Nutrient dynamics in Minnesota watersheds

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Dedication

This thesis, like every great adventure, is dedicated to the memory of Lucas Richardson.

Abstract

While excess nitrogen (N) and phosphorus (P) from anthropogenic activities are known to contribute to the eutrophication of aquatic ecosystems, curbing their inputs poses a management challenge due to poorly understood interactions between land cover, nutrient inputs, and climate. In chapter 1 we examined nutrient inputs, losses and retention in Minnesota watersheds, across a gradient of environmental variables. Fertilizer inputs were dominant sources of N and P inputs to agricultural watersheds, driving nutrient losses. Greater runoff decreased retention relative to inputs, suggesting increasing precipitation and continued hydrological modifications coupled with high nutrient inputs will contribute to sustained high rates of nutrient export. In chapter 2 we examined the factors controlling concentration-discharge relationships describing P and sediment mobilization in agricultural watersheds in Minnesota, assessed via analyses of exponents and coefficients of the relationship for 119 sites. These analyses were complemented by investigation of drivers of statewide annual P export, in which we observed dissolved P made up a significant proportion of annual loads. P and sediment were concentrated with greater discharge at most sites. Mean concentrations were elevated by anthropogenic land uses, and bluffs were associated with greater concentration of particulates. The mobilization of P is highly sensitive to discharge and its different forms deserve explicit consideration when managing nutrient losses.

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Chapter 1

The interactive effects of anthropogenic inputs and climate on nutrient loss and retention in Minnesota watersheds

Abstract

Nonpoint source pollution from anthropogenic activities contributes to eutrophication of aquatic ecosystems, and poses a management challenge due to poorly understood interactions between land cover, nutrient inputs, and climate. We examined nutrient inputs, losses and retention in Minnesota watersheds, ranging from predominantly urban and agricultural to predominantly forested, across a gradient of environmental variables. Anthropogenic inputs were the main driver of nutrient losses in watersheds with high nutrient export. Fertilizer inputs were dominant sources of nitrogen (N) and phosphorus (P) inputs to agricultural watersheds, exceeding, on average, imported food and feed and atmospheric deposition. P retention was high (average 77%) in predominantly urban and agricultural watersheds, but losses remained high in absolute terms (average 55 kg/km^2). N retention was also relatively high with elevated losses in human-dominated watersheds overall (average 75% and 1245 kg/km^2). Retention decreased and river export increased sharply for watersheds with the highest levels of N inputs and runoff. The relationships between retention and inputs were nonlinear and modified by runoff, such that greater runoff decreased retention relative to inputs. Without effective interventions to control nutrient losses from agricultural watersheds, increasing precipitation and continued hydrological modifications coupled with high nutrient inputs will contribute to sustained high rates of nutrient export and further exacerbate eutrophication impacts.

Introduction

Anthropogenic nitrogen (N) and phosphorus (P) inputs strongly impact aquatic ecosystems by altering nutrient cycles and trophic structure and increasing the abundance of pathogens, diseases, and toxic algae blooms (Carpenter et al. 1999; Dubrovsky et al. 2010; Baron et al. 2012). Whereas locating and regulating point sources is relatively straightforward, nonpoint source pollution is challenging to manage because of its diffuse nature and multiple contributors. Agricultural and urban lands are the primary sources of nonpoint nutrient loading (Bernhardt et al. 2008; Baron et al. 2012). In addition to increased nutrient inputs, pervasive influences of hydrological alterations in these areas also impact the export of nutrients from land to water, often leading to increases in concentration and loading to downstream waters.

Managing nutrient cycling in human dominated landscapes is critical toward controlling eutrophication, but is limited by current understanding of the landscape and climate factors driving changes in N and P loading. General relationships between land use/land cover (LULC), nutrient inputs, and water quality are well known (e.g. Walsh et al. 2005; Bernhardt et al. 2008). For example, watersheds with high urban and agricultural land development have elevated nitrogen and phosphorus concentrations compared to undeveloped watersheds (Dubrovsky et al. 2010). However, identifying the nonpoint sources of nutrients leading to eutrophication and the environmental factors that contribute most to their losses presents a scientific challenge because of complex interactions that mediate transport and processing in response to management and climate change. Understanding sources of N and P is critical to reducing pollution of downstream lakes and oceans. To understand the mechanisms driving relationships between LULC and nutrient retention it is important to identify landscape sources and processing in channels and water bodies as elements move downstream.

Differences in the N and P cycles may cause these nutrients to accumulate and mobilize differently in watersheds. In an analysis of three large watersheds in the United States, United Kingdom, and China, those with more prolonged intensive agricultural use exported more P than they received in inputs between 1990 and 2010, suggesting P accumulated during periods of over-fertilization will continue to mobilize long after a decrease in inputs (Powers et al. 2016). N fluxes were found to respond to fertilizer and manure applications on a much shorter timescale in California, where agricultural inputs to a small portion of watersheds were found to be the main driver of variation in N concentrations and led to changes to the seasonality of fluxes (Sobota et al. 2009). While N accumulated during dry periods in these watersheds, N is much less likely than P to adsorb to soils and persist in watersheds over longer periods of time. Quantifying watershed nutrient inputs and losses is one approach to understanding how N and P added to watersheds relates to riverine export.

Nutrient budgets have long proven useful to understanding basic processes controlling the flow of materials through watersheds. More recently, these methods have been applied to landscape-level studies of multiple watersheds to determine the effects of anthropogenic fluxes on nutrient export. Net anthropogenic N and P inputs (NANI & NAPI) quantify the human contribution to watershed nutrient cycling, by calculating the difference between nutrient import (such as fertilizer inputs, imported food, atmospheric deposition, etc.) and export (via food and feed production). In agricultural basins, anthropogenic nutrient fluxes such as fertilizer, food and feed inputs may be much greater than river fluxes (Howarth et al. 2012; Powers et al. 2016). Greater NANI and NAPI result in larger pools of nutrients and increased hydrologic mobilization, thus increasing N and P export from watersheds (Howarth et al. 2012; Sharpley et al. 2014; Chen et al. 2015). Although these methods provide detailed accounts of inputs as they relate to nutrient export, they are limited in their ability to distinguish the mechanisms underlying the wide unexplained variation in observed loads within urban and agricultural watersheds. By looking at inputs and export in the context of landscape and climate variability, we aimed to explain the drivers of nutrient retention.

The Midwestern USA has some of the most productive and intensively farmed cropland in the world. The Midwest region is also the location of large tracts of protected land and an area characterized by gradients in climate, geology, and glacial history. Examining nutrients in the context of intra-regional heterogeneity can provide deeper insight into the factors that might influence nutrient export and the ways we manage it.

The trajectory of agricultural production in Minnesota has followed much of Midwest in transitioning from hay and small grain production to corn and soy production at varying points between the 1950s and present day (Foufoula-Georgiou et al. 2015). Changes in cultivation have been accompanied by different degrees of ditching, tiling, and wetland drainage (Schottler et al. 2014; Lark et al. 2015). These factors, combined with natural landscape heterogeneity and extensive water quality monitoring data, facilitate examining the influence of climate, land use change, and land cover on nutrient movement through human influenced watersheds.

In addition to high nutrient inputs from fertilizer application, hydrologic factors interact with land cover to determine nutrient fluxes. Greater proportions of NANI and NAPI are exported in years with greater precipitation, especially in watersheds modified by agricultural use (Chen et al. 2014; Chen et al. 2015). The combination of increased precipitation and artificial drainage has increased stream flows and decreased hydrologic transit times in agricultural Minnesota watersheds (Schottler et al. 2014; Foufoula-Georgiou et al. 2015; Danesh-Yazdi et al. 2016). Tile drains and channel straightening in agricultural areas may quickly transport water away from fields with minimal uptake and processing of nutrients, especially in years with elevated precipitation. Understanding how hydrology modifies N and P cycles is essential to quantifying the effects of other landscape variables on watershed nutrient export.

In this study we sought to determine how land cover, nutrient inputs, and climate interact to influence N and P losses in watersheds across Minnesota. The study watersheds spanned gradients of agricultural and urban land cover, anthropogenic inputs from fertilizer, food, and feed imports, annual runoff, and wetland presence.

Methods

Study System

Sixty-three watersheds in Minnesota were sampled by the Metropolitan Council (METC) and Minnesota Pollution Control Agency (MPCA) Watershed Pollutant Load

Monitoring Network (Fig. 1). Together these datasets describe comprehensive environmental and LULC gradients. Water samples and flow data were collected throughout the year with a focus on snowmelt and storm event sampling (average 35 samples per year; Minnesota Pollution Control Agency 2016). We averaged annual total phosphorus (TP) and total nitrogen (TN) loads for sites with at least two years of data between 2007 and 2011, a period with substantial variation in precipitation. This study focused on averaged conditions in order to examine watershed mass balances rather than sensitivity to interannual variability in climate or antecedent conditions. Loads were normalized by watershed area to produce average annual yields.

Watersheds also represent a wide variety of land cover, as defined by the National Land Cover Dataset (NLCD). Crop cover ranges from 0-91% (average 48%) and urban cover ranging from 1-47% (average 6%). Connected lake and wetland cover, defined as lakes and wetlands within 100 m of a stream center line, ranged from 0-18% (average 3%) and 0-56% (average 6%) of watershed area respectively. Annual average precipitation and runoff in the watersheds ranged from 0.62 – 0.94 m (average 0.75 m) and 0.10 - 0.34 m (average 0.21 m) respectively during the five-year period. Five year average annual yields (2007 – 2011) encompassed a period of substantial variation in runoff. These conditions represent the typical range of conditions over the past decade, including wet and dry years with no multi-year droughts or flooding.

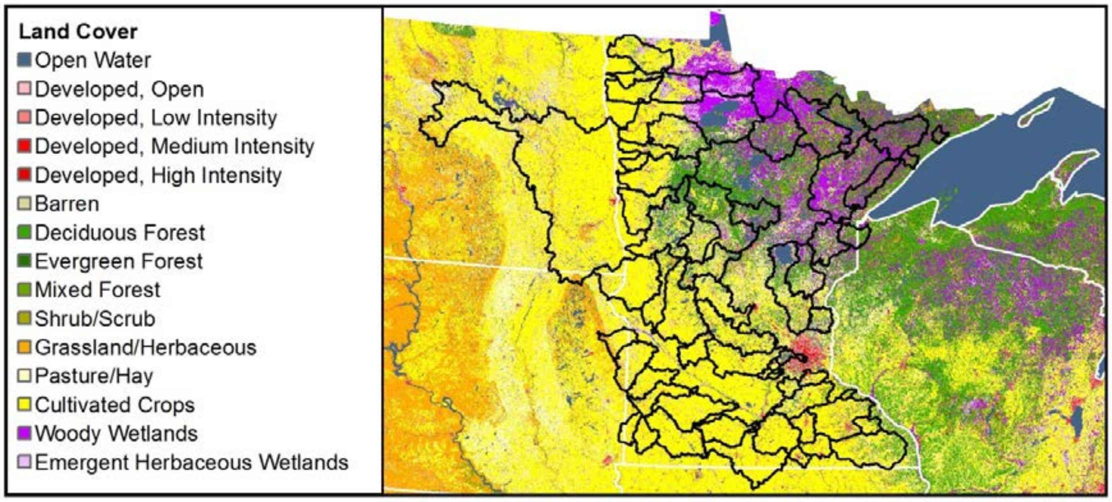


Figure 1. Study watersheds, outlined in black, and land cover information (NLCD 2011). All river monitoring stations were located in Minnesota, and span a broad range of environmental conditions.

GIS-Derived Variables

All watershed analyses were performed using watershed delineations provided by the MPCA and METC. The 2011 NLCD was used to determine the proportion of the watershed in land cover categories such as crops, pasture or hay, urban, open water, or wetland (Homer et al. 2015). The Minnesota Department of Natural Resources (DNR) updated National Wetlands Inventory (NWI), where it was complete for southern and east-central Minnesota, was used instead of the NLCD to determine the wetland land cover (Minnesota Department of Natural Resources 2015; US Fish and Wildlife Service 2015).

The NWI and Minnesota DNR updated NWI were used to determine connected wetland cover in watersheds. The USGS National Hydrography Dataset (NHD) was used to determine the connected lake and pond cover in watersheds. Wetlands and water bodies within 100 m of the center line of an NHD perennial stream were considered connected to the channel (Powers et al. 2013). One kilometer gridded precipitation data for 2007 – 2011 were downloaded from the PRISM Climate Group (2016) and used to determine mean annual precipitation for each watershed.

NANI, NAPI, and Retention

The NANI Version 3.0.1 and NAPI Version 3.0.1β toolboxes available from Cornell University were used to estimate NANI and NAPI for the 63 MPCA monitoring sites completely within the boundaries of the United States. Watersheds smaller than 150 km^2, the recommended minimum watershed size for this method of nutrient input calculation, were removed from this analysis. This removed mostly small urban sites from the dataset. Input data consisted of: (1) National Atmospheric Deposition Program (NADP) model outputs, (2) USGS mineral fertilizer input estimations, (3) the USDA agricultural census of crops and animals, and (4) the USA Census (Howarth et al. 1996; Boyer et al. 2002). This information was used to determine deposition, agricultural and non-agricultural fertilizer imports, agricultural N fixation and net food and feed import or export (natural biological N fixation, rock weathering, and septic leakage, and permitted discharges from point sources such as wastewater treatment plants are not included in input estimates). The balance of these components determines NANI. NAPI was estimated using the same input data, but assumes minimal atmospheric P deposition. The proportion of N and P retained by the watersheds was calculated as follows, in units of kg/km^2:

$$\frac{NANI - Hydrologic\ N\ Export}{NANI} \quad OR \quad \frac{NAPI - Hydrologic\ P\ Export}{NAPI}$$

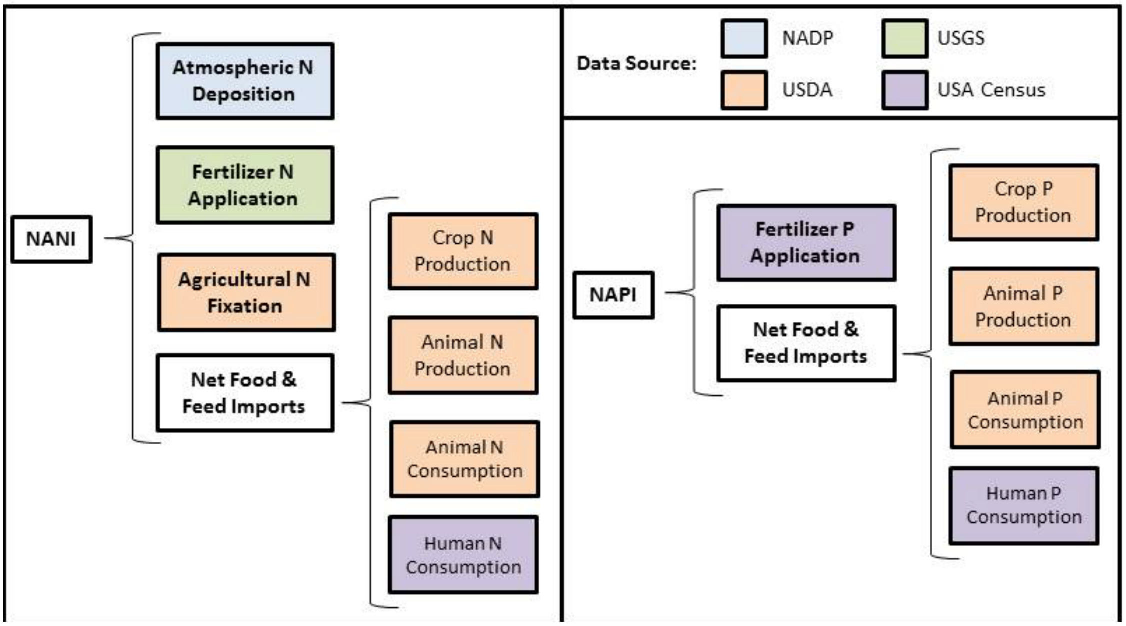


Figure 2. NANI and NAPI Toolbox inputs, intermediates, and output (modified from Hong et al. 2011). In this study, atmospheric deposition data was obtained from NADP model outputs, mineral fertilizer inputs were obtained from the USGS, crop and animal data were obtained from the USDA, and population data was obtained from the USA Census. The sum of these components is used to calculate NANI and NAPI.

Data Analysis

Nutrient losses and retention were related to climate, environmental, and landscape variables calculated using ArcGIS. Candidate explanatory variables included watershed land use, annual precipitation and runoff, NANI and NAPI, and fertilizer inputs of inorganic N and P. Multiple linear regression was performed JMP Pro 12 (SAS Institute, NC, USA). All forms of nutrient export, NAPI, watershed total and connected lake and wetland cover were log transformed to meet statistical assumptions. Backwards stepwise multiple linear regression variable selection was based on AIC.

Additionally, we constructed regression trees (RT) to examine the relationships between nutrient export & retention using our GIS-derived landscape variables. The regression tree approach uses explanatory variables to divide a response variable into increasingly homogeneous groups (De'Ath and Fabricius 2000; Borcard et al. 2011)**.** This approach can handle missing values, nonlinear relationships, and variables that are not normally distributed. The variance in the sum of squares explained by each split can be quantified, as well as the variance explained by the overall model (R^2). All analysis was performed in R with the *rpart* package (R Development Core Team 2014; Therneau et al. 2015)**.** Using the ANOVA method, we overfit each model and pruned the number of splits using the one standard error rule in order to ensure the creation of trees with robust predictive power (Borcard et al. 2011)**.** We used ANOVA to examine whether each group generated by our splits was significantly different.

Results

Study watersheds had a wide gradient of annual precipitation, land cover, and anthropogenic nutrient inputs (Table 1). Nutrient export, nutrient retention, and fertilizer inputs also varied across the study area and were highest in southern and western Minnesota's agricultural watersheds (Fig. 3). Retention of N (average 70%) and P (average 60%) were high at most sites, but variable (Table 1; Fig. 3). Some watersheds retained nearly all NANI and NAPI, while others lost a substantial fraction of net inputs. Watershed N and P retention were lowest in mostly undeveloped watersheds in northern Minnesota with thin soils and high wetland cover, where export was low (Fig. 3). Moderate levels of nutrient retention were observed in southern Minnesota's highly agricultural watersheds, where hydrologic export was high. The highest levels of nutrient retention were observed in central and western Minnesota, where exports tended to be lower than southern Minnesota but still elevated compared to undeveloped watersheds.

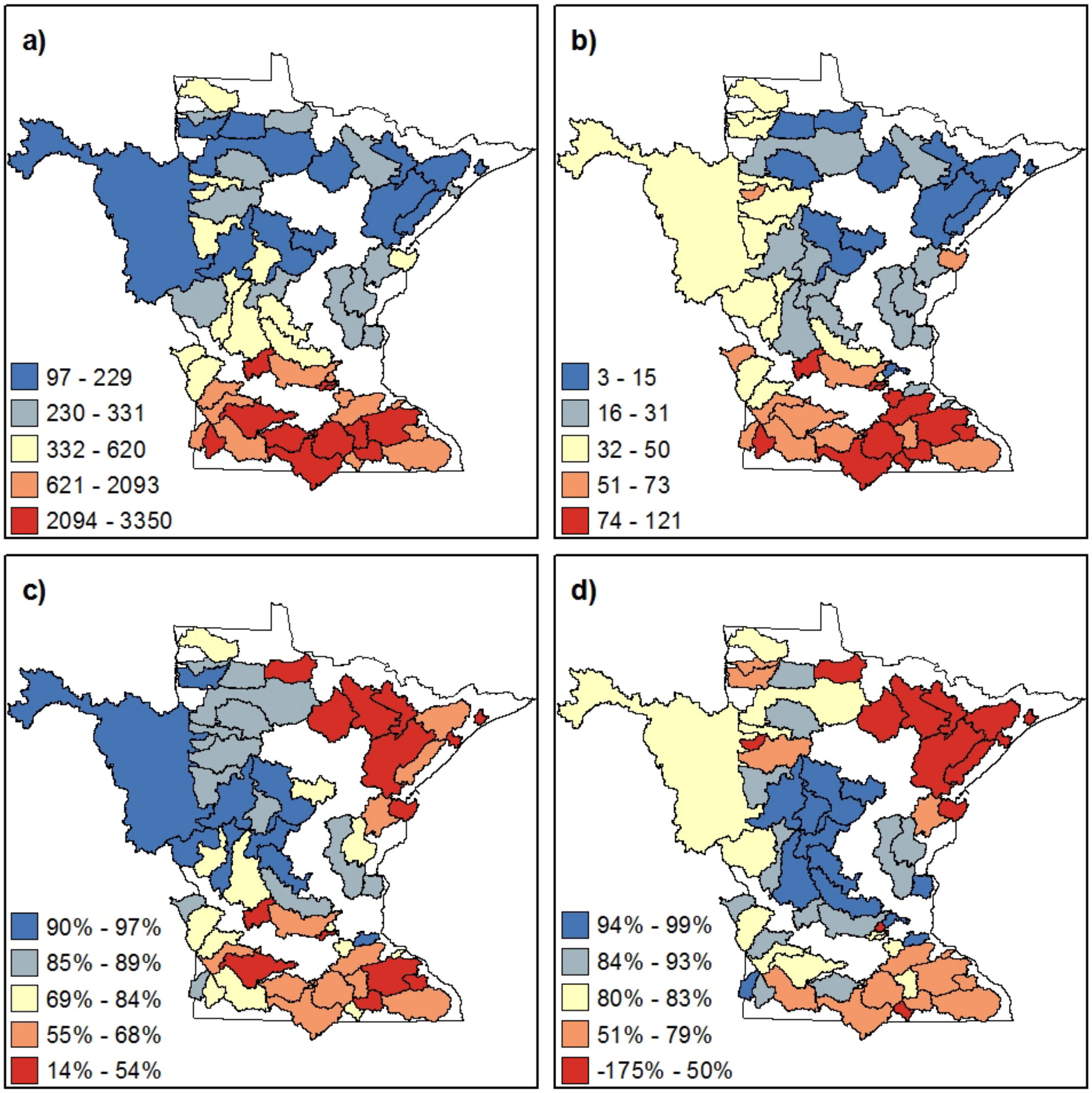


Figure 3. Total nitrogen (a) and total phosphorus (b) export (kg/km^2) and total nitrogen (c) and total phosphorus (d) retention, computed using hydrologic export and estimates of net anthropogenic nutrient inputs.

Table 1. Averages and ranges of values for 62 watersheds used for analyses. All watersheds had an area greater than 150 km^2, and at least two years of water quality data.

