A century of land use and water quality in watersheds of the continental U.S.

Whitney P. Broussard III

Louisiana State University and Agricultural and Mechanical College

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A CENTURY OF LAND USE AND WATER QUALITY IN WATERSHEDS OF THE CONTINENTAL U.S.

A Dissertation

Submitted to the Graduate Faculty of the Louisiana State University and Agricultural and Mechanical College in partial fulfillment of the requirements for the degree of Doctor of Philosophy

in

The Department of Oceanography and Coastal Sciences

by

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B.A., Naropa University, 2000
B.S., University of Louisiana at Lafayette, 2003

August 2008
For my mother, Ione
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Abstract

Human endeavors, particularly the agricultural and industrial activities of the last half century, now produce more biologically available nitrogen (N) than all other natural sources combined. The increased N availability can have consequences for the health of aquatic biota on the local, regional, and global scales. One manifestation of this problem is the formation of coastal hypoxic zones where terrestrial N loading creates eutrophic conditions in coastal waters.

This dissertation examines a century of changes in land use and water quality to quantify the relationships between agricultural land use practices and riverine N yields in the Mississippi River Basin and 56 other watersheds across the continental U.S. A novel and spatially-explicit geographic information system database encompassing the continental U.S. from 1840 to present was developed to test the hypothesis that land use affects water quality. The database was compiled from semi-monthly water quality monitoring records from the U.S. Geological Survey, and from county-level Census of Agriculture data from the U.S. Bureau of the Census and Department of Agriculture.

The results of this research indicate that intensive agricultural land use is statistically associated with riverine N yields at the beginning and end of the 20th Century. These findings imply that agriculture was already affecting N export at the turn of the century, but that intensive contemporary management practices have significantly increased the N export per hectare of cropland in the latter half of the century. The baseline conditions for N export from minimally impacted watersheds, however, did not change over the course of the last century indicating that rehabilitation of surface water quality is possible. Additionally, decreasing in-stream N yields are quantitatively associated with increasing landscape diversity, increasing perennial farmland cover, and decreasing commercial fertilizer applications in agricultural watersheds across the
continental U.S. The results also suggest that government farm payments can be re-distributed to implement land use practices that effectively reduce in-stream N concentrations at the sub-basin scale, while potentially improving groundwater quality, soil quality, and biodiversity. These improvements will require changing land use practices on large spatial scales, however, in ecological units ranging from upland watersheds to coastal bays.
Chapter One
Introduction

The heritage of the past
is the seed that brings forth
the harvest of the future.

Inscribed on the National Archives Building, Washington, DC

Humans first domesticated their surrounding landscapes in the form of agriculture over ten thousand years ago. We have continuously selected species and systems in this endeavor to be efficient and productive. The great challenge of the 21st Century will be to re-evaluate a historical focus on productivity, and to willfully choose a path that incorporates conservation and long-term sustainability into our biologically ingrained mode of survival (Jackson 1980).

We should constructively address this challenge because the effects of this labor have been pervasive and far-reaching, have fundamentally changed the course of human history, have important consequences now, and will continue to have consequences for generations to come. Humans have altered the world ecosystem’s structure and function more in the last 50 years than in the combined history of humanity. Land use changes, particularly the conversion of natural ecosystems to cropland and increasingly employing intensive management practices, have been the most important direct driver of change in terrestrial ecosystems over the last 50 years (MEA 2005). Human endeavors now produce more biologically usable nitrogen than all other natural sources combined (MEA 2005). Agricultural cultivation now covers roughly one quarter of the earth’s land surface (MEA 2005). More land was converted to cropland between 1950 and 1980
than in the 150 years between 1700 and 1850, while at the same time quadrupling the amount of water impounded behind dams, which is 3 to 6 times more water than exists in natural rivers (MEA 2005). Most of this dammed water is used for agriculture.

This dissertation examines one set of agricultural impacts: the relationships between agricultural land use and the concentration of riverine nitrate in surface waters across the continental U.S. Here, history is observation, and historical observations of land use and water quality are the foundation of my research. The questions examined were: 1) How have land use and water quality parameters changed over the course of the 20th Century, i.e. what is the historical context of our current situation? 2) What are the non-point sources of nutrient enrichment in our nation’s rivers and streams? 3) If agricultural land uses are driving this non-point source pollution, which agricultural and economic practices can be quantitatively associated with nutrient enrichment on the watershed scale? 4) Which agricultural and economic practices can feasibly reduce nutrient export from agricultural watersheds? 5) How influential is federal farm policy on land use and, indirectly, water quality? and, 6) Is there evidence that farm policy can be an agent of positive change to develop sustainable agricultural systems and improve water quality?

A detailed description of the materials and methods used in this dissertation are presented in Chapter Two. It contains the description of a comprehensive, spatially-explicit geographic information system (GIS) database containing digitized records from the U.S. Census of Agriculture, in addition to all publicly available nitrate-nitrogen measurements from streams and rivers monitored at the beginning and end of the 20th Century. Recent developments in GIS technology have made this novel method possible to integrate large temporal and spatial scale information at the watershed level. This database was the foundation of my tests and analyses, and I plan to make it publicly available for further research and policy applications.
Chapter Three begins the analytical component of my research with a historical analysis of land use and water quality relationships on the continental scale through the 20\textsuperscript{th} Century. I report changes in U.S. agricultural practices and cropland distribution, and changes in nutrient concentrations from 122 rivers in the beginning and end of the last century. Next, I quantify the relationships between agricultural land use practices and riverine nutrient concentrations in 63 watersheds across the continental U.S. during the last century and discuss implications for the changing relationships. Finally, I quantify the relationship between nutrient concentrations and cropland diversity, and the role cropland diversity may play in reducing surface water nutrient enrichment on the watershed level. This chapter is currently under review for publication in *Frontiers in Ecology and the Environment*.

National agricultural policy played an important role in the development of agriculture in the 20\textsuperscript{th} Century. A study of changing agricultural practices during this time period begs for an in-depth inquiry into the relationship between farm policy, land use, and water quality. Chapter Four reviews the changes in federal farm policy from 1933 to present, and quantifies the relationships between government farm payments paid per area of farmland and cropping patterns, management practices, farm size and income, as well as nutrient export from agricultural watersheds.

The impact of perennial farmlands and annual croplands on riverine nutrient concentrations is examined in Chapter Five. Here, I examine the effects of agricultural cropland on river discharge levels in the 20\textsuperscript{th} Century and quantify the effects of annual and perennial plant cover on nutrient export from agricultural watersheds. A particular aspect of the chapter tests the hypothesis that perennial land cover on farmlands can effectively reduce nutrient loading to surface waters. Pieces of this chapter were published in the journal *Nature*, are...
currently under review for publication in the journal *Science*, and are in preparation for submission as an invited editorial in the *Journal of Soil and Water Conservation*.

It is appropriate to ask fundamentally challenging questions regarding the purpose and ethical obligations of one’s scientific research conducted to fulfill the requirements leading to a professional degree titled “Doctor of Philosophy”. Chapter Six presents a proposed ethical foundation for scientific research and the field of environmental science. I defend the thesis that the principles of Deep Ecology, an eco-centric ethical tradition, offer an appropriate philosophical foundation for the environmental sciences. This approach, I argue, can balance the need for scientific rigor with the desire for personal connection with the natural world, and can be the foundation for environmentally responsible actions.

Finally, I present a summary of the conclusions, implications, and recommendations of my research in Chapter Seven. It is my hope that the database and ecological knowledge gained from this research will be useful resources to improve land management and agricultural policy in the near future.

**LITERATURE CITED**


Chapter Two
Creation and Analysis of the Water Quality, Land Use, and GIS Databases

INTRODUCTION

This dissertation is about the relationships between land use and in-stream nitrogen (N) concentration at the beginning and end of the 20th Century across the continental United States. I developed a method to spatially analyze the relationship between riverine nitrate-N (NN) concentrations and agricultural land uses to test this relationship. A water quality database, a land use database, and a GIS (geographic information system) database were designed and integrated for use in the analyses of the remaining chapters. This chapter describes the methods and techniques used to develop the databases.

The temporal domain of this dissertation study is from 1905 to 2002 and the spatial extent encompasses the continental United States, Alaska, and parts of Canada. It examined 56 watersheds ranging in size from the Cache River Basin in Illinois (976 km²) to the main stem Mississippi River Basin (2,937,502 km²). The variability of nitrate-N (NN) concentrations by watershed is the focus of this research for several reasons. First, NN is a major nutrient source for many eutrophic coastal ecosystems, including ‘Dead Zones’ (Diaz 2001, Rabalais et al. 2002, Howarth et al. 2006). Second, the nutrient concentration is what the organisms sense physiologically, not the nutrient load. Third, riverine discharge data are not available for the monitored watersheds in the early 1900s and the calculation of flux estimates are therefore not possible for this time period. Forth, because riverine discharge, and therefore nutrient flux, is affected by climatic variation, I chose to focus the analyses on nutrient concentration to test the effects of land use management on riverine NN concentrations.
**HISTORICAL WATER QUALITY DATABASE**

Data on the NN concentration in rivers in the early 1900s are from *The Composition of the River and Lake Waters of the United States* (Clarke 1924). This ‘Clarke’ dataset has 192 tables of semi-monthly water sample analyses from 190 monitoring stations on 156 rivers and lakes across the continental United States, with 1 station in Alaska, in the water years 1905-1921. I hand-entered the water quality data, with hired assistance, from 192 tables in the original text into Excel spreadsheets (Microsoft Corporation 2007). See Table 2-1 for a list of monitoring stations, Figure 2-1 for a map of the monitoring stations and their watersheds, and Figure 2-2 for an example of the original datatables. After digitizing the bi-monthly values from a datatable in Clarke, I calculated an annual mean and then checked the calculated annual mean against the published mean value in the original document for quality control. There were several discrepancies between the annual mean values published in Clarke and those calculated in the spreadsheet from the data entries. These discrepancies were resolved by reviewing the individual data entries and recalculating a new annual value. If a discrepancy remained, I retained the calculated value in the spreadsheet. I assumed, in other words, that there was a hand calculation error when the original tables were generated. Most of the data were collected in water years 1906-1907. Most of the west-coast rivers and streams were monitored in water years 1911-1912.

The Clarke water quality dataset was the culmination of a previous project, *The Quality of Surface Waters in the United States, Part 1: Analyses of Surface Waters East of the One Hundredth Meridian* (Dole 1906), that ended prematurely with the death of the lead author, R. B. Dole. In this publication, Dole explains that they used the phenolsulphonic acid method to determine nitrates as described in *American Health Papers and Reports* at the time. He also cautioned that, “Practical considerations make it impossible to perform the test less than 10 days
Table 2-1. A list of the study watersheds from Clarke (1924).

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Table 2-1. A list of the study watersheds from Clarke (1924).
Figure 2-2. A map of the watersheds and monitoring stations reported in Clarke (1924).
**ST. LAWRENCE BASIN.**

**Analyses of water from Grand River at Grand Rapids, Mich.**

*Parts per million.*

<table>
<thead>
<tr>
<th>Date (1906-7)</th>
<th>Turbidity</th>
<th>Total iron (Fe)</th>
<th>Silica (SiO₂)</th>
<th>Iron (Fe)</th>
<th>Calcium (Ca)</th>
<th>Magnesium (Mg)</th>
<th>Sodium and potassium (Na₂K)</th>
<th>Carbonate radicles (CO₃)</th>
<th>Residual radicals (HCO₃)</th>
<th>Sulphate radicals (SO₄)</th>
<th>Nitrate radicals (NO₃)</th>
<th>Chlorides (Cl)</th>
<th>Total dissolved solids</th>
<th>Mean B.C. height (in.)</th>
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*Analyses Oct. 1, 1896, to Nov. 26, 1895, by H. B. Dale; Jan. 2 to Mar. 29, 1897, by H. B. Dale and M. G. Roberts; Mar. 30 to July 4, 1907, by Chas Palmer and W. D. Collins.*

Analyses of water from Kalamazoo River near Kalamazoo, Mich.*

*Parts per million.*

<table>
<thead>
<tr>
<th>Date (1906-7)</th>
<th>Turbidity</th>
<th>Total iron (Fe)</th>
<th>Silica (SiO₂)</th>
<th>Iron (Fe)</th>
<th>Calcium (Ca)</th>
<th>Magnesium (Mg)</th>
<th>Sodium and potassium (Na₂K)</th>
<th>Carbonate radicles (CO₃)</th>
<th>Residual radicals (HCO₃)</th>
<th>Sulphate radicals (SO₄)</th>
<th>Nitrate radicals (NO₃)</th>
<th>Chlorides (Cl)</th>
<th>Total dissolved solids</th>
<th>Mean B.C. height (in.)</th>
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Analyses Sept. 19 to Dec. 7, 1906, by H. B. Dale; Aug. 8, 1906, to Mar. 27, 1907, by H. B. Dale and M. G. Roberts; Mar. 31 to July 2, 1907, by Chas Palmer and M. G. Roberts; July 3 to Sept. 24, 1907, by H. B. Dale, Chas Palmer, and W. D. Collins.*

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Figure 2-2. An example of scanned water quality tables from The Composition of the River and Lake Waters of the United States (Clarke 1924).
after some of the samples had been collected and the estimate was made in a great many samples after a period of 3 to 8 weeks, or even more, had elapsed.” He warned that the reported nitrate values could be problematic due to this time delay from field sampling to laboratory processing.

I compared the Clarke NN values with those reported in Report of Chemical Survey of the Waters of Illinois: Report for the Years 1897-1902 (Palmer 1902) to test whether the delayed analysis compromised my use of the Clarke data. This project monitored in-stream nitrogen conditions in a dozen or more Illinois rivers and streams from 1897 to 1902, matching several Clarke stations in time and space. Palmer used a sodium hydroxide solution instead of the phenolsulphonic acid reagent used in Clarke and Dole. Palmer, a professor in the Department of Chemistry at the University of Illinois, specified that the samples were analyzed in a timely manner: “The sample of water should be collected immediately before shipping by express, so that the shortest possible time shall intervene between the collection of the sample and its examination” (Palmer 1902). These efforts resulted in a 1-3 day delay between field collection and laboratory analysis. For these reasons, I consider the Palmer dataset a dependable dataset to compare with the Clarke dataset. I performed a simple two-tailed t-test for differences between the bi-weekly values of nitrate concentrations at each of these seven stations. I found significant differences in nitrate concentrations at three stations and no significant difference at three stations (alpha = 0.01). I then performed a paired t-test for differences among the grand mean nitrate concentrations by station and found no significant difference (p = 0.70, n = 7) between the two datasets. I concluded that the nitrate values reported in Clarke are reasonably accurate for use in analyses on large spatial and temporal scales.

The Clarke data were then exported to SAS statistical software (SAS Institute, Inc. 2003) where the annual mean, seasonal mean, and five-year average values for each station were calculated. The final dataset was designed to be available for future distribution and analysis via
Table 2-2. A comparison of the annual mean nitrate-N concentrations for 7 Illinois rivers for two studies at the turn of the 20th Century (Clarke 1924, Palmer 1909).

<table>
<thead>
<tr>
<th>Station</th>
<th>NO$_3$ (mg N L$^{-1}$)</th>
<th>Palmer (1896-1902)</th>
<th>Clarke (1906-1907)</th>
<th>p value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mississippi River near Quincy, IL</td>
<td>0.349</td>
<td>0.50</td>
<td>0.006</td>
<td></td>
</tr>
<tr>
<td>Kankakee River near Kankakee, IL</td>
<td>1.818</td>
<td>0.93</td>
<td>0.002</td>
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<tr>
<td>Fox River near Ottawa, IL</td>
<td>0.42</td>
<td>1.11</td>
<td>&lt; 0.001</td>
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<tr>
<td>Illinois River near La Salle, IL</td>
<td>1.85</td>
<td>1.49</td>
<td>0.139</td>
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<tr>
<td>Illinois River near Peoria, IL</td>
<td>1.40</td>
<td>1.76</td>
<td>0.533</td>
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<tr>
<td>Illinois River near Kampsville, IL</td>
<td>1.20</td>
<td>0.97</td>
<td>0.273</td>
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<tr>
<td>Vermillion River near Streator, IL</td>
<td>3.00</td>
<td>2.71</td>
<td>N.A.</td>
<td></td>
</tr>
</tbody>
</table>

Paired t-test for annual means p = 0.695

an internet website or CD-ROM. This product will include semi-monthly records of turbidity, silica, iron, calcium, magnesium, sodium + potassium, carbonate, bicarbonate, sulphate, nitrate, chloride, total dissolved solids, and mean gauge height for a given water year between 1905 and 1921. It is my wish that this historical water quality data will soon be available to others as a reference for contemporary data and the focus of further research.

