MEMO: Limnological Dynamics of the Upper Mississippi River and Tributaries during the Growing Season: A Case for Increased Sampling Effort

Shawn M. Giblin[1] and Robert M. Burdis[2]

[1] Wisconsin Department of Natural Resources Mississippi River Monitoring Field Station, 2630 Fanta Reed Road, La Crosse, Wisconsin 54603
[2] Minnesota Department of Natural Resources Mississippi River Monitoring Field Station, 1801 S. Oak Street, Lake City, Minnesota 55041

Introduction

The mission of the Long Term Resource Monitoring (LTRM) element is to provide decision-makers with information needed to maintain the Upper Mississippi River System (UMRS) as a viable large river ecosystem. The focus of the water quality component of the LTRM is to collect limnological information relevant to the suitability of aquatic habitat for biota and transport of material within the system. The program as designed in 1993 was robust with wide support from the Environmental Management Program partners. In June of 2002, a panel of water quality experts recommended that detailed analyses be conducted before making any changes to the existing water quality program (Soballe and Houser, 2006). However, budget problems plagued the LTRM for many years due to a lack of cost-indexing in the authorizing language, and inadequate annual appropriations. During the 1990s, these problems were dealt with through streamlining of operations and efficiencies gained by implementing practices such as electronic data capture and centralized bottle washing. In late 1999, the program was forced to reduce efforts by approximately twenty-five percent and included a period of no sampling between October 2002 and September 2004. In October 2004 the Minimum Sustainable Program (MSP) was implemented to bring the program within pre-determined fiscal limits. Under the MSP design, cuts were made to the fixed-site portion of the mixed-model design including several tributaries, all out-pool sites, and sampling frequency at the remaining sites was reduced to monthly in summer and bimonthly in winter. Beginning in 2007, the LTRM field stations in Minnesota and Wisconsin, hereafter referred to as FS 1 and FS 2 implemented a sampling regimen that restored some of the sampling effort that was lost under the MSP using additional program element (APE) funds. FS 1 and FS 2 restored biweekly fixed-site water quality monitoring during the summer period with additional sampling in July and August. FS 2 also restored monitoring to four historical fixed-sites in Pools 8 and 9.

The growing season (May-September) represents one of the most critical periods of the year in river ecosystems. The predictable decrease in discharge over this period can result in dramatic shifts in suspended solids concentration, nutrient concentrations, and light availability that can have a profound effect on river biota and productivity (Likens, 2010). Peaks in solar irradiance, temperature, and primary productivity make the growing season one of the most limnologically dynamic periods of the year (Wetzel, 2001). Temporal and spatial shifts in suspended solids,

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nitrogen, and phosphorus concentration can create conditions where different factors (i.e. light, nitrogen, or phosphorus) can limit primary productivity at different times throughout this period (Koch et al., 2004). In addition, summer is often the period that is used to assess waterbodies for water quality impairments.

River discharge is the key variable to which most limnological variables and ultimately biota respond. Changes in discharge result in variable rates of delivery of sediment, nitrogen, and phosphorus (Balogh et al., 1997; Goolsby et al., 2000; Likens, 2010). These changes in discharge can affect the role that point vs. non-point sources play in determining the concentration of suspended solids and nutrients as well as the ratios of nutrients in the river (Goolsby et al., 2000; Withers and Jarvie, 2008). Extreme precipitation events during the growing season can result in a rapid increase in discharge and a decline in water quality (Chick et al., 2009, Likens, 2010).

The concentration of total suspended solids (TSS) is a measure of the amount of material in suspension in the water column. TSS concentration within river systems tends to increase with increasing discharge (Likens, 2010). TSS is an important variable because it can limit primary productivity by blocking light, negatively affects macroinvertebrate respiration and behavior, results in habitat loss, and affects fishes by reducing feeding efficiency and smothering spawning habitat (Walters, 1995). Primary productivity of large rivers has generally been viewed to be light limited as a result of high TSS (Vannote et al., 1980). More recent investigations have suggested that primary productivity of large rivers can be limited by nutrients rather than light, especially during periods of reduced discharge (Reynolds and Descy, 1996; Koch et al., 2004). Additionally, the introduction of the zebra mussel *Dreissena polymorpha* to the Upper Mississippi River in 1994, and quagga mussel *Dreissena bugensis* sometime prior to 2004, has likely increased water clarity throughout the system making light saturation more likely than prior to dreissenid invasion (Strayer et. al., 1999; James et al., 2000).

Nitrogen is an essential plant nutrient that can limit the biomass of phytoplankton and aquatic macrophytes at low concentrations. Nitrogen concentration tends to increase with increasing discharge as non-point input from agriculturally dominated tributary watersheds is delivered to the UMRS (Goolsby et al., 2000). A typical seasonal nitrogen pattern in the UMRS is one of a late-spring/early summer peak followed by a gradual decline later in the growing season as uptake and denitrification increase (Strauss, 2006; James et al., 2008; Houser and Richardson, 2010). Sites associated with high rates of denitrification are typically sites where oxic and anoxic conditions occur simultaneously or alternate in time. These conditions are likely to occur in plant beds and may be a major reason why DIN concentrations are frequently driven below detection during the growing season in highly vegetated systems (Scheffer, 2004). The recent increase in submersed aquatic vegetation (SAV) coverage in the UMRS (Johnson and Hagerty, 2008) suggests denitrification within plant beds could be increasing over time.

Phosphorus is an essential plant nutrient that can limit the biomass of phytoplankton and aquatic macrophytes at low concentrations. Increased phosphorus concentration has been linked to increases in summer biomass of nitrogen-fixing cyanobacteria in numerous aquatic ecosystems (Smith and Schindler, 2009). The prevalence of harmful algal blooms appears to be on the rise worldwide due to increased nutrient loading (Reynolds, 2006). In Lake Pepin in 1988, elevated phosphorus concentration caused severe cyanobacteria blooms and ultimately fish kills (Engstrom et al., 2009). Phosphorus concentration tends to exhibit a seasonal pattern that is the inverse of nitrogen. Growing season concentrations tend to be lowest during late spring/early summer and gradually increase as the season progresses (Houser and Richardson, 2010). This increase is coincident with a reduction in dissolved oxygen in the backwaters which can result in anoxic conditions at the sediment-water interface resulting in a substantial release of iron and aluminum-bound phosphorus (James, 1995). Additionally, loading from point sources such as wastewater treatment plants remains stable as discharge decreases through the summer months which can lead to increased phosphorus concentration during late summer (Withers and Jarvie, 2008).

Excessive loading of nitrogen and phosphorus due to human activities (cultural eutrophication) has become a major problem for surface waters worldwide. Inputs due to intensive agriculture, wastewater discharges, stormwater runoff, construction activities, and other factors have resulted in increased concentration of nitrogen and phosphorus in the UMRS (Goolsby and Battaglin, 2001; Engstrom et al., 2009). Effects of excessive nitrogen and phosphorus loading to surface waters include increased biomass of phytoplankton and macrophytes, shifts to algal species that may be toxic or inedible, increased incidence of fish kills, reductions in species diversity, decrease in water transparency, taste and odor problems with drinking water, and reduction in dissolved oxygen concentration (Smith and Schindler, 2009).

River phytoplankton live in a highly dynamic environment and are subject to multiple stressors. As in lakes, light and nutrients are the primary factors determining phytoplankton biomass and growth (Likens, 2010). Light limitation of phytoplankton is generally viewed to be more common than nutrient limitation among high-order rivers subject to anthropogenic stressors such as intensive agriculture (Wetzel, 2001). However, nutrients can become limiting in some river systems, particularly during periods of low discharge and turbidity (Reynolds and Descy, 1996). Pools 4 and 8 of the UMRS have experienced a statistically significant reduction in TSS concentration in recent years (Johnson and Hagerty, 2008). Reduced discharge, filtration by zebra mussels (*Dreissena polymorpha*) and quagga mussels (*Dreissena bugensis*), and increased sedimentation of TSS due to increased SAV coverage within both Pools 4 and 8 are all factors likely working in concert to increase water transparency in recent years. Reduced discharge in recent years has also likely resulted in the reduction of external nutrient loading of both nitrogen and phosphorus from tributary watersheds (Royer et al., 2006). Reduced nutrient concentrations coupled with improved water transparency may be creating conditions where nutrients rather than light are limiting phytoplankton production under certain spatial and temporal scenarios.

The nutrient ratio of phytoplankton and seston can be directly tied to the nutrient ratio of water being delivered to a particular waterbody. A rich literature has developed using nutrient ratios to predict cyanobacterial dominance (Downing et al., 2001), document changes in nutrient loading and assimilation (Justic et al., 1995; Houser and Richardson, 2010), using nutrient ratios to predict limiting nutrient(s) (Hecky and Kilham, 1988; Elser et al., 1990, Duarte, 1992; Hall and Cox, 1995; Hillebrand and Sommer, 1999), and predict fertility (Downing and McCauley, 1992). Dramatic shifts in N:P over the growing season may provide some indication of spatial and temporal shifts in nutrient availability and limitation in the UMR.

Climate is the primary factor regulating discharge regime in rivers. Extreme events (e.g., drought, storms) are prominent drivers of system-wide change. Increased sampling frequency increases the probability of capturing these events in a stochastic system such as the UMR. A review of the LTRM water quality component suggested that episodic phenomena are important to river biota and that information is needed at fine temporal scales (Soballe and Houser, 2006). Many climatologists now agree that the intensity and frequency of extreme precipitation events are likely to increase in the future (Poff et al. 2002; Jha et al.2006). Intense summer storms often result in significant localized runoff events. Recent floods of historic nature have been observed on the UMR near Pool 8 in 2007 and on the Upper Iowa River in 2008 (Chick et. al. 2009; Fischer and Eash, 2010). Additionally, a severe drought occurred in the UMR from 1987 to 1989 that resulted in a major decline in SAV (Kimber et al., 1995). The loss of SAV resulted in an increase in turbidity (i.e. shift in stable state) that persisted for many years. Anglers and waterfowl hunters expressed a great deal of dissatisfaction with the resource during this turbid period. Various theories have been postulated as to the decline of SAV but the cause remains unknown. A temporally robust sampling schedule during the growing season will likely increase the probability of isolating the mechanism driving such a loss of in the future.

The purpose of this evaluation was threefold: (1) to determine what information would be lost by returning to the MSP sampling design, (2) determine if the additional sampling effort improved our understanding of limnological dynamics and events within the Mississippi River and tributaries, and (3) determine how the assessment of eutrophication in Lake Pepin would differ between the additional monitoring effort and the effort under the MSP.

Study Area

Pool 4 is located between Lock and Dam 3 (Red Wing, MN) and Lock and Dam 4 (Alma, WI; Figure 1). It is 73 km long and includes 14,700 ha of aquatic habitat that has been stratified into the following aquatic areas: main channel; side channel; backwaters; tributaries; and lakes (i.e., Lake Pepin). Lake Pepin, a large, natural widening of the river formed by the Chippewa River delta, divides Pool 4 into upper and lower sections. Pool 8 is located between Lock and Dam 7 (Dresbach, MN) and Lock and Dam 8 (Genoa, WI). It is 39 km long and encompasses 9,000 ha of aquatic habitat that has been stratified into the following aquatic areas: main channel; side channel; backwaters; tributaries; and impounded area. The upper section has a riverine aspect with numerous islands and braided channels whereas the lower section is a large open expanse of water (impounded area). Several tributaries empty into Pools 4 and 8, several of which are monitored by the LTRM.

Methods

The LTRM water quality sampling design since 1993 incorporates biweekly fixed-site sampling (FSS) and quarterly stratified random sampling (SRS). The mixed-model design provides information at both broad spatial scales with low temporal resolution (i.e., SRS), and at small spatial scales with higher temporal resolution (i.e., FSS). SRS tracks conditions at spatial scales corresponding to sampling strata or larger (i.e., whole pool or sampling reach) and at seasonal to annual time scales or longer. In contrast, FSS provides information at more frequent intervals (i.e., within season) at specific points of interest such as tributaries, tailwaters, and backwaters with high habitat value.

Additional fixed-site water quality sampling above the MSP level was conducted from 2007 to 2009 by FS 1 and FS 2. Four fixed-sites that had been discontinued were reestablished by FS 2 in Pools 8 and 9. In addition, sampling was conducted in July and August above the MSP level by both field stations that maintained biweekly sampling through early September. Under the MSP only monthly sampling would occur in July and August.

Field and laboratory methods are described in the LTRM water quality procedures manual (Soballe and Fischer, 2004). Surface measurements and were analyzed by strata to evaluate differences among aquatic areas. Values for parameters that were below the detection limit were assigned values one-half of the detection limit. Dissolved inorganic nitrogen (DIN) was calculated as the sum of ammonia plus ammonium ($NH_{x)}$ and nitrate plus nitrite (NO_x). Fluorometric chlorophyll data from Lake Pepin was calibrated against spectrophotometrically determined chlorophyll using a subset of samples for which both methods of chlorophyll *a* (CHL) determination were used. Water discharge data were obtained from the U.S. Geological Survey gauge in Pool 6 at Winona, MN and the U.S. Corps of Engineers gauge at Lock and Dam 7 near Dresbach, MN.

Fixed-site data from May 15th through September 15th was used to calculate monthly medians for dissolved oxygen (DO); total nitrogen (TN); total suspended solids (TSS); total phosphorus (TP); fluorometric chlorophyll (CHLF); and TN:TP ratio and graphed to examine seasonal dynamics. The coefficient of variation (CV) was calculated for pH; DO; TN; TP; NO_x; NH_x; soluble reactive phosphorus (SRP); TSS; and CHLF by habitat class and field station with and without the additional sampling effort. CV is an indicator of sample variability relative to the mean (Zar, 1984) and was used to evaluate the effect of sampling effort on variability. A higher CV for the data, including the additional sampling effort, would indicate a greater relative dispersion among the data. In addition, data were examined for unique events that occurred in Pools 4 and 8 during the 2007-2009 growing seasons. Analysis was conducted using SAS (version 9.1).

