Development of Phosphorus TMDL for Little Rock Lake, Minnesota

prepared for

Benton Soil and Water Conservation District
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by

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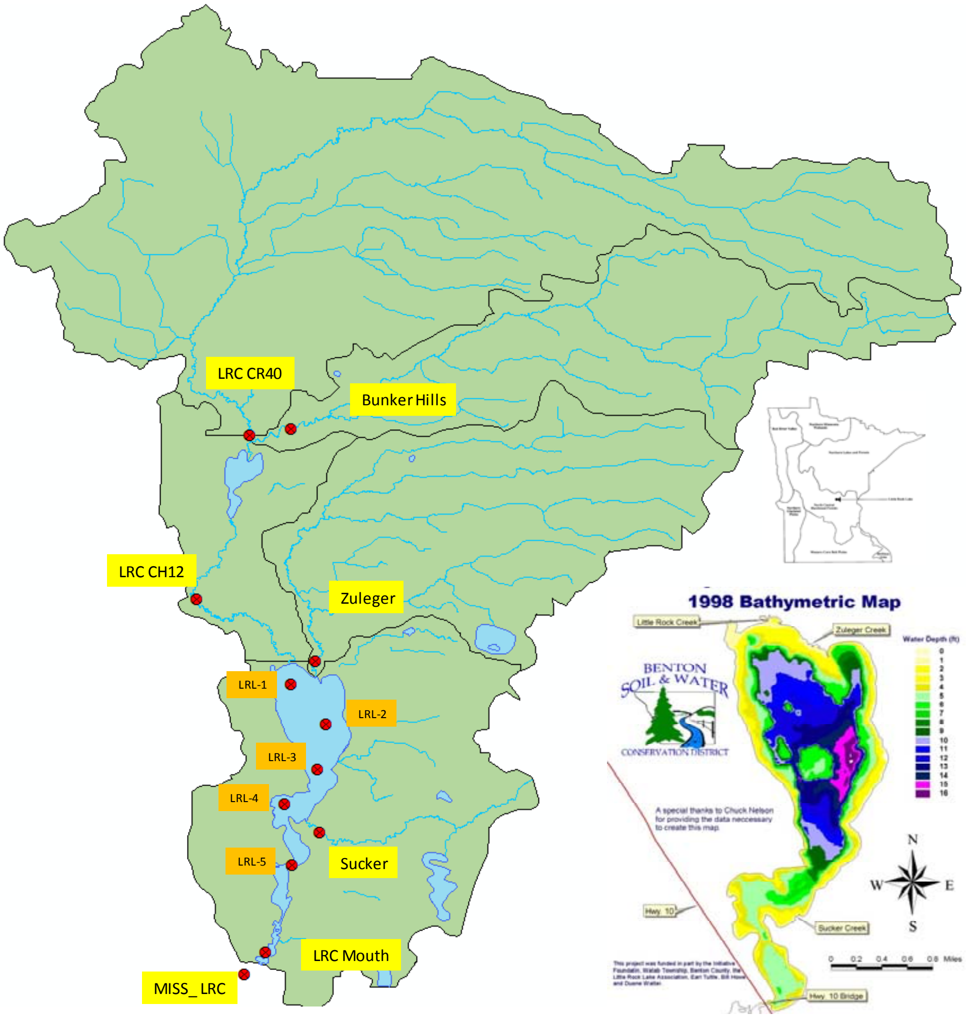
Units

Units	Definition	Conversions
hm^3	cubic hectometer	10^6 m^3 or 810 acre-feet
km^2	square kilometer	0.39 mi^2 or 247 acres
ppb	parts per billion	1 microgram per liter
m	meter	3.28 feet
kg	kilogram	2.2 pounds
mt	metric ton	1000 kg or 2200 lbs
AU	animal unit	1000 lbs live animal weight ~ 1 dairy cow

Introduction

Little Rock Lake (LRL, Figure 1) is a shallow hyper-eutrophic impoundment located in the Minnesota's Central Hardwood Forest (CHF) Ecoregion. It has a surface area of 5.1 km^2, mean depth of 2.4 meters, and total watershed area of 268 km^2. Major watersheds include Little Rock Creek (LRC, 67%), Zuleger Creek (18%), Sucker Creek (4%), local drainage (lakeshed, 9%), and lake (2%). Land uses include cropland (48%), grass/pasture (14%), urban (8%), woodland (15%), wetland (13%), and water (2%). The watershed contains 106 feedlots and 25 to 37 Thousand Animal Units (1 AU = 1000 lbs live animal weight ~ 1 daily cow) consisting of 26% diary cattle, 12% beef cattle, 11% swine, and 51% poultry. There are approximately 300 residences around the shoreline. Considerable erosion in the watershed is indicated by sand deposits in stream channels and at points of discharge into the lake. BSWCD (2009) provides detailed information on the watershed characteristics that impact flow, ecological habitat, nutrient loads.

Figure 1 Little Rock Lake & Watershed Monitoring Sites in 2008



Originally a wetland, the lake basin was formed in 1911 when a dam was constructed on the Mississippi River downstream of the Little Rock Creek (LRC) outlet. Water levels were further raised in 1934 and Little Rock Lake (LRL) evolved from a vegetated marsh to turbid impoundment (Ford et al, 2003; Garrison & LaLiberte, 2009). Major flooding events on the Mississippi River and dam operation have increased both the mean and the variability of lake water levels, although typical seasonal and year-to-year variations in water level are driven primarily by runoff from the LRL watershed (Ford et al., 2003). Shoreline areas are subject to erosion as a consequence of variability in water levels, wind-driven currents, and local runoff.

While it supports an abundant fishery, LRL has extremely high nutrient (phosphorus, P) concentrations that support severe algal blooms (Figure 2). As a result, the lake does not meet nutrient water quality standards established by the Minnesota Pollution Control Agency to support its designated beneficial uses, particularly with respect to aesthetics and recreation (Heiskary & Wilson, 2005; 2008), as listed in the MPCA water quality standards MN Rule CH 7050: "Class 2b, aquatic life use". Toxic bluegreen algal blooms (Figure 2, lower left), anaerobic conditions, and noxious odors resulting from atmospheric releases of hydrogen sulfide were observed in 2007 (Lindon et al, 2007). These conditions were associated with extremely high phosphorus concentrations in spring runoff (>500 ppb) and a relatively dry and warm summer.

Figure 2 LRL Photos

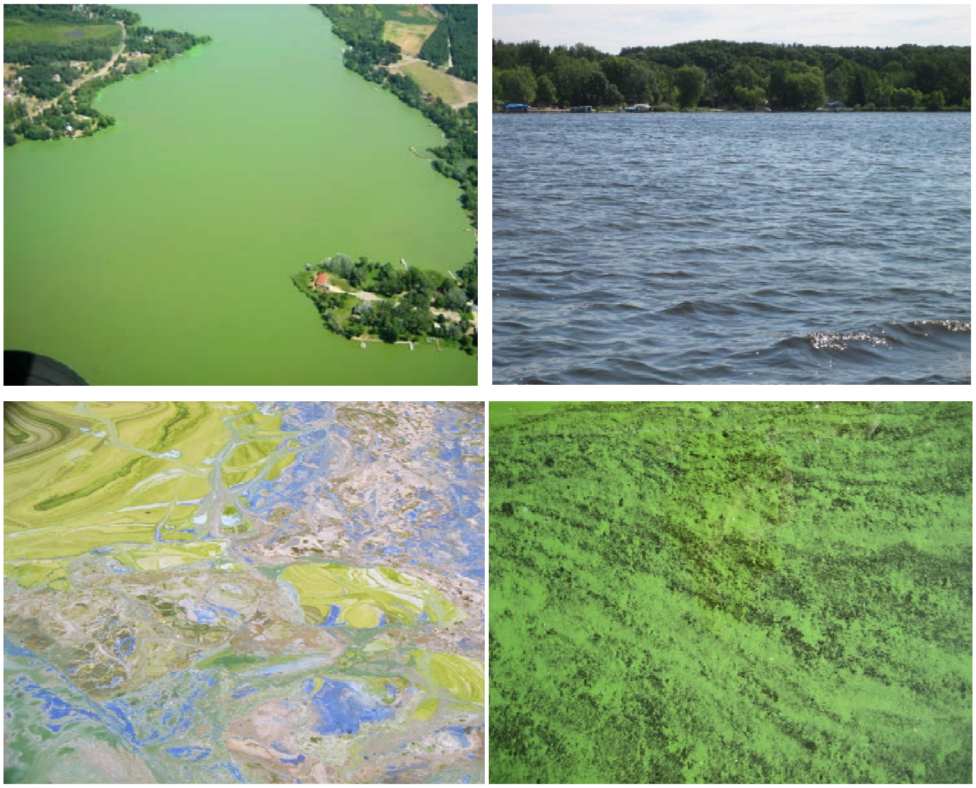
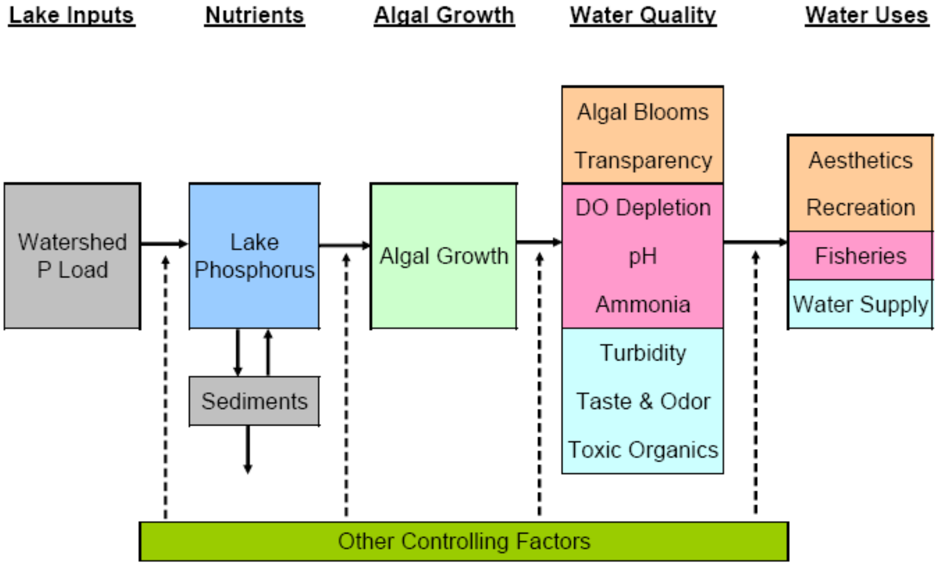


Figure 3 shows cause-effect pathways linking algal blooms to impairment in water quality and water uses. While highly variable depending on such factors as season, hydrology, and climate, algal blooms in eutrophic lakes are ultimately triggered by external phosphorus loads that are stored and recycled between the water column and bottom sediments (Sondergaard et al., 1999, 2005; Hakanson, 2004; Scheffer, 2004). Hyper-eutrophic conditions are not unusual in shallow lakes with large agricultural watersheds, depending on the extent to which phosphorus sources (animal waste, fertilizer, and crop residues) are effectively managed (Schippers et al, 2006; Sharpley et al., 2003, 2006; NRDC, 2010).

Figure 3 Causal Pathways Linking TP Load to Lake Water Quality & Uses



The Clean Water Act requires development and implementation of a plan to reduce watershed nutrient loads sufficiently to achieve water quality standards. The “Total Maximum Daily Load” (TMDL) regulations provide a framework for this process (USEPA, 2009; MPCA, 2009). The TMDL is essentially the assimilative capacity of the lake, or the amount of load that it can accept without exceeding water quality standards. Despite the reference to “daily load” in the regulations, lake phosphorus TMDLs are typically formulated on long-term-average time scales that govern lake water quality responses to nutrient loads and are consistent with derivation of the standards as long-term summer means (Heiskary & Wilson, 2008; Walker, 2003).

Table 1 compares historical LRL water quality conditions with the designated lake standards for shallow lakes in the CHF ecoregion of Minnesota. Comparisons of 1979-2003 with 2006-2008 data indicate significant long-term increases in total phosphorus (TP) and chlorophyll-a (Chl-a, measure of algal density) concentrations, as well as decreases in transparency (Secchi depth). While causal factors responsible for the historical trends are difficult to evaluate because of data limitations and climatologic variations described below, it is clear that significant reductions in phosphorus loads and lake concentrations are needed to meet the standards.

Table 1 Observed Water Quality vs. Shallow Lake Standards

	Standard	1979-2003	2006-2008
Total P (ppb)	< 60	116 - 179	202 - 315
Chl-a (ppb)	< 20	69 - 90	114 - 227
Secchi Depth (m)	> 1	0.5 - 1.1	0.3 - 0.6

This report describes development of a TMDL estimate using a mathematical model that links external phosphorus load to the 60 ppb lake standard (Figure 3). The term "estimate" reflects the uncertainty commonly associated with predicting lake responses to P load reduction, particularly in shallow hyper-eutrophic impoundments (Heiskary & Lindon, 2005). Data limitations preclude development of relatively complex dynamic mass-balance models used in other TMDL assessments for other shallow lakes supported by ten or more years of data (Walker, 2000ab; 2001; 2009; Walker & Havens, 2003). Sufficient site-specific and regional data exist to support estimation of the TMDL using relatively simple, empirical models calibrated to data from other lakes (Canfield & Bachman, 1981; Wilson & Walker, 1988; Heiskary & Wilson, 2008). Despite uncertainties, the TMDL estimate provides an explicit goal that can be refined in the future as additional data are collected, load reductions are achieved, and lake responses are measured.

The TMDL development and supporting data analyses are described in the following sections:

- Data Sources
- Water Quality Standards
- Lake Water Quality Conditions
- Water and Mass Balances
- TMDL Derivation
- Spatial and Temporal Variations
- Margin of Safety
- TMDL Implementation
- Conclusions
- References

The Appendix contains supporting computations, data summaries, data displays, and related information on shallow lake P dynamics derived from the literature reviewed in the course of developing the TMDL model.

Data Sources

The following data and reports provide information for developing TMDL and for tracking changes in the lake and tributaries as P loading controls are implemented:

- Watershed and lake water quality monitoring data collected by Benton County Soil and Water District in 2006-2008 to support development of TMDLs for the lake and tributaries (BSWCD, 2008; 2009).
- Streamflow and lake water level monitoring conducted by MPCA in 2006-2009.
- Water quality data from diagnostic studies performed by MPCA (1974) and Heiskary (1991).
- Transparency and user perception data collected in various years between 1990 and 2008 under that statewide Citizens Monitoring Program (MPCA, 2009b).
- Regional precipitation, runoff, and air temperature data compiled from internet sources.
- Measurements of sediment characteristics and phosphorus release rates at several lake stations in 2008 (James,2008)
- A sediment core study conducted in 2008 to document historical conditions and estimate sediment accumulation rates at the deepest point in the lake (Garrison & LaLiberte, 2009)
- GIS data layers (land use, hydrography, land elevation, soil types, feedlots) derived from statewide databases.
- Analysis of historical fluctuations in water levels, as controlled by water levels in the Mississippi River and runoff from the LRL watershed (Ford et al, 2003).

The lake and tributary water quality data are listed and displayed across various spatial and temporal dimensions in the Appendix.

Six lake sites located along the north-south axis were monitored between July and October of 2008 (Figure 1). The sampling design included field data (transparency and vertical profiles of dissolved oxygen, temperature, conductivity, pH, and turbidity) and 0-2 meter integrated samples analyzed for nutrients, chlorophyll-a, and inorganic chemistry at each site. Bottom samples were collected at the deepest point (Site 204, LRL-4). Limited water quality data from previous years (1979-1981, 1990, 2003, 2006, and 2007) were obtained from the MPCA (2009b) internet database. Codes used to identify lake monitoring sites varied over the years and have been consolidated to reflect the basic downstream order (LRL-1 to LRL-6, Figure 1).

Tributary water quality data were collected at five tributary sites (two on Little Rock Creek, Bunker Hills, Zuleger, and Sucker) between May 2006 and October 2008 (Figure 1). The watershed sites were generally monitored biweekly with supplemental samples collected during high runoff periods. Two additional stream sites were located downstream of the lake at the confluence of Little Rock Creek and the Mississippi River. These sites were included in the 2008 survey design to provide a basis for evaluating the potential effects of phosphorus transport into LRL from the River as a result of backflow and/or dispersion. Under the 2006-2008 monitored conditions, the River functioned primarily as a “dam” for the lake, as opposed to a source of inflow. Hydraulic modeling results indicate that backflow has occurred during infrequent episodes of extremely high water levels in the River (Ford et al, 2003). While these events are likely to trigger shoreline erosion, backflow itself is not likely to represent a significant long-term source, based upon the fact that phosphorus concentrations in local runoff in 2006-2008 exceeded those measured in the River (see Appendix).

The MPCA made daily streamflow measurements between July 2006 and October 2009 at each site except Zuleger, where monitoring was infeasible due to backwater conditions from the lake. The streamflow measurements started in July 2006 and thus did not reflect the entire runoff season. Regression models were used to estimate missing flow data and provide a complete daily flow record for March-October of 2006-2009. Direct inflows to LRL from Little Rock and Sucker Creeks reflected ~71% of total watershed runoff. The remaining inflows were estimated based upon drainage area ratios relative to Little Rock Creek. Lake outflows were computed from the water budget (inflow + precipitation – evaporation – volume increase).

There is considerable uncertainty in characterizing the long-term-average phosphorus budgets and lake water quality conditions based the 2006-2008 data collected to support development of the TMDL (Appendix). The uncertainty results from data gaps, short period of record, and drought conditions. Tributary sampling did not capture early spring runoff periods in 2006 and 2008. LRC spring runoff in peaked at 148 cfs in 2007 as compared with 583 cfs in 2009, when water quality sampling was not conducted. No data were available on tributary flows, P concentrations, or P loads prior to 2006. Because of relatively dry and warm summers, lake water quality conditions observed in 2006-2008 may not have been representative of long-term-average conditions under current watershed conditions and nutrient loading regimes. Because of P storage and recycling between the lake water column and sediments (Figure 3), it is likely that water quality conditions in 2006-2008 were impacted by phosphorus loads that occurred in previous years.

Despite the data limitations, sufficient site-specific and regional data exist to support TMDL estimation using relatively simple, empirical models calibrated to data from other lakes. Continued lake and watershed monitoring over the course of TMDL implementation will provide a basis for refining the water and phosphorus balances and tracking responses to implementation of phosphorus controls using an adaptive management strategy (Walker, 2003).

Water Quality Standards

Absent an approved site-specific standard, regulations require that the TMDL be formulated to meet the existing TP standard (60 ppb) for shallow lakes in the CFH ecoregion (Heiskary & Wilson, 2005; 2008). Heiskary and Lindon (2005) describe the derivation of the standard based upon regional lake datasets and considerations of the following factors:

- Correlations between TP concentration in Minnesota lakes with the following
  - Mean chlorophyll-a and frequency of nuisance algal blooms
  - Secchi depth (transparency)
  - User perceptions of aesthetic qualities and recreational potential
  - Fish populations and vegetation characteristics
- Comparisons with data from reference (minimally impacted) shallow lakes in the ecoregion
- Comparisons with estimates of TP concentrations under pre-settlement (1750-1900) conditions estimated from sediment core studies.
- Review of literature pertaining to effects of TP levels on algal blooms, vegetation, and fisheries.

The TMDL is derived to meet the eutrophication standards with respect to TP, chlorophyll-a, and Secchi depth by reducing the TP load sufficiently to meet each standard. The following text (Heiskary & Lindon, 2005, p. iv) summarizes the objectives, rationale, assumptions, and caveats associated with derivation of the standards:

> *"This study did not develop a predictive model; rather we characterized linkages among nutrient concentration, algal abundance and composition, macrophyte (submergent and floating-leaf) composition and coverage, fishery composition and management and related factors based on a set of representative shallow lakes from across west central Minnesota. These linkages combined with region-wide patterns in lake trophic status (both pre-European and modern-day), user perception and literature review, provide a basis for establishing nutrient criteria to protect uses such as secondary contact (boating and aesthetics) and fish and waterfowl habitat.*
>
> *In summary, based on the various interrelationships among trophic status variables, rooted plant metrics and other considerations it appears that appropriate ranges for selecting eutrophication criteria values for shallow*

> *lakes in the CHF ecoregion are:*
>
> - *Secchi transparency - greater than 0.7 to 1.0 meters;*
> - *Chlorophyll-a - less than 20 – 30 µg/L;*
> - *Total phosphorus – less than 60 – 80 µg/L;*
>
> *Given this range of values, and acknowledging that other biotic and abiotic factors can be very significant in determining whether a lake can support a healthy and diverse population of rooted macrophytes, we are inclined to recommend criteria be set at the lower end of each range of the aforementioned values, i.e. maintain summer average Secchi of 1.0 m or greater, summer average chlorophyll-a of 20 µg/L or lower, and summer average total phosphorus of 60 µg/L or lower. While we are not offering nitrogen criteria at this time, it would appear to be beneficial to keep TKN below 2.0 mg/L when possible. Based on the relationship between TP and TKN, maintaining TP below 60-80 µg/L should yield TKN <2.0 mg/L.*
>
> *Maintaining values at or below these ranges will not absolutely ensure that a shallow lake will remain in a macrophyte-dominated state and support the various uses described for 2b & 2c waters (Minn. Rule Ch. 7050), but should reduce the likelihood that the lake will switch to an algal-dominated state, which as repeatedly noted in the literature can be rather hard to reverse once the change has occurred. Also, maintaining trophic status values at or below these ranges should decrease the likelihood that curly-leaf, a non-native species, will become dominant and further contribute to a shift towards algal dominance.*
>
> *Lakes currently below the TP and chlorophyll-a thresholds should be protected against further increases in TP whenever possible because as these shallow lakes become increasingly nutrient-rich these nutrients will yield distinct increases in chlorophyll-a, which in turn will contribute to reduced transparency and increase the likelihood of a shift from plant-dominance to algal dominance. For lakes currently above these levels reducing TP to 60 µg/L or lower should result in reductions in chlorophyll-a and improved transparency. While this should increase the likelihood of a shift to plant dominance it cannot be guaranteed because of numerous biotic and abiotic factors noted in this study and in the literature on this topic.*

