Historical Changes in Water Quality and Land Use in the Cedar River Basin, IA: Implications for Land Management Activities to Increase Denitrification Rates

Final CRISP Report to the Central Region of the U.S. Geological Survey

By

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Executive Summary

The U.S. Geological Survey has documented historical increases in nitrate levels in the Missouri and Mississippi River Basins. These increased nitrate concentrations have been implicated in the Gulf Hypoxia Syndrome (low dissolved oxygen zone) that threatens valuable marine fishery resources. High levels of nitrate result from a combination of factors including agricultural expansion and increased nitrogen application rates. These factors are exacerbated by land alterations including loss of riparian corridors, wetland drainage, and widespread use of tile drain systems. These alterations have basically altered the functional capabilities of the watershed for nitrogen assimilation, retention, and denitrification.

The U.S Geological Survey has extensive scientific capabilities in the areas of hydrology, water quality assessment, mapping, landscape analysis, biological assessment, and modeling that are critical in development of comprehensive efforts to understand and manage nitrate losses from Midwestern agricultural ecosystems. The Cedar River Basin of eastern Iowa has been identified as a Midwestern agricultural watershed with particularly high levels of nutrients. In addition, the Cedar River Basin is a major focus of the USGS National Water Quality Assessment Program (NAWQA; Eastern Iowa NAWQA Region). Furthermore, the lower reach of the Cedar River between Waterloo and Cedar Rapids, Iowa has been listed on the Iowa Impaired Waters List for high nitrate and bacteria concentrations. The first NAWQA cycle collected considerable spatial and temporal nutrient data that can serve as a data platform to explore testable hypotheses in relation to the fate and effects of nutrients in agricultural ecosystems. In recent years synoptic sampling studies and time of travel work through the USGS cooperative program have added to the knowledge of the Cedar River basin. To leverage these datasets, and to create integrated research opportunities within the USGS, the Central Region of the U.S. Geological Survey provided funds via the Central Region Integrated Science Program (CRISP) to conduct an assessment of the Cedar River Basin and factors related to elevated nitrate concentrations in streams. There were three objectives of this study: 1) Determine historical changes in land use in relation to watershed characteristics; 2) Assess historical changes in water quality that have occurred due to changes in land use, and 3) Determine the denitrification potential of soils among dominant microhabitats of the region.

Results indicate that dramatic changes have occurred in agricultural land use practices in the Cedar River Basin. The Basin is composed of 26 counties with a total human population of approximately 800,000 people. The human population has remained relatively stable over the past 30 years but has shifted from smaller towns to larger cities. The Cedar River Basin covers approximately 4.4 million acres in eastern Iowa and southern Minnesota. The total number of farmed acres has remained relatively constant at approximately 90% of the total acreage. However, dramatic changes in farming practices have occurred. The total number of farms has decreased due to the consolidation of small farms into larger farms. Corn and soybean acreages have increased while pasture and other crops have decreased. Nitrate levels in streams have increased steadily over the past 30 years in association with increased nitrogen fertilizer application. Modeling indicated that approximately 400,000 acres, or approximately 9% of total acreage, has denitrification potential (total 7.91% agricultural; 0.66% bottomland forest; 0.42% grassland; 0.13% backwater sloughs; and 0.13% water). Measurements of denitrification rates indicated that the highest potentials occur in the bottomland forest and grassed waterway habitats. However, denitrification rates varied widely within habitat types.

The Conservation Reserve Program (CRP) is intended to remove marginal land from production to decrease commodity production and provide water quality benefits. However, water quality benefits have not been realized. Thus, in the absence of dramatic changes in farming practices, it is doubtful that desired decreases in nitrate concentrations will occur. However, the results of this CRISP study provides a dataset to promote discussions and future research projects among various agencies including the USGS, USDA, USEPA, and the State of Iowa to re-examine land use activities and practices in relation to nitrate reduction approaches.

Chapter 1: The Cedar River Basin: Identifying Areas With Denitrification Potential

By

Pamela Waisanen, Kristine Verdin, Douglas Schnoebelen, James Fairchild, and Susan Greenlee, and Stephen Kalkhoff

Abstract

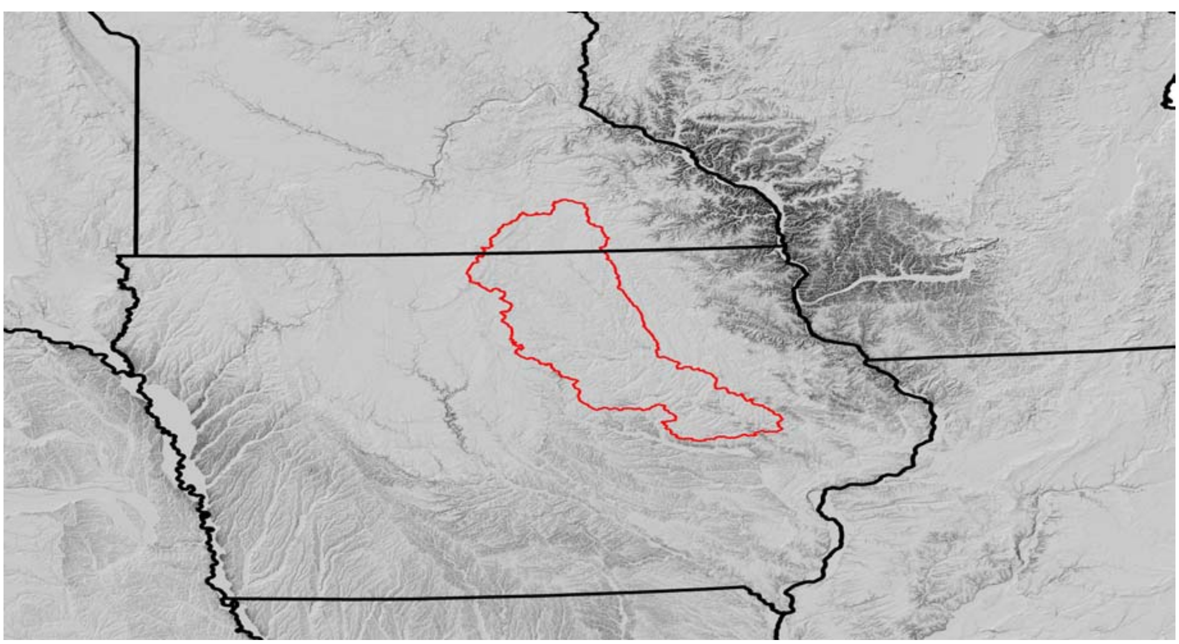
Streams in the Cedar River Watershed, located in eastern Iowa and southern Minnesota, contain some of the highest concentrations of nitrate in the nation. This research, funded by the U.S. Geological Survey's Central Region Integrated Partnership Study (CRISP) Program, had two objectives: 1) To establish land-use history, mainly from Census data, of those variables having the greatest affect on water quality in the study area, and 2) To develop a procedure for identifying the lands with possible denitrification potential. We developed procedures for delineating drainage basins upstream from any given point from the Elevation Derivatives for National Applications dataset. Within these watersheds we compared basin characteristics such as land use, slope, proximity to streams, and soil properties with water quality of streams draining the basins to more precisely predict where higher nitrate concentrations are likely to occur. Results indicate that dramatic changes have occurred in agricultural land use practices in the Cedar River Basin. The Basin is composed of 26 counties with a total human population of approximately 800,000 people. The human population has remained relatively stable over the past 30 years but has shifted from smaller towns to larger cities. The Cedar River Basin covers approximately 4.4 million acres in eastern Iowa and southern Minnesota. The total number of farmed acres has remained relatively constant at approximately 90% of the total acreage. However, dramatic changes in farming practices have occurred. The total number of farms has decreased due to the consolidation of small farms into larger farms. Corn and soybean acreages have increased while pasture and other crops have decreased. Nitrate levels in streams have increased steadily over the past 30 years in association with increased nitrogen fertilizer application. Modeling indicated that approximately 400,000 acres, or approximately 9% of total acreage, has denitrification potential (total 7.91% agricultural; 0.66% bottomland forest; 0.42% grassland; 0.13% backwater sloughs; and 0.13% water). Identification of these areas with denitrification potential is critical in determining which lands could be purchased or managed to provide the greatest water quality benefits.

Introduction

The Cedar River extends from southeastern Minnesota through eastern Iowa to the Mississippi River (fig. 1.1). A 57-mile segment of the Cedar River upstream of Cedar Rapids is listed on the Iowa 303(d) list as impaired by fecal coliform and nitrate-nitrogen (Iowa Department of Natural Resources, 1999, p. 172). Increasing trends in nitrate concentrations in the river have been observed in the last 30 years (Schnoebelen and others, 1999) due to a combination of factors including increased rates of nitrogen application, wetland drainage, and loss of forested riparian corridors. These trends have raised concerns over the quality of water provided to the city of Cedar Rapids.

Much of the excess nitrogen from intensive farming in the Midwest is transported downstream to the Gulf of Mexico (Alexander and others, 2000) where it contributes to the Gulf Hypoxia Syndrome (Rabalais and others, 1996). Numerous land management agencies are considering various strategies for reducing nitrate losses from agricultural areas of the Midwest. Mitsch and others (1999) recommended that 24 million acres of wetlands and riparian buffers be restored, in addition to other farming control practices, to reduce the nutrient loads that ultimately are transported to the Gulf of Mexico. This strategy would serve to promote nitrogen uptake, retention, and removal. The most effective locations for wetland and buffer restoration would be along rivers in watersheds that discharge high amounts of nitrogen near subsurface drainage systems (Mitsch and others, 1999). Denitrification potential is suspected to be greatest in these areas because they optimize conditions (saturated, anoxic conditions containing high levels of organic carbon) for microbial reduction of nitrate (Atlas and Bartha 1993). For example, Mitsch and others (1999) estimated that the Raccoon River Basin in Iowa discharges 27,520 metric tons of nitrogen per year; they estimated that 4%, 10%, and 14% of the 8,912 km^2 watershed would need to be restored to functioning wetlands to reduce nitrogen levels by 20%, 50%, and 70%, respectively. Wetlands may be used to help mitigate nitrate removal for tiles draining fields. The success of nitrogen removal by wetlands is dependant, however, on many factors including hydrology, topography, and economic and political factors. Such rehabilitation efforts are not likely to occur until mechanistic, sound approaches are developed to ensure the success of wetland restorations and other land management activities intended for nutrient reduction.

Figure 1.1. The Cedar River Basin Study Area in relation to regional topographic relief.



Geographic Information Systems (GIS) are tools that can be used to quantify land use changes, landform, and economic data in a spatial and temporal format that allows one to effectively evaluate the potential for land management decisions. In this paper, we apply GIS techniques to evaluate land use changes in the Cedar River Basin that have led to the current elevated nitrate conditions in streams. In addition, we use GIS tool to evaluate areas of greatest denitrification potential in order to determine the economic feasibility of effectively reducing nitrate loads via microbial nitrate reduction.

Background on Historical Changes in Land Use and Agronomics

Land use and cover are critical elements that affect water quality. We used U.S Bureau of Census data to reconstruct a generalized history of the area that may lead to better understanding of how land use and land cover changes may have affected water quality in the Cedar River Basin.

The Cedar River Basin study area covers approximately 4.4 million acres of Eastern Iowa and Southeastern Minnesota. Most of the land has been farmed intensively from the time of European settlement which began in the study area during the middle of the 19th century (Waisanen 2003, and references therein). The National Land Cover Data (U.S. Department of the Interior 1999) indicates that 85% of land use and cover was agricultural in 1992 (Table 1.1).

