Effect of variable annual precipitation and nutrient input on nitrogen and phosphorus transport from two Midwestern agricultural watersheds

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ABSTRACT

Precipitation patterns and nutrient inputs affect transport of nitrate (NO₃-N) and phosphorus (TP) from Midwest watersheds. Nutrient concentrations and yields from two subsurface-drained watersheds, the Little Cobb River (LCR) in southern Minnesota and the South Fork Iowa River (SFIR) in northern Iowa, were evaluated during 1996-2007 to document relative differences in timings and amounts of nutrients transported. Both watersheds are located in the prairie pothole region, but the SFIR exhibits a longer growing season and more livestock production. The SFIR yielded significantly more NO₃-N than the LCR watershed (31.2 versus 21.3 kg NO₃-N ha⁻¹ y⁻¹). The SFIR watershed also yielded more TP than the LCR watershed (1.13 versus 0.51 kg TP ha⁻¹yr⁻¹), despite greater TP concentrations in the LCR. About 65% of NO₃-N and 50% of TP loads were transported during April-June, and <20% of the annual loads were transported later in the growing season from July-September. Monthly NO₃-N and TP loads peaked in April from the LCR but peaked in June from the SFIR; this difference was attributed to greater snowmelt runoff in the LCR. The annual NO₃-N yield increased with increasing annual runoff at a similar

rate in both watersheds, but the LCR watershed yielded less annual NO₃-N than the SFIR for a similar annual runoff. These two watersheds are within 150 km of one another and have similar dominant agricultural systems, but differences in climate and cropping inputs affected amounts and timing of nutrient transport.

Keywords: Agriculture, Nutrient transport, Nutrient yield, Water quality, Land use

Abbreviations: CC: continuous corn; CS: corn-soybean rotation; CV: coefficient of variation (standard deviation divided by mean); LCR: Little Cobb River; Mg: megagrams; NO₃-N: dissolved nitrate- plus nitrite nitrogen; RL: reporting limit; SFIR: South Fork of the Iowa River; SRP: soluble reactive phosphorus; TN: total nitrogen; TP: total phosphorus; WRTDS: weighted regressions on time, discharge, and season

1. Introduction

Nutrients are a topic of water-quality concern because they can contribute to excessive aquatic plant growth, eutrophication, and toxicity (Cерco, 1995; Correll, 1998; Rabalais et al., 2001; Rabalais, 2002). Nutrient export from the U.S. Midwest agricultural states is known to be a contributor to the increasing size of the hypoxic zone in the Gulf of Mexico (Goolsby et al., 2000; Royer et al., 2006; Alexander et al., 2008; Bianchi et al., 2010). Sources of nitrogen and phosphorus include commercial fertilizers, animal manure, wastewater effluent, atmospheric deposition, plant residues, and nitrogen fixation by soil organisms. Nitrogen fertilizer is used extensively throughout the Midwest to increase corn (*Zea mays* L.) yields. Nitrogen application rates are larger for continuous corn (CC) rotations than corn - soybean (*Glycine max* (L.) Merr.) (CS) rotations (Sawyer et al., 2006), where the economic net return for fertilizer application is greatest. With the additional

demand for corn for the ethanol industry and animal production, crop rotations have been shifting from a corn following soybean rotation (CS) to a CC rotation (Feng et al., 2008). Because of the greater application rate, N loadings have been found to be greater in streams with higher percentage of CC rotation (Hernandez-Ramirez et al., 2011).

Traditional patterns of livestock production have changed substantially over the past 20 years in both Iowa and Minnesota. Although the total number of farms raising hogs decreased approximately 74% in Iowa and 67% in Minnesota, from 1992 to 2007, the number of large feeding operations with greater than 1,000 hogs increased during this time period. The number of large feeding operations increased by 27% in Iowa and 39% in Minnesota (USDA, 1992a,b, 2007a,b). Many studies have documented the effects of non-point source pollution associated with increased concentrations of swine and the challenges with manure management (Burkholder et al., 2007; Cook and Baker, 2001). Challenges to efficient use of manure include: variability of nutrient concentrations in manure, subsurface drainage which may transport nutrients from soil before the crop can use them, increased concentrations of nutrients (especially phosphorus) in soil accumulating from previous applications, balancing of application rates for nitrogen and phosphorus (Cook and Baker, 2001; Karlen et al., 2004), and technologies to better synchronize manure nutrient application and crop uptake availability. Manure is often applied to fields during the fall when soils tend to be drier and less prone to compaction by application equipment, but this is also when crops are not growing and nutrients in the manure have the potential to be transported to nearby streams via surface runoff or through subsurface drains (Cook and Baker, 2001). Application of manure and/or fertilizer nutrients exceeding the needs for residue decomposition and direct crop uptake can create an increased risk for off-site

movement resulting in nutrient contamination of surface waters (Burkholder et al., 2007; Cook and Baker, 2001; Geohring et al., 2001; Stamm et al., 2002).

Many studies have evaluated nutrient transport in rivers and streams in relation to nutrient inputs, but few studies have evaluated this relation over periods of more than a few years. It is known that variability in annual precipitation and subsequent stream runoff is an important factor in nutrient transport and thus there is a need to quantify differences between watersheds during periods of low and high runoff that occur in a cyclic nature that may be a decade or more in length. Furthermore, with land use changes that have occurred across the U.S. Midwest in the past decade, the impacts of changes in cropping patterns, land use, and fertilizer application rates and timing on stream water quality need to be considered in strategies to reduce transport of agricultural nutrients to rivers and streams. Iowa (Iowa Nutrient Reduction Strategy, 2014), Minnesota (Minnesota Pollution Control Agency, 2014), and other Midwestern States have developed separate strategies to address these issues.

The objective of this report is to compare transport of nitrate and total phosphorus in two intensively cropped agricultural watersheds as they relate to intensity of nutrient input from crop and livestock production, and observed variability in precipitation and hydrology. Our hypothesis is that although rainfall runoff may be the dominant factor in transport to streams, the effects of nutrient input can be differentiated from those of rainfall runoff. The ability to differentiate these factors will allow producers and land managers to better implement strategies that will efficiently reduce loss of nutrients from fields.