Parameter	Units	Average	Minimum	Maximum
Anthropogenic Nutrient Inputs				
Inorganic Fertilizer N	kg/km^2	3309	1	7281
Atmospheric N Deposition	kg/km^2	3295	1	7281
Crop N Fixation	kg/km^2	3279	0	7473
Net Food and Feed N	kg/km^2	-3375	-7700	2264
Inorganic Fertilizer P	kg/km^2	620	0	1350
Net Food and Feed P	kg/km^2	-314	-1002	587
NANI	kg/km^2	3461	279	7990
NAPI	kg/km^2	289	2	1131
Hydrologic Export				
Average Estimated TN Export	kg/km^2	921	97	3350
Average NOx Export	kg/km^2	636	7	2925
Average TKN Export	kg/km^2	268	85	567
N Retention	.	0.71	0.14	0.97
TKN : NOx	.	3.78	0.15	22.77
Average TP Export	kg/km^2	44	3	121
Average DOP Export	kg/km^2	23	0	82
Average Estimated PP Export	kg/km^2	19	2	69
P Retention	.	0.60	-1.75	0.99
DOP : PP	.	1.17	0.23	6.12
Average TSS Export	kg/km^2	17518	243	75140
Watershed Area, Runoff, and Precipitation				
Area	km^2	3831	176	68117
Average Runoff	m	0.21	0.10	0.34
Average Precipitation	m	0.75	0.62	0.94
Runoff : Precipitation		0.27	0.14	0.48
Land Cover Characteristics				
Cropland	%	48	0	91
Pasture or Hay	%	8	0	24
Urban	%	6	1	23
Forested	%	15	0	61
Connected Lakes	%	3	0	17
Connected Wetlands	%	6	0	56

Regression Analyses

Multiple regression analysis highlighted the importance of agricultural inputs and runoff in determining N and P export. In a multiple regression analysis, log-transformed TN export was best explained by inorganic fertilizer inputs and runoff (Table 2). Log-transformed TP export was best explained by inorganic fertilizer inputs, runoff, and the sum of crop and pasture land as a percentage of watershed area (Table 2). In both cases, greater agricultural intensity and runoff both increased N and P export. Agricultural inputs resulted in more N and P with the potential to be washed into lakes and streams. More precipitation reaching streams via runoff was associated with greater mobilization and transport of nutrients.

Due to a combination of nonlinear responses and highly skewed retention distributions, multiple linear regression did not explain as much of the variation in N and P retention (Table 2). For example, while N and P hydrologic export tended to increase with greater anthropogenic inputs, N and P retention appeared to have threshold responses to nutrient inputs (Fig. 4). Log-transformed TN retention was best explained by runoff and urban land as a percentage of watershed area while TP retention was best explained by the sum of crop and pasture land as a percentage of watershed area and runoff. In both regressions, greater runoff was associated with lower retention, likely due to the positive influence of runoff on nutrient export described previously.

Table 2. Results of multiple regression analyses for total N and P export and retention. Candidate variables included NANI and NAPI as well as their components, runoff and precipitation, and watershed land cover characteristics. All forms of nutrient export, the ratio of DOP to PP, NAPI, watershed total and connected lake and wetland cover were log transformed to meet statistical assumptions. NANI and NAPI were not included as predictors of retention as they directly contributed to its calculation; however, components of these variables were included as potential predictors.

Dependent Variable	*n*	R^2	Term	Coefficient	*P*-value	Cumulative AIC*
ln(Total N Export)	61	0.87	Inorganic N Fertilizer	0.0004	<0.001	99.0
			Runoff	5.86	<0.001	56.8
			Intercept	3.94		
ln(Total P Export)	61	0.84	Inorganic P Fertilizer	0.0004	<0.001	100.7
			Crop + Pasture Cover	0.02	0.003	95.5
			Runoff	0.31	<0.001	75.9
			Intercept	1.11		
ln(Total N Retention)	61	0.54	Runoff	-4.39	<0.001	46.8
			Urban	0.07	<0.001	29.0
			Intercept	0.11		
Total P Retention	61	0.43	Crop + Pasture Cover	0.009	<0.001	105.4
			Runoff	-3.33	0.001	98.0
			Intercept	0.78		

* AIC values are reported for models that include the previous terms, indicating improvement in the model by including an additional term.

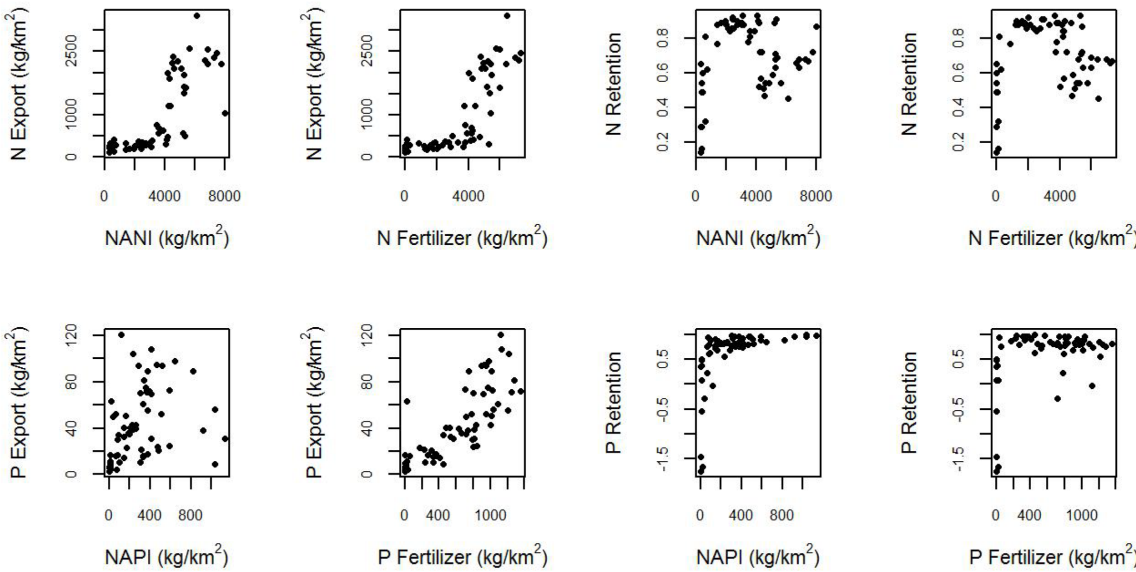


Figure 4. Bivariate plots of N export and retention vs. NANI and N fertilizer inputs and P export and retention vs. NAPI and P fertilizer inputs.

Nitrogen Regression Tree Analyses

Average TN export partitioned into two groups based on NANI (R^2 = 0. 74). Total nitrogen export was lowest at sites where NANI was less than 4150 kg/km^2 (average TN export = 303 kg/km^2; Fig. 5). The lowest NANI watersheds were in northern Minnesota in areas of high forest and wetland cover and minimal inorganic fertilizer inputs (Fig. 5). Some sites with higher anthropogenic activity in western and southern Minnesota also fell into this category.

TN export was higher in sites where NANI was greater than 4150 kg/km^2 (average TN export = 1,860 kg/km^2; Fig. 5). Sites with the highest NANI were highly agricultural watersheds in southern Minnesota. Within this group, fertilizer inputs were much greater than inputs from other sources.

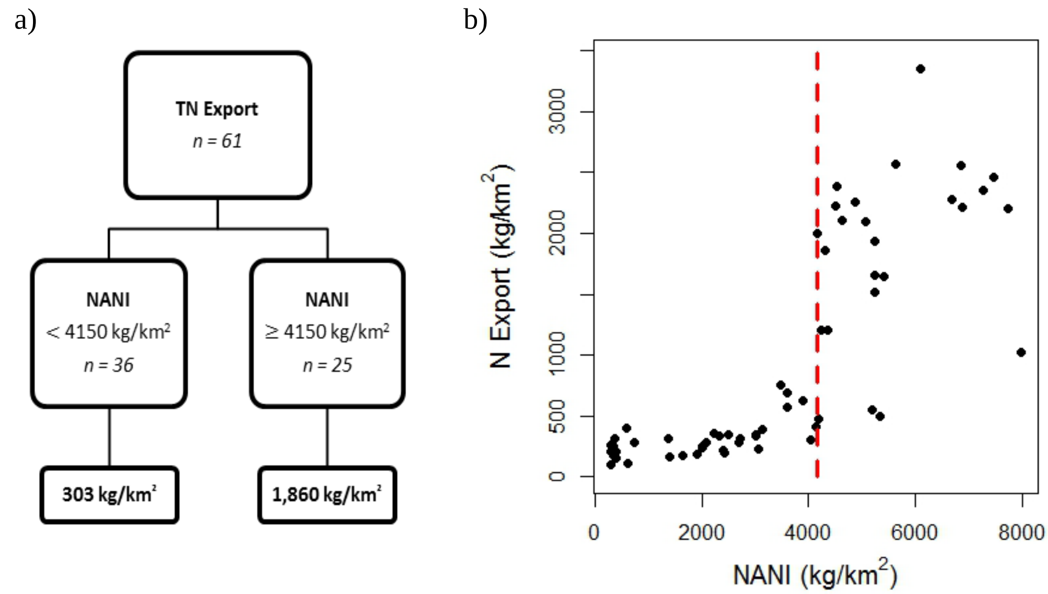


Figure 5. a) Regression tree for TN export, partitioned into two groups: (1) Low NANI and (2) High NANI. b) Bivariate plot of TN export and NANI showing the partition between the two groups. Groups were significantly different ($p < 0.05$).

TN retention partitioned into two groups based on N fertilizer application ($R^2 = 0.46$). Watersheds with low N fertilizer application (i.e. less than 155 kg/km^2) had lower and highly variable levels of N retention (average TN retention = 40% in sites with low fertilizer application compared to 75% in the higher fertilizer application group; Fig. 6). Estimates of retention for northern Minnesota sites were surprisingly low relative to the rest of the state, given their low total N exports (average TN export = 222 kg/km^2 in sites with low fertilizer application compared to 1085 kg/km^2 in the higher fertilizer application group). The dominant form of N inputs where fertilizer inputs were low was atmospheric deposition (Appendix 2).

Of the sites with inorganic N fertilizer inputs greater than 155 kg/km^2, all but three had application rates greater than 1000 kg/km^2. The sites in the higher fertilizer group had higher urban and agricultural land use than those in the lower fertilizer group (average 4% urban and agricultural land in the low fertilizer group and 73% in the higher fertilizer group; Fig. 6). In this group, N retention ranged from 45 to 93%, and variation in N inputs and runoff were important in determining retention (Table 2, Fig. 9). For example, sites with moderate levels of fertilizer application had higher retention than those with the highest levels (Fig. 6b).

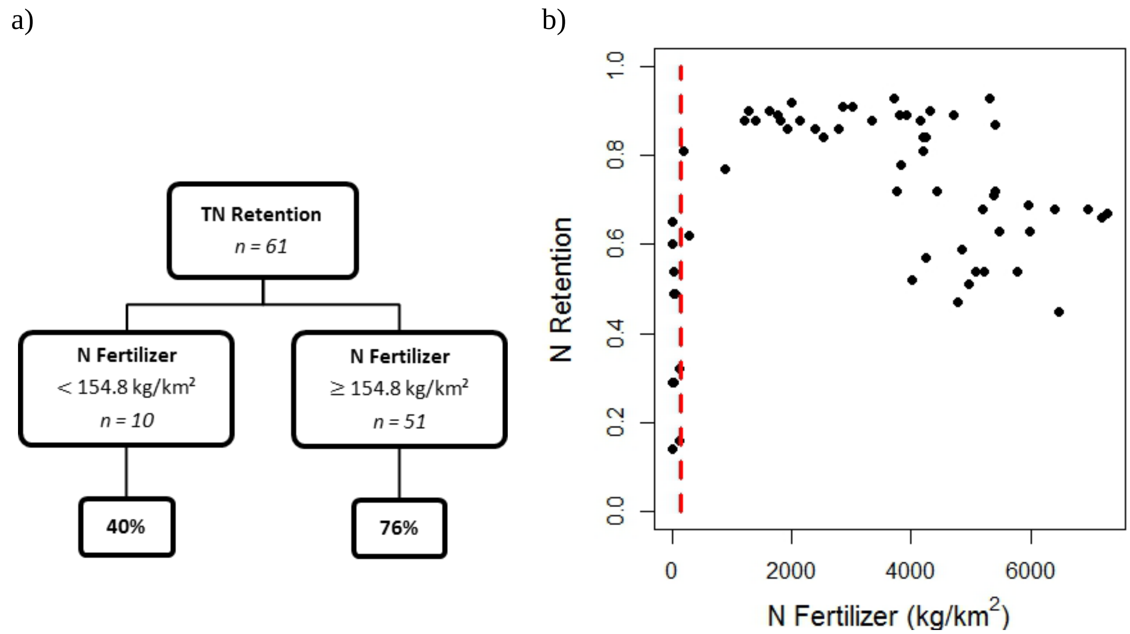


Figure 6. a) Regression tree for N retention, partitioned into two groups: (1) low inorganic N fertilizer and (2) high inorganic N fertilizer. b) Bivariate plot of TN retention and N fertilizer inputs showing the partition between the two groups. Groups are significantly different ($p < 0.05$).

Phosphorus Regression Tree Analyses

Total P export was partitioned into three groups based on inorganic P fertilizer (R^2 = 0.71). Watersheds with P fertilization less than 448 kg/km^2 had lower P export than those with greater fertilizer inputs (average TP export in the lowest fertilizer application group = 14 kg/km^2; Fig. 7). Sites with lower fertilizer inputs were mainly wetland-dominated watersheds in northern Minnesota with low inputs and low export.

Sites with P fertilization in the 448 – 869 kg/km^2 range had higher export, and sites with P fertilization greater than 869 kg/km^2 had the highest export (average TP export =

43 and 78 respectively; Fig. 7). The sites with the highest fertilizer input and export were highly agricultural watersheds in the southern half of Minnesota. In findings consistent with our multiple regression analysis, fertilizer inputs and agricultural intensity were positively related to greater P export.

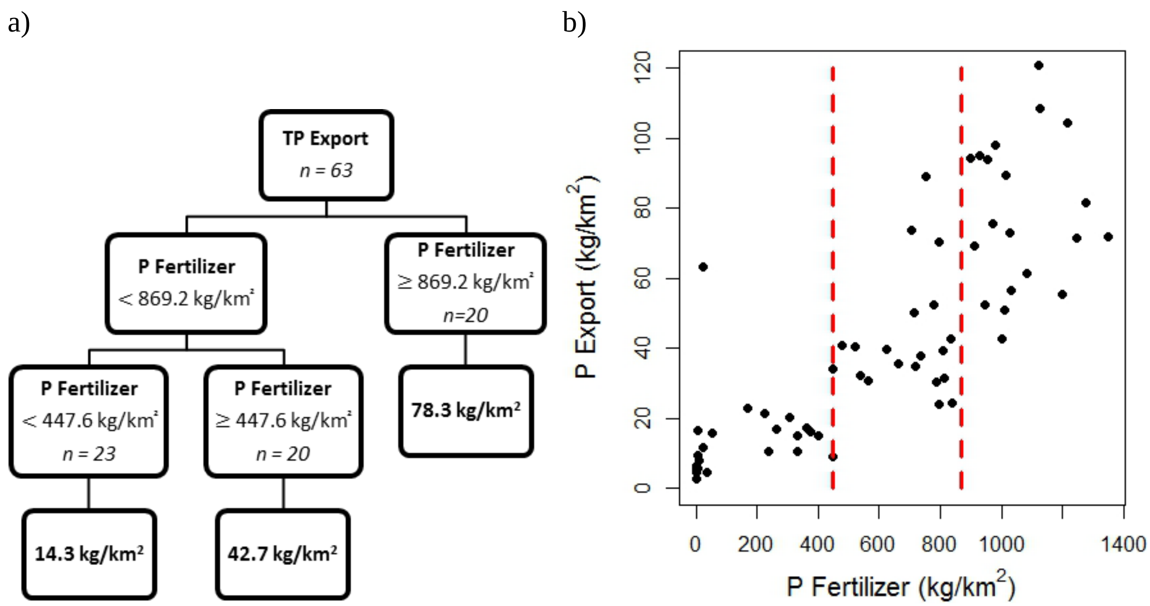


Figure 7. a) Regression tree for TP export, partitioned into three groups: (1) low inorganic P fertilizer; (2) moderate fertilizer; and (3) high fertilizer. b) Bivariate plot of TP export and P fertilizer inputs showing the partitions between the three groups. Groups are significantly different ($p < 0.05$).

Total P retention was partitioned into two groups based on inorganic P fertilizer inputs ($R^2 = 0.51$). P retention was low but highly variable at sites with inorganic P fertilization less than 29 kg/km^2 (average P retention = -0.35; Fig. 8). All of these sites were wetland and forest-dominated sites in northern Minnesota that had low P inputs and exports, which resulted in low calculated P retention (average TP export = 13 kg/km^2 in the group with low fertilizer application and 50 kg/km^2 in the higher application group). As with N retention, losses of P from natural sources that were not accounted for in the NAPI calculations lead to high variability in P retention. At sites with P fertilization greater than 29 kg/km^2, which included major agricultural areas, retention was uniformly high but hydrologic exports were high as well (average TP retention = 0.79; Fig. 8). An average of 21% of NAPI was exported as TP (16% when three outliers are excluded).

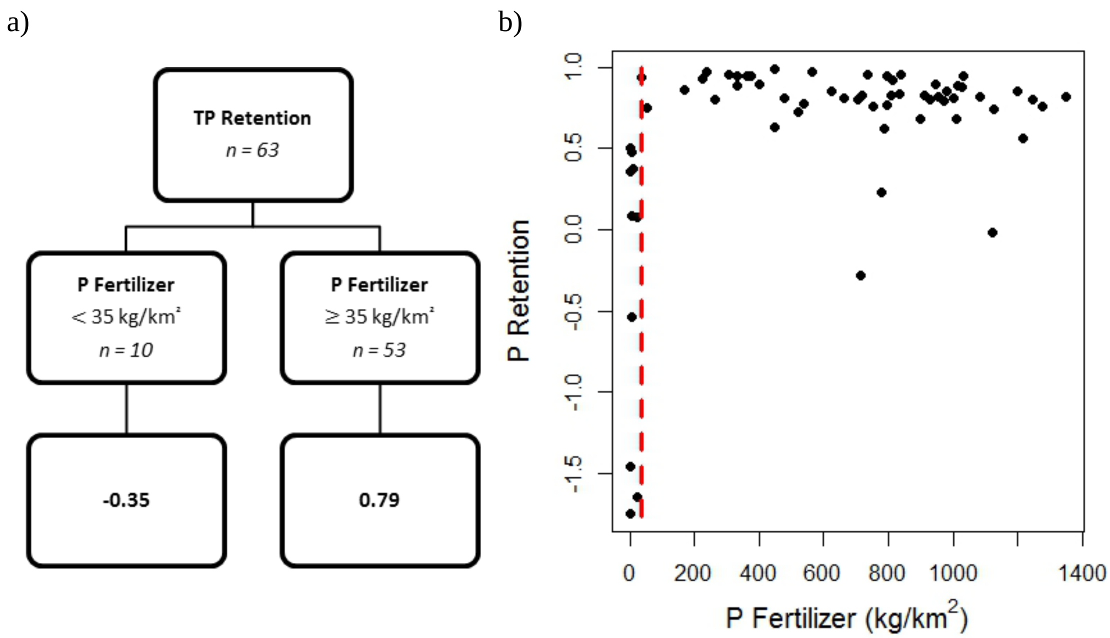


Figure 8. a) Regression tree for P retention, partitioned into two groups: (1) inorganic P fertilizer inputs less than 35 kg/km^2 and (2) inputs greater than 35 kg/km^2. b) Bivariate plot of TP retention and P fertilizer inputs showing the partition between the two groups. Groups were significantly different ($p < 0.05$).

Discussion

Across gradients in human impacts and natural landscape features, our results demonstrate that nutrient transport and retention is driven by a combination of anthropogenic inputs and environmental variables. In highly agricultural and urban watersheds, high losses are driven by high anthropogenic inputs. The relationship between nutrient inputs and retention was nonlinear and modified by watershed runoff. The similarities and contrasts in the factors driving N and P retention in Minnesota's watersheds provide insight into current nutrient management activities and future challenges in the face of climate and land cover change.

Nitrogen Transport and Retention

Total N export was strongly influenced by anthropogenic inputs and runoff. N export was greater from watersheds with higher fertilizer inputs and more N was lost from watersheds with higher runoff. Higher export from sites with higher fertilizer inputs confirms observations from the United States, Europe, and China that intensive agriculture and fertilizer application reduce a watershed's ability to retain N (Howarth et al. 2012; Chen et al. 2014). However, the variability in retention suggests modifiers beyond just inputs to the landscape are important to explaining N losses. Factors such as timing of storms in relation to fertilizer application seasonality of plant growth could affect the amount of N that gets flushed from watersheds. Agricultural practices and crop types have also been shown to influence the amount of excess N in agricultural fields, and thus the amount that might be washed into waterways. Corn-soybean systems relying on fertilizer and fields managed with less crop rotation had more excess nitrogen than those relying on biological nitrogen fixation and those with more complex crop rotation strategies (Blesh and Drinkwater 2013). Landscape features, such as lakes and wetlands, may also retain water and nutrients.

Higher runoff was associated with higher export and lower N retention across all sites, but the effect of runoff differed between sites with high and low anthropogenic influence. Runoff was positively related to export at sites with higher NANI, but not at those with low NANI (Fig. 9). Runoff was associated with a decrease in N retention at both high and low levels of fertilizer inputs, but the intercept was lower and slope steeper than for sites with high levels of fertilizer inputs. These results demonstrate a critical interaction between climate, management and watershed N losses.

The interaction between the timing and amount of precipitation and the hydrological configuration of the watershed determine runoff, but annual nutrient yield and runoff metrics mask much of the variability related to the timing of nutrient export. For instance, lakes have been shown to reduce interannual variability in total N and total P export in wet vs. drought years for agricultural watersheds (Powers et al. 2013). Waterbodies are more likely to be altered or disconnected in developed watersheds (Steele and Heffernan 2013). Fewer waterbodies, coupled with documented higher and faster flows in Minnesota, reduce time for nutrient processing and make hydrologic export more sensitive to flow (Schottler et al. 2014; Danesh-Yazdi et al. 2016). The presence of lakes and wetlands in northern Minnesota may reduce the flow sensitivity of N export by slowing flows in comparison to sites with significant anthropogenic modification of hydrologic pathways, such as storm drains, ditches and tile drainage.

a) b)

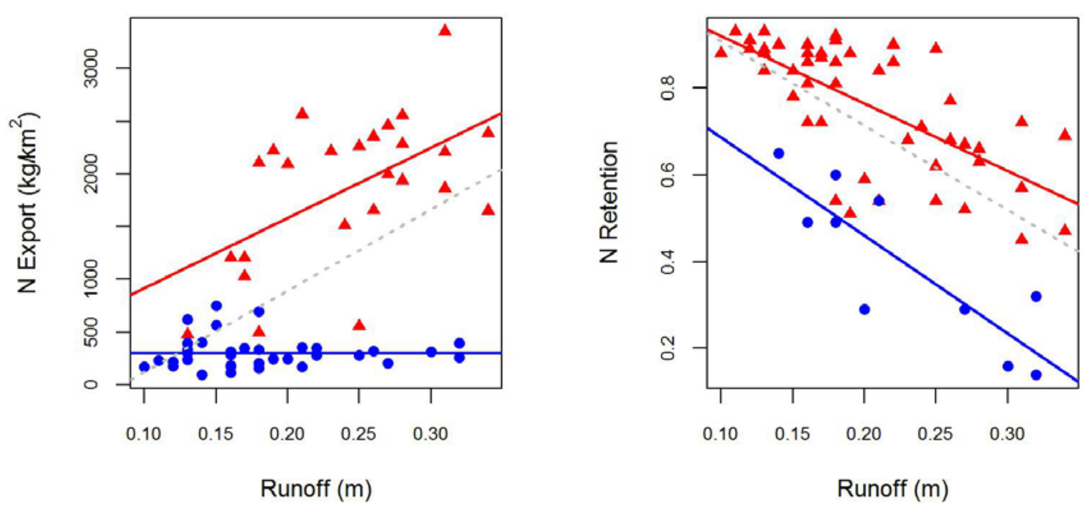


Figure 9. Plots of a) N export vs. runoff in sites with high (triangles) and low (circles) NANI (see Fig. 5 for groups) with regression lines and the regression line for the combined dataset (dashed) and b) N retention vs. runoff in sites with high (triangles) and low (circles) NANI (see Fig. 6 for groups) with regression lines and the regression line for the combined dataset (dashed).