Table 1.1. Land Use and Land Cover in the Cedar River Study Area (National Land Cover Database, U.S. Department of the Interior 1999)

National Land Cover Classification	Area (Square Miles)	% Study Area
Open Water	56	0.81%
Low Intensity Residential	67	0.96%
High Intensity Residential	42	0.60%
Commercial/Industrial/Transportation	122	1.76%
Bare Rock/Sand/Clay	1	0.01%
Quarries/Strip Mines/Gravel Pits	5	0.07%
Deciduous Forest	268	3.85%
Mixed Forest	4	0.06%
Grassland/Herbaceous	356	5.12%
Pasture/Hay	468	6.73%
Row Crops	5,406	77.75%
Small Grains	6	0.08%
Urban/Recreational Grasses	23	0.33%
Woody Wetlands	62	0.89%
Emergent Herbaceous Wetlands	69	0.99%
Total Study Area	**6,953**	**100.00%**

Prior to European settlement the Basin was primarily covered with tallgrass prairie. Poor drainage, from saturated soils and prairie wetlands, hampered the conversion on the Des Moines Lobe to agriculture (Waisanen and Bliss 2002). By 1900, however, open ditches and subsurface tile drains overcame the drainage limitation. The acreage of land in farms that was drained was inventoried for the years 1920, 1930, and 1969. The proportions of drained land in farms for years 1920, 1930, and 1969 were 23%, 24%, and 31% respectively, for the 26 counties of which a portion was contained in the study area boundary (U.S. Bureau of the Census, 1922; 1932; 1973; 1978). Apart from those county data, the history of drainage has largely been lost. Drainage plat maps circa 1880 may exist in individual county engineering or auditor offices (Asell 2003). A current digital accounting of tile drainage would be a valuable input for modeling water-quality conditions and for determining where water will pool for denitrification purposes; however a consistent digital reconstruction of drainage would be time-consuming to produce

Recent Human Demographics

Urban lands in the Cedar River Basin occupy a relatively small percentage of the study area, but impact water quality. Runoff from impervious surfaces in urban areas carries many contaminants, including nitrates from human sewage, lawn fertilizers, and household products. In addition to being a source of nitrates, urban lands have supplanted a natural landscape that could have denitrification potential. Population in the Cedar River study area counties grew from 1970 to 1980, decreased from 1980 to 1990, and increased again from 1990 to 2000 (Tables 1.2 and 1.3). The distribution of these changes was uneven. The majority of losses occurred in most decades for the smaller towns, which are classified as non-metropolitan counties (e.g. not adjacent to larger cities) with a population of 2,500 to 19,999 (Figure 1.2). Population in small towns next to larger cities (non-metropolitan counties with a population from 2,500 to 19,999) tended to lose populations from 1980 to 1990 only. The two metropolitan areas lost population from 1980 to 1990, but gained in other decades.

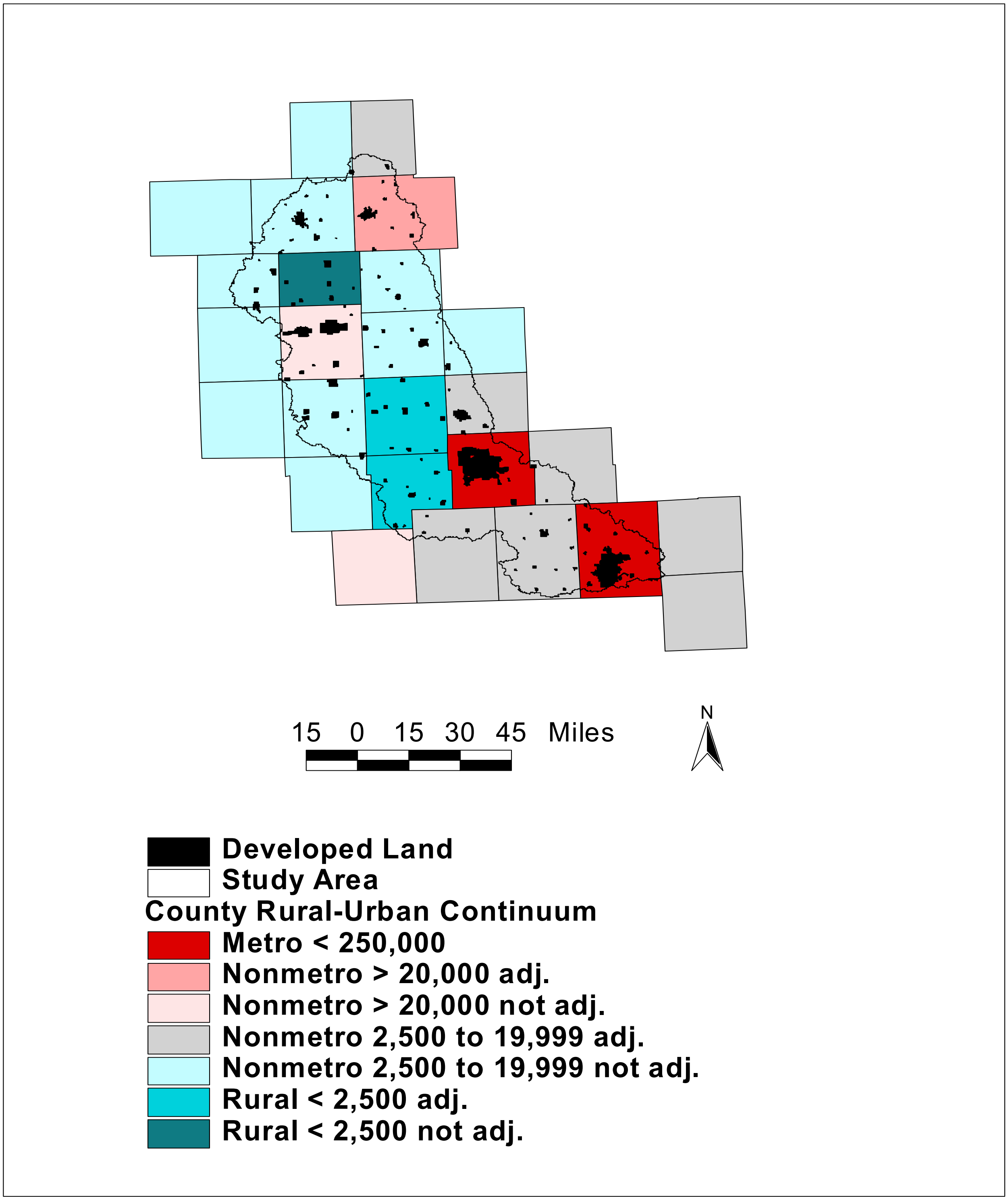
Table 1.2. Population for the 26 Counties over 3 decades.

1970	1980	1990	2000
822,615	835,764	778,328	814,612

Table 1.3. Population Changes for the 26 Counties over 3 decades.

1970 to 1980	1980 to 1990	1990 to 2000
13,149	-57,436	36,284

Figure 1.2. Cedar River Rural-Urban Counties and Urban Areas.

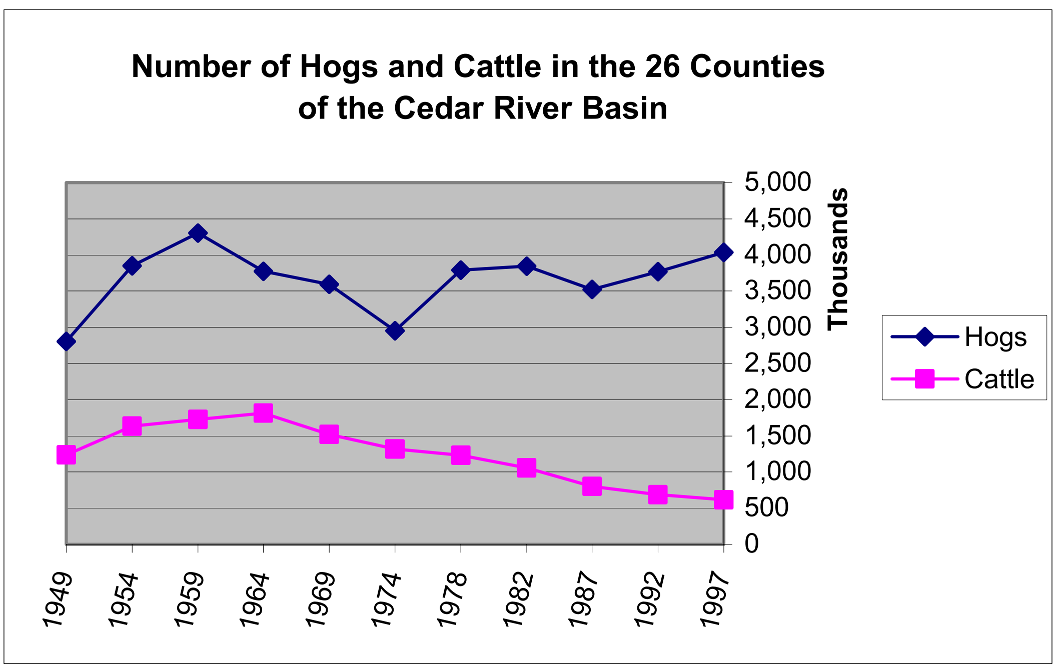


Changes in Hog and Cattle Numbers

The trends for hog numbers in the Cedar River Basin have varied in the last 50 years (Figure 1.3). With the specialization that is occurring, there are more hogs on fewer farms. The numbers from the last two inventories, although less than the 1959 peak, show that hog production is increasing. During the last 20 years, hog production has been moving to more remote locations with few people, both inside and outside of the Corn Belt (Roe et al 2002). Larger pork production facilities have also been expanding in non-metropolitan areas in the last decade (Sharp and others, 2002). Whether or not these shifts are occurring in our study area would be difficult to verify, because no consistent inventories of livestock exist below the county level (changes within a county could mask the migration of animal confinement operations to counties outside of the study area).

Cattle production, unlike hog production, is decreasing in the study area (Figure 1.3). It peaked in 1964, and has steadily declined since that time. Cattle production has remained on the perimeter of where corn would grow and has shifted to the west in association with irrigation expansion (Hudson 1994).

Figure 1.3. Changes in the Numbers of Hogs and Cattle in the Cedar River Basin from 1949 to 1997 (Waisanen 2003)



Changes in Land Use and Cover

The land in farms in the 26 counties in the Cedar River study area declined slowly over the past 50 years (Figure 1.4). A gradual decrease in farmland has occurred, due to a combination of factors that include urbanization and the purchase of lands for natural resource management. Within farmland, there has been a more dramatic shift. Cropland in the 26 counties has slightly increased in terms of acreage, while pastureland decreased, which is reflected in Figure 1.3 in the decreasing number of cattle. Much of the historical pastureland has been converted to cropland as a result of crop subsidies. Farm policy also encouraged marginal lands to be set aside for conservation (Waisanen 2003).

The number of acres of cropland and pastureland with commercial fertilizer applied has increased from 1954 to 1997 (Figure 1.4). This does not take into account applications to farmland that is not pastureland or cropland (for example, to a lawn), or to lands that are not in farms. It does, however, give an idea of the trends over time for applying commercial fertilizer. The number of cropland acres treated with fertilizer peaked in1978, and increased again in 1992.

Major changes in cropping patterns have also occurred in our study area. In the Western Corn Belt Plains Level III ecoregion (Omernik 1999), an area inclusive of the Cedar River Basin study area (shown in yellow in Figure 1.1), there were diverse crop rotations. Oats, barley, alfalfa, sorghum, corn, and soybeans were commonly grown; however, these diverse crops were replaced with a two-crop rotation of corn and soybeans (Figure 1.5). The recent high percentage of cropland dedicated to corn and soybeans indicates an intensification of cropping and a shift away from crop rotations that were more common in previous years (Waisanen 2003).

Changes in cropping practices are the result of a combination of factors resulting from evolving agricultural support programs such as the Farm Bill and Conservation Reserve Program (CRP). These programs have been driven by a combination of economic, supply, and conservation concerns that have led to fluctuating incentives for farmers. For example, CRP peaked in the 144 Western Corn Belt Plains counties in 1993 at approximately 2.6 million acres, but decreased to 1.8 million acres by 2000. The recently observed pastureland decreases and increased soybean acreage may be a result of the 1996 Farm Bill that subsidized the conversion of pastureland to cropland. To make up for lost productivity from lands set aside in CRP and to take advantage of crop subsidies, many producers are cropping marginal farmland with soybeans (referred to as the "slippage" phenomenon). This policy negates the environmental benefits of dollars spent for CRP. Wu (2000) estimated 30 acres of slippage in Iowa and 16 acres of slippage in Minnesota for every 100 acres of land set aside for the CRP.

Figure 1.4. Land in Farms, Cropland, and Pastureland, and Acres of Cropland and Pastureland with Commercial Fertilizer Applied in the Cedar River Study Area, 1949 to 1997. The 1969 and 1974 counts for acres of cropland and pastureland with commercial fertilizer applied were for class I through IV farms that account for 97.2% and 98.7% of farmland in 1969 and 1974, respectively. These acres were for farms with annual sales that exceeded $1,000.