**CONTEMPORARY WATER QUALITY DATABASE**

The next phase of the project compared the historical NN concentrations reported in Clarke to contemporary NN concentrations reported for nearby United States Geological Survey (USGS) water quality sampling stations. To accomplish this goal, I accessed the USGS National Water Information System (USGS 2007) in October and December 2007, and identified the current USGS monitoring stations that analyzed the same river reaches as the Clarke study stations. If there was no station that met these criteria, then a station was chosen that was in the same 8 digit hydrologic unit code (HUC). If no current station was identified at this point, the historical station was dropped from the analysis.
The coordinates for these modern stations were then used to delineate the watersheds using a Geographic Information System (see below for methods on delineating watersheds). Of the original 190 stations in the Clarke dataset, there were 122 modern stations that measured the NN concentration between 1940 and 2002. Sixty-three stations had water quality data between 1993 and 1997. The years of available data varied widely and made consistent analysis difficult. Monitoring data will be crucial for detailed modeling applications in the future.

The annual mean NN concentrations were calculated using SAS. A 5 year grand mean was then calculated from the annual means to match the 5 year intervals between the Census of Agriculture data discussed next. If the NN data were unavailable for a given station, then the NN concentration was approximated from nitrite-nitrate values where NN was estimated to be 99.85% of nitrite-nitrate based on grand averages from the dataset. A simple one-tailed paired t-test was then performed on all available data comparing the NN concentrations for 1906-1912 and 1993-1997 for each watershed station to test if the contemporary NN concentrations were higher than the historical concentrations. All available yearly means were exported to the Century of Change GIS database discussed below.

**Census of Agriculture Statistical Database**

Agricultural land management is the principle land use examined in this study. Agricultural land covered over a quarter of the earth’s land surface in 2000 and will be one of the most important drivers of global changes in biodiversity and coastal eutrophication in the 21st Century (MEA 2005). I developed a statistical database using county-level Census of Agriculture data to assess changes in agricultural land use over the last 100 years. Urban and forestry uses were not included because of logistical constraints, though the importance of these land use practices regarding nutrient export should not be underestimated.
The Census of Agriculture is the primary source of agricultural land use data used in this study. I obtained datafiles of digitized Census of Agriculture data from 1850 to 1910 via public document ICPSR 2896 (Haines and ICPSR 2004). I also obtained digitized datafiles of the Census of the United States on Population data related to agriculture from 1920 – 1940 from public document ICPSR 2896 (Haines and ICPSR 2004). The Census of Agriculture data from 1947-2002 were organized into datafiles by Michael Haines and colleagues at Colgate University using published data records (USDA 1979, 2005; U.S. Bureau of the Census 1987, 1999). See Appendix I for metadata related to this data-set.

Some important and basic things were done to the data entered by Haines. First, all the county-level data were checked by “adding up.” That is, the sum of the data for counties should total up to the sum for states, and the sum for states should total up to the sum for the United States for that time period, whenever counts were involved. The format of the raw datafiles was STATA version 9.0 (StataCorp LP 2005), so I used StatTransfer software (Circle Systems 2008) to convert STATA datafiles to SAS datafiles. I then consolidated tables from the different sources by year (one table per year), and re-assigned some variables to be compatible with GIS .dbf file format label limitations. Some tables were divided into several tables per year to be compatible with GIS .dbf size limitations (i.e., 2 to 4 tables per year for the later censuses). The finalized tables were exported as .dbf formatted tables for use in the GIS database. In addition, I created a single .pdf document of all the variables included in the individual censuses for easy reference when using GIS. See Figure 2-3 for an example of the Census of Agriculture variables document. In all, the Census of Agriculture statistical database covers 17 censuses from 1840 to 2002 using 43 tables with 89 to 966 variables (columns) and 3,078 counties (rows) each.
Figures

14

CENTURY OF CHANGE GIS DATABASE

After compiling and developing statistical databases for water quality and land use individually, I developed a method for spatially analyzing the relationship between the riverine

Figure 2-3. The first page from the list of variables for the Census of Agriculture database.

NN concentrations and agricultural land uses in a GIS environment using ESRI ArcGIS software (ESRI 2005). Here, I spatially related the statistical information to a particular representative area. For example, I associated the Census of Agriculture data for an individual county to the
area of land covered by that respective county in the year of that census. I also associated the water quality data of a particular station to the area of land covered by the respective watershed that drains to that reach of the stream or river.

I created a spatially explicit Census of Agriculture GIS project using county boundary shapefiles from “Historical United States County Boundary Shapefiles” (HUSCO), which is a Louisiana State University Geosciences publication (Earl, 1999), to spatially analyze the agricultural land use data. A shapefile is a set of four or five digital files that are used collectively to spatially represent points, lines, and polygons in a GIS environment. Here, the county boundaries are accurately represented for each decade from 1850 to 1999. Because county boundaries change with time, it was imperative that I use the accurate boundaries for a given year when associating the statistical land use data to a specific spatial extent. With the counties accurately projected using an Albers Equal Area Projection, I used ESRI ArcGIS Desktop software to create ‘layers’ that join (or link) the Census of Agriculture database tables with HUSCO county boundary shapefiles by year (i.e., one layer per year). The statistical data at this point can be displayed as county maps and area calculations can be performed for further spatial analysis. See Figure 2-4, Figure 2-5, and Appendix II for examples of the mapping capabilities of the GIS database.

To spatially analyze the water quality data, I first delineated the drainage basins (used interchangeably with the term ‘watersheds’) for every monitoring station in the database using the National Hydrography Database Plus (NHDPlus) (USEPA and USGS 2005), a valuable product delivered by a mutli-agency project to model water quality parameters. By using high resolution Digital Elevation Models (DEM) from the NHDPlus and ArcGIS, I delineated watersheds for 189 stations (two tables were of the same location, one table is in Alaska and outside the spatial extent of the NHDPlus data set, and one table is in the Pearl River Basin and
Figure 2-4. An example of the capabilities of the GIS database: the ratio of cattle populations to human populations by county.
Figure 2-5. An example of the capabilities of the GIS database: an estimation of farm machinery and equipment, normalized by county.
could not be delineated due to technical difficulties with the NHDPlus data). Each watershed was saved as an individual shapefile and then merged to create a master watershed shapefile where each watershed is a record or row. The water quality datatables were then joined to the master watershed shapefile for mapping and analysis. I assumed that the watershed boundaries did not significantly change during the study period because of the lack of DEMs at the beginning of the century.

**Spatial Analysis Preparation**

I needed two components to analyze land use and water quality relationships by watershed: the water quality parameters associated with a watershed, which was accomplished in the previous step, and land use practices summed by watershed. I used techniques similar to Boyer *et al.* (2002) to sum the county-level agriculture data by watershed. I kept the native units and used acres as the base unit for area in this study because the Census of Agriculture, the primary source of land use data, uses acres as its base unit. In addition, because the Census of Agriculture only reports information on a county-by-county basis, the maximum resolution of the analysis is the county level. I had to assume homogenous land cover at the county level in order to split the agriculture data values between two watersheds if a watershed boundary divides a county, as it usually does. I argue that the analysis is robust at larger regional scales and is an improvement over state-level analyses. See Figures 2-6 and 2-7 for a pictorial demonstration of the data development process.

I first used the Area Calculation function in ArcGIS to calculate the area of each watershed and every county for each year in question. I used North American Datum 1983 for the project datum and Albers Equal Area Projection for the project projection. Then, using the intersect function in ArcGIS, the master watershed shapefile and county shapefiles were joined for each year and the relative proportion of each county area was calculated within each
Figure 2-6. An example of merging county boundaries and watershed boundaries in a GIS environment. A. A county area base map. B. A watershed area base map. C. The intersection of the county and watershed maps used for area calibrations and summation of county data by watershed.

This value was termed the Conversion Factor (Conv_Fact) and was multiplied by the county-level data to determine the relative coverage of a variable in the watershed of interest.

For example, if 40% of a county’s area lies inside the watershed, then Conv_Fact = 0.4. If there
Figure 2-7. A second example of merging county and watershed boundaries with associated data in a GIS environment. A. A county-level base map. B. The county-level data displayed visually as the fraction of county planted in corn. C. A watershed base map with monitoring states as blue triangles. D. The watershed-level data displayed as the concentration of NN exiting the watershed. E. The watershed data accurately re-projected over the land use data as they would be used in a spatial analysis for correlation.

If there are 100 acres of cropland in the entire county, then the number of acres of cropland that lie within the watershed for that county is $100 \times 0.4 = 40$.

Tables were exported to SAS from GIS that included this new variable, Conv_Fact. The conversion factor was then multiplied by the published values for each county-watershed combination to generate a ‘converted value’ for that variable. The converted values were then summed by watershed to determine the total number of acres of a given variable within a given watershed. Finally, the fraction of the watershed in a given variable was calculated as the total number of acres of a given variable divided by the total number of acres in the watershed. The
following equation explains this conversion process from county level data to watershed level data:

\[ F_{LU} = \left( \sum_{i=1}^{n} L_i \times C_i \right) / W \]

where \( F_{LU} \) is the fraction of a watershed’s total area in a particular land use, \( n \) is the number of counties within the watershed, \( L \) is the reported total area (acres) of that land use practice in a specific county, \( C \) is the conversion factor or fraction of the county area that lies within the watershed, and \( W \) is the total watershed area (acres).

**Using the Data: Qualitative Visual Analysis**

The Century of Change GIS database was used to construct county-level maps of the various agricultural practices, e.g., individual crop acreage, particular management techniques, fertilizer usage, farm population and demographics, and many other variables for each digitized census from 1850 to present. See Figure 2-4, Figure 2-5, and Appendix II for examples of the many maps that can be generated with the dataset, which amount to over 27,000 total possibilities. These maps are excellent qualitative indicators of national and regional trends and changes in crop types, land cover, and other agricultural practices. They were indispensable in guiding the quantitative analyses that follow and for explaining the significance of the results to other researchers and the public.

**Using the Data: Quantitative Analysis**

Statistical analyses were performed using SAS Version 9.1 (SAS 2003) and GraphPad Prism 5.01 (GraphPad 2007) statistical software. I focused on the nationally dominant agricultural practices and, in particular, those categories that reported the number of acres of different crop types, such as corn, soybeans, wheat, etc., and different cropland types, such as
harvested cropland, pasture, woodlands, etc. A statistical analysis of the relationships between NN concentration and various land use patterns was performed. I used Pearson’s correlations, simple linear regressions and multiple linear regressions, as well as logarithmic and polynomial models to determine the best fit explanations using r-squared coefficients and simplicity of the model as criteria (alpha = 0.05). Once the best-fit regression models were identified, I performed an ANCOVA analysis to determine if the change in the intercept or slope from the 1906-1912 era was significantly different from that in 1993-1997. The results of these analyses are reported in the following chapters.

**LITERATURE CITED**


GraphPad Software. 2007. *GraphPad Prism Version 5.01*. San Diego, CA: GraphPad Software.


StataCorp LP. 2005. Stata Statistical Software: Release 9. College Station, TX: StataCorp LP.


Chapter Three  
A Century of Changing Land Use and Water Quality Relationships in the Continental U.S.

INTRODUCTION

The global production rate of biologically available nitrogen (N) has doubled in terrestrial ecosystems since 1960, due primarily to human activity that consumed nearly eight times more nitrogenous fertilizer in 2003 than in 1960 (MEA 2005). This human-derived N fixation now produces more biologically available N than all other natural sources combined (Galloway et al. 1995). Food supplies have increased substantially since the industrialization of N fertilizer to the extent that one third of humanity’s protein consumption depends on synthetic N fertilizer (Smil 1997). Excessive nutrient loading, however, has become an important driver of ecosystem change in terrestrial and aquatic ecosystems and is projected to increase in the 21st century (MEA 2005).

Previous studies have identified the roles agricultural landscapes play as sources of riverine N (Howarth et al. 1996, Goolsby et al. 1999, McIsaac et al. 2001, Boyer et al. 2002, Donner 2003, Turner and Rabalais 2003) and how N loading contributes to the formation of oxygen-depleted bottom waters in coastal systems (Turner et al. 2006; Rabalais et al. 1999, 2007). There are, however, uncertainties about what constitutes the baseline values for restoration of riverine water quality (Smith et al. 2003).

This study examines the relationships between various land use practices and riverine N at the beginning and end of the 20th Century in the continental United States. I used 56 watersheds ranging in size from the Cache River Basin (976 km²) to the Mississippi River Basin (2,937,502 km²) to test the hypothesis that there is a statistically significant, quantifiable
association between agricultural land use practices and riverine nitrate-N (NN) concentration. The total N flux from the Mississippi River Basin in 1980-1999 increased nearly 3 fold compared with 1955-1970, due almost entirely to increases in nitrate which is the most abundant form of N (62%) at the mouth of the river (Goolsby and Battaglin 2001). I focused on the variability of NN concentrations by watershed because NN is the most abundant dissolved N ion in river water and historical data on NN concentrations were available for the first part of the 20th Century.

The results show that there is a direct relationship between NN concentrations in surface waters and the extent of agricultural cropland in a watershed. The relationship between increasing NN concentration or decreasing cropland diversity with the intensification of corn agriculture is particularly strong. I argue that it is possible to reduce riverine NN concentrations by focusing policy initiatives on these land use practices.

METHODS

Historical Water Quality Data

I collected two basic types of data to test the hypothesis that land use and water quality are related in the beginning and end of the 20th Century: land use records and water quality records for each study watershed. Data on the NN concentration in rivers in the early 1900s are from Clarke (1924). Contemporary water quality data are from the USGS (U.S. Geological Survey) National Water Information System (USGS 2007). A five year grand mean was calculated from the annual means to match the five year intervals between the Census of Agriculture data discussed below. I identified current USGS monitoring stations located on the same river reaches as the Clarke study stations to compare the historical and contemporary water quality data. The coordinates for the modern stations were used to delineate the drainage basins (watersheds) using the National Hydrography Dataset Plus (USEPA and USGS 2005).
identified the independent watersheds that do not overlay the areal extent of another watershed to avoid pseudo-replication in the regression analysis and present the results from these watersheds.

**Historical Land Use Data**

I developed a GIS (Geographic Information System) database using county-level Census of Agriculture data to quantify changes in land use over the last 100 years. Census of Agriculture data from 1850 to 1910 are from public document ICPSR 2896 (Haines and ICPSR 2004). Census of Agriculture data from 1954 to 2002 were organized into datafiles by Michael Haines and colleagues at Colgate University using published data records (USDA 1979, 2005; U.S. Bureau of the Census 1987, 1999). All datatables were spatially referenced by county with regionally accurate county boundaries (Earl et al. 1999). I summed the county-level Census of Agriculture data by watershed using techniques similar to Boyer et al. (2002).

**Database Preparation and Area Calculations**

NN data in the early 1900’s are from The Composition of the River and Lake Waters of the United States (Clarke 1924). This ‘Clarke’ dataset has 192 Tables of semi-monthly water sample analyses from 190 monitoring stations on 156 rivers and lakes across the continental United States, with 1 station in Alaska, from 1905 to 1921. I digitized the tabular data by hand, calculated an annual mean from the hand-entered data, and compared it against the published value in the original document for quality control.