The study was conducted in the Little Cobb River, Minnesota and the South Fork Iowa River, Iowa from 1996 through 2007 that includes most of one quasi-periodic

runoff cycle that has a characteristic time scale of approximately 12 years (Green et. al, 2014). The primary nutrient species discussed in this report include dissolved nitrate plus nitrite as nitrogen ($\text{NO}_3\text{-N}$) and total phosphorus as phosphorus (TP). Analytical data from periodic samples and daily streamflow data were used to estimate daily and annual nitrogen and phosphorus concentrations and loads. Results are used to discuss differences in nitrogen and phosphorus transport over a range of climatic conditions in two watersheds with similar soil types and precipitation but with substantial differences in nutrient inputs.

2. Materials and methods

2.1 Study Watersheds

Both the Little Cobb River (LCR) and the South Fork Iowa River (SFIR) watersheds are located in the Central Iowa and Minnesota Till plains (Fig. 1), a landform region formed by the Des Moines Lobe of Wisconsin-age glaciation, and both watersheds are intensively managed for row-crop and livestock production. This region, sometimes called the prairie pothole region, has a temperate continental and humid climate. The mean annual temperature near SFIR is 8.1°C with a mean January temperature of -9.0°C and a mean July temperature of 22.8°C (NOAA, 2015). Temperatures are slightly cooler on average in the LCR, with a mean annual temperature of 7.1°C , a mean January and July temperature of 10.8°C and 22.3°C , respectively. The long-term average precipitation (1981-2010) in SFIR is 90.1 cm compared to 70.9 cm in LCR (NOAA, 2015). Annual precipitation during the study was slightly greater in the SFIR than the LCR (Table 1) but about 20% more snow fell

in the LCR (yearly average of 94.1) than in the SFIR (yearly average of 78.2 cm) (NOAA, 2015).

The LCR, located in south central Minnesota in Blue Earth and Waseca counties (Fig. 1), has a drainage area of approximately 321 km² (Table 1) and flows into the Le Sueur River which flows into the Blue Earth River which subsequently drains into the Minnesota River. The SFIR, a tributary to the Iowa River, is located about 150 km south of the LCR in north central Iowa in Hardin and Hamilton counties (Fig. 1) and has a drainage area of approximately 570 km² (Table 1). Soils in both watersheds formed in deep deposits of Wisconsinan-aged glacial till that were deposited during the advance and retreat of the Des Moines Lobe 12,000 to 15,000 years ago. Soils that are poorly or somewhat poorly drained, including the Nicollet and Webster series, have been artificially drained for agricultural production and an extensive network of subsurface (tile) drains is present in both LCR and SFIR.

2.1.1 Land use

Land use and crop rotations were mapped by overlaying USDA National Agricultural Statistics Service (NASS) crop cover data for 2006 and 2007 (USDA National Agricultural Statistics Service, 2012). Land use was similar in the two watersheds with more than 85 percent of the area used for row-crop agriculture (Table 1). However, the SFIR has a slightly greater percentage of grassland/pasture, which is mainly located along riparian valleys in the lower part of the watershed; the LCR which is tree lined in some areas has slightly larger percentages of wetland and open water than the SFIR (Table 1).

Differences in cropping patterns occur that are at least partially related to the intensities of animal production in the two watersheds. Whereas both watersheds are located in counties with some of the highest number of hogs in their respective states, the animal density is substantially greater in the SFIR watershed than in the LCR watershed. Blue Earth and Waseca Counties in Minnesota have the second and eighth (out of 87) greatest numbers of hogs, while Hardin and Hamilton Counties in Iowa have the second and eleventh (out of 99) greatest numbers of hogs (USDA National Agricultural Statistics Service, 2007a, b). In 2007, there were an estimated 200,000 hogs within and around (within 4 km of) the LCR versus an estimated 840,000 hogs in and around the SFIR. (C. Wolter, Iowa Department of Natural Resources [DNR] written communication, May 2012; Minnesota Pollution Control Agency [MPCA] 2009).

2.1.2 Nutrient inputs

Detailed information on nutrient input in the LCR and SFIR watersheds are not readily available, but estimates based on county level estimates (Supplemental Table S1) indicate that annually about 10% more nitrogen is input into the SFIR (134 kg ha^{-1}) than is input into the LCR (121 kg ha^{-1}). There was very little difference in the annual estimated input of phosphorus between the LCR (18 kg ha^{-1}) and the SFIR (19 kg ha^{-1}). More than 60% of the nitrogen input into each watershed originated from fertilizer and animal manure.

Based on a map survey of livestock facilities for the SFIR, Tomer et al. (2008) estimated that 62% of the cropland in the SFIR watershed received manure applications (Supplemental Material). Given differences described above, the number of hectares in each watershed and percent of watershed for CC and CS with and without manure application differ between the

two watersheds; LCR has a higher percent of the watershed of CS than SFIR (Table 2) but SFIR has higher percent of the watershed of CC, CC with manure, and CS with manure.

2.2 Data collection

Stream discharge (streamflow) and water quality data used to estimate loads were collected from the LCR near Beauford, MN and the SFIR northeast of New Providence, IA. (Fig. 1). Water-quality samples were collected to determine concentrations of a full suite of nutrient species and suspended sediment (Supplemental Table S3). Samples were collected one to four times per month beginning in March 1996 and continuing through 2007. The sampling frequency was generally on a monthly basis in fall, winter and early spring increasing to weekly during the growing season from about April through August. Sample collection was discontinued on the LCR from August 1998 to March 1999, February 2000 to March 2001, and August 2003 to March 2004. Samples were not collected from the SFIR from September 1998 through March 1999. Although not collected during every storm event, additional samples were collected during high flow and flood conditions to ensure nutrient data were available through the entire range of streamflow during the study.

