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Net N input and riverine N export from Illinois agricultural watersheds with and without extensive tile drainage

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Abstract. Some of the largest riverine N fluxes in the continental USA have been observed in agricultural regions with extensive artificial subsurface drainage, commonly called tile drainage. The degree to which high riverine N fluxes in these settings are due to high net N inputs (NNI), greater transport efficiency caused by the drainage systems, or other factors is not known. The objective of this study was to evaluate the role of tile drainage by comparing NNI and riverine N fluxes in regions of Illinois with similar climate and crop production practices but with different intensities of tile drainage. Annual values of NNI between 1940 and 1999 were estimated from county level agricultural production statistics and census estimates of human population. During 1945–1961, riverine nitrate flux in the extensively tile drained region averaged 6.6 kg N ha^{-1} year^{-1} compared to 1.3 to 3.1 kg N ha^{-1} year^{-1} for the non-tile drained region, even though NNI was greater in the non-tile drained region. During 1977–1997, NNI to the tile-drained region had increased to 27 kg N ha^{-1} year^{-1} and riverine N flux was approximately 100% of this value. In the non-tile-drained region, NNI was approximately 23 kg N ha^{-1} year^{-1} and riverine N flux was between 25% and 37% of this value (5 to 9 kg N ha^{-1} year^{-1}). Denitrification is not included in NNI and, therefore, any denitrification losses from tile-drained watersheds must be balanced by other N sources, such as depletion of soil organic N or underestimation of biological N fixation. If denitrification and depletion of soil organic N are significant in these basins, marginal reductions in NNI may have little influence on riverine N flux. If tile drained cropland in Illinois is representative of the estimated 11 million ha of tile drained cropland throughout the Mississippi River Basin, this 16% of the drainage area contributed approximately 30% of the increased nitrate N flux in the Lower Mississippi River that occurred between 1955 and the 1990s.

Introduction

Rivers and streams draining temperate, undeveloped areas tend to have low N concentrations and fluxes (Dodds et al. 1996; Clark et al. 2000). The high energetic cost of N fixation appears to limit the quantity of biologically available N in these settings, which leads to competition for available soil N and limits hydrologic loss of N. Anthropogenic N inputs to the terrestrial system in the forms of N fertilizer, deposition of NOy, and production of leguminous crops can increase the availability of N in the soil, which can increase terrestrial primary production and increase hydrologic loss of N.

Howarth et al. (1996) and Jordan and Weller (1996) demonstrated that net N input (NNI) to large, temperate drainage basins is highly correlated to riverine N export. NNI is the difference between anthropogenic N inputs (NOy deposition, fixation associated with crop production, and food, feed and fertilizer imports) and outputs (food and feed exports). In many temperate drainage basins, riverine N export is approximately 25% of NNI (Howarth et al., 1996; Boyer et al. 2002). The fate of the remaining 75% probably varies across watersheds. In 16 basins of the northeastern USA, Van Breemen et al. (2002) estimated that about 51% of the NNI in the early 1990s was denitrified, and 18% had accumulated in soils and woody biomass.

The fraction of NNI that becomes riverine N flux is likely to be influenced by hydrologic transport rates and pathways. David and Gentry (2000) reported that riverine N flux from the state of Illinois was 50% of NNI, which is considerably greater than the average of 25% observed elsewhere. The relatively high percentage of NNI in Illinois rivers was attributed to the presence of artificial subsurface drainage pipes or ''tiles''. Tile drainage is not evenly distributed throughout Illinois, however. Tiles are very common in flat, wet prairie soils of east central Illinois, and very uncommon in southern Illinois (Davison, 1916) where topography provides greater natural drainage and properties of the clay reduce the effectiveness of drainage tile.

Much of the land with extensive tile drainage in Illinois and elsewhere in the north central USA was wetland drained in the late 19th and early 20th centuries (e.g., Bogue 1951). These soils have relatively high organic matter content (4–6%), the decomposition of which may contribute to high riverine N flux. Maize yields from these soils also tend to be high, which encourages relatively high rates of N fertilizer input ($\sim 190\,\mathrm{kg\,N^{-1}\,ha}$) that may also contribute to high riverine N flux. The high riverine N fluxes from these regions may, therefore, be due to high NNIs, enhanced hydraulic transport, and/or mineralization of soil organic N. An analysis of the relationships between NNI and riverine N flux for regions with and without tile drainage can provide insight into how much of the high riverine N flux is due to greater NNI and how much is due to other factors. Riverine nitrate data collected in the 1940s and 1950s in Illinois (Larson and Larson 1957; Harmeson and Larson 1969), in combination with contemporary data on nitrate and other forms of N, allows for a relatively long-term comparison that includes large changes in fertilizer N inputs and agricultural practices.

The objective of this study was to evaluate the influence of tile drainage on riverine N fluxes by comparing contemporary and historical NNI and riverine N fluxes in Illinois agricultural watersheds with and without tile drainage. The limited geographic scope provides a relatively uniform climatic environment in order to focus comparisons on the effects of tile drainage.

The setting

Due to flat topography, fine textured, loess derived soils, and Native American use of fire, much of east central Illinois was wet prairie and wetlands after the end of the Wisconsin glaciation, approximately 10,000 years ago. In the early 1800s

Native Americans were forced to the west of the Mississippi River. European settlement expanded in central Illinois after railroads were constructed in the 1860s. Over time, institutions and technology were developed that enabled the conversion of wet prairie to highly productive cropland (Bogue 1951). The primary technology for increasing the drainage of water from wetlands has been the installation of drain tubing, initially fashioned from clay roof tiles, and called drain tile. Installation of drain tile in Illinois occurred rapidly in the 1880s when the state constitution was amended to give considerable authority to locally organized drainage districts. More drain tile was installed in the 1880s than in any other decade up to 1920, when new installation became minimal and record keeping ended. Since 1920, drainage systems have been expanded and improved, but no reliable statistics have been collected on this activity. Since the 1960s, practically all drainage tubing has been manufactured from plastics rather than clay, yet they are still referred to as drain tiles.