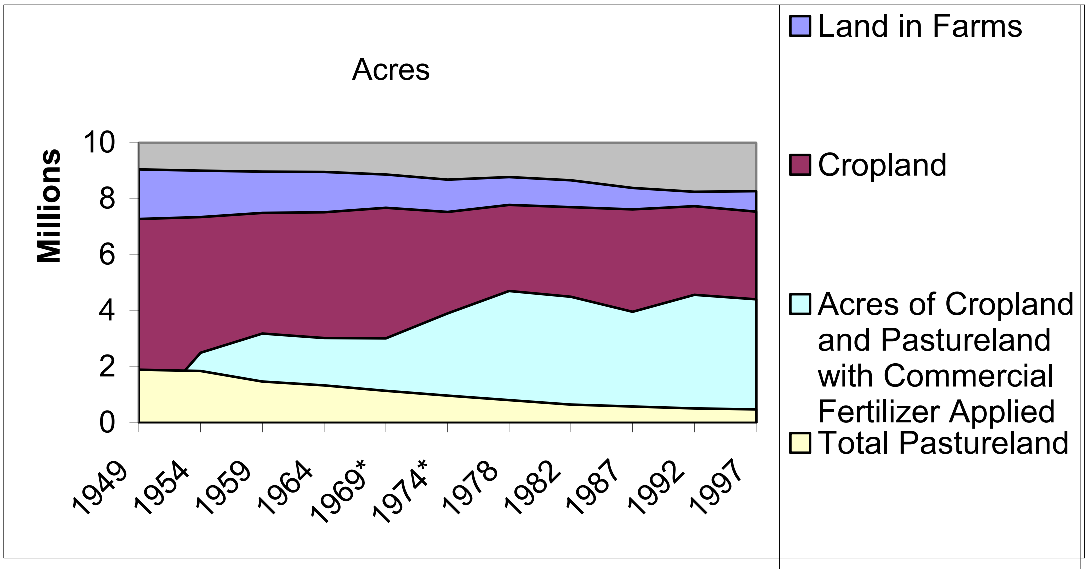
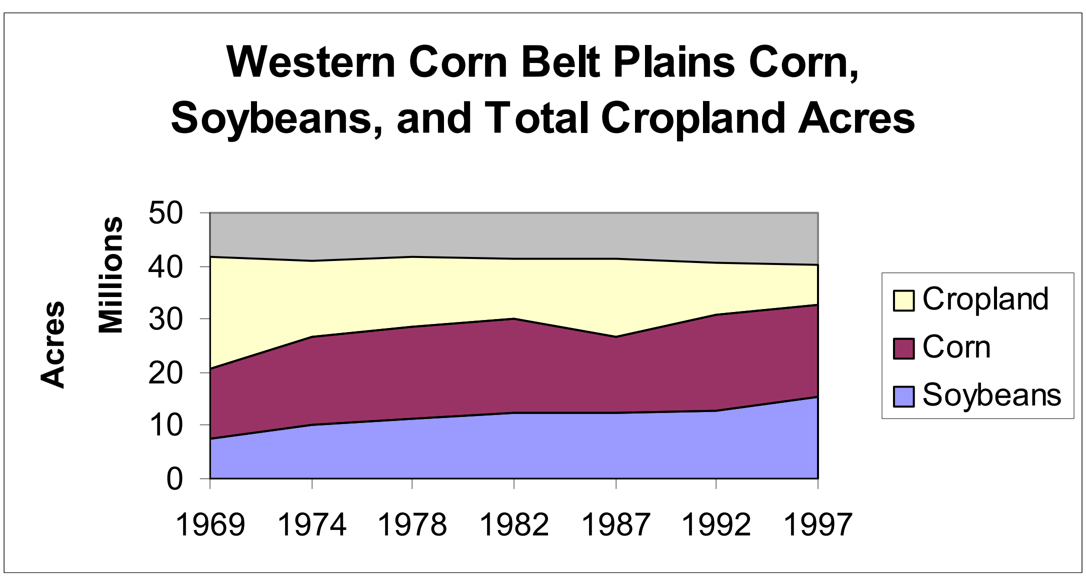


Figure 1.5. Corn. Soybean, and Cropland Acres in the Western Corn Belt Plains, 1969 to 1997.



Methodology

We used a series of GIS applications to determine the location, amounts, and percentages of land in the Cedar River Basin that would exhibit possible denitrification potential. Research indicates that denitrification potential is highest in areas along streams that are frequently saturated during the growing season. This results in high production of organic material that eventually enters into the detrital pool. High respiratory demand of decomposing organic matter results in conditions necessary for denitrification; saturated conditions, high organic matter, and anoxic conditions. Under these conditions, denitrifying bacteria can grow and biologically reduce nitrate to gaseous nitrogen via a series of chemical conversions.

There are several soil and microhabitat types that theoretically exhibit conditions and attributes that will support denitrification, such as bottomland forest wetlands and grassed waterways. A recent study demonstrated that bottomland forest soils and grassed waterways, indeed, have higher potential for denitrification compared to other dominant habitats on the landscape such as agricultural, riverine, and slough habitats (Fairchild and others, 2003). Thus, from a land-management perspective, it is important to determine the location and frequency of occurrence of bottom land forest and grassed waterway habitat in the landscape in order to determine denitrification potential and land management strategies for the Basin.

In order to understand why the water quality varies from site to site, natural and anthropogenic factors in a watershed that affect the quality of water needed to be quantified. The first step in quantifying these factors was to define the area (basin) that is upstream of 21 sampling sites. We intersected the sample points with Elevation Derivatives for National Applications (EDNA) synthetic streams that were derived from the National Elevation Dataset. The advantage of using these streams is that drainage areas upstream or flowpaths downstream from any given point can be traced. We ran upstream traces to delineate the drainage area for each point. The total drainage area boundary for the 21 points was then used to clip out critical inputs for determining possible denitrification potential. The National Land Cover Data (Table 1.1) is a key input. The NLCD was converted to a vector format, and reclassified to match the five microhabitat types (Table 1.4). A visual inspection of the “upland forest” category from the NLCD, draped on an EDNA shaded relief, confirmed that the forests along the river were bottomland forests. These forests were reclassified as “woody wetlands.”

Table 1.4. Relation Between NLCD and Microhabitats

NLCD Code	NLCD Land Use/Cover Description	Microhabitat Description	Microhabitat Code
11	Water	River	5
21,22,23, 85	Urban Development	Excluded	
31, 32	Barren and Mining	Excluded	
41, 42,92	Upland Forest	Woody Wetlands	2
71	Grasslands	Grasslands	1
81-83	Agricultural Lands	Agricultural Lands	4
92	Emergent Herbaceous Wetlands	Slough	3

Another critical data set for identifying HDP is the soils data. Because county soils data were not available for all counties, state soil data from the USSOILS coverage for the Upper Mississippi Basin were used (U.S. DOI 1995). The hydric soils, drainage, and permeability attributes were used to identify poorly drained soils, by the following query:

Select hydgrp > 2.5 and drain > 4.5 and perm < 2

The ranges in the study area for the hydric soils were 1.6 to 4 (with 4 being poorly drained). The ranges for the "drain" attribute were 2.3 to 6.9 (with 6.9 being the most poorly drained). The permeability attribute range was 0.71 to 8.68, and is expressed in inches of permeability per hour. The STATSGO data have been generalized, and therefore, indicate a probability of HDP, rather than an exact location. The reclassified land cover was overlaid on the STATSGO layer. The land cover/soils layers were queried for soils with low permeability and poor drainage, to find areas suitable for wetland restoration of bottomland forest or grass waterways.

The Elevation Derivatives for National Applications (EDNA) streams, which are synthetic flow lines generated from 30-meter Digital Elevation Models from the National Elevation Dataset, were buffered (Figure 1.6) using Pfafstetter codes (Verdin and Verdin 1999). These codes facilitate the connection of the EDNA drainage network. The following buffer widths were used (Table 1.5). A Type 1 stream is buffered by approximately a section on either side, while a Type 4 stream is buffered by a quarter-section on either side. The land use and soils were clipped by the buffered areas. For the last step of the analysis, 30-meter EDNA slopes were queried for slopes greater than or equal to five percent. These slopes were subtracted from the five land use and soil categories that met the drainage criteria.

Figure 1.6. Buffers with Widths Based on the Pfafstetter Codes

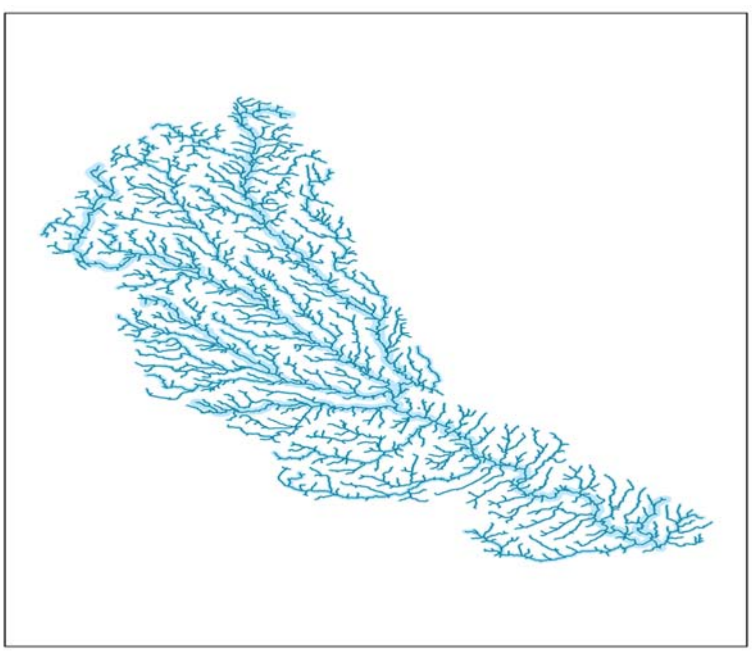


Table 1.5. Buffer Widths Applied to Streams.

Pfafstetter Type	Buffer Distance (square meters, on one side of stream)	Square Miles	Comparable Surveying or Economic Unit
1	1700	1.06	(1 section)
2	900	0.56	
3	500	0.31	
4	400	0.25	(1/4 section)
5	300	0.19	
6	200	0.12	

Results and Discussion

The results of our analysis are provided in Table 1.6 and Figure 1.7. The Cedar River Basin covers approximately 6,953 square miles of land in the Western Corn Belt Plains ecoregion that drain into the 21 sample points. Using the model described above, which evaluated soil moisture, organic carbon, and available data, we determined that 9.25% of the basin was categorized as having possible high denitrification potential (HDP). These lands are suspected of promoting conditions that promote microbial denitrification processes: long periods of saturation; high organic matter; and anoxic conditions.

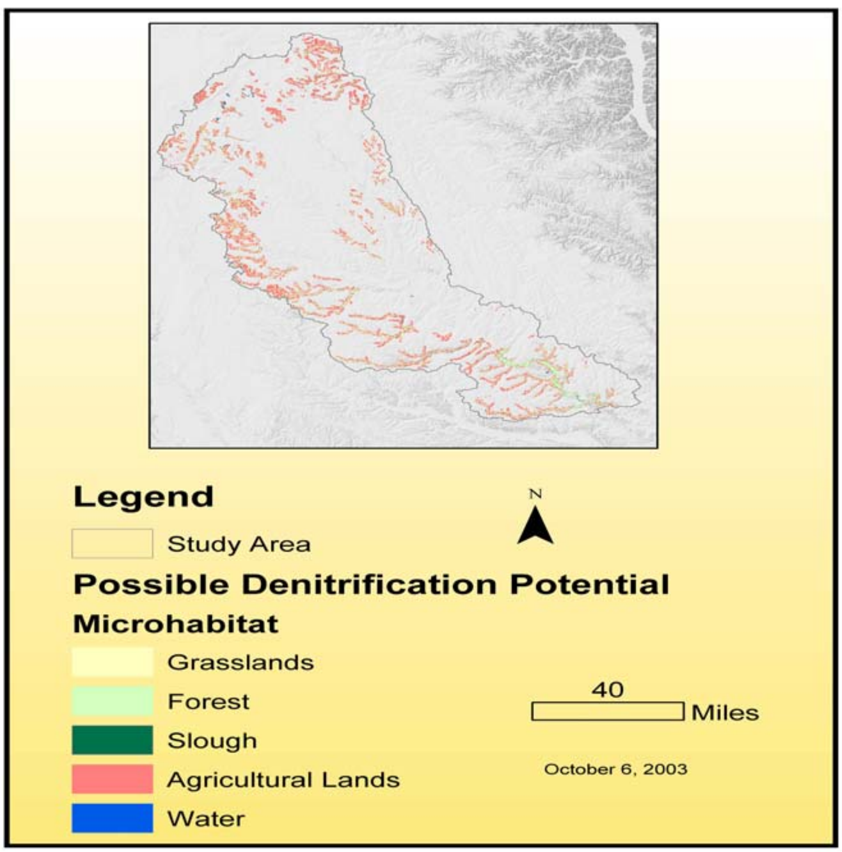
Land that was cropped in 1992 comprised the largest category of HDP (7.91% of the basin). The remaining categories accounted for less than one percent each of the total basin area. Thus, our agriculture category appears to be the major type of land use/habitat that could be used for denitrification purposes. However, ongoing studies (Fairchild and others, 2003) indicate that agricultural soils, although identified by mapping as having HDP, may not actually support what the model predicts. This may be due to agricultural management practices (tillage, tile drainage, etc.) that discourage denitrification.