In northern Minnesota, anthropogenic inputs were low, yet retention was also low across a range of annual runoff conditions. One explanation for this pattern is that natural sources of N, such as biological N fixation, are not accounted for in NANI calculations, and lead to lower estimates of retention in sites with the lowest NANI (Sobota et al. 2013). The relatively few measurements of N fixation in similar areas of northern

Minnesota suggest N inputs from fixation from between 0.5 kg N ha^{-1} yr^{-1} in an acidic bog to 1.82 kg N ha^{-1} yr^{-1} in a fen with greater alder coverage (Urban and Eisenreich 1988; Hill et al. 2016). Thus, N fixation cannot completely account for the low N retention in the northern, wetland dominated sites.

Low N retention in northern MN watersheds was likely due to relatively high losses of organic N, which are less tightly controlled by ecosystem processes compared to inorganic N (Neff et al. 2003). Organic N represents a greater fraction of total N fluxes in watersheds with high wetland cover, such as those in northern Minnesota, compared to those lacking substantial wetland coverage (Pellerin et al. 2004). Inorganic N in lakes of northern MN contribute less than 30% of total N in oligotrophic lakes, 14-50% in other lakes, and less than 10% in a reservoir (Urban and Eisenreich 1988; Axler et al. 1994; Johnston et al. 2001)**.** Streams and rivers are similarly dominated by organic N (Sterner et al. 2007). Finally, although we were unable to directly estimate organic N, NO_3 losses were low (unpublished analyses), further supporting dominance of organic over inorganic N forms. Thus, production and transport of organic N in wetlands likely represents a source of N that is less bioavailable for plants and microbes, and likely accounts for the observed low retention of N in northern watersheds.

Phosphorus Transport and Retention

As with TN export and retention, TP export and retention were strongly dependent on agricultural intensity. The sites with low fertilizer inputs and low crop and pasture cover had low export while increasing fertilizer and crop and pasture cover were associated with greater TP export. Greater runoff appears to mobilize more P and increase export, decreasing retention. As with N retention, P retention was higher at sites with greater agricultural intensity and fertilizer inputs.

Hydrologic changes in agricultural areas may increase P export by modifying runoff. Recent increases in precipitation and tile drain installations may promote the efficient transport of water away from fields, reducing time for P sorption and processing in soils and natural channels (Schottler et al. 2014; Foufoula-Georgiou et al. 2015). Tile drainage is sensitive to climate, with greater flows and nutrient export in years with greater precipitation (Christianson and Harmel 2015). A study of P export from tile drained watersheds in Indiana found approximately half of TP losses originated in tile drain discharge, which remains linked to surface runoff via macropore flow (Smith et al. 2014). This suggests agricultural drainage systems are transporting larger P loads through faster flows compared to watersheds with more infiltration through soils and heterogeneous natural channels. These hydrologic changes can further alter nutrient cycles in agricultural watersheds to interactively increase nutrient export.

Phosphorus retention was uniformly high in all watersheds except for those in northern Minnesota with very low anthropogenic activity and four notable outliers in human dominated watersheds: the Cedar, Shell Rock, and Marsh rivers, as well as Carver creek. Most of the land cover (76-95%) in northern Minnesota watersheds was forest, waterbodies, and wetlands. As with N retention, the inputs and hydrologic export of P in these watersheds were consistently low, leading to some error in retention calculations which only consider anthropogenic inputs and exclude weathering inputs. The Cedar and Shell Rock rivers have high permitted discharge contributions of TP from the Austin and Albert Lea wastewater treatment plants, discussed in detail below. Significant urban land use in Carver creek and the Shell Rock and Cedar rivers (9-12%) also suggest human sewage may be contributing to septic exports to these waterways. Additionally, these outlier watersheds have high crop production and therefore export more food and feed than they import, which lowers their NAPI in relation to runoff.

Phosphorus retention in both the agricultural and mixed urban-agricultural watersheds considered in this study was uniformly high, except for the outliers described above. These P retention findings are consistent with previous studies finding high P retention in human-dominated watersheds. In the Chesapeake Bay area, Lake Michigan and Lake Erie watersheds, and Central California, 5-10 % of NAPI was exported via rivers (Russell et al. 2008; Han et al. 2009; Sobota et al. 2009), while 35 % of net inputs occurred in the

Illinois River where wastewater treatment plant effluent was a major source (David and Gentry 2000).

Permitted discharges are not explicitly considered in NAPI calculations as these exports ultimately originate as, and are represented by, food and feed inputs (Chen et al. 2015). However, the proximity of point sources to river outlets likely enhances their influence on watershed P export. Using data obtained from the MPCA, we calculated the average N and P discharge from all permitted facility point source discharges for watersheds completely in Minnesota between 2007 and 2011 and found they were on average a small proportion of annual river export (N = 4%; P = 9.5%). The Shell Rock and Cedar Rivers stood out as outliers with permitted discharges at 55 and 59% of annual P export respectively, which likely contributed to the lower retention compared to other similar agricultural watersheds. As a small proportion of NAPI is exported each year and a small fraction of this is from permitted sources, nonpoint nutrient sources of nutrients are the main driver of nutrient loads in almost all watersheds.

Implications for Management of Nutrient Retention

Our results underscore the importance of reducing nutrient inputs to achieve lower N and P exports. Nutrient management by farms to optimize rates and types of fertilizer and manure applied, treating and managing water released by tile drains, and managing vegetation (e.g. planting cover crops or perennials to reduce soil and nutrient losses, genetic modification of crops) to require less fertilizer while maintaining crop yields could all reduce NANI and NAPI (Minnesota Pollution Control Agency 2013). Farms that relied on biological nitrogen fixation and crop rotations to maintain productivity had lower N surpluses than those which relied on fertilizer N (Blesh and Drinkwater 2013). Fertilizer is the largest N import in agricultural areas and higher NANI is associated with lower N retention in developed watersheds (Howarth et al. 2012). In our watersheds, inorganic fertilizer was almost always the largest NAPI component where there was significant agricultural activity (i.e. crop cover > 20%), although food and feed imports were important in some watersheds with higher urban land use and livestock impacts. Manure P exceeded inorganic P fertilizer for three sites with agricultural land covering over 20% of watershed area. Continued attention to strategies that minimize excess fertilizer application and maximize N and P removal in crops would increase nutrient retention (e.g. Dodd and Sharpley 2016; McIsaac et al. 2016).

Hydrological and biogeochemical processes are often intertwined, but landscape features connected to flowpaths may affect the two differently. For instance, wetland connectivity has been shown to decrease P loads downstream, but can increase P concentrations because of reduced runoff, anoxic release of dissolved P, and particulate P losses (Zhang et al. 2012; Dupas et al. 2015). Increased discharge may flush more N and P out of terrestrial and into aquatic ecosystems. High discharge also decreases residence time in water bodies and contact time with sediments (Han et al. 2009). Thus, it is essential to include not only the wetland cover present in the watershed, but how those wetlands are connected and configured, to understand variability in watershed N and P retention. The interactive effects of hydrology and chemistry could amplify or dampen the effects of either process alone on nutrient retention.

Even wetlands seemingly disconnected from surface waters may still be connected to groundwater. The USGS has predicted groundwater nitrate concentrations using wetland cover and soil organic carbon to model removal via denitrification (Dubrovsky et al. 2010). Watersheds with high inputs of N to the groundwater and little removal via wetlands may have harmful concentrations of nitrate in drinking water. Depending on the degree of connection between surface and groundwater, high nitrate groundwater may also contribute to greater concentrations and export in streams.

Consequences of Land Cover and Climate Change

Lag effects in the relationship between nutrient inputs and river export should also be taken into consideration when evaluating nutrient management or assessing the effect of land cover change. N fluxes from a watershed in China have been shown to be significantly related to the previous 7 years of NANI and are thought to receive inputs from soil that are not considered in NANI calculations (Chen et al. 2014). Work in the same watershed estimated legacy P at 8-58% of annual TP export and suggested the reason for the exponential relationship between NAPI and TP export may be due to P saturation from past inputs (Haygarth et al. 2014; Chen et al. 2015). It may take years or even decades before reductions in nutrients or other strategies to increase retention, such as wetland restoration, are fully effective (Ulén et al. 2015). Conversely, there may be a lag between the timing of increased nutrient inputs or changes to the landscape and worsening water quality.

To better understand the trends in P accumulation over time and the potential effects of historical inputs on current losses, we examined past P fertilizer inputs to the study watersheds. Using USDA agricultural census data, we estimated historic fertilizer inputs using methods modified from the work of Dietz et al. (2015). We used county-level areas of corn, wheat, and soy planted multiplied by fertilization rate data. Fertilizer inputs have fluctuated but remained high in most watersheds included in this study, increasing, stabilizing or decreasing depending on location and land use (Fig. 10). Watersheds near the Twin Cities metropolitan area especially show decreases in fertilizer inputs as land use has transitioned from agricultural to suburban and urban land use (Fig. 10 c & f). Fertilization and P accumulation, particularly in southern and western Minnesota, show few signs of the major declines that would be necessary to begin depleting P stores in the landscape (Fig. 10 a-b & d-e).

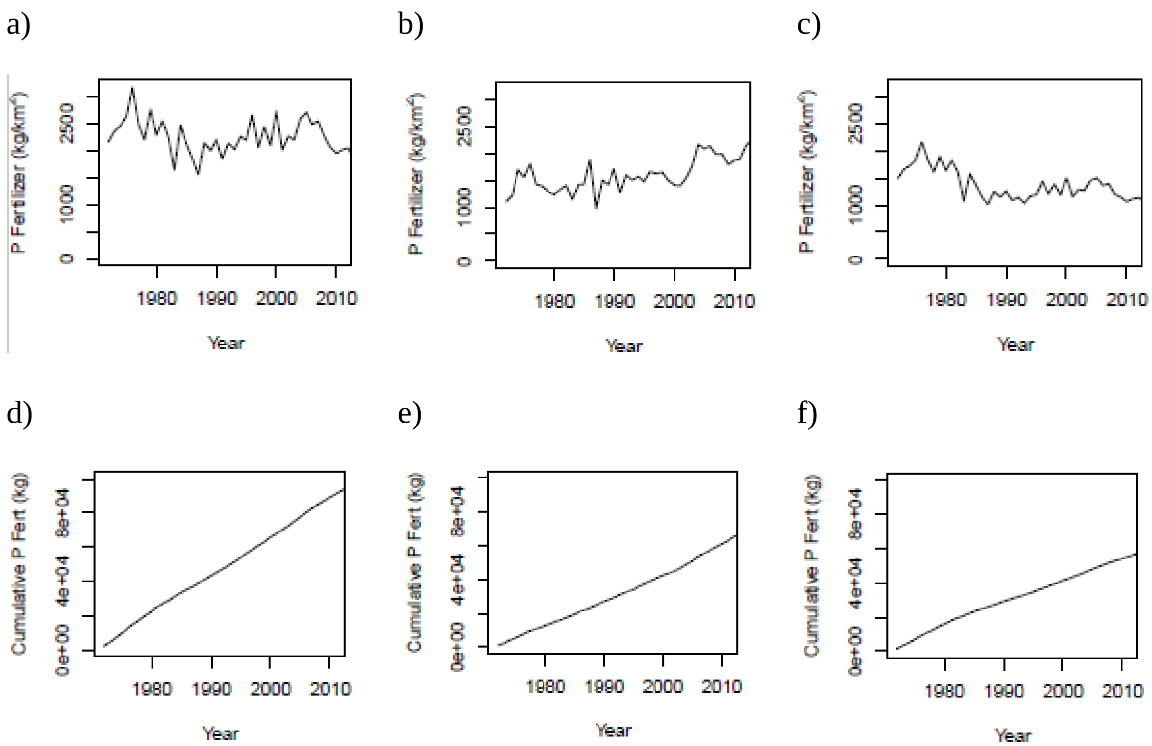


Figure 10. Representative plots showing watersheds with fluctuating fertilizer P inputs that remain relatively high with stable agricultural land cover, increase over time due to agricultural intensification, and decrease over time due to urbanization. Plots show annual estimates of inorganic P fertilizer inputs (a-c) and cumulative P fertilizer inputs (d-f) for the Le Sueur River (a,d), Marsh River (b,e), and Sand Creek (c,f) from 1972 – 2011.

Recent observations have suggested watersheds go through stages of equilibrium between P inputs and outputs, P accumulation where inputs are greater than outputs, and depletion characterized by sustained high outputs after inputs have declined (Haygarth et al. 2014; Powers et al. 2016). Our estimates of past fertilizer inputs suggest in most watersheds we studied, they have fluctuated over time but have not significantly decreased, in contrast to watersheds which have experienced declines in P fertilization and now have greater export than inputs due to this legacy of accumulation (Powers et al. 2016). Following crop price increases due to greater demand for biofuels between 2008 and 2012, Minnesota converted more wetlands to cropland than any state in the US, with the highest rates of conversion in central and western Minnesota (Lark et al. 2015). Because of the sensitivity of nutrient export and retention to agricultural nutrient inputs and high rates of land conversion in the watersheds that retained the most N and P in our analyses, our results indicate more N and P of all forms could be lost from watersheds as wetlands are drained. Conversion of wetlands to cropland would increase the land area receiving N and P fertilizer inputs. In watersheds with significant human presence, nutrient export increases with more intensive agriculture. Wetlands are also likely important to nutrient retention beyond just having low inputs, either for slowing reducing runoff, trapping nutrients, or acting as biogeochemical reactors. Therefore, continued conversion of wetlands to crops is likely to result in water quality deterioration.

Conclusions

Anthropogenic inputs, especially of fertilizer, coupled with hydrologic drivers are important to determining N and P losses from in Minnesota watersheds. By examining inputs and export in the context of landscape and climate variability, we quantified the effects of agricultural intensity on N and P losses and retention across a gradient of land use and precipitation. While retention was high in urban and some agricultural watersheds, losses were also high in the most intensively managed areas, responding strongly to greater fertilizer application and runoff. This suggests there may be an optimal level of nutrient inputs that allow for agricultural productivity while retaining the added nutrients. Climate change and agricultural intensification in watersheds with high nutrient retention threaten to increase nutrient losses throughout the state. Further exploration of the factors affecting nutrient retention can aid in directing management strategies toward a sensible combination of nutrient input reduction, flow retention and infiltration, and promotion of biogeochemical reactions.

Chapter 2

P and sediment dynamics in highly agricultural watersheds

Abstract

Excessive nutrient loading from agricultural watersheds is the dominant contributor to eutrophication in Minnesota and the Gulf of Mexico. While excess phosphorus (P) is known to contribute to this problem, P poses a management challenge because of difficulties tracing its source and curbing hydrologic losses of both its particulate and dissolved forms. We examined the factors controlling annual watershed P losses for 62 sites, and concentration-discharge relationships that describe P and sediment mobilization in streams of primarily agricultural regions in southern and western Minnesota for 119 sites. The exponent, or slope, of the concentration-discharge relationship describes the rate of change in concentration per unit change in discharge, while the coefficient, or vertical offset, of the relationship defines the center of mass of the data determined by the supply of P and water. Particulate and dissolved P both made up a significant proportion of annual export, suggesting the importance of managing both forms of P. Agricultural and urban land cover were significantly related to particulate and dissolved P concentration-discharge coefficients, showing anthropogenic activities elevate mean P concentrations. The presence of bluffs, produced by rapid downcutting of stream channels during the Holocene, was associated with high particulate concentration-discharge exponents showing small near channel areas are highly sensitive to flow conditions and represent major sources of sediment and PP. Lakes and permitted discharges were associated with more variable concentrations across different levels of discharge, resulting in diluting and chemostatic relationships with flow. The unique factors determining dissolved and particulate P export deserve explicit and sometimes different consideration within management strategies to reduce nutrient losses.

Introduction

Decades of study on the lake and watershed conditions that promote algal blooms and other aspects of eutrophication have shed light on the sources of phosphorus (P) that contribute to excessive nutrient loading (e.g. Carpenter and N. F. Caraco, D. L. Correll, R. W. Howarth, A. N. Sharpley 1998; Sharpley et al. 2014; Christianson et al. 2016). P concentrations are highest in areas with agricultural and urban land use, and losses of nutrients from nonpoint sources have not often decreased in response to conservation efforts in the United States (Dubrovsky et al. 2010; García et al. 2016). Despite large investments in improving water quality in lakes and rivers, eutrophication problems persist.

The upper Midwest is a heterogeneous landscape comprising important source areas of P, including agriculture and urban centers, as well as natural features such as forests, lakes and wetlands. Fertilizer and manure are well documented as primary sources of P in agricultural watersheds, but in urban areas a combination of point source wastewater inputs and nonpoint source runoff have historically contributed to high nutrient export (Carpenter et al. 1998; Han et al. 2011; Sharpley et al. 2014; Chen et al. 2015). Investments in improved wastewater treatment and regulations on residential fertilizer use have reduced the point and nonpoint source P contributions to Minnesota's waterways (Barr Engineering Company 2004; Hargan et al. 2011). However, current and historical agricultural P inputs remain a significant contributor to hydrologic P losses.

Minnesota, like much of the upper Midwest, has diverse land use, climate and glacial history, all of which may influence P transport from land to water. The driftless region in the southeast corner of Minnesota was not glaciated during the last ice age (Syverson and Colgan 2011). Watersheds in this area have knickzones at the boundary of glaciated and unglaciated areas with high contributions of sediment from near-channel terraces downstream (Stout et al. 2014). The Red River Basin in northwestern Minnesota is characterized by fine lake bed sediments from glacial Lake Aggasiz which contribute to high silt loads in some of the watershed's low gradient streams, especially those with lower lake and wetland cover (Stoner et al. 1993). The Minnesota River Basin, which was largely glaciated, rapidly downcut when Lake Agassiz drained, and continues to have high rates of erosion from knickzones and bluffs as a result (Engstrom et al. 2009; Belmont et al. 2011).

Sediment has long been emphasized as the primary vehicle for the transport of P and cycling of P in lakes and streams (e.g. Dodd and Sharpley 2016). When P is applied to fields, it can rapidly adsorb to soils and remain there for long periods of time (Sharpley et al. 2014). Since the 1850s, the primary source of sediment to the Mississippi River has shifted from field erosion to bank and bluff erosion (Belmont et al. 2011). Grundtner et al. (2014) found sediment P in the Mississippi River's Lake Pepin originated in the transport of fine particles from stream banks prior to 1850, but afterward only from sediments enriched by historic pollution. Increases in sediment total P in Mississippi River's Lake Pepin since European settlement have been attributed to historic fertilizer application, corn-soy crop cultivation, wastewater treatment discharges, and increases in river flows (Engstrom et al. 2009; Mulla and Sekely 2009). Losses of sediment and P remain high despite investments in best management practices to control soil erosion (Heathcote et al. 2013).

Dissolved P has been shown to contribute significantly to total P losses in some watersheds, but has received less management focus than particulate P (e.g. Gentry et al. 2007; Kröger et al. 2013; King et al. 2014). Phosphorus is usually measured as total phosphorus (TP) and dissolved forms of P, such as soluble reactive phosphorus (SRP) and orthophosphate (OP). Most of the dissolved P pool consists of OP , which is readily available for uptake by organisms (Sharpley and Withers 1994). TP includes less-available forms of P, such as particulate P (PP). Dissolved P losses have been attributed to a number of sources, including fertilizer inputs, sewage point sources, natural weathering, and plant residues (Hansen et al. 2000; Harrison et al. 2005; Jacobson et al. 2006). There is no current consensus on the factors controlling the form and quantity of P exported from agricultural watersheds.

Watershed runoff is important to determining the amount and timing of nutrient losses, and is influenced by changes in climate and land use (Kalkhoff et al. 2016). Precipitation patterns and landscape features interact to determine the timing and quantity of streamflow. Land use in southern Minnesota has shifted from hay and small grain cultivation to corn and soy cultivations during the 20th century, with accompanying increases in watershed ditching and tiling (Foufoula-Georgiou et al. 2015). These changes have been linked to changes in rainfall-runoff relationships such as sharper rising limbs on hydrographs, altered peak flows, and increases in annual water yields (Rahman et al. 2014; Schottler et al. 2014; Foufoula-Georgiou et al. 2015). The combination of agricultural drainage and increased precipitation have resulted in streamflow increases in southern Minnesota's agricultural watersheds (Schottler et al. 2014; Foufoula-Georgiou et al. 2015). How nutrient concentrations will respond to changes in discharge and land use is unknown.

Nutrient and sediment concentration-discharge relationships describe the relationship between the concentration of particulates and solutes across varying levels of discharge. These relationships have been used to infer the source of weathering products and nutrients elevated by anthropogenic activities, as well as the conditions which mobilize them (Godsey et al. 2009; Thompson et al. 2011). These relationships are often described by the power function equation [$C = aQ^b$] where *a* describes the vertical offset of the curve and *b* describes the per-unit increase in concentration as discharge increases (Fig. 11; Godsey et al. 2009). Concentrating relationships ($b > 0$) imply higher flows are mobilizing more of a water-borne constituent, particularly through erosion or greater landscape connectivity. Diluting relationships ($b < 0$) suggest relatively consistent inputs are diluted by greater discharge (Godsey et al. 2009). Chemostatic relationships ($b = 0$) suggest no significant change in concentration across a range of discharge, a pattern observed for mineral weathering products, total nitrogen, and total phosphorus (Godsey et al. 2009; Basu et al. 2010).

This study investigated the factors controlling P and sediment dynamics in predominantly agricultural watersheds, using concentration-discharge relationships to gain insight into the controls on P and sediment mobilization. We examined climate and landscape conditions to explain variation in the coefficients and exponents of the concentration-discharge equations and annual P and sediment export. We hypothesized land cover and nutrient inputs would be related to P concentration-discharge relationships, but expected different factors may be important to determining dissolved vs. particulate P dynamics.

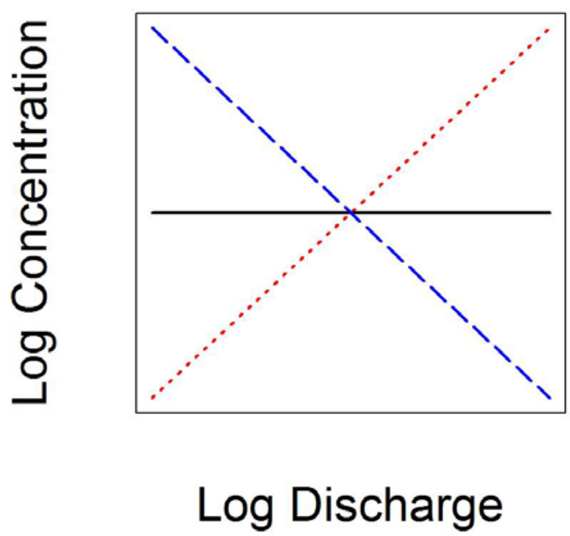


Figure 11. The power functions for concentration-discharge relationships for different constituents may generally follow one of three patterns: concentrating (dotted; exponent > 0), diluting (dashed; exponent < 0), or chemostatic (solid; exponent not significantly different from 0).