To examine the effects of different levels of sampling effort on summer means of TP, CHL, and Secchi transparency in Lake Pepin *t*-tests were performed for each summer (Jun 1st - Sep 15th) comparing sampling efforts with and without additional samples. Additional sampling in Lake Pepin included sampling four fixed-sites, once in late-July and again in late-August resulting in 8 additional samples per summer. Data were logarithmically transformed to normalize the distribution prior to t-tests. Additionally, means were calculated on unaltered data and the percent differences between the means at different levels of sampling effort were calculated. CV was calculated for all three parameters with and without the additional monitoring data for all three parameters to examine relative variability.

Results

Discharge during the Study Period

Discharge during the 2007-2009 growing seasons was notably low. Mean monthly discharge at Winona, MN was well below the long-term mean at some period during all three years (Figure 2). Discharge was frequently less than one-half of the long-term mean during the sampling period. Discharge during 2007 and 2009 was well below the long-term mean for the entire growing season. Localized storms of a record nature occurred in Pool 8 during August of 2007. These storms were localized and occurred largely downstream of the Winona gauge. For this reason, discharge values for August 2007 likely underestimate the true discharge in Pool 8 for this month. Discharge during 2008 was above the long-term mean during May and June but dropped precipitously to values similar to 2007 and 2009 by August.

Water Quality during the Study Period

Examination of monthly median water quality data for Pool 4 revealed strong seasonal shifts and highly dynamic conditions through the growing season (Figure 3). Dissolved oxygen concentrations were at a maximum in May with minima being observed in July or August for all habitat classes. The main channel in Pool 4 typically exhibited the lowest DO concentration of any of the habitat classes. Total nitrogen concentration exhibited a steady decline through the growing season with the exception of tributaries which tended to remain steady. The backwater sites exhibited the lowest TN concentration of any of the habitat classes for all months. Median TSS concentration tended to remain steady for all classes through the growing season with the highest values being observed in the tributaries for all months. The TP concentration exhibited a strong seasonal shift with minima observed in May for all classes and maximal concentration in August for most habitat classes. The concentration of chlorophyll *a*, a surrogate for algal biomass, exhibited a great deal of variability with noticeable peaks in concentration in May, July, and to a lesser extent September. Lake Pepin exhibited the highest chlorophyll concentration among any of the habitat classes. The ratio of TN to TP (TN:TP) exhibited a strong seasonal shift for all habitat classes with maximal observations in May (~20-30) and minima observed during August and September (~5-7).

Examination of monthly median water quality data for Pool 8 revealed strong seasonal shifts and highly dynamic conditions through the growing season equal or greater than those observed in Pool 4 (Figure 4). Dissolved oxygen concentrations indicated maximum concentrations in May with minima observed in August for all habitat classes. Total nitrogen concentration exhibited a steady decline through the growing season for all habitat classes with the exception of the tributaries which remained fairly consistent. Backwater TN concentrations were measurably lower than the other habitat classes for all months. The concentration of TSS exhibited a slight decline as the season progressed for most classes. The highest TSS concentrations were observed in the tributaries for all months. TP concentration generally increased as the season progressed with the backwaters consistently exhibiting the lowest concentrations. Chlorophyll *a* concentration exhibited a general decrease as the season progressed with maxima observed in May in the main channel and impounded classes (25-40 µg/L). The majority of values for June-September were very low (<10 µg/L) for all habitat classes. The TN:TP ratio exhibited a strong seasonal shift with a maximum of 22.86 (main channel- May) and minimum of 5.39 (impounded- September).

Examination of the percent of TP present as SRP for Pool 4 showed a steady increase in the percent of TP as SRP throughout the season (Figure 5). Pool 8 also showed a steady increase in percent SRP with the exception of the tributaries. This phenomenon was observed for both Pools suggesting factors within the UMRS may be causing this trend. An obvious disparity was observed in percent SRP between the main channel and impounded habitat classes (high % SRP) and the backwater and tributary habitat classes (low % SRP) by late summer. Examination of the percent of TN present as dissolved inorganic nitrogen (DIN) also revealed some interesting differences among habitats and months (Figure 6). The percent of TN present as DIN (% DIN) for Pool 4 showed strong seasonal decline as the season progressed with the backwaters generally showing the lowest values. Pool 8 values for % DIN generally showed a slight decline as the season progressed with the backwater habitat class showing substantially lower values than the other classes (minimum- 4.55% in July).

Variability Due to Increased Sampling Effort

Substantial differences in the CV were observed between the MSP and restored monitoring sampling frequencies. Analysis of nine water quality parameters by habitat class revealed that a higher coefficient of variation was observed for 72 % of the comparisons for FS 1 and 61 % of the comparisons for FS 2 when the additional sampling dates were added (Tables 1 and 2).

Lake Onalaska 2007

The French Island Spillway sampling site (M702.2T) is situated at one of two outlets of Lake Onalaska and provides a good indication of water quality within the west side of Lake Onalaska. Limnological observations from this site in 2007 illustrate the dynamic nature of water quality in the UMRS and the need for consistent biweekly monitoring over this period (Figures 7 and 8). Discharge fell from around 38,000 cubic feet per second (cfs) in mid to late-May to around 10,000 cfs in late-July and early-August. Historic storms resulting in precipitation greater than 12" in localized areas, hit the area in late-August causing a rapid increase in discharge to > 40,000 cfs. Following the August storms, discharge abruptly decreased to below 20,000 cfs. Algal biomass was characterized by two spikes- one in early-May and one in early-July. Dissolved silica concentration showed an increase through the season with very low values (< 1250 µg/L) observed in mid-May associated with a diatom bloom. The concentration of SRP was very low in mid-May (15 µg/L) through early-July until discharge dipped below 10,000 cfs at which point record concentrations (280 µg/L on 7/25 and 412 µg/L on 8/6) were observed. Ammonia plus ammonium concentration remained quite low until early-August when unusually high concentrations were observed (210 µg/L on 8/6 and 138 µg/L on 8/23). The high NH_x observation on 8/6 was during low discharge conditions, while the observation on 8/23 was associated with the historic flood event. Nitrate plus nitrite concentration remained fairly high until a substantial algal bloom in early-July when it dropped precipitously to 122 µg/L on 7/9 and 14 µg/L on 7/25. The concentration of NO_x remained low until the mid-August flooding. The concentration of DIN reached a minimum of 30 µg/L on 7/25. Water temperature ranged from around 17 C in mid-May to nearly 30 C during the major algal bloom in early-July. Dissolved oxygen concentrations were near saturation until the major algal bloom in early-July when they became supersaturated. Dissolved oxygen values were well below saturation during August and into early-September. The concentration of TSS showed three noticeable spikes over the growing season. The highest concentration observed was in mid-May which was likely the result of both inorganic solids during elevated discharge and biogenic turbidity as a result of a diatom bloom. Another spike was observed in early July and was largely the result of a substantial algal bloom. The third spike was likely the result of upland sediment exported during late-August flooding. The TN:TP ratio showed a general decline through the growing season will a maximum of 13.97 in mid-June to 2.43 in late-July when discharge was below 10,000 cfs. The ratio of DIN:SRP, inorganic nutrients immediately available to biota, showed an even greater disparity as the season progressed. The range observed was from a maximum in mid-April (30.87) when considerable control was being exerted on SRP during a diatom bloom, to a minimum of 0.11 in late-July on the tail end of a substantial algal bloom in which DIN was driven to near zero.

Upper Iowa River 2008

Historic storms descended on the Upper Iowa Basin on June 7-8th resulting in greater than seven inches of rain at many locations throughout the basin in less than 24 hours. The river at Dorchester crested at an all-time high on June 9th, surpassing the previous record dating back to 1948. A maximum peak discharge of 34,100 cfs was recorded at the USGS gauge in Decorah, IA.

Limnological observations from the Upper Iowa River in 2008 illustrate the extent of intra-annual variability as well as the degree of sediment and nutrient export during extreme events (Figure 9). Discharge in the Upper Iowa River was above average for the entire 2008 growing season but held steady around 1,000 cfs until the historic storms hit the region on June 7-8th. Despite sampling two days after the river crest, discharge was still nearly 10,000 cfs at Dorchester, IA on the day of sampling. A second increase in discharge occurred on 7/8 following roughly two inches of rainfall. Discharge gradually decreased for the remainder of the growing season following the July event. The TSS concentration showed two dramatic spikes (891 mg/L on 6/11 and 1466 mg/L on 7/8) as a result of the two peaks in discharge. The concentration of TP showed two spikes consistent with the flood event as a result of phosphorus adsorbed to sediment delivered to the river. TN also showed two substantial spikes (8.55 mg/L on 6/11 and 7.49 mg/L on 7/8) as a result of increased export from the watershed during the two events. Chlorophyll *a* showed three distinct spikes during the 2008, two during periods of somewhat reduced discharge and one during the July rainfall event.

Upper Iowa River 2009

The 2009 growing season was dramatically different than the conditions observed in 2008. Unlike 2008, discharge during 2009 was generally at or below the long-term mean (Figure 10). Discharge showed a general decrease through the growing season until discharge was well below average at the end of the growing season. As a result of reduced discharge, TSS concentration was reduced creating a more favorable light climate for phytoplankton development. As a result of increased algal abundance and reduced external loads, SRP was frequently < 10 µg/L and was driven below the detection limit (< 1 µg/L) during and following a major algal bloom in late August. Examination of water quality in the Upper Iowa River in 2008-2009 illustrates the degree of inter-annual variability, even in consecutive years, in sediment and nutrient loads as well as algal biomass that can be observed in tributary rivers of the UMR (Table 3).

Main Channel Pool 8 in 2007

The Minnesota Island sampling site (M701.1D) is situated in the main channel thalweg 1.4 miles downstream of Lock and Dam 7. Data from this site illustrates the dynamic nature of water quality in the UMR and the need for consistent biweekly monitoring over the entire growing season (Figures 11 and 12). Discharge was around 40,000 cfs in mid-May and gradually decreased to < 9,000 cfs by late-July. Discharge remained low until intense storms, centered mostly downstream of M701.1D, descended on the area resulting in a rapid increase in discharge. The concentration of chlorophyll *a* was quite high (40-80 µg/L) from mid-May through early-June during a major diatom bloom and again from mid-July (124 µg/L) to early-August (41 µg/L) during a major bloom of the cyanobacterium, *Aphanizomenon sp*. The concentration of TSS showed three distinct spikes in 2007: one in early-June associated with a major diatom bloom, a second during early-July associated with the major *Aphanizomenon* bloom, and a third in late-August due to intense storms. The concentration of TP showed a steady increase through the season with a noticeable spike associated with the late-August storms. The concentration of TN tracked very closely with discharge with a maximum in late-May and minimum in late-July. A steady decrease in TN:TP was observed through the growing season with a maximum in mid-May (23.99) and values < 6 from late-July through early-September. The concentration of dissolved silica was quite low (< 2000 µg/L) from mid-May until early-June in association with a major diatom bloom and showed a general increase throughout the remainder of the growing season. The concentration of DIN was fairly high until mid-July when concentration became very low (73 µg/L on 7/11, 17 µg/L on 7/26, and 249 µg/L on 8/9) until the late-August flooding. The concentration of SRP was quite low (range 5-29 µg/L) during the early season diatom bloom and quite low again during the major *Aphanizomenon* bloom that occurred during the low discharge period from early to late-July (range 13-45 µg/L). The DIN:SRP ratio was very dynamic, with a maximum (> 200) during the spring diatom bloom and minimum (0.378) during the *Aphanizomenon* bloom. The concentration of DO was quite high during the diatom and *Aphanizomenon* blooms, but decreased dramatically as the *Aphanizomenon* bloom waned and discharge increased following the late-August storms.

Main Channel Pool 8 in 2008

Discharge during the 2008 growing season was generally higher than discharge during the 2007 and 2009 growing seasons. Discharge began the season fairly high (79,500 cfs in mid-May) and gradually decreased to less than 20,000 cfs by early-August (Figure 13). Chlorophyll concentration was fairly low in 2008 with two pronounced peaks, one diatom bloom in late-May, following a sharp two-week decrease in discharge from ~80,000 to ~50,000 cfs, and another associated with a cyanobacteria bloom which began in early-August and peaked in mid-August (Figure 14). The concentration of TSS generally decreased through the growing season with notable peaks associated with the late-May diatom and August cyanobacteria blooms. The concentration of TP showed a gradual increase through the growing season while TN tracked very closely with river discharge. The TN:TP ratio ranged from ~40 in late-May and late-June to 5.37 in early September. Dissolved silica concentration showed a general increase throughout the season with a marked reduction in late-May associated with a major diatom bloom. The concentration of DIN peaked in late-June and decreased in association with decreased discharge until a minimum was reached in mid-August (311 µg/L) during the peak of the cyanobacteria bloom. The concentration of SRP was at a minimum (6 µg/L) in late-May during the peak of the diatom bloom and at a maximum in late September (144 µg/L) as discharge dropped below 11,000 cfs. The DIN:SRP ratio was dynamic, ranging from a maximum of 417 during peak diatom biomass to a minimum of 2.74 in early September following a month of persistent cyanobacteria blooms. Dissolved oxygen tracked very favorably with algal biomass, showing pronounced peaks during peak diatom biomass in late-May and peak cyanobacteria biomass in mid-August with a period of reduced DO in association with reduced algal biomass from early-June to late-July.

