Lee, Isenhart et al. 1998; Schmitt, Dosskey et al. 1999)
2	R_P	R_P is nutrient removal rate	Peer reviewed publication (Peterjohn and Correll 1984; Schmitt, Dosskey et al. 1999)

C.6.2 Results and model validation

First paragraph. The figure below is inserted so that there is an item in the sample List of Figures.

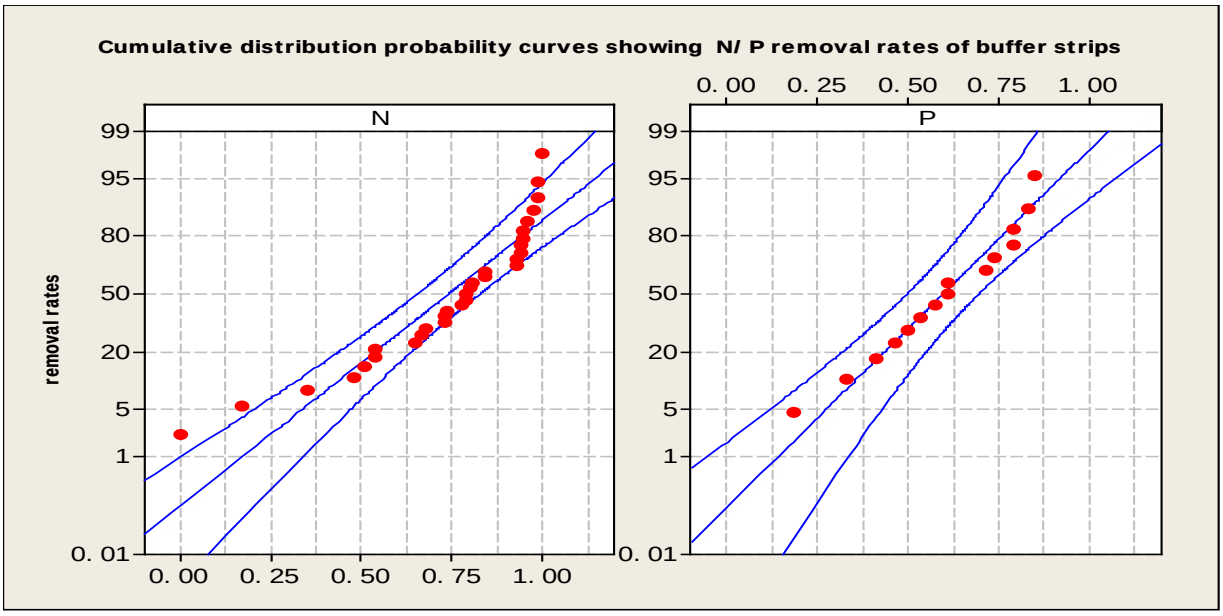

Figure 4. Probability distribution of nutrient removal rates

In Figure 4, the three lines are cumulative probabilities of N/P removal rate via our models, upper lines are estimated values at 90% confident level, middle lines are median values; lower lines are estimated values at 10% confident level. Discrete dots are N/P removal rate collected from publications. These values are used to validate model inputs and reported below.

Model validation is an important part to test and verify the model results. We used multiple sources to verify model results including more than 20 independent studies conducted in Corn Belt states during last 10 years. These studies are listed the table below. These are peer reviewed articles researching the nutrient removal capabilities of buffer strips. The previous studies showed that the buffer strips had significant capabilities to sequester N/P. Our model results are consistent with previous studies.