Methods

Concentration-Discharge Relationships

We obtained concentration and mean daily discharge data from 119 sites primarily in southern and western Minnesota monitored by the Minnesota Pollution Control Agency (MPCA) as part of their Watershed Pollutant Load Monitoring Network. Water samples and flow data are collected throughout the year at major watershed sites (area greater than

1350 mi^2) and during the period of ice-out through October 31st at subwatershed sites (MPCA 2016). For each parameter, 10-15% of sites in our database had no winter sampling. One TP site and three OP and PP sites were sampled in the spring and summer only. Sampling efforts focus on snowmelt and storm events, resulting in observations distributed across the range of flows observed at each site (average samples per year = 25 for subwatersheds and 35 for major watersheds; MPCA 2016). The dataset includes 172,517 observations collected between 2000 and 2016 (Fig. 12). We selected 116 sites where cultivation of crops, pasture, or hay was the predominant land use and three predominantly wetland and forest sites for comparison. For 108 sites, agricultural land use accounted for over 50% of the watershed, allowing us to focus analyses on identifying sources of P and the factors controlling its export in agricultural areas. Primarily urban sites were not the focus of this monitoring, and thus were not included in our dataset. The agricultural watersheds had significant variation in agricultural intensity, lake and wetland cover, point discharges of wastewater, and geomorphology.

We matched constituent concentrations of total phosphorus (TP), orthophosphate-phosphorus (OP), estimated particulate phosphorus (PP), total suspended solids (TSS), and volatile suspended solids (VSS) with mean daily discharge on the date of sampling and ran regressions on log-transformed variables to obtain the concentration-discharge relationship at all sites with at least 25 matched observations (Table 3). PP was estimated as the difference between TP and OP. For the rating curve equation relating concentration (*C*) and discharge (*Q*):

$$[C = aQ^b] \qquad \textit{Equation 1}$$

the curve's coefficient (*a*) and exponent (*b*) tend to be inversely correlated and not independent from one another (Warrick 2015). Therefore, we used the form of the rating curve equation recommended by Warrick (2015) where the daily discharge (*Q*) is divided by the geometric mean of daily discharge (Q_{GM}) corresponding to each sampled date at a site:

$$[C = \hat{a}(Q/Q_{GM})^b] \qquad \textit{Equation 2}$$

Using this form of the equation the vertical offset of the curve ($\hat{a}$) is equal to the center of mass of the data and is an indicator of nutrient, sediment, and water supply. This allowed us to examine the concentration coincident with the geometric mean of discharge in the context of environmental variables.

We used linear regressions on log-transformed concentrations and log-transformed normalized discharge to characterize the concentration-discharge relationships. These analyses were done in R, by fitting the equation recommended by Warrick (2015):

$$\log(C) = b\log(Q/Q_{GM}) + \log\hat{a} \qquad \textit{Equation 3}$$

We evaluated whether the concentration-discharge relationships exhibited a significant trend using the *p*-value of the exponent (*b*). Sites with a significant positive exponent ($p < 0.05$) were classified as concentrating. Sites with a significant negative exponent ($p < 0.05$) were classified as diluting. Sites where the exponent was not significant ($p > 0.05$) were classified as chemostatic. We also performed linear regressions on log-transformed PP:TSS and VSS:TSS versus log-transformed normalized discharge to examine how the dynamics of sediment and P compare across a range of discharge. VSS describes the proportion of TSS that can be combusted, and thus tends to be organic-rich material. We classified these relationships for all sites into those with ratios that significantly increase with discharge, decrease with discharge, or ratios with no significant response to discharge.

Taking antecedent flow conditions into account is important for predictive modeling of sediment and P concentrations, but for this analysis we aimed to determine the landscape variables most related to overall relationships between discharge, sediment and P. For matched TSS and mean daily flow samples in Minnesota, Vaughan and Belmont (2016) found separating observations by rising and falling limb improved the fit of sediment rating curves. However, similar explanatory variables were important for both the coefficients and exponents of the sediment rating curves for the combined and divided data, with similar model fits (Vaughan and Belmont 2016). We expect the fit of equations relating dissolved and particulate P concentrations with discharge would be improved by taking hysteresis into account, but combining samples taken on the rising and falling limbs of storms provides an overall assessment of the sensitivity of P to hydrologic conditions.

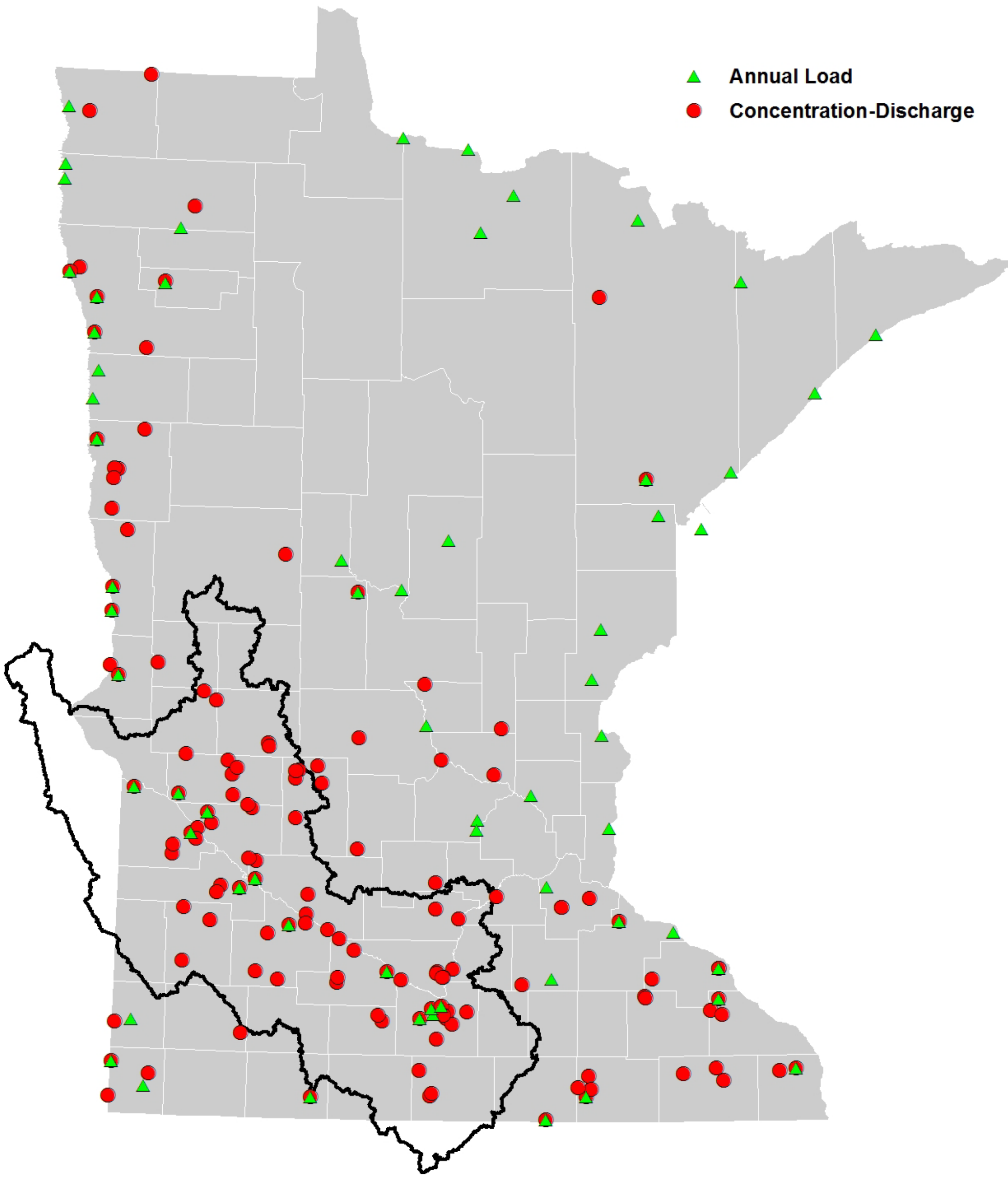


Figure 12. Sites used for concentration-discharge and annual load analyses. Most of the watersheds are primarily agricultural drainages located in the Minnesota River Basin, outlined in black.

Table 3. Number of concentration observations matched with daily flow data for each parameter obtained and the number of sites from our database with more than 25 matched observations.

Parameter	**Total P**	**Ortho-P**	**Estimated PP (TP-OP)**	**TSS**	**VSS**
# of observations matched with discharge data	19011	17076	16606	18498	12518
# of sites with >25 observations	118	114	112	117	105

Explanatory Variables

We delineated watersheds using the locations of monitoring sites provided by the MPCA, the USGS National Hydrography Dataset (NHD) flowlines, and the USGS National Elevation Dataset 30 m digital elevation model. We used the 2011 National Land Cover Dataset (NLCD) to calculate the proportional land cover of each watershed (Homer et al. 2015). We combined all levels of urban cover into one urban class and all levels wetland cover into one wetland class. We used county-level fertilizer input data from the USGS, normalized by the proportion of each county within our watersheds to estimate P and N fertilizer inputs to our watersheds (Gronberg and Spahr 2012). Using 30-year average precipitation downloaded from the PRISM Climate Group (2016), we determined mean annual precipitation for each watershed.

To better understand the role of lake retention and processing in P transport, we calculated the percentage of each watershed that drains through a lake. We selected lakes larger than 4 hectares, placed a pour point at the area of greatest flow accumulation in the lake and delineated the upstream watershed. We compared the area of these watersheds with the watershed area upstream of the sites where concentration and discharge data were collected.

We obtained data from the MPCA for all permitted facility discharges of P between 2007 and 2011. We used the 5-year averaged sum of total P loads from these point sources normalized by watershed area. We calculated fertilizer inputs using USGS county-level inorganic P fertilizer input estimations (Gronberg and Spahr 2012). We weighted fertilizer inputs by the proportion of each county overlapping the watershed, and calculated the sum of total P inputs normalized by watershed area.

We obtained estimates of bluff area for each site (Danesh-Yadzi and Foufoula-Georgiou, unpublished data). River bluffs were mapped using LiDAR elevation data from the Minnesota Geospatial Information Office. Within a moving 12 meter by 12 meter window, areas with elevation differences greater than 4 meters were converted to polygons and clipped to a buffer that extended 3 meters beyond the calculated channel size for each watershed (Danesh-Yazdi and Foufoula-Georgiou, unpublished data; Danesh-Yazdi et al. 2016). We normalized the total bluff area by watershed area.

We did not include annual metrics for flow as continuous flow measurements are not available for all sites. We did not consider variation in background weathering derived sources of P due to the low contributions of P from this source relative to anthropogenic inputs and previous research that has examined the effects of soil type and geology on sediment dynamics (Johnes and Hodgkinson 1998; Vaughan and Belmont 2016).

Statistical Analysis

Variation in concentration-discharge coefficients and exponents was examined using land cover, watershed lake drainage, precipitation, normalized bluff area, normalized inorganic P fertilizer inputs, normalized permitted discharges, and watershed area. Multiple linear regressions were performed in JMP Pro 12 (SAS Institute, NC, USA). Pasture and hay cover, wetland cover, watershed area, and normalized bluff area were log transformed to meet statistical assumptions. Forward stepwise multiple linear regression variable selection was based on AIC. We calculated partial R^2 values by squaring partial correlations obtained in R using the *ppcor* package (Kim 2015).

Annual Nutrient Loads

To complement our analyses of P sources, we examined the dissolved and particulate contribution to annual loads in 62 watersheds monitored by the MPCA and Metropolitan

Council (METC). Flow and water quality data were collected by these organizations throughout the year at watersheds ranging in size from 79 to 68,117 km^2 and calculated by the MPCA using the model FLUX32 (MPCA 2016). We averaged annual loads over the period of 2007-2011 to obtain estimates of the contribution of dissolved loads over a representative period which included high and low flow years. We estimated PP by subtracting orthophosphate-P (OP) from TP. Our analyses showed most (mean = 88%, SE = 9%) of the dissolved phosphorus in highly agricultural watersheds is present as OP (unpublished data). We calculated land cover and fertilizer inputs as described above and mean annual precipitation using annual precipitation data from 2007 – 2011 downloaded from the PRISM Climate Group (2016). Additionally, we estimated connected lake and wetland cover by intersecting USGS National Hydrography Dataset (NHD) flowlines with NHD waterbodies and the National Wetlands Inventory (NWI) or an updated NWI, where it was complete for southern and east-central Minnesota (Minnesota Department of Natural Resources 2015; US Fish and Wildlife Service 2015).

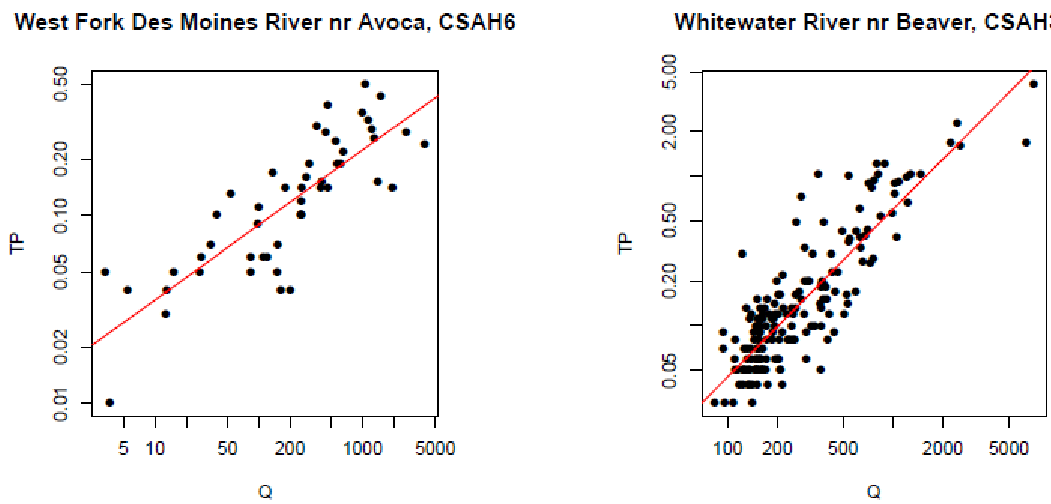
Results

Concentration Discharge Relationships

Sites exhibited a range of concentrating, diluting, and chemostatic concentration-discharge relationships for TP, OP, PP, TSS and VSS (Fig. 13; Appendix 3). One striking finding of this study was the prevalence of concentrating relationships across most of the study sites for all parameters (Table 4).

The spread in these relationships suggest the relationship between P concentration and discharge may be sensitive to variation in environmental factors such as land cover and topography (Fig. 14). Few concentrations increase at a rate greater than or equal to 1:1 with discharge. The median exponents (b) for all concentrations were less than 0.5, suggesting variability in discharge was much greater than the variability in nutrient concentrations, as expected (Table 4).

a)



b)

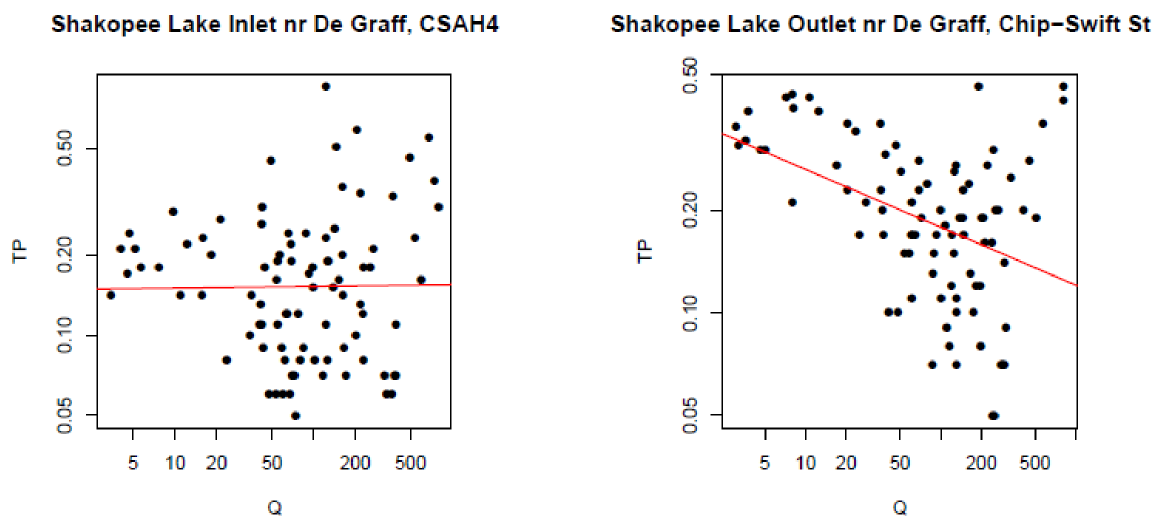


Figure 13. Sample plots showing (a) different concentrating relationships where TP increases with different slopes across a similar range of discharges and (b) a lake inlet-outlet pairing that switches from a chemostatic to a diluting relationship. Untransformed data are plotted in log-log space.

Table 4. Number of concentrating, diluting, and chemostatic power function exponents for concentration discharge relationships with at least 25 matched observations and the median, mean, and range of exponent values (*b*).

Parameter	Concentrating	Diluting	Chemostatic	*b* Median	*b* Mean	*b* range
TP	86 (73%)	11 (9%)	21 (18%)	0.21	0.27	-0.40 – 1.17
OP	76 (67%)	8 (7%)	30 (26%)	0.28	0.29	-0.55 – 1.07
PP	72 (64%)	11 (10%)	29 (26%)	0.16	0.26	-0.30 – 1.36
TSS	93 (79%)	6 (5%)	18 (15%)	0.43	0.46	-0.32 – 1.83
VSS	77 (73%)	24 (23%)	4 (4 %)	0.25	0.30	-0.38 – 1.60

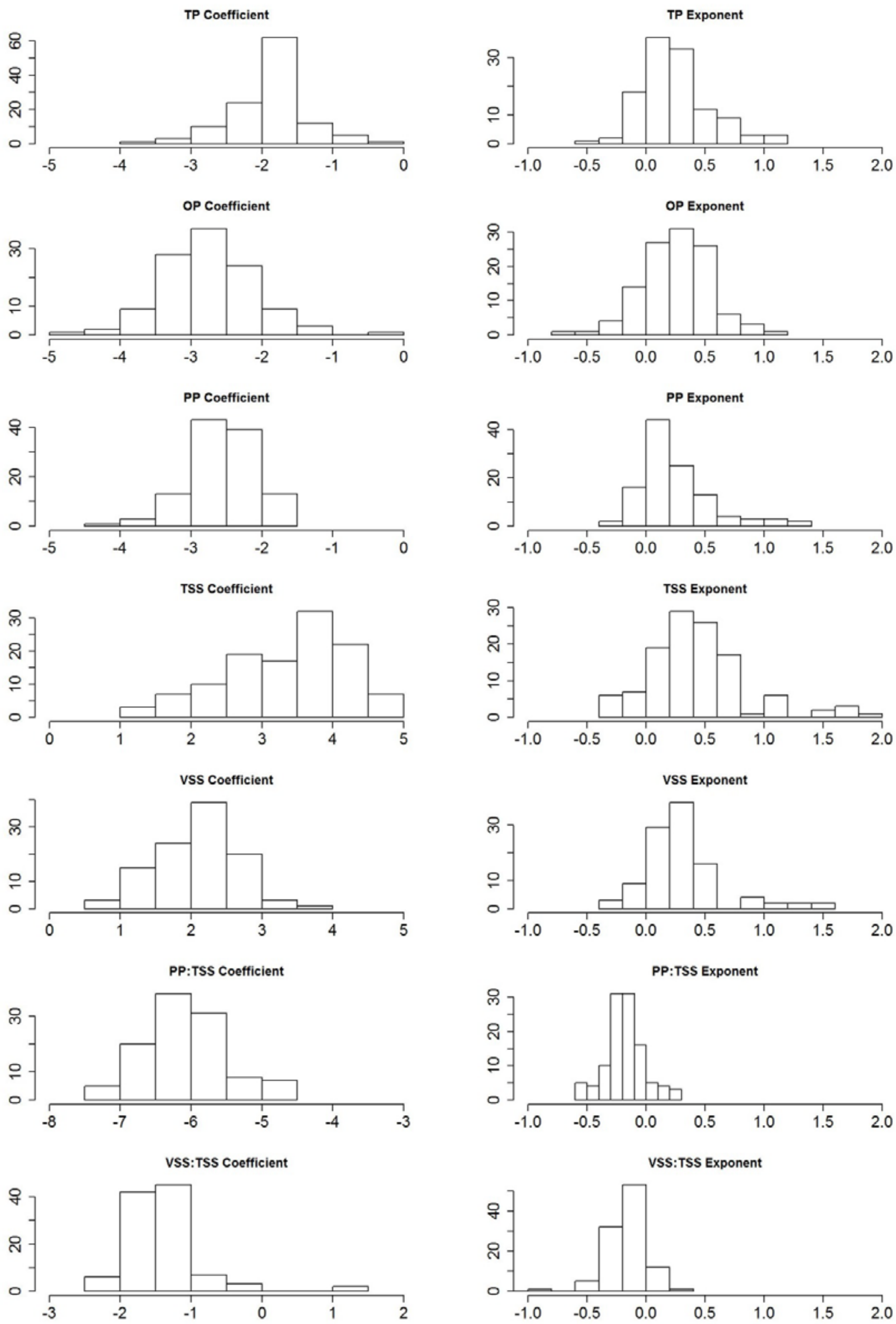


Figure 14. Distributions of power function coefficients and exponents.

Drivers of Concentration-Discharge Relationships

The concentration-discharge coefficient ($\hat{a}$), representing the center of mass of concentration data and determined by the supply both of P and discharge, showed sensitivity to watershed land use metrics associated with anthropogenic activities (Table 5). The coefficient for dissolved P was positively related to crop and urban cover. The PP coefficient was positively related to P fertilizer inputs and pasture and hay land use. TP coefficient was positively correlated with permitted discharge P and P fertilizer inputs. Together these results consistently suggest anthropogenic nutrient inputs to agricultural and urban lands raise the concentration of dissolved P concentrations in receiving waters.

For TSS and VSS concentration-discharge relationships, which were largely characterized by concentrating exponents (b), higher coefficients were associated with greater watershed area. This could indicate greater sediment supply in larger watersheds. Watershed area was also highly correlated with bluff area, so to remove this influence when comparing bluff presence at sites ranging in size from 6 to 4,620 km^2, bluff area was normalized by drainage area for analysis. This suggests larger drainage areas are associated with greater watershed runoff and therefore higher stream power, which may mobilize more sediment and P from near-channel sources.

Flow sensitivity of dissolved P, or represented by the OP concentration-discharge exponent (b), were best explained by the percentage of the watershed that drained through lakes, permitted discharge P, crops, and wetlands. The negative relationship between the exponent and its predictor variables suggest these represent sources consistent across different magnitudes of flow, sources that contribute greater dissolved P at low flows, or that there is greater P retention where lakes and wetlands are present.

The PP exponent was strongly positively related to watershed area normalized bluff area, suggesting areas prone to erosion may contribute more PP as discharge increases. As with dissolved P, lower PP exponents were also associated with more drainage through lakes, crops, and wetlands. The TP exponent was significantly related to the combined predictor variables for PP and dissolved P. Similar to the P results; the TSS and

VSS exponents were also negatively related to lake drainage and crop cover and positively related to bluff area. Bluff area was not correlated with crop cover. However, bluff area was inversely correlated with the proportion of the watershed lake draining through a lake, likely because lakes existed in flatter areas of the landscape than bluffs.