Main Channel Pool 8 in 2009

Discharge during the 2009 growing season was generally the lowest of any during the 2007-2009 period. Discharge reached a maximum in mid-May (33,100 cfs) and a minimum in mid-July (9,700 cfs) (Figure 15). Chlorophyll *a* was extremely low during 2009 with a peak (19.1 µg/L) in late-May associated with a diatom bloom and another modest peak from mid-August to early-September (range 9.6-10.9 µg/L) due to an *Aphanizomenon* bloom (Figure 16). The concentration of chlorophyll *a* was very low (< 4 µg/L) from mid-June through late-July. The concentration of TSS was also extremely low in 2009, with a maximum of only 11.7 mg/L in mid-May to < 5 mg/L in mid-June and mid-July. The concentration of TP showed a steady increase as the season progressed. Unlike 2007 and 2008, the concentration of TN did not show a close association with discharge in 2009, although concentrations were low. The TN:TP ratio decreased during the growing season with a maximum in early-June (18.9) and a minimum in early-September (5.28). The concentration of dissolved silica generally increased through the growing season with observations of < 2000 µg/L from mid-May to early-June during a modest diatom bloom. The concentration of DIN was notable in that maximum concentrations were observed in mid and late-July when discharge was very low (7/16, DIN 1227 µg/L, discharge 9,700 cfs; 7/27, DIN 1229 µg/L, discharge, 14,700 cfs). This pattern was substantially different than that observed in 2007 and 2008 when DIN dropped to extremely low levels when discharge decreased- especially below 20,000 cfs. The concentration of SRP generally increased through the growing season with a minimum in mid-May during the diatom bloom (18 µg/L) and a maximum in mid-August (140 µg/L), followed by a modest decrease associated with an *Aphanizomenon* bloom in late summer. The DIN to SRP ratio ranged from 24.5 during the diatom bloom to 4.01 during the *Aphanizomenon* bloom. The DO concentration tended to track fairly well with algal biomass with the highest observation (10.2 mg/L) being observed in mid-May during peak diatom biomass and the second highest being observed in early-September during peak *Aphanizomenon* biomass. The DO concentration was generally the lowest from mid-June to late-July when algal biomass was very low. Zebra mussel veliger sampling at Lock and Dam 8 conducted by the Wisconsin Department of Natural Resources indicated a strong increase in veligers from 2006-2009 (Figure 17). Given the two-year life expectancy among *Dreissena polymorpha*, it seems reasonable to assume that a large population of adult *Dreissena polymorpha* existed in Pool 8 in 2009 (Nalepa and Schloesser, 1993).

Lake Pepin 2007-2009

Results of the t-tests revealed there were no significant differences in summer mean CHL, TP, or Secchi during any year when comparing data with and without the additional sampling. However, mean summer CHL in 2008 was over 30% higher when the additional monitoring was included in the analysis (Table 4). Mean CHL was 18.6% higher in 2009 with the additional sampling and would have likely been higher in 2007 as well had a heavy rain event in August not reduced phytoplankton populations to very low levels. Similar to CHL, TP summer means were also higher with the additional sampling in all three years. Secchi showed the least amount of change between the levels of effort but did show a consistent pattern of decreasing transparency with the additional sampling (Table 4). Coefficient of variation was greatest for CHL, TP was intermediate and Secchi was the least variable parameter (Table 5). Comparison of variability between the data with and without the additional sampling revealed an increase in CV for two of the three variables analyzed (Table 5).

Discussion

Our data indicate that the UMR can exhibit a high degree of limnological variability during the growing season. This variability can be especially pronounced during periods of reduced discharge and intense precipitation as was observed in 2007-2009. Substantial seasonal variability was observed during the growing season including reduced DO concentrations during mid-summer (especially July and August), improved light availability due to reduced TSS, shifts in both algal abundance and species composition, a reduction in TN, an increase in TP, and dramatic shifts in TN:TP (Figures 3 and 4). Increased sampling resulted in an increase in the CV amongst the data. Additionally, implications for the attainment water quality goals within Lake Pepin were observed when the additional sampling effort was removed from the analysis.

A major goal of limnological monitoring is to sample during a wide range of environmental conditions so the full range of limnological conditions can be observed and analyzed. The result of the additional sampling effort was a substantive change in the relative variability of the data (Tables 1 and 2). A threshold in which increased sampling fails to produce an increase in variability among the data was not reached for 2007-2009. Failure to reach this threshold suggests the range of limnological conditions have not been fully sampled and demonstrates the need for additional sampling during the growing season in order to achieve a more accurate representation of conditions.

The examination of the percent of nitrogen and phosphorus present in dissolved, inorganic forms can provide an indication of the amount of nutrients immediately available to biota, the amount of assimilation and transformation occurring, and potential sources of nutrients. The availability of dissolved inorganic nutrients can be a primary driver of primary productivity, especially under conditions where light is not limiting. Analysis of SRP data revealed three major findings: 1) the concentration of SRP increases through the growing season, 2) the percentage of TP as SRP increases through the growing season, and 3) the percent of TP as SRP is quite different among habitat types (Figure 5). The increase in SRP concentration and percent of TP as SRP through the growing season has been frequently observed and is likely the result of release of phosphorus from the sediments under anoxic conditions, the increased role of wastewater discharge in total discharge under low flow conditions, and nutrient transformations as a result of dreissenid invasion (James, 1995, Haggard et al., 2001, Raikow et al., 2004 and Withers and Jarvie, 2008). The percentage of TP as SRP was consistently higher in the main channel than in the backwaters (Figure 5). Although inputs from wastewater discharges are important, it seems likely that nutrient transformation by zebra mussels is likely playing a role (Arnott and Vanni, 1996, Haggard et al., 2001). The reliance of *Dreissena polymorpha* on a hard substrate (rock, docks, bridges, etc.) for colonization, of which there is more of in the main channel, may be an important factor driving this difference (Scheffer, 1994, Caraco et al., 1997).

Analysis of DIN showed three major findings: 1) the concentration of DIN tends to decrease through the growing season, 2) the percentage of TN as DIN decreases through the growing season, and 3) the percent of TN as DIN is quite different among habitat types (Figure 6). The decrease in DIN concentration and percent of TN as DIN is likely the result of increases in nitrogen uptake, increased denitrification, and reduced export from tributary watersheds as discharge decreases through the growing season (Strauss, 2006; James et al., 2008; Houser and Richardson, 2010). The main channel percentage of TN as DIN was consistently higher in the main channel when compared to the backwaters (Figure 6). This trend was likely the result of reduced input of nitrogen to the backwaters from the main channel, high nitrogen uptake, and high denitrification within the backwaters (Strauss, 2006; James et al., 2008).

Data collected at the Lake Onalaska outlet in 2007 provides an illustration of how dynamic the UMR can be during a single growing season (Figures 7 and 8). Early in the growing season, a diatom bloom developed that resulted in reduced concentration of SRP (15 µg/L) and dissolved silica (1213 µg/L). High sinking rates are a common trait among diatoms, so dominance of diatoms during late spring and early summer when discharge, water velocity, and wind are high is a common pattern on the UMR (Huff, 1986). The concentration of DIN became extremely low as discharge dropped and appeared to become low enough (30 µg/L on 8/9) to result in nutrient limitation of a major mid-summer algal bloom. DIN concentration of < 50 µg/L has been suggested as a point where nutrient requirements among phytoplankton are no longer met and phytoplankton become nutrient-limited (Reynolds and Descy, 1996). Dissolved oxygen concentration dropped substantially following the major July algal bloom as discharge dropped below 10,000 cfs. During this period, it appears there was a substantial release of iron and aluminum-bound SRP and release of NH_x as a result of anoxia at the sediment-water interface (James, 1995). This sediment-derived release resulted in record values at the site for both parameters. Less than a month after these low flow phenomena, record rainfall descended on the area resulting in substantial loading of sediment and nutrients to the system.

The Upper Iowa River flows through a highly cultivated driftless landscape and is notable for the substantial sediment and nutrient load it delivers to Navigation Pool 9 of the UMR. The Upper Iowa sampling site (UI02.9M) is located 2.9 miles upstream of its confluence with the Mississippi and is an example of a site that was dropped altogether under the MSP and reinstated in 2007 through the APE: Restoration of Specific Monitoring Elements to the Long Term Monitoring Program. Observations from the Upper Iowa River in 2008 and 2009 provide an example of the inter-annual variability that can be observed within tributaries of the UMR during consecutive growing seasons (Table 3). During 2008, a June rainfall of historic proportions descended on the basin. A lesser, but significant rainfall event occurred in early-July that resulted in extremely high TSS and nutrient values. The high TSS and nutrient concentrations observed in early-July were likely elevated as a result of disturbance from the June storm one month earlier. Sampling indicated TSS exceeded 800 mg/L on both occasions, TP exceeded 1000 µg/L on both occasions, and TN exceeded 7000 µg/L on both occasions (Figure 9). The combined effect of these two rainfall events resulted in substantial loading of sediment and nutrients to Pool 9 of the UMR.

The 2009 growing season on the Upper Iowa River was characterized by below-average discharge and was a sharp contrast from conditions observed in 2008 (Figure 10). By late-August, discharge was below 400 cfs and SRP concentration was below the detection limit (< 1 µg/L). Research regarding nutrient limitation among phytoplankton suggests that nutrient requirements for phosphorus are not met when concentrations drop below 3 µg/L (Reynolds and Descy, 1996). This is consistent with our observations as a major late-August algal bloom appeared to end with the onset of SRP values below 1 µg/L. The observation of record nutrient loading in 2008 to apparent nutrient limitation the following growing season provides some insight into the range of conditions that can be observed in consecutive years within tributaries of the UMR.

Data collected from the main channel of Pool 8 during 2007-2009 provides some insight into the limnological dynamics that can be observed under a reduced discharge regime. In 2007, an early diatom bloom appeared to exert considerable control on SRP concentrations from mid-May to mid-June (Figure 12). Although SRP never did reach the 3 µg/L threshold, values of 5 and 9 µg/L were observed over this period. Actual thresholds for nutrient limitation can vary by waterbody and algal assemblage, resulting in higher values observed for nutrient-limited algal growth in some systems (Sterner and Grover, 1998). The nutrient ratios over this period (N:P ~19-24, DIN:SRP 50-225) suggest that there may have been some P limitation over this period. Although Reynolds (2006) noted that nutrient limitation rarely occurs when TN:TP ranges from 10 to 30, the observation of extremely high DIN:SRP values suggest this may have been a case when P limitation did occur within this range. By July, a substantial bloom of the N-fixing cyanobacterium, *Aphanizomenon* developed and persisted until the end of July. The bloom began as discharge dropped to ~10,000 cfs and DIN dropped to < 80 µg/L. Heterocysts are cells unique to cyanobacteria that allow them to fix atmospheric nitrogen (N_2). The ratio of heterocysts to vegetative cells in *Aphanizomenon* can range from 1:10,000 to 1:10. This ratio begins to increase quite rapidly as water column DIN drops below 350 µg/L and becomes particularly pronounced as DIN drops below 80 µg/L. The observation of a substantial *Aphanizomenon* bloom when DIN dropped below 80 µg/L suggests that the ability to fix N_2 allowed *Aphanizomenon* to gain a competitive advantage over this period (Reynolds, 2006). The concentration of SRP dropped from 93 µg/L prior to the *Aphanizomenon* bloom to 13 µg/L at the peak of this substantial bloom demonstrating the considerable control that blooms of this nature can exert on available P. The nutrient ratios (N:P 4.4-8.2, DIN:SRP 0.38-5.58) during this bloom were also very consistent with ratios commonly observed during cyanobacteria blooms (Smith, 1983). The ratio of N:P tends to decrease with increasing fertility and waters with TN:TP < 5 tend to more closely resemble the ratio observed in sewage rather than unpolluted waters (Sterner and Elser, 2002). The shift in TN:TP from early season values suggestive of P limitation to late season values suggestive of N limitation illustrates the dynamic nature of growing season limnology on the UMR.

The 2008 growing season was different than 2007 and 2009 in that above average discharge was observed early in the growing season (Figure 13). A substantial diatom bloom developed in late-May in which SRP dropped to 6 µg/L, TN:TP was ~40, and DIN:SRP was > 400 (Figure 14). All three of these values are suggestive of P limitation. Chlorophyll concentration was moderate following this bloom until a fairly substantial cyanobacteria bloom developed in mid-August when discharge (< 16,000 cfs), DIN (< 350 µg/L), TN:TP (~7), and DIN:SRP (~3) all dropped to values conducive to cyanobacterial dominance while SRP concentration remained high (> 100 µg/L). The shift from an early season diatom bloom to a late season cyanobacteria bloom demonstrates the response that the algal assemblage can have to changing water quality within a single growing season.

The 2009 hydrograph was characterized by low discharge and was very similar to the 2007 hydrograph (Figure 15). Despite similarities in discharge, substantial differences in chlorophyll concentration were observed between 2007 and 2009. There was a modest diatom bloom in late-May, but following this bloom, chlorophyll-*a* concentrations dropped to < 5 µg/L from mid-

June to mid-August (Figure 16). These low chlorophyll values were observed despite low TSS and ample SRP and DIN suggesting grazing pressure may have played a role in the reduction in algal biomass. Similar conditions in 2007 resulted in a dramatic *Aphanizomenon* bloom that persisted for over one month. If the adult *Dreissena* population can be inferred from veliger surveys conducted by the Wisconsin DNR, it appears that a substantial adult population of *Dreissena polymorpha* may have existed in Pool 8 by 2009 (Figure 17). It seems possible that this population may have exerted considerable control on phytoplankton biomass during this low flow period (Figure 16). The observations are consistent with observations on the Hudson River where a 17-fold decrease in algal density and reduction in cyanobacteria resulted following dreissenid invasion (Smith, et al., 1998). The concentration of dissolved oxygen was generally lower in 2009 than in 2007-2008 and was likely due to decreased phytoplankton biomass and high respiratory activity as a result of high adult zebra mussel biomass. This is consistent with observations of reduced DO concentration on the Hudson River following zebra mussel invasion due to an increase in respiratory activity (Caraco et al., 2000).

Our analysis of Lake Pepin data found no statistical difference in summer means of CHL, TP, or Secchi transparency between the two levels of sampling effort. However, mean differences were relatively large for CHL in 2008 and 2009. The additional sampling resulted in consistently higher mean concentrations of CHL and TP and a decrease in mean Secchi. The only exception was in 2007 when mean CHL was lower with the additional sampling. Seasonal variability in the three parameters was highest for CHL. Variability of the Lake Pepin data showed an increase for two of the three parameters when calculated with the additional monitoring effort.

Currently, there are two water quality impairments listed in Pool 4. A turbidity/total suspended solids impairment in the main channel above Lake Pepin and an eutrophication impairment in Lake Pepin. Total maximum daily load (TMDL) efforts are ongoing for these impairments and the water quality goals set for these waterbodies will be based on summer mean data. Minnesota lakes are typically assessed for eutrophication using the causative water quality parameter TP and response parameters CHL and Secchi (Heiskary and Wilson, 2008). If TP is in excess of set criteria along with one of the response variables then the lake is considered impaired and becomes listed as required by the federal Clean Water Act. Total phosphorus concentrations are relatively high in Lake Pepin and are likely to be above the forthcoming criteria being established for the lake. Consequently, summer means of CHL and Secchi become critical in the assessment of whether Lake Pepin is achieving water quality standards.