In contrast to central Illinois, the rolling topography of southern Illinois provides greater natural drainage, and was settled several decades earlier than central Illinois. Native vegetation was more commonly forest, and soils tend to have a higher clay content and lower organic matter ($\sim 2\%$) than those in central Illinois ($\sim 4\%$) (Alexander and Darmody 1991). Many cultivated soils in southern Illinois have a low permeability horizon (fragipan) at approximately 1 m depth, which restricts water movement and root penetration. These soils have a tendency to become saturated during the growing season, but tile drainage has not been effective because of the low hydraulic conductivity and the dispersive nature of the clays. Statistics on tile drainage installation collected in the early 20th century indicate there was less than 0.7 m of drainage tile installation per hectare of cultivated land in the southern portion of Illinois (Davison 1916) compared to 21 m ha^{-1} in the central portion of the state. In southern Illinois, the modification of the surface topography is a more common practice for reducing the probability of saturated soil in agricultural fields than the installation of tiles.

Annual precipitation and stream flow in southern Illinois (1100 mm $year^{-1}$ and 350 mm $year^{-1}$, respectively) are slightly greater than in central Illinois (1000 mm $year^{-1}$ and 300 mm $year^{-1}$, respectively). Much of the difference in precipitation occurs during the dormant season, which contributes to higher streamflows in southern Illinois during that season.

In both southern and central Illinois, cropland (including hay) is the predominant land use (Table 1). Maize and soybean are the most common annually cultivated crops in both central and southern Illinois, although the proportion of land devoted to maize–soybean production is greatest in the central region. The proportion of the land in each region that is annually cultivated for maize, soybean, wheat or oats generally increased between 1940 and 1980 and was relatively constant between 1980 and 1997 (Figure 1).

The productivity and uniformity of the soils in east central Illinois, in combination with trends in economics and mechanization, have led to a high degree of specialization in production of maize and soybeans. In the southern region, where soils are less uniform and generally less productive, a greater portion of the land is

Table 1. Average land-cover classifications in Illinois regions according to the 1982–1997 Natural Resource Inventories (USDA).

Region*	Cropland (including hay)	Forest	Urban	Grassland pasture
East central	89	3	5	3
Southeast	78	11	4	7
Southwest	67	17	5	10

* Defined by counties identified in Table 2.

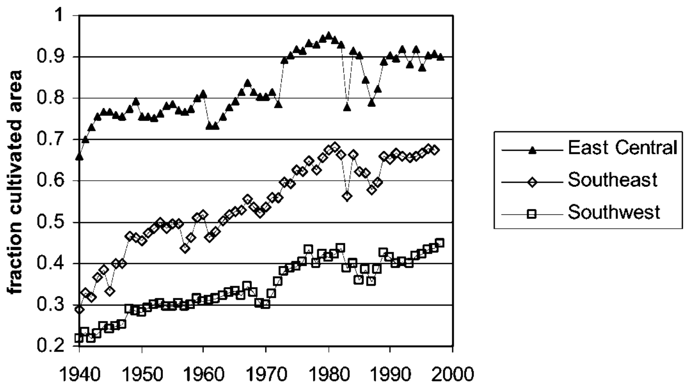


Figure 1. Fraction of the land in each region used to produce annual cultivated crops (maize, soybean, wheat and oats) 1940–1997.

forested, and a greater portion of the cropland is devoted to wheat and hay production. Livestock density is also greater in southern than in east central Illinois.

Methods

Calculation of NNI

Annual estimates of NNI for 1940–1964 and 1977–1997 were calculated for three regions in Illinois using annual agricultural production statistics and decadal population census statistics for 27 counties in central and southern Illinois. Eleven counties in east central Illinois were used to calculate NNI for the intensively tile drained region of the state (Table 2). Ten counties in southeastern and six counties in southwestern Illinois were used to calculate NNI for areas with little tile drainage. These counties were selected to correspond to drainage areas of rivers that have been monitored for discharge and N concentrations.

With some minor exceptions, methods of estimating NNI were similar to that described by Howarth et al. (1996), David and Gentry (2000) and McIsaac et al. (2001, 2002). N harvested in maize and soybeans was estimated to be 0.331 and

Table 2. Counties used to compile partial N budgets for different regions and drainage basins in Illinois.

Region	Tile density	Drainage basins	Counties
East central	High	Sangamon, Vermilion (Illinois River), Vermilion (Wabash River), Mackinaw, Upper Embarras, West Okaw	Champaign, Christian, De Witt, Douglas, Ford, Iroquois, Livingston, Macon, McLean, Piatt, Vermilion
Southeast	Low	Little Wabash, North Fork of Embarras	Clay, Coles, Cumberland, Edwards, Effingham, Jasper, Richland, Shelby, Wayne, White
Southwest	Low	Big Muddy	Franklin, Jackson, Jefferson, Perry, Washington, Williamson

1.55 kg N bu^{-1}, respectively. N harvested in hay was assumed to be 23.6 kg N t^{-1} for alfalfa and 20 kg N t^{-1} for other hay.

For a variety of reasons, we assumed that pastures were not significant net contributors of N to the system. The trampling of vegetation by livestock and recycling of manure N deposited in pastures tends to inhibit biological N fixation, especially in permanent pastures. Additionally, in our budgeting method, N fixed in pastures that is harvested by grazing livestock would essentially substitute for feed grains in meeting the nutritional needs of the animals, and thereby allow an equal increase in grain N exported from the basin. Thus, biological N fixation in pastures would influence the form of N input and outputs, but would not alter the net regional N balance. Additionally, data on the area used for pasture was not consistently collected during the study period, but it appeared to be less than 12% in most years. On the other hand, pastures could have provided a significant source of N when the pastures were rotated with cultivated crops. Establishing legume pasture vegetation would have led to accumulation of biologically fixed N in the pasture, which would have been mineralized when cultivated in preparation of planting an annual crop. No data could be found to determine the extent to which pastures were rotated with annual crops in the different regions. Our assumption of no NNI from pastures represents an underestimate of the N input from this source. Since pastures appear to be a small portion of the area, the NNI is also small, and probably on the order of 0–5 kg N ha^{-1} $year^{-1}$ when distributed over each region.