Table 5. Results of forward stepwise multiple regression analyses of concentration-discharge relationship exponents and coefficients. All are significant at $p < 0.05$, with variables significant at $p < 0.001$ shown in bold. Partial R^2 and slope directions (+/-) are shown for predictor variables in the final models.

Predictor	**OP**		**PP**		**TP**		**TSS**		**VSS**	
	a	***b***	***a***	***b***	***a***	***b***	***a***	***b***	***a***	***b***
Mean Annual Precipitation										
% of Watershed Area that Drains through a Lake		0.01 (-)		**0.21 (-)**		**0.10 (-)**		**0.17 (-)**		**0.31 (-)**
Permitted Discharge TP Inputs		**0.09 (-)**			0.07 (+)	0.07 (-)				
P Fertilizer Inputs			**0.27 (+)**		**0.35 (+)**					
Crops	0.10 (+)	**0.04 (-)**		**0.33 (-)**		**0.35 (-)**		**0.23 (-)**		**0.22 (-)**
ln(% Pasture and Hay)			0.06 (+)							
ln(% Wetlands)		**0.11 (-)**				0.05 (-)			0.08 (-)	
ln(% Urban)	**0.13 (+)**									
ln(Watershed Area)							**0.43 (+)**		**0.28 (+)**	
ln(Normalized Bluff Area)				**0.22 (+)**		**0.05 (+)**	0.07 (+)	**0.22 (+)**		**0.16 (+)**
Total R^2	**0.27**	**0.49**	**0.32**	**0.63**	**0.34**	**0.67**	**0.26**	**0.59**	**0.22**	**0.54**

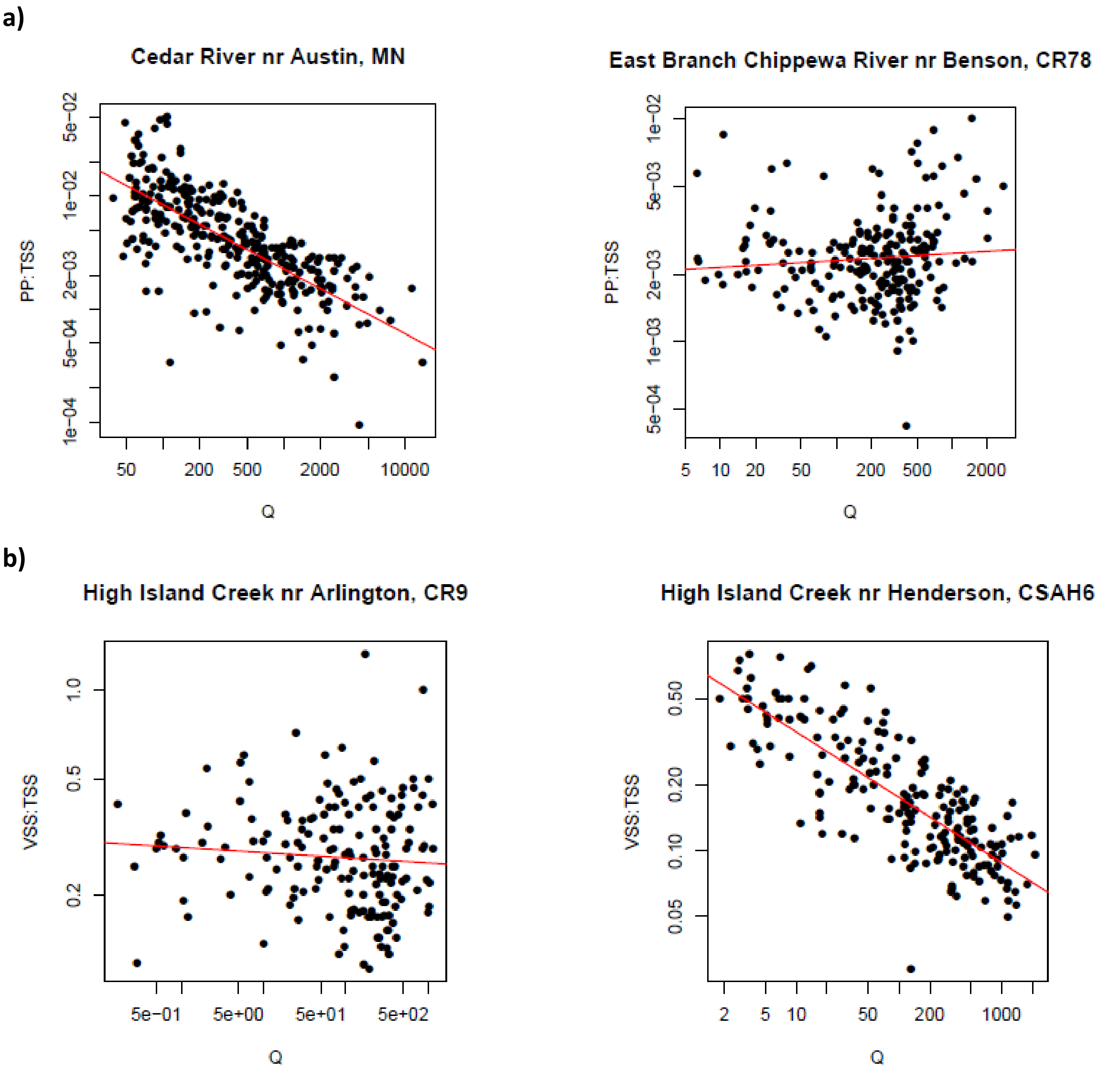


Figure 15. Sample plots showing (a) the ratio of PP to TSS decreasing with increasing discharge and a relationship with no significant difference between PP:TSS across different levels of discharge and (b) an upstream to downstream continuum in which the downstream site has higher bluff area where VSS:TSS switches from no significant response across a range of discharge to decreasing with greater discharge. Untransformed data are plotted in log-log space.

Although PP, VSS, and TSS exhibited concentrating relationships with discharge at most sites, the magnitude of TSS concentrations increased more relative to discharge than

PP and VSS concentrations (Tables 6-7; Fig. 15-16). Lower exponents (*b*) for the power function equations relating these ratios and discharge were associated with greater bluff area. In contrast to TSS and VSS results, lower coefficients ($\hat{a}$) in the power function equation relating the ratios with discharge were associated with greater watershed area. This suggests at sites with larger drainage areas, which have greater bluff area, more TSS is present relative to PP and VSS. The larger sites in the Minnesota River Basin and driftless region tend to be downstream of knickzones, which are areas with a slope change that is uncharacteristic for the watershed, and associated with high near-channel erosion (Belmont et al. 2011; Stout et al. 2014).

The proportion of the watershed that drained through a lake was associated with a greater coefficient ($\hat{a}$) in the power function equation relating the PP:TSS ratio and discharge. Particles rich in P, such as algae, are more common downstream of lakes while particles low in P may settle out in lakes. We did not include a seasonal analysis, but acknowledge the ratio of P and organic matter to TSS is likely to vary seasonally, and affect the chemostatic and concentrating relationships observed.

Table 6. Number of increasing, decreasing, and no significant response power function exponents for relationships of PP:TSS and VSS:TSS with discharge with at least 25 matched observations and the median, mean, and range of exponent values (*b*).

Parameter	Increasing	Decreasing	No Response	*b* Median	*b* Mean	*b* range
PP:TSS	6 (6%)	81 (74%)	22 (20%)	-0.18	-0.18	-0.58 – 0.29
VSS:TSS	3 (3%)	87 (83%)	15 (14%)	-0.16	-0.17	-1.28 – 0.30

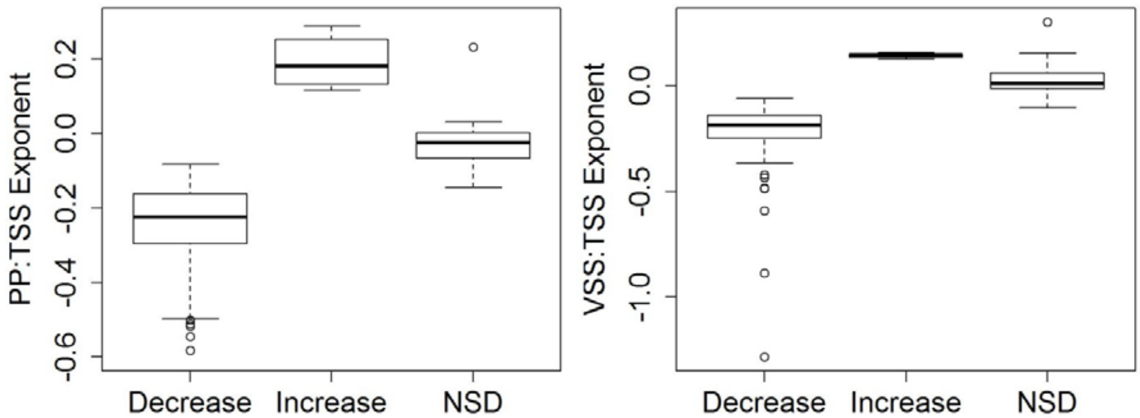


Figure 16. Summary of power function exponents (*b*) for the relationships between log-transformed PP:TSS and VSS:TSS versus log-transformed normalized discharge. Exponents were categorized by whether the ratios significantly decreased, increased, or showed no significant response (NSD) with increasing discharge.

Table 7. Results of forward stepwise multiple regression analyses of PP:TSS and VSS:TSS vs. discharge exponents and coefficients. All are significant at $p < 0.05$, with variables significant at $p < 0.001$ shown in bold. Partial R^2 and slope directions (+/-) are shown for predictor variables in the final models.

Predictor	**PP:TSS**		**VSS:TSS**	
	a	***b***	***a***	***b***
Mean Annual Precipitation				
% of Watershed Area that Drains through a Lake	**0.18 (+)**			
Permitted Discharge TP Inputs				
P Fertilizer Inputs				
Crops				
ln(% Pasture and Hay)				
ln(% Wetlands)				
ln(% Urban)		0.12 (-)		
ln(Watershed Area)	**0.38 (-)**		0.09 (-)	
ln(Normalized Bluff Area)		**0.08 (-)**		**0.27 (-)**
Total R^2	**0.33**	**0.25**	**0.09**	**0.27**

Annual Nutrient Loads

A significant portion of Minnesota's annual total P is exported in the dissolved form (Fig. 17). The range in dissolved P contribution to annual total phosphorus in loads calculated by the MPCA ranged from 7 to nearly 100 % (average 46%) at 62 sites during the study period (2007 to 2011). In southern Minnesota's highly agricultural watersheds, where total P export is the highest in the state, the dissolved phosphorus load ranges from about half of to over 100% of the particulate P load. Lightly developed watersheds in northern MN had the lowest dissolved P and lowest total P loads overall.

Forward stepwise multiple regression showed OP export was best explained by fertilizer inputs, urban land use, and connected lake cover in the watershed (Table 8). PP export was best explained by TSS export (which was highly influenced by bluff and bank erosion), urban land cover, and mean annual runoff. As with the concentration-discharge relationships, human impacts from agricultural and urban land uses are associated with greater OP, while lakes are associated with lower OP. PP increases were closely associated with increases in TSS, but also greater urban land cover and runoff. The association between PP and runoff confirms the flow sensitivity of PP, while both historic and current P inputs may elevate losses from urban areas.

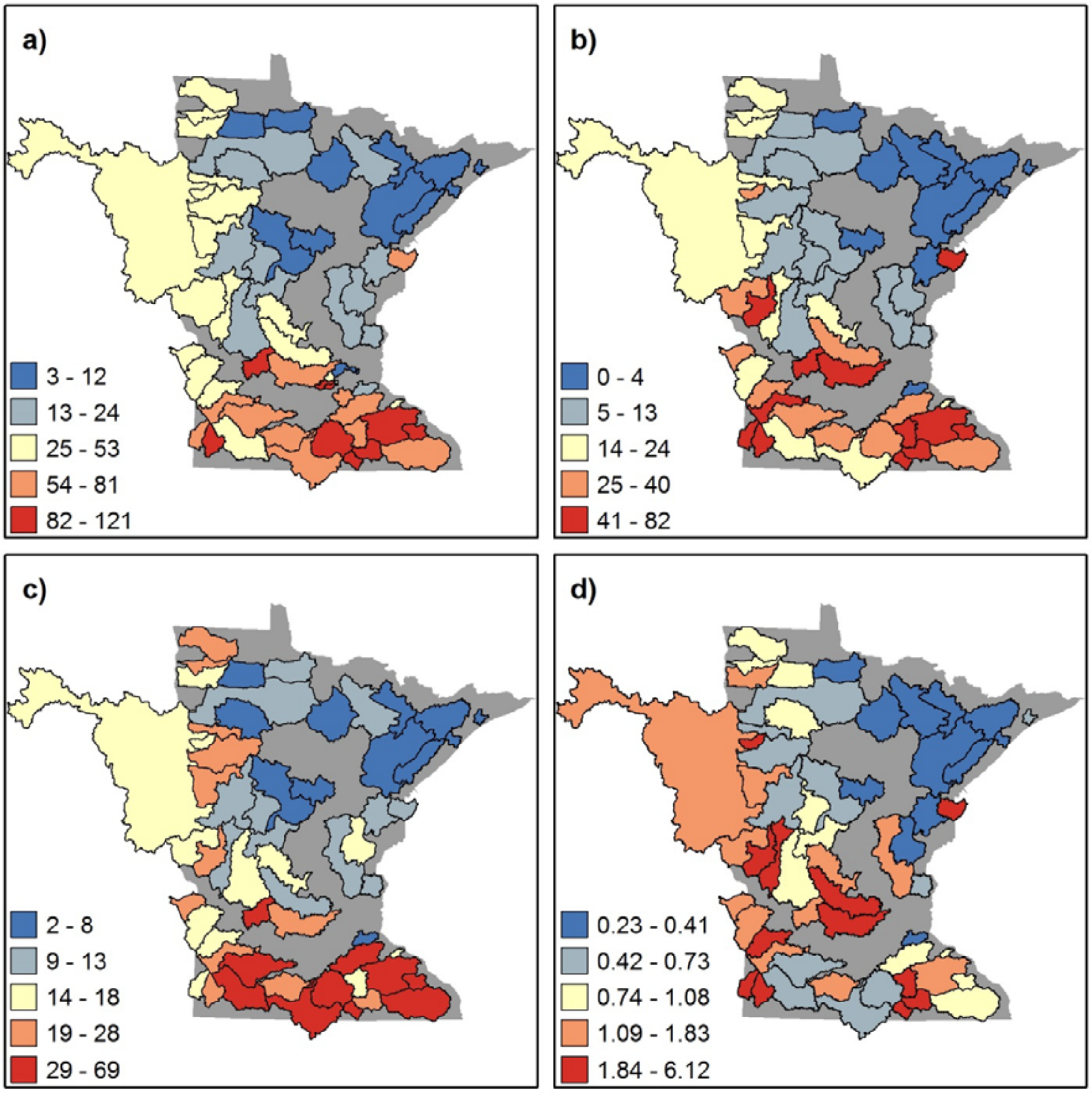


Figure 17. Total phosphorus (a), orthophosphate-phosphorus (b), and estimated particulate phosphorus (c) export (kg/km^2). The ratio of OP:PP (d).

Table 8. Results of forward stepwise multiple regression analyses of OP and estimated PP export. Partial R^2 and slope directions (+/-) are shown for predictor variables in the final models.

Predictor	**OP** (n=56)	**PP** (n=56)
Mean Annual Precipitation		
Mean Annual Runoff		0.16 (+)
P Fertilizer Inputs	0.55 (+)	
ln(TSS Export)		0.73 (+)
ln(Watershed Area)		
% Crops		
ln(% Urban)	0.35 (+)	0.34 (+)
ln(Connected Lakes)	0.22 (-)	
ln(Connected Wetlands)		
Total R^2	**0.88**	**0.87**

Discussion

Prevalence of Concentrating Relationships

The exponents (*b*) for the parameters in this study were much higher than those observed for the concentration-discharge relationships of weathering products. Godsey et al. (2009) found concentrations of weathering products (Ca, Mg, Na, and Si) were relatively stable across a range of several orders of magnitude in discharge, suggesting hydrology drives loading but not concentrations. Similar chemostatic behaviors observed for P were hypothesized to stem from a consistent transport-limited source of P, such as legacy P from anthropogenic inputs to the landscape (Basu et al. 2011). The prevalence of exponents significantly higher than zero, including some near one in driftless-region watersheds, suggests hydrology drives both loading and concentrations in many of the sites we examined. In an unpublished analysis, concentrating relationships tended to remain concentrating across all seasons. Higher discharge may connect more sediment and P source areas, as well as increase stream power and mobilize more near-channel materials (Heathwaite et al. 2005).

While concentrations at most sites increased with discharge, the ratios of PP:TSS and VSS:TSS were negatively related to discharge. This shows for a large majority of sites, increasing flow mobilizes sediments that are relatively depleted in P and organic matter, indicating inputs from deeper soil layers. Increases in discharge have been associated with a shift in sediment from erosion of agricultural fields to bluff erosion, a sediment source that is less rich in P and organic matter (Belmont et al. 2011; Grundtner et al. 2014). Sediment sources lower in P and organic matter, such as bluffs, become more important during higher flows. Solids rich in P and organic matter, derived within channels, riparian zones or in some cases point sources, represent a more consistent input and an important source of particulates during low flow conditions, when there is more time for sorption and assimilation of dissolved P. These can include algae from ditches, lakes, and wetlands as well as sediments with bound P from fertilizer or wastewater treatment facilities.

Factors Controlling P Concentrations

Concentrating relationships for PP, TP (which includes PP), TSS, and VSS with discharge were positively associated with watershed bluff area. Additionally, bluff area was associated with higher TSS and VSS baseline concentrations. The local relief of rivers, encompassing bluffs and other large features that contribute sediment to rivers, has been associated with the exponents and coefficients of concentration-discharge relationships between sediment and TSS in Minnesota (Vaughan and Belmont 2016). The only exponents not associated with bluff area were those of dissolved P, suggesting a different source drives its loss across different levels of discharge. Therefore, a small area of the watershed influenced by hydrology and landscape development is highly influential in determining the sensitivity of particulates to discharge.

Diluting exponents were much less common than concentrating exponents. Lower exponents were associated with higher proportion of watershed area draining through a lake, higher wetland cover, and higher permitted discharge inputs. Nutrient processing in waterbodies can reduce variability in P export across different flow regimes (Powers et al. 2015). Sites with relatively high permitted discharge inputs, such as the Shell Rock River downstream of Albert Lea, MN (63 kg/km^2) and Cedar River downstream of Austin, MN (55 kg/km^2), have elevated baseflow concentrations that reduce P variability across different flows. Sites characterized by dilution of P may require special consideration for load calculations and designing best management practices for nutrient reduction. For instance, reducing point sources of P may assume additional importance in reducing loads at these sites. Additionally, concentration-discharge relationships characterized by lake and wetland nutrient processing indicate potential difficulties identifying the original sources of P and seasonal differences in the flows at which they are mobilized.

In the sites examined, crop cover was also associated with lower exponents across the study sites, suggesting P and sediments were less concentrating with discharge as agricultural intensity increased. This result is somewhat surprising, given crop cover was inversely correlated with lake drainage and wetland area. However, across the study sites, crop cover was consistently high (27-92%; average 72%) and crop cover was positively correlated with urban land cover. Sites with high crop cover also included sites with the highest permitted discharge inputs from urban wastewater treatment facilities, which contributed to lower exponents, while driftless region sites had moderate crop cover and higher exponents.

Fertilizer inputs, crop cover, pasture and hay, permitted discharge inputs, and urban land were associated with higher concentration-discharge relationship coefficients ($\hat{a}$) across the sites examined, suggesting these factors can increase P concentration baselines. The coefficient of TSS rising limb concentration-discharge relationships was positively related to agricultural land use in another study of Minnesota watersheds, suggesting the importance of anthropogenic landscape modifications to both P and sediment dynamics (Vaughan and Belmont 2016). Urban land use is likely associated with greater permitted discharge inputs. Crop cover is associated with more fertilizer inputs. Pasture and hay are associated with lower fertilizer inputs, but may have greater P inputs from manure and vegetation. Higher soil P has been shown to contribute to higher dissolved P in runoff, likely leading to higher coefficients in areas with more agricultural land use (Sharpley et al. 2014).

Although hydrology drives P losses in most of the watersheds examined, considering the original source of the phosphorus is still important. Higher fertilizer inputs and crop cover are associated with greater P coefficients, suggesting nutrient enrichment of Minnesota's soils elevates the baseline for nutrient export, particularly from agricultural watersheds.

Contrasts in Controls on Dissolved and Particulate P

Although management attention has historically been directed toward reduced PP loads, we found dissolved P was a significant portion of annual P export in 62 watersheds across Minnesota. Both PP export and OP export increased with greater urban land cover, but PP export was strongly positively associated with greater TSS and runoff whereas OP export was positively associated with greater P fertilizer inputs and negatively associated with connected lake area. Urban land cover and permitted discharges were important to determining dissolved P concentration-discharge coefficients and exponents respectively, suggesting urban septic and point source discharges may be important sources to dissolved P.

The likelihood different sources contribute to dissolved and particulate P suggests they may respond differently to changes in flow within storm events. Clockwise and counterclockwise hysteresis loops have been observed in TSS, dissolved P and PP concentrations during storm events, suggesting the importance of sediment and P source proximity and ease of mobilization to the sampled stream (Meyer and Likens 1979; House and Warwick 1998; Bowes et al. 2005). Preferential flow from ditched and drained fields, coupled with near-channel P sources, could increase the magnitude of hysteresis loops in agricultural landscapes (Heathwaite and Dils 2000; Bowes et al. 2005). Although near-channel sources of dissolved P are likely significant, evidence for the importance of near-channel factors in determining TSS and PP concentrations suggests particulate mobilization may be especially sensitive to the rising limb of storm hydrographs (Dodd and Sharpley 2016).

Implications for P Management

The importance of flow in determining sediment and P concentrations and export suggests best management practices that reduce flows, such as controlled tile drain releases and constructed wetlands, may effectively reduce losses (Basu et al. 2010; Lemke et al. 2011). These measures may be especially effective at reducing TSS and PP losses in watersheds with naturally erosive landscape features and high concentration-discharge relationship exponents, but how dissolved P might respond is less clear. While we found PP and dissolved P are both important contributors to watershed loads, the factors associated their baseline concentrations and relationships with discharge were different and likely require individual attention when considering management strategies.

In addition to quantity of flow, hydrologic connectivity affects the forms of P that are lost. Lakes and wetlands, which we found were important to reducing concentrations of sediment and all forms of P, are often disconnected or drained as land is converted to agricultural use (Steele and Heffernan 2014; Lark et al. 2015). Furthermore, while tile drains may decrease particulate P losses in surface runoff, a significant proportion of discharge and annual P, particularly dissolved P, export can come from tile drains (Robertson and Saad 2013; King et al. 2014; Smith et al. 2014). This suggests P applied to agricultural fields may be more efficiently transported to streams through this additional landscape connectivity where there are fewer basins to retain nutrients and more ditch and tile drainage.