Chlorophyll *a* was the more variable of the two response parameters in Lake Pepin and our results are consistent with other studies that examined variability and found CHL to be more variable than TP or Secchi (Ferris and Lehman, 2008; Knowlton and Jones, 2006a; Knowlton and Jones, 2006b and references therein). Knowlton and Jones (2006a) analyzed data from Missouri reservoirs to examine temporal variability of trophic state parameters. Their simulations demonstrated that increased sample frequency during the summer increased the precision of the data and decreased the number of years to detect a trend in trophic state parameters. Because of the high variability found in CHL, this parameter benefitted the most from increased sampling frequency by increasing precision around the means.

The pattern of higher TP and CHL and lower Secchi transparency we found with the additional sampling is likely due to the fact that the additional data was collected in late summer. Chlorophyll *a* and TP concentrations are generally higher in late summer in the UMR (Houser and Richardson, 2010). Consequently, when the additional monitoring data collected in late summer is included in the calculation of the summer mean the above patterns were observed. This bias could make the difference in whether or not the lake is meeting established water quality goals. For example, the difference in mean values for CHL between the levels of effort was large enough that a summer mean could easily fall on either side of the proposed criteria of a summer mean of 32µg/L CHL. Furthermore, the original impairment for Lake Pepin was listed based on LTRM data collected at the bi-weekly frequency and future assessment of the lake will rely on the LTRM monitoring as the sole source of data. Maintaining biweekly sampling throughout the summer months appears critical to providing consistent, high-quality data for the management of Lake Pepin. Certainly sample size and frequency of monitoring may depend on the question to be answered. However, Lovett et al. (2007) point out environmental monitoring data can be used for multiple purposes. Use of the LTRM data for TMDL efforts is a great example of an unforeseen use not originally intended in the design of the program. In addition use of LTRM data for TMDLs fits in the mission of the LTRM by providing decision makers with sound scientific information.

The information presented in this report illustrates limnological dynamics during the growing season and the importance of biweekly sampling to maintain a robust program that is capable of capturing information driving ecological change. We have shown through our analysis that increased sampling frequency often increased the variability of the dataset and allowed the field stations to collect valuable data that would have been missed under the MSP sampling effort. The MSP monitoring schedule reduces sampling frequency during a period (July-September) when stressors on biota can be high (e.g., low DO), limnological variability is high, and recreational use by the public is at a maximum (Kramer, 1987; Houser and Richardson, 2010, Penaloza, 1987). Discontinued sites and decreased sampling frequency at the remaining fixed-sites under the MSP design have seriously diminished an important component of the mixed-model sampling design. The LTRM has been a major provider of fixed-site water quality data on the Mississippi River, representing 46% of the data in a recent water quality assessment (UMRCC, 2002), emphasizing the importance of that information to river managers. In addition, LTRM water quality data has been instrumental in TMDL efforts on the UMR and has been used to establish water quality goals and will continue to be heavily relied upon for future assessments. Lovett et al. (2007) suggest the cost of monitoring is minimal relative to the value of the resources it protects and the policy it informs. We argue that this is exactly the case for the restoration of bi-weekly sampling and reinstatement of historical sampling sites.

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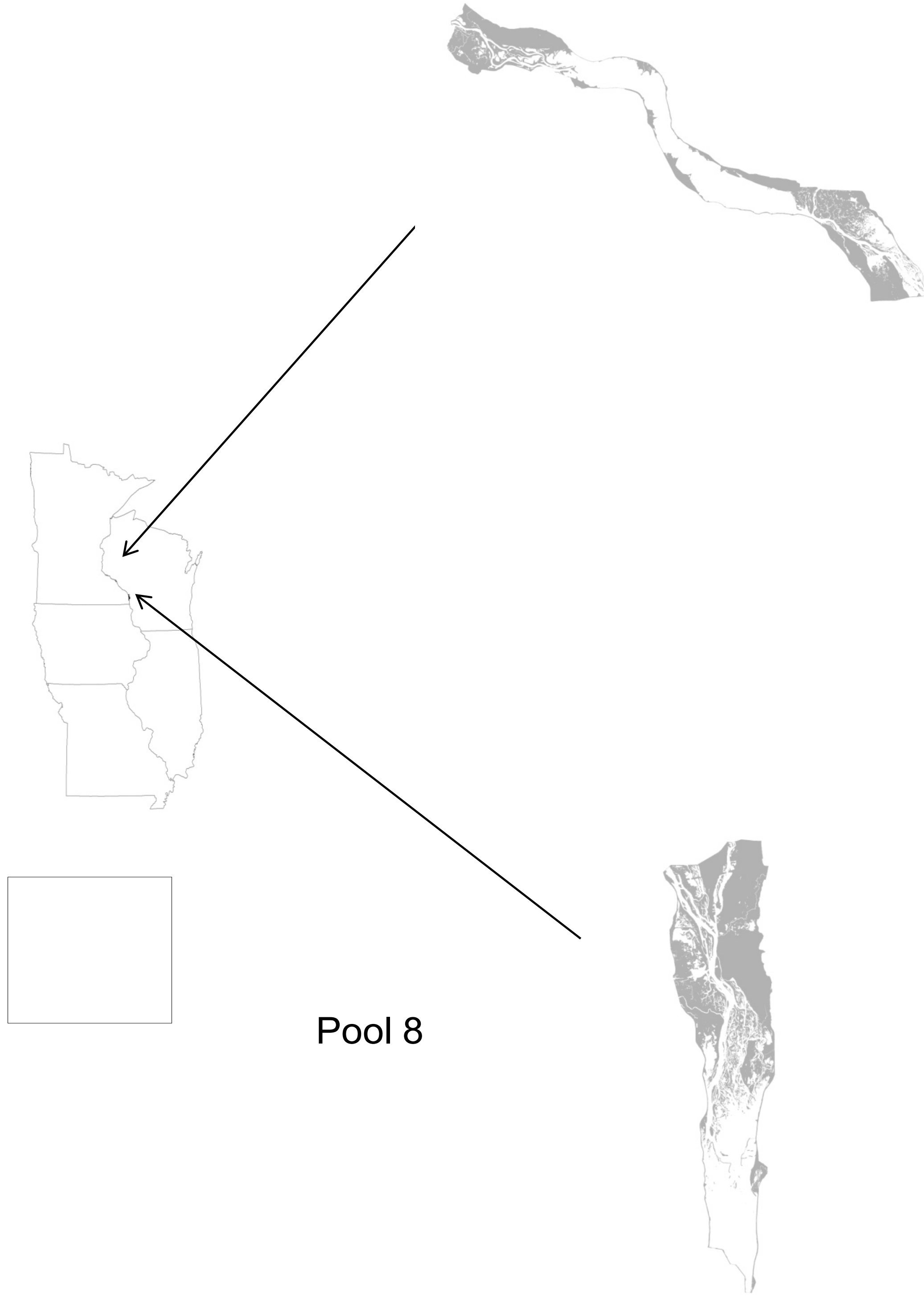

Figure1. Location of Pools 4 and 8 of the Upper Mississippi River.

Light editorial correction by Jennifer Sauer, USGS

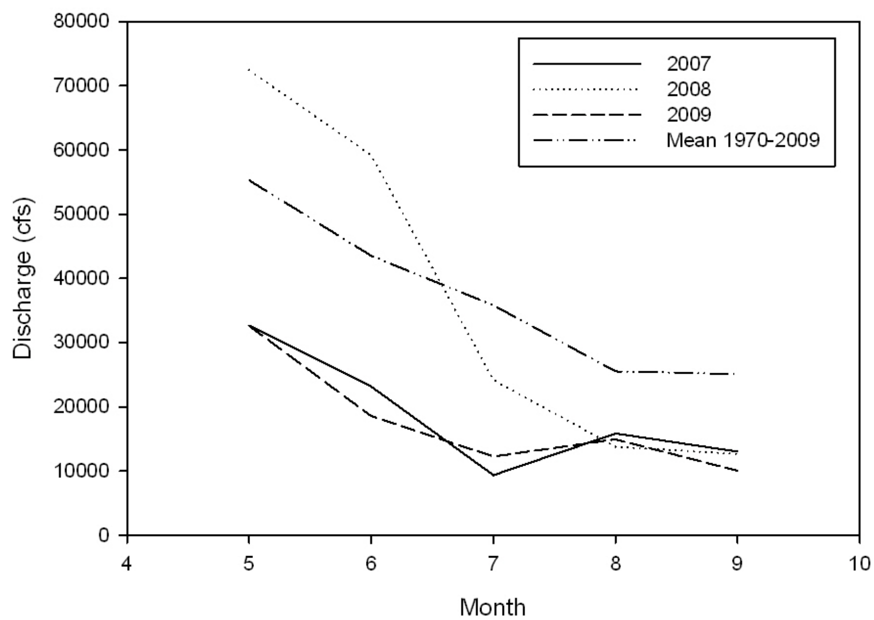

Figure 2. Monthly mean discharge at Winona, MN (cfs) for 2007-2009 compared to long term mean (1970-2009).

Light editorial correction by Jennifer Sauer, USGS

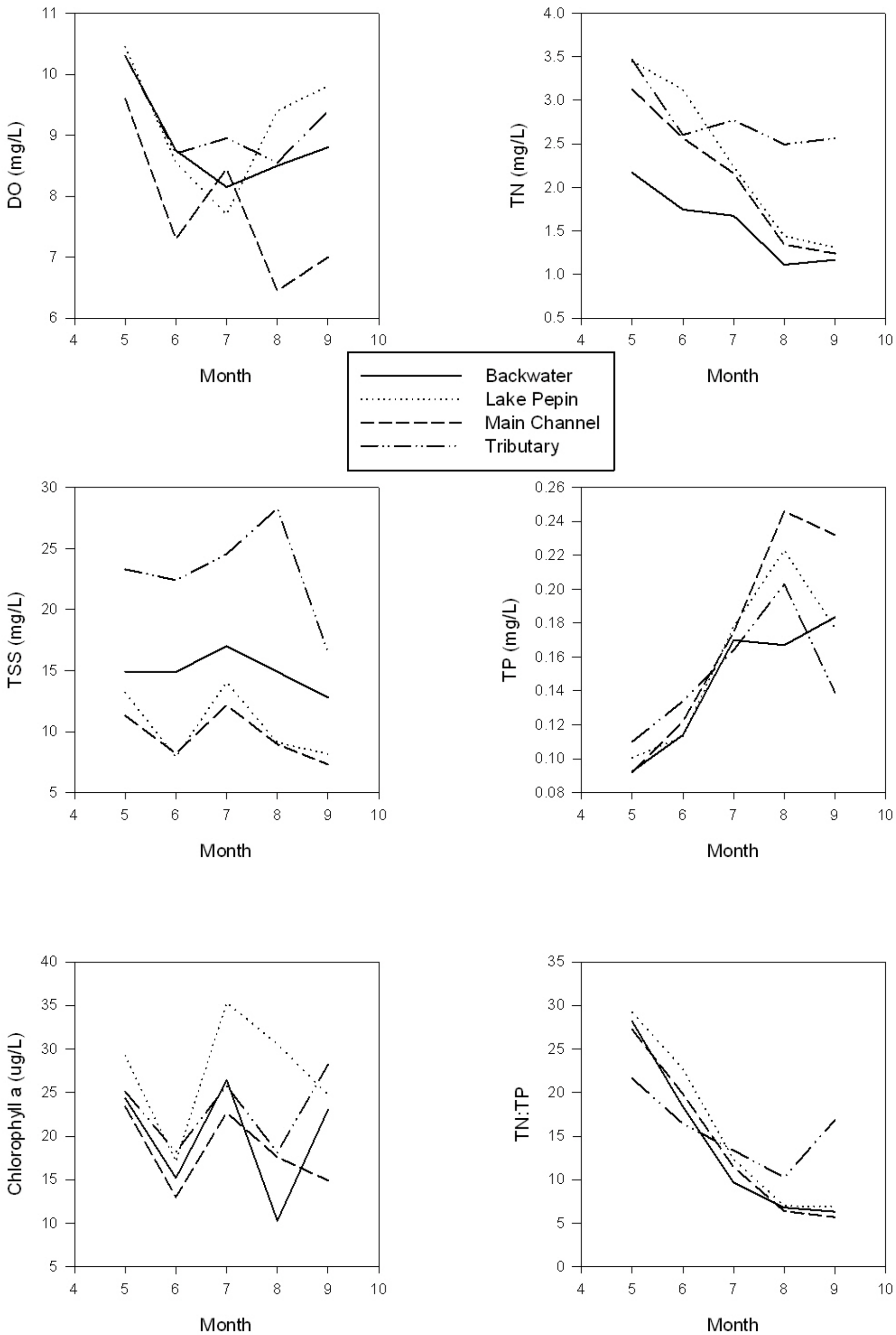

Figure 3. Pool 4 monthly median water quality values by habitat class for 2007-2009.