Representative samples were collected using an isokinetic sampler from multiple verticals across the stream following equal width-increment protocols and composited into a single sample (USGS, 2006). In addition to the regular ambient samples, quality assurance samples were collected that included blank samples to ensure cross contamination did not occur and replicate samples were collected to document analytical variability (Mueller et al., 1997).

Once samples were collected from the river and composited, the water was processed at the field site. One sub-sample, designated for analysis of TP, was collected unfiltered. A second sub-sample designated for analysis of dissolved NH_4 , NO_3 , and SRP was filtered using a 0.45- μm capsule filter and was preserved with 2.25 M sulfuric acid. Processed samples were chilled and shipped on ice to maintain 4°C to the USGS National Water Quality Laboratory (NWQL) in Denver, CO for analysis. Analyses conducted on water samples at the NWQL were as follows. Total N was analyzed using alkaline persulfate digestion with a RL of 0.1 mg N L⁻¹ (Patton and Kryskalla, 2003). Total P and SRP were analyzed using semi-automated colorimetry with an RL of 0.008 mg P L⁻¹ (USEPA method 365.1; Clesceri et al., 1998). Dissolved N species were analyzed using colorimetry and NO_3 -N had an RL of 0.04 mg N L⁻¹ (Patton and Kryskalla, 2003).

Daily streamflow was calculated from stream stage recorded at 15-minute intervals using a bubble gage coupled with a nonsubmersible pressure transducer (Sauer and Turnipseed, 2010) and stage-discharge ratings developed for each site from periodic flow measurements (Rantz, 1982). Daily streamflow data were available for the LCR from April 1996 to September 1999 and from June 2001 to September 2007. Daily discharge was available for the SFIR site through the entire study period from October 1995 to September 2007. Stream flow data at both sites were downloaded from the USGS National Water Information System (NWIS) database (U.S. Geological Survey, 2014).

2.3 Data analysis

Periodic sample concentrations and continuous streamflow data were used to calculate daily, monthly, and yearly loads using the Weighted Regressions on Time, Discharge, and Season (WRTDS) model (Hirsch et al., 2010) ; summarized in the Supplemental Material).

The WRTDS model was calibrated using daily flow and periodic water quality data that extended beyond (through 2011) the study period to eliminate potential model instability near the end of the modeled period. However, model results were used for this report only from the study period of 1996-2007. The WRTDS estimates generally were within 10 percent of the observed concentration and load (Supplemental Table S2; Fig. S2).

Annual $\text{NO}_3\text{-N}$ and TP load and yield were calculated by water year (October through September of the following year); the beginning of the water year roughly demarks the time of year when water use by annual row crops ceases. Estimated yearly loads for the entire study period at SFIR are summarized but only loads from the 1996-1999 and 2002-2007 periods were used for direct comparison to those from LCR.

3. Results

3.1 Streamflow

Streamflow, generated by runoff of snowmelt and rainfall, annually averaged $2.07 \text{ m}^3 \text{ s}^{-1}$ in the LCR and $4.36 \text{ m}^3 \text{ s}^{-1}$ in the SFIR during the 1996 to 2007 period (Table. 1). Average streamflow at the time of sample collection was $2.83 \text{ m}^3 \text{ s}^{-1}$ in the LCR and was $5.69 \text{ m}^3 \text{ s}^{-1}$ in the SFIR (Supplemental Table S3). The annual average streamflow was much more variable in the SFIR than in the LCR. The annual average streamflow had a CV of about 35% in the LCR and the CV of discharge was >50% in the SFIR. Streamflow was not recorded in the LCR in 2000, but the minimum annual average streamflow was $0.72 \text{ m}^3 \text{ s}^{-1}$ in 2002 and the maximum annual average streamflow was $3.23 \text{ m}^3 \text{ s}^{-1}$ in 1999 (Table. 3). The annual average streamflow in the SFIR ranged from a low of $1.04 \text{ m}^3 \text{ s}^{-1}$ in 2000 when regional rainfall was below normal (NOAA, 2015) to a maximum of $9.83 \text{ m}^3 \text{ s}^{-1}$ in 2007

(Table. 3) when heavy rains fell during August (NOAA, 2015). The timing of the peak monthly streamflow in the SFIR was delayed by two months in relation to the peak monthly streamflow in the LCR (Fig. 2). The average peak monthly streamflow in LCR occurred in April during or just after snowmelt. In contrast, the average peak monthly streamflow in SFIR was in June near the beginning of the growing season.

Accounting for differences in drainage area, the amount of water discharged from the SFIR watershed was greater than that from the LCR watershed. The above differences in streamflow resulted in an annual median annual runoff of 18.3 cm from the LCR in comparison to of 19.1 cm of runoff in the SFIR watershed (Table 1). Annual runoff occurred in a cyclical pattern in both watersheds with relatively high runoff in late 1990's followed by the by low runoff in the early 2000's with a subsequent increase in runoff through the last half of the study (Table. 3).

3.2 Nutrient Concentrations

Significant water quality differences were observed between samples collected from the LCR and the SFIR. Differences at the time of sample collection were noted in physical and chemical characteristics such as sampled streamflow, pH, dissolved oxygen, and specific conductance (Supplemental Table S3). Differences in nutrient concentrations were not consistent between streams for selected nitrogen and phosphorus species. There were differences in nitrogen and phosphorus concentrations that were related to stream discharge and seasonality.

Nitrate concentrations differed seasonally between the rivers and generally varied in relation to streamflow. The $\text{NO}_3\text{-N}$ concentrations increased with increasing streamflow

during the spring into the summer (Fig 3) and then decreased through the remainder of the growing season, as crop water use increased and streamflow decreased. Concentrations then increased during the fall, when streamflow responses to rainfall increased as crops matured and were harvested. Monthly median $\text{NO}_3\text{-N}$ concentrations in the SFIR increased steadily from $4.8 \text{ mg NO}_3\text{-N L}^{-1}$ in January when flow was minimal, to $19.4 \text{ mg NO}_3\text{-N L}^{-1}$ in June when average streamflow was the greatest. Monthly median $\text{NO}_3\text{-N}$ concentrations then decreased as summer progressed and crop development was greatest, and the lowest monthly concentration ($0.12 \text{ mg NO}_3\text{-N L}^{-1}$) occurred in September before flow increased again through the fall and into the winter (Fig. 3).