The Clarke water quality dataset was the culmination of a previous project, The Quality of Surface Waters in the United States, Part 1: Analyses of Surface Waters East of the One Hundredth Meridian (Dole 1909), that ended prematurely with the death of the lead author, R. B. Dole. Dole explained that they used the phenolsulphonic acid method to determine nitrate concentrations as prescribed in American Health Papers and Reports at the time. He cautioned that, “Practical considerations make it impossible to perform the test less than 10 days after some
of the samples had been collected.” He warned that the reported nitrate values could be problematic due to this time delay from field sampling to laboratory processing. I tested whether using data from a delayed sample analysis compromised the Clarke dataset by comparing the Clarke nitrate values with those reported in *Report of Chemical Survey of the Waters of Illinois: Report for the Years 1897-1902* (Palmer 1903). Palmer reported in-stream nitrate concentrations for 514 samples in Illinois rivers and streams from 1897 to 1902. The Palmer dataset had seven stations that matched the location of the Clarke stations in time and space. Palmer specified that the samples were analysed in a timely manner: “The sample of water should be collected immediately before shipping by express, so that the shortest possible time shall intervene between the collection of the sample and its examination” (Palmer 1903). These efforts resulted in a 1-3 day delay between field collection and laboratory analysis. For these reasons, I consider the Palmer data a dependable dataset to compare with the Clarke dataset. I performed a simple two-tailed t-test for differences between the semi-monthly values of nitrate concentrations at six of the seven stations. I found significant differences in nitrate concentrations at three stations and no significant difference at three stations (alpha = 0.01). I then performed a paired t-test for differences among the grand mean nitrate concentrations by station and found no significant difference (p = 0.70, n = 7) between the two datasets. I concluded that the nitrate values reported in Clarke are reasonably accurate and that use of the Clarke dataset is acceptable for analyses on large spatial and temporal scales. In addition, the Clarke dataset has been used by the U.S. Geological Survey (USGS) in official publications, implying their confidence in the dataset (Goolsby *et al.* 2001). I do not know of any other comparable dataset for water quality on the national scale for this time period.

I accessed the USGS National Water Information System, as described in the methods section, to identify current USGS monitoring stations located on the same river reaches as the
Clarke study. If there were no stations that met these criteria, then a station was chosen that was in the same 8 digit hydrologic unit code (HUC). A station was removed from the analysis if contemporary data could not be obtained. Of the original 157 watersheds in the Clarke dataset, there were 122 modern stations that measured NN between 1940 and 2002. Sixty-three stations had water quality data between 1993-1997. If nitrate data was unavailable, then the nitrate concentration was estimated from nitrate-nitrite values assuming that nitrate was 99.85% of nitrate-nitrite based on grand averages estimated from the contemporary USGS data. The water quality data were then spatially referenced by watershed as described in the methods section and compiled into a GIS (geographic information systems) database using ArcGIS 9.1 (ESRI 2008) for spatial analysis.

All available Census of Agriculture data (see methods section) were similarly compiled and included in the database. I spatially referenced the raw land use data files with regionally accurate county boundary shapefiles for each respective year using Historical United States County Boundary Shapefiles (Earl 1999). Because county boundaries change with time, it was imperative that I used accurate boundaries for a given year when associating the statistical land use data to a specific spatial extent.

I followed techniques similar to Boyer et al. (2002) to sum the county-level agriculture data by watershed for a single time period using an Albers Equal Area projection and the following equation:

\[
F_{LU} = \left(\sum_{i=1}^{n} L_i * C_i \right)/W
\]

where \(F_{LU}\) is the fraction of a watershed’s total area in a particular land use, \(n\) is the number of counties within a watershed, \(L\) is the reported total area (acres) of that land use practice in a
specific county, C is the fraction of the county area that lies within the watershed, and W is the total watershed area (acres).

The maximum resolution of the analysis is the county level because the Census of Agriculture only reports information at that level of detail. I assumed that there was homogenous land cover at the county level in order to split the agricultural data values between two watersheds if a watershed boundary divides a county, as it usually does. The technique represents an improvement over state-level analyses, and is a robust technique to analyse land use and water quality relationships on larger spatial and temporal scales.

I used SAS statistical software (SAS Institute Inc. 2003) for database management and Prism statistical software (GraphPad Software Inc. 2007) to develop regression models.

**RESULTS AND DISCUSSION**

**Water Quality Changes**

The grand mean NN concentration for all study watersheds increased threefold from 1906-1912 to 1993-1997 (0.60 +/- 0.07 to 1.79 +/- 0.24 mg N L\(^{-1}\), \(\mu +/- S.E., t = -5.90, p < 0.0001\)). The minimum and maximum values were 0.03 and 2.71 mg N L\(^{-1}\) in 1906-1912, and 0.06 and 8.54 mg N L\(^{-1}\) in 1993-1997, respectively. The NN concentrations increased more than tenfold in the Iowa, Minnesota, and Des Moines Rivers.

The spatial distribution of the NN variation among watersheds is similar in both time periods with the higher concentrations in the Midwest and the lower concentrations in the Northwest, southern Great Plains, and the Piedmont (Figure 3-1a and 3-1b). Only one watershed (out of 56) in 1906-1912 reported a NN concentration higher than 2 mg N L\(^{-1}\), while 18 watersheds reported NN concentrations higher than 2 mg N L\(^{-1}\) in 1993-1997. Thirty-three watersheds (out of 56) in 1906-1912 and 16 watersheds in 1993-1997 had NN concentrations
lower than 0.5 mg N L\(^{-1}\). Figure 3-1c is a frequency histogram of the NN distribution among watersheds for both time periods showing the increasing proportion of watersheds with higher NN concentrations at the end of the century. The three lowest NN averages from 1906-1912 were from the Penobscot River, ME, Columbia River, WA, and Wateree River, SC. The three lowest NN averages from 1993-1997 were from the John Day River, OR, Columbia River, WA, and Penobscot River, ME. The three highest NN averages from 1906-1912 were from the Vermillion River, IL, Miami River, OH, and the Illinois River, IL. The three highest NN averages from 1993-1997 were from the Vermillion River, IL, Minnesota River, MN, and Iowa River, IA. Note that two rivers (the Columbia and Red rivers) remained among the lowest concentrations for both time periods and that one river (the Vermillion River) remained among the highest concentrations for both time periods. This suggests that the attributes of the anthropogenic activities driving the NN concentrations at the beginning of the 20\(^{th}\) Century are similar to the present-day drivers.

The water quality data by watershed were separated into three geographical regions: the Mississippi River Basin (MRB), west of the MRB, and east of the MRB. Figure 3-1d is a five year running average time series of the available NN data, normalized to values reported in 1906, showing the increasing concentration and range of values toward the end of the century. The concurrent rise in fertilizer consumption in the U.S. (USDA 2008) (Figure 3-1d) is evidence of the relationship between intensive agricultural land use and water quality degradation. Others have found similar results indicating that the currently high NN values were not common until after World War II and the start of modern agriculture in the 1950s and 1960s (Goolsby and Battaglin 2001). A rise in NN concentration in the late 1940s is noteworthy (Figure 3-1d) and likely explained by the start of intensive mechanical tillage and drainage. A fall in NN
concentrations in watersheds within and east of the MRB during the 1990s, at a time when fertilizer use continued to increase, is interesting and deserves further investigation.

**Land Use Changes**

The 20th Century was marked by specialization and consolidation in American agriculture (Table 3-1 and Figure 3-2). Although the area of total farmland increased by 11.5% from 1900 to 2002, the total number of farms fell almost 63% while the number of farms with more than 405 ha (1,000 acres) increased 65%, and the average farm size doubled. Farms larger than 405 ha occupied nearly 67% of the total U.S. farmland in 2002. Corn for grain occupied nearly 23% of
the U.S. cropland in 1900 and 17% in 2002 (Table 3-1 and Figure 3-2). Importantly, I think, there was a spatial redistribution of corn agriculture. Whereas corn was grown throughout the midwestern and southeastern U.S. in the early 1900s, today these efforts are primarily concentrated in the Midwest (Figure 3-2).

This pattern is evident in the Piedmont region of the Southeast where, for example, Georgia and Alabama lost 56% and 57% of their cropland between 1900 and 2002, respectively. Corn plantings likewise dropped by 1,298,219 ha (92.2%) and 1,033,972 ha (93.1%) for these two states in the same period. In the heart of the Corn Belt for the same period, corn cultivation in Iowa and Illinois increased 1,125,314 ha (28.4%) and 497,153 ha (12.0%), respectively. The

Table 3-1. Summary statistics of agriculture in the United States at the beginning and end of the 20th Century. Column A: the absolute change of the variable from 1900 to 2002; Column B: the fractional change. Data are rounded to two significant digits. Columns C and D reflect calculations of raw data. 1 hectare = 2.471 acres, M ha = Million hectares, ND = No Data, N/A = Not Applicable.

<table>
<thead>
<tr>
<th>Variable (unit)</th>
<th>A 1900</th>
<th>B 2002</th>
<th>C Absolute change [Column B – A]</th>
<th>D Relative change [Column C / A]</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Number of farms (farms)</td>
<td>5,700,000</td>
<td>2,100,000</td>
<td>- 3,600,000</td>
<td>- 63%</td>
</tr>
<tr>
<td>2. Farms &gt; 405 ha (farms)</td>
<td>29,000</td>
<td>47,000</td>
<td>+ 19,000</td>
<td>+ 65%</td>
</tr>
<tr>
<td>3. % all farms as large farms [Row 2 / Row 1]</td>
<td>0.8%</td>
<td>8.3%</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td>4. Farmland (M ha)</td>
<td>340</td>
<td>380</td>
<td>+39</td>
<td>+ 12%</td>
</tr>
<tr>
<td>5. Farms &gt; 405 ha (M ha)</td>
<td>ND</td>
<td>250</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td>6. % of farmland in large farms [Row 5 / Row 4]</td>
<td>ND</td>
<td>67%</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td>7. Avg. farm size (ha)</td>
<td>59</td>
<td>180</td>
<td>+120</td>
<td>+ 200%</td>
</tr>
<tr>
<td>8. Cropland (M ha)</td>
<td>170</td>
<td>180</td>
<td>+7.8</td>
<td>+ 5%</td>
</tr>
<tr>
<td>9. Corn for grain (M ha)</td>
<td>38</td>
<td>27</td>
<td>-11</td>
<td>- 28%</td>
</tr>
</tbody>
</table>
Figure 3-2. The percent of land area in each county that is cropland (left) and corn harvested for grain (right). Cropland is defined as the “Number of Improved Acres” in the 1900 Census of Agriculture and “Total Cropland” in 1949 and 2002.

A substantial increase in corn plantings and infrastructure in Iowa and Illinois appear to account for the majority of the spatial redistribution of corn agriculture in the U.S. over the last century.
Land Use and Water Quality Relationships

There is a direct relationship between the fraction of a watershed’s total area in cropland or cropland planted in corn with the concentration of NN exiting the watershed (Figure 3-3). The best-fit simple linear regressions developed are shown in Table 3-2 where NN is the nitrate concentration exiting a watershed in mg N L\(^{-1}\), and \(F_{\text{Cropland}}\) and \(F_{\text{Corn}}\) are the fractions of a watershed’s total area in cropland or corn for grain, respectively. While the study watersheds contain urban-industrial, forestry, and other land uses that potentially impact N export and should be considered in a full budget analysis, the evidence presented here indicates a strong association between agricultural production and riverine NN concentration.

An analysis of covariance performed for both variables has similar results: the regressions for 1906-1912 and 1993-1997 have significantly different slopes (\(p < 0.0001\)), but the intercepts are not significantly different from zero or from each other.

Surprisingly, the NN concentrations in the early 20\(^{th}\) Century are highly correlated with the extent of agriculture then. Manures were the primary source of fertilizer for these farm operations while drainage practices were minimal, and the primary work was powered by animals. As a result, the net loss of NN from the fields to the streams and rivers was lower than today (Figures 3-1, 3-3) and probably driven by the mineralization of virgin soil organic matter with agricultural tillage. The significant linear regressions between land use and NN in both 1906-1912 and 1993-1997 indicate that there are practices inherent to agriculture that drive the variations in NN concentrations among watersheds and across timescales, while the increase in the slopes of both relationships between 1906-1912 and 1993-1997 reveal the effects of changing management practices. The use of mechanical tillage, tile drainage, and industrial fertilizers, for example, are current practices that could explain the changing relationships. Interestingly,
Figure 3-3. (a & b) A linear regression between the fraction of cropland cover (a) or harvested corn (b) and the concentration of nitrate-N (NN) by watershed at the beginning and end of the 20th Century. Data at New Orleans (not included in the analysis) is a bold triangle or circle, for 1906-12 and 1993-97, respectively. (c) The average NN concentration for watersheds with 0%-30% and 30%-60% cropland cover. The features are the minimum and maximum values, 25th and 75th percentiles, and the median (alpha = 0.01, separate analyses by group). (d) A linear regression between a watershed’s Cropland Diversity Index and the NN concentration in 1993-1997 (left) or the fraction of the watershed in corn (right).

Watersheds with over 60% cropland or 25% corn demonstrate the highest NN concentrations and seem to be driving the observed shift in the latter half of the 20th Century.

There was no change in the NN concentration for watersheds with 0% to 30% cropland cover for four intervals from 1906-1912 to 1993-1997 (Figure 3-3c). This result can be
Table 3-2. Regression models and supporting statistics describing land use and water quality relationships at the beginning and end of the 20th Century.

<table>
<thead>
<tr>
<th>Year</th>
<th>Regression model</th>
<th>$r^2$</th>
<th>p value</th>
</tr>
</thead>
<tbody>
<tr>
<td>1906-1912</td>
<td>$NN = 0.03 + 0.97*F_{\text{cropland}}$</td>
<td>0.46</td>
<td>$&lt; 0.0001$</td>
</tr>
<tr>
<td>1993-1997</td>
<td>$NN = -0.23 + 4.54*F_{\text{cropland}}$</td>
<td>0.61</td>
<td>$&lt; 0.0001$</td>
</tr>
<tr>
<td>1906-1912</td>
<td>$NN = 0.16 + 2.43*F_{\text{corn}}$</td>
<td>0.41</td>
<td>$&lt; 0.0001$</td>
</tr>
<tr>
<td>1993-1997</td>
<td>$NN = 0.33 + 9.70*F_{\text{corn}}$</td>
<td>0.70</td>
<td>$&lt; 0.0001$</td>
</tr>
</tbody>
</table>

explained by the buffering capacity of soil ecosystems and suggests that soil systems in minimally impacted watersheds (<30% cropland) generally exported similar NN concentrations in both the beginning and end of the last century. There was a significant change, however, in riverine NN concentrations from 1974-1976 to 1993-1997 in watersheds with 30% to 60% cropland cover (Figure 3-3d). This result suggests that higher riverine NN concentrations observed in intensively managed watersheds (>30% cropland) is a functional response to agricultural N inputs, and that the response interval was most significant after the mid 1970s when fertilizer consumption was about half of present-day values (Figure 3-3d). The mid 1970s is also when the formation of the hypoxic zone at the mouth of the Mississippi River became a regular summertime event (Turner et al. 2008). The decreasing median NN concentration in watersheds with 30%-60% cropland cover from 1955-1959 to 1974-1978 is interesting and may be a reflection of the small sample size used for the analysis ($n = 26$) or possibly due to an increase in the production of wheat during this time period in the U.S.

**Cropland Diversity**

I examined the GIS database for those practices negatively associated with NN export. I used a simple Cropland Diversity Index (CDI), modified from the biological standard Simpson’s Diversity Index (Simpson 1949) which was developed to estimate species diversity within a
known population, to measure the probability that two hectares of cropland randomly chosen from a watershed are planted in distinct crops (Figure 3-3d). The equation used to determine the CDI is:

\[ CDI = 1 - \sum_{i=1}^{9} \frac{Crop_i^2}{Cropland^2} \]

where CDI is the Cropland Diversity Index for a given watershed, Crop is the number of hectares of a particular crop type within that watershed, and Cropland is the number of hectares of cropland within that watershed.

To generate the CDI, I included reported harvested areas of barley, corn, cotton, hay, oats, rice, sorghum, soybeans, and wheat. A low CDI score indicates a low probability that two random hectares are planted in different crops. A CDI score of 0.5, for example, implies that there is a 50% chance that two randomly chosen hectares of cropland belong to distinct crops, e.g., cropland covered exclusively in a corn-soybean rotation system. A surprising result showed that cropland diversity in 1993-1997 was inversely related to riverine nitrate-N concentrations (\( r^2 = 0.45, p < 0.0001 \)) (Figure 3-3d). The diversity of croplands in these watersheds is also directly related to the fraction of a watershed harvested in corn (\( r^2 = 0.80, p < 0.001 \)) (Figure 3-3d). Watersheds with a larger fractional area in corn demonstrated lower cropland diversity and higher NN concentrations. This result, coupled with the results from Figures 3-3a, 3-3b, and 3-3c, suggests that the rise in intensive homogenous croplands, especially corn production systems, is responsible for a significant proportion of the increase in NN concentration values observed in the latter half of the 20th Century. Similar to recent model predictions (Donner and Kucharik 2008), the results of this analysis demonstrate that there will be potentially more NN loading as the fraction of watersheds in corn agriculture rises with biofuel production.
CONCLUSIONS

The average NN concentrations of the 63 rivers monitored in the beginning and end of the last century was 3-4 times higher at the end of the century. The magnitude of this change was especially strong in the Mississippi River Basin and its Midwest region. A linear relationship existed at the beginning of the 20th Century between NN concentrations and the extent of agricultural cropland, and particularly with the area of harvested corn. Agricultural cropland planted in corn was the most significant agricultural driver of riverine NN concentration identified from the factors considered here. In addition, the slopes of the regression models increased in the last 100 years suggesting that management practices, such as commercial fertilizer application, mechanical tillage, and intensive drainage, are responsible for the increase in NN export per hectare of cropland in the latter half of the century. I recognize that the national specialization and intensification of corn agriculture in the Midwestern states has met social goals to produce high-yielding grain crops in large quantities. I also recognize that complex socio-economic structures determine regional land use patterns. The results of this research, however, indicate that the continued expansion of modern corn agriculture will likely increase the NN concentration in rivers and streams, particularly in those watersheds where corn cropland occupies more than 25% of the total area.