N fixation in soybeans was estimated to be 0.9 kg N bu^{-1}. N fixation in alfalfa and other hay was estimated by two different methods. Following methods used in previous studies, we estimated biological N fixation in alfalfa to be 218 kg N ha^{-1} $year^{-1}$ and 116 kg N ha^{-1} $year^{-1}$ in other hay (Howarth et al. 1996; Jordan and Weller 1996; David and Gentry 2000). In some instances this led to hay N fixation estimates that were less than estimated N harvested in the hay. While this may occur in settings where high N availability in the soil inhibits fixation by legumes, it is more common for N fixation in forage legumes to be greater than the N harvested. Meisinger and Randall (1991) suggested that N fixation associated

Table 3. Comparison of estimates of fertilizer used in the study areas based on county level estimates presented by Alexander et al. (1990), N fertilizer sales reported at the county level and surveys conducted by USDA and the Illinois Department of Agriculture reported for major land resource areas and crop reporting districts.

Years	kg N ha^{-1} $year^{-1}$								
	Central			Southeast			Southwest		
	Alexander	Survey	Sales	Alexander	Survey	Sales	Alexander	Survey	Sales
1954	7	7	–	5	5	–	3	3	–
1959	12	14	–	8	7	–	5	4	–
1964	30	29	–	21	18	–	13	10	–
1977–1985	58	–	87	64	–	45	45	–	27
1992–1994	–	71	79	–	44	47	–	51	50

with hay crops is approximately 1.25 times the amount of N harvested legume hay. Thus, we also calculated this estimate of N fixation in alfalfa and other hays.

Estimates of annual N fertilizer use prior to 1945–1964 were based on the county level estimates of Alexander et al. (1990). These estimates compared well with those found in Ibach and Adams (1967), and Ibach et al. (1964) who compiled fertilizer use statistics for sub-regions within states for the 1954, 1959 and 1964 crop years (Table 3). Fertilizer input prior to 1945 use was assumed to be negligible. After 1977, fertilizer input estimates were based on county level N fertilizer sales published by the Illinois Department of Agriculture. Sales reported by fertilizer dealers to the Illinois Department of Agriculture are sometimes several months late, and these late reports are usually added to the next year total. To account for delayed reports, we used a 2 year moving average for each year, where the reporting year was defined as occurring between July of the previous year and June of the current year.

On average, the estimated fertilizer N input values based on sales were within 15% of survey data collected by the Illinois Department of Agriculture between 1992 and 1994 (Illinois Department of Agriculture, unpublished data). The estimates of Alexander et al. (1990) differed by 40–60% for the same period. Alexander et al. (1990) assumed statewide fertilizer use would be distributed equally on all fertilized acres within individual counties within each state. In the 1950s and early 1960s, the proportion of fertilized land and the quantities of fertilizer applied in each region were similar. During the 1960s and 1970s, the central region became more specialized in maize production, which requires greater N input than other crops. Thus, the estimates of Alexander et al. (1990) tend to underestimate the fertilizer use in the central portion of the state and overestimate the fertilizer use in the southern portion of the state, where wheat and hay are more common and receive less fertilizer than maize. Since we did not have reliable estimates for fertilizer input between 1964 and 1977, we did not estimate NNI for that period.

Differences between estimates of N fertilizer input based on sales and those based on surveys may result from differences between the area covered by the survey and the counties used for estimating sales. Greater N fertilizer input estimated from sales as opposed to surveys may result from N fertilizer application to crops that were not included in the surveys (e.g., oats, vegetables, lawns). Additionally, N fertilizer sold in a reporting district may be applied outside the district, and there may be systematic under reporting of fertilizer use in the surveys, particularly in regions where water quality concerns have raised concerns about regulation of fertilizer use.

Atmospheric deposition of NOy was estimated to be 5 kg N ha^{-1} $year^{-1}$ after 1980, based on long-term monitoring of wet deposition in the state and assuming dry deposition is 0.7 times wet deposition (David and Gentry 2000). Atmospheric deposition of NOy prior to 1980 was assumed to be proportional to national emissions of NO_x (David and Gentry 2000). Estimated annual NO_x emissions in the 1940s were approximately one third of the emissions in the 1980s, and consequently, we estimated the annual deposition in the 1940s to be one third of the annual deposition after 1980.

N deposition as ammonia and organic N was assumed to be roughly equal to volatilized N and explicit estimates of these fluxes were not included as inputs or outputs. While this assumption appears to be appropriate when calculating NNI for large regions, it may introduce some error at smaller scales, particularly in east central IL during the 1980s and 1990s, which had relatively few livestock and is down-wind of regions with more intensive livestock production. Average annual wet deposition of ammonia in central and southern Illinois ranged from 2 to 3 kg N ha^{-1} $year^{-1}$ between 1985 and 1999 (National Atmospheric Deposition Program 2000). If dry deposition is equal to wet deposition, and if ammonia volatilization in the region is zero, then the net input of ammonia N to the region would be between 4 and 6 kg N ha^{-1} $year^{-1}$. Since ammonia volatilization is likely greater than zero, the net input of ammonia deposition is probably less than 6 kg N ha^{-1} $year^{-1}$.

Riverine N flux estimates

We attempted to make use of all relevant riverine discharge and N concentration data in the study regions for estimating riverine N fluxes. Daily stream flow measurements were obtained from the USGS National Water Information Service (NWIS) web (USGS 2002). For the 1940–1961 period, only nitrate concentrations were available for eight rivers that had been monitored by the Illinois State Water Survey (ISWS) (Larson and Larson 1957; Harmeson and Larson 1969). ISWS analyzed one sample per month for periods of 4–6 years at each location. Riverne N concentrations measured collaboratively by USGS and Illinois EPA were used to estimate fluxes after 1978 for 14 rivers. These data were obtained from the US EPA Storet system. Approximately nine samples per year were collected and analyzed.

Riverine nitrate-N, total Kjeldahl N (TKN) and total N (TN) fluxes in rivers draining the study regions (Table 4 and Figure 2) were estimated using a time period weighting method described by Likens et al. (1977) and Coats et al. (2002). The average concentration of successively collected samples was assumed to represent the concentration of the discharge that occurred between these samples. Coats et al. (2002) reported that this method of load estimation had less bias for estimating annual nitrate flux than other methods tested, including the commonly used rating curve approach of Cohn et al. (1992). Guo et al. (2002) also reported that the rating curve approach tended

Table 4. River monitoring stations used for estimation of riverine N flux. Stations marked with an asterisk (*) represent subbasins of a basin for which nitrate or TN flux were calculated.