The seasonal pattern of $\text{NO}_3\text{-N}$ concentrations in the LCR was similar to that in the SFIR with a couple notable exceptions. First, the greatest monthly median concentration in June was $11.5 \text{ mg NO}_3\text{-N L}^{-1}$; almost $8.0 \text{ mg NO}_3\text{-N L}^{-1}$ less than in the SFIR. Second, monthly median $\text{NO}_3\text{-N}$ concentrations in the LCR decreased through the summer but did not drop as low as those measured in the SFIR. The smallest monthly median concentration in the Little Cobb was $1.7 \text{ mg NO}_3\text{-N L}^{-1}$ in September. Third, rather than continuing to increase steadily from winter to late spring and early summer, median monthly $\text{NO}_3\text{-N}$ concentrations decreased from January through March before again increasing to peak concentrations in June.

The monthly median TP concentrations were present in a bimodal pattern with peak concentration in March and in June in the SFIR (Fig 3). Concentrations of TP increased through the winter and peaked in the March snowmelt period. Monthly median TP concentrations increased from $0.03 \text{ mg TP L}^{-1}$ in December to $0.13 \text{ mg TP L}^{-1}$ in March. After again peaking at $0.13 \text{ mg TP L}^{-1}$ in June when streamflow was the greatest, monthly

median TP concentrations in the SFIR decreased steadily during the remainder of the summer as streamflow decreased (Fig. 2) until October when the median monthly concentration was $0.05 \text{ mg TP L}^{-1}$. Monthly median TP concentrations also were present in a bimodal pattern (Fig. 3) in the LCR with a pronounced peak of $0.27 \text{ mg TP L}^{-1}$ in the March snowmelt period and a less distinct peak of $0.24 \text{ mg TP L}^{-1}$ in July as streamflow was decreasing. Median monthly TP concentrations remained between $0.20 \text{ mg TP L}^{-1}$ and $0.24 \text{ mg TP L}^{-1}$ between June and September in the LCR as streamflow decreased (Fig. 2).

Concentrations of $\text{NO}_3\text{-N}$ and TP varied with streamflow. The relation between the constituent and streamflow differed between the dissolved $\text{NO}_3\text{-N}$ and TP. $\text{NO}_3\text{-N}$ concentrations increased as flow increased from very low flow to average flow (Fig. 4). As flow increased beyond average flow, the $\text{NO}_3\text{-N}$ concentration became constant but then decreased because of dilution or source limitation during the highest flow conditions.

Rather than increasing, TP concentrations remained relatively constant as flow increases from low to medium flow conditions in the LCR and SFIR (Fig.4). TP concentrations only increased as streamflow exceeded average flow conditions. Total P concentrations increased rapidly in the SFIR as streamflow exceeded $10 \text{ m}^3 \text{ s}^{-1}$ but this rapid increase at high flows was not as apparent in the LCR (Fig. 4). The TP concentrations were greater in the LCR than in the SFIR during low to medium flows, but were greater in the SFIR than in the LCR during high flows.

3.3 Nutrient Loads

The annual estimated $\text{NO}_3\text{-N}$ load in the LCR ranged from about 237 Mg in 2002 when runoff was 6.74 cm to 1,370 Mg in 1999 when runoff was $>30 \text{ cm}$. The annual estimated

NO₃-N load in the SFIR ranged from about 460 Mg in 2000 when runoff was least (5.65 cm) to more than 3,500 Mg in 2007 when runoff was greatest (53.4 cm; Table 3). The SFIR watershed yielded significantly ($P < 0.05$) more NO₃-N (average of 31.2 kg NO₃-N ha⁻¹ y⁻¹) than did the LCR watershed (average of 21.3 kg NO₃-N ha⁻¹ y⁻¹; Table 3). The estimated annual NO₃-N yield in the LCR during this study is almost identical to that estimated (21.0 kg NO₃-N ha⁻¹ y⁻¹) by Randall and Mulla (2001) in watersheds of the Blue Earth, Le Sueur, and Watonwan Rivers of MN during the late 1980's and early 1990's. Schilling and Wolter (2005) used a regression equation based on the percent of row crop in watersheds to estimate that 21.5 kg NO₃-N ha⁻¹ y⁻¹ was contributed from Des Moines Lobe watersheds. This estimate is similar to our average annual yield from the LCR (21.3 kg ha⁻¹ y⁻¹) but substantially lower than our estimate for the SFIR (31.2 kg ha⁻¹ y⁻¹). Schilling and Wolter's (2005) approach does not account for the amounts of manure that are applied to row crops in the SFIR. Average annual total N loads from a previous 2002-2005 study at four sites in the South Fork Iowa River (Tomer et. al., 2008) ranged from 18.5 to 25.8 kg NO₃-N ha⁻¹ y⁻¹. The average annual yield from 1997-2007 for the SFIR estimated using the WRTDS was slightly greater than the annual average of 27.2 kg NO₃-N ha⁻¹ y⁻¹ found recently (2009-2013) for the Boone River in Iowa (Ikenberry et al., 2014). In Illinois, the annual NO₃-N yields from the Embarrass and Kaskaskia River watersheds were 28.3 and 29.9 kg NO₃-N ha⁻¹ yr⁻¹ during the 1998-2005 period (Royer et al., 2006). The averages are similar to the annual average yield (29.0 kg NO₃-N ha⁻¹ yr⁻¹) for the SFIR but were significantly ($P < 0.05$, Wilcoxon ranked sum) greater than the average annual yield from the LCR (21.0 kg NO₃-N ha⁻¹ yr⁻¹) for this period.