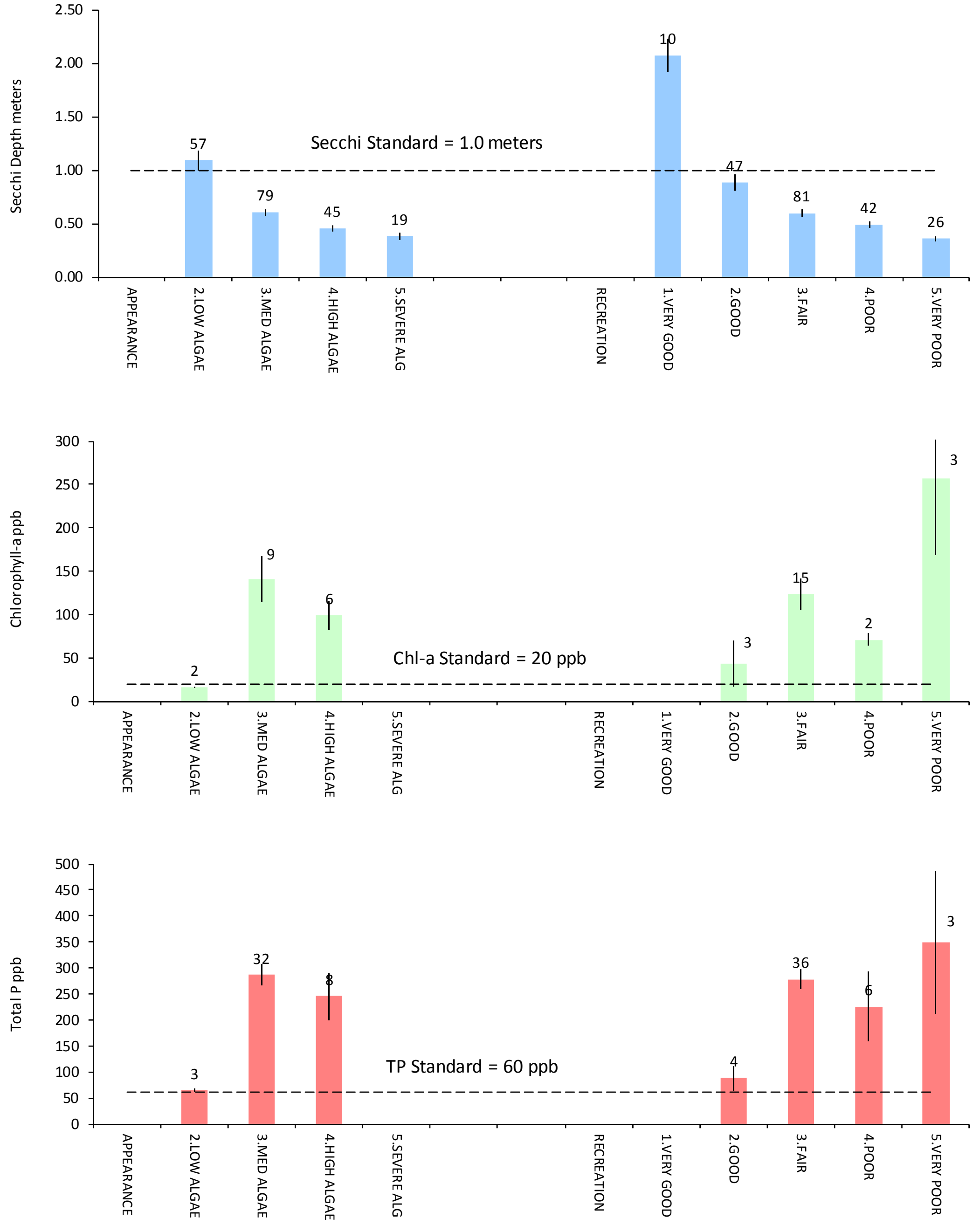
The 60 ppb standard was set at the lower end of the 60-80 ppb range consistent with sustaining a plant-dominated ("clear-water") state as opposed to an algal-dominated (turbid) state. For enriched lakes similar to LRL that are already in the turbid category, achieving reductions in lake TP would be expected to provide reductions in algal density (chlorophyll-a) and increases in transparency. Sas (1989) noted significant reductions in bluegreen bloom frequencies at TP concentrations below 100 ppb. While a shift towards a plant-dominated state may occur at lower TP levels, there is no expectation that it would be complete. A partial shift to native vegetation species, as manifested by increased growth in the shoreline areas, for example, could be considered beneficial because it would improve fish habitat and help to stabilize bottom sediments. As noted by Heiskary and Lindon (2005), achieving reductions in TP levels could also reduce the risk of excessive growth of the exotic curly-leafed pondweed, which has been observed in portions of the lake (LRLA & MDNR, 2007). The derivation of the standard acknowledges that there is considerable uncertainty in forecasting the trajectory of hyper-eutrophic shallow lakes such as LRL to reductions in P load. The uncertainty can be addressed through adaptive implementation of the TMDL.

Lake Water Quality Conditions

LRL water quality and user survey data collected under MPCA's Citizen Lake Monitoring Program over the 1990-2008 period are summarized in Figure 4. Water quality data from each sampling event are paired with user perceptions of aesthetic quality and suitability for recreational uses (Heiskary & Walker, 1985). Survey results are expressed on a scale of 1 to 5 (generally, excellent to poor). On four sampling dates when recreational potential was ranked in the second category ("good"), the average TP concentration was 88 ± 24 ppb. On three days when the aesthetic quality was ranked in the second category ("low algae"), the average TP concentration was 64 ± 4 ppb. The user survey results are also reasonably consistent with the distributions of the chlorophyll-a and transparency data.

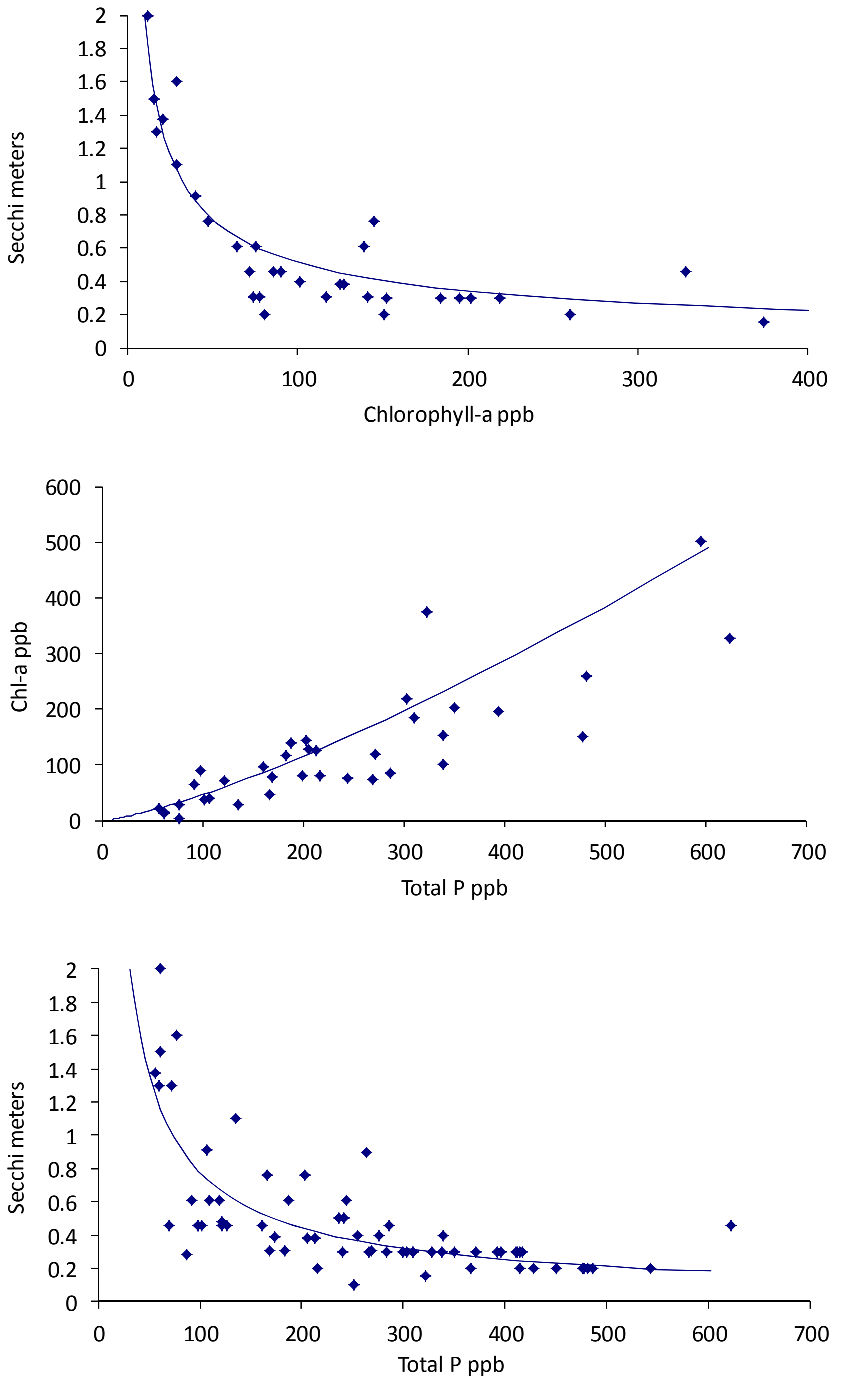
Figure 5 shows that phosphorus levels are highly correlated with chlorophyll-a levels and Secchi depths across individual sampling events. These correlations indicate that achieving incremental reductions in lake TP levels over the course of TMDL implementation would provide significant reductions in algal blooms that would be perceptible by lake users.

Figure 4 User Perceptions vs. Water Quality Measurements



Means and standard errors of trophic state indicators in user perception categories
for physical appearance and recreational potential. Labeled with number of observations.
All Little Rock Lake sites, 1990-2008, Citizens Monitoring Program
Dashed lines indicate MPCA watedr quality for shallow lakes in the CHF Ecoregion.
Units: ppb = micrograms per liter (µg/L)

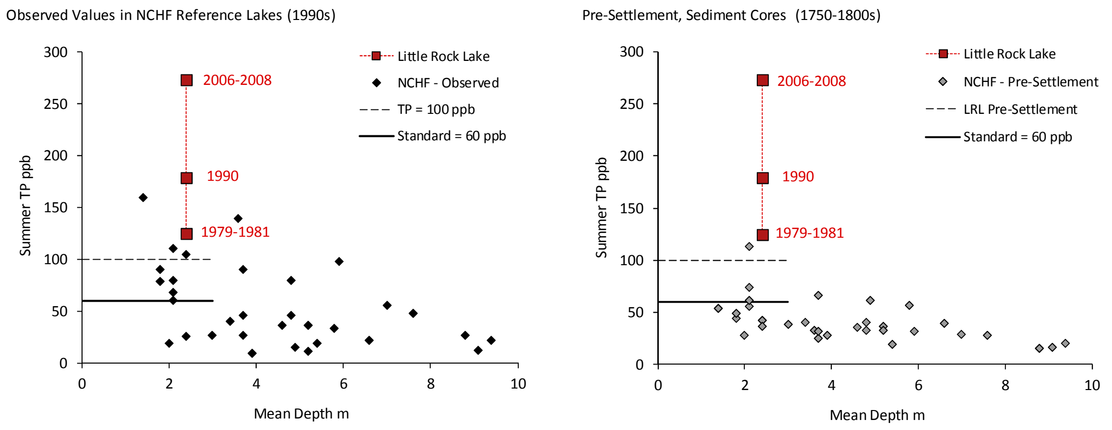
Figure 5 Correlations vs. MPCA Statewide Regressions



LRL data are from May-September, 1990-2008; each point represents a paired sample. Lines based upon state-wide regressions of summer-mean data from other Minnesota Lakes (Heiskary & Wilson, 2008). Units ppb = micrograms per liter (µg/L).

Figure 6 compares historical summer-mean TP levels with data from other regional lakes classified as “reference” or “minimally-impacted” (Heiskary & Wilson, 2005). The susceptibility of shallow lakes to eutrophication problems is reflected by the negative correlation between TP concentrations and water depth. The left panel shows TP levels measured in the 1990s. The right panel shows estimates for pre-settlement conditions (1750-1900) derived from sediment cores. TP concentrations in LRL more than doubled over the years to levels that far exceed the standard and values observed in the other shallow lakes. TP concentrations averaged 125 ± 5 ppb in 1979-1981, 179 ± 23 in 1990, and 273 ± 35 ppb in 2006-2008. As discussed below, high values measured 2006-2008 may be partially attributed to extreme climatologic conditions (warm and dry), as opposed to a long-term trend in the lake water quality. The mean value for the reference lakes under pre-settlement conditions (on right) is similar to the 60 ppb standard.

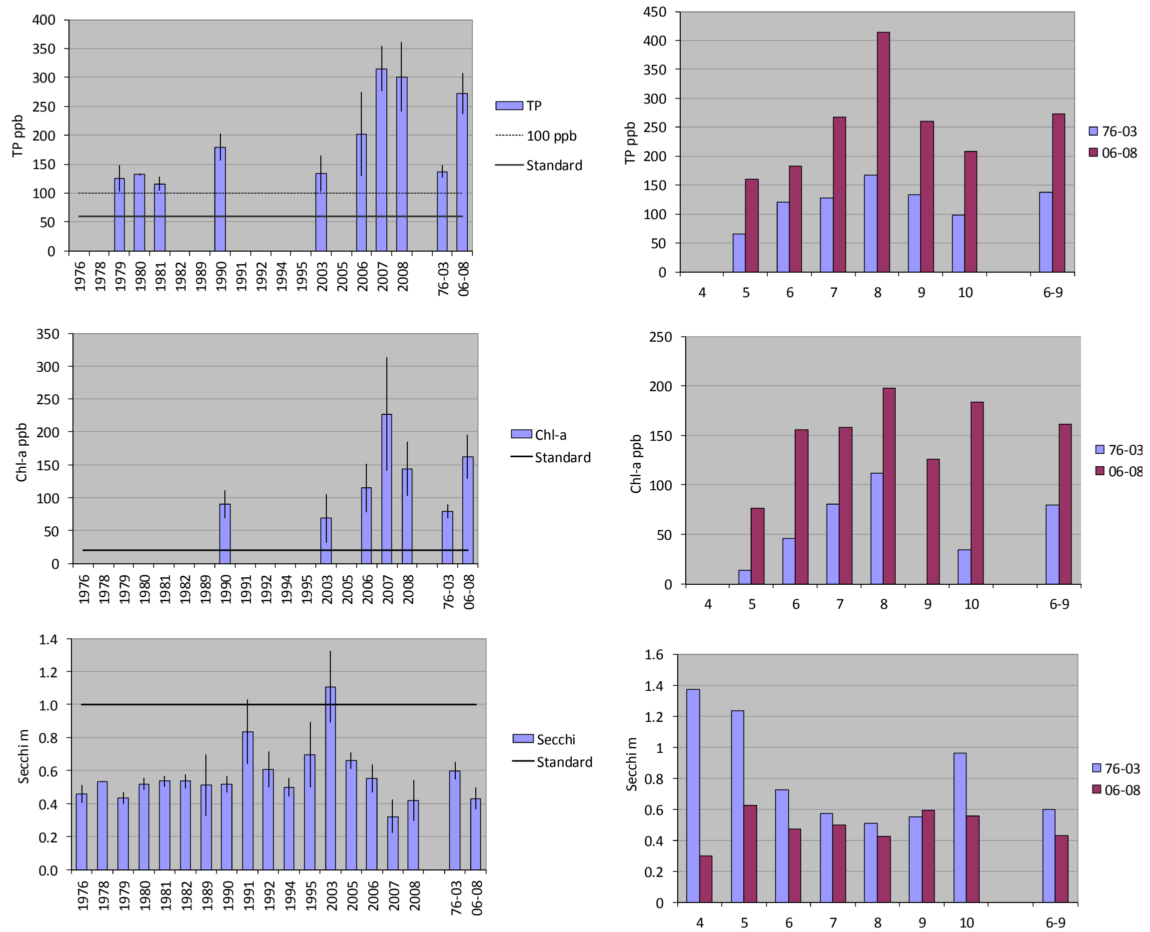
Figure 6 LRL Summer TP Concentrations vs. Data from Other NCHF Lakes



Sediment core studies indicate that LRL historical summer-average TP concentrations ranged from 109 ppb in 1911 to 176 ppb in 2008 (Garrison et al, 2009). These estimates were based upon diatom species distribution at sediment depths of 50-52 cm and 0-2 cm, respectively. While the relevance of the 1911 estimate (109 ppb) is questionable because LRL was a wetland at that time, it is similar to the 100 ppb TP criterion for extreme bluegreen blooms in turbid lakes (Sas, 1989; MPCA, 1974). While within the range of historical data, the 2008 estimate is relatively uncertain because it required extrapolation of the dating methodology beyond its calibration range. Other sediment profile data, including lower iron-bound P levels in the surface sediments (James, 2008) and increased dominance of microcystis (Garrison et al, 2009), are consistent with increases in nutrient enrichment and transition from a wetland to a turbid hyper-eutrophic lake over the years since LRL was formed.

Trends in summer-mean TP, chlorophyll-a, and transparency data over the 1976-2008 period are shown in Figure 7. The means are based upon samples collected in at least three out of the four summer months (June-September) in each year. The data for each parameter indicate that LRL was considerably more eutrophic during the 2006-2008 TMDL study, as compared with previous years. That conclusion is supported by apparent trends in the yearly time series and by comparisons of the 1976-2003 with the 2006-2008 averages by month shown at the right in Figure 7. While insufficient to support computation of summer means, TP concentrations of 200 ppb in September 1970 and 70 ppb in June 1971 were reported in the first LRL diagnostic study, which also noted a “heavy bloom” of bluegreen algae (Aphanizomenon) in September 1970 (MPCA, 1974).

Figure 7 Long-Term Trends in Phosphorus, Chlorophyll-a & Transparency


Summer Means (June-Sept)

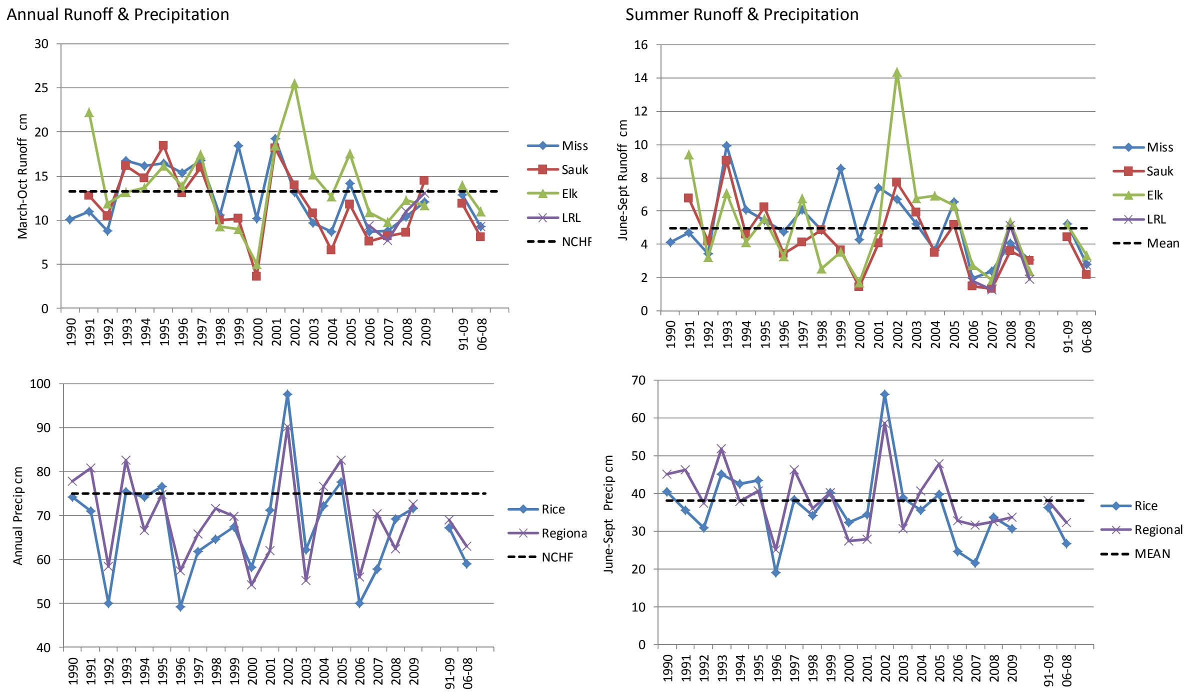
Monthly Means

Summer Means +/- 1 Standard Error

Interpretation of the apparent trends is difficult because of variations in climatologic conditions and limitations in the data. Data collected in 1990 under the second MPCA diagnostic study (Heiskary, 1991) provide the best historical frame of reference. While the overall percentage of developed land apparently did not change between 1990 (~68%, Heiskary, 1991) and 2006-2008 (~70%, BSWCD, 2009), an increase in TP load could have occurred as a result of increases in the intensity and/or types of agricultural and/or urban land uses. Precipitation and runoff from other regional watersheds were well below long-term averages, particularly in the summers of 2006-2007 (Figure 8). The relatively high TP levels and heavy algal blooms observed in those years may partially reflect warm and dry summers relative to the 1970-2009 period of record Figure 9.

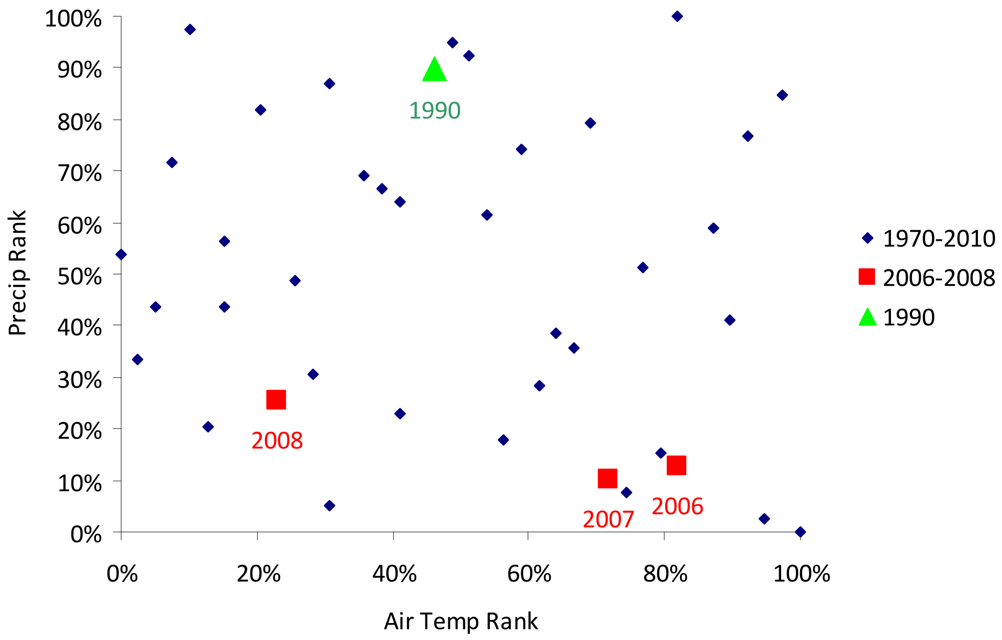
While external P loads would be higher in wet years, excessive algal blooms are more likely in shallow lakes during dry and hot summers, when lower base flows provide less dilution for P loads recycled from lake bottom sediments, algal growth rates and sediment decomposition rates are increased by warmer temperatures, and longer water residence times allow development of intense blooms. For example, summer chlorophyll-a levels are inversely correlated with flow in the Sauk River mainstem lakes, which are also in the NCHF ecoregion (Walker, 2009).

Figure 8 Regional Runoff and Precipitation Time Series



NCHF = Mean Runoff for NCHF Ecoregion (MINLEAP, Wilson & Walker, 1988); Regional = US Climatologic Database, Minnesota Region 5; Rice = NWS at Rice; Mean = Mean of Long-Term Datasets
Conditions were relatively dry during the TMDL study period (2006-2008).