River station description	USGS Gage #	Drainage area (km^2)	N Measured 1987–1998	Period of NO_3^- measurement during 1946–1961
Rivers draining areas with extensive tile drainage (east central Illinois)				
Sangamon R. at Riverton	05576500	6783	TN	
*Sangamon R. at Fisher	05570910	622	TN	
*Sangamon R. at Monticello	05572000	1485	TN	1957–1961
*South Fork Sangamon R. at Rochester	05576000	2246	NO_3^-	
Vermilion R. at Danville (Wabash Basin)	03339000	3343	TN	
*Vermilion R. at Oakwood (Wabash Basin)	03336645	1119	TN	
Vermilion R. at Catlin (Wabash Basin)	03338500	2485		1951–1955
West Okaw R. at Lovington	05591700	290	TN	
Mackinaw R. at Green Valley	05568000	2780	NO_3^-	1951–1955
Vermilion R. at Leonore (Illinois Basin)	05555500	3241	NO_3^-	
Vermilion R. at Lowell (Illinois Basin)	05555300	3187		1958–1961
Embarras R. at Camargo	03343400	482	NO_3^-	
Rivers draining areas with minimal tile drainage (southern Illinois)				
Southeastern Illinois				
Little Wabash R. at Carmi	03381500	8001	TN	1958–1961
*Little Wabash R. below Clay City	03379500	622	NO_3^-	1951–1955
North Fork of Embarras R. at Oblong	03346000	824	NO_3^-	
Southwestern Illinois				
Big Muddy R. at Murphysboro	05599500	5620	TN	1956–1961
*Big Muddy R at Plumfield	05597000	2057	NO_3^-	1946–1950

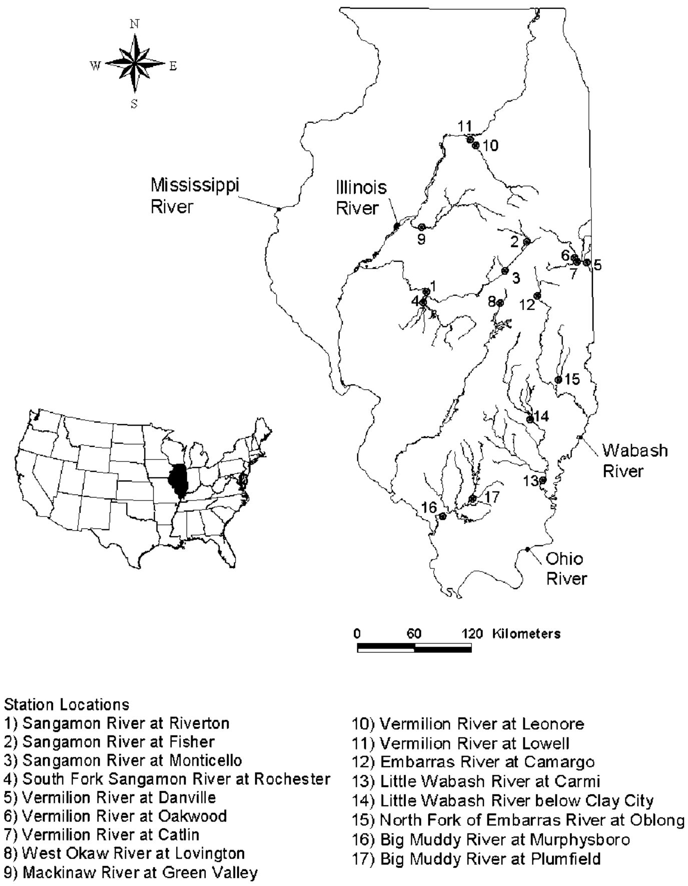


Figure 2. River monitoring locations used in this study.

to over estimate nitrate loads by 4–13% for the Sangamon River at Monticello, IL, and that rating curve estimates were more variable at low sampling frequencies than ratio or flow weighting type methods of estimating nitrate flux. Using the time period weighting method and nine concentration values per year for the Sangamon River at Monticello from May 1993 to April 1999, we estimated an average annual nitrate flux of 24.1 kg N ha^{-1} year^{-1}, which is within 9% of the value that Guo et al. (2002) estimated from daily concentrations for this period (26.4 kg N ha^{-1} year^{-1}) and considered the 'true value'. The 9% deviation from the 'true value' is similar to the root mean square error (RSME) that Guo et al. (2002) reported for ratio and flow weighting methods of estimating 6-year average flux with approximately nine observations of concentration per year, and less than the 10–16% RSME reported for the rating curve methods.

Reverse flows were occasionally recorded for the Big Muddy River at Murphysboro (USGS Station 05599500) caused by exceptionally high flow in the

Mississippi River. No water samples for N concentration determination were collected on days with reverse flow. In estimating annual load, the reverse flows, and an equal volume of subsequent positive flow (the discharge of the reversed flow) were set to zero. Additionally, daily discharge for the Big Muddy River at Murphysboro during the 1994 and 1995 water years was not reported by the USGS. We estimated this from discharge measured upstream at Plumfield (USGS station 05597000) using a regression equation developed from daily observations at both sites between 1990 and 1993, and 1996–2000, which could account for 90% of the variation in observed discharge at Murphysboro. For the Mackinaw River at Green Valley (USGS station 5568000), discharge for the 1987 and most of the 1988 water years was similarly estimated based on discharge measured upstream at Congerville, IL (USGS Station 05567500).

Total riverine N and nitrate-N fluxes were calculated for six river monitoring stations draining east central Illinois and two monitoring stations in southern Illinois. Nitrate flux alone was calculated for an additional four stations in central Illinois and three stations in southern Illinois. The drainage areas above these stations range from 290 to 8100 km^2.

For the Sangamon, Vermilion (Wabash Basin), Little Wabash and Big Muddy Rivers, N concentration and discharge data were available at more than one spatial scale. Riverine N fluxes were calculated for these subbasins to evaluate spatial variability of N fluxes within the regions. To estimate riverine N export from the regions during 1987–1998, an area-weighted average riverine flux was calculated using only the riverine N flux values estimated at the largest scale for each basin with monitoring. In other words, for the east central region, the Sangamon R. at Riverton and Vermilion R. at Danville were used to represent the riverine N flux from those basins. To compare the 1951–1961 riverine nitrate flux estimates to contemporary values, we calculated an area-weighted average nitrate flux using the four contemporary stations that were most similar in drainage area to the stations monitored in the earlier period. Three of the four stations were practically identical across time periods. The fourth pair of stations, the Vermilion River at Catlin and Danville, differed in drainage area by approximately 30%.