The annual estimated TP load in the SFIR ranged from 7.9 Mg in 2000 to 155.0 Mg in 2007, and in the LCR ranged from 4.9 Mg TP in 2002 to 31.2 Mg TP in 1999 (Table 3). As with the estimated NO₃-N load, the greatest TP loads occurred during the greatest runoff and the smallest loads were estimated during the years with the smallest runoff in both watersheds. Although concentrations were greater (Supplemental Table 3) in the LCR than in the SFIR, the SFIR watershed yielded significantly more TP ($P < 0.05$) than the LCR watershed (1.13 versus 0.51 kg TP ha⁻¹yr⁻¹). Estimated yields ranged from 0.38 kg TP ha⁻¹ to 0.85 kg TP ha⁻¹ in the SFIR from 2002-2005; this was similar to the range of 0.4 to 0.7 kg TP ha⁻¹ estimated for four sites in the SFIR watershed as part of an independent study by Tomer et al. (2008). Annual average yields from the Embarrass (0.83 kg ha⁻¹) and Kaskaskia (0.67 kg ha⁻¹) River watersheds in Illinois during the 2000-2006 periods (Royer et al., 2006) were greater than average annual yields from the LCR and SFIR (Table 3) during the same period.

Data collected at the LCR and SFIR sites were also used by Sprague and Gronberg (2012) to document nutrient transport in the United States. Yearly average load during this 1997-2007 study was similar to that at other Midwestern rivers. Average annual NO₃-N yield in the LCR (21.3 kg NO₃-N ha⁻¹) was in the highest category modeled by David et al., (2010) in the Mississippi River Basin during the 1997 to 2006 period. The estimated average annual NO₃-N yield in SFIR (31.2 kg NO₃-N ha⁻¹) exceeded the maximum per county estimates of NO₃-N loads made by David et al (2010).

3.3.1 Relation to runoff

NO₃-N and TP loads varied from year to year during the 1997-2007 study period due to differences in rainfall and runoff. Annual yield of NO₃-N and TP increased linearly with

increases in annual runoff (Fig.5, Table 4). The slope of the relation between annual $\text{NO}_3\text{-N}$ yield and annual runoff are not significantly ($P>0.05$) different in the LCR and SFIR (Fig. 5). Although annual yield increased with increasing annual runoff from both watersheds, the LCR watershed yielded smaller amounts of $\text{NO}_3\text{-N}$ than the SFIR for a similar runoff. For example an annual runoff of 25 cm would yield an estimated 28.9 kg $\text{NO}_3\text{-N ha}^{-1}$ from the LCR and 33.1 kg $\text{NO}_3\text{-N ha}^{-1}$ from the SFIR. These results are similar to those modeled by Nangia et al. (2010) for two small watersheds on the Des Moines Lobe in south central Minnesota. Nangia et al. (2010) concluded that a negative offset in the $\text{NO}_3\text{-N}$ yield-precipitation relation would result from a reduction in nitrogen application rates. The annual $\text{NO}_3\text{-N}$ yield from a watershed would be reduced with a similar runoff. The LCR and SFIR watersheds have similar landuse, soils, drainage, and topography but the SFIR has substantially more nitrogen input from a combination of fertilizer (Table 2) and manure from hogs (Supplemental Material).

Annual TP yield increased linearly with increased annual runoff in the LCR and SFIR watersheds during 1997-2007 (Fig.5). The increased TP yield being disproportionate to runoff increases results from much of the TP being delivered during snowmelt- and rainfall-runoff events.

3.3.2 Relation to land use

Smaller amounts of $\text{NO}_3\text{-N}$ for a given amount of runoff transported in the LCR in relation to the SFIR suggests that land use factors in addition to streamflow or runoff affect the amount of N and P transported from these two Midwestern watersheds. Annual $\text{NO}_3\text{-N}$ and TN yield estimates accounting for differences in streamflow (flow normalized) were

constant through a range of dry to wet climatic conditions in both the LCR and SFIR. The slope of the regression equations (Table. 4) relating annual yield to annual runoff was not significantly [$P>0.05$] different than zero indicating that the flow normalized yield for both $\text{NO}_3\text{-N}$ and TP at both sites does not change with an increase in annual runoff. Because the slope of the flow normalized regression equations is zero, the y intercept is the estimated flow normalized yield. The estimated flow normalized $\text{NO}_3\text{-N}$ yield was 22.0 kg/ha in the LCR and 37.0 kg/ha in the SFIR. The estimated flow normalized TP yield was 0.58 kg/ha in the LCR and was 1.48 kg/ha in the SFIR. Differences in nitrogen and phosphorus input (Supplemental Table S1) and cropping patterns (Table 2) are possible land use factors that account for the significant difference in the estimated flow normalized yields from the LCR and SFIR watersheds. Flow normalized yields may be considered the “base” yield with the current landuse practices. Changes in the land-use practices, for example reduced nitrogen and phosphorus input, potentially would result in smaller base yield.

3.3.3 Seasonal transport

Transport of $\text{NO}_3\text{-N}$ and TP in the LCR and SFIR varied seasonally with runoff during the study period (Fig. 6). The three months with greatest monthly $\text{NO}_3\text{-N}$ and TP load were April, May, and June during late spring and early summer. However, the maximum $\text{NO}_3\text{-N}$ and TP monthly loads in the LCR occurred in April; two months before the maximum in the SFIR. Transport of $\text{NO}_3\text{-N}$ and TP in the latter half of the growing season generally accounted for less than 20 percent of the yearly total.

The average monthly nutrient load generally correlated to the average monthly runoff. Average monthly runoff in the LCR ranged from 0.3 cm in the winter months of December and January to 3.6 cm in April (Fig. 2). Peak runoff in the LCR occurred in April and was

caused by snowmelt and or rainfall on snow and frozen ground. The majority of runoff (72%) in the LCR occurred in January through June with more than 60% occurring during the last three months of the period. Average monthly runoff in the SFIR ranged from 0.5 cm in October to 5.2 cm in June (Fig. 2). Peak runoff in the SFIR occurred two months later than in the LCR, which was caused by proportionally greater snowmelt-runoff contributions occurring in the more northern LCR. Snowfall averaged 94 cm at Mankato, MN (NOAA,2015) near the LCR during the 1996-2007 study; 17 percent greater than at Eldora, IA near the SFIR (78 cm, NOAA, 2015). Peak runoff in the SFIR coincides with seasonal precipitation patterns that peak at a time before crop canopies are fully closed, after which crops intercept more precipitation and maximize their water use (Dinnes et al., 2002). A majority (75%) of the runoff in SFIR occurred from January through June with 57 percent occurring after most snowmelt, from April through June.