The NN concentrations in low-disturbance watersheds, on the other hand, have not shifted in the last 100 years. That is to say, the intercept for all land use – water quality relationships examined in this study are not significantly different from zero in the beginning and end of the last century. This implies that water quality may be rehabilitated if there are changes in land use practices such as a reduction in cropland area dedicated to corn agriculture, increases in cropland diversity, or changes in intensive land management. Increasing the area of perennial
systems could meet these objectives and will likely return major improvements in water quality, ecosystem services, and on-farm benefits (Cox et al. 2006, Glover et al. 2007, EPA2008).

Policy initiatives directed at these land use practices will have a direct effect on the water quality of both higher-order streams and coastal areas by improving soil quality while benefiting the farming enterprise. A full assessment of the direct and indirect impacts of these improvements amidst complex social, political, and economicsystems will require, I think, large-scale (5,000 km²) and long-term (decades) watershed-level empirical studies that capture this complexity (Jordan et al. 2007).

**LITERATURE CITED**


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Chapter Four
Linking Federal Farm Policy, Agricultural Landscapes, and Water Quality in 20th Century America

INTRODUCTION

The 20th Century was marked by growing farm size, mechanization, and specialization in American agriculture (Table 4-1). Although the area of total farmland increased by 11.5% from 1900 to 2002, the total number of farms fell almost 63% and the average farm size doubled. The average number of commodities produced per farm in the U.S. decreased from five in 1900 to just over one in 2002. In this time period, the percent of the U.S. population employed in agriculture fell from 41% to 2%, and 93% of farmers earned off-farm income in 2002 compared to 30% in 1900. The mechanization of American agriculture can be observed in the substantial increase in the number of farm tractors with a subsequent decrease in animals used for farm power (Table 4-1). Thus, the average American farm is now producing fewer agricultural products, is more dependent on intensive machinery, and is more reliant on off-farm income.

The Mississippi River Basin (MRB) is the ‘iconic’ agricultural watershed in the US. It is the largest watershed in North America (Van der Leeden et al. 1990) and has more agricultural land than any other watershed in the world (IUCN et al. 2005) (Figure 4-1). The land cover of

Table 4-1: Changes in 20th Century American agriculture, including number of farms, farm size, production, workforce, off-farm income, work animals and tractors (N.D. = no data; number of farm animals were no longer recorded after 1960). Data source: Dimitri et al. 2005.

<table>
<thead>
<tr>
<th></th>
<th>1900</th>
<th>1945</th>
<th>2002</th>
</tr>
</thead>
<tbody>
<tr>
<td>Number of farms (million)</td>
<td>5.7</td>
<td>5.9</td>
<td>2.1</td>
</tr>
<tr>
<td>Average size of farms (acres)</td>
<td>146</td>
<td>195</td>
<td>441</td>
</tr>
<tr>
<td>Average number of commodities</td>
<td>5.1</td>
<td>4.6</td>
<td>1.3</td>
</tr>
<tr>
<td>Percent of U.S. workforce</td>
<td>41</td>
<td>16</td>
<td>1.9</td>
</tr>
<tr>
<td>Percent of farmers (1900, 1945)</td>
<td>30</td>
<td>27</td>
<td>93</td>
</tr>
<tr>
<td>Number of work animals (million)</td>
<td>21.6</td>
<td>11.6</td>
<td>N.D.</td>
</tr>
<tr>
<td>Number of tractors (million)</td>
<td>0</td>
<td>2.4</td>
<td>4.6</td>
</tr>
</tbody>
</table>
today’s MRB is a consequence of the anthropogenic transformation of the predominately prairie-woodland habitat of 200 years ago into a landscape that is now 58% cropland, 21% range or barren land, 18% woodland, 2.4% wetlands or water, and 0.6% urban land (Goolsby and Battaglin 2001; Turner and Rabalais 2003) (Figure 4-1). It has become known, not unfairly, as the “bread basket” of the U.S. Fifty-seven percent of all U.S. farmland lies within the MRB, which encompasses 81% of all U.S. corn and soybeans acres and 61% of all U.S. wheat acres (U.S. Department of Agriculture (USDA) 2005).

This land use transformation into an industrial model of agricultural production, however, has multiple consequences away from the farmfield, including consequences for what economists call ‘public goods’ and ecologists call ‘ecological services’. Farm management decisions are clearly based on the impacts to the farming enterprise and its social network, but...
this is often at the expense of weaker incentives for mitigating off-site impacts. Excessive nutrient loading, for example, has become one of the most dominant drivers of ecosystem change in the last 40 years for both terrestrial and aquatic ecosystems (MEA 2005). The nitrogen flux from the MRB from 1955-1970 to 1980-1999 increased nearly 3 fold, due almost entirely to increases in nitrate as a consequence of the intensification of agricultural practices (Goolsby and Battaglin 2001, Turner and Rabalais 2003). Furthermore, nutrient loading to terrestrial and aquatic ecosystems is projected to increase in the 21st century and remain the dominant driver of changes in biodiversity in these ecosystems (MEA 2005).

A specific example of nutrient enrichment is from the Des Moines Water Works (DMWW) in Des Moines, IA. The DMWW employs an expensive ion exchange process to reduce the nitrate concentration in their drinking water, pumped from the Raccoon and Des Moines Rivers, to meet national health standards. In 2007, the DMWW reported the average nitrate-N concentration in their drinking water after processing was the maximum allowed - 10 mg L\(^{-1}\) (DMWW 2007). Downstream, the average size of bottom water hypoxia in the Northern Gulf of Mexico is growing (Figure 4-1, Rabalais et al. 2007) and the collective conclusion of hypoxia-related research over the last decade has confirmed the original hypothesis that nitrogen flux from the Mississippi River, particularly in the form of nitrate, is the dominant driver of hypoxia formation in the Northern Gulf of Mexico (Rabalais et al. 2007, Turner et al. 2008). A 30% to 50% reduction in nitrate export from the MRB may be necessary to reduce the average annual size of the Gulf of Mexico hypoxic zone to 5,000 km\(^2\), the target goal (Mississippi River/Gulf of Mexico Watershed Nutrient Task Force 2001, Scavia et al. 2003, EPA 2008).

**Rationale**

The focus of this dissertation is the relationship between land use and nitrate-N (NN) loading from the upland watersheds in the MRB and other major watersheds in the continental
Chapter 4 is about the statistical relationships between several agricultural land use practices and increasing NN concentrations in the watersheds of the U.S. In this chapter, I investigate relationships between federal farm policies and agricultural land use practices as well as NN concentrations.

National farm policy is intended to support the income of farming operations and to influence the supply of agricultural products via price support programs. Farm policies potentially influence cropping choices through both direct economic subsidies and agronomic advice. Many other factors influence cropping choices by individuals, families, and corporate entities, including family health and community based factors, labor supply, infrastructure costs, water supply, and net profits. Although the $24 billion in farm subsidies in 2005 were only 1% of total U.S. farm expenditures, they accounted for 32% of the total net income of the agricultural sector (BEA 2007, ERS 2008a). Farm policy, therefore, has a potentially large influence on the success or failure of individual farms and long-term management decisions.

**History and Description of Government Farm Programs**

The central purpose of U.S. agricultural policy has been commodity-specific price supports, supply controls, and income support programs for U.S. farmers. The first “Farm Bill” was established in the Agricultural Adjustment Act of 1933. This legislation was part of the New Deal and created a mix of commodity-specific and income support programs that supported the typically small, diversified farms of rural America in the 1930s (Dimitri *et al.* 2005).

Subsequent farm bills introduced soil conservation programs (1936), fixed-price commodity support programs (1949), flexible price commodity support programs (1954), rural development programs (1970), food stamp programs (1977), conservation reserve programs (1985), direct payment programs for specific commodities (1997), and counter-cyclical payment programs to offset low market prices for specific commodities (2002) (Bowers *et al.* 1985,
Dimitri *et al.* 2005). The complete texts of all U.S. farm bills are available on the National Agricultural Law Center website (NALC 2008). See Appendix III for a detailed description of the current government programs.

The direct commodity payments accounted for the largest share (26%) of government farm payments in 2002-2005, with loan deficiency, conservation program, ad hoc and emergency, and counter-cyclical payments collectively accounting for 54% of the total U.S. payments (Figure 4-2) (ERS 2008). The direct commodity payments are provided principally for grain and oilseed crops including: barley, canola, corn, crambe, flaxseed, grain sorghum, mustard, oats, peanuts, rapeseed, rice, safflower, sesame, soybeans, sunflowers, upland cotton, and wheat (FSA 2006). These payments are based on county-level historical yields and farm-level historical planting areas, also known as ‘base acres’. Once the direct payment rate is determined by the USDA Farm Service Agency (e.g., $0.28 per bushel of corn in 2006), then the total direct payment for an individual farm is equal to the payment rate, times the number of base acres for that crop on that farm, times 85% to account for non-harvested plantings, times the

![Figure 4-2. The average annual payments of U.S. government farm programs from 2002 through 2005. See Appendix III for a description of the farm programs.](image_url)
historical yield for that crop for that farmer (base yield) or the Farm Service Agency (FSA) determined county average yield. The direct commodity payments are, therefore, de-coupled from the actual farm production and market price. Once a farming operation is enrolled in a direct commodity program, it receives this fixed direct payment regardless of the farm’s production that year. Five hundred thousand farms received these fixed direct commodity payments in 2002 (Dimitri 2005), equal to 25% of all farms, while corn and wheat payments accounted for over 60% of all U.S. direct commodity payments in that year (USDA 2006).

Questions Addressed

The questions I address here are: 1) how much influence does federal farm policy, i.e., government farm subsidies or programs, have on crop choices and farming practices; 2) how are these crop choices and farming practices influencing riverine NN concentrations in our nation’s agricultural watersheds? and 3) what role, if any, did government commodity payments have in the consolidation and specialization of modern agriculture? I address these questions using data on land use, fertilizer use, cropping patterns, and water quality at the watershed scale.

Materials and Methods

The data sources and data preparation techniques used in the spatial regression analyses in this chapter are identical to those used in Chapter 4. Chapter 3 has a detailed explanation of the data sources and preparation techniques for agricultural land use data (Haines and ICPSR 2004; USDA 1979, 2005; U.S. Bureau of the Census 1990, 2001), water quality data (Clarke 1924, USGS 2007), and the GIS databases used in this study. Government farm payments were documented in most of the Census of Agriculture reports as “Total Government Payments” and include all payments made to farming operations by the U.S. federal government. I did not distinguish between fixed commodity payments, payments made for conservation practices, and other miscellaneous payments, because these distinctions were not included in the Census of
Agriculture until 2002. The agricultural economic data used in the discussion of government payment distributions are from published datasets from the USDA, the USDA Economic Research Service, and the Bureau of Economic Analysis. The farm income and government payment data used herein are published datasets from the Economic Research Service (2008a) and the Bureau of Economic Analysis (BEA 2007). The Consumer Price Index (CPI-U) was used to adjust for inflation (BLS 2008) with 1982-1984 as the base period.

I used 2002 Census of Agriculture data (USDA 2005) summed by watershed and 1997-2002 averaged NN measurements in 29 independent watersheds across the continental U.S. (USGS 2007), explained in detail in Chapter 3, to test if there is a statistically-significant relationship between government farm payments, fertilized farmland, and NN concentrations exiting the study watersheds. These watersheds have a cumulative total area of 227,344,487 ha (561768226 acres) and cover 54% of the entire MRB and 30% of the continental U.S.

All regression analyses presented here are watershed-level analyses and employ a log transformation to NN concentration values to normalize the residuals of the regression models. I use data for three Midwestern states (Illinois, Iowa, and Minnesota) to document the transformation of crop choices in the center of the MRB. A Cropland Diversity Index (CDI) for each watershed was created where the CDI measures the probability that two randomly chosen acres of cropland in a watershed belong to distinct crops. A ‘high’ CDI score of 0.90 indicates that there is a 90% chance that the two acres are different crop types (see Chapter 3 for a detailed explanation of the development of the CDI).

These data were used to describe the changes in cropping choices, the importance of farm payments to farm income and farm size, and the relationships between government payments and various indices of water quality and agricultural land management.
RESULTS AND DISCUSSION

Indices of the Consolidation of Crop Types

The specialization and consolidation of agriculture in four Midwestern states is illustrated by changes in farm size and crop acreage from 1900 to 2002 in Figures 4-3 and 4-4 (soybeans is an exception and is shown as the relative change in area since 1949 because soybeans were not introduced on a large scale in American agriculture until after World War II). There was a decreasing number of farms with relatively little change in farmland and cropland area over the 102 year period. As a result, the average farm size doubled in Minnesota and Iowa, and tripled in Illinois from 1900 to 2002. Corn and soybeans were the only two commodity crops to increase their planted area in all three states during the 20th Century. Corn acreage rose 4%, 20%, and 355% from 1900 to 2002 in Illinois, Iowa, and Minnesota, respectively. The dramatic rise in soybean acreage during the last half of the century, from nonexistent in 1900 to 43%, 38%, and 31% of the total 2002 cropland in Illinois, Iowa, and Minnesota, respectively, is offset by a concurrent reduction in small grain and hay plantings. Oats, barley, rye, buckwheat, and potatoes were virtually absent in these three states in 2002.

Another way to describe the shift in planted area during the last 100 years is a cumulative bar graph presented in Figure 4-5. Here the figure demonstrates decreasing cropland diversity in the three Midwestern states. By 2002, corn and soybean acreage occupied 88%, 82%, and 60% of the total cropland in Illinois, Iowa, and Minnesota, respectively. This pattern of decreasing cropland diversity is especially evident in Minnesota where wheat, hay, oats, barley, and flax occupied 72% of the cropland in 1900, compared to 17% in 2002. Croplands in the Midwest have specialized over the course of the 20th Century and now primarily produce corn, soybeans, and a nominal amount of wheat and hay (Figure 4-5).
Government Payments and Farm Size

The structural changes in American agriculture are shifting production towards consolidation in very large farms (>\$500,000 annual sales). These large farms accounted for 45% of the total U.S. value of production in 2003, compared to 32% in 1989 (MacDonald 2006).

Figure 4-3. The relative change in the number of farms, acres of farmland and cropland, and the average farm size from 1900 to 2002 for three Midwestern states.

Figure 4-4. The relative change in crop cover from 1900 to 2002 for three Midwestern states. The percent changes in soybean cover are from 1949 to 2002.
With the production of program commodities shifting to larger farms, government commodity payments have also shifted to larger farms because payments are linked to planting and yield histories. Key and Roberts (2007a) demonstrated that, at the regional scale, the higher commodity payments per acre of cropland are statistically correlated with subsequent farm growth and farm survival. These regions with higher commodity payments per acre of cropland also demonstrate faster rates of cropland concentration where croplands are increasingly consolidated into larger farms (Key and Roberts 2007b). Thirteen percent of all U.S. farm payments in 1989 were paid to very large farms (farms with over $500,000 in annual sales) compared to 32% of all payments in 2003 (MacDonald 2006).