Two of the major rivers that drain the southern portion of the state, the Kaskaskia and the Embarras Rivers, originate in the central region, and thus could not be used to represent the non-tile drained areas. Consequently there are fewer rivers that could be used to characterize the agricultural regions with minimal tile drainage.

Results

NNI

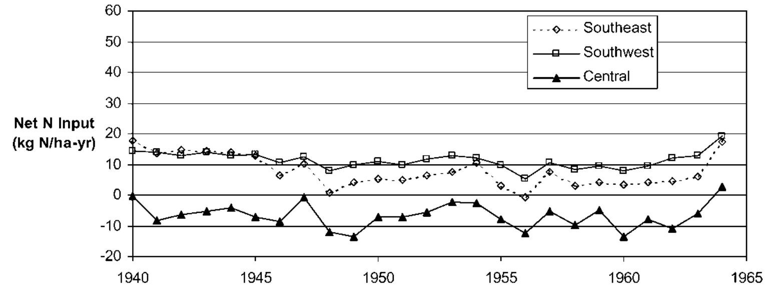
The average annual value of NNI in the east central region of Illinois during 1940–1959 was less than zero, indicating that more N was exported in grain and animal products than was input to the region (Table 5). The average net deficit was between

Table 5. Estimated average annual N inputs and outputs for two periods based on county agricultural statistics for southeastern, southwestern, and east central Illinois.

	kg N ha^{-1} $year^{-1}$					
	1940–1959			1977–1997		
	East central	Southeast	Southwest	East central	Southeast	Southwest
Fertilizer	3.4	2.3	1.5	89.0	51.3	29.0
NOy deposition	2.2	2.2	2.2	5.0	5.0	5.0
Fixation						
Soybean	12.5	6.2	2.6	39.6	23.6	13.6
Hay (based on yield)	9.3	10.3	8.6	2.9	4.7	6.9
Hay (based on area)	6.6	5.6	5.8	2.5	3.0	6.1
Harvest						
Maize	17.1	6.8	2.9	47.5	23.3	9.0
Soybean	21.3	10.3	4.2	65.4	39.0	22.5
Wheat	1.2	1.5	1.9	0.7	4.2	2.5
Hay	4.0	4.5	4.7	2.4	3.8	5.5
Oats	5.2	1.6	0.5			
Manure N						
Cattle	9.0	6.7	5.4	3.1	5.0	5.6
Hog	3.5	2.5	1.5	2.3	3.8	1.9
Human	0.7	0.7	1.2	1.6	0.7	1.4
Horses	0.7	0.9	1.3			
Chickens	1.0	1.0	1.1			
NNI						
W/fixation in hay based on yield	−6.6	8.1	11.1	27.6	23.9	23.8
W/fixation in hay based on area	−9.3	3.4	8.4	27.2	22.2	23.0

6.6 and 9.3 kg N ha^{-1} $year^{-1}$, depending on the assumption used for N fixation in hay. This may represent depletion of soil organic N and/or errors in estimating N inputs, which will be discussed later. In the two southern regions, NNI was positive and ranged from 3.4 to 11.1 kg N ha^{-1} $year^{-1}$. Values were slightly greater in the southwest region, in spite of lower N fertilizer, fixation and manure N inputs, because crop N harvest in the southwest was considerably less than in other regions.

NNI to all regions was substantially greater during 1977–1997 than the NNI for 1940–1959. During 1977–1997, NNI to the east central region averaged either 27.6 and 27.2 depending on the assumption used for N fixation in hay. NNI to the southern regions (24.1 and 22.8 kg N ha^{-1} $year^{-1}$) was approximately 15% less than in east central region. Even though the NNI values were similar in central and southern Illinois, the sources of the N are somewhat different, which may influence the portion that is transported to streams. Fertilizer N input was greatest in the east central region, where N harvested in grain was also greatest. In southern Illinois, both N fertilizer and N harvested in grain were lower than in central Illinois, but N fixation in hay crops and N in manure were greater than in central Illinois.



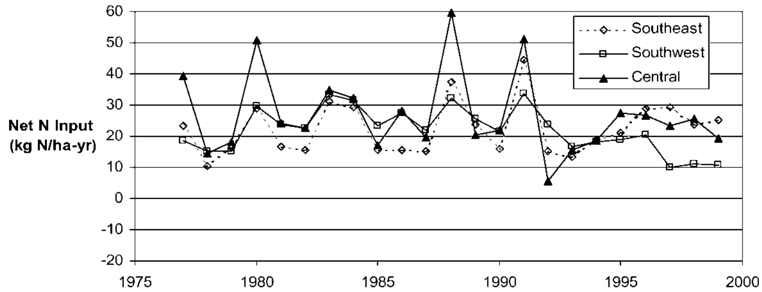


Figure 3. Annual variation in NNI to the three regions in Illinois 1940–1964 (top) and 1977–1999 (bottom).

Annual NNI values tended to decline during the 1940s and early 1950s (Figure 3). This was due in part to the decline in the numbers of horses and chickens and the increased production of soybeans. The decline in livestock in the region allowed more of the harvested N to be exported in grains. Additionally, N harvested in soybeans tends to be greater than the quantity acquired from biological fixation, and consequently the expansion of soybeans represents a reduction in the NNI. In the early 1960s, NNI values began to increase as fertilizer N application increased faster than N harvested in crops.

In the 1977–1999 period, values of NNI were similar among the regions, although the largest annual values of NNI occurred in the east central region during drought years that reduced crop yields. Maize yields are more susceptible to reduction by drought than wheat, soybeans or hay. Furthermore, considerably more fertilizer is applied to maize than to wheat and little is applied to soybeans. Consequently, drought results in more unutilized N fertilizer in maize fields than other crops. The higher annual variability of NNI in the east central region is a consequence of a higher proportion of land devoted to corn production and greater fertilizer input.

Riverine N flux

Riverine nitrate N flux in east central Illinois during the 1950s ranged from 5.8 to 8.2 kg N ha^{-1} $year^{-1}$, which was consistently greater than values for the southern

Table 6. Riverine water and nitrate-N flux in the 1945–61 period.