Figure 9 Climatologic Conditions in Lake Study Years



US Long-TermTerm Climate Monitoring, Minn. Divison 5; 1970-2009, June-August

Figure 10 Seasonal Variations in LRL Water Quality During 1990 and 2006-2008.shows seasonal variations in water temperature and trophic state indicators in 1990 and 2006-2008. Lower TP and chlorophyll-a levels observed in 1990 are consistent with lower water and air temperatures in July and August. Relatively high water temperatures, TP levels, high Chl-a levels, and low Secchi depths were observed in June and July of 2007. Toxic bluegreen algae (Microcystis specie) and atmospheric hydrogen sulfide releases from anoxic bottom sediments were also reported (Lindon et al., 2007). Comparisons with data from other years indicate that severe conditions in 2007 were triggered by high tributary TP loads in March-April (see below) followed by summer low flow and high temperatures.

Figure 10 Seasonal Variations in LRL Water Quality During 1990 and 2006-2008.

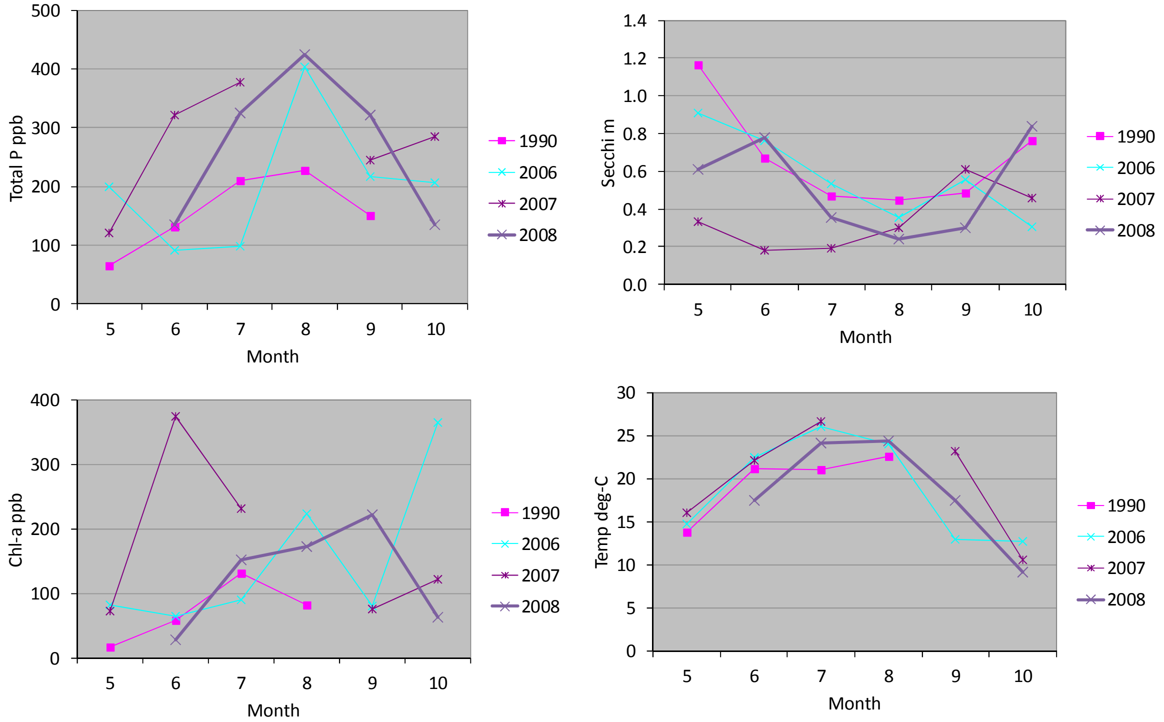
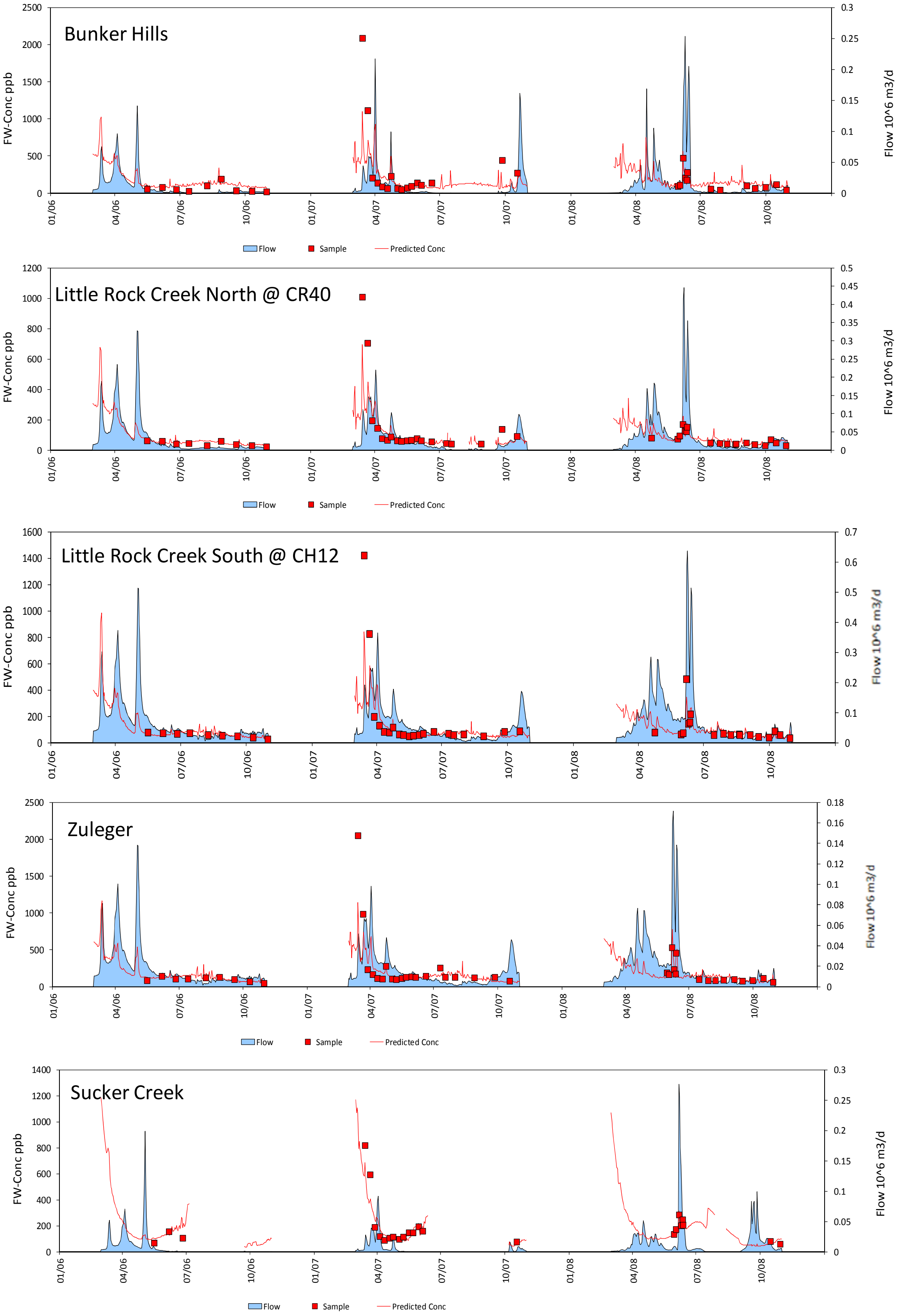


Figure 11 shows daily flows and TP concentrations in the tributaries over the 2006-2009 period. The Appendix contains more detailed displays and summaries of the data. Red lines show daily estimated TP concentrations predicted from regression equations relating sampled values to flow and season (Walker & Havens, 2003). Extremely high TP levels (~500-1000 ppb), as well as high concentrations of fecal coliforms and other nutrients indicative of animal waste, were measured in early spring runoff of 2007 (Appendix). Concentration spikes also occurred during the June 2008 runoff event.

Early spring rains in 2007 would have promoted the transport of nutrients from watershed sources to the lake. LRC spring runoff peaked at 148 cfs in 2007 as compared with 583 cfs in 2009, when water quality sampling was not conducted. Lake water levels rose by 0.6 ft in spring of 2007 as compared with 2.5 ft in spring of 2009 (Appendix). It is likely that the much larger spring runoff event in 2009 would have contributed substantially more phosphorus to the lake, as compared with spring runoff in 2006-2008.

Figure 11 Daily Flows & Flow-Weighted Mean Phosphorus Concentrations at Tributary Sites



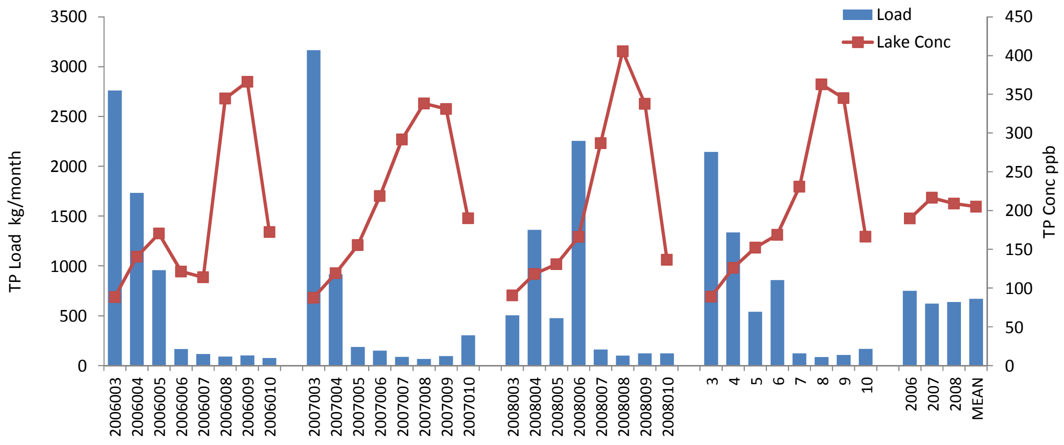
Water and Mass Balances

Estimated water and phosphorus mass balances for the March-October of 2006-2008 period of record are summarized in the Appendix (A-1). These provide cornerstones for developing the TMDL using the methods described in the next section (see TMDL Derivation). The assumptions and calculation results are listed in Appendix A-1. Components of the water and mass balances include the following:

1. Monitored Inflows in Port
    - Little Rock Creek above Lake Inflow (CH 12)
        - LRC North (above CR 8)
        - Bunker Hills
    - Zuleger Creek (Concentration Only)
    - Sucker Creek
    - Rainfall (measured at Rice, approximately 3 miles from LRL)
2. Unmonitored Inflows
    - Areas the drain directly into the lake (lakeshed)
    - Wastewater disposal systems (septic tanks) on shoreline lots.
    - Atmospheric deposition
3. Unmonitored Outflows
    - Outflow volumes computed from water budget (inflow + precipitation – evaporation – increase in storage)
    - Outflow concentrations based upon data from the monitoring site at the south end of the lake (LRL-4) in 2008; estimated at 92% of the lake-mean concentration in other years, based upon calibration to the 2008 data.
    - Evaporation based upon regional data
4. Storage in Lake
    - Lake volume computed from surface area and stage.
    - Lake TP concentration

Gaps in the tributary flow and phosphorus data were filled using regression techniques, interpolation, and drainage area ratios. While data to evaluate groundwater inflows and outflows are not available, as is typical of lake studies (Walker, 1985), they are likely to be small relative to the surface TP loads, given the relatively large watershed and high concentrations TP in the tributaries. Any contributions from the groundwater are assumed to be unchanged relative and would not impact the calculation of the tributary TP loads under the TMDL. Several assumptions were also necessary to evaluate the unmonitored inflow and outflow components. Given the data limitations and assumptions required, the inflow loads and mass balances are considered approximations to be refined using future monitoring data.

Figure 12 Monthly TP Loads and Lake TP concentrations



Monthly variations in TP load and lake concentration are shown in. Figure 12. Despite uncertainty in the monthly values, the pattern is typical of P dynamics in shallow eutrophic lakes (Sheffer, 2004; Sondergaard, et al, 1999; Heiskary & Linton, 2005):

- The initial buildup in lake TP levels is triggered by spring runoff loads.
- The mid summer P dynamics are dominated by P recycled from the bottom sediments, which is fueled by P deposited to the sediments earlier in the season and in previous years (Figure 3
- The P buildup is accelerated by changes in hydrology and chemistry as the summer blooms develop. Mechanisms are related to decreases in flow (less dilution), increases in temperature, increases in sunlight, increases in pH, and decreases in dissolved oxygen levels at the sediment-water interface. Mass-balance calculations (Appendix) indicate that rates of P buildup in the summer are reasonably consistent with laboratory studies of sediment cores collected in LRL and other lakes, when sensitivities to pH, temperature, and intermittent oxygen depletion are considered (James, 2008).
- Phosphorus decreases in the fall reflect die-off and sedimentation of algal blooms.
- As illustrated in Figure 3, P deposited to the sediments over the years is either recycled to the water column or buried below the sediment horizon that interacts with the water column (typically ~ 10 cm). The sediment accretion rate in LRL is estimated at 1-2 cm/yr (Garrison & LaLiberte, 2009).
- Data from other shallow lakes (Appendix) indicate that P buildup over the summer is highly correlated with the initial TP concentrations in the winter and

> early spring. In lakes with spring TP concentrations < 50 ppb, the summer buildup is negligible; i.e. the spring and summer concentrations are equal. In lakes similar to LRL with spring TP levels of ~150 ppb, the summer means are ~ 300 ppb. The pattern is likely to reflect feedback loops that accelerate the rate of sediment P recycling as the lake becomes more eutrophic. The loops are driven by increases in pH, increases in organic matter production, and decreases in aquatic vegetation that otherwise stabilize the sediments. The linkage between spring and summer P indicates the importance of decreasing TP loads in spring runoff in order to achieve the summer TP standard.

Based upon review of the water quality and sediment data, P cycling mechanisms operating in LRL are likely similar to those observed in other lakes. Explicit mass-balance models of sediment-water interactions have been developed for other shallow-lake TMDL assessments supported by several years of data (Walker, 2000ab, 2001, 2009). These allow simulation of seasonal and year-to-year variations in lake conditions in response to variations in hydrology, climate, and reductions in P load. While additional monitoring data would be needed to support development of a similar model for LRL, the simpler empirical approach described below is sufficient for an initial TMDL assessment.

TMDL Derivation

Starting from the existing phosphorus loads derived in the previous section, load reductions sufficient to achieve the lake TP target are derived by applying an empirical phosphorus balance model, as described in detail below. Sufficient site-specific and regional data exist to support TMDL estimation using relatively simple, empirical models calibrated to data from other lakes. Generalized models of this type are robust to uncertainty in site-specific data and have been widely used in lake management for a few decades (Vollenweider, 1976; Canfield & Bachman, 1981; Wilson & Walker, 1988; Walker, 1984; 2006). These models typically have uncertainties ranging from 30-40% of the predicted value, depending on dataset and model (Walker, 1985). Continued lake and watershed monitoring over the course of TMDL implementation will provide a basis for tracking progress, refining the model, and reducing uncertainty in the TMDL estimate (Walker, 2003).

The Canfield-Bachman (1981) model is widely applied in Minnesota lake P assessments and provides a robust basis for TMDL development when sufficient data are not available for developing site-specific models. The model predicts lake summer-mean TP concentration based upon average-annual inflow volume, TP load, lake mean depth, and lake surface area. It was originally developed from a nationwide dataset representing 290 lakes. A slightly different version was originally calibrated to reservoir data and subsequently tested against data from Corps of Engineer reservoirs and other large datasets (Walker, 1985). The model is used in the Minnesota Lake Eutrophication Analysis Procedure (MINLEAP, Wilson and Walker, 1988) to predict water quality conditions in relatively unimpaired lakes in each ecoregion of the state. LRL water quality impairment is indicated by comparisons with data from MINLEAP predictions and calibration lakes (Appendix). Equations are listed in the Appendix (A-3).

The TMDL can be defined as the long-term average TP load consistent with achieving the long-term-average TP standard (summer mean = 60 ppb). As explained below, tributary runoff volumes for the 1991-2009 baseline period were applied to a range of assumed flow-weighted mean concentrations and the model was repeatedly run in order to estimate the runoff concentration required to achieve the lake TP target. Equations used in the derivation are listed in the Appendix. The derivation involves the following steps:

- To establish a hydrologic baseline, water and phosphorus loads measured in 2006-2008 are adjusted to reflect long-term-average (1991-2009) conditions using regional streamflow and precipitation data (Figure 8). Average tributary flows are increased by 37% based upon runoff data for the Sauk and Elk River watersheds. Application of data from other regional watersheds was necessary in order to extend the hydrologic record because long-term hydrologic records are not available for the LRL watershed. The resulting average runoff (13 cm) is similar to that assumed for this ecoregion in MINLEAP (Wilson & Walker, 1988).

- Loading scenarios are constructed by applying a hypothetical flow-weighted-mean concentration limit to each tributary discharging directly into the lake (Little Rock, Zuleger, and Sucker). The scenarios cover a range of 40 to 200 ppb in runoff concentration (Figure 13). While the TMDL is independent of the allocation across sub-basins, the same concentration target could be applied to the LRC sub-basins (northern LRC and Bunker Hills) during implementation. Other allocations across tributaries could be used if they provide a more cost-effective method to achieve the same total tributary load. The predicted lake TP concentration is driven by the sum of the TP loads from all sources and is independent of how the loads are split among the individual tributaries or individual sub-basins.

- Total P loads are computed by applying the assumed flow-weighted-mean, March-October runoff concentration to the adjusted 1991-2009 baseline flow for each tributary.

- Direct discharge from septic systems is not permitted under MN state law; therefore the load allocation for septic systems is set to zero.

- TP concentrations in lakeshed runoff are assumed to equal the baseline values. These are conservative assumptions to the extent that additional measures are taken to reduce these sources over the course of TMDL implementation and hence provide a margin of safety in the TMDL allocation, as discussed below.

- The Canfield-Bachman model is applied to predict confidence intervals for lake TP concentration over a 40-200 ppb range in runoff concentration (Figure 13).

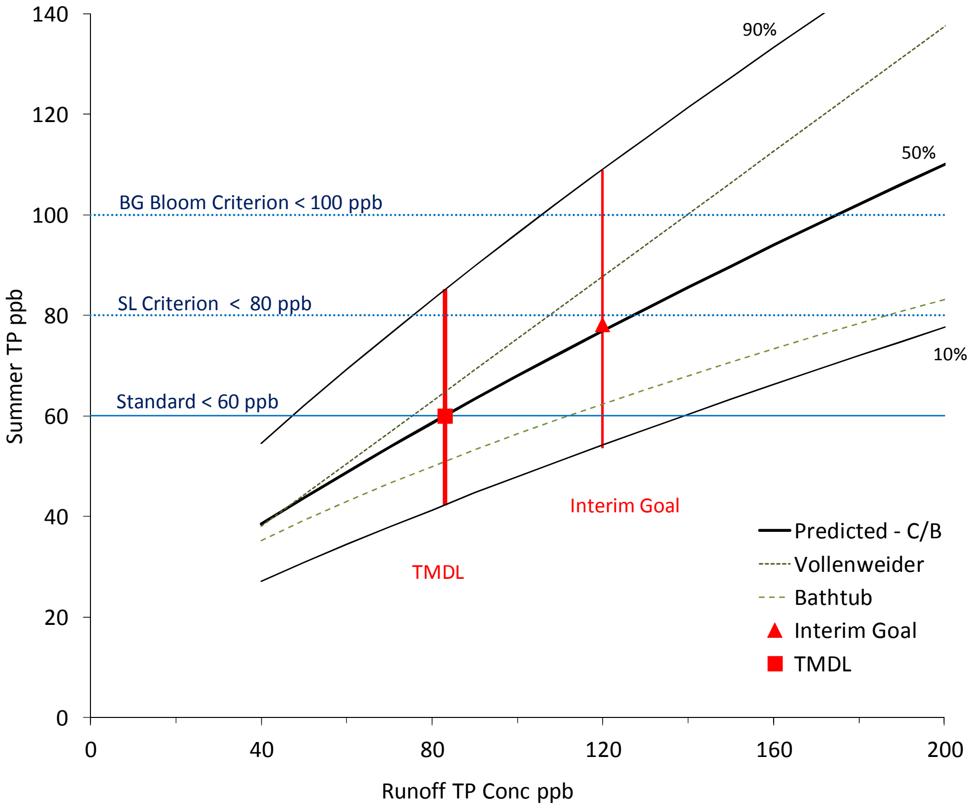
Testing against large datasets indicates that empirical models of this type typically have log-normal error distributions and 80% confidence intervals (10^{th} to 90^{th} percentiles) ranging from approximately 70% to 140% of the predicted lake TP concentration (Walker, 1985).

- To evaluate sensitivity to modeling assumptions (Appendix), other empirical models (Vollenweider, 1976; Walker, 2006) are also applied to predict lake P concentrations. These results generally fall within the confidence intervals predicted by the Canfield-Bachman model.

- Transparency and chlorophyll-a levels are predicted from lake TP concentrations using empirical equations developed by MPCA based upon data from other shallow lakes in this region of Minnesota (Heiskary & Lindon, 2005; Heiskary & Wilson, 2008), as listed in Appendix A-3.

Lake TP responses to variations in runoff concentration are shown in Figure 13. At a runoff concentration of 83 ppb uniformly applied to each gauged tributary, the predicted lake concentration is 60 ppb and the 80% confidence interval is 42 to 85 ppb. The estimated risk of exceeding the 80 ppb criterion (upper end of the 60-80 ppb range derived by Heiskary & Lindon (2005) is 15%. The estimated risk of exceeding the 100 ppb criterion for extreme bluegreen blooms (Sas, 1989) is 3%. At a runoff concentration of 120 ppb from each tributary, the estimated risks of exceeding the 80 ppb and 100 ppb levels, which might be considered as interim targets for the TMDL, are 47% and 18%, respectively. This scenario could be considered as an interim target for implementation of the TMDL. The 83-120 range in runoff concentration is within the inter-quartile range of values measured in relatively unimpacted streams in the NCHF ecoregion (25th percentile = 70 ppb, 75th percentile = 120 ppb, Heiskary & Wilson, 2005).

Figure 13 Predicted Response to Variations in Runoff TP Concentration



Runoff TP	Lake Inflows from Gauged Tributaries (Little Rock Creek, Zuleger Creek, Sucker Creek)
Predicted TP	Canfield/Bachman (1981) Lake Model, Black Lines = 10th - 50th - 90th Percentiles
Vollenweider	Vollenweider (1976) Model
Bathtub	Bathtub Model 1 (Walker, 2006)
Runoff TP	Mean Runoff TP Conc for Gauged Tribs (Little Rock, Zuleger, Sucker) Adjusted to Achieve Standard
	Ungauged Sources (Lakeshed, Shoreline, Atmospheric) Assumed Constant
Lake TP Criteria	
60 ppb	Shallow Lake Standard (Heiskary & Wilson, 2008)
80 ppb	Upper End of 60-80 ppb Criterion Range for Clear-Water Shallow Lakes (Heiskary & Lindon, 2005)
100 ppb	Criterion for Extreme Bluegreen Algal Blooms (Sas, 1989).