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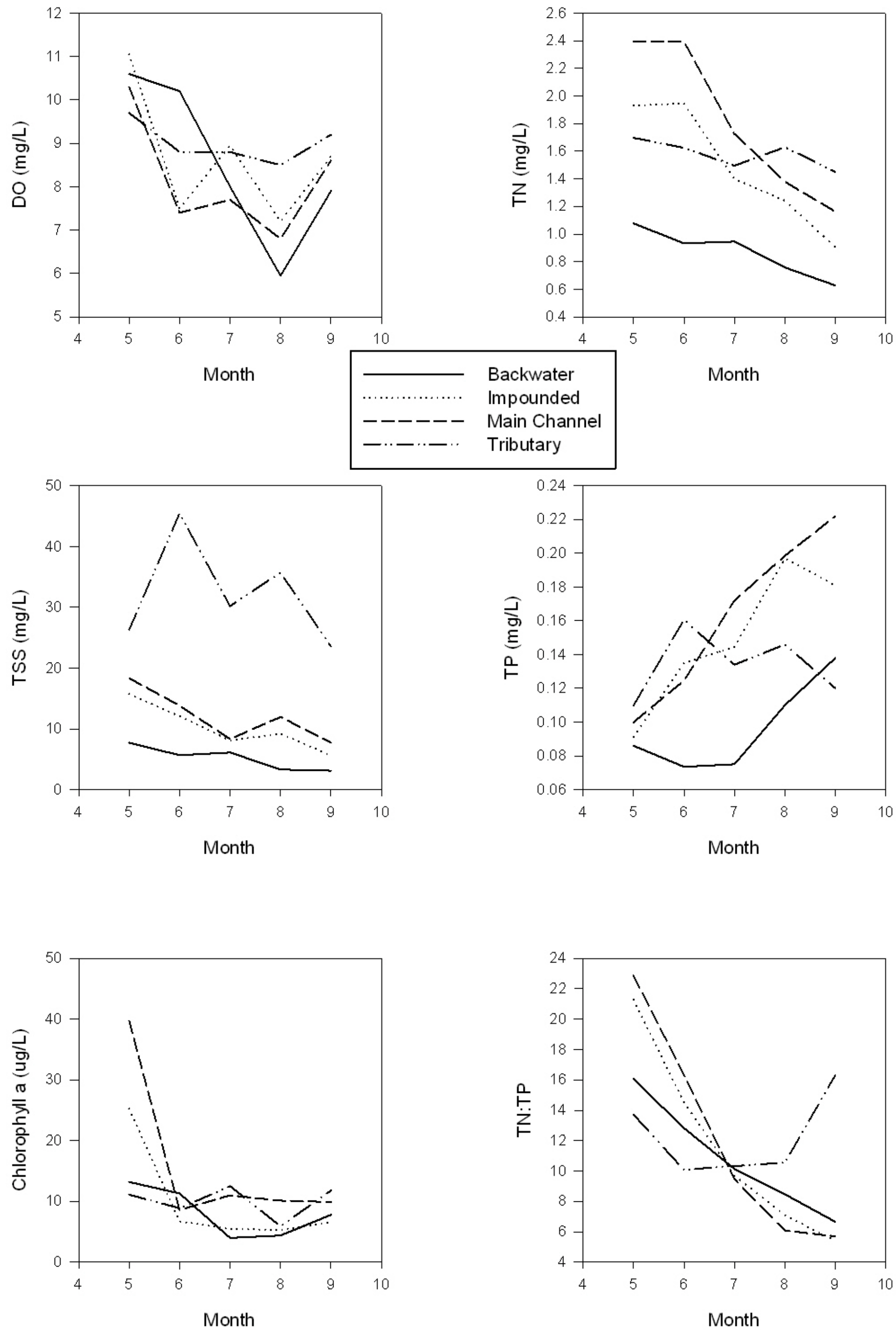

Figure 4. Pool 8 monthly median water quality values by habitat class for 2007-2009.

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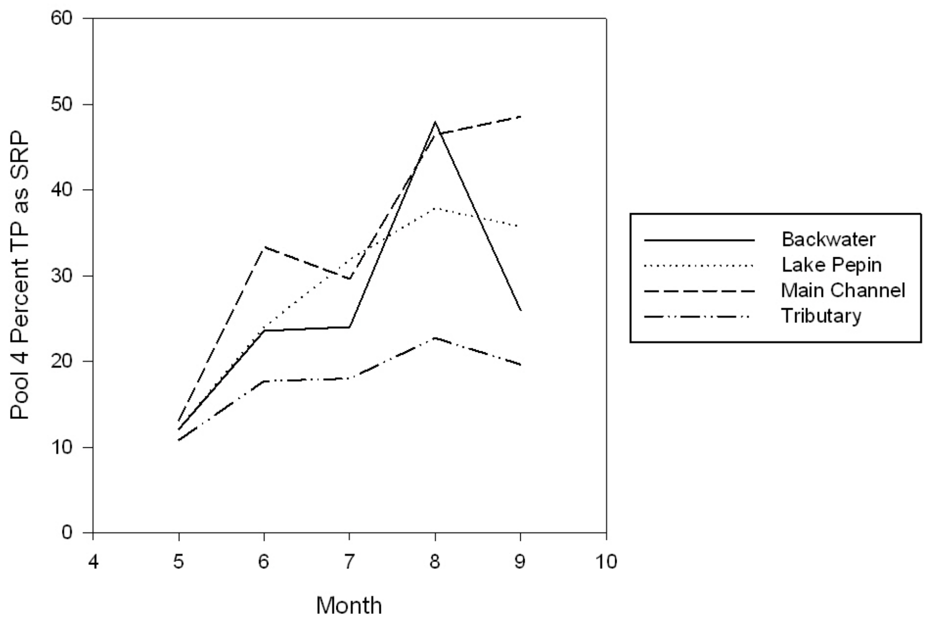

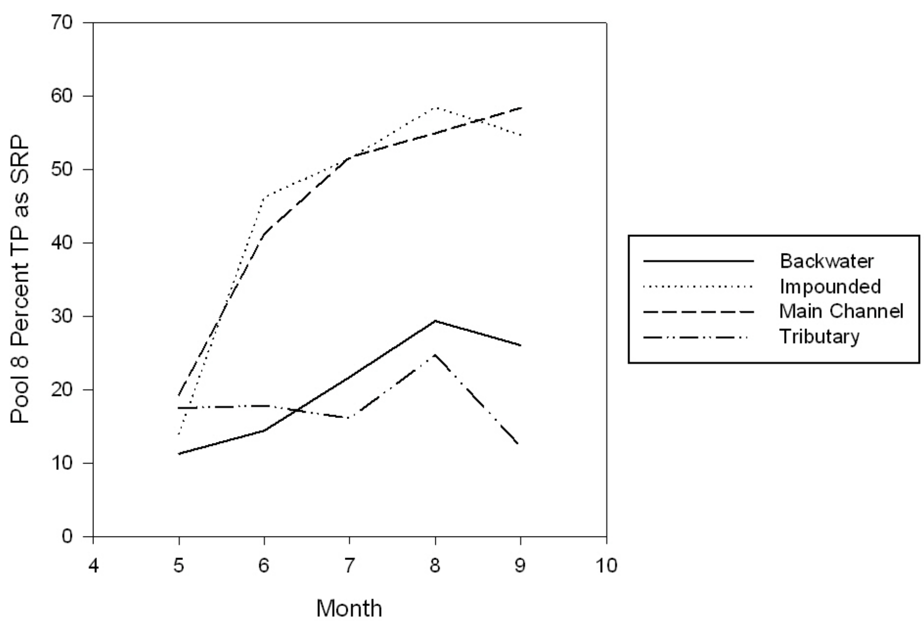

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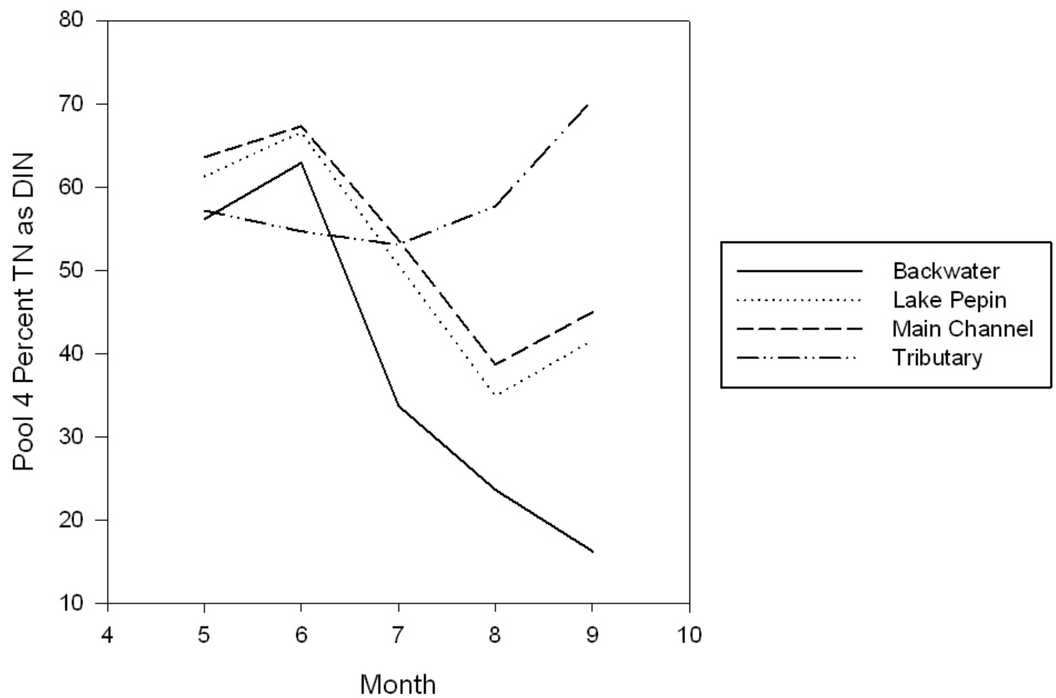
Figure 5. Percentage of total phosphorus comprised of soluble reactive phosphorus for Pools 4 and 8 from 2007-2009.



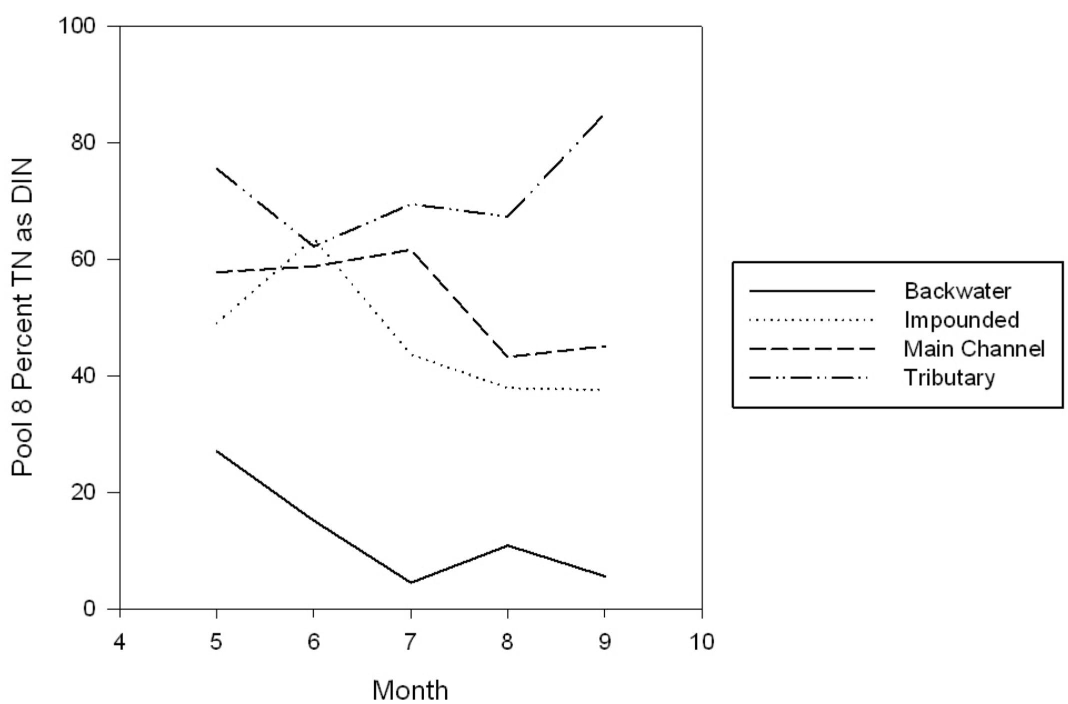


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Figure 6. Percentage of total nitrogen comprised of dissolved inorganic nitrogen for Pools 4 and 8 from 2007-2009.

Table 1. Field Station 1 coefficient of variation with the MSP and biweekly sampling frequencies.

Habitat Class	TN_{MSP}	$TN_{BIWEEKLY}$	NOX_{MSP}	$NOX_{BIWEEKLY}$
BWC	57.95	58.44	102.47	109.82
Lake Pepin	48.86	50.07	82.56	89.50
MC	50.13	50.61	80.04	84.80
TRIB	55.35	53.88	84.02	82.85

Habitat Class	NHX_{MSP}	$NHX_{BIWEEKLY}$	TP_{MSP}	$TP_{BIWEEKLY}$
BWC	102.93	118.01	38.69	48.74
Lake Pepin	106.63	110.49	52.31	54.64
MC	93.82	99.04	47.34	52.74
TRIB	121.61	169.89	46.93	52.66

Habitat Class	SRP_{MSP}	$SRP_{BIWEEKLY}$	TSS_{MSP}	$TSS_{BIWEEKLY}$
BWC	83.28	99.70	84.06	84.17
Lake Pepin	105.30	119.61	53.80	49.47
MC	89.75	106.04	90.93	85.25
TRIB	73.29	81.19	75.70	81.66

Habitat Class	$CHLF_{MSP}$	$CHLF_{BIWEEKLY}$
BWC	73.53	80.59
Lake Pepin	57.65	61.39
MC	76.41	84.57
TRIB	97.40	96.41

Table 2. Field Station 2 coefficient of variation with the MSP and biweekly sampling frequencies.