River monitoring location	Monitoring period	Water Yield (mm $year^{-1}$)	Nitrate-N flux (kg N ha^{-1} $year^{-1}$)
East central region			
Mackinaw R. at Green Valley	1951–1955	193	5.8
Vermilion R. at Catlin	1951–1955	197	6.9
Sangamon R. at Monticello	1957–1961	226	8.2
Vermilion R. at Lowell	1958–1961	188	6.2
Area weighted average		197	6.6
Southeast region			
Little Wabash R. at Carmi	1958–1961	321	2.5
Little Wabash R. below Clay City	1950–1955	162	1.3
Southwest region			
Big Muddy R. at Murphysboro	1956–1961	372	2.8
Big Muddy R. at Plumfield	1946–1950	476	3.1

region which ranged from 1.3 to 3.1 kg N ha^{-1} $year^{-1}$ (Table 6). The riverine nitrate N fluxes were between 25 and 37% of the NNI for the southwestern region and between 16 and 74% of NNI for the southeastern region. Since the terrestrial N budget for the east central region was less than zero during this period, calculating riverine N flux as a percentage of NNI would not be meaningful. Adding the riverine N loss to the terrestrial N deficit suggests that soils in the region were losing 13–16 kg N ha^{-1} $year^{-1}$ on average and/or estimates of NNI were in error by at least this margin.

During the 1987–1997 period, the average riverine nitrate-N flux within in east central Illinois ranged from 13.7 to 38.1 kg N ha^{-1} $year^{-1}$ for different monitoring stations, which was considerably greater than values observed during the 1950s and greater than that observed in the southern regions (Table 7). The spatial variability of riverine nitrate flux may be a reflection of spatial variation in NNI, tile drainage intensity, and/or the presence of surface water reservoirs.

The area-weighted average nitrate flux for the east central region, based on the six stations that represent the largest drainage area of basins with monitoring was 23.9 kg N ha^{-1} $year^{-1}$ which represents about 87–88% of the NNI for the 1977–1997 period. The area weighed average TKN and TN fluxes for the region, based on three monitoring stations that represent the largest drainage area of the monitored basins were 3.3 and 23.6 kg N ha^{-1} $year^{-1}$, respectively. The similarity between the nitrate N flux and the TN fluxes is a consequence of the specific basins for which nitrate and TN data are available. TN concentration data were not available for three monitoring stations with high nitrate N flux (Vermilion River at Leonore, Embarras River at Camargo and Mackinaw River at Green Valley) and it was available for a station with relatively low nitrate-N flux and large drainage area (Sangamon at Riverton).

Table 7. Water yields, and riverine nitrate, TKN and TN fluxes for the 1987–1998 period. Stations marked with an asterisk (*) represent subbasins of a basin for which nitrate or TN flux were calculated.

River monitoring locations	Water yield (mm $year^{-1}$)	kg N ha^{-1} $year^{-1}$		
		Nitrate-N flux	TKN flux	TN flux
East central region				
Sangamon R. at Riverton	257	16.7	2.9	19.5
*Sangamon R. at Fisher	313	28.9	3.2	32.1
*Sangamon R. at Monticello	300	25.8	3.5	29.3
*S. Fork Sangamon R. at Rochester	235	13.7		
Vermilion R. at Danville	336	26.5	4.1	30.5
*Vermilion R. at Oakwood	317	25.2	2.6	27.4
West Okaw R. at Lovington	342	38.1	2.4	40.5
Mackinaw R. at Green Valley	258	25.6		
Vermilion R. at Leonore	297	32.1		
Embarras R. at Camargo	346	33.9		
Area weighted average flux (*sub-basins not included)	284	23.9	3.3	23.6
Area weighted average flux of basins sampled during 1951–1961	299	27.8		
Southeastern region				
Little Wabash R. at Carmi	353	3.7	5.0	8.6
*Little Wabash R. below Clay City	304	4.1		
North Fork of Embarras R. at Oblong	316	4.7		
Area weighted average flux (*sub-basins not included)	350	3.8	5.0	8.6
Southwestern region				
Big Muddy R. at Murphysboro	341	2.1	3.4	5.6
*Big Muddy R. at Plumfield	320	1.0		

For the basins that were monitored for nitrate flux in the 1950s, the area weighted nitrate-N flux during 1987–1998 was 27.9 kg N ha^{-1} $year^{-1}$, compared to 6.6 kg N ha^{-1} $year^{-1}$ during 1951–1961. Some of this increase is likely a consequence of the 51% increase in water yield during 1987–1998 (299 mm $year^{-1}$) compared to that during 1951–1961 (197 mm $year^{-1}$). Assuming a proportional response of nitrate flux and water yield, the estimated 1951–1961 nitrate flux at 299 mm $year^{-1}$ of water yield would be 10 kg N ha^{-1} $year^{-1}$.

Riverine TKN flux for the region appeared to be relatively consistent across monitoring locations at 3.3 kg N ha^{-1} $year^{-1}$. If we assume this value is applicable to all stations where nitrate N flux was determined, then a regional average riverine TN flux would be 27.2 kg N ha^{-1} $year^{-1}$, which is approximately equal to the NNI for the region 1977–1997.

In contrast, the riverine TN flux was 5.6 kg N ha^{-1} $year^{-1}$ in southwestern Illinois and 8.6 kg N ha^{-1} $year^{-1}$ in southeastern Illinois, which represents approximately

25% and 37% of the NNI to the regions, respectively. Riverine TN flux as a percentage of NNI in these regions was within the range observed in other settings (Howarth et al. 1996, Boyer et al. 2002).

It is interesting that riverine nitrate N flux in the Big Muddy River at Plumfield in the 1987–1997 period (1.0 kg N ha^{-1} $year^{-1}$) was only one third of the value estimated for the 1946–1950 period (3.1 kg N ha^{-1} $year^{-1}$) (Table 6). This may be due, in part, to the construction of a large reservoir (Rend Lake) above Plumfield in 1970, which increased denitrification losses in the basin. In order for riverine nitrate-N flux in the Big Muddy at Murphysboro to be 2.1 kg N ha^{-1} $year^{-1}$ during the 1987–1998 period, the 63% of the basin that does not drain from above Plumfield must have had a nitrate yield of 2.7 kg N ha^{-1} $year^{-1}$. This is similar to the nitrate flux estimated for the entire basin above Murphysboro during 1956–1961 (2.8 kg N ha^{-1} $year^{-1}$). It should also be noted that water yield during 1956–1961 was 16% greater than water yield in the later period.