Best management practices have tended to place focus on reducing particulate P, but this emphasis may not effectively reduce dissolved P. A review by Dodd and Sharpley (2016) found conservation practices, such as vegetated buffers and wetlands, implemented to reduce particulate P may increase organic and inorganic dissolved P. While vegetated buffers have been shown to trap particulate P transported via overland flow, some studies have shown buffers are net exporters of dissolved P due to conditions that promote microbial processing and desorption processes (Roberts et al. 2012; Dupas et al. 2015). Flushing of dissolved reactive P was found to occur when soils were re-wetted after dry periods in the fall and when anoxic conditions released bound P in late winter (Dupas et al. 2015). Additionally, conservation practices to promote soil health in some cases contribute to fertilizer build-up in the soil, which may then be released via desorption and microbial transformations (Dodd and Sharpley 2015). Thus, while buffers may trap some PP from upland erosion of fields, they may increase dissolved P export while doing little to curb bluff erosion.

The flow-sensitivity of sediment and P losses suggests efforts to reduce flows may result in water quality improvements. For instance, lakes and wetland conservation coupled with controls on agricultural drainage would likely decrease nutrient and sediment export by maximizing settling times and in-channel processing (Bowes et al. 2008). Slower flows may also reduce sediment loss from areas prone to erosion. Reductions of urban point and nonpoint sources of P may additionally decrease dissolved P export. Minimizing fertilizer inputs to agricultural fields would likely curb both dissolved and particulate P export. Although the legacy effect of P renders the translation of management actions to reductions in nutrient export a slow process, tailoring these actions to the dominant P sources in a watershed is likely to yield the largest reduction in losses.

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Appendix 1: Watershed N, P, and Environmental Characteristics

Table 9. Watershed N inputs, export and retention.

Watershed	Monitoring Organization	Organization Site Identifier	NANI (kg/km^2)	N Fertilizer Inputs (kg/km^2)	TN (kg/km^2)	NOx (kg/km^2)	TKN (kg/km^2)	N Retention	N:P
Baptism River	MPCA	H01092001	298	1	257	52	205	0.14	39.24
Bevens Creek	METC	BEL	4,511	4,954	2,221	1,775	446	0.51	23.42
Bevens Creek	METC	BEU	4,624	5,079	2,105	1,657	449	0.54	22.41
Big Fork River	MPCA	E77069001	346	49	178	8	170	0.49	22.77
Blue Earth River	MPCA	W30091001	7,479	7,281	2,458	2,146	312	0.67	30.18
Blue Earth River	MPCA	E30092001	7,281	6,945	2,354	2,029	325	0.68	32.89
Bois de Sioux River	MPCA	E54018001	4,027	5,312	302	112	190	0.93	7.08
Buffalo River	MPCA	H58033001	3,000	3,332	348	143	204	0.88	8.75
Cannon River	MPCA	E39004002	5,248	5,194	1,654	1,264	390	0.68	21.91
Carver Creek	METC	CA	3,479	3,819	749	384	365	0.78	14.89
Cedar River	MPCA	E48020001	6,110	6,463	3,350	2,925	426	0.45	32.15
Chippewa River	MPCA	E26057001	3,901	4,238	620	275	237	0.84	25.95
Clearwater River	MPCA	E66050001	2,002	2,132	240	83	157	0.88	16.14
Cloquet River	MPCA	H04048001	397	20	157	14	143	0.60	33.99
Cottonwood River	MPCA	E29001001	5,623	5,771	2,565	1,683	462	0.54	41.86
Crow Wing River	MPCA	E12039001	1,902	1,271	185	55	130	0.90	17.32
Hawk Creek	MPCA	H25037001	4,878	5,228	2,262	1,752	511	0.54	23.12
Kawishiwi River	MPCA	E72061001	279	5	97	12	85	0.65	38.16
Kettle River	MPCA	E35065001	726	289	277	31	246	0.62	17.72
Lac qui Parle River	MPCA	E24023001	3,593	4,195	566	362	204	0.84	14.44
Le Sueur River	MPCA	E32077001	6,854	5,991	2,558	2,133	424	0.63	23.56
Leaf River	MPCA	H13058001	2,484	1,929	348	142	206	0.86	20.07
Little Fork River	MPCA	E76090001	347	33	246	13	232	0.29	14.74
Long Prairie River	MPCA	H14034001	2,691	1,639	279	70	210	0.90	13.68
Marsh River	MPCA	E59008001	3,141	4,144	392	204	188	0.88	7.47
Minnehaha Creek	METC	MH	6,805	2,631	NA	NA	NA	NA	NA
Nemadji River	MPCA	E05011002	576	124	394	21	373	0.32	6.25
North Fork Crow River	MPCA	H18088001	5,201	3,925	554	263	291	0.89	14.59

Otter Tail River	MPCA	H56105001	2,428	2,002	197	56	141	0.92	12.22
Pine River	MPCA	H11051001	614	185	114	12	102	0.81	26.13
Pomme De Terre River	MPCA	E23007001	4,141	4,322	405	212	192	0.90	12.90
Poplar River	MPCA	H01063003	284	1	203	50	153	0.29	37.30
Rapid River	MPCA	H78007001	375	123	314	13	301	0.16	27.02
Red Lake River	MPCA	E63078001	1,384	1,396	167	48	119	0.88	9.89
Red River of the North	MPCA	E61046001	3,060	3,701	229	84	145	0.93	6.42
Redwood River	MPCA	E27035001	5,080	4,857	2,093	1,286	415	0.59	30.17
Rock River	MPCA	H83016001	7,744	5,391	2,207	1,742	465	0.72	24.70
Root River	MPCA	H43007002	4,315	4,243	1,862	1,557	305	0.57	26.53
Rum River	MPCA	H21101001	2,014	1,212	246	72	174	0.88	11.46
Sand Creek	METC	SA	4,353	3,763	1,203	819	384	0.72	16.33
Sand Hill River	MPCA	E61039001	3,019	3,816	333	157	176	0.89	9.55
Sauk River	MPCA	W16058002	5,344	3,009	494	259	236	0.91	16.05
Shell Rock River	MPCA	H49009001	5,402	5,960	1,651	1,084	567	0.69	13.69
Snake River	MPCA	H68011001	2,395	2,861	220	62	158	0.91	6.84
Snake River	MPCA	E36076001	1,369	893	318	33	285	0.77	13.82
South Fork Crow River	MPCA	H19001001	5,245	5,473	1,939	1,486	453	0.63	26.54
Split Rock Creek	MPCA	H82015001	7,990	5,394	1,027	742	285	0.87	18.18
St. Louis River	MPCA	E03174001	396	29	203	22	181	0.49	21.33
Straight River	MPCA	E39101001	6,685	7,188	2,283	1,930	352	0.66	31.84
Sunrise River	MPCA	H37030001	2,711	1,805	314	152	163	0.88	21.05
Tamarac River	MPCA	H69051002	2,335	2,780	331	107	224	0.86	8.17
Thief River	MPCA	E65014001	1,626	1,775	177	20	157	0.89	16.97
Two Rivers	MPCA	H70012001	2,232	2,542	355	88	267	0.84	8.73
Vermillion River	METC	VR	4,986	4,547	NA	NA	NA	NA	NA
Vermillion River	MPCA	E73002001	365	25	168	7	161	0.54	29.07
Watonwan River	MPCA	E31051001	6,883	6,388	2,212	1,856	356	0.68	40.06
Wells Creek	MPCA	H38006002	3,588	4,202	692	575	117	0.81	22.83
West Fork Des Moines River	MPCA	E51107001	5,250	5,370	1,512	1,090	422	0.71	29.80
Whitewater River	MPCA	H40016001	4,159	4,011	1,997	1,570	427	0.52	22.41
Wild Rice River	MPCA	E60112001	2,088	2,388	283	103	180	0.86	8.32
Yellow Bank River	MPCA	E22012001	4,189	4,699	473	254	219	0.89	9.03

Yellow Medicine River	MPCA	E25075001	4,237	4,434	1,204	828	261	0.72	28.35
Zumbro River	MPCA	H41043001	4,535	4,787	2,387	1,974	413	0.47	25.29

Table 10. Watershed P inputs, export and retention.

Watershed	**Monitoring Organization**	**Organization Site Identifier**	**NAPI (kg/km^2)**	**P Fertilizer Inputs (kg/km^2)**	**P Manure Inputs (kg/km^2)**	**TP (kg/km^2)**	**DOP (kg/km^2)**	**P Retention**
Baptism River	MPCA	H01092001	3	0	1	7	2	-1.46
Bevens Creek	METC	BEL	469	929	520	95	NA	0.8
Bevens Creek	METC	BEU	516	953	539	94	NA	0.82
Big Fork River	MPCA	E77069001	13	9	13	8	2	0.38
Blue Earth River	MPCA	W30091001	346	1,275	597	81	24	0.76
Blue Earth River	MPCA	E30092001	364	1,246	589	72	26	0.8
Bois de Sioux River	MPCA	E54018001	227	1,001	187	43	25	0.81
Buffalo River	MPCA	H58033001	257	626	198	40	21	0.85
Cannon River	MPCA	E39004002	362	974	515	75	38	0.79
Carver Creek	METC	CA	39	714	342	50	NA	-0.28
Cedar River	MPCA	E48020001	236	1,214	487	104	79	0.56
Chippewa River	MPCA	E26057001	478	796	495	24	11	0.95
Clearwater River	MPCA	E66050001	148	401	120	15	7	0.9
Cloquet River	MPCA	H04048001	9	4	8	5	1	0.5
Cottonwood River	MPCA	E29001001	335	1,085	594	61	25	0.82
Crow Wing River	MPCA	E12039001	308	239	277	11	4	0.97
Hawk Creek	MPCA	H25037001	643	982	702	98	51	0.85
Kawishiwi River	MPCA	E72061001	4	1	2	3	0	0.36
Kettle River	MPCA	E35065001	62	54	90	16	4	0.75
Lac qui Parle River	MPCA	E24023001	231	808	341	39	21	0.83
Le Sueur River	MPCA	E32077001	411	1,126	603	109	40	0.74
Leaf River	MPCA	H13058001	377	362	329	17	8	0.95
Little Fork River	MPCA	E76090001	11	6	10	17	4	-0.54
Long Prairie River	MPCA	H14034001	481	308	479	20	10	0.96

Marsh River	MPCA	E59008001	69	779	95	53	36	0.23
Minnehaha Creek	METC	MH	1,033	446	114	9	NA	0.99
Nemadji River	MPCA	E05011002	24	22	60	63	54	-1.65
North Fork Crow River	MPCA	H18088001	924	737	823	38	26	0.96
Otter Tail River	MPCA	H56105001	333	376	312	16	6	0.95
Pine River	MPCA	H11051001	74	35	66	4	1	0.94
Pomme De Terre River	MPCA	E23007001	414	812	496	31	21	0.92
Poplar River	MPCA	H01063003	2	0	0	5	2	-1.75
Rapid River	MPCA	H78007001	13	23	14	12	3	0.08
Red Lake River	MPCA	E63078001	82	262	74	17	6	0.8
Red River of the North	MPCA	E61046001	193	664	130	36	20	0.81
Redwood River	MPCA	E27035001	417	913	657	69	45	0.83
Rock River	MPCA	H83016001	824	1,013	816	89	64	0.89
Root River	MPCA	H43007002	301	796	507	70	32	0.77
Rum River	MPCA	H21101001	313	225	182	21	12	0.93
Sand Creek	METC	SA	364	704	367	74	NA	0.8
Sand Hill River	MPCA	E61039001	200	717	87	35	14	0.83
Sauk River	MPCA	W16058002	1,131	565	983	31	18	0.97
Shell Rock River	MPCA	H49009001	118	1,120	390	121	82	-0.02
Snake River	MPCA	H68011001	147	538	84	32	18	0.78
Snake River	MPCA	E36076001	168	168	136	23	7	0.86
South Fork Crow River	MPCA	H19001001	591	1,026	534	73	52	0.88
Split Rock Creek	MPCA	H82015001	1,040	1,031	819	56	41	0.95
St. Louis River	MPCA	E03174001	10	5	11	10	3	0.09
Straight River	MPCA	E39101001	399	1,350	570	72	54	0.82
Sunrise River	MPCA	H37030001	331	334	114	15	6	0.95
Tamarac River	MPCA	H69051002	148	523	84	40	21	0.73
Thief River	MPCA	E65014001	97	334	67	10	5	0.89
Two Rivers	MPCA	H70012001	209	478	81	41	20	0.81
Vermillion River	METC	VR	589	840	314	24	NA	0.96
Vermillion River	MPCA	E73002001	11	4	10	6	1	0.48
Watonwan River	MPCA	E31051001	380	1,201	569	55	30	0.85
Wells Creek	MPCA	H38006002	80	789	497	30	14	0.62

West Fork Des Moines River	MPCA	E51107001	160	1,009	508	51	19	0.68
Whitewater River	MPCA	H40016001	374	752	528	89	44	0.76
Wild Rice River	MPCA	E60112001	91	449	126	34	13	0.63
Yellow Bank River	MPCA	E22012001	508	947	372	52	32	0.9
Yellow Medicine River	MPCA	E25075001	262	833	482	42	29	0.84
Zumbro River	MPCA	H41043001	292	898	533	94	53	0.68

Table 11. Watershed area, precipitation, and selected land cover characteristics.

Watershed	**Monitoring Organization**	**Organization Site Identifier**	**Area (km^2)**	**Precipitation (m)**	**Runoff (m)**	**Crops (%)**	**Urban (%)**	**Wetlands (%)**	**NANI: NAPI**	**Export N:P**
Baptism River	MPCA	H01092001	358	0.77	0.32	0	2	3	112.0	39.2
Bevens Creek	METC	BEL	335	0.79	0.19	66	7	4	9.6	23.4
Bevens Creek	METC	BEU	223	0.79	0.18	64	7	4	9.0	22.4
Big Fork River	MPCA	E77069001	3,833	0.7	0.16	0	2	49	27.4	22.8
Blue Earth River	MPCA	W30091001	3,994	0.84	0.27	86	7	27	21.6	30.2
Blue Earth River	MPCA	E30092001	6,242	0.83	0.26	79	7	15	20.0	32.9
Bois de Sioux River	MPCA	E54018001	4,869	0.69	0.13	77	4	5	17.8	7.1
Buffalo River	MPCA	H58033001	2,869	0.74	0.17	86	4	6	11.7	8.8
Cannon River	MPCA	E39004002	3,471	0.83	0.26	56	8	7	14.5	21.9
Carver Creek	METC	CA	213	0.76	0.15	41	10	4	88.6	14.9
Cedar River	MPCA	E48020001	1,033	0.87	0.31	80	9	5	25.9	32.1
Chippewa River	MPCA	E26057001	4,869	0.7	0.13	67	5	3	8.2	25.9
Clearwater River	MPCA	E66050001	3,574	0.7	0.13	33	4	3	13.5	16.1
Cloquet River	MPCA	H04048001	2,028	0.71	0.18	0	2	3	42.9	34.0
Cottonwood River	MPCA	E29001001	3,367	0.78	0.21	85	6	3	16.8	41.9
Crow Wing River	MPCA	E12039001	9,738	0.7	0.16	10	4	3	6.2	17.3
Hawk Creek	MPCA	H25037001	1,307	0.76	0.25	82	7	40	7.6	23.1
Kawishiwi River	MPCA	E72061001	3,186	0.69	0.14	0	1	27	70.7	38.2
Kettle River	MPCA	E35065001	2,248	0.8	0.25	2	4	3	11.7	17.7
Lac qui Parle River	MPCA	E24023001	2,486	0.73	0.15	62	4	3	15.5	14.4
Le Sueur River	MPCA	E32077001	2,875	0.85	0.28	82	7	2	16.7	23.6

Leaf River	MPCA	H13058001	2,253	0.68	0.22	10	4	3	6.6	20.1
Little Fork River	MPCA	E76090001	4,351	0.7	0.2	0	2	6	32.1	14.7
Long Prairie River	MPCA	H14034001	2,030	0.72	0.22	26	6	5	5.6	13.7
Marsh River	MPCA	E59008001	570	0.66	0.13	85	4	10	45.8	7.5
Minnehaha Creek	METC	MH	439	0.74	0.1	4	47	8	6.6	NA
Nemadji River	MPCA	E05011002	1,088	0.79	0.32	1	3	6	24.1	6.2
North Fork Crow River	MPCA	H18088001	3,496	0.77	0.25	57	6	39	5.6	14.6
Otter Tail River	MPCA	H56105001	4,941	0.73	0.18	27	6	48	7.3	12.2
Pine River	MPCA	H11051001	2,018	0.69	0.16	1	3	15	8.3	26.1
Pomme De Terre River	MPCA	E23007001	2,344	0.69	0.14	63	5	34	10.0	12.9
Poplar River	MPCA	H01063003	295	0.77	0.27	0	2	26	143.3	37.3
Rapid River	MPCA	H78007001	1,813	0.63	0.3	1	1	46	29.6	27.0
Red Lake River	MPCA	E63078001	14,711	0.65	0.1	24	2	45	16.8	9.9
Red River of the North	MPCA	E61046001	68,117	0.65	0.11	66	5	24	15.9	6.4
Redwood River	MPCA	E27035001	1,629	0.77	0.2	77	6	34	12.2	30.2
Rock River	MPCA	H83016001	1,085	0.81	0.31	76	6	40	9.4	24.7
Root River	MPCA	H43007002	4,120	0.94	0.31	43	6	24	14.3	26.5
Rum River	MPCA	H21101001	4,065	0.79	0.19	20	14	22	6.4	11.5
Sand Creek	METC	SA	614	0.77	0.16	52	6	10	12.0	16.3
Sand Hill River	MPCA	E61039001	1,088	0.67	0.13	70	5	5	15.1	9.6
Sauk River	MPCA	W16058002	2,698	0.77	0.18	49	7	2	4.7	16.1
Shell Rock River	MPCA	H49009001	495	0.86	0.34	70	12	14	45.7	13.7
Snake River	MPCA	H68011001	1,974	0.62	0.12	77	5	3	16.3	6.8
Snake River	MPCA	E36076001	2,523	0.83	0.26	8	4	15	8.1	13.8
South Fork Crow River	MPCA	H19001001	3,286	0.77	0.28	72	6	0	8.9	26.5
Split Rock Creek	MPCA	H82015001	803	0.76	0.17	79	6	1	7.7	18.2
St. Louis River	MPCA	E03174001	8,884	0.69	0.18	0	3	1	37.9	21.3
Straight River	MPCA	E39101001	1,127	0.87	0.28	74	10	0	16.8	31.8
Sunrise River	MPCA	H37030001	985	0.73	0.16	25	9	3	8.2	21.1
Tamarac River	MPCA	H69051002	931	0.64	0.18	91	5	7	15.7	8.2
Thief River	MPCA	E65014001	2,551	0.67	0.12	33	2	3	16.7	17.0
Two Rivers	MPCA	H70012001	2,745	0.64	0.21	85	5	8	10.7	8.7
Vermillion River	METC	VR	605	0.81	0.21	51	23	2	8.5	NA

Vermillion River	MPCA	E73002001	2,344	0.7	0.21	0	1	3	32.8	29.1
Watonwan River	MPCA	E31051001	2,204	0.8	0.23	87	6	94	18.1	40.1
Wells Creek	MPCA	H38006002	176	0.85	0.18	41	4	1	44.9	22.8
West Fork Des Moines River	MPCA	E51107001	3,237	0.82	0.24	81	6	1	32.9	29.8
Whitewater River	MPCA	H40016001	702	0.92	0.27	50	6	5	11.1	22.4
Wild Rice River	MPCA	E60112001	4,040	0.72	0.16	52	3	3	23.0	8.3
Yellow Bank River	MPCA	E22012001	1,189	0.72	0.13	53	4	1	8.2	9.0
Yellow Medicine River	MPCA	E25075001	1,725	0.74	0.17	77	5	4	16.2	28.4
Zumbro River	MPCA	H41043001	3,674	0.88	0.34	56	8	3	15.5	25.3

Appendix 2: NANI and TN components for with lowest NANI

Table 12. NANI and TN components for the 10 sites monitored by the MPCA with lowest NANI

Monitoring Site	**NANI (kg/km^2)**	**Fertilizer N (kg/km^2)**	**Atmospheric Deposition (kg/km^2)**	**Agricultural N Fixation (kg/km^2)**	**Net Food & Feed Imports (kg/km^2)**	**TN (kg/km^2)**	**NOx (kg/km^2)**	**TKN (kg/km^2)**	**TKN: NOx**
Kawishiwi River	279.3	5.2	252.1	6.3	15.8	96.9	11.5	85.4	7.4
Poplar River	283.8	0.6	274.0	0.4	8.8	202.5	49.7	152.9	3.1
Baptism River	297.8	1.5	284.3	0.9	11.1	256.6	51.7	204.9	4.0
Big Fork River	346.2	48.9	251.7	60.7	-15.1	178.0	8.1	170.0	21.1
Little Fork River	346.9	33.3	256.0	45.0	12.5	245.6	13.4	232.2	17.3
Vermillion River	364.8	25.3	262.3	35.4	41.8	168.3	7.4	160.9	21.7
Rapid River	374.8	123.0	230.7	149.2	-128.1	314.5	13.2	301.3	22.8
St. Louis River	396.0	28.5	294.6	39.5	33.4	202.8	21.6	181.2	8.4
Cloquet River	397.4	20.1	314.5	27.8	35.0	157.0	13.7	143.3	10.4
Nemadji River	575.5	124.4	317.2	194.1	-60.1	394.1	21.2	372.9	17.6

Appendix 3: Concentration-discharge relationships

Table 13. Concentration-discharge relationship sample size (*n*), coefficient (*â*), exponent (*b*), R^2, p-value, and classification (class) for total phosphorus, orthophosphate phosphorus and particulate phosphorus. The concentration-discharge relationships at sites with at least 25 observations were classified as concentrating (conc.; exponent > 0), diluting (dil.; exponent < 0), or chemostatic (chem.; exponent not significantly different from 0).