Ownership was also considered in our model. In the buffered zones along the rivers, there were 15 square miles of land in public ownership in Iowa. Data were not available for Minnesota. Public land managers may actively manage for increased denitrification. If these lands are identified by the model as having HDP, but further research indicates low rates of denitrification, then the land could be more actively managed for nitrogen removal. This could be done using a variety of rehabilitation efforts including tile drain removal, grass plantings, diking, and other procedures that would increase organic matter, moisture levels, and anoxic conditions associated with denitrification processes. These management efforts are being practiced in habitat types in addition to agricultural lands, although at a smaller spatial scale such as in the bottomland forest and grassed waterways that are known to support higher denitrification rates (Fairchild and others, 2003).

Table 1.6. Agricultural Lands in the Cedar River Basin Study Area Meeting Possible Denitrification Potential Criteria

Microhabitat/ Land Use and Cover	Total Study Area in Square Miles	Buffered Areas in Square Miles	Areas in Buffers Meeting Slope and Soil Criteria in Square Miles	Acres in Buffers Meeting Slope and Soil Criteria	Area Meeting Criteria as a Percent of Total Study Area
Grasslands	356	172	29	18,560	**0.42%**
Bottomland Forest	333	250	46	29,440	**0.66%**
Slough	68	46	9	5,760	**0.13%**
Agriculture	5,880	2,207	550	352,000	**7.91%**
Water	56	49	9	5,760	**0.13%**
Excluded Uses	260	92			
Total	**6,953**	**2,816**	**643**	**411,520**	**9.25%**

Figure 1.7. Areas in the Cedar River Basin Study Area with High Denitrification Potential.



The same rehabilitation management practices used on public lands can be used on privately owned agricultural lands. However, there are currently a lack of economic incentives to do so. Most active producers are paying for both land and machinery and are therefore under intense financial pressure. Therefore, they may be unwilling to implement conservation practices resulting in decreased nitrate runoff due to tradeoffs between environmental gains and losses in productivity and direct financial returns.

There is much debate over farmland management in regard to incentives, regulatory policy, and personal responsibility to the environment through stewardship. The advantages of incentive-based policy include ownership of externalities, such as the degraded downstream water that affects other communities. The disadvantage of the incentive approach is that it does not address the "sustaining the unsustainable" issue, and fosters a reliance on monetary reward for stewardship. The regulatory approach includes incorporation of new laws into the legal system and the assimilated costs of monitoring and enforcement (Costanza et al 1997).

Our model has indicated that approximately 9 percent of land in the Cedar River Basin may have HDP. We estimated buffer widths along streams, calculated lands with a five-percent slope, and devised a query for soil drainage. All of these parameters could be restricted (i.e., selecting lesser slopes, or soils with less permeability), to closer match a given conservation budget. The National Agricultural Statistics Service (NASS) estimated that cropland in Minnesota was valued at $1,420 an acre, and cropland in Iowa was valued at $2,120 per acre, as of January 1, 2002 (2003). To purchase the amount of land to provide the required nitrogen reduction would be impractical. The costs and benefits of various nitrogen reduction strategies are extensively documented by Mitsch et al (1999). However, Hey (2002) suggested that CRP lands could also be used for nitrogen farming. A combination of land purchases, CRP protection, and nitrogen farming might achieve the dual goals of commodity price control and nitrate reduction.

Conclusions

We identified land use and land cover trends in the Cedar River Basin to illustrate the demographic, agricultural, and ecological trends that may affect water quality in the Eastern Iowa NAWQA Region. We developed a procedure for identifying lands with possible denitrification potential. This procedure has been illustrated using the Cedar River Basin study, and using generalized data for the approximately 7,000 square mile area. Refinements can be made as high-resolution digital data are generated. A digital layer of CRP lands and an ownership layer are essential for the accuracy of estimates. A wall-to-wall land use and land cover database that contains consistent categories over time is also essential for these calculations. County level soil (SSURGO) maps will provide data more appropriate for local analysis, as these maps become available. EDNA synthetic streams, flow accumulations, elevations, upstream tracing capabilities, and data linkages provide spatial analysis capabilities for identifying these areas that have possible denitrification potential, and these data will improve as 10-meter elevation data are incorporated into EDNA layers. Lastly, a current and detailed drainage map would better identify HDP lands.

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Chapter 2: Historical and Current Water Quality of the Cedar River Basin

By

Doug Schnoebelen, Steve Kalkhoff, James Fairchild, and Pamela Waisanen

Abstract

The Cedar River above Cedar Rapids, Iowa drains approximately 6500 square miles of heavily cropped farm land (93-98 percent agricultural land use) in east-central Iowa. Agricultural chemicals, and in particular nutrients, are a major contaminant of concern in impairing water-quality. A 57-mile reach of the Cedar River above Cedar Rapids, Iowa has been placed on the impaired waters list for nitrate and fecal coliform bacteria concentrations. The City of Cedar Rapids, Iowa has had nitrate concentrations in their alluvial wells above 10 mg/L at certain times of the year putting these wells in above the maximum contaminant level for this constituent in drinking water. The increased nutrient concentrations from corn-belt states have been linked to hypoxia in the Gulf of Mexico. Limited historical water quality data on the Cedar River shows nitrate concentrations in the early 1900s as typically less than 4 mg/L compared to concentrations since the 1970s at over 10 mg/L. Increasing nitrate concentrations in the Cedar River appear to correspond with increasing fertilizer use in the Basin over the past 30 years. Five sites in the Cedar River Basin, where long-term (1970-95) nitrate concentration data was available, showed increasing nitrate concentration trends. Recent synoptic studies on the Cedar River main stem and tributary sites confirm higher concentrations of nitrate with median nitrate concentrations ranging from 3.3 to 10.3 mg/L. In general, tributary sites had higher concentrations of nitrate than main stem sites. Higher concentrations of nitrate at tributary sites may be from small to nonexistent riparian zones, less dilution from runoff, ground water input, and/or less in-stream processing by algae as compared to the main stem sites.

Introduction

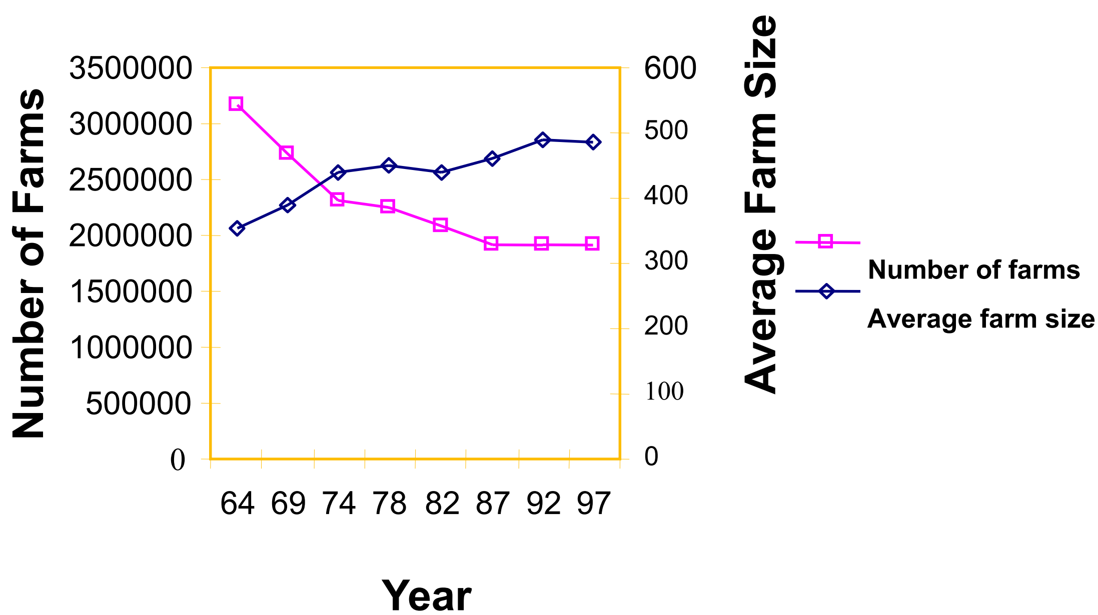
The occurrence, fate, and transport of nutrients are an important component of understanding the water-quality in the Cedar River Basin located in east-central Iowa. Several recent studies in eastern Iowa have identified nutrients as a major contaminant of concern in impairing water quality in streams (Goolsby and Battaglin, 1993; Hallberg and others, 1996; Schnoebelen and others, 1999; Kalkhoff and others, 2000; Becher and others, 2001). Nutrients were investigated along with pesticides in the Cedar River Basin by Squillace and Engberg (1988) from 1984 through the summer of 1985. In addition, increased concentrations of nitrogen and phosphorus from Iowa streams are discharging into the Mississippi River and have been linked to the occurrence of hypoxia in the Gulf of Mexico (Turner and Rabalais, 1994; Goolsby and others, 1999). A 57-mile reach of the Cedar River upstream of Cedar Rapids, Iowa, has been identified as having high nitrate nitrogen (NO_3-N) and fecal coliform bacteria concentrations. This 57-mile reach has been placed on the impaired waters list for these constituents. In particular, high nitrate concentrations (above 10 mg/L) in the Cedar River are of concern to the City of Cedar Rapids, Iowa because it obtains its water from the alluvial aquifer that is directly affected by the Cedar River. Approximately 70 percent of the ground water in the alluvial aquifer is recharged by the Cedar River (Schulmeyer and Schnoebelen, 1998). The Cedar River is the source of most nitrate in the alluvial aquifer in the Cedar Rapids area because of induced infiltration from the river due to pumping (Schulmeyer and Schnoebelen, 1998; Boyd, 1999). The U.S. Environmental Protection Agency (EPA) has set 10.0 mg/L as the limit of nitrate in drinking water (USEPA, 1986) and there are times when municipal wells in the Cedar Rapids well field exceed the nitrate limit as set by the USEPA.

The Biological Resources, National Mapping, and Water Resources Disciplines of the U.S. Geological Survey are conducting an investigation in cooperation with the City of Cedar Rapids and the Iowa Department of Natural Resources to improve the understanding of land use and biological processes that affect the transport and fate of nitrate in the Cedar River.

Background and Historical

The Cedar River Basin at Cedar Rapids, Iowa includes approximately 6,500 square miles that is dominated by agricultural land use (93 to 98 percent). The majority of the area has been in farm production since the early 1900s (Waisanen and Bliss, 2002). Over the last 30 years the trend in Iowa and other parts of the Midwest has been toward a reduction of the number of farms with a corresponding increase in farm size (fig. 2.1).

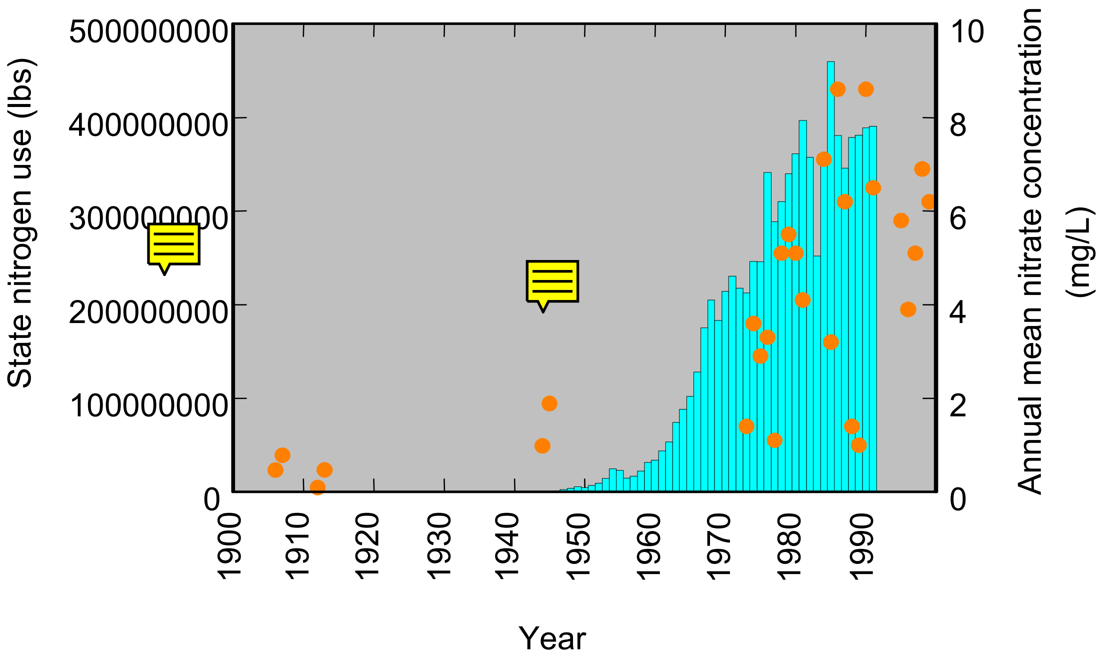
Figure 2.1. Number of farms and average farm size in the United States



Historically, as farming increased in Iowa the total number of wetlands have decreased. It is estimated that approximately 89 percent of the wetlands in Iowa have been lost from 1780 to 1980 (Dahl, 1990). The historical changes (50-100 years) in water-quality for the Cedar River are difficult to document due to a lack of historical water quality data. However, there is limited historical data on nitrate concentrations available for the Cedar River and the nearby Iowa River.