Discussion

According to data presented by Boyer et al. (2002), NNI to the 16 basins in the northeastern USA was 27 kg N ha^{-1} $year^{-1}$, and the average riverine TN flux was 7.2 kg N ha^{-1} $year^{-1}$. The NNI in the northeastern USA and east central Illinois are similar, but the riverine fluxes in east central Illinois are approximately four times greater. The difference is not necessarily explained by differences in water yield, which are greater in the northeast USA (average 545 mm $year^{-1}$) than in east central Illinois (290 mm $year^{-1}$). Greater water yield generally promotes greater riverine N losses. The difference is likely due to the presence of tile drains or characteristics correlated with tile drains such as high soil organic N. In southern Illinois, where tile drains are rare, riverine N flux as a fraction of NNI was similar to the range observed by Boyer et al. (2002).

It is not surprising that tile drainage increased the transport efficiency of NNI to streams, but it is surprising that riverine N flux was equal to or greater than NNI in tile drained regions. This is probably due, in part, to reduced in-field denitrification in this setting. Tile drainage reduces the spatial extent and duration of water saturation in fields, and thereby reduces the anoxic conditions in soils that promote in-field denitrification. Randall and Meisinger (1991) suggested that estimates of denitrification losses should be approximately 30–40% lower in fields with tile drainage than without tile drainage. Additionally, tiles reduce interaction between drainage water and riparian zones, which can serve as significant sinks of N (Osborne and Kovacic 1993).

High riverine N flux relative to NNI in tile drained regions may also be due to sources of N that are not included in our calculation of NNI. Depletion of soil organic N and atmospheric deposition of ammonia and organic N, and inaccuracies in our estimates of biological N fixation are possible sources of N that are not considered in our estimation of NNI.

During the 1940s and 1950s, the area under annual cultivation in southern Illinois expanded by 50–60% (Figure 1). This probably contributed to increased N availability for crop production and for transport to rivers, due to increased net mineralization and depletion of soil organic N in the areas brought into annual cultivation. This may explain why southern Illinois riverine nitrate fluxes in the 1950s were a larger percentage of NNI than TN flux was during the 1987–1998 period. During the later period, the area under cultivation also increased, but the increase was smaller in both absolute and relative terms.

Between 1940 and 1960 in east central Illinois, the area cultivated for annual crops expanded approximately 20%. Although this represents a less dramatic increase in area under cultivation than occurred in the southern regions, the soils in the central region have, on average, approximately twice the organic N content as the soils in the southern region. Thus, the quantity of N mineralized per hectare by the expansion of cultivation during 1940–1960 in central and southern Illinois may have been roughly equal. In addition to expansion of annual cultivation in the east central region, expansions of tile drainage systems were likely to have improved soil aeration and thus increased mineralization of soil organic N in the east central region.

Our estimates of NNI does not include all N inputs associated with green manure or pastures. Few statistics could be found to develop reliable estimates of these sources, which were undoubtedly more important sources of N for crop production prior to the 1960s when fertilizer N was relatively expensive. Statistics on hay production consider only the areas where hay was harvested and not areas where leguminous hay species (e.g., alfalfa, clover, and vetch) were grown only to be plowed into the soil for 'green manure'. Estimates of these N sources would increase the NNI during 1940–1959 for all regions in this study, and could possibly increase the NNI for the east central region to a value that was greater than zero. It is also possible, however, that crop production in the east central region was depleting soil organic N during the 1940s and 1950s. Green manure and rotated pastures are very likely negligible sources of N in the 1977–1997, especially in the east central region where livestock, hay and pasture have become relatively uncommon.

In large regions, atmospheric deposition of NH_x and organic N may be considered to be part of an internal cycle of volatilization and redeposition (Howarth et al. 1996). But smaller regions may behave as source or sink areas for atmospheric NH_x and organic N. In the 1977–1997 period, east central Illinois was probably a sink area for NH_x deposition because livestock densities were greater in the predominant up-wind direction (north, west and south). As discussed in the Methods section, the NNI from NH_x and organic N deposition is likely less than $6\,kg\,N\,ha^{-1}\,year^{-1}$ and may be on the order of $2–3\,kg\,N\,ha^{-1}\,year^{-1}$ for the east central region in the 1977–1997 period. In the 1940s and 1950s, there was greater livestock density in the east central region than in the southern regions. At that time, the east central region may have been a net source of atmospheric NH_x, while the southern regions may have been sinks of NH_x.

Our NNI estimates do not include in-field or in-stream denitrification losses. In order for NNI to equal riverine N flux in the east central region, any denitrification losses must be equal to N sources that are not included in NNI, such as depletion of soil organic N and non-symbiotic biological N fixation. We are not aware of reliable estimates of these fluxes in tile drained settings and recommend research to quantify these N fluxes. Long-term measurements and simulations in both tile drained and non-tile drained settings have indicated that soils lose organic N following initial cultivation but achieve a steady state when under constant management for about 60 years (Stevenson 1986). Simulations of cultivated soils in the north central US suggest that soil organic C values in the top 20 cm declined until about 1950 and then began increasing in the 1970s (Donigian et al. 1997). These simulations consider only the top 20-cm of the soil and did not consider the influence of changes in drainage. Artificial drainage systems probably promote mineralization of soil organic matter by improving aeration of the soil. The rate of increased mineralization associated with improvements in drainage has not been measured. Furthermore, there are no statistics collected on installation of or improvements to drainage systems in the USA. Thus, it is currently impossible to accurately quantify the contribution of this potential source of riverine N at a watershed scale.