The average estimated monthly $\text{NO}_3\text{-N}$ load in the LCR ranged from 12.1 Mg in January to 180 Mg in April (Fig. 6). The three greatest monthly loads were April, May, and June which accounted for 66 percent of the yearly $\text{NO}_3\text{-N}$ load. The average monthly $\text{NO}_3\text{-N}$ load in the SFIR ranged from 35.6 Mg in January to 465 Mg in June. The $\text{NO}_3\text{-N}$ load in April through July was almost 80 percent of the yearly total in the SFIR.

The average estimated monthly TP load in the LCR ranged from 0.10 Mg in January to 3.56 Mg in April. Slightly more than 50 percent of the TP was transported during the months of April, May, and June. The smallest average estimated monthly load (0.41 Mg) in the SFIR occurred in October during early fall. The maximum estimated monthly TP load was 21.7 Mg in June. Sixty five percent of the yearly TP load in the SFIR occurred during the April through June period.

4. Discussion

More $\text{NO}_3\text{-N}$ and TP are transported from the SFIR watershed than from the LCR watershed and this difference was attributed to both differences in rainfall runoff and several related land use factors (animal production and crop rotation) that result in greater nutrient inputs into the SFIR watershed. Streamflow in the LCR and SFIR watersheds was important in determining the amounts and timings of nitrogen and phosphorus transported from these two watersheds. Increased amounts of $\text{NO}_3\text{-N}$ and TP concentration occurred during the months and years with the greatest rainfall.

As streamflow increased from low to mean flow, $\text{NO}_3\text{-N}$ concentrations increased dramatically and then remained constant or continued to slowly increase before decreasing as streamflow approached highest flow conditions (Fig. 4). In contrast, TP concentrations remained relatively constant as flow increased through mean flow and concentrations only began to increase when flow exceeded several times the mean flow (Fig.4). The dissimilarity in the relations between concentration and flow for $\text{NO}_3\text{-N}$ and TP suggest different transport paths.

During periods of low flow, transport of $\text{NO}_3\text{-N}$ was almost negligible because groundwater-sourced baseflow was dominant. Baseflow occurs along subsurface paths with low velocities and long travel times that are subject to biogeochemical processes including denitrification. As flow increases, greater contributions are sourced from tile drainage that occur over shorter distances and durations than groundwater flow; tile drainage water can carry substantial amounts of $\text{NO}_3\text{-N}$ leached from agricultural soils (Dinnes et al., 2002; Tomer et al., 2010). At high flows when overland flow from rainfall-runoff events substantially contributes to stream flow, lower $\text{NO}_3\text{-N}$ concentrations may occur due to

dilution especially at the greatest discharge rates. Transport of TP dominantly occurs with suspended sediment. Although tile flows can carry significant concentrations of SRP (Tomer et al 2008), most TP transport occurs during runoff events; overland flow may transport sediment into the stream at these times but stream banks and bed loads are also important in TP transport (Tomer et al., 2010). Greater TP loads and yield occurred in LCR during early spring with snowmelt and rainfall on bare ground. Greater TP loads and yields occurred in SFIR in late spring/early summer when the greatest rainfall occurred.

Streambank erosion has been implicated in the transport of phosphorus during high streamflow including in the SFIR (Tomer et al., 2010; Yan et al., 2010). In Minnesota, one study in the Blue Earth River (Sekely et al., 2002) determined that stream bank erosion contributed from 31 to 44 percent of the sediment but less than 10 percent of the TP transported from the watershed. The small amount of TP transport is due to the small TP concentrations in the lower bank strata (Sekely, et al., 2002).

4.1 Timing of Transport

Not only is the amount of NO₃-N and TP transported from the LCR and SFIR different, but the greatest transport occurs in early spring in the LCR in contrast to early summer in the SFIR. The difference in timing of the transport of NO₃-N and TP is likely due to a combination of factors that include the timing of peak streamflow and availability of nutrients. Greater snowpacks accumulate and persist during winter as one moves north from Iowa to Minnesota, hence substantial snowmelt transport of nutrients occurs more consistently in the LCR compared to the SFIR. Timing of peak discharge occurs later in the SFIR, supporting this interpretation. Previous studies (Kalkhoff et al., 2000; Tomer et al., 2003; Tomer et al., 2008) have shown that a substantial amount of the annual NO₃-N load is

transported from Des Moines Lobe watersheds in late spring before establishment of a crop cover. The greater extent of slopes >2% (Table 1) also provide an environment in which snowmelt is more subject to runoff in the LCR.

4.2 Effect of nutrient input on nitrogen and phosphorus transport

Less NO₃-N and TP were transported from the LCR than the SFIR and this difference may be attributed to several land cover factors including crop rotation, use of harvested crops for both ethanol and livestock production, and a greater extent of wetlands and lakes in the LCR. Although the proportion of land used for row crops is similar in each watershed, SFIR has a greater percent of CC rotation (see above) compared to LCR in 2006-2007 (Table 2). The N application rate on average is more for CC than CS (Sawyer et al., 2006) and therefore, CC is associated with a larger N loading to streams (Hernandez-Ramirez et al., 2011). In addition, the N application rate is higher in the SFIR because the longer growing season in Iowa results in larger expected grain yields (Sawyer et al., 2006). The increase in extent of CC rotations in the SFIR was in response to an increased demand for corn for ethanol, as well as for swine production. Livestock facilities within the SFIR have almost 3.5 times more swine than those in the LCR, according to state permit information, although both watersheds are used for hog production. The total hogs increased from 1992 to 2007 and Hardin County in Iowa had the highest number compared to the other counties in Iowa. Animal density has been linked to high NO₃-N concentration in eastern Iowa Rivers (Welden and Hornbuckle, 2006) and long-term manure application can lead to accumulation of nutrients and eventually leaching of nitrogen (Bergstrom and Kirchmann, 1999; Edmeades, 2003). Hog feeding operations require roughly one acre (0.4 ha) of corn plus one acre of soybeans, in relatively equal proportions, to provide feed to produce 20