These data can provide a valuable historical context particularly when compared to more recent data collected as part of the present study.

Figure 2.2. Total nitrogen use in Iowa from 1945 to 1990 (blue bars) and annual mean nitrate in the Iowa River (orange dots)



Nitrate concentrations in the Iowa River from the early 1900s typically were 1 mg/L or less (fig. 2.2) (Hershey, 1955). During the period from 1950 to 1980, fertilizer use increased dramatically in Iowa (figs. 2.2, 2.3). Nitrate concentrations from the early 1900s in the Cedar River are low with concentrations less than 4 mg/L nitrate as nitrogen (Clarke, 1924). Recent nitrate concentrations in the Cedar River are substantially greater (fig. 2.3). Corresponding increases in nitrate concentrations in the Cedar River appear to have occurred with increased fertilizer use (figs. 2.2, 2.3).

Trends in nutrient concentrations can indicate long-term improvement or deterioration in stream water quality and may be caused by various conditions within a drainage basin. In the Cedar River Basin, seasonal Kendall Tau Tests for trend analysis for nutrient concentrations have been compiled for all available monthly sampling data for the period 1970-1995 (Schnoebelen and others, 1998). Results are presented in table 2.1. Results that are significant (p-value less that or equal to 0.05) are highlighted in bold type in table 2.1. The magnitude of the Kendall slope estimate (trend slope p) of the relation of concentration to time trend are listed when the p-value is less than or equal to 0.05. The trend slopes are listed in mg/L per year. Increasing trends for nitrate concentrations occurred during the period 1970-95 period in the Cedar River Basin at the following sampling sites: Cedar River at Lansing Minnesota; Cedar River at Austin, Minnesota;

Shell Rock River near Gordonsville; Cedar River at Cedar Falls, Iowa; and Cedar River at Palo, Iowa (table 2.1).

Current Water Quality

Water samples were collected at 21 sites: 15 main stem stream sites and 6 smaller tributary sites (fig. 2.4). Samples were collected in September 2002. In addition, water-quality data were available from previous USGS synoptic samplings at the 21 sites in November 2000 and May 2001. Each site was sampled according to U.S. Geological Survey protocols (USGS, 1998). Samples were filtered through a 0.45 micron capsule filter in the field and were shipped overnight on ice to the U.S. Geological Survey National Water Quality Laboratory (NWQL) in Denver, Colorado for analysis. Water samples were analyzed for dissolved nutrient species that included ammonia, ammonia plus organic, nitrate plus nitrite, nitrite, phosphorus and orthophosphorus. Physical properties (specific conductance, pH, temperature, dissolved oxygen) were measured at each site. A discharge measure was made at each site unless there was a USGS gaging station at the site, and then the discharge was obtained from the gaging station record.

Results

In comparing the September 2002 results and results from previous synoptic studies (November 2000 and May 2001) to limited historical data discussed earlier it is clear that nitrate concentrations are higher now than they have been in the past (table 2.2, figs. 2.2, 2.3). Median nitrate concentrations for all the synoptic studies (main stem and tributary sites) ranged from 3.3 mg/L to 10.3 mg/L (table 2.2). This is considerably higher than the majority of nitrate values that were reported in the early 1900s, which were typically less than 4 mg/L nitrate as nitrogen. Other comparisons in table 2.2 that should be noted are differences in nitrate concentrations between the main stem sites and tributary sites. The tributary sites had higher nitrate concentrations than the main stem sites for each synoptic sampling (table 2.2). The higher concentrations of nitrate at the tributary sites are not unexpected since many of the small tributaries draining cropland riparian zones are small or absent in the basin. Other factors for this difference may be a larger groundwater influx in the main stem sites with more dilution of nitrate from runoff and perhaps more in-stream processing of nitrate by algae in the larger streams. The in-stream processing of algae may also explain the lower overall nitrate concentrations in the September 2002 sampling when compared to the other synoptic sampling (table 2.2). In general the 2002 nitrate concentrations were thought to be lower because during this time there was considerable algae in the stream. Algae can sequester significant amounts of nitrate. However, once algae begin to die off during the late fall and winter, nitrate concentrations tend to rise which was most likely the case in the November 2000 sampling (table 2.2). Climatic conditions may also play a role in that 2002 was a dry year with less runoff to transport nitrate to streams. September and October are typically the driest months in Iowa and any precipitation in November may cause increased transport of nitrate to the river when evapotranspiration is low. Another explanation for higher nitrate concentrations in November of 2000 may be the increased use of the fall application of fertilizer. Fall fertilizer applications can

Table 2.1. Summary of seasonal Kendall trend analysis of concentrations of nitrite plus nitrate nitrogen at surface-water-quality monitoring sites in the Cedar River Basin, 1970–95 (Schnoebelen and others, 1999) . (--) indicates data not corresponding to current study. Probabilites in bold are significantly different in Kendall's Tau test

Corresponding synoptic site numbers labeled on figure 2.4	Site name	Median nitrite plus nitrate nitrogen concentration (mg/L) as N	Kendall correlation coefficient, tau	Probability coefficient	Trend slope [(mg/L)/yr]
--	Cedar River near Lansing, Minnesota(about 10 miles north of Austin, Minnesota)	4	0.33	**<0.0001**	0.1451
--	West Fork Cedar River near Austin, Minnesota	3.86	0.29	**0.0001**	**0.0938**
2	Cedar River near Charles City, Iowa	5.6	0.06	0.4953	Na
4	Shell Rock River near Gordonsville, Minnesota	2.2	0.4	**<.0001**	**0.1682**
6	West Fork Cedar River near Finchford, Iowa	5.35	0.14	0.3134	Na
7	Cedar River at Cedar Falls, Iowa	4.1	0.22	**0.0241**	**0.0944**
--	Cedar River at Gilbertville, Iowa (about 8 miles south of Waterloo, Iowa)	4.1	0.23	0.1209	Na
13	Cedar River near Palo, Iowa	5	0.33	**<.0001**	**0.1853**

Figure 2.3. Yearly mean nitrate concentrations in the Cedar River at the Cedar Rapids, Iowa and Palo, Iowa sites

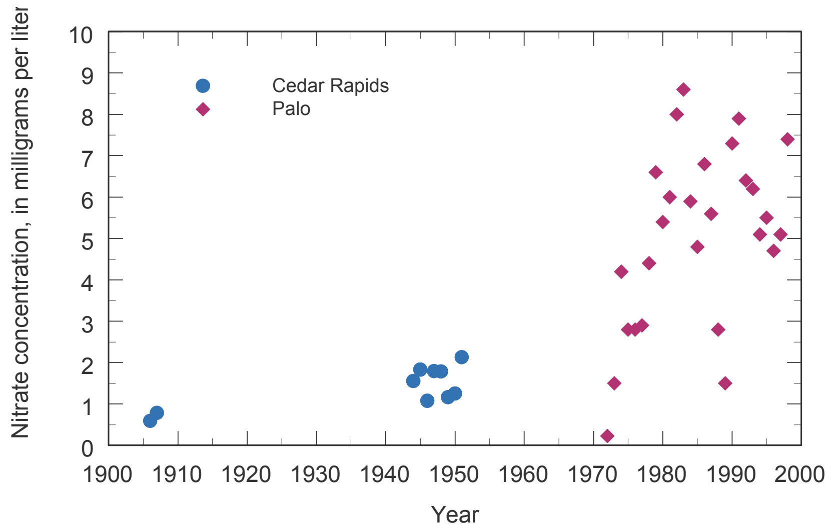
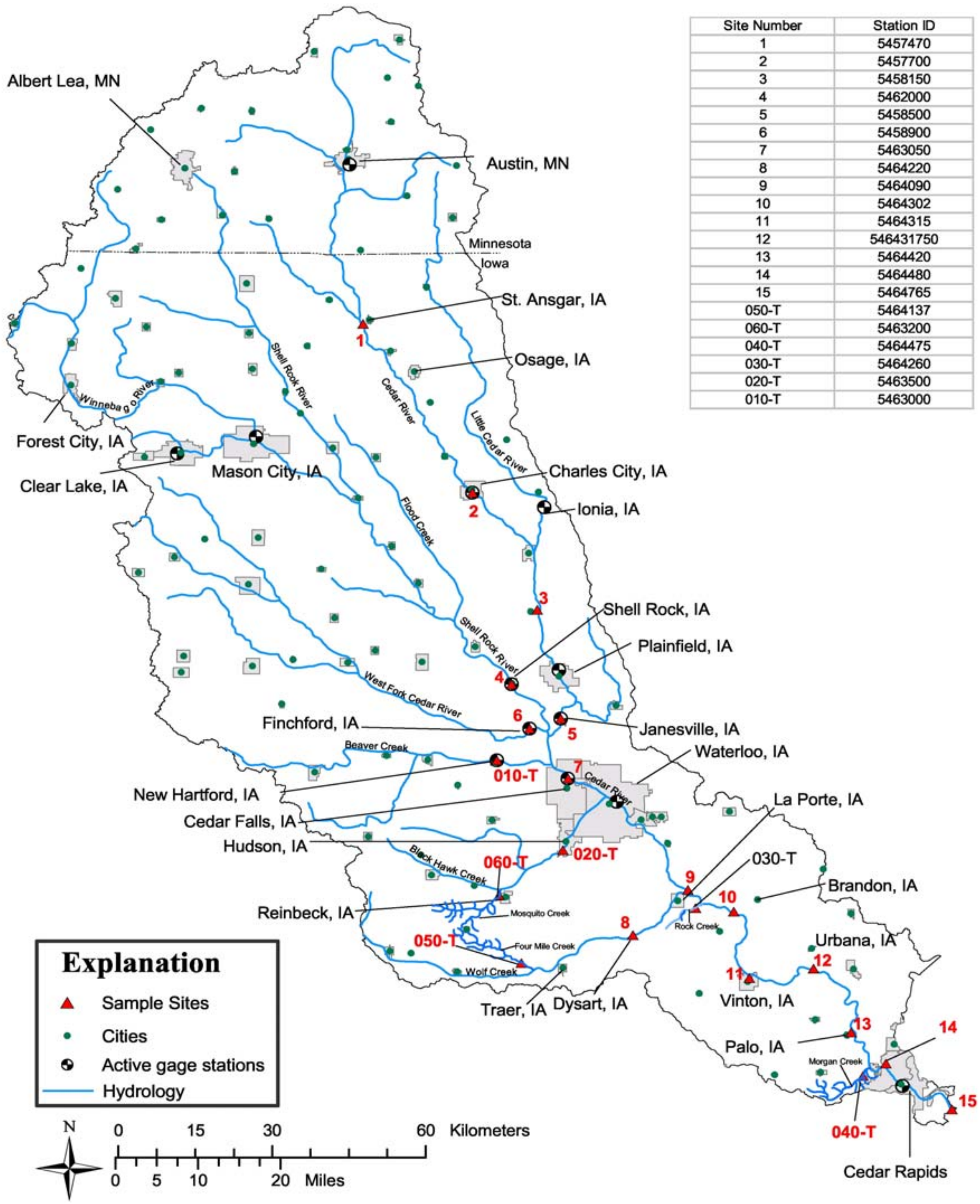


Table 2.2. Median, average, maximum and minimum nitrate concentrations for main stem and tributary sites in the Cedar River Basin sampled in September 2002 (shaded) compared to November 2000 and May 2001 samplings. [concentrations are in milligrams per liter, mg/L; --, no data]

	November 2000		May 2001		September 2002	
Statistic	Main stem sites	Tributary sites	Main stem sites	Tributary Sites	Main stem sites	Tributary sites
Median	6.9	10.3	9.9	--	3.3	5.6
Average	8.1	10.3	10.8	--	3.2	5.6
Maximum	10.4	3.6	14.5	8.9	6.4	8.0
Minimum	12.6	7.5	--	--	1.6	3.3

Figure 2.4 Cedar River Basin above Cedar Rapids, Iowa and water-quality sampling sites.