If denitrification losses in tile drained settings are large, and non-symbiotic N fixation inputs are small, then the depletion of soil organic N may be a major contributor to riverine N in tile drained settings. Depletion of soil organic N may be a concern for the long-term fertility of the soils (Jaynes et al. 2001). On the other hand, if denitrification losses are small, then net depletion of soil organic N may be a minor contribution of riverine N. Burkart and James (1999) estimated that denitrification throughout most of Illinois in the 1990s would range from 14 to 38 kg N ha^{-1} year^{-1}. This estimate is partly based on suggested denitrification rates of Meisinger and Randall (1991), who speculated that the accuracy of their estimates would be in the range of $\pm$20% to $\pm$50%. Mesinger and Randall (1991) suggested a 30–40% reduction in estimated denitrification rates where tile drainage was installed, but Burkart and James (1999) were not able to apply this adjustment because of a lack of reliable data on tile drainage density. Thus, denitrification in tile drained settings is likely to be 30–40% less than the estimate of Burkart and James (1999). If we reduce the Burkart and James (1999) estimate by 35% for tile drained settings, and consider the estimated accuracy of $\pm$50%, the range of denitrification losses in tile drained settings may be on the order of 5–40 kg N ha^{-1} year^{-1}. The lower estimate suggests that denitrification and depletion of soil organic N play a relatively minor role in the N cycle, but the larger estimate suggests that denitrification and depletion of soil organic N are greater than average annual NNI or riverine N fluxes. If denitrification and net depletion of soil organic N are significant, then marginal reductions in NNI in tile drained watersheds may result in reductions in denitrification and N harvest but have little influence on riverine N output. Plot and field scale experiments indicate that relatively large changes in fertilizer N input result in relatively small changes in nitrate concentrations and export in tile drainage waters (Baker 2001; Jaynes et al. 2001; Randall and Goss 2001). This is consistent with mineralized soil organic N providing a significant portion nitrate losses in tile drainage water. These studies, however, are relatively short-term (3–5 years) and may not fully reflect the long-term effects of reduced NNI on riverine N export.

Jaynes et al. (2001) reported average nitrate N export in tile drainage water from a field under maize–soybean rotation in Iowa ranged from 30 to 50 kg N ha^{-1} year^{-1} depending on the quantity of fertilizer N input. For the high fertilizer N treatment (the farmer's normal practice) nitrate export in the tile drainage water was slightly less than NNI calculated using a partial N budget method similar to our calculation of NNI. For medium and low fertilizer N treatments, nitrate-N output in tile drainage water was greater than NNI, indicating a depletion of soil organic N. The tile drainage system studied by Jaynes et al. (2001) had been installed only 4 years prior to the experiment. Improved aeration of the soil provided by the enhanced drainage may have promoted mineralization of soil organic N that contributed to N flux measured in the drainage water. Other plot and field scale studies have also reported annual average nitrate N flux in tile drainage under maize–soybean production ranging from 15 to 50 kg N ha^{-1} year^{-1} (Mitchell et al. 2000; Randall and Goss 2001).

David et al. (1997) reported that riverine nitrate N export from an intensively tile drained watershed in Illinois was 50% of the available inorganic N. The available inorganic N included an estimated 57 kg N ha^{-1} year^{-1} mineralized from the soil during the growing season that would not be included in the more commonly used estimates of NNI that assume soil organic N is at steady state. Immobilization of inorganic N in crop residues was also included in their watershed N budget, but it was less than half the rate of mineralization, which would suggest a net decline in soil organic N of approximately 30 kg N ha^{-1} year^{-1}. Observations from a limited number of long-term experimental plots and regional simulations suggest that soil organic N in the region has been steady state since the 1950s (Donigian et al. 1997; Aref and Wander 1998), but additional data are needed to accurately assess the long term changes in soil organic N, including the influence of new tile drains.

According to USDA (1987) estimates, in 1985 there were approximately 11 million ha of cropland with tile drainage in the states that drain to the Mississippi River. This represents 4% of the total land area and 16% of the area used for annual crop production in the Mississippi River Basin. If Illinois watersheds are representative of tile drained watersheds in the north central USA, the 20 kg N ha^{-1} year^{-1} increase in riverine N flux that occurred after 1961 from these 11 million ha would have contributed 0.22 Tg N year^{-1} to the Mississippi River system. This is approximately 30% of the 0.70 Tg N year^{-1} increase in nitrate flux recorded in the lower Mississippi between 1960 and 1980 (Goolsby et al. 1999).

N conservation in these settings could have a significant impact on water quality. Randall et al. (1997) and Mitchell et al. (2000) demonstrated that N transport in tile drainage under perennial vegetation (grass or alfalfa) ranged from 1 to 3 kg N ha^{-1} year^{-1}, an order of magnitude less than N transport from fertilized corn-soybeans fields. Additionally, Goldstein et al. (1998) reported significantly lower nitrate concentrations in tile drainage water from a corn–wheat–hay rotation without fertilizer compared to N fertilized maize–soybean systems. Thus, tile drainage alone does not increase riverine N flux, but cultivation and intensification of N use on tile drained lands cause large rates of N transport to streams. Diversification of crop production systems in these regions to include perennial grasses, drought tolerant crops, and crops that require less N fertilizer would likely reduce riverine N transport in these settings (Randall and Goss 2001). Additionally, controlled drainage systems can promote in-field denitrification and reduce transport to surface waters (Zucker and Brown 1998). Finally, construction and restoration of wetlands can also reduce the quantity of N transported from fields to rivers.

Summary

In the 1940s and 1950s, average annual riverine nitrate flux in central and southern Illinois ranged from 1.3 to 7 kg N ha^{-1} $year^{-1}$ with the greatest values occurring in the east central region where tile drainage was extensive. At that time, however, estimated annual NNI to the central region was less than zero, suggesting that crop production was subsidized by depletion of soil organic N and/or we underestimated N inputs from biological N fixation. In southern Illinois, where tile drainage is uncommon, average annual NNI ranged from 3 to 11 kg N ha^{-1} $year^{-1}$ during the 1940s and 1950s. In the 1977–1997 period, NNI had increased to between 22 and 24 kg N ha^{-1} $year^{-1}$ in the southern region and to approximately 27 kg N ha^{-1} $year^{-1}$ in the central region. Riverine N flux in the central region of Illinois during 1987–1998 was approximately 100% of the average annual NNI during 1977–1997, which is 3–4 times greater than observed in other temperate locations (Howarth et al. 1996; Boyer et al. 2002). In southern Illinois, riverine N flux during this period was 25–37% of NNI, which is similar to percentages reported elsewhere. Thus, the exceptionally high riverine N fluxes in tile drained regions cannot be explained by proportionally higher NNI alone. The larger riverine N flux in the tile drained region may be a result of low denitrification losses and/or ongoing depletion of soil organic N and/or greater biological N fixation inputs. Further research is needed to more accurately quantify these fluxes in tile drained settings. Improved understanding of the fluxes and processes that govern riverine N fluxes in tile drained settings can help inform conservation efforts in these regions.

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