Flow and load allocations across sources for the baseline, interim and TMDL scenarios are listed in Table 2. For an average March-October inflow of 37 hm^3, the TMDL estimate is 3,236 kg and the combined inflow concentration for all sources is 87 ppb. The corresponding daily-average TP load over the March-October runoff season is 13.2 kg/day. Load reductions relative to the baseline range from 54 to 69% for the individual tributaries discharging directly into the lake, although these estimates could vary considerably because of uncertainty in baseline loads derived from the 2006-2008 data. The net effects of internal P loads recycled from the lake sediments are not explicit in the allocation because they are implicit in the calibration of the Canfield/Bachman empirical model, which relates lake summer P concentration to external load. It is assumed that the existing high rates of internal P recycling will decrease as the lake and sediments equilibrate to lower external P loads (Figure 3).

At a lake P concentration of 60 ppb, the regional regression models for shallow lakes (Heiskary & Lindon, 2005; Heiskary & Wilson, 2008) predict a chlorophyll-a concentration of 18 ppb and a mean Secchi depth of 1.08 meters (Table 2). The equations involved are listed in Appendix A-3. These results indicate that achieving the 60 ppb lake TP standard would provide compliance with the lake standards for chlorophyll-a and transparency (< 20 ppb and > 1.0 meters, respectively).

Table 2 Load Allocations for Baseline, Interim, & TMDL Scenarios

	Baseline 1991-2009 Hydrology, March-Oct				Interim Goal			TMDL Allocation (c)			
Source	Area km2	Flow hm3	Load kg	Conc ppb	Load kg	Conc ppb	Reduc %	Load kg	Load kg/d	Conc ppb	Reduc %
LRC Subwatersheds											
Bunker Hills	50.5	4.4	1267	287	529	120	58%	a			
LRC North	104.5	11.1	1766	160	1328	120	25%	a			
Direct Lake Inflows											
LR Creek - CH12	178.2	22.0	4046	184	2641	120	35%	1827	7.5	83	55%
Zuleger	48.0	5.9	1570	265	712	120	55%	492	2.0	83	69%
Sucker	11.2	3.0	551	182	364	120	34%	252	1.0	83	54%
Total Gauged	237.5	31.0	6167	199	3717	120	40%	2571	10.5	83	58%
Lakeshed	25.2	3.1	571	184	571	184	0%	571	2.3	184	0%
Total Watershed	262.6	34.1	6739	198	4288	126	36%	3142	12.8	92	53%
Shore. Septic Tanks			90		90		0%	b			
Stormwater (d)								2			
Total External	262.6	34.1	6829	200	4379	128	36%	3144	12.8	92	54%
Rainfall	5.1	3.1	94	30	94	30	0%	94	0.4	30	0%
Total Inflow	267.7	37.2	6923	186	4473	120	35%	**3238**	13.2	87	53%

Predicted Lake	Standard	Mean	10%	90%	Mean	10%	90%
Total P ppb	< 60	78	55	111	60	42	85
Chlorophyll-a ppb	< 20	24	14	42	18	10	32
Secchi m	> 1.0	0.9	0.6	1.2	1.1	0.8	1.5

a TMDL allocations for Little Rock Creek Subwatersheds are reflected in the total allocation for LR Creek at CH12

b Direct discharge from septic systems is not permitted under MN state law; therefore the load allocation for septic systems is set to zero . .

c The **TMDL** is the Total Inflow Load (3238 kg)

d Allowance for stormwater P loads associated with urban growth (0.05% of total)

Seasonal Variations

EPA regulatory guidelines (EPA, 2009; 40 C.F.R Part 130) require consideration of spatial and temporal water quality variations in formulating the TMDL. The 2008 monitoring data (Appendix) indicate that spatial variations across the lake monitoring sites were not significant, especially in the context of the large seasonal and random variations. Given the strong correlation between chlorophyll-a and TP levels across individual sampling events the lake TP concentrations achieved under TMDL conditions (

Figure 5) would provide significant reductions in the magnitude and frequency of extreme algal blooms. Considerations of seasonal variations in water quality and critical conditions associated with severe mid-summer algal blooms and resulting use impairment are embedded in the derivation of the 60 ppb TP standard (Heiskary & Wilson, 2008).

Margin of Safety

Regulatory guidelines (EPA, 2009) also require that the TMDL include a margin of safety (MOS) to provide assurance that the lake water quality standards will be achieved. The following factors can be considered as MOS components:

- The load allocation (Table 2) assumes that there will be no reduction in runoff P load from ungauged lakeshed relative to baseline conditions. These sources are estimated to account for 18 % of the TMDL allocation (571 / 3238 kg). The percentage of developed land in the lakeshed is 45%, as compared with 56% for the entire watershed (BCSWD, 2009). Implementing runoff P controls in the lakeshed similar to those implemented in the gauged watersheds would be expected to provide reductions in P load relative to baseline conditions that are slightly below those predicted for the gauged tributaries (58%, Table 2), but still significantly greater than the 0% assumed in the allocation.

- Achieving runoff total P load reductions would require greater percentage reductions in soluble reactive P (likely from animal waste & fertilizer), which has a greater impact on lake algal productivity, as compared with other forms of phosphorus that are less biologically available (Walker, 1985).

- Best Management Practices for reducing phosphorus loads from agriculture (Sharpley et al., 2006) and other sources could be conservatively designed in the process of implementation.

- The TMDL derivation was based upon data from relatively dry years which had high potential for phosphorus and algae buildup within the lake during the summer months because of low flushing rates. The highest tributary TP loads occurred in response to relatively intense early spring rains in 2007. These conditions indicate that the percentage reductions in tributary TP concentrations required to meet the TMDL goals expressed as a long-term averages may be lower than those estimated in the derivation.

- The 60 ppb lake standard is at the lower end of the 60-80 ppb range derived by Heiskary & Lindon (2005) as a TP criterion for shallow lakes. While this does not provide a margin of safety for achieving the lake P standard, it could be interpreted to provide a margin of safety for achieving the beneficial uses, upon which the lake P standard is conservatively based.

TMDL Implementation

Adaptive implementation of the TMDL is necessary given the uncertainties associated with predicting the effectiveness of management measures, as well as the time scales and ultimate responses of shallow lakes to load reductions. Continued monitoring is essential to improve the baseline and track responses to implementation of P loading controls over a range of climatologic conditions. Relevant responses include the tributary water quality and TP loads, lake water quality, vegetation, algae, fish, and user perception.

The monitoring plan for the lake TMDL can be integrated with the plan for the Little Rock Creek TMDL, the Citizens Monitoring Program, and MDNR surveys of fish and vegetation. The recommended design for the lake TMDL is similar to that designed for the 2008 (BSWCD, 2008) with the following emphases and exceptions:

- Monitor the entire spring-summer-fall season in tributaries and lake. While the tracking compliance with the lake standards require June-September sampling, spring and fall data are needed to evaluate responses to watershed P controls, the lake phosphorus mass balance, and the buildup of phosphorus and blooms over the growing season.

- The number of lake sites can be decreased from five to three: LRL-1, LRL-2 (deepest point) and LRL-5 (representing outflow from the lake). The lake outlet can be sampled during spring runoff if ice cover precludes access to the lake.

- The lake can be sampled monthly and parameters should include at a minimum (every year) TP (surface & bottom at LRL-2), chlorophyll-a, transparency, field profiles, and user perception survey. The remaining parameters specified in the 2008 design can be monitored every third year (LRL-2 only).

- Monitoring of tributary flow and water quality should be performed each year and integrated with the creek TMDL plan. The plan should include sufficient samples to capture the rising and falling limbs of the spring runoff period (at least weekly frequency).

- The downstream sites at the LRC basin outlet and Mississippi River can be eliminated. Special sampling is recommended to document lake responses to extreme flooding events on the Mississippi and shoreline flooding.

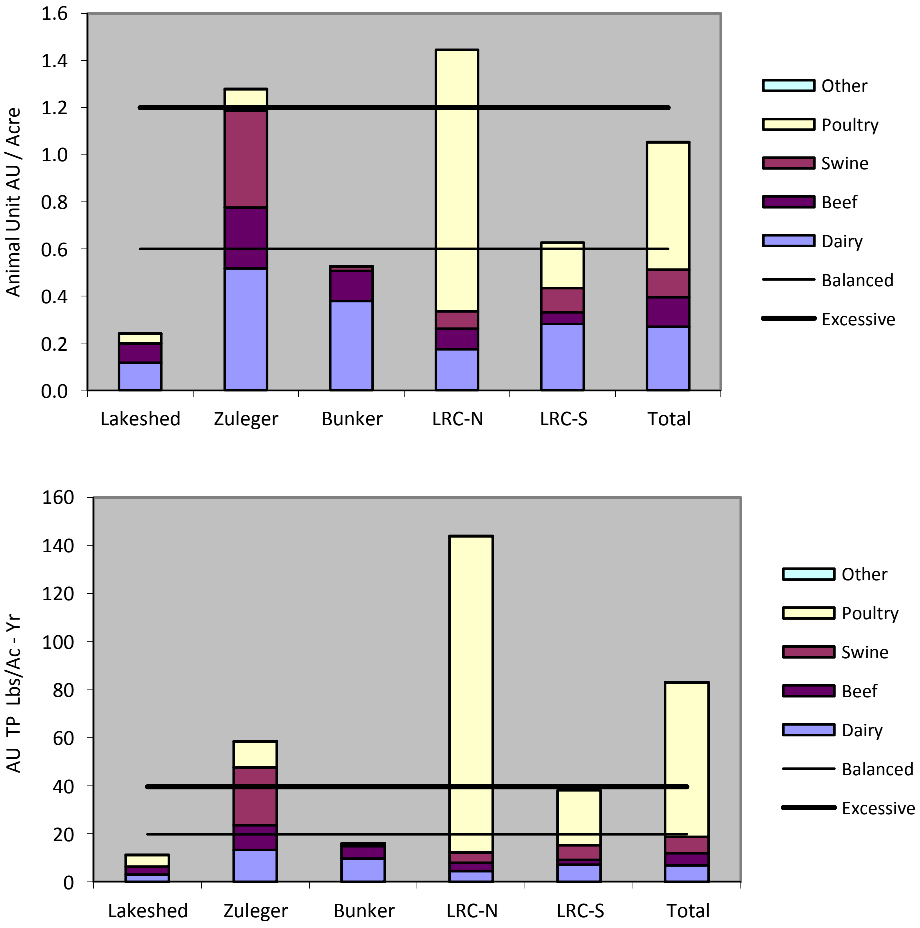
The results should be compiled and reported yearly to track progress. A comprehensive review of the data, mass balances, and modeling update should be performed after 3-5 years of continuous monitoring.

The extremely high concentrations, seasonal distribution, flow-dependence of several water quality constituents in spring runoff (fecal coliforms, BOD, ammonia-N, Kjeldahl-

N, Total P, and soluble reactive P, Appendix) indicate that animal waste is an important component of nutrient loads to the lake. Based upon watershed Animal Unit (1000 lbs live animal weight) estimates ranging from 25,471 (Felix, A., BSWCD, 2009) to 37,076 (GIS data) and unit waste loading factors ranging from 12 kg P / AU-year for dairy cows to 54 kg P/ AU-year for poultry (NRCS, 1995), the amount of phosphorus in animal waste generated and cycled on the farms is approximately 132-192 times the existing long-term-average P load reaching the lake (6,292 kg/yr, Table 2). Considering that this does not account for fertilizer P, only a small fraction of the P associated with agricultural operations would have to be transported in runoff to the lake in order to account for a significant portion of the total load. Figure 14 shows AU densities and manure P production expressed per unit of cropland in each watershed relative to guidance values developed for managing farm phosphorus balances in Vermont. These inventories can be refined with additional site-specific information on AU densities and manure management in each basin.

Phosphorus loading controls can be implemented on an incremental, cost-effective basis and tracked relative to the interim and TMDL goals developed above. Abundant technology, guidance, and statewide management programs exist to support design of BMPs for reducing phosphorus loads from cropland and feedlots (Sharpley et al., 2006; NRDC, 2010). The long-term strategy involves farm management to minimize excess phosphorus (fertilizer + animal feed – crop export – animal export), which eventually builds up on the soils or is transported to the lake. While transport is generally considered to occur primarily in surface runoff, sub-surface flows are expected to become increasingly important as soluble P concentrations build up in soils subject to excess P applications (Schippers et al., 2006; Sharpley et al, 2003). Farm-scale and watershed-scale phosphorus budgets guided by soil testing can be used as a basis for managing excess phosphorus and buildup of soluble P in the soils; this type of program could be coupled with traditional BMPs to reduce surface runoff and phosphorus transport from feedlots and cropland. As a component of the margin of safety, additional measures can be taken to reduce P sources in the lakeshed (septic tanks and runoff from shoreline lots, highways and other impervious surfaces).

Figure 14 Animal Unit Densities in LRL Watershed



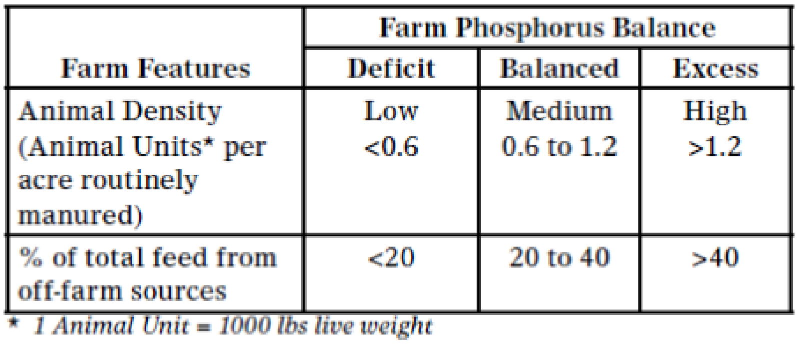
Animal Unit data from GIS Layer (bmms_FL-P_mn009)

Animal Unit densities expressed per acre of total cropland.

NRCS (1995)	Dairy	Beef	Swine	Poultry	Other
Lbs - TP / AU - Yr	26	40	58	119	26

Guidance values for AU Densities to Manage Farm P Balance, Vermont

http://www.sera17.ext.vt.edu/Documents/BMP_phosphorus_balance.pdf



Farm Features	Farm Phosphorus Balance		
	Deficit	Balanced	Excess
Animal Density (Animal Units* per acre routinely manured)	Low <0.6	Medium 0.6 to 1.2	High >1.2
% of total feed from off-farm sources	<20	20 to 40	>40

* *1 Animal Unit = 1000 lbs live weight*

If significant improvements in lake water quality are not achieved within a few years after significant reductions in P load are accomplished, application of alum or other chemicals would help to accelerate recovery by trapping historical P loads in the lake sediments. Incremental reductions in phosphorus and turbidity may promote growth of aquatic vegetation, which would help to stabilize the sediments and accelerate recovery (Heiskary & Lindon, 2005). Vegetation management programs should consider the possibility that excessive herbicide applications would make it more difficult to achieve water quality standards by promoting recycling of P loads from bottom sediments.

Depletion of stream base flow resulting from increased groundwater pumping for irrigation has been identified as a management concern for Little Rock Creek (BSWCD, 2009). Lower summer inflows resulting from drought and/or groundwater pumping could have adverse impacts on lake water quality through various mechanisms. Lower inflows would provide less dilution for P recycled from the lake bottom sediments and accelerate the buildup of P in the water column and algal blooms, as observed in 2007. Development of stagnant conditions could induce backflow and associated phosphorus loads from the outlet channel in periods when evaporation exceeds the total inflow from the tributaries and rainfall. The predominance of bluegreen algae could be enhanced by decreases in summer nitrate loads, potentially significant because of the high nitrate concentrations in summer base flows (NOX-N ~ 5 to 10 ppm). Nitrate loads could have beneficial impacts by oxidizing bottom sediments and decreasing P recycling. The mechanisms and scales are recommended for further evaluation supported by results of the ongoing watershed modeling study (BSWCD, 2009) and future monitoring.

Conclusions

1. Historical data indicate that LRL summer mean TP concentrations increased from ~125 ppb in 1979-1981 to ~270 ppb in 2006-2008, as compared with the 60 ppb water quality standard. Corresponding increases in chlorophyll-a and decreases in transparency were observed. While it is clear that significant reductions in TP concentration are required to achieve the standard, interpretation of the historical water quality deterioration is complicated by climatologic variations, data limitations, potential long-term effects of P buildup in the lake sediments, and potential trends in land use types and intensities.

2. Lake TP concentrations are highly correlated with chlorophyll-a levels, Secchi depths, and user perceptions of aesthetic qualities and suitability for recreational use. Algal blooms in LRL are highly responsive to variations in watershed P loads, recycling of historical P loads from bottom sediments, and climate. Toxic bluegreen algal blooms and noxious hydrogen sulfide odors were observed in 2007, when spring runoff contained the highest TP concentrations and loads. High concentrations of other nutrients and fecal coliforms indicate that animal waste was an important source. The blooms were likely accelerated later in the summer by low inflows and warm temperatures.

3. Modeling results indicate that achieving the 60 ppb lake TP standard would require reducing the tributary flow-weighted-mean concentrations to 83 ppb or less. Reductions in load relative to existing conditions range from 54% to 69% for the individual tributaries, although these estimates could vary considerably because of uncertainty in baseline loads derived from the 2006-2008 data.

4. Despite uncertainty in forecasting the ultimate lake responses to reducing external loads as prescribed by the TMDL, achieving incremental reductions in TP load and lake concentrations over the course of TMDL implementation are expected to provide incremental reductions in algal bloom severity and increases in transparency that would be perceptible by lake users.

5. Continued monitoring is essential to improve the TMDL baseline and track lake responses to implementation of P loading controls over a range of climatologic conditions.

6. Adaptive implementation of the TMDL is necessary, given the uncertainties associated with predicting the time scales and ultimate responses to load reductions. The opportunity to revise the lake goal and/or load allocation in the future based upon additional data and model refinements will reduce the uncertainties and provide greater assurance that lake management goals will eventually be achieved.

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Appendix
Development of Phosphorus TMDL for Little Rock Lake, Minnesota

A-1

Approximate Water & Mass Balances, March-October 2006-2008

Term	Area km^2	Flow hm^3	Flow-Weighted Load kg	Conc ppb	Std Error	Samples	Runoff cm	Unit Area Load kg/km^2	Percent of Total Lake Inflow % Area	% Flow	% Load	Flow cfs
Little Rock Creek Gauged Sub-Basins												
Bunker Hills	50.51	3.2	927	287	0.20	38	6.4	18	19%	12%	18%	8.1
LRC North - CR40	104.50	8.1	1292	160	0.13	46	7.7	12	39%	29%	25%	20.3
Sum of Gauged	155.01	11.3	2218	196		84	7.3	14	58%	41%	44%	28.4
Lake Inflows												
LR Creek - CH12	178.25	16.1	2958	184	0.18	47	9.0	17	67%	58%	58%	40.4
Zuleger	48.03	4.3	1148	265	0.16	44	9.0	24	18%	16%	23%	10.9
Sucker **	11.20	2.2	403	182	0.08	25	19.8	36	4%	8%	8%	5.6
Total Gauged	237.48	22.7	4510	199	0.12	116	9.5	19	89%	82%	88%	56.9
Ungauged Local	25.17	2.3	418	184	0.25		9.0	17	9%	8%	8%	5.7
Total Watershed	262.64	24.9	4927	198	0.16		9.5	19	98%	90%	97%	62.6
Shoreline Septic Tanks			90		0.50						2%	
Total External	262.64	24.9	5018	201	0.16		9.5	19	98%	90%	98%	62.6
Rainfall	5.10	2.7	81	30	0.30		53.0	16	2%	10%	2%	6.8
Total	267.74	27.6	5099	185	0.16	116	10.3	19	100%	100%	100%	69.4
Evaporation		3.8					75.2			14%		9.6
Net Inflow	267.74	23.8	5099	214	0.16	116		19			100%	59.7
Outflow	267.74	23.6	3195	135	0.13	26	8.8	12	100%	85%	63%	59.2
Net Inflow - Outflow		0.21	1904		0.47						37%	
Initial Storage		11.51	1151	100								
Final Storage		11.72	1485	127								
Mean Storage		11.85	2416	204								
Storage Increase		0.21	334									
Net Retention		0.00	1570		0.57						31%	

Red Cells Are Input Values

Lake Area	5.1	km^2
Lake Mean Depth	2.4	m
Rainfall P	30	ppb
Length of Averaging Period	0.67	March-October
Hydraulic Resid Time	127	days

RSE *	Relative Std Error of Load & FWM estimate
Ungauged (Lakeshed)	Drainage Area Ratio; Flow & Load = 0.14 x LRC
Zuleger Flow (Ungauged)	Drainage Area Ratio; Flow = 0.27 x LRC
Net Inflow	Inflow + Rainfall - Evaporation
Lake Outflow Volume	Water Balance
Storage	Computed from lake volume and lake-mean concentrations
Lake Outlet P Conc	0.92 x Lake Mean Conc (Calibrated to 2008 data)
Net Retention	Net Inflow - Outflow - Increase in Storage
Precipitation	http://climate.umn.edu; NWS stations Near Rice
Evaporation	Regional monthly means. Van der Leen at al, Water Encyclopedia. 1990.
Missing Flows	< July 2006; Regression vs. LRC_CH40;
Load Calculations	Gauged Sites; Daily Time Step; Regress Conc vs. Flow & Season; Interpolate Residuals; Walker & Havens, 2003.

Computation of Shoreline Septic Tank Loads

Septic Tanks	300	
People/Tank	3	
Seasonal Load Factor	1	
Unit Load To Tank	0.66	kg/cap-yr
Total Source Load	475.2	kg
Functioning Tank Reduc	90%	
Percent Failing Tanks	10%	
Load to Lake Per Tank	0.30	kg
Total Load to Lake	90.3	kg/yr

* Relative standard error = standard error / predicted value; estimates do not reflect uncertainty resulting from data gaps, flow estimates, ungauged watersheds, and drought conditions
Loads and mass balances are at best approximations and not representative of long-term averages; additional data needed to refine estimates and provide baseline for TMDL implementation

** Runoff from Sucker Creek exceeds values measured in other watersheds (20 cs. 6-9 cm/yr). It is possible that this reflects inflow from the adjacent Mayhew Creek basin
Aerial photography and GIS hydrography layer indicate that these basins are connected by a drainage ditch, which is not reflected in the hydrologic unit boundary.