Site Number	Station ID
1	5457470
2	5457700
3	5458150
4	5462000
5	5458500
6	5458900
7	5463050
8	5464220
9	5464090
10	5464302
11	5464315
12	546431750
13	5464420
14	5464480
15	5464765
050-T	5464137
060-T	5463200
040-T	5464475
030-T	5464260
020-T	5463500
010-T	5463000



increase concentrations of nitrate in streams in eastern Iowa during the fall and winter months depending on when runoff occurs (Becher and others, 2001, p. 20). In the spring when the largest amounts of fertilizers are applied, nitrate concentrations typically tend to increase in streams due to increased runoff after fertilizer application (May 2001 sampling, table 2.2).

Summary

The Cedar River Basin has been dominated by agriculture since the early 1900s. Much of the natural wetland areas have been lost. With increased agricultural production there has also been an increase in the use of fertilizers since the 1950s. A comparison of more recent nitrate data with historical data from the early 1900s in the Cedar River and Iowa River basins show an increase of nitrate concentrations of 3 to 4 times what historical concentrations were. The recent synoptic studies and previous trend work on available data for Cedar River indicate that nitrate concentrations have increased with time at several sites.

Recent synoptic studies have also identified that nitrate concentrations are typically higher in tributary streams than the main stem Cedar River. Monitoring has helped identify watersheds that may be "hot spots" for nitrate. These may be areas that could be targeted first for implementation of future best management practices. Monitoring work has also identified seasonal trends in nitrate concentrations that may be related to in-stream processing by algae or other factors such as the fall application of fertilizers.

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Chapter 3: An Assessment of the Denitrification Potential of Soils in the Cedar River, Basin, IA

By

James Fairchild, Kathy Echols, Doug Schnoebelen, Pamela Waisanen, Stephen Kalkhoff, and B. Thomas Johnson

Abstract

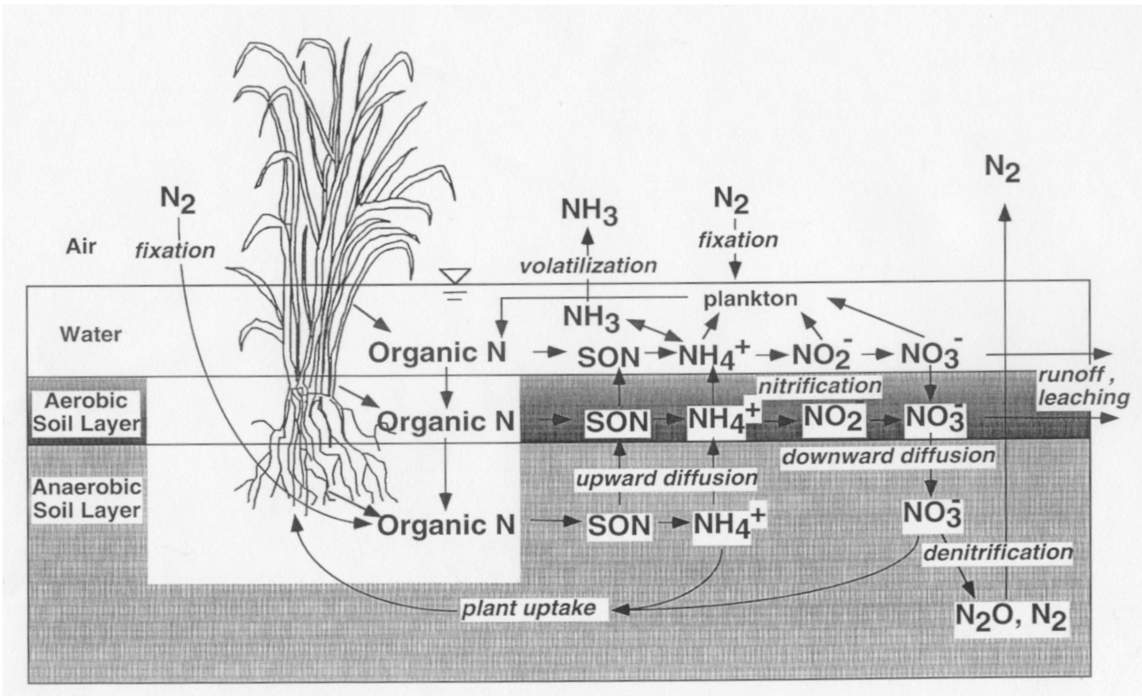
Water quality monitoring by the U.S. Geological Survey has documented that nitrate concentrations in the Cedar River of eastern Iowa are increasing in response to increased nitrogen application associated with row-crop agriculture. Wetland drainage, loss of riparian corridors, and installation of tile drains have exacerbated the problem by decreasing the retention time of nitrogen at the land/water interface. Thus, normal processes of nitrate uptake, assimilation, and denitrification have been altered. We conducted a study to evaluate the denitrification potential of five microhabitat types in the Cedar River Basin: river sediments, slough sediments, agricultural soils, bottomland forest soils, and grassed waterway soils. Highest denitrification rates occurred in soils from the bottomland forest and grassed waterways. However, there was wide spatial variation in denitrification rates. Denitrification rates were associated with increased levels of organic matter. However, denitrification rates were not limited by available nitrate. These results are being used in landscape modeling to determine areas of highest denitrification potential. This effort should facilitate ecological restoration efforts directed at decreasing nitrate levels in Iowa streams.

Introduction

Increasing ecological concern has arisen over the impacts of increasing nitrogen levels entering the Gulf of Mexico (Rabalais and others, 1996). It is widely known that marine systems, unlike freshwater systems, are predominately nitrogen limited (Howarth and others, 1988). Increasing nitrogen levels in the Mississippi River have been associated with increased primary productivity, algal community shifts, increasing organic loads, and subsequent declines in dissolved oxygen in the Gulf of Mexico. This phenomenon, known widely as the Gulf Hypoxia problem, is considered a major threat to commercial and recreational values of the Gulf (Rabalais and others, 1996). Similar observations have been made in the Chesapeake Bay Region (Boynton and others, 1996).

It is now evident that anthropogenic activities have led to basic disruption of the nitrogen cycle (fig. 3.1). Nitrogen entry into the environment under pristine conditions is tightly coupled between the atmosphere and soil organic matter (Mitch and Gosslink 1993; Vitousek and others, 1997). Microorganisms are critical in reducing nitrogen to ammonia, which is rapidly assimilated into plants as living tissue. Ammonium is returned to soils or water as organic matter decomposes. Unassimilated ammonia, if exposed to oxygen, is rapidly oxidized to nitrate by nitrifying bacteria. Nitrate not taken up by plants can be returned to the atmosphere by denitrifying bacteria under conditions of anoxia in carbon-rich environments. However, excess nitrate under oxidized conditions is prone to transport in ground and surface waters.

Figure 3.1. Conceptual diagram of the nitrogen cycle



(From Mitsch and Gosselink, 1993)

Nitrogen can also have significant impacts in freshwater systems. Ammonia nitrogen, under alkaline conditions, shifts to the unionized form which is highly toxic to fish and other aquatic organisms (Sheehan and Lewis, 1986). Further, nitrate levels have increased significantly in groundwater and underground drinking wells. Nitrate in drinking water is toxic to infants at concentrations exceeding 10 mg/L due to its binding with methemoglobin; such nitrate poisoning leads to numerous infant deaths in the U.S. each year (Johnson and Kross, 1990).

Nitrogen levels in freshwater have increased for several reasons. First, there have been major increases in human populations in urbanized areas, which have increased inputs of ammonia associated with sewage treatment systems. These point-source problems have been managed using secondary sewage treatment systems and the establishment of effluent limits for ammonia nitrogen (Reed and others, 1995). Secondly, there have been increased applications of nitrogen in agriculture due to row-cropping and manure applications associated with waste disposal from the expansion of large confined animal feeding operations (Staver and others, 1996; Rabalais and others, 1996).

Non-point sources of nitrogen can exist in dissolved forms including nitrate, nitrite, ammonia, urea, and dissolved organic nitrogen (Horne and Goldman, 1994). Nitrate is the predominant dissolved form in surface waters and is readily mobile and prone to move in the dissolved phase of surface and ground water (Lucey and Goolsby, 1993). In contrast, ammonia is prone to sorb to soils similar to phosphate and is therefore less mobile (Horne and Goldman, 1994).

Retention of nitrogen within a watershed can be increased by controlling the timing and rate of nitrogen applications in addition to increasing the sorption and uptake of nitrogen that is applied. Ammonia losses from terrestrial systems are minimized by controlling erosion and use of vegetated systems for ammonium uptake and removal. However, ammonia is easily oxidized to the highly mobile form of nitrate, which is prone to dissolved losses in runoff. Microbial populations can remove significant quantities of nitrate nitrogen via denitrifying processes that occur under specific conditions of anoxia in the presence of high organic matter (Seitzinger, 1988) (fig. 3.2). However, during senescent periods such as midwinter the ability of biological systems to assimilate nitrogen is reduced due to decreases in rates of biological metabolism; thus, losses of nitrogen tend to increase during senescent periods (Goolsby and Battaglin, 1993).

Figure 3.2. Pathway of microbial conversion of nitrate to gaseous nitrogen

$$\mathbf{NO_3^- \rightarrow NO_2^- \rightarrow N_2O\uparrow \rightarrow N_2\uparrow \rightarrow NO\uparrow}$$

Although these seasonal processes of nitrogen uptake and loss have always occurred, it is increasingly believed that the capacity for natural biological systems to assimilate, store, and return nitrogen to the gaseous phase has been altered due to massive changes at the landscape and watershed levels (National Research Council, 1992). These landscape changes are numerous. For example, over half of the wetland habitats in the United States have been drained and converted to uplands since colonial times (Dahl, 1990). Wetlands represent highly productive systems that not only produce large amounts of organic matter but also retain organic allochthonous matter from upland runoff (Maltby, 1991). This results in high organic sediments that under anoxic conditions support significant populations of denitrifying bacteria (Seitzinger, 1988). The loss of wetlands has resulted in major loss of anoxic, highly retentive soil systems that are major factors in the denitrifying process at the local and landscape level.

The capacity for natural systems to assimilate, retain, and remove nitrogen is also reduced due to other local and landscape-level activities. For example, installation of tile drains increases the rates of dewatering of soils. Similarly, tillage and grazing practices result in localized soil compaction, which can reduce soil infiltration and accelerates runoff. Loss of riparian corridors and native plant communities are also significant factors that reduce soil percolation, soil organic matter pools, and water retention. Collectively, these factors reduce the residence time and alter the soil conditions necessary for nitrogen retention and removal.

The Cedar River is a major stream draining much of the Eastern Iowa NAWQA Unit (EIWA) and extends from southern Minnesota to the Mississippi River in southeastern Iowa. The Cedar River has shown increasing trends in nitrate concentrations in the last 30 years (Schnoebelen and others, 1999). The Iowa 303(d) list specifies a 57-mile segment of the Cedar River above Cedar Rapids, Iowa as impaired by fecal coliform and nitrate-nitrogen (NO_3-N) (Iowa Department of Natural Resources, 1999, p. 172). In addition, increasing trends of nitrate concentrations in the Cedar River have raised concerns from the City of Cedar Rapids, Iowa for protecting their water supply. Cedar Rapids obtains its water supply from a series of wells completed in the alluvial aquifer along the Cedar River. Approximately seventy percent of the recharge for the alluvial aquifer used by the City of Cedar Rapids comes from the Cedar River (Schulmeyer, 1995; Schulmeyer and Schnoebelen, 1998).

Reductions of nitrate levels in streams and groundwater of the Eastern Iowa Basin, and ultimately, decreased nitrate input the to Gulf of Mexico, will require a comprehensive program of reduced nutrient applications, increased nutrient retention, and increased rates of microbial denitrification. Reductions in nutrient applications are the most direct approach to nitrogen reduction; however, there currently are no monetary or regulatory incentives to reduce nitrogen application rates. Therefore, it is important to evaluate factors related to nitrate uptake and denitrification. Denitrification is known to be an important factor in nitrogen removal under optimum conditions. However, much less is known about denitrification rates in specific soil types and conditions in specific geographic areas such as the Eastern Iowa NAWQA Unit.