The increase in household income on these very large family farms has consequently outpaced the growth in overall U.S. incomes. The median U.S. household income grew 1% from $42,892 in 1989 to $43,318 in 2003, while the median income of households receiving
commodity payments increased 65% from $45,808 in 1989 to $75,772 in 2003 (MacDonald 2006). Note that the median income of households receiving commodity payments was similar to the national average in 1989, but was substantially higher in 2003. The shift in commodity payments toward very large farms with relatively high household incomes demonstrates the structural changes that took place in the latter part of the 20th Century. One consequence of this shift is that large farms are less likely than small farms (average annual sales less than $50,000), particularly those small farms operated by land owners, to practice sustainable agricultural practices such as agroforestry and organic agriculture (Tavernier and Tolomeo 2004). What are the economic implications of these changes in American agriculture to farm income?

**Government Payments and Farm Income**

The total net farm income of all U.S. farms declined over the second half of the 20th Century by 35%, or from $62 billion in 1950-1955 to $40 billion in 2001-2005 (real 1982-1984 dollars) (Figure 4-6). The total U.S. government farm payments increased over the same interval by 10 fold - from $1 billion in 1950-1955 to $11 billion in 2001-2005 (real 1982-1984 dollars) (Figure 4-6). The proportion of total net farm income in the U.S. from government farm payments has subsequently increased from 9% in 1933-1937, the years of the first farm bill, to 29% in 2001-2005 (Figure 4-7). During this same time period, the minimum proportion of net farm income from government payments was 1.4% in 1949, a year of near record-high farm income. The maximum proportion of net farm income from government payments in this period was 65% in 1983, the year with the minimum net farm income. Government farm payments, from this perspective, appear to accomplish a central goal of federal farm policy - to provide income support for farming operations when farm income from market sales is lower than average. The question remains, however, if government payments are an important factor in a
Figure 4-6. A time series of decreasing net farm income and increasing total government farm payments for the U.S. from 1933 to 2006. The dollars are adjusted for inflation using the Consumer Price Index (1982-1984 price level).

Figure 4-7. A time series of the increasing proportion of U.S. net farm income that is derived from government farm payments.

The farm operator’s decision to consolidate more cropland onto larger farms and to plant specific commodities in favor of other crop types.
Government Payments, Commercial Fertilizer Usage, and Riverine Nitrate Concentrations

The total amount of government farm payments per acre of farmland is concentrated in the Heartland, Great Plains, Mississippi River corridor, the Southeastern Piedmont, and the California valley (Figure 4-8a). These are the areas of the U.S. that produce the majority of the nation’s grain, oil seeds, and cotton - the principle recipients of commodity program payments. The nationally averaged direct commodity payment per base acre of cropland in 2002, was $96.21 for rice, $45.73 for peanuts, $34.26 for cotton, $24.35 for corn, $16.78 for sorghum, $15.25 for wheat, and $11.52 for soybeans, while the remaining commodity crops received between $0.008 and $10.00 per acre. These estimates were calculated from published national records of historical crop base acres, crop yields per base acre, and direct payment rates (assuming payments were 85% of the maximum possible payout to account for non-harvested cropland) (FSA 2006, ERS 2008b).

These same regions of the country, the Midwest in particular, also apply inorganic commercial fertilizer to larger proportions of farmland and exhibit higher concentrations of NN concentrations exiting their watersheds (Figure 4-8b, 4-8c).

A significant linear relationship exists in these watersheds between the total amount of government farm payments per acre of farmland and the fraction of a watershed’s farmland that is fertilized ($r^2 = 0.82, p < 0.0001$) (Figure 4-9a). A similar relationship exists between the fraction of farmland that is fertilized and the concentration of NN exiting the study watersheds ($r^2 = 0.40, p < 0.0001$) (Figure 4-9b). These results suggest a significant association between the amount of government payments a farm received per acre, the proportion of the watershed’s farmland that was fertilized, and the concentration of riverine NN exiting the agricultural watersheds. Finally, a direct relationship can be demonstrated between the total amount of government farm payments per acre of farmland and riverine NN concentrations by watershed
Figure 4-8. The magnitude and spatial distribution of: (a) government farm payments per acre of farmland, (b) percent of farmland area treated with commercial fertilizer, and (c) riverine nitrate-N exiting the study watersheds in 2002.
(r^2 = 0.43, p < 0.0001) (Figure 4-9e). This association suggests that farm programs are playing an important role in driving land use practices that ultimately affect riverine NN concentrations.

**Government Payments, Cropland Diversity, and Riverine Nitrate Concentrations**

A negative, non-linear relationship exists between the CDI of a watershed and the total amount of government farm payments per acre of farmland in that watershed (r^2 = 0.82, p < 0.0001) (Figure 4-9c). The diversity of the cropland decreases as government payments per acre of farmland increase, suggesting that farmlands receiving higher payments are specialized agricultural landscapes with low cropland diversity. Watersheds with low cropland diversity were also associated with higher riverine NN concentrations (r^2 = 0.64, p < 0.0001) (Figure 4-9d).

These results suggest that agricultural landscapes receiving higher government payments per acre of farmland are dominated by a few specific commodities and tend to export water with a higher concentration of NN per acre of farmland than farmland receiving lower government payments. One possible explanation for this observation is that less diverse farmlands tend to have larger fields and could be described as homogenized landscapes. This agricultural system is easier to farm on a large scale, with the advent of mechanized tillage and synthetic chemical applications, and accompanies the increase in farm size discussed above.

Another reason for the observed effect is that lower cropland diversity is associated with more intensive corn agriculture (Chapter 4). Those watersheds with a larger fraction of land in corn or soybeans (this is the most common agricultural rotation in the Midwest where corn and soybeans are grown in the same field in consecutive years) are linearly related to higher riverine NN concentrations (r^2 = 0.67, p < 0.0001) (Figure 4-10a). Watersheds that receive more government farm payments per acre also demonstrate a larger fraction of land in corn or soybeans (r^2 = 0.94, p < 0.0001) (Figure 4-10b). No other commodity crop I examined was
Figure 4-9. A simple linear regression analysis for 29 independent watersheds across the continental U.S. between (a) total government farm payments per acre of farmland and the fraction of fertilized farmland (farmland that is treated with commercial fertilizers), (b) the fraction of fertilized farmland in a watershed and the nitrate-N (NN) concentration exiting the watershed, (c) government farm payments per acre of farmland and the Crop Diversity Index of a watershed, (d) the Cropland Diversity Index and the NN concentration exiting a watershed, and (e) government farm payments per acre of farmland in a watershed and the NN concentration exiting the watershed (five year averages, 1993-1997). See the text for an explanation of the Cropland Diversity Index. Dotted lines represent 95% confidence bands.
significantly related to the payment of government farm payments per acre, suggesting that corn and soybean agricultural systems are driving government payments in the watersheds. Nitrogen application rates for corn (131 lbs acre\(^{-1}\) or 147 kg ha\(^{-1}\)) were substantially higher than those for soybeans (18 lbs acre\(^{-1}\) or 26 kg ha\(^{-1}\)) or wheat (66 lbs acre\(^{-1}\) or 74 kg ha\(^{-1}\)) in 2000 (NASS 2001) and could explain why croplands planted in corn explain more of the variation in NN concentration by watershed than any other crop type examined. Additionally, the 2000 U.S. corn crop accounted for 40% of the nation’s total N fertilizer consumption, which far exceeded that of the wheat (15%), cotton (5%), or soybean (1%) crops (ERS 2008c). Thus, a connection can be made between government farm payments, specific commodities and their associated agricultural practices, and riverine NN concentrations in receiving surface waters.

**CONCLUSIONS**

The total value of government farm payments in the U.S. increased, and the total U.S. net farm income decreased, in the 20\(^{th}\) Century. Consequently, the proportion of net farm income from government farm payments increased to nearly 30% in 2005, suggesting that farming operations are dependent on these payments to be economically viable and that government payments are a determining factor in agronomic decisions. It is possible that the income support provided by these programs has reduced the risk associated with specialization and allowed farmers the opportunity to focus on one or two crops. This focus, however, ignores the benefits associated with diversification that farmer’s in the early 20\(^{th}\) Century needed to survive the vagaries of the weather, insect, pest, and disease problems. On the field level, agricultural landscapes that received more direct government payments per acre of farmland are associated with increasing farm size, and are associated with agricultural practices that tend to fertilize larger proportions of farmland, tend to have lower cropland diversity, and are associated with higher riverine NN concentrations. These observed artifacts of 20\(^{th}\) Century American
agriculture, such as specialization, consolidation, and intensification, may have occurred regardless of government support programs. The transition, however, would most likely have been slower and more gradual because government support has reduced the risk of farming operations (Dimitri 2005) and allowed larger operations to invest in land and capital acquisitions.

I suggest that using government commodity programs to support a wider variety of crop types, particularly on smaller farms (Tavernier and Tolomeo 2004), could decrease the risk to diversify croplands in impaired agricultural landscapes and stimulate economic markets for other crops. Additionally, increasing government farm programs for soil conservation could protect
valuable soil resources, investing in long-term fertility and agricultural sustainability. These goals can be accomplished while supporting working farms and could potentially distribute income support benefits more broadly across the U.S. agricultural sector (Claassen et al. 2007), although particular local conditions (Knowler and Bradshaw 2007) and land tenure (Soule et al. 2000) will have to be accounted for when promoting conservation agriculture. The U.S. Government Accountability Office found that conservation programs and commodity programs are currently working at cross purposes in that commodity program payments influence landowner’s decisions to convert grasslands to croplands (GAO 2007). These programs could work cooperatively if, for instance, farming operations are required to implement certain conservation practices to receive commodity payments (Westra 2008). Examples of conservation management practices that could increase soil fertility, thereby reducing a farm’s dependency on inorganic fertilizer applications and subsequently reduce N leaching, include continuous living cover on the soil for the majority of the growing season (Jordan et al. 2007), reduced dependency on field drainage systems (Randall et al. 1997), and increased production of perennial field crops (Cox et al. 2006, Glover et al. 2007).

The recurring farm bill is one of the most expensive and far-reaching pieces of legislation in U.S. agriculture. It can be used as a tool to implement large scale changes in land use, the order of which will be necessary to address national water quality issues like hypoxia in the Northern Gulf of Mexico. To resolve problems of this magnitude, we will need large-scale, long-term watershed studies (Jordan et al. 2007) and organizing bodies that cross political boundaries to apply land use practices in ecological units from upland micro-watersheds to coastal bays.
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Chapter Five
Can Perennial Farmlands Reduce Nitrate Export to Surface Waters?

INTRODUCTION

The production of humanity’s food and fiber has created agricultural systems (areas where at least 30% of the landscape is in croplands, shifting cultivation, confined livestock production, or freshwater aquaculture) that cover one quarter of the Earth’s land surface (MEA 2005). Annual crops grown as monocultures cover about two thirds of the world’s croplands (FAO 2008). Recently, however, there is concern about the environmental off-site impacts of agricultural production on food productivity and on the integrity of our natural ecosystems upon which food production depends (Faeth 2000, Tilman et al. 2001).

Agricultural lands, for example, have been shown to influence the in-stream nitrogen (N) concentration, including in the Mississippi River Basin (MRB) (Goolsby et al. 1999, Turner and Rabalais 2003). The nitrogen flux from the MRB increased nearly 3 fold from 1955-1970 to 1980-1999 (Goolsby and Battaglin 2001). The N from agricultural runoff accounted for 66% of the in-stream N additions to the Mississippi River from 1955-1970 to 1980-1996, with Minnesota, Iowa, and Illinois accounting for 40% - 50% of the total nitrogen inputs (Goolsby et al. 1999). Agricultural practices currently account for 70% of the N exported from the MRB, with corn and soybean cultivation being the largest contributor (52%; Alexander et al. 2008). This nitrogen export from the rivers of the MRB influences local water quality and controls the frequency and size of bottom water hypoxia in the Northern Gulf of Mexico (Rabalais et al. 2007, Turner et al. 2008). The research presented here is part of an effort to examine the relationships between different agricultural land use practices and nitrogen exports from
watersheds. This chapter focuses primarily on the comparison of perennial and annual agricultural systems and their effects on water quality.

**Natural Ecosystems as a Model for Sustainable Agriculture**

If we examine natural land-based ecosystems to understand properties that may strongly influence the effective management of soil and water resources, then we find that most of the plant communities are diverse and are comprised of perennials, i.e., plants that live for more than two years. The native prairie community that once dominated the Great Plains and upper Midwest (Figure 5-1) is an example of a diverse perennial system that can be a model for current agricultural systems in this area (Jackson 1999, Jackson 2002). The conversion of natural ecosystems, such as perennial prairie communities, to agricultural cropland planted largely in annual crops has effectively reduced the stored ecological capital that was built up and maintained by natural systems with few exceptions (Pimm 2001). This depletion is partially a

![A photograph of an undisturbed native prairie landscape in the Flint Hills of Kansas, U.S.A. Photo taken in the Summer of 2005.](image)
consequence of how the annual crops are grown and harvested. The annual tillage disturbs the soil and annual crops that, in general, have smaller root structures compared to perennials (Jackson et al. 1996). Figure 5-2 demonstrates the substantial below-ground difference between perennial grassland communities and annual wheat fields. This picture was taken from a demonstration soil pit at The Land Institute in Salina, Kansas. One ecological benefit of perennial plants is reduced N leaching through the soil profile because of their robust root architecture and mitigated soil erosion because they provide a continuous living cover on the soil surface for longer periods of the growing season (Glover et al. 2007). Efforts to constructively mimic perennial plant structures or include these processes inherent to perennial ecosystems within agricultural systems do appear, therefore, to be important evolutionary aspects of building sustainable agricultural systems.

Figure 5-2. A comparison of typical plant density, plant diversity, and root structures between native perennial prairie vegetation (left) and annual wheat vegetation (right) typical of Kansas, U.S.A. Photo credit: Jim Richardson.
The prospects of developing perennial grain and biomass crops are promising, albeit long-term endeavors. Advances in embryo rescue plant breeding of perennial grains (Cox et al. 2006, Glover et al. 2007) and post-harvest processing of bio-based chemicals and fuels (Tilman et al. 2006, Regauskas et al. 2006), however, make the transition to perennial crops more feasible and the need for complimentary ecological research more urgent.

In this chapter, I quantify the effects of perennial farmland and annual cropland on riverine nitrate-N (NN) concentrations from the beginning to the end of the 20th Century in two regional studies of: (1) grassland-dominated watersheds in the Great Plains, and, (2) representative watersheds across the continental U.S. I find that there is a direct relationship between the area of annual cropland and NN concentrations, and an inverse relationship between NN concentration and the area perennial farmland. These contrasting results are, in part, consistent with the differences in vegetative vitality, or “greenness”, between perennial and annual croplands throughout the growing season and have implications for how we might more effectively manage soil and water resources to reduce surface water NN export.

**Materials and Methods**

I used 5-year averaged water quality and land use data from eight grassland-dominated watersheds (Figure 5-3) for 1906-1912, 1974-1978 and 1993-1997 to test for relationships between the area of annual croplands and regional riverine NN concentrations. I used 36 watersheds across the continental U.S. to test for relationships between the area of perennial farmlands and regional riverine NN concentrations. The 1906-1912 water quality data are from US Geological Survey (USGS) river monitoring stations reported in Clarke (1924). The more modern water quality data (1974-1997) are from the USGS National Water Information System (2007). The selection of monitoring stations and years was based on the completeness of NN
Figure 5-3. The spatial distribution of grassland-dominated watersheds used in the multiple regression analysis.

concentration data sets, which were most complete for the years 1906-1912, 1974-1978 and 1993-1997.
I compiled digitized Census of Agriculture county-level data regarding land use practices, crop and livestock production, and human population for these same time periods. The original datafiles used in this study were digitized by M. Haines and colleagues from Census of Agriculture reports from 1905 (Haines 2004), 1978 (U.S. Bureau of the Census 1990), 1997 (U.S. Bureau of the Census 2001), and 2002 (U.S. Department of Agriculture 2005). County-level crop data were used to determine annual and perennial crop cover for spatial and statistical analyses. Annual cropland is defined herein as all cropland reported in the Census of Agriculture minus cropland that is pastured or cropland in hay production. Perennial cropland is defined herein as all cropland minus annual cropland. Perennial farmland is defined as all farmland reported in the Census of Agriculture minus annual cropland. This land use definition includes cropland covered in perennial vegetation, pastures, woodlands, and miscellaneous land cover such as wetlands and surface waters on farms. I converted the county-level land use data to units per watershed area using methods similar to those used by Boyer et al. (2002). Chapter 3 has a detailed explanation of the data sources and preparation techniques.