Table 10. Independent observations used to validate nutrient removal rates

Items		Datasources
Nutrient removal by buffer strips	R_N	Peer reviewed publication (Lewis L and David A 1993; Barling and Moore 1994; Hubbard and Lowrance 1994; B.-M. Vought, Pinay et al. 1995; Schultz, Collettil et al. 1995; John, Stanley et al. 2000; Ducros and Joyce 2003; Jon and Karl 2003; Turner and Rabalais 2003; Lee, Smyth et al. 2004; Schultz, Isenhart et al. 2004)
	R_P	Peer reviewed publication (Lewis L and David A 1993; Barling and Moore 1994; Hubbard and Lowrance 1994; B.-M. Vought, Pinay et al. 1995; Schultz, Collettil et al. 1995; John, Stanley et al. 2000; Ducros and Joyce 2003; Jon and Karl 2003; Turner and Rabalais 2003; Lee, Smyth et al. 2004; Schultz, Isenhart et al. 2004)

C.7 MODELS USED TO CREAT LIFE CYCLE INVENTORY

Table 6 shows the median value and upper/lower bound for each parameter for Chapter 6. The data can be used to construct life cycle inventory in future research.

Table 11. Values of all parameters

Categories		variable	units	range	median value
Basic information	Corn yield rate	L	kg/m^2	0.69-1.13	0.97
	Nitrogen fertilizer application rate	R_N	kg-N $/km^2$	1.14-1.60	1.38
	Phosphorus fertilizer application rate	R_P	kg-P$/km^2$	0.46-0.70	0.54
Fertilizers practices	Energy consumption for synthetic fertilizer	Calculated result from GREET	MJ/kg Corn	1.4-2.1	1.5
	Energy consumption for manure as coproduct	Collected data	MJ/kg Corn	0.05-0.3	0.2
	CO_2 emissions for synthetic fertilizer	Calculated result from GREET	g/kg Corn	82-114	87
	CO_2 emissions for manure as coproduct	Collected data	g/kg Corn	69-177	87
	CH_4 emissions for synthetic fertilizer	Calculated result from GREET	g/kg Corn	1.7-2.6	1.9
	CH_4 emissions for manure as coproduct	Collected data	g/kg Corn	6.7-94.1	68.3
	N_2O emissions for synthetic fertilizer	Calculated result from GREET	g/kg Corn	3.1-2117	177
	N_2O emissions for manure as coproduct	Collected data	g/kg Corn	320-2019	335
	Nitrogen nutrient leaching for synthetic fertilizer	L^{runoff}_N, nutrient linear model	g/kg Corn	4.0-8.0	5.0
	Phosphorus nutrient leaching for synthetic fertilizer	L^{runoff}_P, nutrient linear model	g/kg Corn	0.018-1.0	0.15
	Nitrogen nutrient leaching for manure	L^{runoff}_N, nutrient linear model	g/kg Corn	0.05-10	3.81
	Phosphorus nutrient leaching for manure	L^{runoff}_P, nutrient linear model	g/kg Corn	0.05-0.9	0.18

Table 11. (Continued)

Tillage practices	Energy consumption for conventional tillage	Calculated result from GREET	kJ/kg Corn	99-126	108
	Air emissions for conventional tillage	Calculated result from GREET	g/kg Corn	5.3-6.7	5.8
	Energy consumption for no tillage	Calculated result from GREET	kJ/kg Corn	76-97	83
	Air emissions for no tillage	Calculated result from GREET	g/kg Corn	4.3-5.4	4.7
	Carbon sequestration	Collected data	g/kg Corn	13.1-16.7	14.3
	Nitrogen nutrient leaching for conventional tillage	L^{runoff}_{N}, nutrient linear model	g/kg Corn	0.09-15	4.8
	Phosphorus nutrient leaching for conventional tillage	L^{runoff}_{P}, nutrient linear model	g/kg Corn	0.05-0.4	0.2
	Nitrogen nutrient leaching for no tillage	L^{runoff}_{N}, nutrient linear model	g/kg Corn	0.01-15	2.4
	Phosphorus nutrient leaching for no tillage	L^{runoff}_{P}, nutrient linear model	g/kg Corn	0.001-0.1	0.007
Buffer strips	Energy consumption	Calculated result from GREET	MJ/ kg Corn	0.009-0.012	0.01
	Carbon sequestration	Collected data	g/kg Corn	13.5-17.1	14.7
	Nitrogen removal rate	R_N, buffer strips linear model	N/A	0.41-0.85	0.75
	Phosphorus removal rate	R_P, buffer strips linear model	N/A	0.32-0.81	0.67

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