Site	TP						OP						PP					
	n	***a***	***b***	***R^2***	***p***	**Class**	***n***	***a***	***b***	***R^2***	***p***	**Class**	***n***	***a***	***b***	***R^2***	***p***	**Class**
Beaver Creek nr Beaver Falls, CSAH2	400	-1.62	0.25	0.27	0.00	Conc.	367	-2.34	0.30	0.24	0.00	Conc.	370	-2.43	0.25	0.26	0.00	Conc.
Beaver Creek nr Valley Springs, 10th Ave	50	-1.48	0.46	0.46	0.00	Conc.	37	-2.98	0.61	0.59	0.00	Conc.	37	-1.70	0.37	0.41	0.00	Conc.
Big Cobb River nr Beauford, CR16	252	-1.65	0.20	0.36	0.00	Conc.	247	-2.83	0.29	0.21	0.00	Conc.	240	-2.17	0.14	0.19	0.00	Conc.
Blue Earth River nr Blue Earth, 105th St	62	-1.66	0.23	0.26	0.00	Conc.	62	-2.43	0.43	0.39	0.00	Conc.	61	-2.45	0.06	0.02	0.27	Chem.
Blue Earth River nr Rapidan, MN	358	-1.75	0.38	0.46	0.00	Conc.	357	-3.12	0.49	0.44	0.00	Conc.	343	-2.12	0.33	0.30	0.00	Conc.
Blue Earth River nr Winnebago, CSAH12	57	-1.52	0.25	0.26	0.00	Conc.	57	-2.76	0.51	0.47	0.00	Conc.	56	-1.94	0.09	0.03	0.17	Chem.
Bluff Creek nr Bluffton, 585th Ave	11	-2.48	-0.01	0.00	0.91		0						0					
Bois de Sioux River nr Doran	204	-1.24	0.13	0.22	0.00	Conc.	203	-1.69	0.15	0.17	0.00	Conc.	189	-2.46	0.10	0.05	0.00	Conc.
Bois de Sioux River nr White Rock, SD	22	-1.32	-0.04	0.02	0.52		22	-1.73	-0.03	0.00	0.77		22	-2.59	-0.05	0.02	0.56	
Buffalo Creek nr Glencoe, CR1	122	-1.16	-0.22	0.30	0.00	Dil.	62	-1.77	-0.25	0.25	0.00	Dil.	61	-2.49	-0.16	0.22	0.00	Dil.
Buffalo River nr Georgetown, CR108	321	-1.67	0.28	0.36	0.00	Conc.	301	-2.27	0.28	0.26	0.00	Conc.	284	-2.53	0.21	0.11	0.00	Conc.
Buffalo River nr Glyndon, CSAH19	34	-1.74	0.73	0.79	0.00	Conc.	34	-2.61	0.54	0.49	0.00	Conc.	34	-2.37	0.93	0.83	0.00	Conc.
Buffalo River nr Glyndon, CSAH68	45	-1.93	0.34	0.37	0.00	Conc.	37	-2.69	0.27	0.12	0.04	Conc.	36	-2.65	0.60	0.60	0.00	Conc.
Cannon River at Morristown, CSAH16	52	-1.52	-0.09	0.07	0.05	Chem.	33	-1.71	-0.25	0.29	0.00	Dil.	38	-2.49	-0.04	0.01	0.50	Chem.
Cannon River at Welch, MN	40	-1.73	0.32	0.25	0.00	Conc.	30	-3.05	0.18	0.05	0.24	Chem.	30	-2.27	0.58	0.41	0.00	Conc.
Cedar River nr Austin	393	-1.14	-0.20	0.17	0.00	Dil.	387	-1.53	-0.26	0.17	0.00	Dil.	380	-2.59	0.05	0.01	0.05	Chem.
Cedar River nr Lansing, CR2	64	-1.93	0.12	0.08	0.02	Conc.	34	-2.30	0.18	0.14	0.03	Conc.	34	-3.07	0.15	0.18	0.01	Conc.
Chetomba Creek nr Maynard, 880th AVE	341	-2.19	0.17	0.17	0.00	Conc.	300	-2.97	0.25	0.24	0.00	Conc.	315	-2.85	0.11	0.09	0.00	Conc.
Chippewa River at Benson, US12	40	-1.89	0.19	0.11	0.03	Conc.	41	-3.40	0.45	0.21	0.00	Conc.	40	-2.23	0.07	0.02	0.45	Chem.
Chippewa River nr Clontarf, CSAH22	291	-1.94	0.07	0.02	0.03	Conc.	270	-3.38	0.40	0.15	0.00	Conc.	275	-2.30	-0.12	0.04	0.00	Dil.

Chippewa River nr Cyrus, 140th St	199	-1.83	0.04	0.01	0.22	Chem.	166	-3.61	0.43	0.22	0.00	Conc.	192	-2.13	-0.13	0.05	0.00	Dil.
Chippewa River nr Milan, MN40	400	-1.90	0.28	0.24	0.00	Conc.	396	-3.14	0.38	0.18	0.00	Conc.	383	-2.30	0.16	0.08	0.00	Conc.
Clear Creek nr Seaforth, CR56	170	-1.86	0.14	0.11	0.00	Conc.	165	-2.49	0.24	0.21	0.00	Conc.	154	-2.87	0.03	0.00	0.45	Chem.
Clearwater River nr Clearwater, CR145	15	-3.70	0.20	0.24	0.06		15	-4.53	-0.01	0.00	0.95		14	-4.20	0.20	0.16	0.15	
Cloquet River nr Burnett, CR694	159	-3.91	-0.01	0.00	0.83	Chem.	145	-4.59	0.00	0.00	0.88	Chem.	114	-4.27	0.03	0.00	0.74	Chem.
Cottonwood River nr Lamberton, US14	214	-1.74	0.18	0.21	0.00	Conc.	209	-2.86	0.29	0.36	0.00	Conc.	209	-2.28	0.10	0.06	0.00	Conc.
Cottonwood River nr Leavenworth, CR8	240	-1.73	0.30	0.41	0.00	Conc.	235	-3.02	0.42	0.47	0.00	Conc.	232	-2.15	0.25	0.27	0.00	Conc.
Cottonwood River nr New Ulm, MN68	501	-1.93	0.50	0.57	0.00	Conc.	474	-3.09	0.49	0.44	0.00	Conc.	442	-2.29	0.46	0.45	0.00	Conc.
Crow Creek nr Morton, Noble Ave	37	-2.05	0.22	0.17	0.01	Conc.	36	-2.41	0.18	0.09	0.07	Chem.	26	-2.97	0.35	0.46	0.00	Conc.
Deerhorn Creek nr Lawndale, 240th Ave	21	-2.45	-0.15	0.01	0.64		0						0					
Dobbins Creek at Austin, CR61	32	-2.65	0.44	0.48	0.00	Conc.	33	-3.12	0.47	0.50	0.00	Conc.	31	-3.55	0.45	0.53	0.00	Conc.
Dry Weather Creek nr Watson, 85th Ave NW	268	-2.11	0.19	0.17	0.00	Conc.	252	-2.88	0.24	0.16	0.00	Conc.	250	-2.77	0.14	0.14	0.00	Conc.
Dry Wood Creek nr Hancock, CSAH7	68	-0.87	0.00	0.00	0.92	Chem.	66	-1.44	0.10	0.10	0.01	Conc.	65	-1.79	-0.11	0.13	0.00	Dil.
East Branch Blue Earth River at Blue Earth, Main St	57	-1.56	0.11	0.08	0.03	Conc.	57	-2.52	0.17	0.06	0.06	Chem.	55	-2.17	0.04	0.01	0.47	Chem.
East Branch Chippewa River nr Benson, CR78	264	-2.18	0.08	0.04	0.00	Conc.	272	-3.20	0.09	0.02	0.02	Conc.	247	-2.67	0.04	0.01	0.14	Chem.
Elk River nr Big Lake	98	-2.40	0.01	0.00	0.80	Chem.	87	-3.94	0.05	0.00	0.55	Chem.	86	-2.66	0.00	0.00	0.99	Chem.
Gilchrist Lake Inlet nr Sedan, TWP170	36	-3.08	0.03	0.00	0.78	Chem.	35	-4.12	-0.09	0.01	0.63	Chem.	32	-3.52	-0.02	0.00	0.91	Chem.
Gilchrist Lake Outlet nr Sedan, CSAH10	36	-3.18	-0.19	0.17	0.01	Dil.	34	-4.46	-0.06	0.02	0.49	Chem.	35	-3.50	-0.25	0.16	0.02	Dil.
Hawk Creek nr Granite Falls, CR52	446	-0.99	0.00	0.00	0.96	Chem.	432	-1.72	-0.03	0.00	0.34	Chem.	407	-1.97	0.13	0.07	0.00	Conc.
Hawk Creek nr Maynard, MN23	393	-0.69	-0.19	0.13	0.00	Dil.	373	-1.34	-0.24	0.10	0.00	Dil.	362	-1.76	-0.01	0.00	0.69	Chem.
Hawk Creek nr Priam, CR116	332	-0.03	-0.59	0.54	0.00	Dil.	315	-0.49	-0.77	0.49	0.00	Dil.	298	-1.66	-0.10	0.03	0.00	Dil.
High Island Creek nr Arlington, CR9	189	-1.68	-0.02	0.00	0.56	Chem.	190	-2.51	0.09	0.02	0.04	Conc.	185	-2.56	-0.09	0.06	0.00	Dil.
High Island Creek nr Henderson, CSAH6	216	-1.58	0.31	0.34	0.00	Conc.	219	-2.77	0.29	0.19	0.00	Conc.	212	-2.11	0.32	0.34	0.00	Conc.
Huse Creek nr Sunburg, 62nd St NW	36	-2.63	-0.06	0.02	0.40	Chem.	33	-3.02	-0.12	0.09	0.10	Chem.	25	-3.57	-0.08	0.02	0.47	

Kandiyohi CD27 nr Sunburg, CSAH1	139	-1.59	-0.11	0.09	0.00	Dil.	125	-2.04	-0.15	0.10	0.00	Dil.	128	-2.80	-0.02	0.00	0.58	Chem.
Kittleson Creek nr Fertile, CSAH1	35	-2.67	0.05	0.04	0.24	Chem.	29	-3.80	-0.04	0.01	0.54	Chem.	29	-3.19	0.10	0.12	0.07	Chem.
Lac qui Parle River nr Lac qui Parle, CSAH31	378	-2.00	0.28	0.39	0.00	Conc.	343	-2.95	0.34	0.31	0.00	Conc.	354	-2.54	0.18	0.17	0.00	Conc.
Lac Qui Parle River nr Providence, CSAH23	201	-1.97	0.18	0.29	0.00	Conc.	187	-3.22	0.19	0.12	0.00	Conc.	190	-2.40	0.16	0.20	0.00	Conc.
Le Sueur River at St. Clair, CSAH28	182	-1.62	0.18	0.28	0.00	Conc.	187	-2.66	0.24	0.17	0.00	Conc.	181	-2.22	0.21	0.29	0.00	Conc.
Le Sueur River nr Rapidan, CR8	250	-1.59	0.44	0.60	0.00	Conc.	256	-2.93	0.50	0.47	0.00	Conc.	248	-1.97	0.43	0.52	0.00	Conc.
Le Sueur River nr Rapidan, MN66	233	-1.48	0.44	0.56	0.00	Conc.	222	-2.61	0.48	0.45	0.00	Conc.	218	-1.89	0.43	0.39	0.00	Conc.
Little Beauford Ditch nr Beauford, MN22	282	-1.87	0.24	0.18	0.00	Conc.	283	-2.40	0.27	0.18	0.00	Conc.	269	-2.81	0.22	0.16	0.00	Conc.
Little Cottonwood River nr Courtland, MN68	23	-2.08	0.41	0.39	0.00		23	-3.28	0.49	0.26	0.01		23	-2.49	0.34	0.39	0.00	
Little Fork River nr Linden Grove, TH73	29	-2.53	0.02	0.03	0.38	Chem.	29	-4.00	-0.01	0.00	0.86	Chem.	29	-2.81	0.03	0.04	0.30	Chem.
Little Rock Creek nr Rice, 15th Ave NW	40	-2.66	0.34	0.61	0.00	Conc.	40	-3.78	0.25	0.31	0.00	Conc.	40	-3.12	0.39	0.53	0.00	Conc.
Long Prairie River nr Philbrook, 313th Ave	191	-2.70	0.20	0.10	0.00	Conc.	170	-3.44	0.01	0.00	0.82	Chem.	163	-3.45	0.31	0.16	0.00	Conc.
M.F.Crow River nr New London, Town Hall Road	158	-3.17	0.08	0.05	0.01	Conc.	23	-3.96	-0.05	0.01	0.66		21	-3.52	0.09	0.07	0.26	
Maple River nr Rapidan, CR35	413	-1.58	0.33	0.56	0.00	Conc.	399	-2.28	0.43	0.45	0.00	Conc.	387	-2.49	0.27	0.24	0.00	Conc.
Maple River nr Sterling Center, CR18	264	-1.55	0.14	0.18	0.00	Conc.	264	-2.12	0.21	0.19	0.00	Conc.	261	-2.68	0.05	0.02	0.04	Conc.
Middle Branch Root River nr Fillmore, CSAH5	52	-1.86	0.75	0.52	0.00	Conc.	52	-2.34	0.54	0.33	0.00	Conc.	44	-2.79	1.04	0.56	0.00	Conc.
Middle Fork Crow River nr Spicer, 275th st	71	-2.89	0.06	0.02	0.21	Chem.	5	-2.76	0.28	0.03	0.80		3	-2.91	0.05	0.00	0.97	
Minnesota River at St. Peter, MN22	93	-1.35	0.33	0.39	0.00	Conc.	93	-2.81	0.56	0.32	0.00	Conc.	92	-1.73	0.23	0.22	0.00	Conc.
Minnesota River nr Jordan	0						0						0					
Minnesota River nr Lac Qui Parle	149	-2.01	0.13	0.07	0.00	Conc.	148	-2.74	0.12	0.02	0.09	Chem.	145	-3.08	0.15	0.08	0.00	Conc.
Mustinka River nr Norcross, MN9	50	-1.31	0.14	0.23	0.00	Conc.	47	-2.29	0.22	0.15	0.01	Conc.	50	-1.92	0.06	0.03	0.22	Chem.
Mustinka River nr Wheaton, CSAH9	102	-1.34	0.22	0.41	0.00	Conc.	102	-1.76	0.26	0.30	0.00	Conc.	96	-2.76	0.17	0.23	0.00	Conc.
Nicollet CD13A nr North Star, MN99	180	-2.15	-0.02	0.00	0.57	Chem.	173	-2.66	-0.03	0.00	0.51	Chem.	154	-3.06	0.04	0.01	0.29	Chem.

Nicollet CD46A nr North Star, CSAH13	177	-1.74	0.01	0.00	0.56	Chem.	167	-2.22	0.02	0.00	0.42	Chem.	161	-2.98	0.05	0.01	0.24	Chem.
North Branch Kandiyohi CD29 nr Sunburg, CSAH1 (W side)	75	-2.26	-0.01	0.00	0.81	Chem.	67	-3.21	-0.15	0.04	0.09	Chem.	67	-2.92	0.12	0.07	0.03	Conc.
North Branch Middle Fork Zumbro River nr Oronoco,5th St	49	-1.74	0.72	0.62	0.00	Conc.	49	-2.58	0.62	0.55	0.00	Conc.	49	-2.40	0.81	0.63	0.00	Conc.
North Eden Creek nr Franklin, CSAH10	41	-2.42	0.36	0.46	0.00	Conc.	38	-2.72	0.20	0.16	0.01	Conc.	27	-3.10	0.46	0.55	0.00	Conc.
North Fork Whitewater River nr Elba, Fairwater St	84	-1.96	0.88	0.65	0.00	Conc.	0						0					
North Fork Zumbro River nr Mazeppa, CSAH7	49	-1.55	0.90	0.71	0.00	Conc.	48	-2.22	0.69	0.53	0.00	Conc.	47	-2.29	1.14	0.79	0.00	Conc.
Otter Tail River at Breckenridge, CSAH16	152	-2.70	0.78	0.40	0.00	Conc.	137	-3.47	0.65	0.19	0.00	Conc.	128	-3.24	0.43	0.14	0.00	Conc.
Pipestone Creek nr Pipestone, CSAH13	67	-2.08	0.29	0.41	0.00	Conc.	67	-3.07	0.51	0.47	0.00	Conc.	66	-2.76	0.05	0.02	0.26	Chem.
Plum Creek nr Walnut Grove, CSAH10	156	-1.93	0.43	0.36	0.00	Conc.	148	-3.04	0.31	0.30	0.00	Conc.	142	-2.51	0.48	0.31	0.00	Conc.
Pomme De Terre River at Appleton	340	-1.84	0.45	0.41	0.00	Conc.	259	-2.74	0.70	0.34	0.00	Conc.	253	-2.62	0.13	0.03	0.01	Conc.
Pomme de Terre River nr Hoffman, CR76	89	-2.24	0.44	0.28	0.00	Conc.	89	-3.73	1.07	0.35	0.00	Conc.	88	-2.67	-0.03	0.00	0.79	Chem.
Red Lake River at Fisher, MN	287	-2.26	0.50	0.47	0.00	Conc.	248	-3.47	0.51	0.35	0.00	Conc.	258	-2.55	0.41	0.41	0.00	Conc.
Red Lake River nr Red Lake Falls, CR13	62	-2.81	0.35	0.47	0.00	Conc.	43	-3.79	0.41	0.33	0.00	Conc.	53	-3.14	0.30	0.42	0.00	Conc.
Red River of the North at Grand Forks, ND	178	-1.28	0.23	0.25	0.00	Conc.	178	-1.94	0.14	0.11	0.00	Conc.	166	-2.09	0.35	0.22	0.00	Conc.
Redwood River at Russell, CR15	172	-2.29	0.28	0.40	0.00	Conc.	171	-3.30	0.23	0.22	0.00	Conc.	165	-2.81	0.30	0.39	0.00	Conc.
Redwood River nr Redwood Falls, MN	430	-0.81	-0.15	0.17	0.00	Dil.	421	-1.41	-0.18	0.09	0.00	Dil.	409	-1.98	0.04	0.01	0.17	Chem.
Rock River nr Hardwick, CR8	37	-2.26	0.00	0.00	0.97	Chem.	37	-3.50	0.07	0.04	0.27	Chem.	37	-2.72	-0.04	0.02	0.42	Chem.
Root River nr Houston, MN	117	-1.67	1.17	0.71	0.00	Conc.	116	-2.57	0.89	0.47	0.00	Conc.	114	-2.27	1.36	0.76	0.00	Conc.
Root River nr Mound Prairie, CSAH25	222	-1.90	1.12	0.72	0.00	Conc.	180	-2.64	0.99	0.64	0.00	Conc.	173	-2.62	1.17	0.68	0.00	Conc.
Root River nr Pilot Mound	328	-1.98	0.87	0.55	0.00	Conc.	283	-3.06	0.73	0.44	0.00	Conc.	271	-2.32	0.81	0.46	0.00	Conc.
Roseau River below State Ditch 51 nr Caribou, MN	8	-2.30	0.10	0.09	0.48		8	-2.99	0.13	0.06	0.57		8	-3.10	0.01	0.00	0.94	
S Br. Wild Rice River at CR27 nr Felton, MN	39	-2.01	0.35	0.42	0.00	Conc.	39	-2.36	0.36	0.32	0.00	Conc.	39	-3.37	0.32	0.48	0.00	Conc.

Sand Hill River at Climax, MN	293	-2.06	0.50	0.56	0.00	Conc.	271	-2.87	0.46	0.38	0.00	Conc.	242	-2.57	0.42	0.32	0.00	Conc.
Sauk River nr St. Martin, CR12	129	-1.94	0.20	0.14	0.00	Conc.	62	-2.68	0.15	0.04	0.14	Chem.	58	-2.71	0.20	0.10	0.01	Conc.
Seven Mile Creek nr North Star, 0.3mi us of US169	6	-2.22	0.42	0.34	0.22		12	-2.47	0.65	0.76	0.00		4	-3.56	0.40	0.67	0.18	
Seven Mile Creek nr North Star, 0.6mi us of US169	22	-2.30	0.61	0.60	0.00		35	-2.55	0.57	0.69	0.00	Conc.	13	-2.43	0.50	0.28	0.07	
Shakopee Creek nr Benson, UNN TWP road (1 mile W MN29)	308	-1.52	-0.12	0.11	0.00	Dil.	287	-3.01	0.00	0.00	0.97	Chem.	295	-1.89	-0.18	0.26	0.00	Dil.
Shakopee Lake Inlet nr De Graff, CSAH4	89	-1.88	0.01	0.00	0.90	Chem.	84	-2.82	-0.10	0.02	0.24	Chem.	83	-2.65	0.12	0.05	0.04	Conc.
Shakopee Lake Outlet nr De Graff, Chip-Swift St NE	88	-1.68	-0.17	0.17	0.00	Dil.	78	-3.46	0.04	0.00	0.66	Chem.	84	-1.99	-0.30	0.41	0.00	Dil.
Shell Rock River nr Gordonsville, CSAH1	166	-0.98	-0.40	0.55	0.00	Dil.	163	-1.68	-0.55	0.30	0.00	Dil.	158	-2.37	-0.14	0.06	0.00	Dil.
Sleepy Eye Creek nr Cobden, CR8	245	-1.84	0.27	0.36	0.00	Conc.	241	-2.73	0.43	0.57	0.00	Conc.	237	-2.63	0.16	0.10	0.00	Conc.
South Branch Buffalo River nr Baker, CR57	24	-2.14	0.10	0.08	0.17		2	-2.59	0.20	1.00			2	-3.57	-1.04	1.00		
South Branch Buffalo River nr Glyndon, CR79 (28th AveS)	88	-1.62	0.13	0.16	0.00	Conc.	79	-1.99	0.18	0.19	0.00	Conc.	79	-2.92	0.01	0.00	0.88	Chem.
South Branch Middle Fork Zumbro River nr Oronoco,5th St	47	-1.46	0.65	0.64	0.00	Conc.	47	-2.54	0.50	0.42	0.00	Conc.	47	-2.03	0.79	0.70	0.00	Conc.
South Branch Root River at Lanesboro, Rochelle Ave N	118	-1.77	0.72	0.57	0.00	Conc.	55	-2.26	0.49	0.34	0.00	Conc.	52	-2.37	0.77	0.48	0.00	Conc.
South Branch Two Rivers River at Hallock, MN175	27	-1.78	0.13	0.11	0.09	Chem.	27	-2.42	-0.02	0.00	0.85	Chem.	27	-2.65	0.34	0.53	0.00	Conc.
South Branch Yellow Medicine River nr Minneota, CSAH26	34	-1.86	0.35	0.57	0.00	Conc.	33	-3.17	0.32	0.42	0.00	Conc.	34	-2.19	0.36	0.59	0.00	Conc.
South Fork Crow River nr Cosmos, MN7	158	-1.69	0.06	0.05	0.01	Conc.	114	-2.75	0.09	0.03	0.07	Chem.	121	-2.21	0.02	0.01	0.42	Chem.
South Fork Watonwan River nr Madelia, CSAH13	54	-1.88	0.16	0.24	0.00	Conc.	46	-3.21	0.23	0.10	0.03	Conc.	46	-2.27	0.24	0.31	0.00	Conc.
South Fork Whitewater River nr Altura, CR112	105	-1.76	0.73	0.61	0.00	Conc.	27	-3.14	0.37	0.11	0.10	Chem.	27	-2.56	0.63	0.33	0.00	Conc.
Split Rock Creek nr Jasper, 201st St	111	-1.77	0.42	0.63	0.00	Conc.	109	-2.24	0.50	0.56	0.00	Conc.	97	-2.93	0.21	0.19	0.00	Conc.