The land use and water quality data were examined using geographic information system software and commercially-available statistical software Prism 5 statistical software (GraphPad Software, Inc. 2007) and S-Plus 2000 statistical software (MathSoft Engineering and Education, Inc. 1999) were used to perform a simple and multiple linear regression analyses. The application of a Cook’s D test revealed that data from a single monitoring station were consistently and severely impacting the model coefficients in a multiple regression analysis of relationships between landuse and water quality. Data for this watershed were removed on the basis of its influence and the unusual conditions in the watershed: at the first sampling date, its human and cattle population densities were both several times greater than any other watershed.
A linear regression analysis was used to test for the strength of the relationships between annual croplands and riverine NN concentrations for eight grassland-dominated watersheds. The dependent variable, NN concentration, was converted to the natural logarithm to achieve normality. The independent variables included: the fraction of the watershed cropland planted to annual crops, cattle population per watershed area, human population per watershed area, and year. A step-wise regression analysis was then performed for model selection via backward elimination.

The scarcity of water quality data in 1906-1912 reduced the number of watersheds examined in the Great Plains to a total of eight watersheds. Several of the watersheds examined are nested within other study watersheds. I developed a test to determine if a difference existed between the nested and the independent watersheds regarding the relationship between NN concentrations and annual cropland. I used 49 additional watersheds from the same dataset presented in this study and classified the watersheds as independent if they were not encompassed by another study watershed. Nested watersheds are embedded within other study watersheds.

RESULTS AND DISCUSSION

Annual and Perennial Croplands Influence NN Concentrations

The results of the regression analysis using the 1993-1997 data showed that nitrate concentrations are significantly related to the fraction of a watershed in annual cropland for both the independent and nested categories ($p < 0.0001$, $r^2 = 0.61$, $n = 34$ and $p = 0.0013$, $r^2 = 0.56$, $n=15$; respectively) (Figure 5-4). An ANCOVA analysis showed that the slope and intercept of the regression lines were not significantly different from each other ($p = 0.81$ and $p = 0.12$, respectively). I concluded that the nested watersheds in this analysis are not statistically different from the independent watersheds. Because the nested watersheds are not different from
Figure 5-4. A simple linear regression analysis between the fraction of a watershed’s area in annual cropland and the riverine nitrate-N concentration exiting the watershed for (a) independent and (b) nested watersheds across the continental U.S. in 1993-1997. See the text for an explanation of independent and nested watersheds. Dotted lines are 95% confidence bands.

Independent watersheds, and historical data is limited for this region, I chose to use all possible watersheds for the analysis with adequate data from 1906 that lie within the study area.

Riverine NN concentrations throughout the 20th Century are statistically-related to the fraction of a watershed’s cropland in annual crops, human population density and cattle density (F = 12.02, p < 0.0001, n = 20) (Table 5-1). This result indicates that land use quality had an impact on NN concentrations before commercial fertilizers were widely applied. The influence
Table 5-1. The statistical output of the multiple regression model used to determine the influence of annual crop cover and human and cattle population densities on nitrate-N concentrations in eight watersheds across the Great Plains (Figure 5-4). The resulting model had a p-value <0.0001.

<table>
<thead>
<tr>
<th>Model parameter</th>
<th>Value</th>
<th>Standard error</th>
<th>t value</th>
<th>p value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Intercept</td>
<td>-0.15</td>
<td>0.30</td>
<td>-0.66</td>
<td>0.5149</td>
</tr>
<tr>
<td>Annual cropland (Fraction of watershed area)</td>
<td>4.20</td>
<td>1.53</td>
<td>2.74</td>
<td>0.0131</td>
</tr>
<tr>
<td>Cattle population (Cattle per watershed acre)</td>
<td>-37.57</td>
<td>6.51</td>
<td>-5.77</td>
<td>&lt; 0.0001</td>
</tr>
<tr>
<td>Human population (Persons per watershed acre)</td>
<td>17.39</td>
<td>7.47</td>
<td>2.33</td>
<td>0.0311</td>
</tr>
<tr>
<td>Year</td>
<td>0.016</td>
<td>0.00</td>
<td>3.76</td>
<td>0.0013</td>
</tr>
</tbody>
</table>

that cattle and human populations have on NN concentrations in surface waters is most likely from excrements and waste.

Conversely, there is a inverse association between the fraction of a watershed’s total area in perennial farmland and the concentration of NN exiting the watershed (Figure 5-5). The best-fit simple linear regressions developed are shown in Table 5-2. An analysis of covariance performed on the regression models shows that the 1906-1912 and 1993-1997 models have significantly different slopes (p < 0.0001) and significantly different Y-intercepts (p < 0.0001). The significantly higher slope of the modern regression model (1993-1997) is a result of the higher riverine NN concentration in agriculturally intensive watersheds, and is similar to findings presented in Chapter 3. This observation is further evidence that the most intensive agricultural landscapes (>50% cropland) are responsible for the higher NN concentrations observed in the latter half of the 20th Century.

Interestingly, the X-intercept for both regression models is not significantly different from 1.0 (X = 1.024 and 0.973 when Y = 0.0 for 1906-1912 and 1993-1997, respectively),
Figure 5-5. A simple linear regression analysis between the fraction of a watershed’s area in perennial farmland and the riverine nitrate-N concentration exiting the watershed in 1906-1912 (gray triangles) and 1993-1997 (black circles) across the continental U.S. Dotted lines are 95% confidence bands.

Table 5-2. Regression models and supporting statistics describing perennial farmland cover and water quality relationships at the beginning and end of the 20th Century, where NN is the nitrate concentration exiting a watershed in mg N L\(^{-1}\), and FPerennial is the fraction of a watershed’s total area in perennial farmland.

<table>
<thead>
<tr>
<th>Year</th>
<th>Regression model</th>
<th>r(^2)</th>
<th>p value</th>
</tr>
</thead>
<tbody>
<tr>
<td>1906-1912</td>
<td>NN = 1.22 – 1.19*F(_{\text{Perennial}})</td>
<td>0.48</td>
<td>&lt; 0.0001</td>
</tr>
<tr>
<td>1993-1997</td>
<td>NN = 4.60 – 4.73*F(_{\text{Perennial}})</td>
<td>0.65</td>
<td>&lt; 0.0001</td>
</tr>
</tbody>
</table>

indicating that watersheds covered almost exclusively in perennial farmland cover are exporting very little, if any, NN to surface waters. This is further evidence that the baseline conditions for NN export from non-impacted watersheds have not shifted over the last 100 years. This constancy supports the hypothesis that the concentration of NN could be reduced in watersheds with high NN concentrations and low perennial farmland by increasing perennial plant cover.
Agricultural Production and River Discharge

The increase in N export from agricultural watersheds is partially due to increases in river discharge, because export or flux is equal to concentration times discharge. Raymond et al. (2008) demonstrated a strong correlation between the fraction of a watershed’s area in cropland and an increase in river discharge at average precipitation for sub-watersheds of the MRB (Figure 5-6). Seventy-five percent of the total cropland area in the Mississippi River Basin was planted in annual crops in 2002 (USDA 2005). An increase in the river discharge normalized for average precipitation demonstrates that modifications of watershed properties are changing the relationship between precipitation and discharge, causing the amount of discharge to increase in a normal precipitation year. They argued that the relationship between agricultural land cover and discharge at average precipitation demonstrates that agricultural land use practices have increased the Mississippi River discharge by roughly 50 km$^3$ yr$^{-1}$, comparable to the total

Figure 5-6. The change in discharge at average precipitation versus the percentage of a watershed’s area in cropland for sub-watersheds of the Mississippi River Basin. The change was calculated by averaging the time periods before 1966 and after 1987. The black filled circles represent watersheds that are independent, while the grey filled triangles are watersheds that have nested watersheds within them. Adapted from Raymond et al. (2008)$^1$. 

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discharge of the Rhone River (~54 km$^3$ yr$^{-1}$) (Raymond et al. 2008). This finding is consistent with the results of recent studies demonstrating increased river discharges driven by agricultural landscapes (Shilling 2005, Zhang and Shilling 2006) and suggests that mechanical tillage and intensive drainage are potentially affecting the hydrology of nearby rivers and streams.

**Annual and Perennial Croplands Influence Vegetation “Greenness”**

An inherent property of annual croplands is that the annual plant cover dies every winter leaving the soil bare and exposed. The field crops are therefore not actively growing in the early spring and are vulnerable to excessive nutrient leaching during the springtime rain events (Jackson et al. 1996). Additionally, annual crop types are typically harvested in the fall, leaving the fields bare in the late fall when farmers typically apply commercial fertilizers, especially in the Midwest. This temporal pattern is evident in Figure 5-7 where annual and perennial cropland cover, estimated from 2002 Census of Agriculture data presented above, is superimposed over spatially explicit Normalized Difference Vegetation Index (NDVI) data (USDA 2008) for 2007. The NDVI measures vegetation vigor caused by chlorophyll activity. This attribute is often referred as “greenness” and is used by USDA officials to monitor vegetation conditions in croplands across the U.S. (Wade et al. 1994). The lack of green cover in regions of Iowa and Missouri dominated by annual croplands during the spring and fall can be observed in Figures 5-7b and 5-7d. Intensive row-crop production of annual croplands in Iowa during the summer growing season can be observed in Figure 5-7c. Perennial croplands, conversely, demonstrate a higher greenness index during the early spring and late fall (Figures 5-7b and 5-7d), the times of the year when annual croplands are bare, because perennial crop types can begin to grow as soon as environmental conditions (e.g., temperature) are favorable. This observation can be explained by the extensive root systems of perennial plants that survive the winter and are ready for regrowth in early spring (Figure 5-2) (Glover et al. 2007). Harvested perennial croplands can also
Figure 5-7. (a) The spatial distribution of annual and perennial cropland in the Mississippi River Basin and the spatial distribution of annual cropland, perennial cropland, and the Normalized Difference Vegetation Index in Iowa and Missouri for (b) April, (c) August, and (d) October 2007.

continue to grow after harvest and retain living cover on these croplands until the first freezing temperatures. The temperature gradient from southern Missouri to northern Iowa is an important factor that would stimulate the southern plants of Missouri to grow earlier in the Spring and later in the Fall. The areas of annual croplands in the southeastern most portion of Missouri and north
central Missouri, however, would also be green in April and October (Figures 5-7a, 5-7d) if temperature was the most important driver in these regions.

The ability to retain continuous living cover on the farm is a desirable trait for reducing N leaching when annual croplands are susceptible to excessive leaching (Jordan et al. 2007). The prospects of satisfactory economic returns from current perennial cropping systems, however, are an obstacle to perennial cropping systems (Randall and Mulla 2001). This obstacle may explain why the majority of perennial croplands are found on the fringe of the fertile corn belt, grain belt, and Mississippi River corridor (Figure 5-7a).

**CONCLUSIONS**

Intensive agricultural production, especially in areas with over 50% cropland cover, is strongly correlated with increasing river discharge (Figure 5-6). The increased discharge has a positive influence on the total export of nutrients from these agricultural watersheds. Increasing perennial land cover and decreasing annual land cover are associated with decreasing riverine NN concentrations by watershed (Table 5-1, Figure 5-5). The combined effect of annual agricultural croplands is to increase river discharge and nutrient concentrations, thereby increasing the total nutrient export. Croplands that mimic the physical structure and seasonal patterns of perennial farmlands, however, could reduce river discharge, nutrient concentrations, and total nutrient export.

The multiple regression analysis presented here shows that croplands planted in annual crops, but not perennial crops, are directly associated with higher NN concentration leaving agricultural landscapes. Perennial crops grow from early spring (a time of heavy rainfall) through the late fall (a time of heavy fertilizer applications). Nutrient uptake occurs throughout the long growing period, and the plant biomass above- and belowground reduces erosion, resulting in enhanced soil structure and function (Glover et al. 2007). Compared to annual
vegetation, therefore, the diverse and long-lived root structures of perennial vegetation utilize soil and water resources more efficiently and can reduce NN leaching through the soil profile (Ewell 1999, Glover et al. 2007) (Figure 5-2), thereby exhibiting lower riverine NN concentrations on the watershed scale.

A fundamental challenge of the 21st Century will be to simultaneously produce the basic needs for a growing human population while protecting ecosystem services that are necessary for the long-term sustainability of global food production systems. The solution to such a challenge will necessitate an agricultural system that can produce food and fiber with minimal energetic inputs and benefit, rather than impact, environmental conditions. Soil health and water quality are two interrelated environmental conditions that can be improved with perennial plant cover. This research presents evidence of the environmental benefits of perennial plant cover on riverine water quality and would recommend that land managers utilize perennial crop types in an effort to reduce the impacts of agricultural land use on water quality. The effect can be observed in riverine NN concentrations on the regional scale and could have implications for other water quality parameters.

ENDNOTES

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LITERATURE CITED


Chapter Six
A Deep Ecological Ethos for Environmental Science

INTRODUCTION

The field of environmental science has grown substantially over the last half century for several reasons, one of which is an innate connection to the natural environment on the part of environmental scientists. E. O. Wilson argued that human beings, in general, may have a “biophilia”, or an innate tendency to connect with life and life-like processes (Wilson 1986). The field of horticultural therapy uses the psychological effects of gardening as therapeutic treatments (Watson 1960). Outdoor education instructors use wilderness training as a means to rehabilitate troubled inner-city teenagers (Ford 1981). Journal entries from Henry David Thoreau (1854), John Muir (1911), and countless others testify to the effects of natural places on the human psyche. These examples share a common theme - humans possess an unexplainable desire to connect with the natural environment in a way that does not recognize the natural world as separate from oneself. It is from this innate desire that many environmental scientists pursue their field of study and work for a lifetime to this end. Here I argue that recognition of one’s connection to the natural world is a defensible foundation for an environmental ethos that can guide environmental scientists, including those working on the problems of our 10,000 year old use of land for agricultural purposes, that is the subject of this dissertation.

Perhaps the most difficult task one could undertake is to question the established, philosophical foundation of one’s own ethos. The essence of Deep Ecology, according to Bill Devall and George Sessions, is to do just this (Devall and Sessions 2005). It challenges humanity “to keep asking more searching questions about human life, society, and nature in the Western philosophical tradition of Socrates” (Devall and Sessions 2005, p. 200). It goes beyond
the limits of fact-based science and attempts to describe a comprehensive ethos and worldview that would more accurately be described as ‘ecological wisdom’. Arne Næss, Professor Emeritus at the University of Oslo and founder of the journal of philosophy Inquiry, coined the term “Deep Ecology” in 1973 to articulate this notion of ecological wisdom or equilibrium (Næss 1973). Here I will use the principles set forth by Næss, Devall, and Sessions to defend the use of Deep Ecology as an appropriate philosophical foundation for a responsible environmental ethos. This chapter is by no means exhaustive, but written as a supplement to the established readings on the subject.

WHAT IS DEEP ECOLOGY?

Næss claims the perspectives and foundations of Deep Ecology lie in the experience and intuitions of the ecologist (Næss 1995, p. 7; Devall and Sessions 2005, p. 201). This experience provides a unique description of the natural world that takes the biocentric approach of Paul Taylor (Taylor 1986) and others one step further. Næss explains that Deep Ecology rejects the image of man-in-environment as a part in a whole. Organisms, he continues, are knots in a “net or field of intrinsic relations.” Intrinsic relations, he explains, are necessarily included in the very definitions of the organisms. The field of ecology, by definition, is the study of these relationships. This is an important philosophical distinction of Deep Ecology. The very constitution of two things A and B cannot be completely realized without mention of the intrinsic relations that connect A and B. For Næss, this leads to a theory of thinking in terms of “gestalts” (Næss 2005, p. 193). Such a perspective, when questioned deeply, leads one to appreciate our complex experiences and lays the groundwork for the two foundational principles of Deep Ecology: biocentric equality and self-realization.

This axiom of biocentric equality is intuitively clear to Næss; thus, all beings have “the equal right to live and blossom” (Næss 1995, p. 4). The intrinsic value of all life is considered
one of two ultimate norms in Deep Ecology. Because it is a fundamental principle, there is no need to logically derive “biospherical egalitarianism”, as Næss calls it. It should be noted, however, that although Deep Ecology is fundamentally biocentric, it does not exclude non-living beings and relationships from moral consideration, as we shall see from the final development of Næss’ position. Indeed, equal consideration should be given to all beings, with which the greater Self identifies, seeing the mature Self in all things and all things in the Self.