A-2

Water & Mass Balances for 2006-2008 & 1991-2009 Baseline Scenarios

2006-2008							Load
Source	Area km^2	Flow hm^3	Load kg	Conc ppb	Runoff cm	Load kg/km^2	% Total
Bunker Hills	50.5	3.2	927	287	6.4	18.3	18%
LRC North - CR40	104.5	8.1	1292	160	7.7	12.4	25%
Lake Inflows							
LR Creek - CH12	178.2	16.1	2958	184	9.0	16.6	58%
Zuleger	48.0	4.3	1148	265	9.0	23.9	23%
Sucker	11.2	2.2	403	182	19.8	36.0	8%
Total Gauged	237.5	22.7	4510	199	9.5	19.0	88%
Ungauged Local	25.2	2.3	418	184	9.0	16.6	8%
Total Watershed	262.6	24.9	4927	198	9.5	18.8	97%
Shoreline Septic Tanks			90				2%
Total External	262.6	24.9	5018	201	9.5	19.1	98%
Rainfall	5.1	2.7	81	30	53.0	15.9	2%
Total	267.7	27.6	5099	185	10.3	19.0	100%
Evaporation		3.8					
Net Inflow	267.7	23.8	5099	214	8.9	19.0	100%

TMDL Baseline Conditions, 1991-2009 Hydrology							Load
Source	Area km^2	Flow hm^3	Load kg	Conc ppb	Runoff cm	Load kg/km^2	% Total
Bunker Hills	50.5	4.4	1267	287	8.7	25.1	18%
LRC North - CR40	104.5	11.1	1766	160	10.6	16.9	26%
Lake Inflows							
LR Creek - CH12	178.2	22.0	4046	184	12.3	22.7	58%
Zuleger	48.0	5.9	1570	265	12.3	32.7	23%
Sucker	11.2	3.0	551	182	27.1	49.2	8%
Total Gauged	237.5	31.0	6167	199	13.0	26.0	89%
Ungauged Local	25.2	3.1	571	184	12.3	22.7	8%
Total Watershed	262.6	34.1	6739	198	13.0	25.7	97%
Shoreline Septic Tanks			90				1%
Total External	262.6	34.1	6829	200	13.0	26.0	99%
Rainfall	5.1	3.1	94	30	61.6	18.5	1%
Total	267.7	37.2	6923	186	13.9	25.9	100%
Evaporation		3.8					
Net Inflow	267.7	33.4	6923	207	12.5	25.9	100%

Adjustment of 2006-2008 Hydrology to 1991-2009 Baseline

March-October	2006-2008	1991-2009	Ratio	
Elk River Runoff cm	8.1	11.9	1.46	
Sauk River Runoff cm	11.0	14.0	1.28	Mean of Elk & Sauk = 1.37
LRL Runoff cm	9.5	13.0	1.37	
Precipitation @ Rice cm	53.0	61.6	1.16	

A-3 Water & Mass Balances for TMDL & Interim Scenarios

TMDL Conditions	Tributary TP = 83 ppb					Unit Area	Load	Load
Source	Area km^2	Flow hm^3	Load kg	Conc ppb	Runoff cm	Load kg/km^2	% Total	% Reduc
Bunker Hills	50.5	4.4	366	83	8.7	7.2	11%	71%
LRC North - CR40	104.5	11.1	919	83	10.6	8.8	28%	48%
Direct Lake Inflows								
LR Creek - CH12	178.2	22.0	1827	83	12.3	10.2	56%	55%
Zuleger	48.0	5.9	492	83	12.3	10.2	15%	69%
Sucker	11.2	3.0	252	83	27.1	22.5	8%	54%
Total Gauged	237.5	31.0	2571	83	13.0	10.8	79%	58%
Lakeshed	25.2	3.1	571	184	12.3	22.7	18%	0%
Total Watershed	262.6	34.1	3142	92	13.0	12.0	97%	53%
Shoreline Septic Tanks *			0				0%	
Stormwater **			2				0%	
Total External	262.6	34.1	3144	92	13.0	12.0	97%	54%
Rainfall	5.1	3.1	94	30	61.6	18.5	3%	0%
Total	267.7	37.2	3238	87	13.9	12.1	100%	53%
Evaporation		3.8						
Net Inflow	267.7	33.4	3238	97	12.5	12.1	100%	53%
Outflow	267.7	33.4	2271	68	12.5	8.5	70%	53%

Predicted Lake Water Quality	Predicted	10%	90%	Standard
Total Phosphorus ppb	60	42	85	60
Chlorophyll-a ppb	18	10	32	20
Secchi Depth m	1.08	0.79	1.47	1.0

Interim Goal	Tributary TP = 120 ppb					Unit Area	Load	Load
Source	Area km^2	Flow hm^3	Load kg	Conc ppb	Runoff cm	Load kg/km^2	% Total	% Reduc
Bunker Hills	50.5	4.4	529	120	8.7	10.5	12%	58%
LRC North - CR40	104.5	11.1	1328	120	10.6	12.7	30%	25%
Direct Lake Inflows								
LR Creek - CH12	178.2	22.0	2641	120	12.3	14.8	59%	35%
Zuleger	48.0	5.9	712	120	12.3	14.8	16%	55%
Sucker	11.2	3.0	364	120	27.1	32.5	8%	34%
Total Gauged	237.5	31.0	3717	120	13.0	15.7	83%	40%
Lakeshed	25.2	3.1	571	184	12.3	22.7	13%	0%
Total Watershed	262.6	34.1	4288	126	13.0	16.3	96%	36%
Shoreline Septic Tanks			90				2%	
Total External	262.6	34.1	4379	128	13.0	16.7	98%	36%
Rainfall	5.1	3.1	94	30	61.6	18.5	2%	0%
Total	267.7	37.2	4473	120	13.9	16.7	100%	35%
Evaporation		3.8		0				
Net Inflow	267.7	33.4	4473	134	12.5	16.7	100%	35%
Outflow	267.7	33.4	3137	94	12.5	11.7	70%	35%

Predicted Lake Water Quality	Mean	10%	90%	Standard
Total Phosphorus ppb	78	55	111	60
Chlorophyll-a ppb	24	14	42	20
Secchi Depth m	0.88	0.64	1.19	1.0

Model Equations:

Q = Net Inflow hm^3

L = TP Load kg

PI = Avg Inflow Conc ppb = L / Q

f = fraction of year, march-oct = 0.67

A = Lake Area = 5.1 km^2

Z = Mean Depth = 2.4 m

T = Hydraulic Resid Time years = A Z f / Q

SE = prediction standard error, ln-transformed, typical value for empirical models (Walker,1985)

F10 = Scale Factor, 10th Percentile = EXP (-1.28 SE)

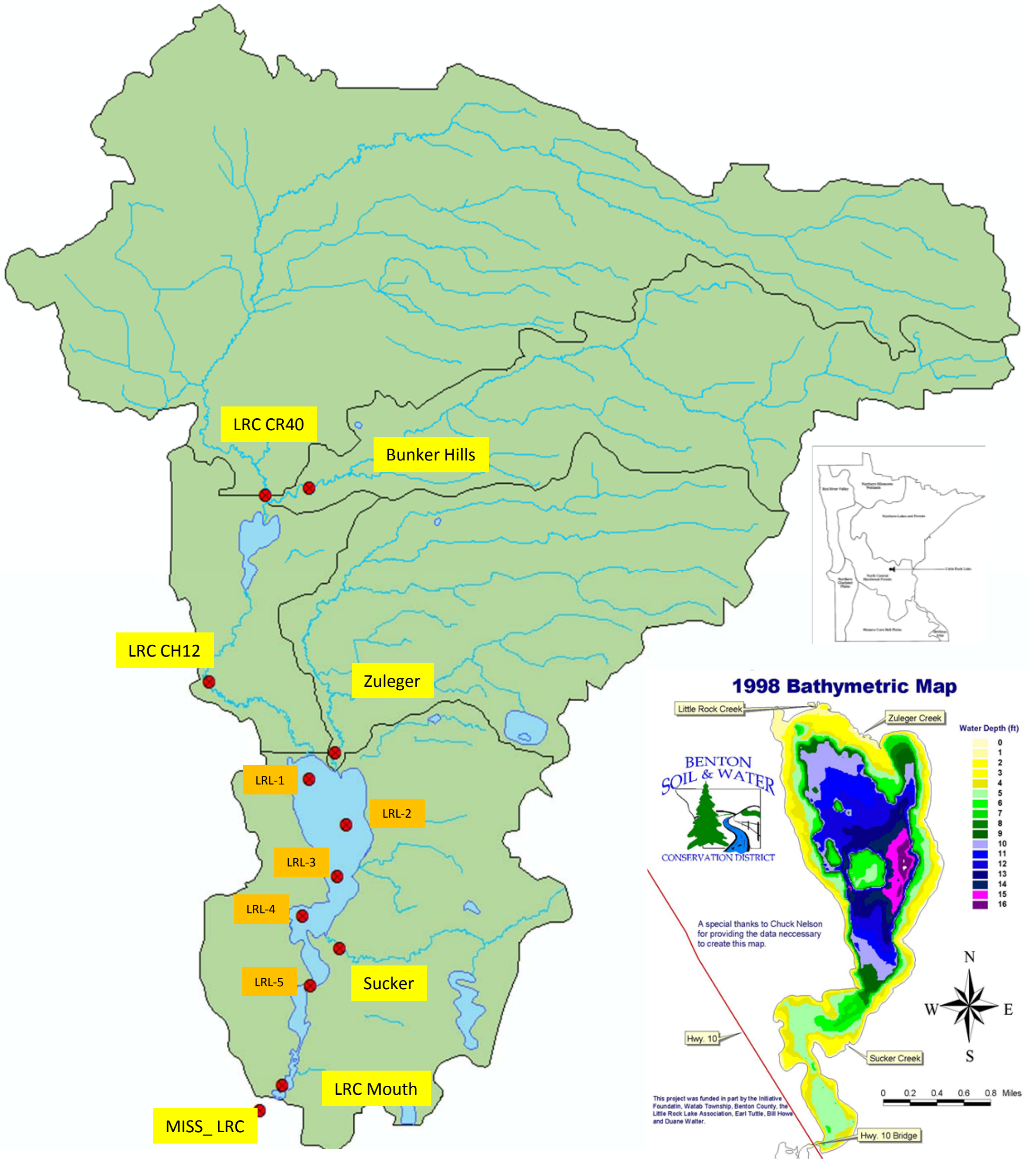
F90 = ErrorScale Factor, 90th Percentile = EXP (+ 1.28 SE)

Predicted Values	Equation	SE	F10	F90	Reference
P = Lake TP ppb	$PI \ / \ [\ 1 + 0.162\ T^{0.542}\ PI^{0.458}\ ]$	0.272	0.71	1.42	Canfield/Bachman (1981) Lake Model
Chla = Chlorophyll-a ppb	$\log10(Chl\text{-}a) = 1.08 \log10(TP) - 0.66$	0.437	0.57	1.75	Shallow Lakes (Heiskary & Wilson, 2008, Eq 4)
S = Secchi Depth m	$26.985\ P^{-0.7861}$	0.242	0.73	1.36	Shallow Lakes (Heiskary & Lindon, 2005, Fig 13)

* Direct discharge from septic systems is not permitted under MN state law; therefore the TMDL allocation for septic systems is set to zero .

** Allowance for stormwater loads associated with future urban development set a 0.05% of the TMDL (2 kg)

A-4 Lake & Watershed Monitoring Sites



ALIAS	STORET	DESCRIPTION	LAT	LONG	TYPE
BUNKER	S004-063	BUNKER HILL CR AT CR 56, 4 MI NE OF RICE, MN	45.803	-94.176	Stream
LRCR_CR40	S004-062	LITTLE ROCK CR AT CR 40, 3.5 MI NE OF RICE, MN.	45.801	-94.189	Stream
LRCR_CH12	S004-061	LITTLE ROCK CR AT CSAH 12, 1 MI NE OF RICE, MN	45.764	-94.205	Stream
SUCKER	S004-064	SUCKER CR AT SUCKER CR RD, 3.8 MI SE OF RICE, MN	45.711	-94.165	Stream
ZULEGER	S002-447	ZULEGER CR AT CSAH-2. 2.5 MI E OF RICE, MN	45.750	-94.167	Stream
LRC_MOUTH	S005-004	LITTLE R CK AT HARRIS CHANNEL, 4.5 MI NE OF SARTELL	45.684	-94.182	Stream
MISS_LRC	S004-320	MISSISSIPPI RIVER ABOVE LRC	45.679	-94.188	Stream
LRL_1	205	LITTLE ROCK LAKE (05-0013)	45.745	-94.174	Lake
LRL_2	204	LITTLE ROCK LAKE (05-0013) Deepest Point	45.736	-94.163	Lake
LRL_3	209	LITTLE ROCK LAKE (05-0013)	45.726	-94.166	Lake
LRL_4	211	LITTLE ROCK LAKE (05-0013)	45.718	-94.176	Lake
LRL_5	212	LITTLE ROCK LAKE (05-0013) Above Lake Outlet	45.704	-94.173	Lake
LRL_DS	05-0012	UNNAMED (LITTLE ROCK CHAIN) 4.3 MI SE OF RICE	45.699	-94.176	Lake

A-5

Monthly Mean Water Quality Data by Site and Parameter, 2006-2008

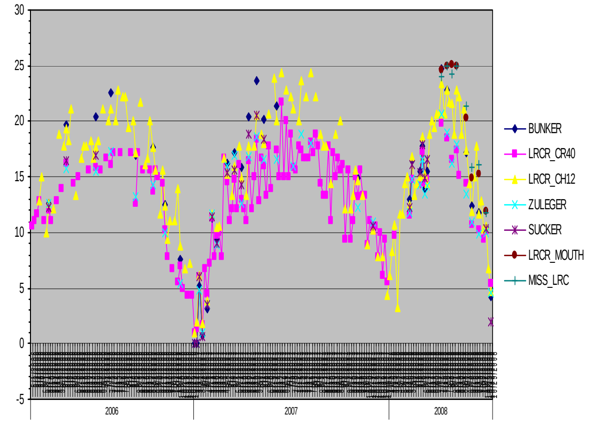
		Month								Spring	Summer	Year
Data	SITE	3	4	5	6	7	8	9	10	3-5	6-9	3-10
TP Samples	LRC_CR40	3	5	7	9	6	6	5	6	15	26	47
	BUNKER_HILL	3	4	7	9	3	2	5	5	14	19	38
	LRC_CH12	3	5	7	9	6	6	5	6	15	26	47
	ZULEGER	3	4	7	9	6	5	5	5	14	25	44
	SUCKER	3	4	7	8				3	14	8	25
	LAKE		1	2	3	14	13	16	11	3	46	60
	LRC_MOUTH					2	2	3	1	0	7	8
	MISS_LRC					2	2	3	1	0	7	8
TP ppm	LRC_CR40	0.64	0.09	0.07	0.10	0.05	0.04	0.06	0.05	0.26	0.06	0.14
	BUNKER_HILL	1.13	0.13	0.08	0.18	0.04	0.14	0.14	0.09	0.45	0.13	0.24
	LRC_CH12	0.81	0.10	0.06	0.15	0.07	0.06	0.06	0.06	0.32	0.08	0.17
	ZULEGER	1.09	0.17	0.13	0.23	0.14	0.11	0.10	0.07	0.46	0.14	0.25
	SUCKER	0.54	0.11	0.13	0.19				0.07	0.26	0.19	0.21
	LAKE		0.12	0.16	0.18	0.32	0.42	0.31	0.15	0.14	0.31	0.24
	LRC_MOUTH					0.16	0.16	0.17	0.09		0.16	0.14
	MISS_LRC					0.05	0.05	0.03	0.03		0.04	0.04
SRP ppm	LRC_CR40	0.44	0.04	0.03	0.04	0.03	0.03	0.04	0.03	0.17	0.03	0.08
	BUNKER_HILL	0.73	0.08	0.04	0.06	0.02	0.12	0.11	0.07	0.28	0.08	0.15
	LRC_CH12	0.54	0.04	0.02	0.04	0.04	0.03	0.03	0.03	0.20	0.04	0.10
	ZULEGER	0.77	0.11	0.08	0.11	0.09	0.08	0.07	0.06	0.32	0.09	0.17
	SUCKER	0.46	0.04	0.07	0.11				0.05	0.19	0.11	0.15
	LAKE		0.02	0.01	0.03	0.13	0.17	0.09	0.03	0.01	0.10	0.07
	LRC_MOUTH					0.03	0.06	0.01	0.01		0.03	0.03
	MISS_LRC					0.02	0.01	0.01	0.01		0.01	0.01
TKN ppm	LRC_CR40	5.86	1.67	0.89	1.18	0.42	0.41	0.50	0.82	2.80	0.63	1.47
	BUNKER_HILL	7.10	1.89	1.48	1.56	0.89	1.05	1.09	1.38	3.49	1.15	2.05
	LRC_CH12	6.26	1.54	0.92	1.34	0.53	0.65	0.59	0.88	2.91	0.78	1.59
	ZULEGER	6.63	1.79	1.19	1.54	0.49	0.45	0.38	0.72	3.20	0.71	1.65
	SUCKER	3.73	1.67	1.08	1.53				0.45	2.16	1.53	1.69
	LAKE					2.86	2.97	2.97	1.99		2.93	2.70
	LRC_MOUTH					1.79	1.56	1.70	1.20		1.68	1.56
	MISS_LRC					0.74	0.72	1.41	0.60		0.95	0.87
NH4-N ppm	LRC_CR40	2.14	0.06	0.01	0.03	0.01	0.01	0.02	0.02	0.74	0.02	0.29
	BUNKER_HILL	2.25	0.07	0.02	0.04	0.01		0.02	0.03	0.78	0.02	0.35
	LRC_CH12	2.07	0.09	0.02	0.10	0.01	0.01	0.01	0.06	0.73	0.03	0.30
	ZULEGER	2.19	0.17	0.04	0.09	0.02	0.01	0.01	0.01	0.80	0.03	0.32
	SUCKER	1.13	0.16	0.02	0.07				0.04	0.44	0.07	0.28
	LAKE		0.10			0.11	0.10	0.13	0.18	0.10	0.12	0.13
	LRC_MOUTH					0.01	0.03	0.01	0.01		0.02	0.02
	MISS_LRC					0.01	0.03	0.02	0.01		0.02	0.02
NOX-N ppm	LRC_CR40	2.43	2.82	5.29	4.03	10.71	10.48	7.77	6.12	3.51	8.25	6.21
	BUNKER_HILL	1.22	2.78	3.77	4.22	1.58	3.47	6.43	6.21	2.59	3.92	3.71
	LRC_CH12	2.43	3.12	4.84	3.47	6.06	6.27	6.41	5.70	3.47	5.55	4.79
	ZULEGER	1.47	3.03	4.31	4.46	4.28	3.55	3.66	5.37	2.94	3.99	3.77
	SUCKER	0.84	0.70	0.21	0.87				0.03	0.58	0.87	0.53
	LAKE		1.50			0.02	0.02	0.03	0.22	1.50	0.02	0.36
	LRC_MOUTH					0.05	0.01	0.01	0.01		0.02	0.02
	MISS_LRC					0.19	0.09	0.09	0.18		0.12	0.14
TSS ppm	LRC_CR40	16.93	3.47	3.40	2.27	3.30	1.65	2.32	1.84	7.93	2.38	4.40
	BUNKER_HILL	14.87	5.33	4.40	3.07	1.47	9.00	3.12	5.52	8.20	4.16	5.85
	LRC_CH12	20.17	6.27	4.02	4.27	2.07	2.25	2.56	1.72	10.15	2.79	5.41
	ZULEGER	19.93	11.73	2.30	3.20	2.13	2.50	4.52	1.54	11.32	3.09	5.98
	SUCKER	6.13	3.00	2.83	2.10				1.27	3.99	2.10	3.07
	LAKE					44.00	66.75	58.50	21.00		56.42	47.56
	LRC_MOUTH					21.50	24.00	31.00	12.00		25.50	22.13
	MISS_LRC					4.60	4.60	3.07	3.20		4.09	3.87
BOD ppm	LRC_CR40	18.43	1.25	0.88	0.77	1.00	1.00	1.35	1.00	6.85	1.03	3.21
	BUNKER_HILL	24.47	1.45	1.22	2.13	1.00	1.00	1.75	1.30	9.05	1.47	4.29
	LRC_CH12	22.97	1.50	2.62	1.38	0.95	1.25	1.20	1.03	9.03	1.20	4.11
	ZULEGER	20.50	1.65	0.90	0.87	0.80	1.00	1.05	0.97	7.68	0.93	3.47
	SUCKER	11.37	1.63	1.84	0.95				1.10	4.94	0.95	3.38
	LAKE											
	LRC_MOUTH											
	MISS_LRC											