head of hogs (B. Kerr, USDA Agricultural Research Station, personal communication, 2012). If facilities send two herds of swine to market in a year, about twice the land in CS rotation in the SFIR is required to provide feed to meet the demand of swine feeding operations in the watershed (at capacity). The land in CC provides a location for manure disposal near the animal production facility to minimize the average haul distances for manure. Whereas the close proximity of land in CC provides benefit to the cost of swine production, the CC rotation satisfies the market for corn ethanol rather than a balance of corn and higher-protein soybean that best suits nutritional requirements for swine production. This results in a net importation of feed to the SFIR watershed. We also note that whereas fall applications of anhydrous NH_3 must be delayed until the soil cools in the fall to minimize nitrification to leachable NO_3 , this is not required for manure applications despite the fact that the dominant form of N in swine manure is NH_3 . Additionally, runoff from approximately one-third of the LCR watershed passes through 0.34 km² Severson Lake which could lower nutrient yields by increasing residence time and attenuating nutrients, however, no known nutrient data exists for lakes in the LCR.

Given the detailed land use data for 2007 (Supplemental Figure S1), the $\text{NO}_3\text{-N}$ yield in streamflow was compared on a unit area basis and the assumed unit area application of N (fertilizer plus manure). During 2007, yields of $\text{NO}_3\text{-N}$ were estimated to be 17.9 kg $\text{NO}_3\text{-N}$ ha⁻¹ from the LCR and 60.4 kg $\text{NO}_3\text{-N}$ ha⁻¹ from the SFIR (Table 3). If the rates of N application in the two watersheds based on Sawyer et al. (2006) and the maximum economic rates are assumed, the estimated streamflow transport represents the equivalent of 25% of the watershed wide N application rate (assuming non cropped lands had no N applied) in the LCR but were equivalent to 45% of the watershed-wide N application rate in the SFIR

during 2007. This result is for only one year of data and the difference could be due to a number of reasons including N uptake or denitrification in Severson Lake in the LCR. Whereas actual N applications in either watershed are not known, the magnitude of the difference in stream N losses relative to a reasonable assumption of maximum N application does suggest increased rates of N loss from cropland under the CC rotation. Plot-scale research demonstrated NO₃-N losses substantially increased where fall applications of swine manure were combined with CC rotations (Hernandez-Ramirez et al., 2011).

5. Conclusions

The results of this study document the importance of both runoff and intensity of nutrient inputs on the quantity of nutrients transported in two Midwestern rivers. These data provide useful information to water managers and producers as they consider which nutrient reduction strategies to use to efficiently reduce nutrient transport in Midwestern streams. This work suggests that strategies that mitigate both the effects of increased runoff and nutrient inputs could produce the best results. This work also shows that consideration of the timing of transport is important in the selection of a strategy that will produce the optimum level of reduction. The data collected during this decade-long study provide baseline information on nutrient transport. These data along with other State and Federal monitoring data could be used by water managers and producers to evaluate the implemented nutrient reduction strategies and to inform water managers and producers if modification to reduction strategies are needed.

These results provide information that can be used to select the nutrient management strategy that will most effectively reduce transport in spring and during the growing season.

The differences in the amount and timing of nutrient transport from watershed in close proximity also suggest that a regional approach to nutrient reduction strategies may be problematic and that nutrient transport can be reduced when differences in climatic factors such as timing of precipitation and length of frost-free conditions are considered.

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Tables

1. Watershed characteristics of the Little Cobb River, MN (LCR) and South Fork Iowa River, IA (SFIR).
2. Crop rotation in the Little Cobb River, MN (LCR) and the South Fork Iowa River, IA (SFIR), 2007.
3. Estimated annual nitrate and total phosphorus load and yield Little Cobb River near Beauford, MN and South Fork Iowa River ne of New Providence, IA, 1997-2007.
4. Relation between annual runoff and annual nitrate and total phosphorus yield in the South Fork Iowa River, IA and the Little Cobb River, MN, watersheds, 1997-2007

Illustrations

- Figure 1.- Location of the Little Cobb River, MN(LCR) and the South Fork Iowa River, IA (SFIR) watersheds and data collection sites . [Central Iowa and Minnesota Till Prairies from USDA, 2006].
- Figure 2.- Monthly average precipitation, streamflow, and runoff in the Little Cobb River, MN and South Fork Iowa River, IA basins. [Average (1981-2010) monthly precipitation from NOAA, 2015 ; South Fork data from Edora, IA weather station and Little Cobb data from Mankato, MN weather station]

Figure 3.- Variability in nitrate nitrogen and total phosphorus concentrations in the Little Cobb River, MN and the South Fork Iowa River, IA, 1997-2007.

Figure 4.- Nitrate-N, total phosphorus, and suspended sediment concentrations in relation to streamflow in the Little Cobb River, MN and the South Fork Iowa River, IA and, 1997-2007 [trend line is loess smooth; average 1997-2007 streamflow, vertical lines (solid-South Fork Iowa River; dashed-Little Cobb River)]

Figure 5.- Annual nitrate-N and total phosphorus yield in relation to annual watershed runoff, in the Little Cobb River, MN and the South Fork Iowa River, IA , 1997-2007 [The regression equations for yield versus runoff relation are given in table 4]

Figure 6.- Average monthly nitrate nitrogen and total phosphorus load in the Little Cobb River, MN and the South Fork Iowa River, IA, 1997-2007.