This process of Self-realization is the final manifestation of the deep ecological foundation. Self-realization is the result of an expanded sense of identification beyond the boundaries of our own person and species. Næss contrasts this understanding of Self-realization with the shallower process of self-realization that is “more or less consistent egotism – the narrowest experience of what constitutes one’s self” (Næss 2005, p. 194). Here, Næss refers to the body, immediate family, or species, and draws a distinction between the greater “Self” and the narrow “self”. Identification is the “process through which the interest or interests of another being are reacted to as our own interest or interests” (Næss 2005, p. 194). According to Næss, the process of identification with the Natural world is a product of maturity, and it will result in the norms and conditions that describe the position of Deep Ecology. “Self-realization in its absolute maximum is the mature experience of oneness in diversity” (Næss 2005, p. 194). The resulting ethos obliges one to “live as if Nature matters” (Næss 2005, p. 194).

**Reasons to Support Deep Ecology as an Environmental Ethos**

There are several reasons why Deep Ecology is an appropriate philosophical foundation for a responsible environmental ethos that could guide ecologists and other environmental scientists. The first reason, previously mentioned, is the claim that all living beings are considered equal and equally possess intrinsic worth. This claim is considered an ultimate, fundamental norm, in line with other biocentric theories such as those set forth by Paul Taylor.
(Taylor 1986). The justifying reasons for granting intrinsic value to living beings is different, however, between Deep Ecology and biocentric theories. Taylor argues that we ought to grant intrinsic value to all living things because we understand the value of ecology and it makes sense to adopt an attitude of respect for everything that has a “good-of-its-own” (Taylor 2005, p.123-127). Deep Ecology agrees that we ought to grant intrinsic value to all living beings, but that we should do so because we identify all living beings through the process of identification presented above. Borrowing from the famous Kantian doctrine ‘never use a person solely as an end,’ Næss claims that identification tells us that if I have a right to live, then you have a right to live. This process, whereby we see ourselves as one in the same with other living beings, is the ultimate reason for environmentally responsible action. I should not save the river because it is a precious resource or even a necessary node in the web of life. I should save the river because I am part of the river, and because I identify with the river. Næss points out, “…if your Self in the wide sense embraces another being, you need no moral exhortation to show care. You care for yourself without feeling any moral pressure to do so…” (Næss 1988, p. 26-27). This produces a rich and passionate ethic that goes beyond the traditional limits of environmentalism to include many aspects of a morally responsible life.

A second reason why Deep Ecology is a sufficient ethical theory lies in Næss’ concept and promotion of the ‘total-field image’. Here, he goes one step further than most ecologists to include the existence of relationships in the very constitutions of organisms and natural objects. This is again a significant distinction between Deep Ecology and many other environmental ethics because Deep Ecology takes these intrinsic relationships to an existential level of importance, meaning the existence of organisms can only be fully understood when including the relationships that connect the organism to the rest of the world. With a richer appreciation for the connections, and unity, of living beings and systems, Deep Ecology facilitates the process of
identification and an environmental ethos that is more readily defended by its followers. A person is more likely to be concerned and called to act if he or she recognizes the necessary connections and relations that permeate all systems, and can then identify with the subject(s) in question. Recent discoveries in the fields of physics, chemistry, biology, and ecology can offer substantiated, reasonable support to these claims that all living systems are intricately connected and composed of the same materials (Capra 1997).

Thirdly, Deep Ecology is a normative doctrine that obliges all those who subscribe to its platform principles to act in such a way that would bring about the necessary changes to manifest these principles (Næss and Sessions 1995, p. 49-53). It is therefore an ethos of action, openly normative, and expresses a value priority system that goes further than other fact-based, scientific theories. Næss comments, “There are political potentials in this movement which should not be over-looked…” (Næss 1995, p.7).

Finally, the deep ecological perspective allows for regional and temporal differences; it encourages diversity in all scales of time and space. This principle is related to an appreciation for the complexity of ecosystems and the maxim that diversity increases the stability and resiliency of natural systems. Thinking in terms of vast systems, such as ‘gestalts’ coupled with the ‘total field image’, offers a theoretical framework that can meld and reasonably discern the incredibly diverse information being generated by the modern natural sciences. This perspective necessarily includes a dynamic concept of space and time, the ability to change the details of one’s position as the situation demands, all the while adhering to a unified, fundamental principle.

**Objections and Rebuttals**

There are a number of criticisms and objections to Næss’ proposed ethical system. One to consider is what I call the anthropocentric argument. One such argument by Richard Watson
claims that Deep Ecology seeks to remove the human element from the natural world and is therefore unduly anti-human, that is, it condones undue anti-anthropocentric attitudes (Watson 2005). This argument raises some legitimate concerns, but it misinterprets the process of identification and Self-realization. Watson argues that if one were to take a biocentric egalitarian approach on the matter, then one would not hold humans to different ethical standards than any other species. For Watson, there is no difference between the mass extinction that followed the evolution of plants, due to the resulting increase in atmospheric oxygen, and the mass extinction that is accompanying the evolution of humans. If we are truly egalitarian, he might say, then we must consider our actions, no matter what they are, to be simply natural and no more bound to moral considerations than the process of photosynthesis.

Næss would agree with Watson that humans are natural, i.e. an equal part of nature, and deserve to be treated on an even keel with all species. He would also agree that we ought not remove the human element from the natural world. He rebuts Watson, however, by arguing that humans may not know what is in the best interest for complex ecosystems, so it would be in the best interests for all concerned parties if humans practiced restraint when trying to control and dominate natural processes. This has become known as the “Precautionary Principle.” This point of view is rooted in an appreciation for the astounding level of complexity found in organisms and interactions in the biosphere. What follows is a perception, an acknowledgement, of “profound human ignorance” regarding these relationships and our ability to effectively ‘manage’ or even ‘facilitate’ them. Næss therefore proposes that we err on the side of humility and act with caution towards life traditions and our state of ignorance (Næss 1995, p. 5-6).

Watson might reply, ‘Why should we err on the side of humility and act with caution?’ If the motive of this action is the good of human beings, then this would be an anthropocentric ethos. Conversely, if we act for the good of other organisms, why should we demand that
humans respect these ethical principles when we do not expect other organisms to do the same? One response to Watson’s argument is that different organisms have different characteristics, and one of our distinctively human characteristics is to reflect on and care about ethical matters. This unique ability, if it is indeed unique, obligates humanity to an ethical responsibility that may not receive a reciprocal treatment. Deep Ecology does not advocate acting for the good of oneself, it advocates acting for the good of the whole.

Another principle used to defend Deep Ecology’s position regarding the role of humanity amid the Whole is the principle of biocentric equality, whereby one gains respect for the interests of all living beings. This interpretation of biocentric equality is superior to that of Watson because it identifies and empathizes with the interests and needs of other species and forms of life. Watson hides behind the veil of equality to promote what is more aptly called an enlightened anthropocentric approach, which places the needs and interests of humanity above all and is thus only indirectly concerned for non-human beings. To repeat, Deep Ecology does not call for a massive withdrawal from the natural world. To do so would surely mean the end of the human species. It acknowledges the fact that all living beings must intrude upon or feed upon other living beings (Naess 2005, p. 195). And the norms that follow simply ask humanity to live ‘lightly’ in nature with a minimum impact rather than a maximum impact on the surrounding world. Here, another aspect of the guiding principle evolves: simple in means, rich in ends (Devall and Sessions 2005, p. 202).

A second objection to consider is the claim that Deep Ecology lacks the scientific rigor of objective reason and relies too heavily on experiences and intuitions (some would include spirituality). As such, it is unfit to act as a philosophical foundation for any environmental ethic regarding reasoned philosophical ethics or science. To that, Devall and Sessions reply, “…the search for deep ecological consciousness is the search for a more objective consciousness and
state of being through an active deep questioning and meditative process and way of life” (Devall and Sessions 2005, p. 201). One problem with this objection is that it mistakes ethics for science. One could argue that we should base ethical principles on deeper, more intuitive commitments that simply cannot be justified in the same way that we validate scientific knowledge. Another problem with the objection is that it presupposes a debatable interpretation of science that is purely objective and rational. Science is a process and the methods of this process are constantly influenced by the practicing scientist. I have learned through the process of this dissertation that it is also possible to act intuitively and spontaneously while adhering to the scientific method. The objective collection of data from government archives in a systematic manner was a necessary feature of my experimental design. At the same time, countless footsteps and experiences in the agricultural fields of Kansas and Iowa were necessary to grasp the purpose and scope of the study as a whole-field *gestalt* and to interpret the results in a meaningful way that will lend itself to meaningful recommendations on the ground.

Næss, similarly, is calling for an ethos that goes beyond the traditional limits of environmental science to include an obligation to act in such a way that will bring about the necessary changes to manifest the principles of the ethical foundation. These changes challenge the philosophical, political, social, and scientific boundaries of the traditional Western defense of nature. “Deep Ecology goes beyond a limited piecemeal shallow approach to environmental problems and attempts to articulate a comprehensive religious and philosophical worldview” (Devall and Sessions 2005, p. 201). The ecological movement should be “ecophilosophical rather than ecological,” he explains (Næss 1995, p. 8). So he proposes a philosophy as a kind of *sofia* or wisdom; one that is openly normative. “It should contain norms, rules, postulates, value priority announcements and hypotheses concerning the state of affairs in our universe. Wisdom is policy wisdom, prescription, not only scientific description and prediction” (Næss 1995, p. 8).
It follows from this discussion that an adequate philosophical foundation is not restricted or selfish regarding individual species or academic fields, but, rather, inclusive and dynamic, able to change with the time and place.

**SIGNIFICANCE AND SUMMARY**

While I do not advocate that one is rationally obligated to adopt the perspective presented here, I do suggest that the principles of Deep Ecology offer an appropriate, reasonable, and enlightening philosophical foundation for my work because it builds on a broad perspective of ecology that recognizes the importance of interacting components, and intimately connected observers, in understanding the studied ecosystems. The Mississippi River Basin, one focus of this dissertation, is the largest watershed in the North America and its condition and use is an example of our relationship with the natural world. Acknowledging the ethical foundation presented in this chapter, the previous chapters of this dissertation examined the interrelationships among people, soil, and water in watersheds across the continental U.S. through the 20th Century.

The philosophical principles of Deep Ecology additionally call for normative action and informed prescription. What follows, for the environmental scientists, is an applied approach whereby the objective methodologies of data collection and analysis lead to prescriptive action. I argue that there is a time and a place for both aspects of this approach. The value of our ecological understanding of environmental problems cannot be underestimated. The prospects of not acting on this information, at the same time, cannot be underestimated. An ethos founded in deep ecological principles allows the environmental scientist to be both objective and prescriptive, to see the river simultaneously as a subject of study and a part of oneself.
ENDNOTES

1 Næss uses the tradition of *The Bhagavadgita* and the life/work of Mahatma Gandhi to describe the concept of “self” or *atman*. For further reading, see “Ecosophy T: Deep Versus Shallow Ecology,” p. 193-4.

2 I am indebted to Suzanne Duarte for her teachings on the process and fundamental position of Self-realization within Deep Ecology.

3 Normative ethics, such as Deep Ecology and Paul Taylor’s biocentric theory, seek to answer questions about which acts should be advocated and which acts should be prohibited. As such, they prescribe standards of behavior. An ultimate norm in this sense refers to a prescribed standard of behavior that is a final goal of the advocating ethos.

4 For further discussion on the role of modern science in the continuing development of the Western worldview, see Fritof Capra, *The Web of Life* and *The Tao of Physics*.

LITERATURE CITED


Chapter Seven
Summary and Synthesis

Rationale and Objectives

Agricultural cultivation now covers roughly one quarter of the earth’s land surface and has altered the world’s ecosystem structure and function more in the last 50 years than in the combined history of humanity (MEA 2005). Land use changes, particularly the conversion of natural ecosystems to cropland and the use of intensive management practices, have been the most important direct driver of change in terrestrial ecosystems over the last 50 years (MEA 2005). The great challenge of the 21st Century will be to re-evaluate a historical focus on agricultural productivity, and willfully choose a path that incorporates conservation and long-term sustainability (Jackson 1980). Soil health and water quality are two measures of sustainability that can be influenced by changes in agriculture.

This dissertation addresses Wes Jackson’s challenge by quantifying the relationships between agricultural land use and water quality in watersheds across the continental U.S. through the 20th Century. Information about this relationship will be useful for implementing changes to land use practices that could positively affect water quality conditions. I examined five basic research questions in light of this objective: 1) How have land use and water quality parameters changed over the course of the 20th Century, i.e., what is the historical context of our current situation? 2) What are the non-point sources of nutrient enrichment in our nation’s rivers and streams? 3) If agricultural land uses are driving this non-point source pollution, which agricultural and economic practices can be quantitatively associated with nutrient enrichment on the watershed scale? 4) Which agricultural and economic practices can feasibly reduce nutrient export from agricultural watersheds? and, 5) How influential is federal farm policy on land use
and, indirectly, water quality? Is there evidence that farm policy can be an agent of positive change to develop sustainable agricultural systems and improve water quality? The following is a summary and synthesis of the results and answers to these questions.

**GIS Database and Historical Trends in Land Use and Water Quality**

I created a deliverable, spatially-explicit GIS (geographic information system) database to qualitatively map historical land use patterns and to test the relationship between land use and water quality at the watershed level across the continental U.S., and throughout the 20th Century. I used Census of Agriculture data representing all 3078 counties in the U.S. in 20 censuses from 1840 to 2002. In-stream nitrate-nitrogen (NN) records measured in the beginning and end of the last century were included from Clarke (1924) and the U.S. Geological Service National Water Information System for 156 rivers and streams across the continental U.S. and Alaska.

The water quality data show that average NN concentrations have more than tripled in the last century across the U.S. The increase in concentration is particularly large in watersheds with more than 50% cropland, especially in the Midwest region. Agricultural cropland has also concentrated in the Midwest during this same time period. Corn and soybean plantings have increased at the expense of other grain crops such as wheat, barley, and oats resulting in a more homogenized landscape. This reduction in cropland diversity over the last half century was accompanied by a reduction in the number of farms, an increase in the average farm size, an increase in fertilizer application, and an increase in mechanical tillage resulting in more intensively managed landscapes.

**The Effects of Land Use Practices on Water Quality and the Role of Environmental Ethics**

Perennial cropping systems, agricultural production systems that mimic the structure and function of natural ecosystems have the potential to meet the demands for food and fiber while
simultaneously reducing the environmental impacts of agricultural production (Cox et al. 2006, Glover et al. 2007). The results of my research show that perennial plant cover is inversely related to NN concentrations by watershed, while annual crop cover is positively related to riverine NN concentrations. The robust root architecture of perennial crop types in addition to their longer growing season and relatively minor input requirements could explain this relationship (Glover et al. 2007). Additionally, agricultural cropland is strongly correlated with increasing river discharge at normal precipitation in the latter half of the 20th Century. In fact, annual discharge rates from the Mississippi River, on the order of the Rhone River (~50 km$^3$ yr$^{-1}$), can be attributed to agricultural practices (Raymond et al. 2008). This extra river discharge is in addition to the normal baseflow conditions. I argue that increasing river discharge coupled with increasing NN concentrations from agricultural cropland, and particularly annual cropland, can explain the significantly higher NN export to surface waters of the U.S. over the last century.

Intensively managed agricultural cropland, especially those areas dominated by corn plantings, is strongly correlated with higher riverine NN concentrations. This relationship was significant at the beginning of the 20th Century but was amplified in the latter half of the 20th Century indicating that some modern agricultural practices including commercial fertilizer applications, intensive tillage, and improved drainage, are significantly increasing NN concentrations in present-day surface waters. The baseline conditions, however, for NN export from minimally impacted watersheds have not changed over the course of the last century indicating that rehabilitation of in-stream water quality is possible.

One potential land use characteristic that could influence the rehabilitation of surface water quality is cropland diversity. Decreasing cropland diversity is strongly correlated with rising NN concentrations by watershed indicating that efforts to increase cropland diversity can effectively reduce nutrient loading to surface waters. This effect is due, in part, to a reduction in
corn agriculture and a subsequent reduction in fertilizer applications, which are the dominant drivers of the statistical relationships presented above. Increasing cropland diversity, however, could increase the areal extent of less intensive grain crops thereby reducing the net export of NN from agricultural watersheds. Increasing cropland diversity could also reduce the average field size and foster a more intimate relationship between the farmer and the farmland.