A-5 Monthly Mean Water Quality Data by Site and Parameter, 2006-2008

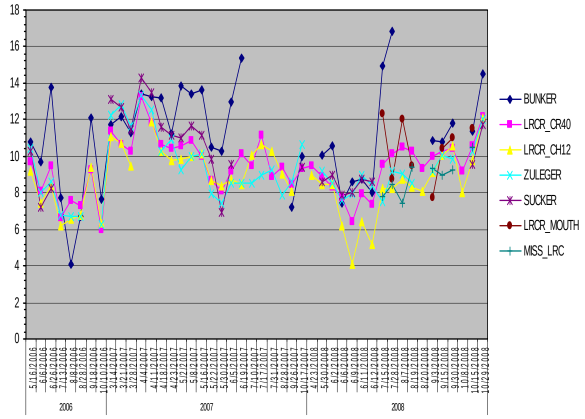
		Month								Spring	Summer	Year
Data	SITE	3	4	5	6	7	8	9	10	3-5	6-9	3-10
Turbidity NTU	LRC_CR40	9.80	4.74	3.47	6.01	3.63	1.73	1.59	2.91	6.00	3.24	4.24
	BUNKER_HILL	5.15	1.65	1.00	11.99	1.14	1.30	1.61	1.84	2.60	4.01	3.21
	LRC_CH12	11.23	5.54	7.07	5.08	1.68	2.52	2.17	3.53	7.95	2.86	4.85
	ZULEGER	10.23	3.20	3.31	19.69	1.96	2.16	2.13	1.76	5.58	6.48	5.56
	SUCKER	3.77	1.43	1.54	10.48				0.87	2.24	10.48	3.62
	LAKE			15.05	63.29	57.07	77.85	64.46	21.88	15.05	65.67	49.93
	LRC_MOUTH					25.04	26.43	22.50	8.72		24.66	20.67
	MISS_LRC					4.11	6.18	2.52	2.58		4.27	3.85
FCOLI cfu/100 ml	LRC_CR40	1215.33	25.00	146.67	440.00	234.00	656.67	260.00	75.00	462.33	397.67	381.58
	BUNKER_HILL	2016.67	2107.25	70.83	202.67	230.00	185.00	390.00	3220.00	1398.25	251.92	1052.80
	LRC_CH12	1654.33	1395.50	36.42	65.33	135.00	300.00	101.50	119.33	1028.75	150.46	475.93
	ZULEGER	876.67	256.25	274.00	233.33	262.50	270.00	321.00	80.67	468.97	271.71	321.80
	SUCKER	457.33	47.00	96.33	93.00				90.00	200.22	93.00	156.73
	LAKE											
	LRC_MOUTH											
	MISS_LRC											
CL ppm	LRC_CR40	13.33	15.75	13.75	11.27	12.65	12.88	13.06	12.75	14.28	12.47	13.18
	BUNKER_HILL	19.33	30.33	28.50	24.83	20.07	27.00	27.54	32.54	26.06	24.86	26.27
	LRC_CH12	16.00	17.83	15.00	12.43	13.05	13.32	13.50	14.97	16.28	13.08	14.51
	ZULEGER	19.00	28.67	20.25	16.07	11.25	11.28	13.36	18.31	22.64	12.99	17.27
	SUCKER	13.00	16.67	11.28	7.40				8.63	13.65	7.40	11.39
	LAKE		11.80			14.53	14.98	14.37	14.22	11.80	14.62	13.98
	LRC_MOUTH					9.01	10.50	12.23	12.60		10.58	11.08
	MISS_LRC					6.72	7.69	9.60	9.11		8.00	8.28
Temp deg-C	LRC_CR40	2.81	8.46	12.92	15.53	17.32	15.69	12.92	8.09	8.06	15.37	11.72
	BUNKER_HILL	1.71	6.17	16.98	17.85	23.39	17.30	13.75	8.17	8.28	18.07	13.16
	LRC_CH12	2.98	8.35	14.68	18.04	21.75	19.38	14.31	9.10	8.67	18.37	13.57
	ZULEGER	1.64	6.27	15.06	15.29	18.28	15.43	11.29	7.74	7.65	15.07	11.37
	SUCKER	2.01	6.14	15.59	16.58				7.62	7.91	16.58	9.59
	LAKE		11.64	15.35	20.69	24.80	24.30	17.52	9.89	13.49	21.83	17.74
	LRC_MOUTH					24.79	24.99	16.81	11.94		22.20	19.63
	MISS_LRC					24.45	24.61	17.79	11.82		22.28	19.67
DO ppm	LRC_CR40	10.75	11.11	9.56	8.29	9.31	9.07	9.69	9.44	10.47	9.09	9.65
	BUNKER_HILL	11.75	12.79	11.78	10.57	13.19	5.41	10.56	10.87	12.10	9.93	10.86
	LRC_CH12	10.44	10.24	9.24	7.03	8.94	7.93	9.45	9.07	9.97	8.34	9.04
	ZULEGER	12.23	11.77	9.18	8.34	8.36	7.76	9.41	9.80	11.06	8.47	9.61
	SUCKER	12.43	12.62	9.91	8.40				10.22	11.65	8.40	10.72
	LAKE		13.80	11.24	9.16	9.48	8.79	8.81	11.37	12.52	9.06	10.38
	LRC_MOUTH					10.54	10.75	9.70	11.54		10.33	10.63
	MISS_LRC					8.14	8.42	9.19	10.51		8.58	9.07
Cond umhos/cm	LRC_CR40	266.33	311.80	384.57	338.56	418.00	415.83	406.80	397.20	320.90	394.80	367.39
	BUNKER_HILL	262.00	359.25	461.86	369.89	394.67	456.50	476.20	501.50	361.04	424.31	410.23
	LRC_CH12	282.00	323.40	396.86	338.78	401.67	415.33	419.80	412.60	334.09	393.89	373.80
	ZULEGER	275.67	372.75	419.43	368.44	395.67	391.00	409.40	456.00	355.95	391.13	386.04
	SUCKER	202.00	268.75	291.71	259.50				315.00	254.15	259.50	267.39
	LAKE		260.00	260.50	268.50	292.66	291.12	319.57	305.85	260.25	292.96	285.46
	LRC_MOUTH					314.08	317.00	335.08	317.00		322.06	320.79
	MISS_LRC					375.25	335.92	360.50	353.00		357.22	356.17
pH	LRC_CR40	7.53	7.79	8.06	7.66	8.10	7.97	7.88	7.86	7.79	7.90	7.86
	BUNKER_HILL	7.43	7.94	8.40	8.01	8.73	7.68	7.83	7.89	7.92	8.06	7.99
	LRC_CH12	7.46	7.74	8.05	7.51	8.15	7.91	7.83	7.89	7.75	7.85	7.82
	ZULEGER	7.55	7.83	8.01	7.60	8.34	7.85	7.77	7.92	7.79	7.89	7.86
	SUCKER	7.49	7.89	7.81	7.27				7.40	7.73	7.27	7.57
	LAKE		9.22	9.30	8.85	9.24	9.26	8.76	8.92	9.26	9.03	9.08
	LRC_MOUTH					8.73	8.78	8.65	8.70		8.72	8.71
	MISS_LRC					8.15	8.26	8.29	8.04		8.23	8.19
Flow cfs	LRC_CR40	24.89	31.70	20.73	18.36	2.55	1.87	2.35	10.00	25.78	6.28	14.06
on Sample Day	BUNKER_HILL	11.33	16.91	2.57	25.28	0.34	0.35	0.79	1.61	10.27	6.69	7.40
	LRC_CH12	30.31	58.86	40.06	36.19	10.51	7.22	8.60	18.28	43.08	15.63	26.25
	ZULEGER	14.28	14.94	6.75	18.90	2.41	2.04	2.26	5.28	11.99	6.40	8.36
	SUCKER	6.83	8.05	1.05	26.83				2.94	5.31	26.83	9.14
Computed	LAKE		129.35	53.29	123.00	9.57	3.88	24.03	25.52	91.32	40.12	52.66
Computed	LRC_MOUTH					13.52	4.03	25.28	34.88		14.28	19.43
	MISS_LRC											

Stream Water Quality by Site & Date, 2006-2008

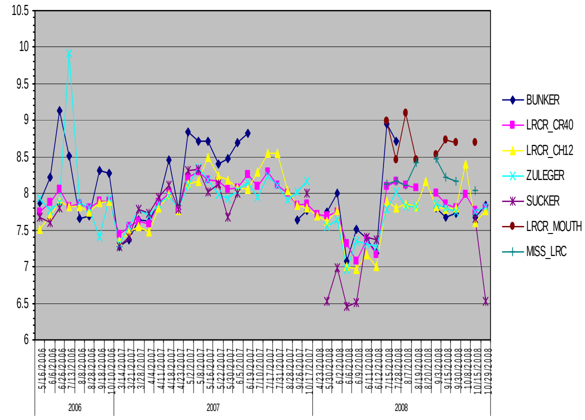
Temperature, water deg C



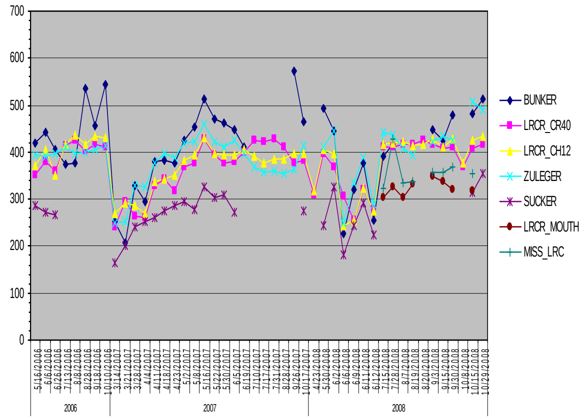
Dissolved oxygen (DO) mg/l



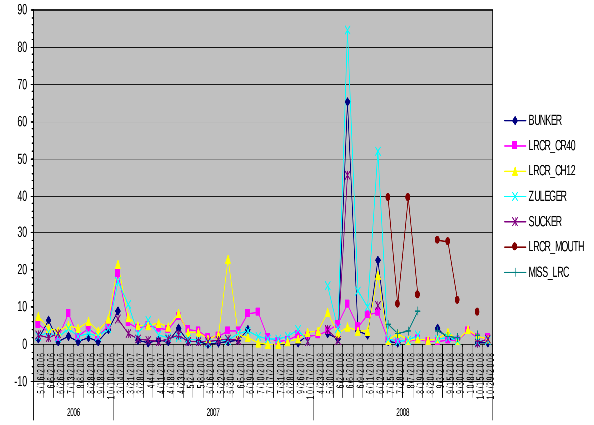
pH None



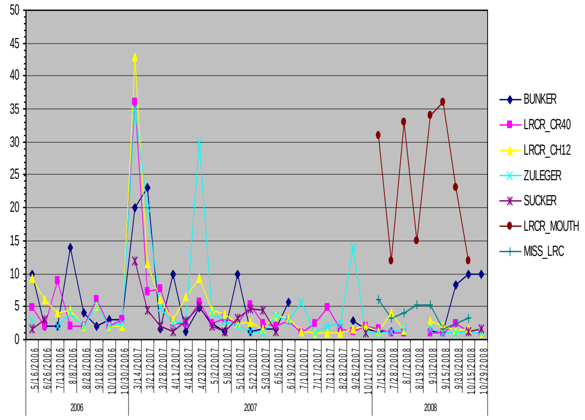
Specific conductance uS/cm



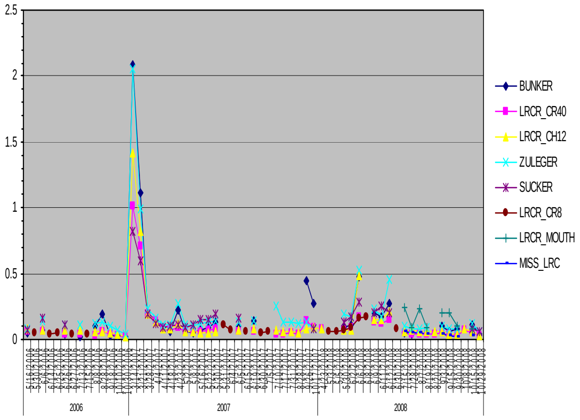
Turbidity NTRU



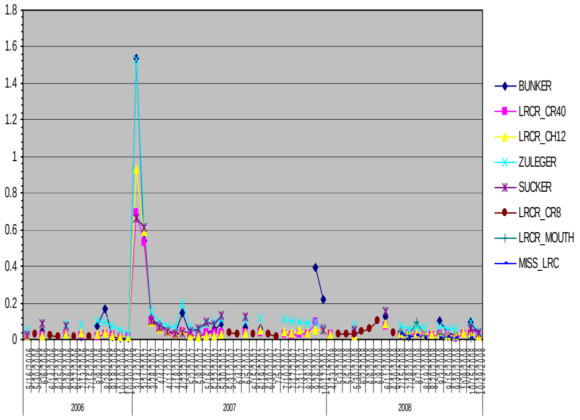
Solids, Total Suspended (TSS) mg/l



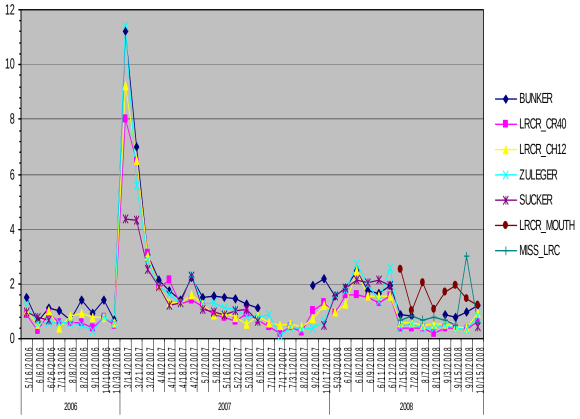
Phosphorus as P mg/l



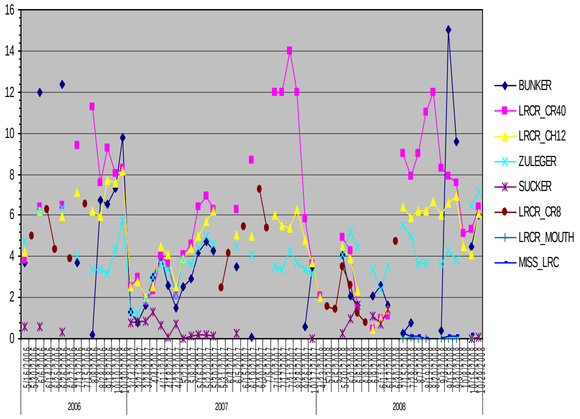
Phosphorus, orthophosphate as P mg/l



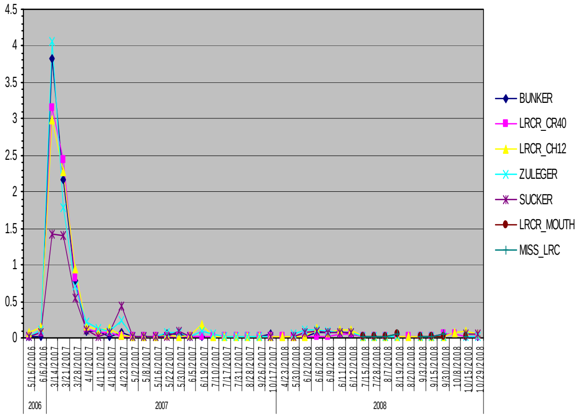
Nitrogen, Kjeldahl mg/l



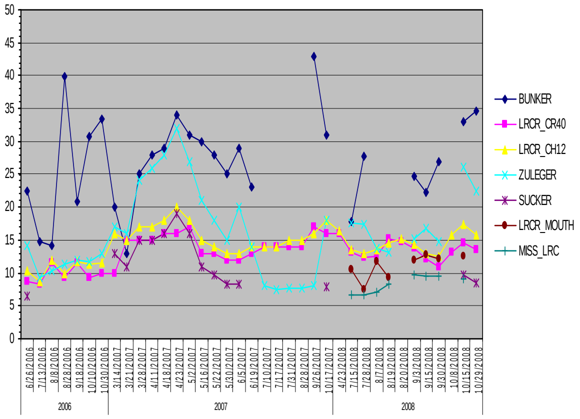
Nitrogen, Nitrite (NO2) + Nitrate (NO3) as N mg/l



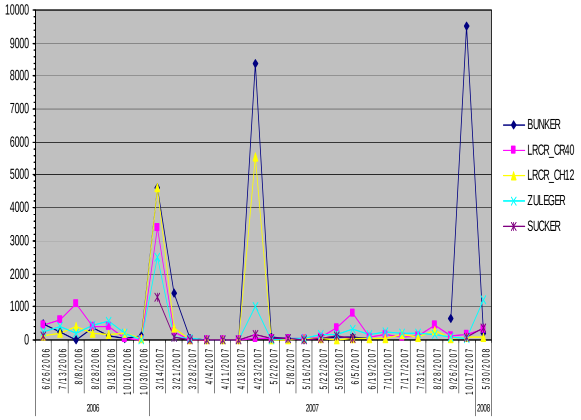
Nitrogen, ammonia (NH3) + ammonium (NH4) mg/l



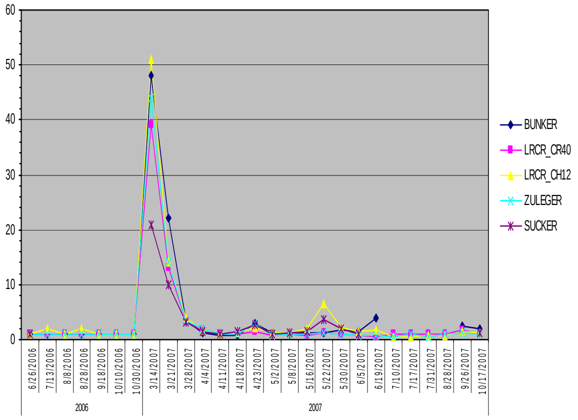
Chloride mg/l



Fecal Coliform cfu



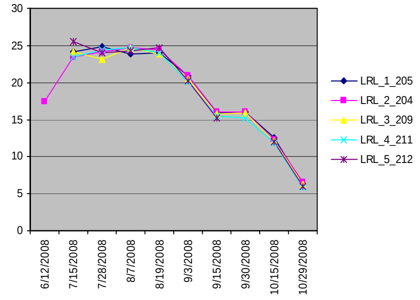
BOD, Biochemical oxygen demand mg/l



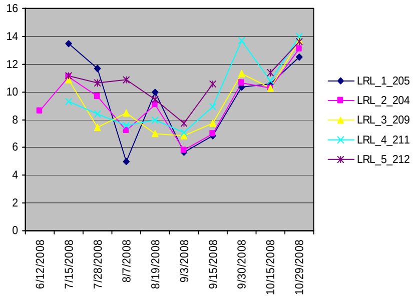
A-7 LRL Water Quality Time Series by Variable & Site, 2008

Surface Samples Stations Sorted in Downstream Order 2008

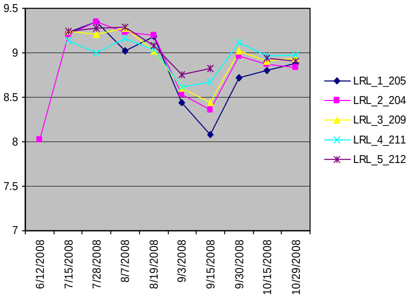
Temperature, water deg C Year: 2008



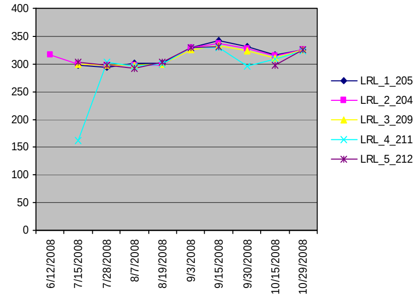
Dissolved oxygen (DO) mg/l Year: 2008



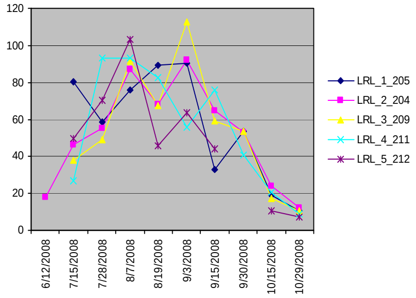
pH None Year: 2008



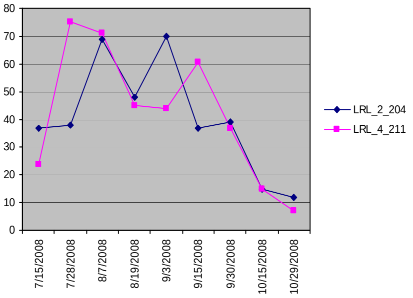
Specific conductance uS/cm Year: 2008



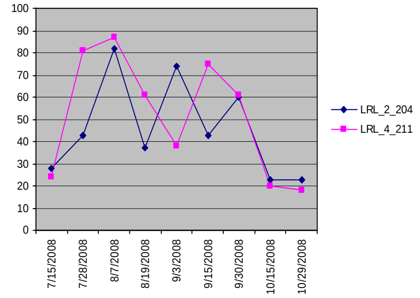
Turbidity - Probe FNU Year: 2008



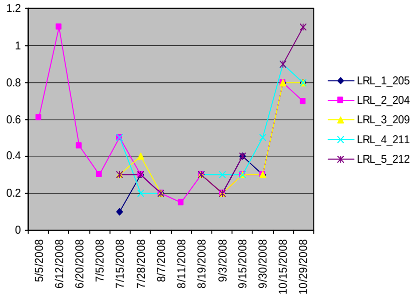
Turbidity NTRU Year: 2008



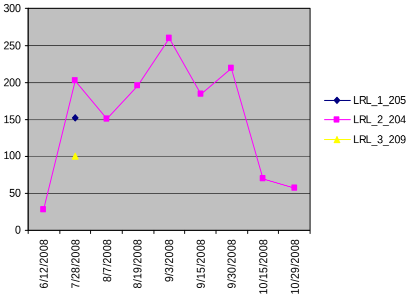
Solids, Total Suspended (TSS) mg/l Year: 2008



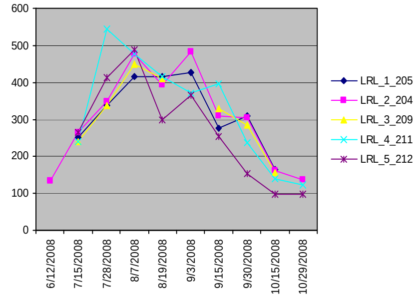
Depth, Secchi Disk Depth m Year: 2008



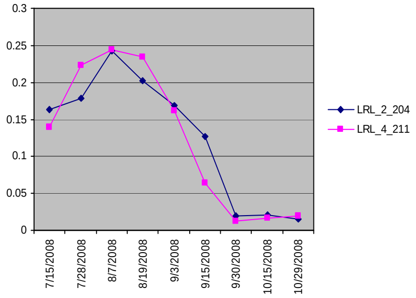
Chlorophyll a, corrected for pheophytin ug/l Year: 2008



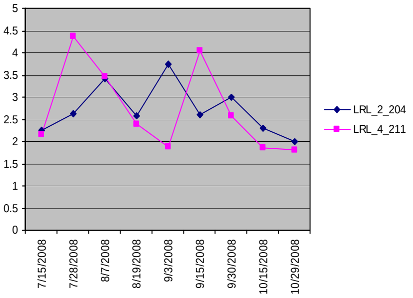
Phosphorus as P ppb Year: 2008



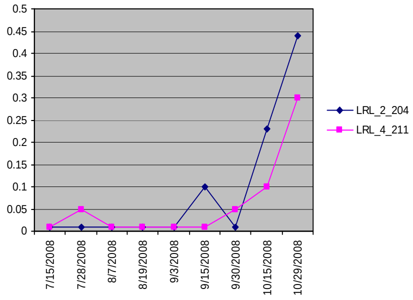
Phosphorus, orthophosphate as P mg/l Year: 2008



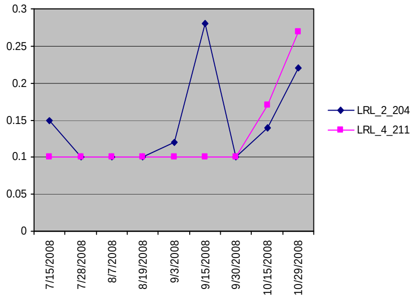
Nitrogen, Kjeldahl mg/l Year: 2008



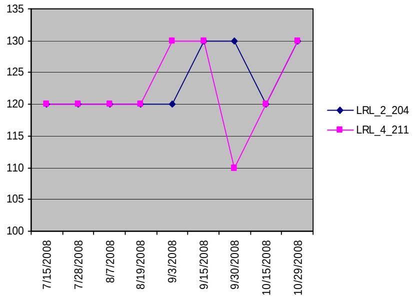
Nitrogen, Nitrite (NO2) + Nitrate (NO3) mg/l Year: 2008



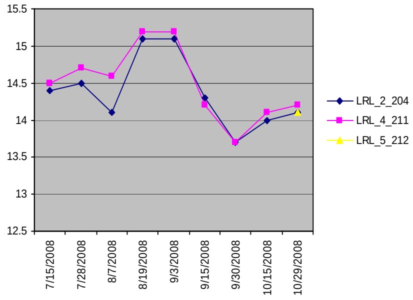
Nitrogen, ammonia (NH3) + ammonium mg/l Year: 2008



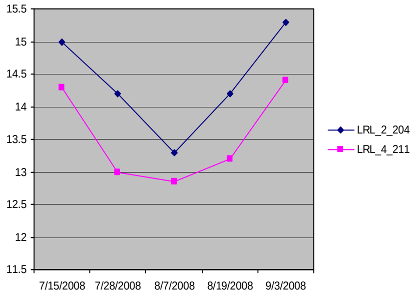
Alkalinity, Total (total hydroxide+carbon mg/l CaC Year: 2008



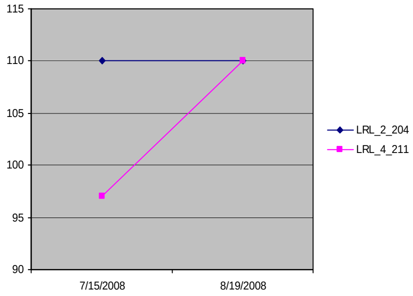
Chloride mg/l Year: 2008



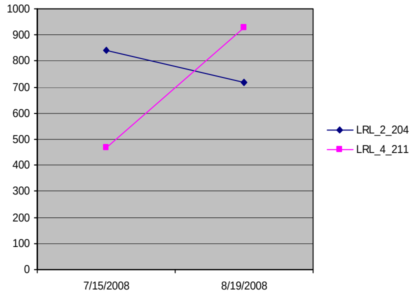
Sulfur, sulfate (SO4) as SO4 mg/l Year: 2008