Spring Creek nr Hanley Falls, 480th St	98	-1.61	-0.11	0.06	0.02	Dil.	95	-2.31	-0.03	0.00	0.63	Chem.	92	-2.48	-0.14	0.13	0.00	Dil.
Spring Creek nr Sleepy Eye, CSAH10	42	-2.32	0.61	0.69	0.00	Conc.	39	-2.75	0.40	0.38	0.00	Conc.	28	-2.79	0.68	0.72	0.00	Conc.
Ten Mile Creek nr Lac Qui Parle, CSAH18	52	-2.04	0.31	0.24	0.00	Conc.	49	-2.63	0.31	0.17	0.00	Conc.	49	-3.07	0.40	0.34	0.00	Conc.
Thief River nr Holt, CSAH7	76	-2.46	0.02	0.00	0.60	Chem.	58	-3.55	0.02	0.00	0.78	Chem.	74	-2.85	0.01	0.00	0.73	Chem.
Threemile Creek nr Green Valley, CR67	171	-2.02	0.21	0.25	0.00	Conc.	168	-3.07	0.30	0.27	0.00	Conc.	166	-2.68	0.14	0.10	0.00	Conc.
Turtle Creek at Austin, 43rd St	138	-1.76	0.28	0.34	0.00	Conc.	139	-2.64	0.46	0.34	0.00	Conc.	138	-2.57	0.05	0.01	0.22	Chem.
Vermillion River at Farmington, Denmark Ave	101	-2.71	0.62	0.42	0.00	Conc.	0						0					
Vermillion River nr Vermillion, CSAH85	208	-1.82	0.34	0.05	0.00	Conc.	7	-1.15	-0.67	0.07	0.58		7	-2.22	-0.32	0.01	0.81	
Wabasha Creek nr Franklin, CSAH11	41	-1.50	0.18	0.17	0.01	Conc.	38	-2.06	0.10	0.04	0.23	Chem.	35	-2.66	0.37	0.39	0.00	Conc.
Watonwan River nr Garden City, CSAH13	414	-1.52	0.08	0.04	0.00	Conc.	373	-2.22	0.25	0.18	0.00	Conc.	371	-2.50	0.04	0.01	0.14	Chem.
Watonwan River nr La Salle, CSAH3	125	-1.48	0.05	0.01	0.22	Chem.	113	-2.12	0.02	0.00	0.74	Chem.	113	-2.49	0.07	0.02	0.18	Chem.
West Branch Lac Qui Parle River at Dawson, Diagonal St	214	-2.07	0.08	0.05	0.00	Conc.	201	-2.84	0.13	0.06	0.00	Conc.	204	-2.83	0.05	0.02	0.03	Conc.
West Branch Rum River nr Princeton, CR102	68	-2.41	0.16	0.33	0.00	Conc.	57	-3.22	0.08	0.04	0.13	Chem.	57	-3.05	0.23	0.57	0.00	Conc.
West Fork Beaver Creek nr Bechyn, 320th St	110	-1.52	0.15	0.13	0.00	Conc.	100	-2.22	0.36	0.33	0.00	Conc.	98	-2.41	-0.07	0.03	0.08	Chem.
West Fork Des Moines River at Jackson, River St	173	-1.57	-0.04	0.02	0.11	Chem.	172	-2.79	-0.03	0.00	0.56	Chem.	171	-2.28	0.08	0.05	0.00	Conc.
West Fork Des Moines River nr Avoca, CSAH6	50	-2.16	0.40	0.68	0.00	Conc.	50	-3.53	0.40	0.36	0.00	Conc.	49	-2.54	0.34	0.51	0.00	Conc.
Whitewater River nr Beaver, CSAH30	179	-1.98	1.12	0.76	0.00	Conc.	180	-2.49	0.92	0.61	0.00	Conc.	169	-2.99	1.34	0.72	0.00	Conc.
Yellow Bank River nr Odessa, CSAH40	174	-2.08	0.52	0.53	0.00	Conc.	149	-2.61	0.51	0.40	0.00	Conc.	137	-3.06	0.50	0.55	0.00	Conc.
Yellow Medicine River nr Granite Falls	96	-1.73	0.25	0.20	0.00	Conc.	96	-2.78	0.28	0.20	0.00	Conc.	91	-2.17	0.20	0.14	0.00	Conc.
Yellow Medicine River nr Hanley Falls, CR18	109	-1.89	0.25	0.26	0.00	Conc.	104	-3.05	0.26	0.18	0.00	Conc.	103	-2.33	0.25	0.23	0.00	Conc.

Table 14. Concentration-discharge relationship sample size (*n*), coefficient (*â*), exponent (*b*), R^2, p-value, and classification (class) for total suspended solids and volatile suspended solids. The concentration-discharge relationships at sites with at least 25 observations were classified as concentrating (conc.; exponent > 0), diluting (dil.; exponent < 0), or chemostatic (chem.; exponent not significantly different from 0).

Site	TSS						VSS					
	n	***a***	***b***	R^2	***p***	**Class**	***n***	***a***	***b***	R^2	***p***	**Class**
Beaver Creek nr Beaver Falls, CSAH2	396	3.20	0.60	0.53	0.00	Conc.	155	2.21	0.44	0.47	0.00	Conc.
Beaver Creek nr Valley Springs, 10th Ave	48	4.69	0.64	0.45	0.00	Conc.	48	2.82	0.38	0.23	0.00	Conc.
Big Cobb River nr Beauford, CR16	253	4.20	0.35	0.50	0.00	Conc.	250	2.33	0.19	0.30	0.00	Conc.
Blue Earth River nr Blue Earth, 105th St	60	3.40	0.32	0.25	0.00	Conc.	60	2.14	0.11	0.05	0.08	Chem.
Blue Earth River nr Rapidan, MN	361	4.26	0.63	0.47	0.00	Conc.	356	2.47	0.30	0.24	0.00	Conc.
Blue Earth River nr Winnebago, CSAH12	55	4.35	0.28	0.16	0.00	Conc.	55	2.76	0.12	0.05	0.10	Chem.
Bluff Creek nr Bluffton, 585th Ave	9	1.60	0.14	0.16	0.23		9	1.16	0.20	0.38	0.04	
Bois de Sioux River nr Doran, MN	159	3.75	0.25	0.17	0.00	Conc.	156	2.17	0.14	0.09	0.00	Conc.
Bois de Sioux River nr White Rock, SD	20	2.44	-0.14	0.04	0.36		20	1.14	-0.12	0.04	0.36	
Buffalo Creek nr Glencoe, CR1	171	3.19	0.05	0.02	0.07	Chem.	97	1.95	-0.01	0.00	0.91	Chem.
Buffalo River nr Georgetown, CR108	312	3.93	0.45	0.26	0.00	Conc.	190	2.08	0.34	0.25	0.00	Conc.
Buffalo River nr Glyndon, CSAH19	32	4.71	1.10	0.86	0.00	Conc.	32	2.77	0.84	0.77	0.00	Conc.
Buffalo River nr Glyndon, CSAH68	43	3.67	0.63	0.47	0.00	Conc.	31	1.78	0.52	0.44	0.00	Conc.
Cannon River at Morristown, CSAH16	50	2.00	0.12	0.06	0.09	Chem.	49	1.27	-0.03	0.00	0.65	Chem.
Cannon River at Welch, MN	1	2.66	1.11	0.92	0.18		0					
Cedar River nr Austin, MN	325	2.83	0.64	0.59	0.00	Conc.	301	1.52	0.38	0.38	0.00	Conc.
Cedar River nr Lansing, CR2	65	2.22	0.56	0.47	0.00	Conc.	49	0.90	0.46	0.46	0.00	Conc.
Chetomba Creek nr Maynard, 880th AVE	338	2.86	0.20	0.14	0.00	Conc.	98	1.58	0.22	0.17	0.00	Conc.
Chippewa River at Benson, US12	39	3.91	0.27	0.10	0.05	Conc.	38	2.63	0.10	0.02	0.39	Chem.
Chippewa River nr Clontarf, CSAH22	291	3.71	-0.09	0.01	0.07	Chem.	91	2.28	-0.10	0.02	0.18	Chem.
Chippewa River nr Cyrus, 140th St	215	3.39	-0.31	0.09	0.00	Dil.	50	2.29	-0.27	0.14	0.01	Dil.
Chippewa River nr Milan, MN40	399	3.84	0.31	0.15	0.00	Conc.	214	2.29	0.26	0.11	0.00	Conc.
Clear Creek nr Seaforth, CR56	167	3.45	0.20	0.14	0.00	Conc.	90	1.77	-0.04	0.01	0.30	Chem.
Clearwater River nr Clearwater, CR145	13	1.66	0.53	0.59	0.00		0					
Cloquet River nr Burnett, CR694	160	1.14	0.51	0.27	0.00	Conc.	143	0.60	0.27	0.22	0.00	Conc.
Cottonwood River nr Lamberton, US14	212	4.12	0.27	0.21	0.00	Conc.	111	2.25	0.12	0.11	0.00	Conc.
Cottonwood River nr Leavenworth, CR8	238	4.25	0.46	0.40	0.00	Conc.	154	2.55	0.19	0.18	0.00	Conc.

Cottonwood River nr New Ulm, MN68	442	4.21	0.76	0.58	0.00	Conc.	278	2.63	0.36	0.39	0.00	Conc.
Crow Creek nr Morton, Noble Ave	35	2.47	0.53	0.54	0.00	Conc.	35	1.45	0.27	0.32	0.00	Conc.
Deerhorn Creek nr Lawndale, 240th Ave	21	2.95	0.30	0.18	0.04		0					
Dobbins Creek at Austin, CR61	58	2.45	0.67	0.46	0.00	Conc.	31	0.83	0.57	0.62	0.00	Conc.
Dry Weather Creek nr Watson, 85th Ave NW	281	2.60	0.44	0.44	0.00	Conc.	67	1.56	0.20	0.19	0.00	Conc.
Dry Wood Creek nr Hancock, CSAH7	66	3.86	0.01	0.00	0.88	Chem.	66	2.57	-0.05	0.02	0.28	Chem.
East Branch Blue Earth River at Blue Earth, Main St	55	3.99	0.24	0.13	0.01	Conc.	55	2.56	0.10	0.05	0.11	Chem.
East Branch Chippewa River nr Benson, CR78	260	3.45	0.02	0.00	0.54	Chem.	83	1.92	0.10	0.03	0.10	Chem.
Elk River nr Big Lake, MN	93	2.17	-0.16	0.03	0.07	Chem.	91	3.19	0.11	0.01	0.51	Chem.
Gilchrist Lake Inlet nr Sedan, TWP170	34	1.73	-0.15	0.02	0.47	Chem.	5	1.32	0.32	0.15	0.39	
Gilchrist Lake Outlet nr Sedan, CSAH10	34	1.80	-0.21	0.09	0.08	Chem.	5	1.46	-0.52	0.35	0.16	
Hawk Creek nr Granite Falls, CR52	441	3.64	0.48	0.39	0.00	Conc.	270	2.24	0.37	0.34	0.00	Conc.
Hawk Creek nr Maynard, MN23	391	3.69	0.35	0.27	0.00	Conc.	152	2.40	0.26	0.17	0.00	Conc.
Hawk Creek nr Priam, CR116	328	2.84	0.41	0.29	0.00	Conc.	99	1.77	0.42	0.34	0.00	Conc.
High Island Creek nr Arlington, CR9	189	3.37	-0.12	0.07	0.00	Dil.	172	2.08	-0.14	0.12	0.00	Dil.
High Island Creek nr Henderson, CSAH6	218	4.41	0.67	0.61	0.00	Conc.	195	2.63	0.36	0.37	0.00	Conc.
Huse Creek nr Sunburg, 62nd St NW	34	1.50	-0.21	0.06	0.16	Chem.	2	0.92	0.27	0.50	0.30	
Kandiyohi CD27 nr Sunburg, CSAH1	136	2.20	0.03	0.00	0.51	Chem.	6	0.78	-0.15	0.22	0.24	
Kittleson Creek nr Fertile, CSAH1	33	2.33	0.33	0.39	0.00	Conc.	33	1.12	0.19	0.29	0.00	Conc.
Lac qui Parle River nr Lac qui Parle, CSAH31	361	3.60	0.49	0.55	0.00	Conc.	244	2.02	0.27	0.32	0.00	Conc.
Lac Qui Parle River nr Providence, CSAH23	195	3.89	0.29	0.33	0.00	Conc.	105	2.27	0.21	0.18	0.00	Conc.
Le Sueur River at St. Clair, CSAH28	186	4.27	0.37	0.45	0.00	Conc.	182	2.29	0.24	0.30	0.00	Conc.
Le Sueur River nr Rapidan, CR8	256	4.73	0.65	0.60	0.00	Conc.	253	2.60	0.41	0.47	0.00	Conc.
Le Sueur River nr Rapidan, MN66	193	4.87	0.65	0.49	0.00	Conc.	128	2.65	0.51	0.44	0.00	Conc.
Little Beauford Ditch nr Beauford, MN22	267	3.09	0.34	0.24	0.00	Conc.	178	1.36	0.20	0.11	0.00	Conc.
Little Cottonwood River nr Courtland, MN68	6	4.17	0.72	0.78	0.00		0					
Little Fork River nr Linden Grove, TH73	27	2.98	0.13	0.37	0.00	Conc.	27	1.48	0.05	0.12	0.07	Chem.
Little Rock Creek nr Rice, 15th Ave NW	NA						0					
Long Prairie River nr Philbrook, 313th Ave	200	1.34	0.58	0.28	0.00	Conc.	79	1.06	0.30	0.21	0.00	Conc.
M.F.Crow River nr New London, Town Hall Road	158	1.34	0.35	0.27	0.00	Conc.	0					
Maple River nr Rapidan, CR35	428	4.41	0.56	0.55	0.00	Conc.	390	2.47	0.32	0.36	0.00	Conc.
Maple River nr Sterling Center, CR18	273	3.83	0.15	0.11	0.00	Conc.	255	2.13	0.04	0.01	0.10	Chem.
Middle Branch Root River nr Fillmore, CSAH5	50	4.05	1.18	0.82	0.00	Conc.	50	2.33	0.99	0.80	0.00	Conc.

Middle Fork Crow River nr Spicer, 275th st	69	1.54	0.27	0.18	0.00	Conc.	0					
Minnesota River at St. Peter, MN22	91	4.63	0.65	0.53	0.00	Conc.	90	2.95	0.38	0.38	0.00	Conc.
Minnesota River nr Jordan, MN	10	5.51	0.71	0.75	0.00		10	3.48	0.43	0.50	0.01	
Minnesota River nr Lac Qui Parle, MN	146	3.15	0.26	0.10	0.00	Conc.	146	1.85	0.06	0.01	0.15	Chem.
Mustinka River nr Norcross, MN9	48	4.19	0.23	0.21	0.00	Conc.	47	2.26	0.09	0.04	0.18	Chem.
Mustinka River nr Wheaton, CSAH9	103	3.75	0.34	0.45	0.00	Conc.	100	2.07	0.20	0.26	0.00	Conc.
Nicollet CD13A nr North Star, MN99	184	2.41	0.20	0.09	0.00	Conc.	0					
Nicollet CD46A nr North Star, CSAH13	177	2.83	0.37	0.25	0.00	Conc.	0					
North Branch Kandiyohi CD29 nr Sunburg, CSAH1 (W side)	73	1.85	0.11	0.03	0.15	Chem.	2	1.00	0.00	0.00	0.97	
North Branch Middle Fork Zumbro River nr Oronoco,5th St	47	3.88	1.11	0.71	0.00	Conc.	47	1.93	0.82	0.58	0.00	Conc.
North Eden Creek nr Franklin, CSAH10	39	2.21	0.68	0.70	0.00	Conc.	39	1.30	0.41	0.62	0.00	Conc.
North Fork Whitewater River nr Elba, Fairwater St	81	2.62	1.80	0.69	0.00	Conc.	81	1.25	1.32	0.67	0.00	Conc.
North Fork Zumbro River nr Mazeppa, CSAH7	47	3.88	1.67	0.87	0.00	Conc.	47	2.01	1.12	0.79	0.00	Conc.
Otter Tail River at Breckenridge, CSAH16	153	3.58	0.43	0.12	0.00	Conc.	144	1.74	0.23	0.06	0.00	Conc.
Pipestone Creek nr Pipestone, CSAH13	41	3.75	0.27	0.24	0.00	Conc.	41	2.30	0.16	0.14	0.01	Conc.
Plum Creek nr Walnut Grove, CSAH10	152	4.05	0.66	0.34	0.00	Conc.	64	1.86	0.30	0.38	0.00	Conc.
Pomme De Terre River at Appleton, MN	314	3.64	0.28	0.07	0.00	Conc.	231	2.06	0.14	0.03	0.01	Conc.
Pomme de Terre River nr Hoffman, CR76	87	3.12	-0.32	0.05	0.03	Dil.	87	2.13	-0.20	0.03	0.09	Chem.
Red Lake River at Fisher, MN	242	3.98	0.72	0.48	0.00	Conc.	215	2.01	0.48	0.45	0.00	Conc.
Red Lake River nr Red Lake Falls, CR13	60	2.67	0.49	0.41	0.00	Conc.	38	1.31	0.26	0.20	0.00	Conc.
Red River of the North at Grand Forks, ND	176	4.60	0.63	0.38	0.00	Conc.	144	2.39	0.50	0.41	0.00	Conc.
Redwood River at Russell, CR15	162	3.15	0.51	0.59	0.00	Conc.	129	2.05	0.30	0.46	0.00	Conc.
Redwood River nr Redwood Falls, MN	375	3.98	0.44	0.34	0.00	Conc.	251	2.41	0.15	0.09	0.00	Conc.
Rock River nr Hardwick, CR8	33	3.42	-0.04	0.01	0.59	Chem.	0					
Root River nr Houston, MN	115	4.88	1.67	0.74	0.00	Conc.	111	2.72	1.50	0.70	0.00	Conc.
Root River nr Mound Prairie, CSAH25	220	4.34	1.55	0.73	0.00	Conc.	220	2.68	1.03	0.30	0.00	Conc.
Root River nr Pilot Mound, MN	291	4.06	1.43	0.66	0.00	Conc.	117	3.40	0.34	0.06	0.01	Conc.
Roseau River below State Ditch 51 nr Caribou, MN	NA						0					
S Br. Wild Rice River at CR27 nr Felton, MN	40	2.63	0.61	0.66	0.00	Conc.	39	1.35	0.36	0.54	0.00	Conc.
Sand Hill River at Climax, MN	253	4.02	0.57	0.47	0.00	Conc.	212	1.97	0.41	0.47	0.00	Conc.
Sauk River nr St. Martin, CR12	124	2.52	0.09	0.01	0.27	Chem.	0					
Seven Mile Creek nr North Star, 0.3mi us of US169	1	3.74	2.25	0.56	0.46		0					
Seven Mile Creek nr North Star, 0.6mi us of US169	26	2.19	0.76	0.50	0.00	Conc.	0					

Shakopee Creek nr Benson, UNN TWP road (1 mile W MN29)	284	4.17	-0.12	0.05	0.00	Dil.	88	2.69	-0.07	0.05	0.04	Dil.
Shakopee Lake Inlet nr De Graff, CSAH4	87	2.92	0.15	0.04	0.06	Chem.	30	1.95	0.15	0.08	0.12	Chem.
Shakopee Lake Outlet nr De Graff, Chip-Swift St NE	84	3.69	-0.27	0.23	0.00	Dil.	31	2.45	-0.38	0.40	0.00	Dil.
Shell Rock River nr Gordonsville, CSAH1	161	2.60	0.17	0.05	0.00	Conc.	158	1.81	0.05	0.01	0.35	Chem.
Sleepy Eye Creek nr Cobden, CR8	231	3.57	0.37	0.28	0.00	Conc.	146	2.07	0.16	0.13	0.00	Conc.
South Branch Buffalo River nr Baker, CR57	22	3.09	0.14	0.12	0.10		0					
South Branch Buffalo River nr Glyndon, CR79 (28th AveS)	89	3.37	0.03	0.00	0.55	Chem.	75	1.74	0.03	0.01	0.53	Chem.
South Branch Middle Fork Zumbro River nr Oronoco,5th St	45	4.15	1.17	0.77	0.00	Conc.	45	2.29	0.85	0.67	0.00	Conc.
South Branch Root River at Lanesboro, Rochelle Ave N	116	4.29	1.19	0.68	0.00	Conc.	116	3.85	0.32	0.08	0.00	Conc.
South Branch Two Rivers River at Hallock, MN175	25	3.51	0.63	0.65	0.00	Conc.	25	1.36	0.47	0.60	0.00	Conc.
South Branch Yellow Medicine River nr Minneota, CSAH26	32	4.24	0.49	0.66	0.00	Conc.	31	2.69	0.36	0.61	0.00	Conc.
South Fork Crow River nr Cosmos, MN7	75	3.38	0.00	0.00	0.99	Chem.	75	2.10	0.03	0.01	0.37	Chem.
South Fork Watonwan River nr Madelia, CSAH13	52	3.67	0.40	0.53	0.00	Conc.	52	2.38	0.25	0.36	0.00	Conc.
South Fork Whitewater River nr Altura, CR112	126	2.92	0.38	0.15	0.00	Conc.	75	1.55	1.22	0.69	0.00	Conc.
Split Rock Creek nr Jasper, 201st St	107	3.91	0.35	0.31	0.00	Conc.	107	2.33	0.24	0.21	0.00	Conc.
Spring Creek nr Hanley Falls, 480th St	94	2.76	-0.07	0.02	0.17	Chem.	49	1.80	-0.04	0.02	0.39	Chem.
Spring Creek nr Sleepy Eye, CSAH10	40	2.67	1.04	0.82	0.00	Conc.	40	1.54	0.60	0.70	0.00	Conc.
Ten Mile Creek nr Lac Qui Parle, CSAH18	50	2.65	0.43	0.39	0.00	Conc.	50	1.24	0.22	0.14	0.01	Conc.
Thief River nr Holt, CSAH7	73	3.05	0.17	0.08	0.02	Conc.	58	1.30	0.07	0.03	0.23	Chem.
Threemile Creek nr Green Valley, CR67	140	3.76	0.18	0.11	0.00	Conc.	98	2.14	0.06	0.03	0.09	Chem.
Turtle Creek at Austin, 43rd St	137	3.60	0.08	0.02	0.08	Chem.	119	2.05	0.14	0.05	0.01	Conc.
Vermillion River at Farmington, Denmark Ave	99	2.14	0.57	0.24	0.00	Conc.	99	1.01	0.53	0.29	0.00	Conc.
Vermillion River nr Vermillion, CSAH85	206	2.56	0.56	0.15	0.00	Conc.	206	1.39	0.39	0.12	0.00	Conc.
Wabasha Creek nr Franklin, CSAH11	39	2.84	0.50	0.36	0.00	Conc.	39	1.72	0.33	0.33	0.00	Conc.
Watonwan River nr Garden City, CSAH13	412	4.18	0.47	0.44	0.00	Conc.	374	2.54	0.22	0.19	0.00	Conc.
Watonwan River nr La Salle, CSAH3	123	4.02	0.33	0.22	0.00	Conc.	123	2.43	0.22	0.16	0.00	Conc.
West Branch Lac Qui Parle River at Dawson, Diagonal St	206	2.86	0.26	0.30	0.00	Conc.	139	1.55	0.14	0.13	0.00	Conc.
West Branch Rum River nr Princeton, CR102	66	1.65	0.33	0.46	0.00	Conc.	66	3.17	0.42	0.14	0.00	Conc.
West Fork Beaver Creek nr Bechyn, 320th St	108	3.32	-0.22	0.15	0.00	Dil.	63	1.80	-0.10	0.02	0.22	Chem.
West Fork Des Moines River at Jackson, River St	153	3.93	0.38	0.37	0.00	Conc.	152	2.64	0.17	0.12	0.00	Conc.
West Fork Des Moines River nr Avoca, CSAH6	48	4.33	0.43	0.49	0.00	Conc.	48	2.77	0.30	0.46	0.00	Conc.
Whitewater River nr Beaver, CSAH30	181	3.90	1.83	0.80	0.00	Conc.	178	1.84	1.59	0.80	0.00	Conc.

Yellow Bank River nr Odessa, CSAH40	195	3.17	0.80	0.65	0.00	Conc.	172	1.74	0.49	0.53	0.00	Conc.
Yellow Medicine River nr Granite Falls, MN	73	3.72	0.58	0.56	0.00	Conc.	72	2.46	0.38	0.52	0.00	Conc.
Yellow Medicine River nr Hanley Falls, CR18	106	3.90	0.44	0.42	0.00	Conc.	54	2.61	0.23	0.30	0.00	Conc.