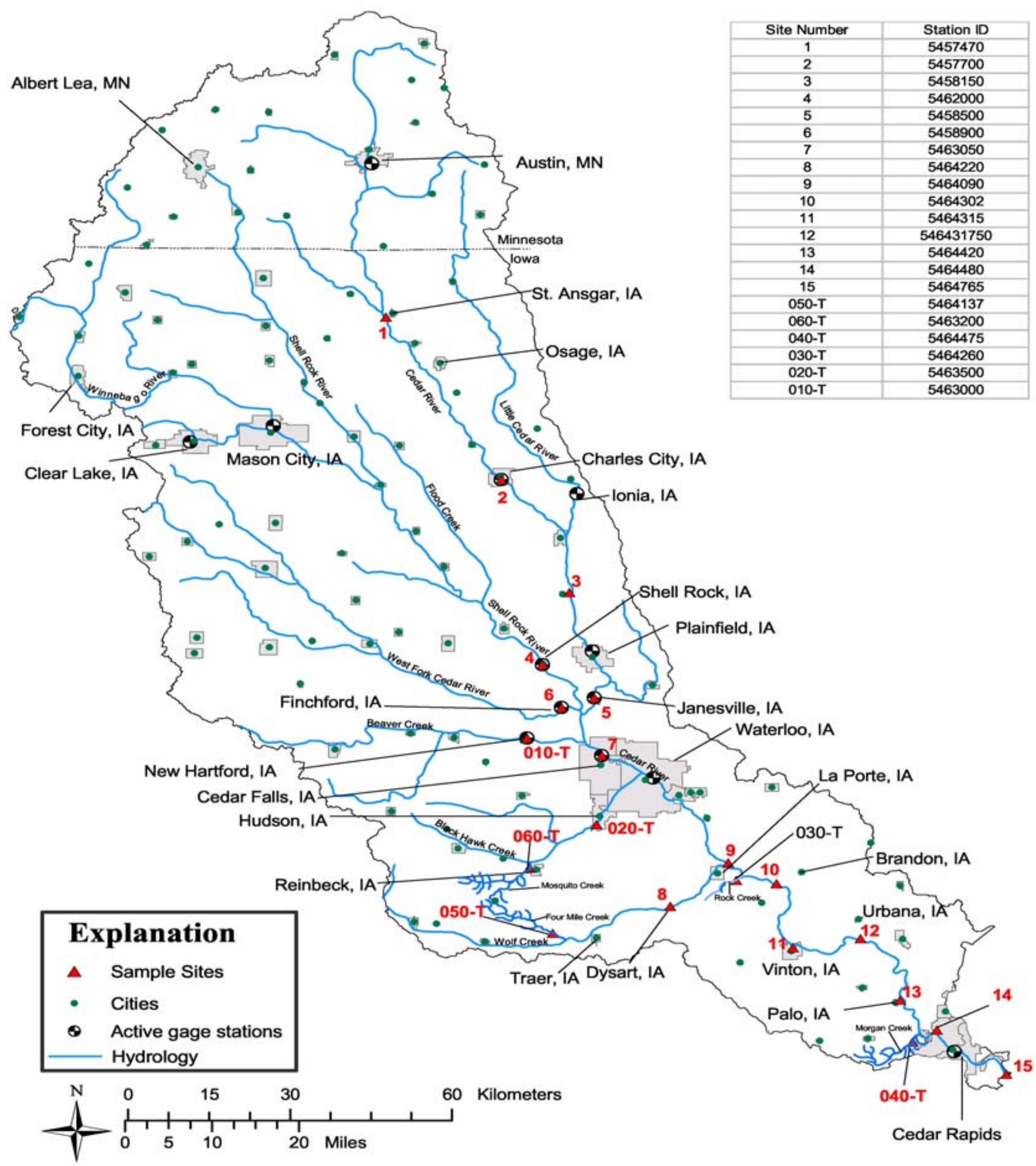
In this study we evaluate the denitrification potential of various soil types in the Cedar River Basin of the Eastern Iowa NAWQ Unit. This study had 2 specific objectives: 1) compare the denitrification potential of different soil microhabitats in the Cedar River Basin, and 2) determine the primary factors associated with denitrification potential. This information is provided in association with the information in Chapters 1 and 2 of this report in order to develop a conceptual landscape model of the Cedar River Basin to expand future efforts at nitrogen reduction by increasing land management activities that foster denitrification processes in soils and sediments.

Methods

Soils Collection and Analysis

Soils and sediments were collected September 23-26, 2002 during the same water sampling interval described in Chapter 2. Soils and sediments were sampled at two main-stem sites (Seminole Valley Park, near site 14 at Cedar Rapids, IA; and Dudgeon State Wildlife Area near site 11 at Vinton, IA) and two tributary sites (private land adjacent to Shell Rock County Park, near site 5 at Shell Rock, IA; adjacent to Site 050-T, Wolf Creek, east of Traer, IA) (fig. 3.3 and Table 3.1). All site locations were within 0.5 km of stream sampling sites described in Chapter 2. Three microhabitat types, expected to reflect a gradient of soil conditions and land use, were sampled at each site: bottomland forest, river/stream, and agricultural (soybean or corn field). An additional microhabitat type was sampled at the main river (slough) and tributary (grassed waterways) sites due to dominance as landscape features and potential for the occurrence of denitrifying conditions. Thus, a total of five microhabitat types were sampled (figs. 3.4-3.8). We attempted to obtain intact soil/sediment cores at each site. However, the coring method was unsuccessful due to the high variation in moisture content and degree of soil/sediment consolidation across sites. Thus, soils and sediments were sampled using a stainless steel trowel at each site and habitat. The top 2.5 cm of soils and sediments were gently scraped away to remove accumulations of large organic matter (stems, leaves, and detritus). The trowel was used to remove the sample from approximately a 5 x 10-cm core area. Samples were transferred to airtight ball jars, sealed under ambient moisture and atmospheric conditions, and chilled on ice to <4 ^{o}C until analysis in the laboratory.

Figure 3.3 Cedar River Basin above Cedar Rapids, Iowa and water-quality sampling sites.



Site Number	Station ID
1	5457470
2	5457700
3	5458150
4	5462000
5	5458500
6	5458900
7	5463050
8	5464220
9	5464090
10	5464302
11	5464315
12	546431750
13	5464420
14	5464480
15	5464765
050-T	5464137
060-T	5463200
040-T	5464475
030-T	5464260
020-T	5463500
010-T	5463000

Table 3.1. Sampling locations used during study[1]

Site	Microhabitat or Soil/Sediment	GPS location		Field Notes
		Latitude	**Longitude**	
Seminole Valley	Bottomland Forest	42° 00' 7.3	91° 43' 40.4"	40m West of well 6
Seminole Valley	River	42° 00' 7.3"	91° 43' 40.4"	100m North of well 6
Seminole Valley	Slough	42° 00' 7.3"	91° 43' 40.4"	35m East of well 6
Seminole Valley	Agriculture	42° 00' 25.3"	91° 43' 64.1"	80m West of walnut tree in fence line by red barns and 50m west into bean field
Dudgeon Lake	Bottomland Forest	42° 11' 34.7"	92° 01'43.7"	Silver maple stand 75m East of road.
Dudgeon Lake	River	42° 10' 40.6"	92° 01' 47.5"	Boat ramp above 150 bridge; semi-backwater East of road
Dudgeon Lake	Slough	42° 11' 34.7"	92° 01' 43.7"	Long u-shaped slough 20m North of Road
Dudgeon Lake	Agriculture	42° 11' 34.7"	92° 01' 43.7"	10 acre soybean field 100-150m South of bottomland forest plot
Shell Rock	Bottomland Forest	42° 43' 61.1"	92° 35' 62.3"	75m West of grass waterway
Shell Rock	River	42° 42' 69.1"	92° 34' 85.3"	50m below Highway T63 Bridge in Shell Rock
Shell Rock	Grassed Waterway	42° 43' 61.1"	92° 35' 62.3"	5m West of Road; 30m wide; sampled middle
Shell Rock	Agriculture	42° 43' 61.1"	92° 35' 62.3"	Soybean farm; 100m North of bottomland forest and grass waterway
Wolf Creek	Bottomland Forest	42° 15' 6.0"	92° 17' 55"	Silver maple/grasses; 100m West of highway; 30m buffer strip middle
Wolf Creek	River	42° 15' 6.0"	92° 17' 55"	20m upstream of bridge
Wolf Creek	Grassed Waterway	42° 15' 6.0"	92° 17' 55"	Road ditch, Northwest side of highway across river from gage
Wolf Creek	Agriculture	42° 15' 6.0"	92° 17' 55"	50m North of riparian forest; 75m North of bottomland forest sample

[1]Sites correspond with Figure 3.1 as Seminole Valley (Site 14, Cedar Rapids, IA), Dudgeon Lake (site 11, near Vinton, IA), Shell Rock (site 5. near Shell Rock, IA), and Wolf Creek (site 050-T, near Traer, IA).

Figure 3.4. Photograph of typical agricultural habitat sampled in study

Figure 3.5. Photograph of typical riverine habitat sampled in study.

Figure 3.6. Photograph of typical bottomland forest habitat sampled in study

Figure 3.7. Photograph of typical slough habitat sampled in study

Figure 3.8. Photograph of typical grassed waterway habitat sampled in study

Soil Analysis

Soils were stored for approximately 3 months under refrigerated conditions (4 ^{o}C) until analysis of denitrification potential, soil moisture, total organic carbon, particle size, ammonia, and nitrate:nitrite concentrations. Concentrations of nitrate/nitrite and ammonia of soils/sediments (air-dried at ambient conditions) were determined by extraction with 2 N KCl as described in Aelion and Shaw (2000) and subsequent analysis using a Technicon AAII system with colorimetric detection. Organic carbon content of soils/sediments was determined using combustion/analysis of a 30 mg sample (dried at 105 °C) in a Coulometrics Model 5020 Analyzer (Joliet, IL). Soil moisture was determined by loss of weight after drying at 105 °C. Particle size analysis was determined on a 100 gm dried sample using the Bouycous Method (ASTM, 2000).

Measurement of Denitrification Potential

Denitrification potential of soils was measured using the acetylene-block method (Yoshinari and others, 1979) similar to that described by Aelion and Shaw (2000). Denitrification potential was determined under two conditions: 1) anoxic saturation, and 2) anoxic saturation +10 mg/L NO_3 in a sequential assay.

Fifty grams of soil or sediment was removed from the original sample storage jar under ambient moisture conditions and added to a screw-top ball jar. Jar lids were modified using two airtight luer-lock fittings that facilitated addition and removal of gases using airtight syringes or gas cylinders. Following soil addition, we added 50 ml of anoxic well water (purged 2 h with N_2 gas) at 25 °C. The jars were then sealed and purged by bubbling N_2 gas (luer-lock delivery from a pressurized gas cylinder while venting the jar with the remaining luer-lock) for 5 minutes to remove residual oxygen. We then introduced 25 ml high purity acetylene gas using a gas tight syringe (approximately 10% of headspace volume) to block the conversion of NO to N_2 gas. Jars were then incubated for 3 h 25 °C in the dark. At the end of 3 hours incubation a 3-ml sample of the jar headspace was removed using an airtight syringe and injected into 2.5 ml serum Vacutainer® and stored chilled (<4 °C) until subsequent analysis of nitrous oxide as described below.

Once sampled, the jars were maintained for 24 h in the dark (25 °C) until the following day. Jars were then amended with $NaNO_3$ in anoxic water to a final NO_3-N concentration of 10 mg/L N and bubbled for a 5-m period with N_2 gas as described above. The acetylene was added; the jar was incubated for 3 h; and the headspace sampled as above. Thus, denitrification potential was determined under anoxic, flooded conditions under ambient and nitrate-amended conditions.

Nitrous oxide, produced by blockage of the microbial pathway to nitrogen gas, was analyzed using gas chromatography. One ml of gas was sampled from each Vacutainer® for GC analysis with the syringe and transfer device flushed with helium gas to prevent cross contamination between each sample. Nitrous oxide was measured using a 5890 II Hewlett-Packard gas chromatograph (Hewlett-Packard, Palo Alto, CA) equipped with a carbon dioxide cryo-valve; electron capture detector (ECD); split-splitless injector with a Merlin Microseal; and a 30 m x 0.32 mm GS-Gaspro capillary column (Agilient Technologies, Palo Alto, CA). A 1 m retention gap was attached to the front of the column with a press-tight union (Restek Corp, Bellafonte, PA). The injector contained a glass wool liner and was kept at 150 °C; injections were run in the splitless mode. The detector temperature was 250 ºC. Helium (UHP) was used as the carrier gas at a flow rate of 3 mL/min. Nitrogen makeup to the detector was 63 mL/min. The septum purge was 4.4 mL/min. The oven temperature was 20 ºC isothermal for a run of 5 minutes per sample. The capillary GC/ECD data were collected, archived in digital form, and processed using a Perkin Elmer chromatography data system, which included the model 970 interface and version 6.1 of Turbochrom Workstation chromatography software, on a Pentium III microcomputer. The method of collecting and quantifying nitrous oxide in samples was calibrated with a five-point nitrous oxide gas standard in helium (Linweld, Lincoln, NE). The calibration used was a point-to-point with the origin forced through zero.