Habitat Class	TN_{MSP}	$TN_{BIWEEKLY}$	NOX_{MSP}	$NOX_{BIWEEKLY}$
BWC	50.42	54.12	181.67	177.49
IMP	50.03	50.65	88.88	92.95
MC	44.15	44.52	76.83	80.27
TRIB	76.78	75.24	105.41	103.97

Habitat Class	NHX_{MSP}	$NHX_{BIWEEKLY}$	TP_{MSP}	$TP_{BIWEEKLY}$
BWC	101.44	97.22	61.23	108.94
IMP	83.75	78.49	41.84	43.26
MC	98.40	96.99	33.77	40.95
TRIB	108.78	109.73	115.24	108.26

Habitat Class	SRP_{MSP}	$SRP_{BIWEEKLY}$	TSS_{MSP}	$TSS_{BIWEEKLY}$
BWC	98.61	100.41	84.56	245.03
IMP	91.61	81.87	90.66	87.17
MC	78.65	69.26	62.16	109.59
TRIB	84.01	90.40	263.87	252.77

Habitat Class	$CHLF_{MSP}$	$CHLF_{BIWEEKLY}$
BWC	104.19	111.35
IMP	110.58	110.05
MC	114.69	111.53
TRIB	120.88	123.32

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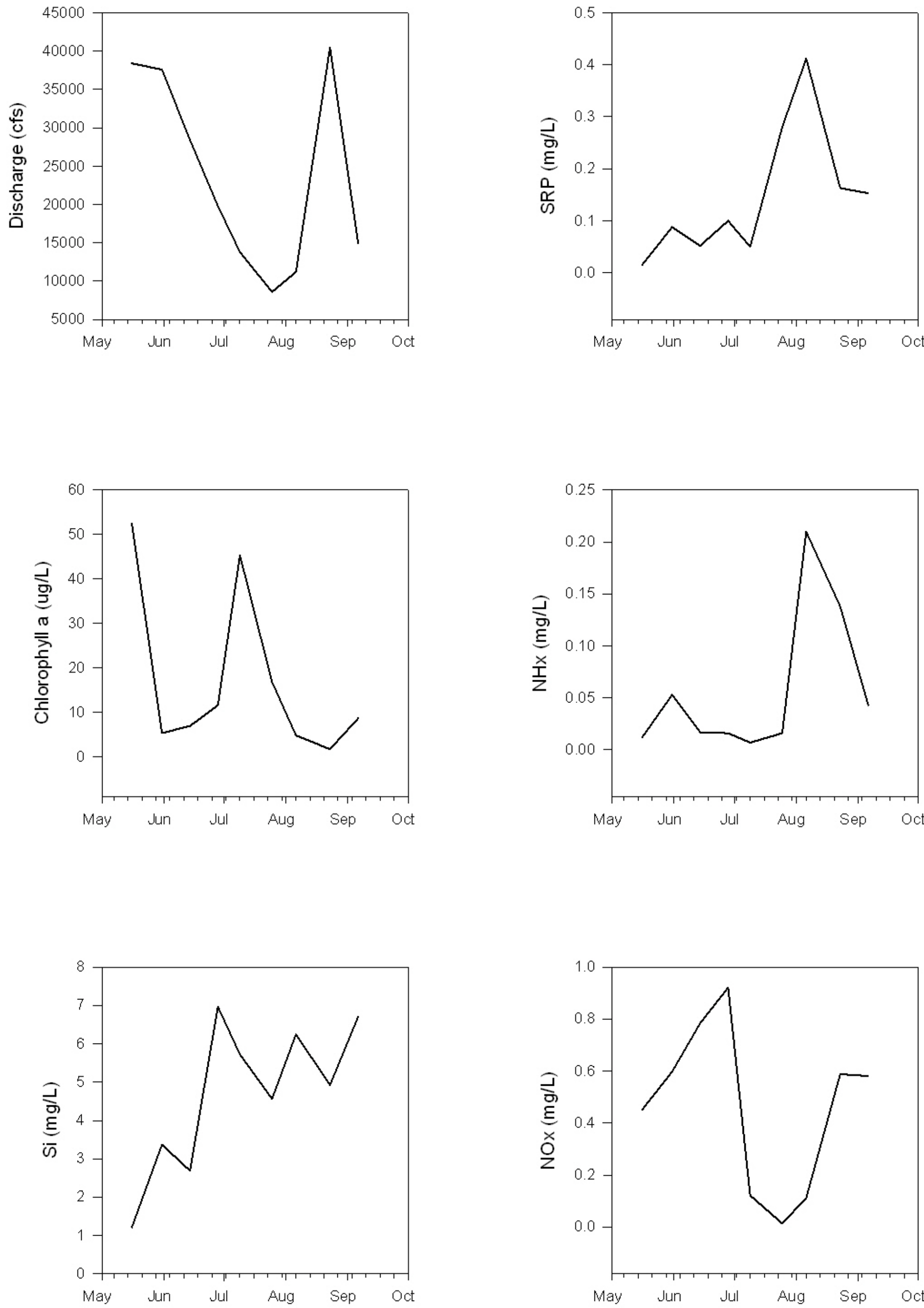

Figure 7. Discharge and water quality values collected at the French Island Spillway (M702.2T) during the 2007 growing season.

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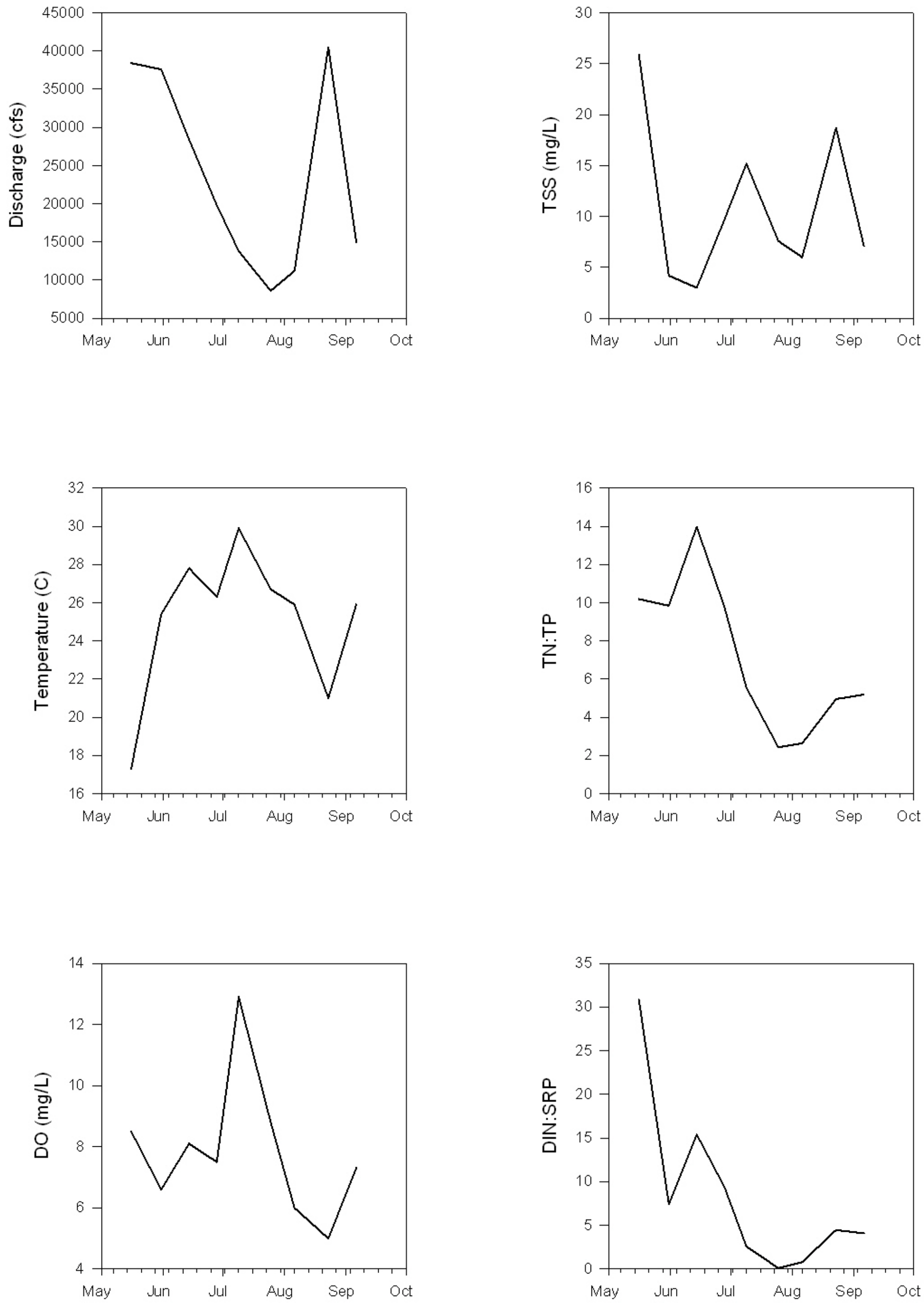

Figure 8. Discharge, water quality and nutrient ratio values collected at the French Island Spillway (M702.2T) during the 2007 growing season.

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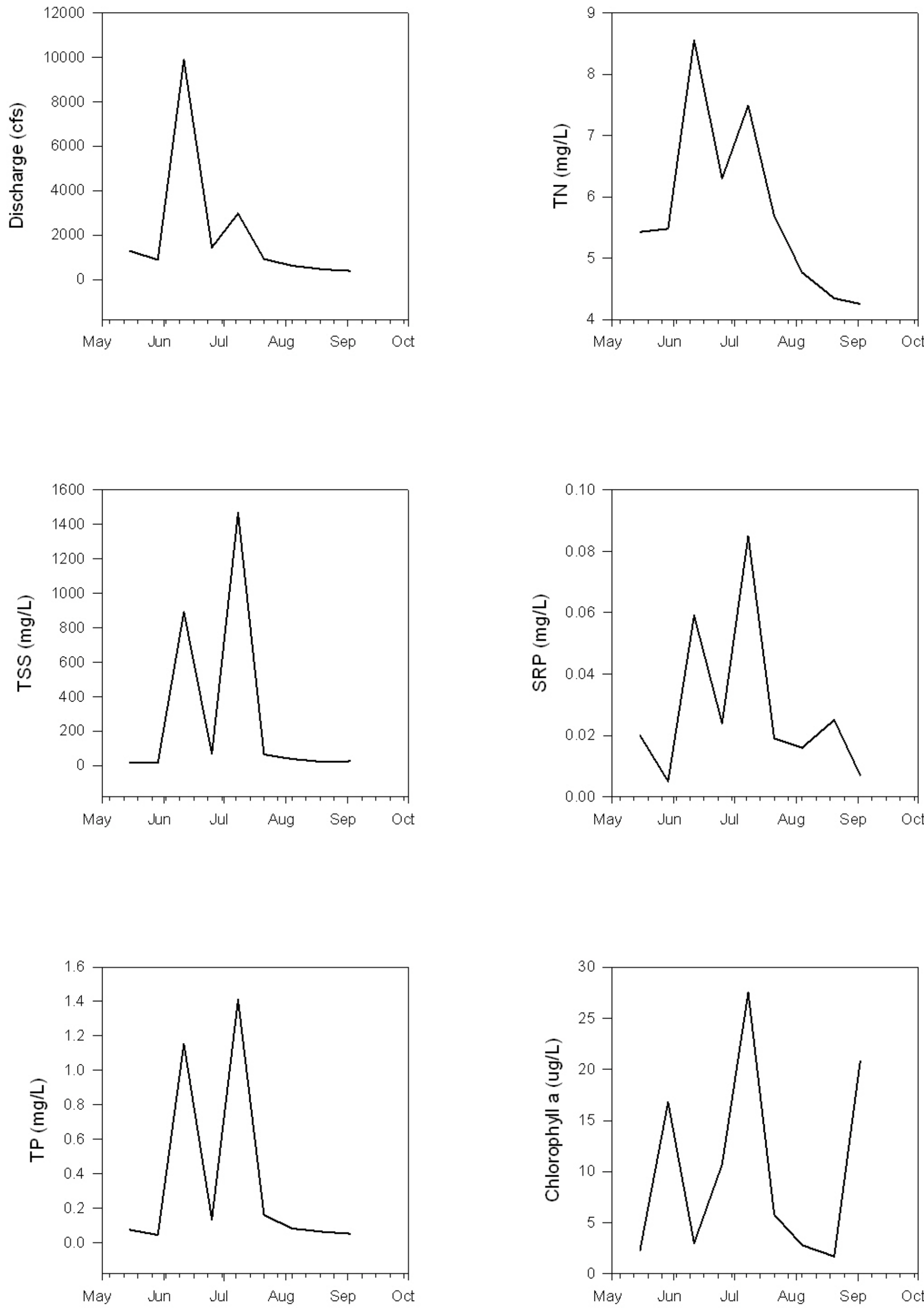

Figure 9. Discharge and water quality values collected at the Upper Iowa River (UI02.9M) during the 2008 growing season.

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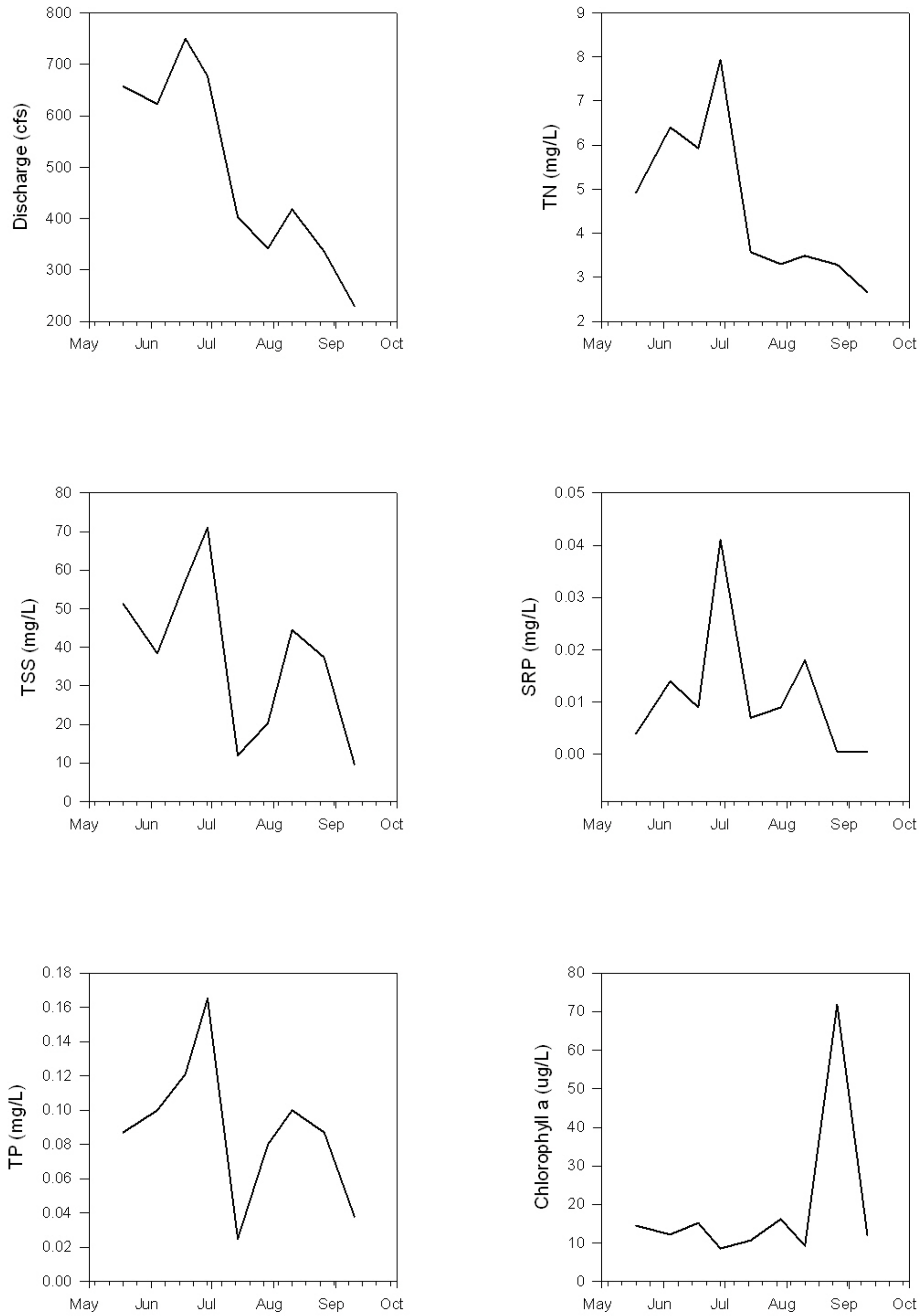

Figure 10. Discharge and water quality values collected at the Upper Iowa River (UI02.9M) during the 2009 growing season.