Table 1--. Watershed characteristics of the Little Cobb River, MN (LCR) and South Fork Iowa River, IA (SFIR).
[km², square kilometers; cm, centimeter; m³ s⁻¹, cubic meters per second; %, percent; rainfall data from NOAA (2015); runoff and streamflow from U.S. Geological Survey (2014); and National Land Cover Data from Homer et al. (2004)]

Sampling site			Watershed							
USGS site identification number	Site name	Drainage area (km ²)	Hydrology (period of record 1997- 1999; 2002-2007)			Landuse				
			Median annual rainfall (cm)	Median annual runoff (cm)	Median annual streamflow (m3 s-1)	0-2% Slope (percent of watershed)	Row-crop (%)	Wetland (%)	Open water (%)	Grassland/ pasture (%)
05320270	Little Cobb River near Beauford, MN	321	81.8	18.3	1.95	81	87	2.8	1	2.6
05451210	South Fork Iowa River ne of New Providence, IA	570	82.0	19.1	3.52	98	86	0.4	0.1	5.1

Table. 2 --Crop rotation in the Little Cobb River, MN
and the South Fork Iowa River, IA watersheds, 2007
[ha, hectares; %, percent]

Watershed	Crop rotation	Hectares (ha)	Fraction of watershed (%)
Little Cobb	Continuous Corn	4,730	14.74
South Fork	Continuous Corn	21,200	37.19
Little Cobb	Corn/Soybeans	20,000	62.31
South Fork	Corn/Soybeans	25,600	44.91
Little Cobb	Continuous Corn w/ manure	1,810	5.64
South Fork	Continuous Corn w/ manure	14,500	25.44
Little Cobb	Corn/Soybeans w/ manure	7,370	22.96
South Fork	Corn/Soybeans w/ manure	15,500	27.19

Table 3. --Estimated annual nitrate and total phosphorus load and yield
 Little Cobb River near Beauford, MN and South Fork Iowa River ne of New Providence, IA, 1997-2007
 [m³ s⁻¹, cubic meters per second; cm, centimeter; Mg, megagrams ; kg ha⁻¹, kilograms per hectare; N, nitrogen; P, phophorus; NA, no data]

Water Year	Annual average discharge (m ³ s ⁻¹)	Annual runoff (cm)	Annual nitrate load (Mg-N)	Annual nitrate yield (kg-N ha-1)	Flow weighted annual nitrate load (Mg-N)	Flow weighted annual nitrate yield (kg-N ha-1)	Total phosphorus load (Mg-P)	Total phosphorus yield (kg-P ha-1)	Flow weighted total phosphorus load (Mg-P)	Flow weighted total phosphorus yield (kg-P ha-1)
Little Cobb River near Beauford, MN										
1997	2.34	22.0	717	21.3	690	20.5	21.2	0.63	29.0	0.86
1998	1.93	18.1	659	19.6	719	21.4	16.6	0.49	27.5	0.82
1999	3.23	30.3	1,367	40.6	737	21.9	31.2	0.93	25.5	0.76
2000	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2001	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2002	0.72	6.74	237	7.03	750	22.3	4.88	0.14	21.3	0.63
2003	1.44	13.5	492	14.6	740	22.0	10.6	0.32	20.4	0.60
2004	1.95	18.3	573	17.0	728	21.6	23.1	0.69	19.3	0.57
2005	2.34	21.9	863	25.6	714	21.2	15.3	0.45	18.4	0.55
2006	2.66	24.9	943	28.0	699	20.8	16.8	0.50	17.6	0.52
2007	1.92	18.0	601	17.9	674	20.0	13.5	0.40	16.7	0.50
South Fork Iowa River ne of New Providence, IA										
1997	5.33	29.0	1,916	33.0	1,855	32.0	110	1.90	143	2.47
1998	6.17	33.6	2,431	41.9	1,970	34.0	154	2.65	127	2.19
1999	5.71	31.0	2,601	44.8	2,060	35.5	91.37	1.57	113	1.95
2000	1.04	5.65	459	7.91	2,110	36.4	7.90	0.14	99.7	1.72
2001	4.01	21.8	1,825	31.5	2,136	36.8	62.4	1.08	90.0	1.55
2002	3.20	17.4	1,460	25.2	2,142	36.9	22.0	0.38	82.9	1.43
2003	3.31	18.0	1,557	26.8	2,132	36.7	23.1	0.40	77.3	1.33
2004	3.05	16.7	1,460	25.2	2,116	36.5	29.0	0.50	72.7	1.25
2005	3.52	19.1	1,451	25.0	2,094	36.1	49.5	0.85	68.8	1.19
2006	2.93	15.9	1,261	21.7	2,064	35.6	17.9	0.31	65.5	1.13
2007	9.83	53.4	3,506	60.4	2,016	34.8	155	2.67	63.0	1.09

Table 4 -- Relation between annual runoff and annual nitrate and total phosphorus yield in the South Fork Iowa River, IA and the Little Cobb River, MN, watersheds, 1997-2007
 [All yields are in kilograms per hectare; runoff in centimeters; NO₃, nitrate nitrogen; TP, total phosphorus, FN, flow normalized]

Watershed	Constituent	Equation	Residual standard error	Adusted R- squared	P-value
†South Fork	nitrate	NO ₃ yield = 1.20(runoff)+ 3.11	2.61	0.92	<0.001
	total phosphorus	TPyield = 0.09(runoff) - 0.93	0.33	0.84	<0.001
Little Cobb	nitrate	NO ₃ yield = 1.35(runoff) - 4.84	2.79	0.91	<0.001
	total phosphorus	TPyield = 2.88(runoff) - 0.05	0.12	0.70	<0.001
Flow normalized					
South Fork	nitrate	FNNO ₃ yield = - 0.062(runoff) + 37.0	1.35	0.20	0.10
	total phosphorus	FNTPyield = - 0.004(runoff)+ 1.48	0.48	-0.10	0.75
Little Cobb	nitrate	FNNO ₃ yield = -0.04(runoff) + 22.04	0.75	-0.001	0.35
	total phosphorus	FNTPyield = 0.004(runoff) + 0.58	0.14	-0.10	0.64

† Regression does not include extremely wet year (2007)

Figure 1

Figure 2

Figure 3

Figure 4

Figure 5

Figure 6