Results

Soil Characteristics

Ambient soil characteristics measured at the time of sampling are presented in table 3.2. Average organic carbon levels of soils and sediments, expressed on a dry-weight basis, ranged from 0.30 % (river sediment from Cedar River at Seminole Valley site) to 5.90% (bottomland forest soil at Dudgeon Lake site). Levels of organic carbon were generally lowest in river sediments/agricultural soils and highest in grassed waterway/bottomland forest soils. Soil moisture levels were lowest in agricultural soils, intermediate in bottomland forest soils, and highest in grassed waterways. River and slough sediments were totally saturated in all cases.

Table 3.2. Summary of habitats and soils sampled in Iowa denitrification study[1].

Site and Type	Habitat	Organic Carbon (%)	Moisture. (%)	Sand (%)	Silt (%)	Clay (%)	NH_3 (ug/g)	NO_2-NO_3 (ug/g)	Denitrification Rate (ng/g/h)
Seminole Valley	Bottomland Forest	2.71 (0.41)	27 (1)	35 (3)	41 (6)	23 (6)	1.16 (0.12)	26.9 (40.3)	11.9 (17.0)
Seminole Valley	Slough	2.52 (0.27)	56 (1)	33 (7)	54 (7)	13 (3)	145.2 (12.2)	0.0 (0.0)	13.3 (26.2)
Seminole Valley	River	0.30 (0.06)	24 (2)	90 (2)	5 (2)	5 (1)	0.45 (0.36)	2.5 (0.6)	9.4 (17.0)
Seminole Valley	Agricultural	0.84 (0.16)	9 (1)	78 (2)	14 (1)	7. (2)	0.19 (0.06)	27.2 (41.6)	8.1 (13.4)
Dudgeon Lake	Bottomland Forest	5.90 (0.76)	32 (2)	62 (6)	20 (4)	13 (4)	2.99 (0.26)	27.0 (15.9)	97.2 (88.2)
Dudgeon Lake	Slough	0.54 (0.11)	31 (6)	88 (4)	4 (2)	4 (1)	13.0 (8.4)	2.4 (0.5)	8.9 (16.4)
Dudgeon Lake	River	0.30 (0.09)	30 (16)	92 (1)	3 (1)	3 (1)	4.14 (1.82)	2.3 (0.8)	8.5 (14.0)
Dudgeon Lake	Agricultural	0.83 (0.8)	9 (1)	79 (1)	12 (2)	9 (1)	0.30 (0.10)	10.3 (5.6)	1.0 (0.6)
Wolf Creek	Bottomland Forest	2.50 (0.31)	19 (1)	28 (9)	50 (7)	20 (3)	1.04 (0.19)	16.7 (2.7)	4.4 (3.6)
Wolf Creek	Grassed Waterway	3.31 (0.26)	29 (2)	30 (4)	40 (6)	30 (3)	1.14 (0.19)	27.4 (6.9)	25.1 (21.9)
Wolf Creek	River	0.31 (0.08)	24 (0.4)	83 (7)	12 (6)	5. (2)	7.28 (3.17)	0.54 (0.42)	10.0 (19.9)
Wolf Creek	Agricultural	2.53 (0.13)	19 (1)	38 (2)	44 (4)	18 (2)	0.68 (0.14)	21.3 (2.4)	12.0 (15.2)
Shell Rock	Bottomland Forest	4.07 (1.65)	22 (5)	56 (3)	28.2 (2.3)	14 (2)	1.08 (0.47)	15.3 (10.0)	12.6 (13.2)
Shell Rock	Grassed Waterway	3.31 (1.29)	31 (4)	71 (5)	21.2 (2.4)	7. (3)	0.69 (0.30)	22.0 (2.40)	16.0 (22.7)
Shell Rock	River	0.55 (0.16)	23 (1)	86 (10)	2.0 (0.7)	2 (1)	0.79 (0.46)	3.19 (1.20)	8.1 (15.4)
Shell Rock	Agricultural	1.59 (0.10)	14 (1)	64 (2)	24.6 (1.9)	11 (2)	0.36 (0.10)	21.6 (3.4)	12.2 (14.1)

[1]Numbers represent mean + 1 standard deviation of five replicates. Parameters represent ambient soil conditions at time of sampling.

Soil and sediment particle sizes ranged widely both between microhabitats and sites. For example, river sediments consisted of over 86% sand and roughly equal amounts of silt and clay. However, slough sediments ranged widely from 33% sand (Seminole Valley) to 88% sand (Dudgeon Lake); variation is not surprising given the potential differences in hydrologic dynamics in slough habitats during flooding. Similar ranges were observed in agricultural soils (range 38-79%); grassed waterways (range 30-71%), and bottomland forest soils (range 28-62%).

Highest sediment ammonia concentrations occurred in slough sediments (range 13-145 µg/g) and river sediments (range <1-8 µg/g) that contained correspondingly lower levels of nitrate (≤ 3 µg/g). In contrast, upland soils contained elevated levels of nitrate (range 10-27 µg/g) compared to the river/slough habitats.

Denitrification Rates

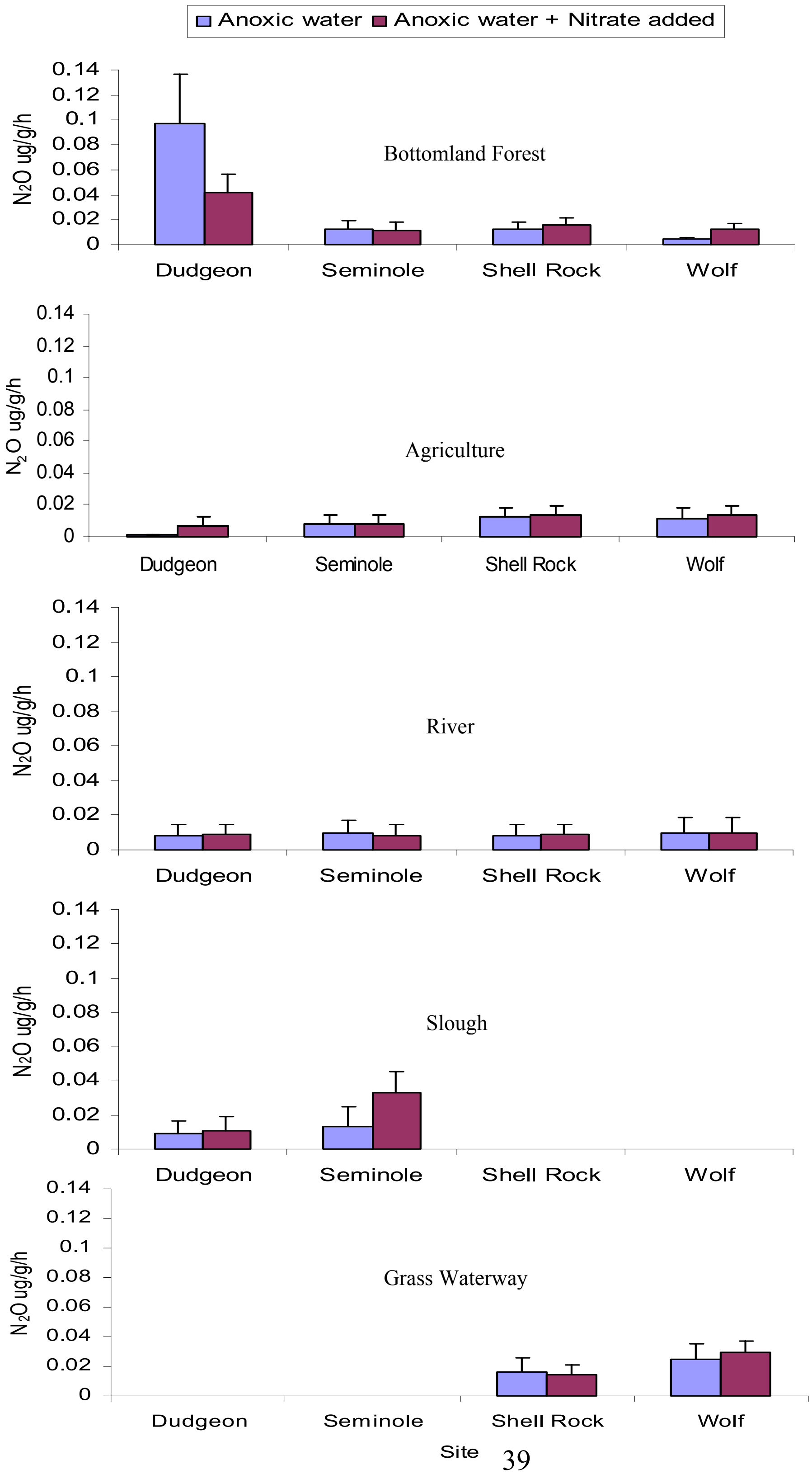
Soil and sediment denitrification rates for both the anoxic water treatment and anoxic water + nitrate manipulations are provided in figure 3.9. Highest levels of denitrification potential occurred in bottomland forest soils at the Dudgeon Lake Site (97 ng/g/h) and were five-fold higher than those measured at other sites (fig. 3.9). Denitrification rates at Dudgeon Lake were not nitrate-limited; in fact, rates decreased following nitrate addition in the second phase of the assay (fig. 3.9). On average, bottomland forest soils had statistically higher denitrification rates compared to the other microhabitats. However, denitrification rates varied widely across bottomland forest sites with lowest levels occurring at Wolf Creek (4 ng/g/h). The Wolf Creek bottomland forest site was atypical compared to the other sites and consisted of a 10-m wide riparian corridor elevated approximately 10 m above the stream, which may limit the annual amount of saturated conditions at this site.

Denitrification rates at the two grassed waterway microhabitats (Shell Rock and Wolf Creek tributaries) were consistently higher (range 16-25 ng/g/h) compared to all sites except the Dudgeon Lake Bottomland forest site (97 ng/g/h) and the Seminole slough site (13 ng/g/h). The slough site at Seminole contained no measurable levels of nitrate under ambient sampling conditions. However, nitrifying bacteria may have converted sufficient quantities of ammonia to nitrate during the study to contribute to the elevated denitrification activity observed. Subsequent addition of nitrate resulted in a significant increase in the observed denitrification rate to approximately 40 ng/g/h. This was the only site where nitrate addition had a significant effect on denitrification rates. Denitrification rates in agricultural soils and river sediments were the lowest observed in the study and were consistently less than 12 ng/g/h.

Discussion

Denitrification is a common microbial process among prokaryotic bacteria that results in the reduction of nitrate to a number of intermediate reduced forms of nitrogen. Denitrifying bacteria compose from 1-5% of the soil bacteria that can be cultured (Tiedje and others, 1988). Although denitrifying bacteria are ubiquitous, population densities vary spatially due to differences in ambient soil conditions that promote their growth (Parkin, 1987). Anoxia, high organic carbon, and high moisture generally favor conditions for denitrifying bacteria. Thus, it is predicted that highest denitrification potential would occur in the bottomland forest and grassed waterway habitats that are expected to be saturated for a large portion of the year.

Figure 3.9. Relative denitrification rates of soils and sediments in each microhabitat type by site. Numbers represent mean ± 1 standard deviation of five replicates



Results in general supported predictions: highest rates of denitrification potential occurred in the bottomland forest, grassed waterway, and slough habitats where anoxic conditions are likely to occur during some part of the year. However, rates varied widely (4-97 ng/g/h) across sites that were predicted to have high denitrification potential. River sediments and agricultural soils had consistently the lowest denitrification potential, most probably due to infrequent conditions of anoxia (river sediments) or anoxia/saturation (agricultural soils) over the annual cycle. Denitrification potentials measured in this study were significantly higher than those measured in intact soil cores (0-4 ng N_2O/g/h) from Rose Lake Wildlife Research Area by Robertson and Tiedje (1986). However, our use of soil slurries is known to produce higher estimates of denitrification than studies using intact cores. Furthermore, we optimized conditions for denitrification by controlling temperature, moisture, and anoxia. However, our primary objective in this study was to compare various locations and microhabitats under standardized conditions. The high variation observed in denitrification potential, observed both within and across sites, has been previously documented (Robertson and Tiedje, 1986; Parkin, 1987). Variation in this study, which maximized conditions for denitrification, is most likely explained by differences in microbial biomass among samples. We did not measure biomass of denitrifying bacteria in this study due to financial limitations. However, this is a priority for future studies of denitrification potential.

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