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Table 3. Maximum, minimum and percent differences observed for discharge and selected water quality parameters from the Upper Iowa River (UI02.9M) during the 2008 and 2009

Parameter				
Discharge (cfs)	MAX	9890	Date	6/11/2008
	MIN	230	Date	9/10/2009
	% difference	4200		
TSS (mg/L)	MAX	1466	Date	7/8/2008
	MIN	9.7	Date	9/10/2009
	% difference	15013		
TP (mg/L)	MAX	1.41	Date	7/8/2008
	MIN	0.025	Date	7/14/2009
	% difference	5540		
TN (mg/L)	MAX	8.55	Date	6/11/2008
	MIN	2.66	Date	9/10/2009
	% difference	221		
SRP (mg/L)	MAX	0.085	Date	7/8/2008
	MIN	< 0.001	Date	8/26/2009
	% difference	16900		
Chla (µg/L)	MAX	71.8	Date	8/26/2009
	MIN	1.7	Date	8/20/2008
	% difference	4124		

growing seasons.

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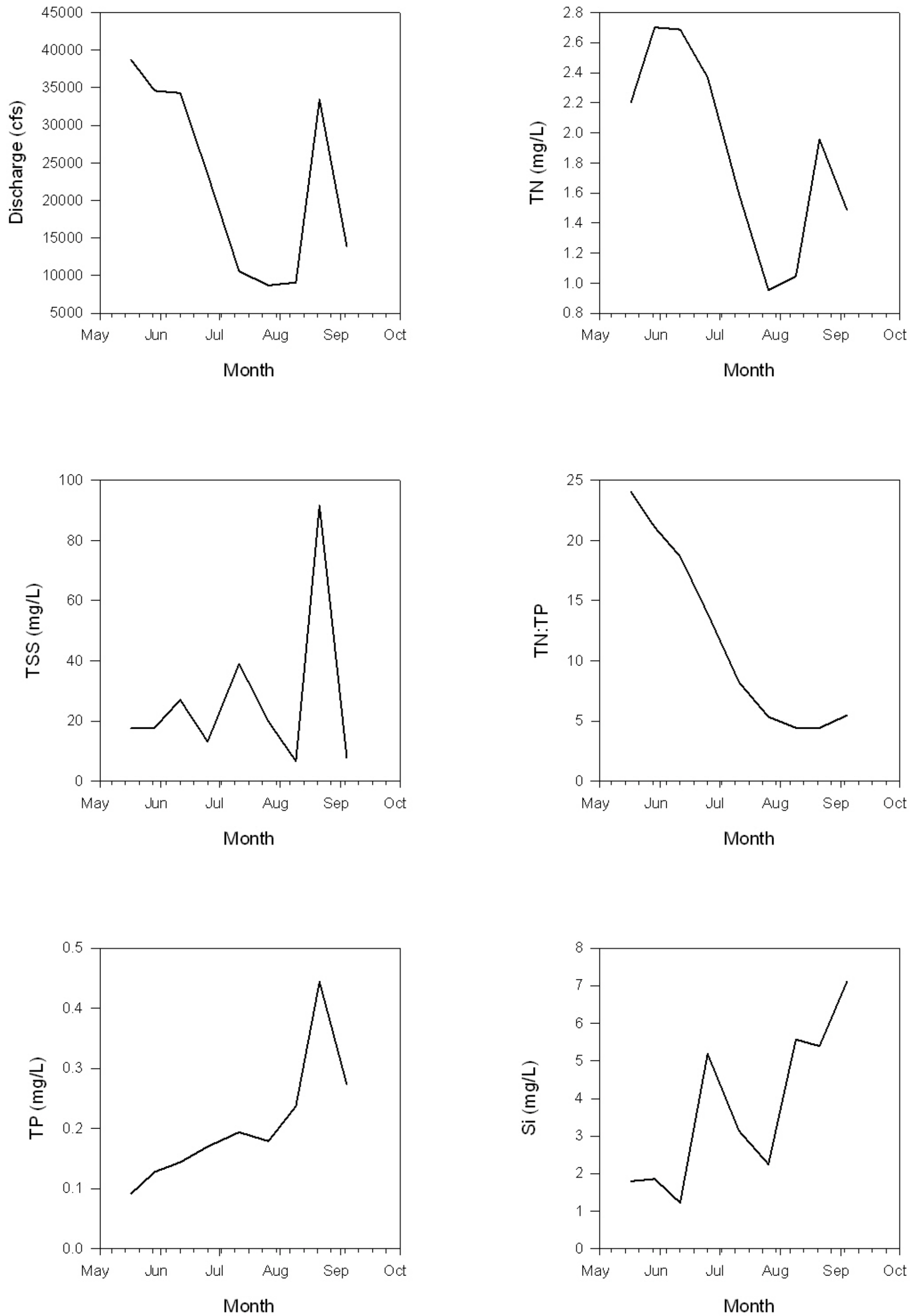

Figure 11. Discharge and water quality values collected from the main channel of Pool 8 (M701.1D) during the 2007 growing season.

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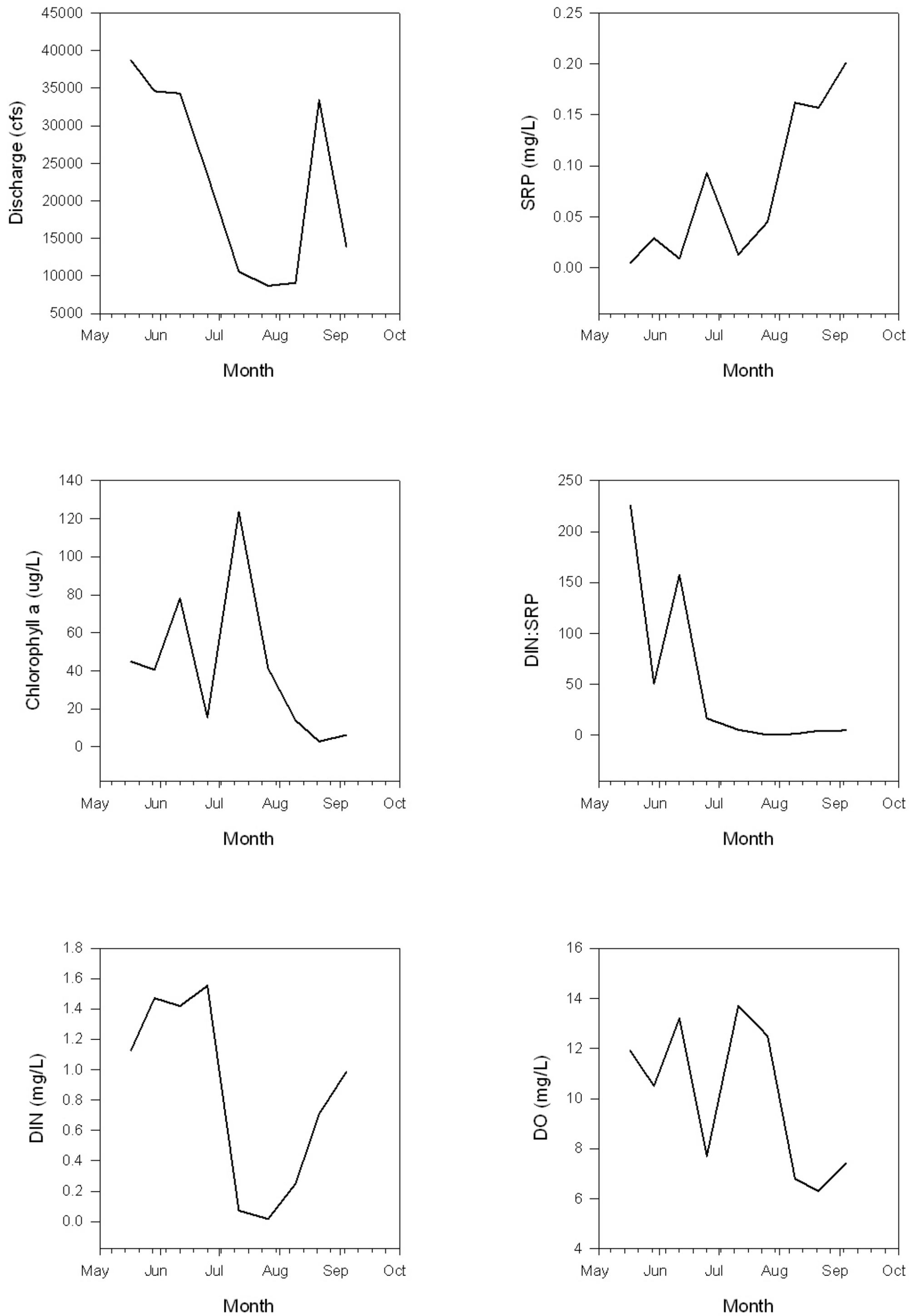

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Figure 12. Discharge and water quality values collected from the main channel of Pool 8 (M701.1D) during the 2007 growing season.

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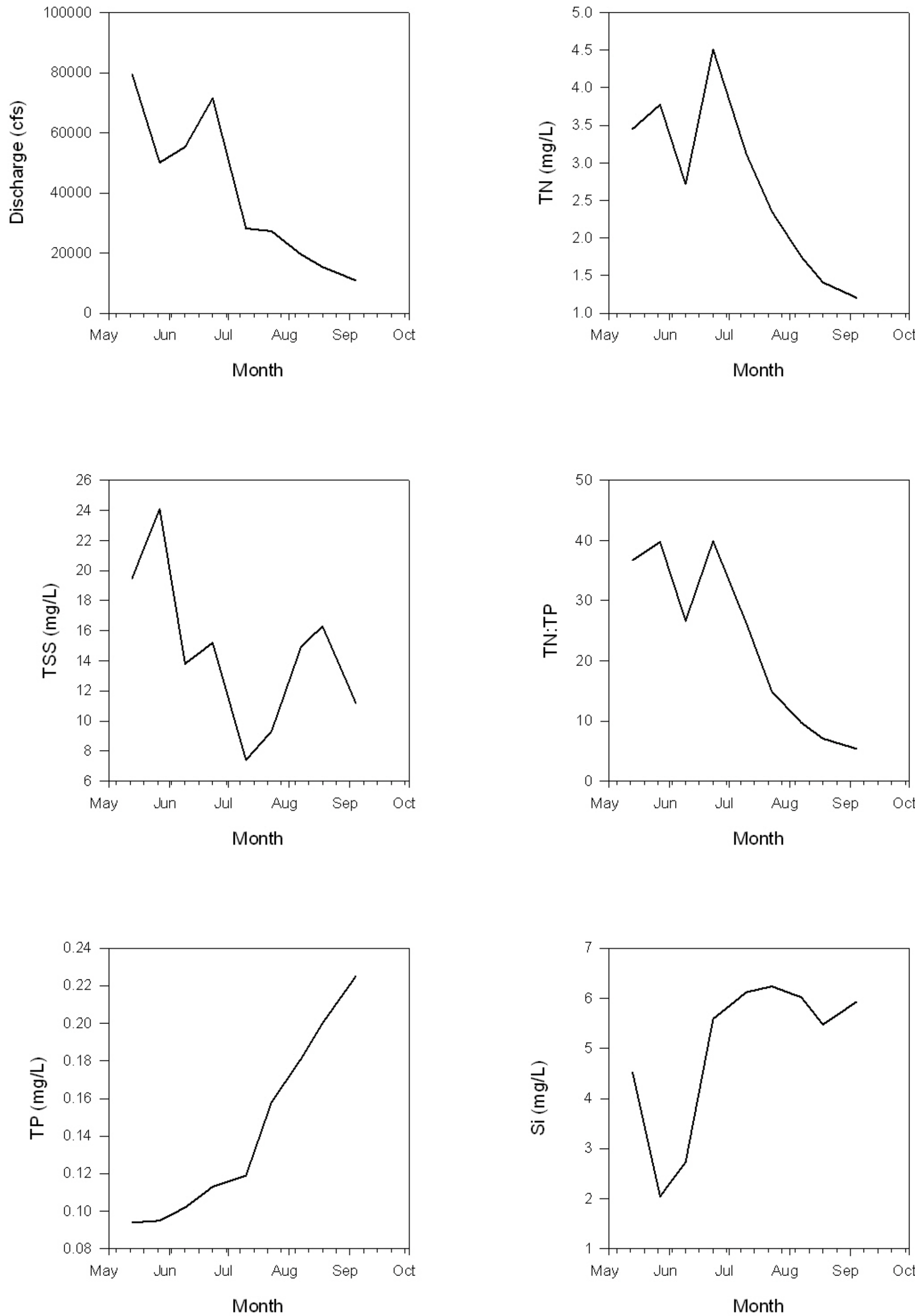

Figure 13. Discharge and water quality values collected from the main channel of Pool 8 (M701.1D) during the 2008 growing season.