This relationship between people and the land is the focus of the Deep Ecological ethos presented in Chapter Two. Because humans have the capacity to discern ethical matters, and we are willfully choosing to act in such a way that may inflict harm on the natural environment, I argue that we are ethically obligated to consider the good of the living beings, and the natural world in general, when decisions and actions have environmental implications, either direct or indirect. I advocate an eco-centric ethical foundation to this end that allows the observer to maintain scientific rigor and identify with the surrounding natural world. This process of identification with nature can facilitate a positive influence on one’s ‘ecological footprint’. When considering a course of action that may impact the soil health on one’s farm or the water quality in the distant coastal zone, an intimate connection, or identification, with these natural systems can justify those actions that are environmentally responsible.

**FEDERAL FARM POLICY, CHANGING LAND USE, AND WATER QUALITY IMPLICATIONS**

Federal farm policy likely accelerated the process of consolidation and specialization of American croplands in the last century that indirectly increased average riverine NN concentrations. The indicators of these processes, especially decreasing cropland diversity, increasing farm size, decreasing number of farms, and increasing commercial fertilizer applications, can be quantitatively linked to government farm payments by watershed. Further,
government farm payments and each of these indicators can be statistically associated with rising riverine NN concentrations.

Figures 7-1 and 7-2 summarize and illustrate these results regarding the relationships between land use, water quality and the role of government farm payments. Rising riverine NN concentrations are associated with increasing cropland cover, corn cropland cover, cropland planted in annuals, increasing fertilizer use, and increasing government farm payments per acre of farmland. Lower riverine NN concentrations are associated with increasing cropland diversity and perennial farmland cover. Additionally, as government farm payments increase per acre of farmland, the average farm size and the use of commercial fertilizers appear to increase while the number of farms receiving payments and the diversity of the cropland tends to decrease.

I argue that it is possible to use the federal farm bill, the legislation that authorizes government farm payments, to increase cropland diversity, minimize the national specialization

Figure 7-1. A summary of the results and conclusions of this dissertation relating agricultural land use practices and riverine nitrate-nitrogen concentrations.
Figure 7-2. A summary of the results and conclusions of this dissertation relating governments farm payments to farm size, the number of farms in a given area, cropland diversity, commercial fertilizer applications, and riverine nitrate concentration.

of croplands, decrease the dependency on corn agriculture and commercial fertilizer applications, and increase perennial cropland cover in an effort to reduce nutrient enrichment of surface waters. There are many potential benefits to nutrient reductions including improved surface water and groundwater quality, wildlife heath, biodiversity, and soil quality (EPA 2008). If we make water quality improvement a goal and national priority, then agricultural land uses will inevitably be a focus of this endeavor. In other words, we will need to address the source of our water quality problems with an appropriate form of agriculture.

LITERATURE CITED


Appendix I
A List of the Original Sources of the Land Use Database

The following appendix describes the original sources of the Census of Agriculture data presented in this dissertation. The information was provided by Michael Haines, Department of Economics, Colgate University, who digitized and compiled the raw data.

1840 Data Set


1850 Data Set


1860 Data Set


1870 Data Set


1880 Data Set


1890 Data Set


1900 Data Set


1910 Data Set


1920 Data Set


1930 Data Set


1940 Data Set


1950 Data Set


1954 Data Set


1959 Data Set


**1964 Data Set**


**1969 Data Set**


**1974 Data Set**


**1978 Data Set**


**1982 Data Set**


**1987 Data Set**


**1992 Data Set**


**1997 Data Set**

2002 Data Set

Appendix II
A Sample of Maps Generated from the GIS Database

The follow appendix is a sample of maps generated from the GIS database that was created for this dissertation. The maps were designed at the request of various scientists whom I have collaborated with over the course of this project. The examples included here are a representative sample of the various mapping capabilities of the database.
1899
Harvested Cropland: Corn for Grain

Data Sources:

W. Broussard¹, R. E. Turner¹, and M. Haines²
¹Dept. of Oceanography and Coastal Sciences
Louisiana State University
Baton Rouge, LA 70803, USA
²Dept. of Economics
Colgate University
Hamilton, NY 13346, USA

Percent of County Area

No Data
0% - 5%
6% - 10%
11% - 15%
16% - 20%
21% - 25%
26% - 30%
31% - 35%
36% - 40%
41% - 45%
46% - 50%
51% - 55%
56% - 60%
1949
Harvested Cropland: Corn for Grain

Data Sources:

Percent of County Area

No Data
0% - 5%
6% - 10%
11% - 15%
16% - 20%
21% - 25%
26% - 30%
31% - 35%
36% - 40%
41% - 45%
46% - 50%
51% - 55%
56% - 60%

W. Broussard\textsuperscript{1}, R. E. Turner\textsuperscript{1}, and M. Haines\textsuperscript{2}
\textsuperscript{1}Dept. of Oceanography and Coastal Sciences
Louisiana State University
Baton Rouge, LA 70803, USA
\textsuperscript{2}Dept. of Economics
Colgate University
Hamilton, NY 13346, USA
2002
Harvested Cropland: Corn for Grain

Data Sources:
US Department of Agriculture, 2005.

Percent of County Area

No Data
0% - 5%
6% - 10%
11% - 15%
16% - 20%
21% - 25%
26% - 30%
31% - 35%
36% - 40%
41% - 45%
46% - 50%
51% - 55%
56% - 60%

W. Broussard*, R. E. Turner*, and M. Haines*
*Dept. of Oceanography and Coastal Sciences
Louisiana State University
Baton Rouge, LA 70803, USA
Dept. of Economics
Colgate University
Hamilton, NY 13346, USA

132
1969
Harvested Cropland: Soybeans

Data Sources:

Percent of County Area

No Data
0% - 5%
6% - 10%
11% - 15%
16% - 20%
21% - 25%
26% - 30%
31% - 35%
36% - 40%
41% - 45%
46% - 50%
51% - 55%
56% - 60%

W. Broussard¹, R. E. Turner¹, and M. Haines²
¹Dept. of Oceanography and Coastal Sciences
Louisiana State University
Baton Rouge, LA 70803, USA
²Dept. of Economics
Colgate University
Hamilton, NY 13346, USA
1909
Harvested Cropland: Wheat
1978

Harvested Cropland: Wheat

Data Sources:

Percent of County Area

W. Broussard1, R. E. Turner1, and M. Haines2
1Dept. of Oceanography and Coastal Sciences
Louisiana State University
Baton Rouge, LA 70803, USA
2Dept. of Economics
Colgate University
Hamilton, NY 13348, USA
Percent of County in Indian Corn, 1900
Census of Agriculture

Legend

- Interstate
- Highway
- State Boundaries

Indian Corn (ac) / Cnty Area (ac)

- 0.00% - 1.44%
- 1.45% - 4.02%
- 4.03% - 6.75%
- 6.76% - 9.57%
- 9.58% - 12.82%
- 12.83% - 17.09%
- 17.10% - 22.74%
- 22.75% - 29.01%
- 29.02% - 37.19%
- 37.20% - 52.01%

Data Source: ICPSR 0003
US Census Bureau, Census of Agriculture

Whitney Broussard, RE Turner, LSU
Jerry Glover, The Land Institute

155
2002 Farm Act Base Acres
Major 8 Commodity Crops
(C-8 Base Acres as % of Total Cropland)

Legend
- Major Rivers
- Commodity Base Acres / Cropland
- State Boundaries
- N.A. Boundaries

0% - 25%
26% - 50%
51% - 75%
76% - 100%
101% +

Projection: Albers Equal Area Conic
Data Source: USDA-ERS

W. Broussard, LSU
R. E. Tumer, LSU
J. Westra, LSU

165
2002
Net Cash Farm Income of Operations
per Acre of Farmland

Legend

- Major Rivers
- State Boundaries
- N.A. Boundaries

Net Income per Acre Farmland:
- $25 - $50
- $10 - $25
- $50 - $75
- $75 - $100
- > $150

Projection: Albers Equal Area Conic
Data Source: USDA Census of Agriculture

W. Broussard, LSU
R. E. Turner, LSU
J. Westra, LSU
2002
Percent of Net Farm Income from Government Payments

Legend
- Mississippi River Basin: Gov. Payments / Net Farm Income
- State Boundaries: < 1%
- N.A. Boundaries: 1% - 10%
- No Data

Projection: Albers Equal Area Conic
Data Source: Census of Agriculture, US Census Bureau
ICPSR 003, M. Haines
Compiled by: W. Broussard, LSU
R. E. Turner, LSU
J. Westra, LSU

170
1969
Farms Receiving Government Payments

Legend
- Mississippi River Basin
- 1 Dot = 200 Farms
- State Boundaries
- Government Payments (# Farms)

Projection: Albers Equal Area Conic
Data Source: Census of Agriculture, US Census Bureau
ICPSR 003, M. Haines
Compiled by: W. Broussard, LSU
R. E. Turner, LSU
J. Westra, LSU
2002
Farms Receiving Government Payments

Legend
- Mississippi River Basin
- 1 Dot = 200 Farms
- State Boundaries
- Government Payments (# Farms)

Projection: Albers Equal Area Conic
Data Source: Census of Agriculture, USDA
ICPSR 003, M. Haines
Compiled by: W. Broussard, LSU
R. E. Turner, LSU
J. Westra, LSU
1959 Census of Agriculture
Number of Acres per Operator

Legend
- Major Lakes: 719 - 954
- Major Rivers: 955 - 1,211
- State Boundaries: 1,212 - 1,543
- N.A. Boundaries: 1,544 - 2,005
- Acres per Operator
  - 0 - 70: 2,006 - 2,626
  - 71 - 135: 2,627 - 3,533
  - 136 - 193: 3,534 - 4,814
  - 194 - 257: 4,815 - 6,583
  - 258 - 333: 6,584 - 8,765
  - 334 - 427: 8,766 - 13,334
  - 428 - 550: 13,335 - 22,899
  - 551 - 718: 22,900 - 80,000

Projection: Albers Equal Area Conic
Data Source: US Bureau of the Census
Census of Agriculture, 1959

W. Broussard, LSU
R. E. Turner, LSU
1995
Water Use per Day
Total Consumptive Use (10,000 gal)

Legend
Water Use
Consumptive Use (10,000 gal)
0 - 1463
1464 - 4251
4252 - 8171
8172 - 14504
14505 - 24367
24368 - 40192
40193 - 64477
64478 - 112968
112969 - 180454
180455 - 265375
Major Lakes
State Boundaries
N.A. Boundaries

Projection: Albers Equal Area Conic
Data Source: US Bureau of the Census, 2001

W. Broussard, LSU
R. E. Turner, LSU
1995
Water Use per Day
Groundwater Withdraws (gal)

Legend
Water Use
GW Withdraws (gal)
0 - 950
951 - 2740
2741 - 5501
5502 - 9276
9277 - 14060
14061 - 21469
21470 - 31791
31792 - 54380
54381 - 103193
103194 - 175411
Major Lakes
State Boundaries
N.A. Boundaries

Projection: Albers Equal Area Conic
Data Source: US Bureau of the Census, 2001

W. Broussard, LSU
R. E. Turner, LSU
Appendix III
A Description of the 2002-2005 Government Farm Program Payments


Direct government payment amounts include cash payments made directly to farmers, not including Farmer-owned Reserve Payments as these data are not available by State. The amounts also include the net value of certificates.

Direct payments and counter-cyclical payments are authorized by the Farm Security and Rural Investment Act of 2002 for 2002 through 2007 crops. The Act also increases the number of crops authorized to receive payments.

Tobacco transition payments include both the Commodity Credit Corporation payments to quota holders and producers and the third party payments to quota holders and producers who opted for the lump sum payment option.

Conservation programs include: the Agricultural Conservation Program, the Agricultural Management Assistance Program, the Agricultural Management Assistance Program - NRCS, the Auto Conservation Reserve Program - Cost Shares, the Colorado River Basin Salinity Control Program - NRCS, the Conservation Reserve Program - Annual Rental, the Conservation Reserve Program - Cost Shares, the Conservation Reserve Program - Incentives, the Conservation Security Program, the Environmental Quality Incentives Program, the
Environmental Quality Incentives Program - NRCS, EQIP - Ground and Surface Water Conservation - NRCS, EQIP - Klamath Basin - NRCS, EQIP - 1996 Farm Bill, the Farmland Protection Program - NRCS, the Forestry Incentives Program - NRCS, the Grasslands Reserve Program, the Great Plains Program - NRCS, the Soil and Water Conservation Assistance Program - NRCS, the Wetlands Reserve Program, the Wetlands Reserve Program - NRCS, and the Wildlife Habitat Incentive Program - NRCS.

Ad Hoc and emergency programs includes all programs providing disaster and emergency assistance payments to growers. Programs include: the Cottonseed Payment Program, the Crop Disaster Program, the Crop Disaster Program - North Carolina, the Crop Disaster Program - Virginia, the Crop Disaster Program 2005, the Crop Disaster Assistance Program 2001/2002, the Dairy Indemnity Program, the Dairy Marketing Loss Assistance Program, the Disaster Program, the Emergency Conservation Program, the Emerging Markets Program, the Feed Indemnity Program, the Florida Hurricane Citrus Disaster Assistance Program, the Florida Hurricane Nursery Disaster Assistance Program, the Florida Hurricane Vegetable Disaster Assistance Program, the Hurricane Indemnity Program, the Lamb Meat Adjustment Assistance Program, the Livestock Assistance Program, the Livestock Compensation Program, the Livestock Emergency Assistance Program, the Livestock Indemnity Program, the Market Loss Assistance Program, the Noninsured Assistance Program, the Nursery Losses Program - Florida, the Quality Losses Program, the Sugar Beet Disaster Program, the Trade Adjustment Assistance Program, and the Tree Assistance Program.

Miscellaneous programs include Acreage Grazing Payments, Additional Interest Payments, the American Indian Livestock Feed Program, the American Indian Livestock Feed Program - Apportioned, Direct payments and counter-cyclical payments - Fruit and Vegetable Violations, Direct payments and counter-cyclical payments - Late Filing Fees, Direct payments
and counter-cyclical payments - Payment Limitation Overpayment, the Feed Grain Deficiency Program, the Hard White Winter Wheat Program, the Interest Payments, and the Wheat Deficiency Program.
Vita

Whitney P. Broussard III was born in Lafayette, Louisiana, and raised in Lake Charles, Louisiana. He attended grade school at Immaculate Conception Cathedral School and graduated from Alfred M. Barbe High School in 1996. He pursued studies in philosophy and the humanities at Christian Brothers University in Memphis, Tennessee, from 1996 to 1998 where he volunteered much of his time at the M. K. Gandhi Institute for Nonviolence. He began his studies of the environment in 1998 at Naropa University in Boulder, Colorado, where he lived and worked on the University’s Hedgerow Farm. He graduated from the University in 2000 with a Bachelor of Arts degree in environmental studies with concentrations in horticulture and Native American studies and a minor in traditional Eastern arts. He was the greenhouse manager at the local nursery in Lake Charles for the next year and worked as a horticulture student at Dechen Chöling Shambhala Center in Limoges, France, in the Spring of 2001. He entered the University of Louisiana at Lafayette in the Fall of 2001 and focused on soil and water science where he worked as a research assistant monitoring the field-level effects of agricultural practices on water quality conditions in the Vermillion, Teche, and Mermentau River Basins. He graduated Summa Cum Laude in 2003 with a Bachelor of Science degree in environmental and sustainable resources. Whitney was awarded a Louisiana Board of Regents fellowship to pursue doctoral studies in the Department of Oceanography and Coastal Sciences at Louisiana State University (LSU) in the Fall of 2003 where he continued his study of agricultural land use practices and their effects on water quality, using a watershed approach on large temporal and spatial scales. His doctoral work has also been funded by a Graduate Research Fellowship from The Land Institute in Salina, Kansas, a Teaching Assistantship from the Department of Oceanography and Coastal Sciences at LSU, and a Dissertation Fellowship from the Graduate
School at LSU. He studied landscape ecology and the philosophy of science as an international exchange student at Utrecht University in Utrecht, The Netherlands, in the Fall of 2005, and received an academic scholarship to study aquatic ecology at Iowa State University in Lake Okoboji, Iowa, in the Summer of 2006. Whitney was awarded The Joseph Lipsey, Sr., Memorial Scholarship Award from the Department of Oceanography and Coastal Sciences in the Spring of 2008, and will receive a Doctor of Philosophy degree in oceanography and coastal sciences with a minor in philosophy and religious studies from LSU on August 7, 2008, with a 4.0 GPA.