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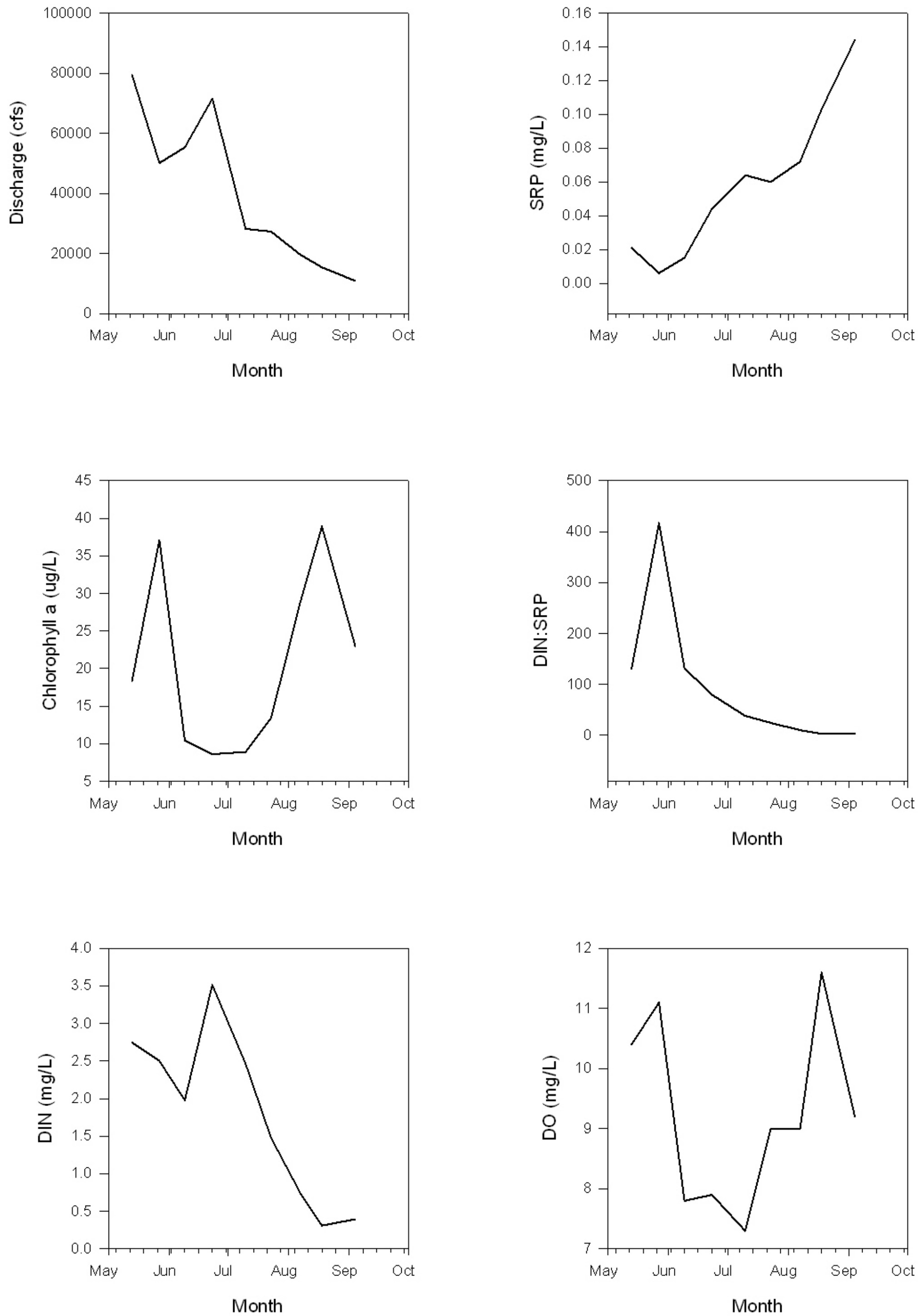

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Figure 14. Discharge and water quality values collected from the main channel of Pool 8 (M701.1D) during the 2008 growing season.

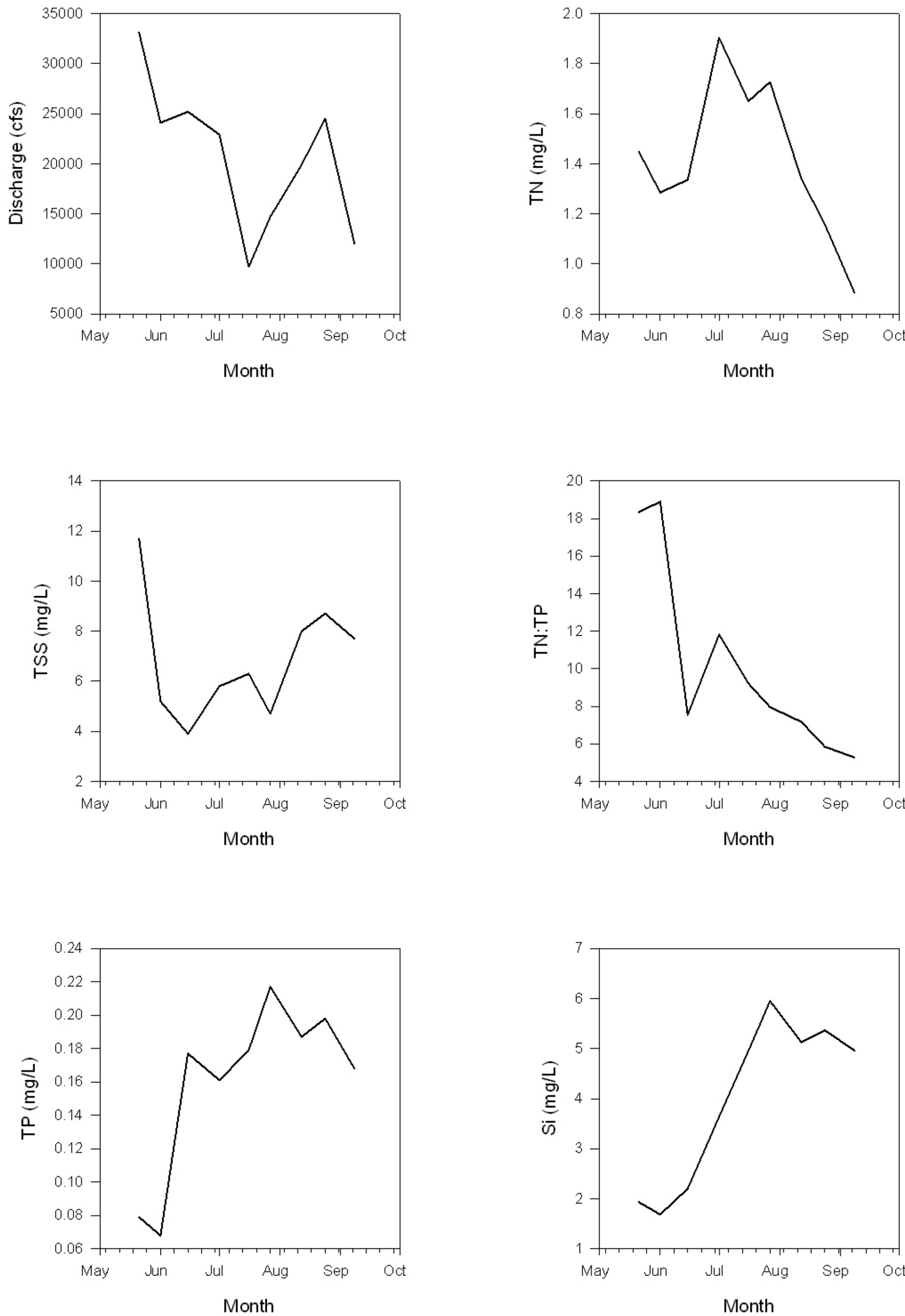

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Figure 15. Discharge and water quality values collected from the main channel of Pool 8 (M701.1D) during the 2009 growing season.

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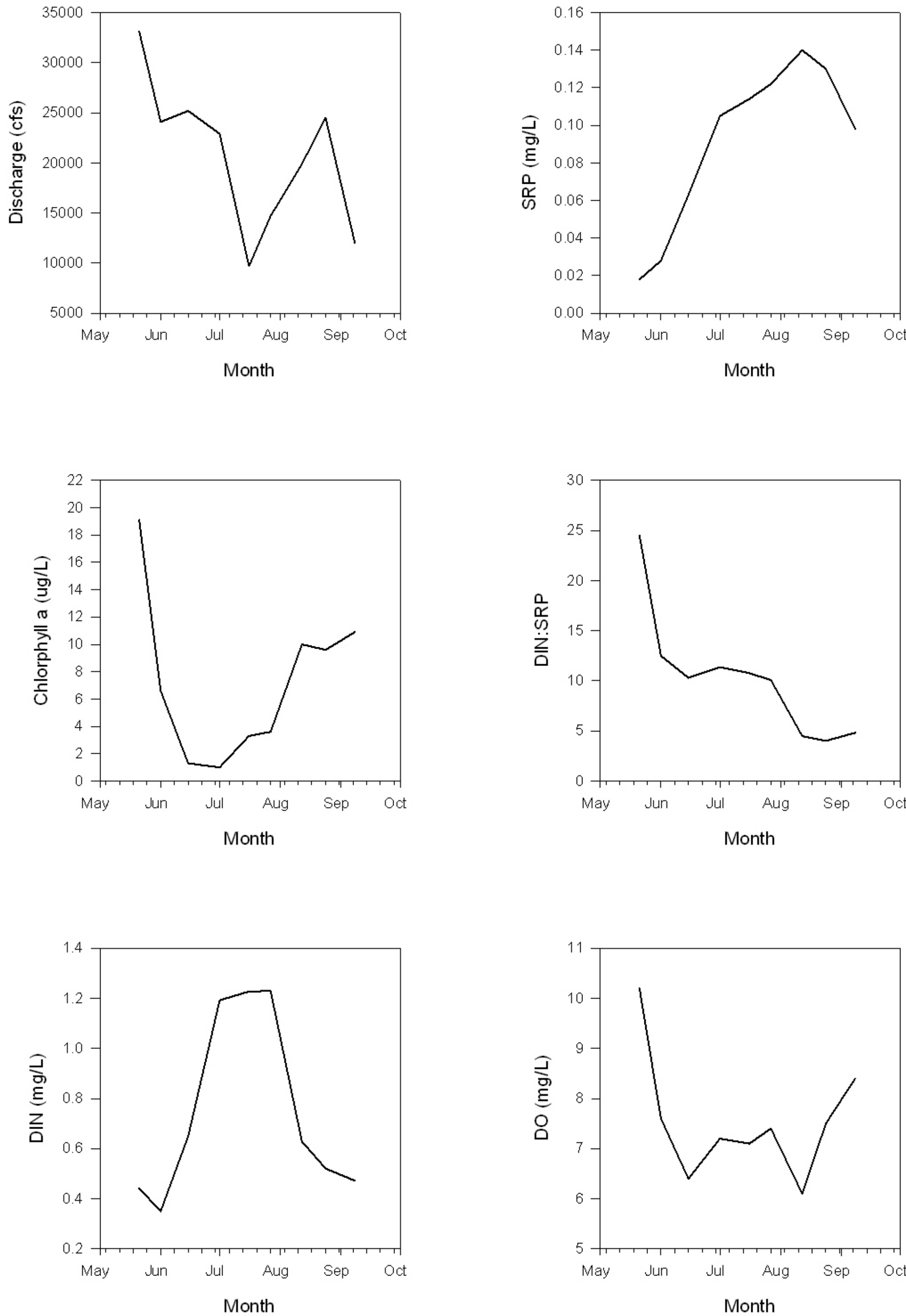

Figure 16. Discharge and water quality values collected from the main channel of Pool 8 (M701.1D) during the 2009 growing season.

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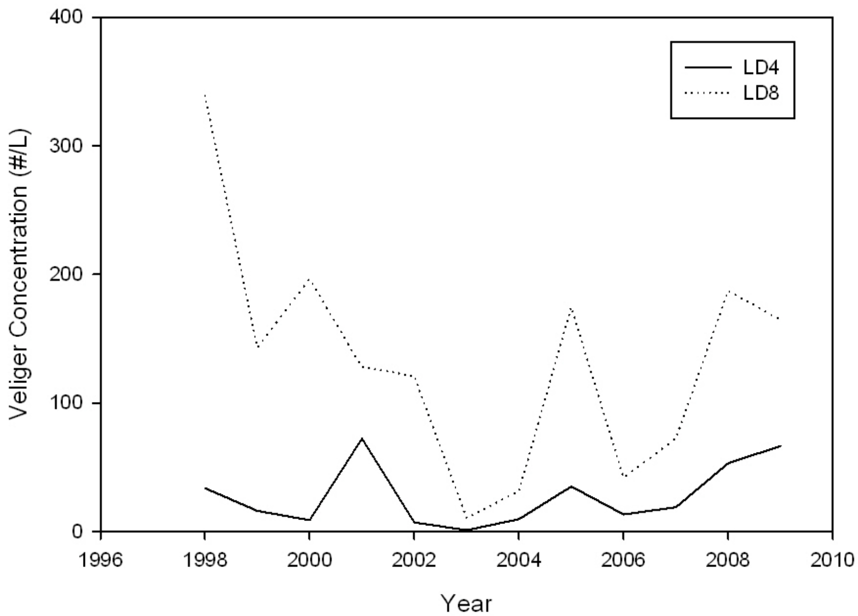

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Figure 17. Mean zebra mussel veliger concentration at Lock and Dam 4 and 8 during July-September sampling from 1998-2009.

Table 4. Mean summer values in Lake Pepin with and without additional monitoring effort and percent difference between means.

Year	CHL[a] *without*	CHL[a] *with*	% diff.	TP[b] *without*	TP[b] *with*	% diff.	Secchi[c] *without*	Secchi[c] *with*	% diff.
2007	38.9	38.5	-1.0	0.22	0.26	15.4	79.8	76.4	-4.2
2008	21.1	30.3	30.4	0.17	0.18	5.5	77.4	74.6	-3.6
2009	32.8	40.3	18.6	0.14	0.15	6.6	94.0	86.9	-7.6

[a] Chlorophyll *a* (μg/L)
[b] Total phosphorus (mg/L)
[c] Secchi transparency (cm)

Table 5. Coefficient of variation with and without additional monitoring effort in Lake Pepin from 2007 - 2009.

CHL[a] *without*	CHL[a] *with*	TP[b] *without*	TP[b] *with*	Secchi[c] *without*	Secchi[c] *with*
58.0	62.5	49.1	50.4	40.3	39.4

[a] Chlorophyll *a* (μg/L)
[b] Total phosphorus (mg/L)
[c] Secchi transparency (cm)

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