Calcium mg/l Year: 2008

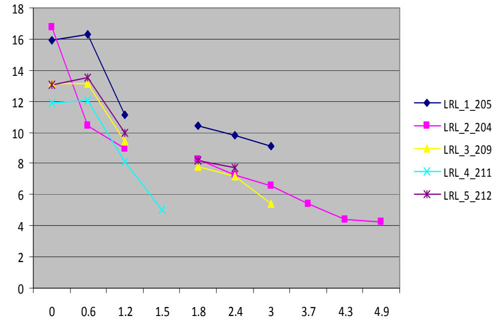


Iron ug/l Year: 2008

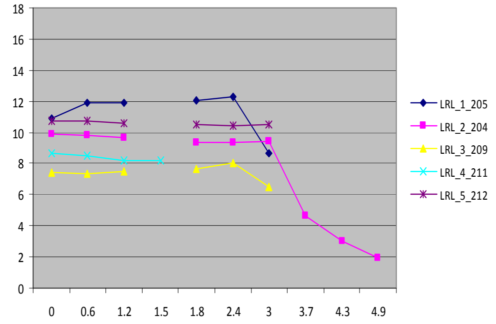


Vertical Profiles by Date & Lake Site, 2008 X-Axis = Depth meters Dissolved Oxygen (ppm)

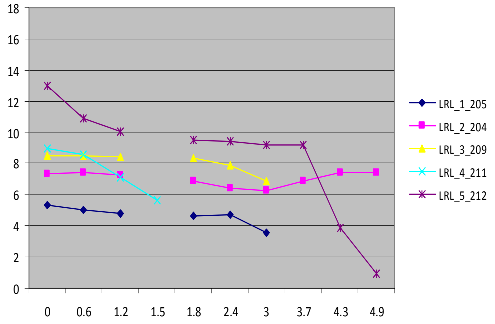
7/15/2008



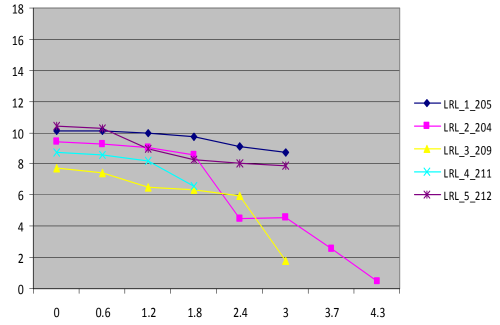
7/28/2008



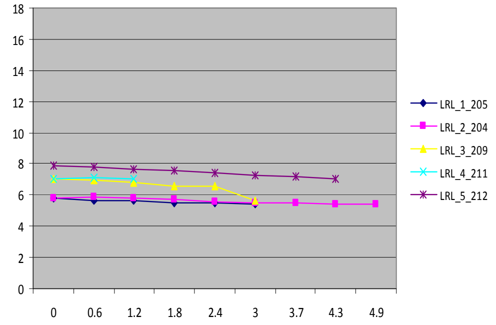
8/7/2008



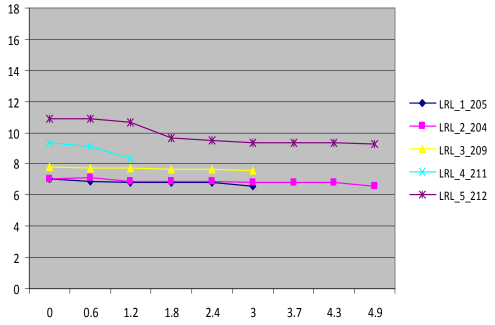
8/19/2008



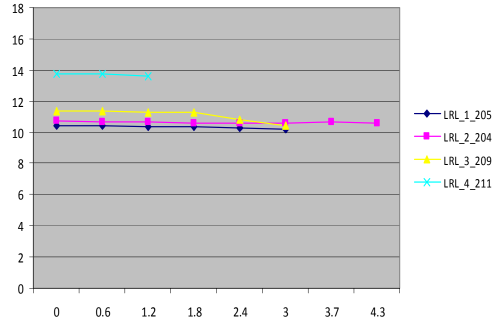
9/3/2008



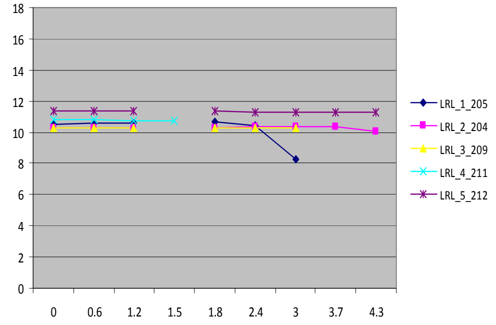
9/15/2008



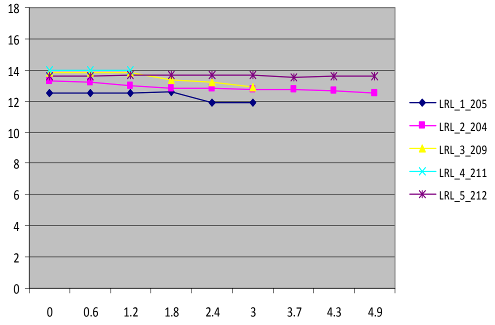
9/30/2008



10/15/2008

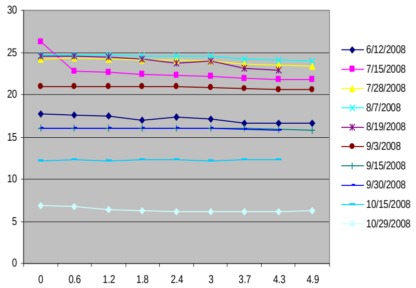


10/29/2008

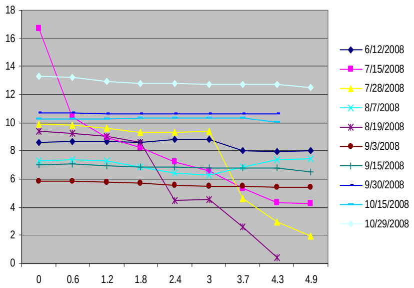


A-9 Vertical Profiles by Date & Parameter, 2008, Site 204 X-Axis = Depth (meters)

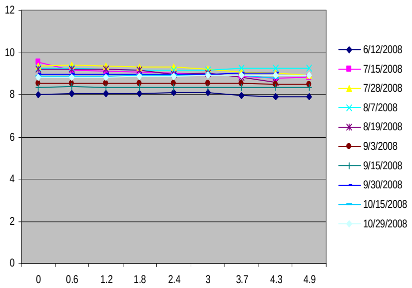
Temperature, water deg C



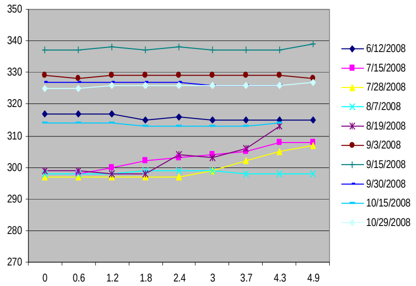
Dissolved oxygen (DO) mg/l



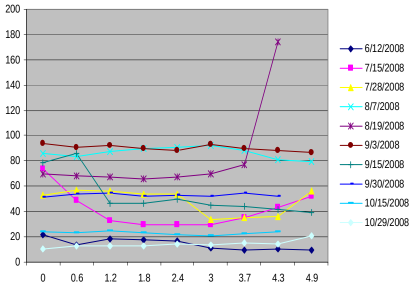
pH None



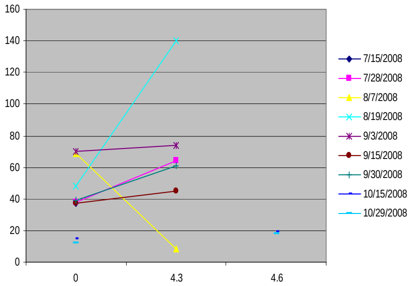
Specific conductance uS/cm



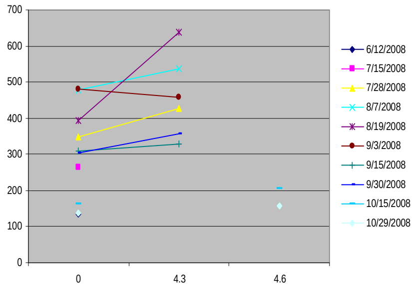
Turbidity - Probe FNU



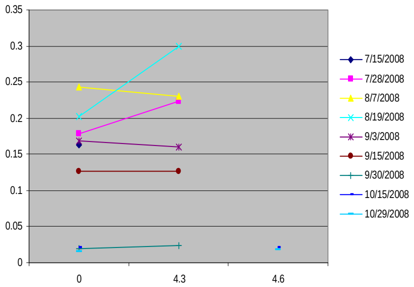
Turbidity NTRU



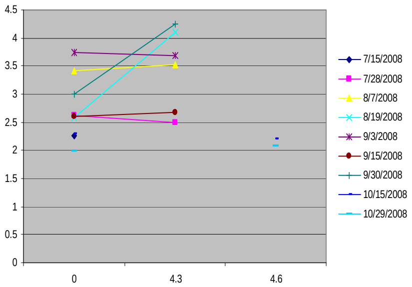
Phosphorus as P ppb



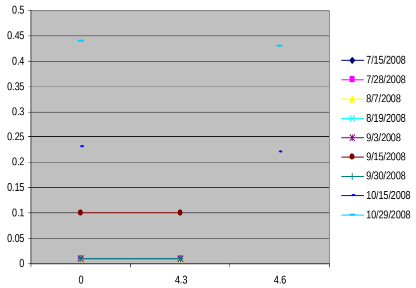
Phosphorus, orthophosphate as P mg/l



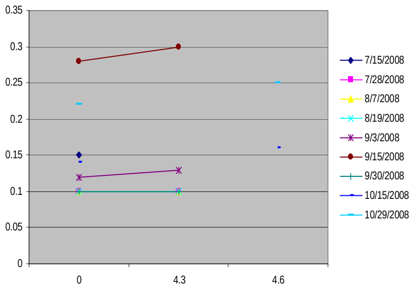
Nitrogen, Kjeldahl mg/l



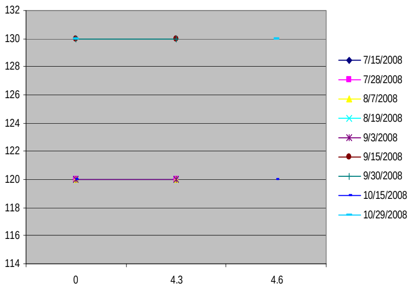
Nitrogen, Nitrite (NO2) + Nitrate (NO3) as N mg/l



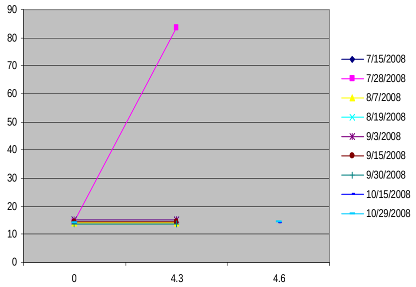
Nitrogen, ammonia (NH3) + ammonium (NH4) mg/l



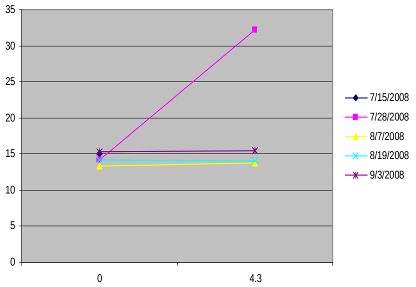
Alkalinity, Total (total hydroxide+carbonate+bic mg/l CaCO3



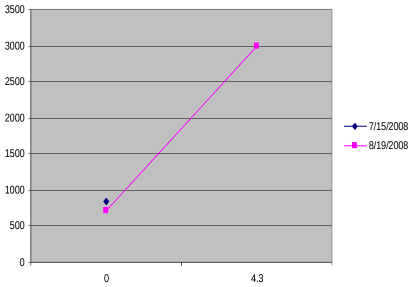
Chloride mg/l



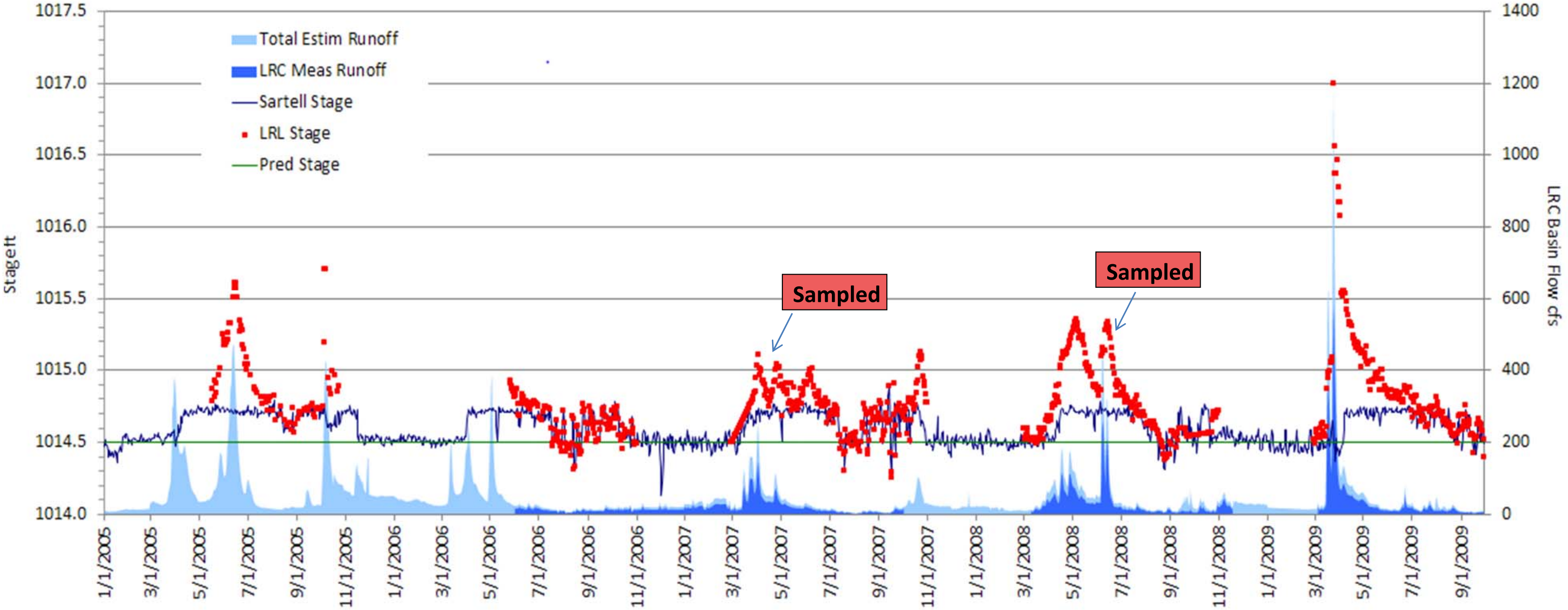
Sulfur, sulfate (SO4) as SO4 mg/l



Iron ug/l



A-10 Daily Stage & Flow Data for Mississippi River and Little Rock Lake, 2005-2009



Sartell & Little Rock Lake stage data provided by MDNR (D. Heneley, D. Lais) & MPCA (M. Evenson).

Little Rock Lake stage shown for March - October; datums adjusted to fit Sartell stage on days with low runoff (< 10 cfs, $r^2 > 0.95$) in 2005-2009.

MDRN lake stage data from lake water quality monitoring database; datum adjustment = -0.18 feet.

MPCA lake stage data from continuous monitor at Hwy 10 bridge in July 2006 - October 2009; datum adjustment = -0.53 feet.

LRC Basin total runoff to Little Rock Lake estimated from measured flows LRC CH12 & Sucker Creek (dark blue); missing values filled by correlations with LRC CR8 or Platte River.

Runoff from ungauged watersheds (Zuleger, lakeshed) estimated based upon drainage area ratios relative to LRC CH12

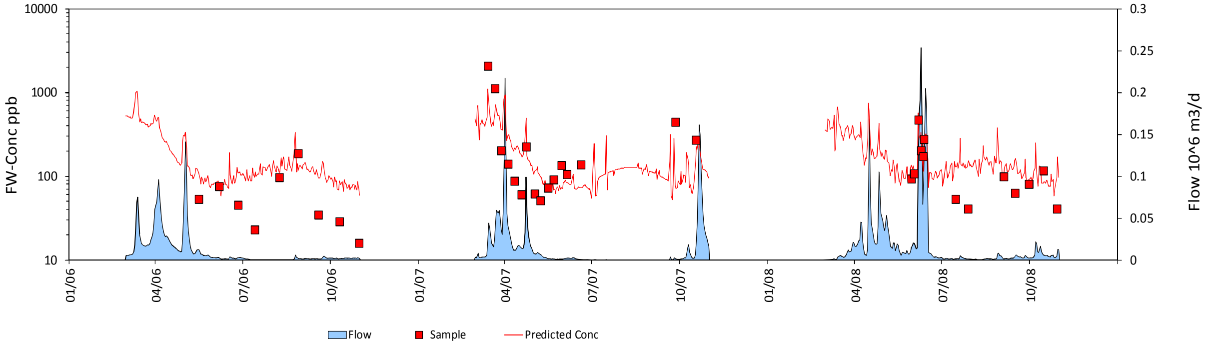
Red boxed indicate high-flow events when water quality samples were collected.

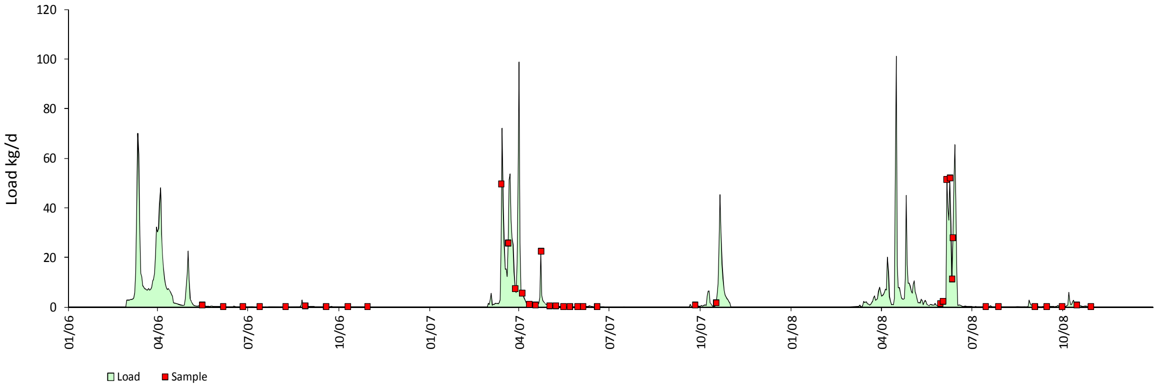
A-11 Flow & TP Load Time Series - Bunker Hills Creek

Site: BUNKER_HILL Bunker Hill Crk Variable: TP Total P ppb Months: 3 10 Dates: 01/01/2006 - 12/31/2008

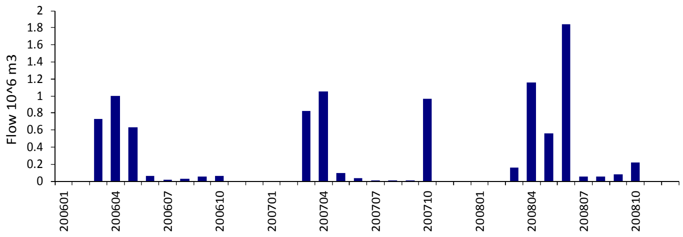
Flow Data: BUNKER_HILL Flow 10^6 m3/yr = 3.224 Load kg/yr = 926.7 FW-Conc ppb = 287.4 RSE = 20% Method Used= 5

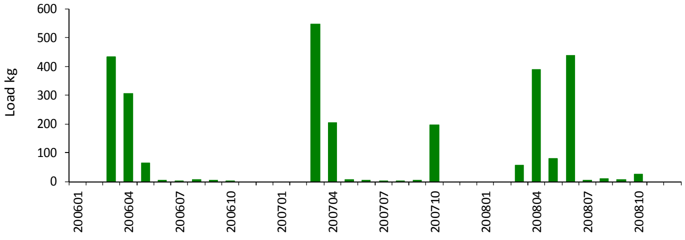
Daily Time Series:

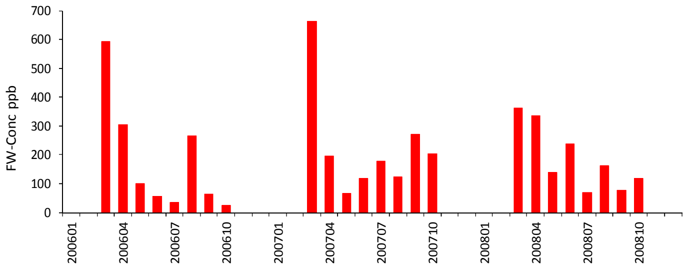




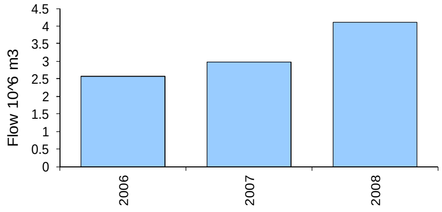
Monthly Time Series:

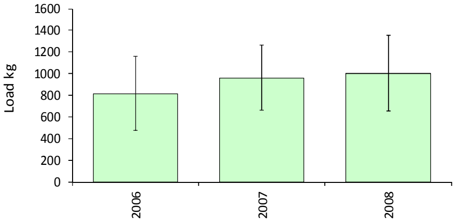


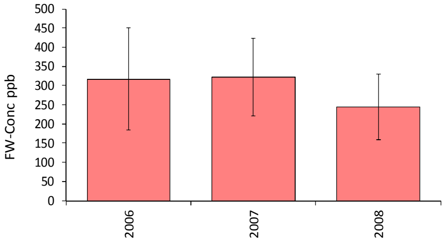




Yearly Time Series:







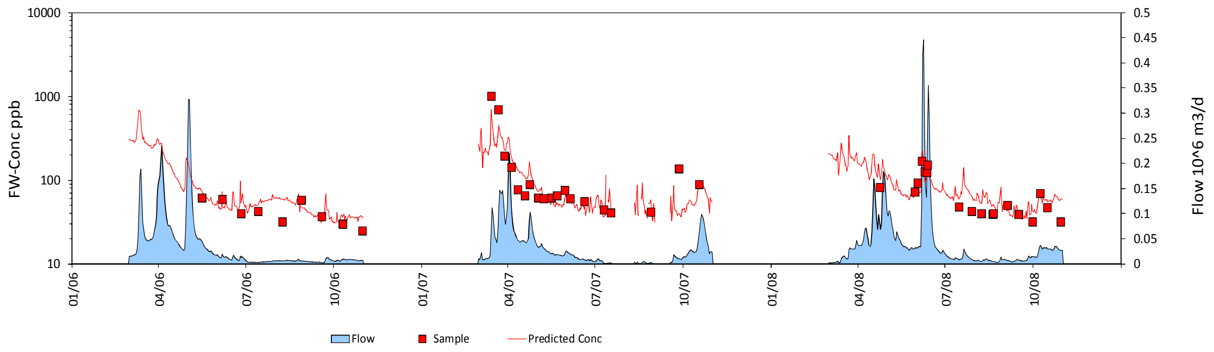
12/20/2010

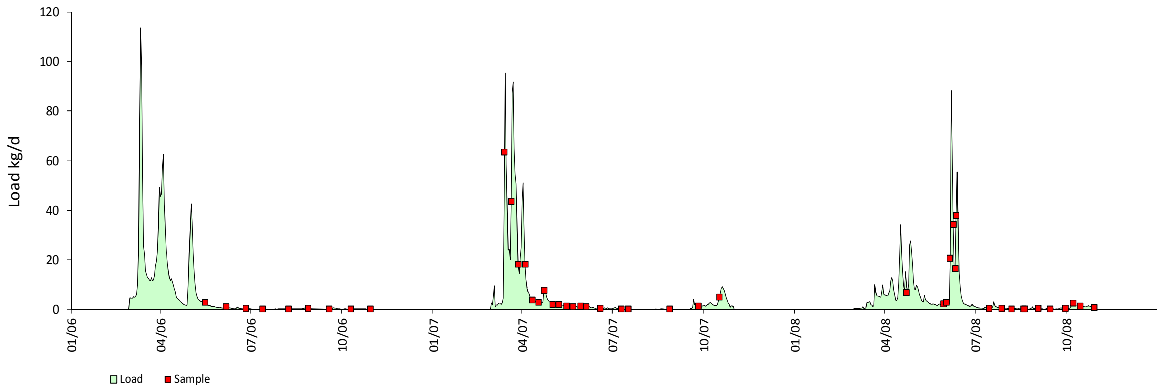
A-12 Flow & TP Load Time Series - Little Rock Creek North

Site: LRC_CR40 LRC at CR 40 Variable: TP Total P ppb Months: 3 10 Dates: 01/01/2006 - 12/31/2008

Flow Data: LRC_CH40 Flow 10^6 m3/yr = 8.094 Load kg/yr = 1291.7 FW-Conc ppb = 159.6 RSE = 13% Method Used= 5

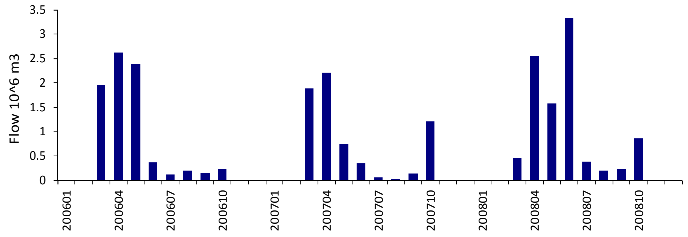
Daily Time Series:

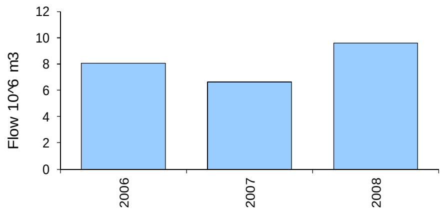


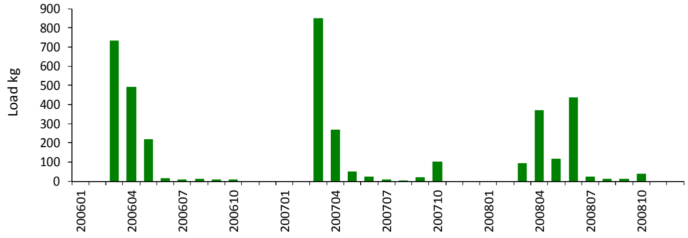


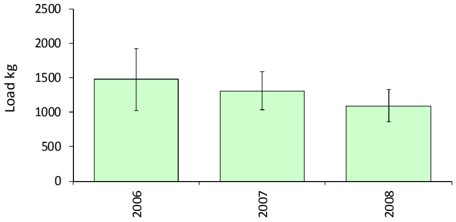
Monthly Time Series:

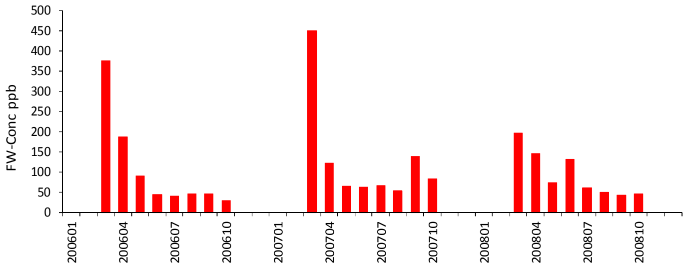
Yearly Time Series:

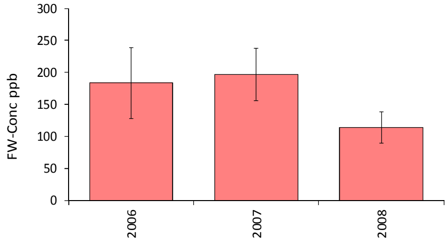












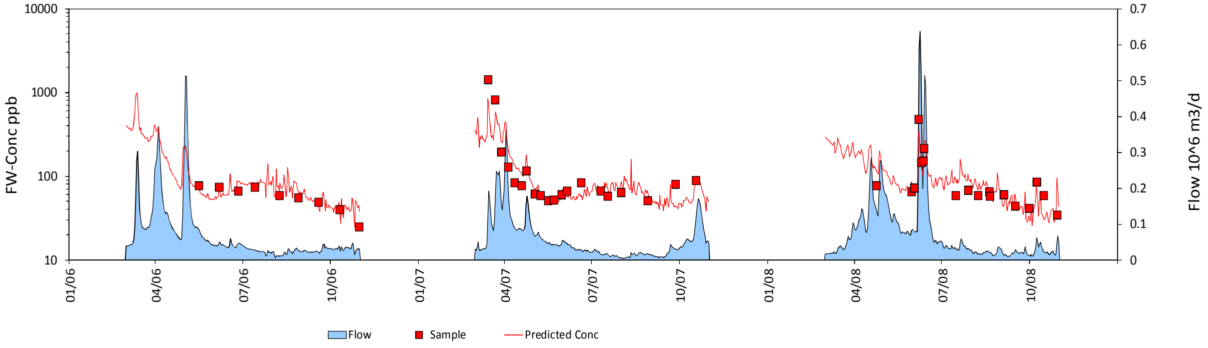
12/20/2010

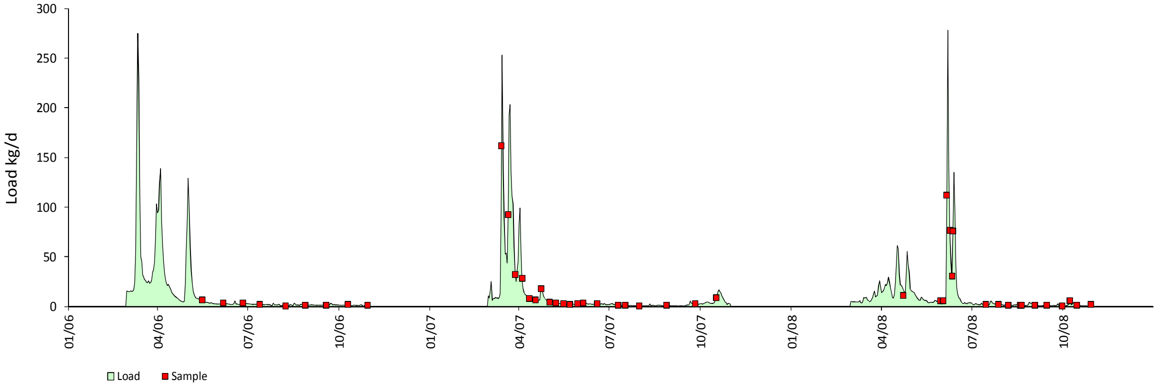
A-13 Flow & TP Load Time Series - Little Rock Creek @ CH12, Lake Inflow

Site: LRC_CH12 LRC at CH12 Inflow to LRL Variable: TP Total P ppb Months: 3 10 Dates: 01/01/2006 - 12/31/2008

Flow Data: LRC_CH12 Flow 10^6 m3/yr = 16.096 Load kg/yr = 2958.4 FW-Conc ppb = 183.8 RSE = 18% Method Used= 5

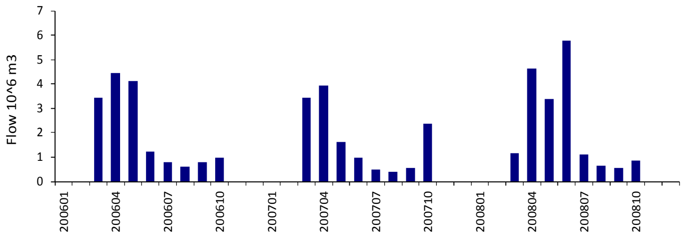
Daily Time Series:

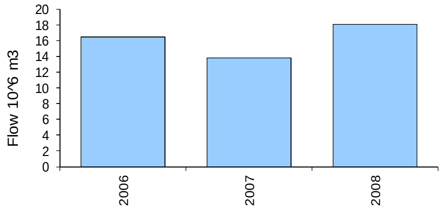


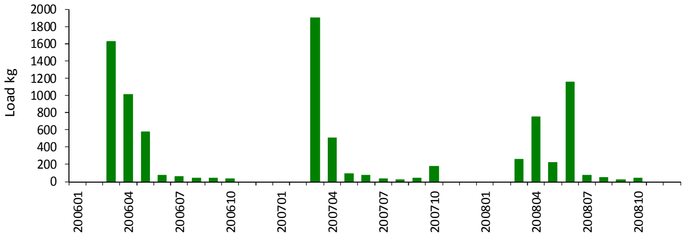


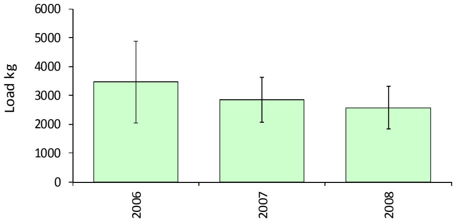
Monthly Time Series:

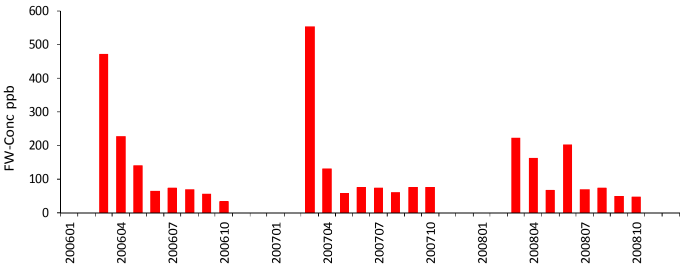
Yearly Time Series:

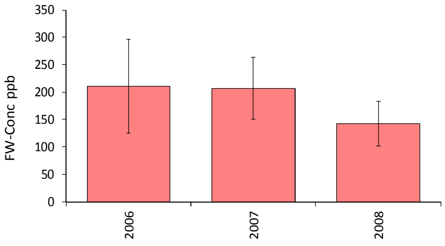












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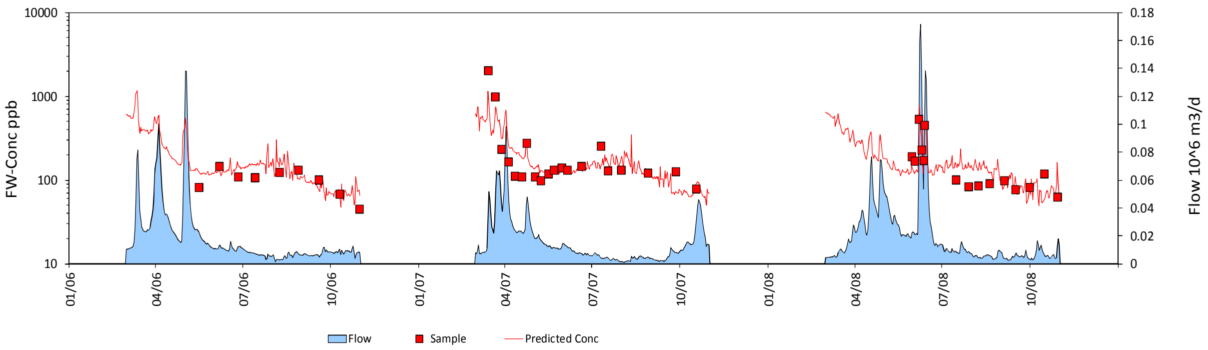
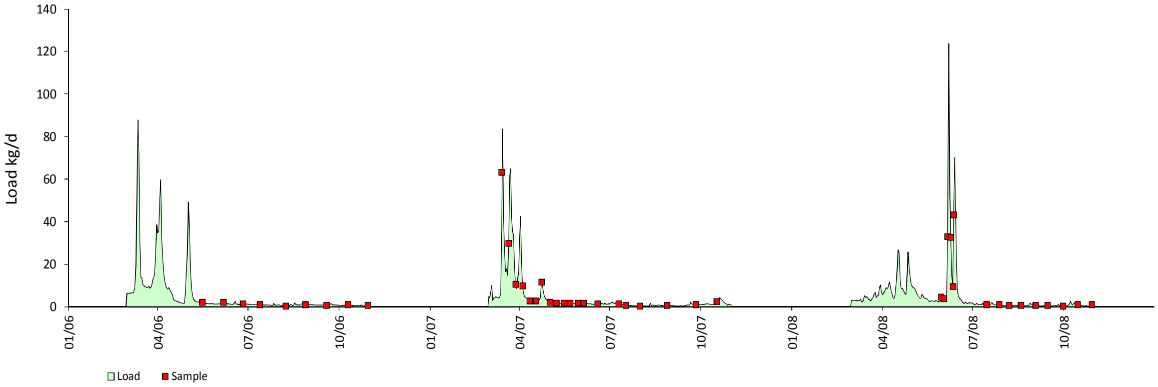
A-14 Flow & TP Load Time Series - Zuleger Creek

Site: ZULEGER Zuleger Creek Inflow to LRL Variable: TP Total P ppb Months: 3 10 Dates: 01/01/2006 - 12/31/2008

Flow Data: ZULEGER Flow 10^6 m3/yr = 4.337 Load kg/yr = 1148.4 FW-Conc ppb = 264.8 RSE = 16% Method Used= 5

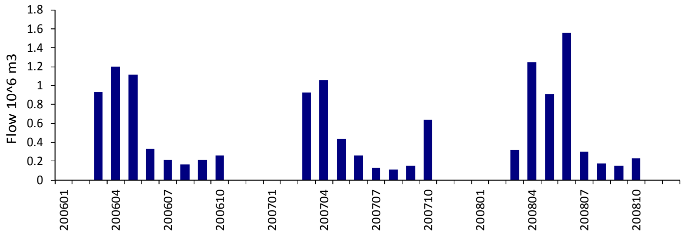
Daily Time Series:

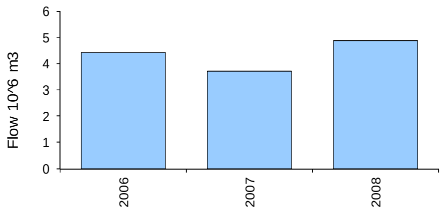


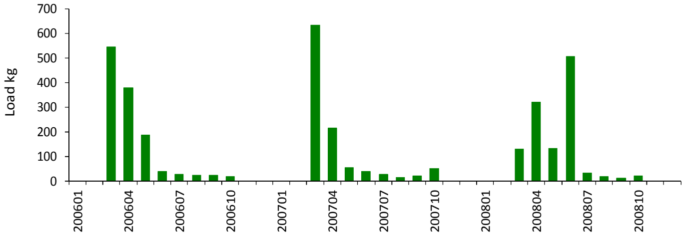


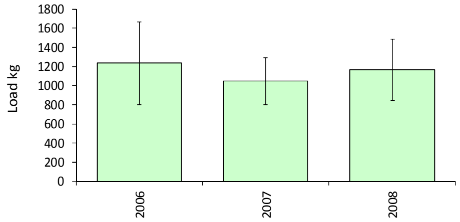
Monthly Time Series:

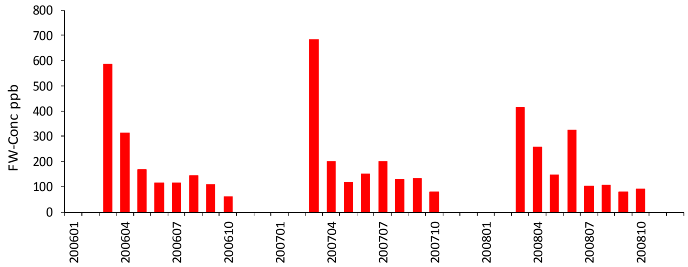
Yearly Time Series:

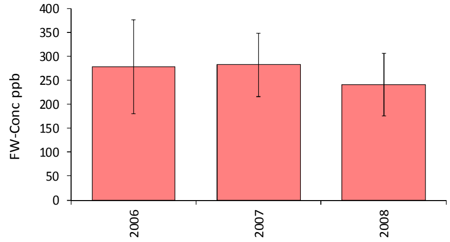








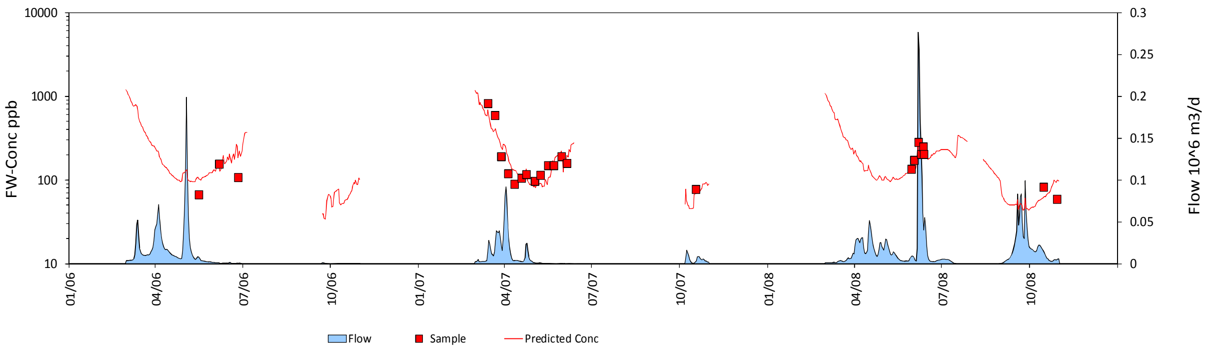


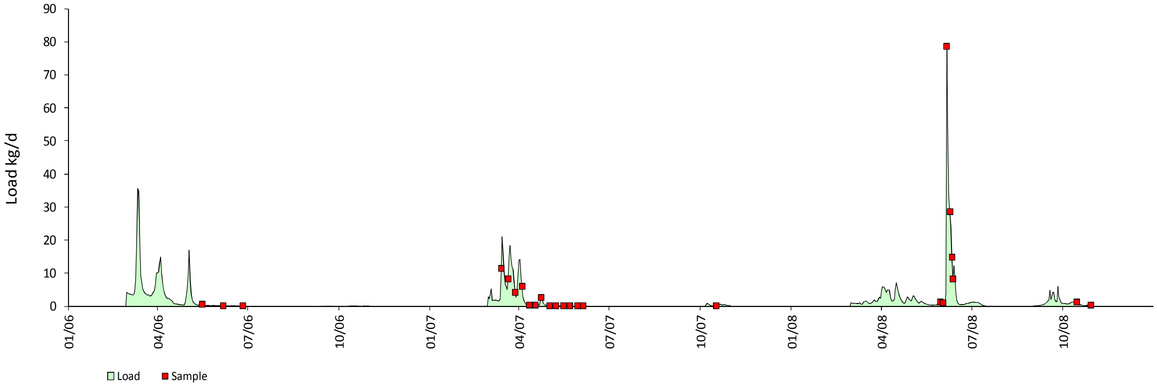


A-15 Flow & TP Load Time Series - Sucker Creek

Site: SUCKER Sucker Creek Inflow to LRL Variable: TP Total P ppb Months: 3 10 Dates: 01/01/2006 - 12/31/2008
Flow Data: SUCKER Flow 10^6 m3/yr = 2.218 Load kg/yr = 403.0 FW-Conc ppb = 181.6 RSE = 8% Method Used= 5

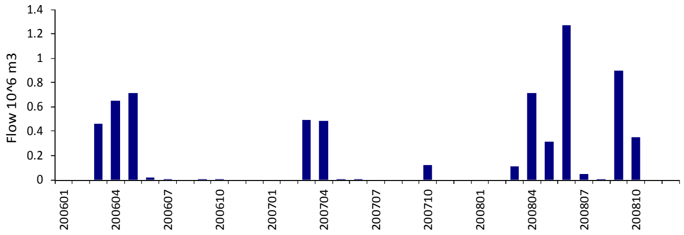
Daily Time Series:

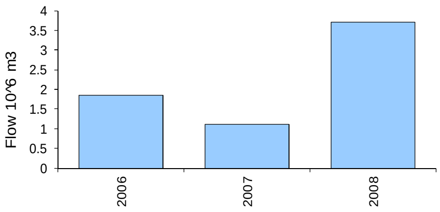


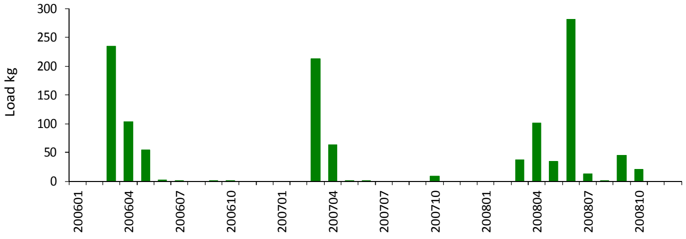


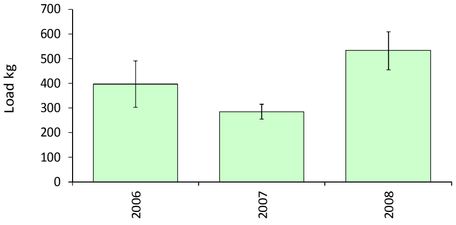
Monthly Time Series:

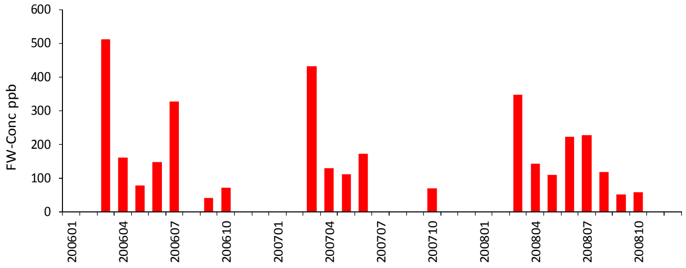
Yearly Time Series:

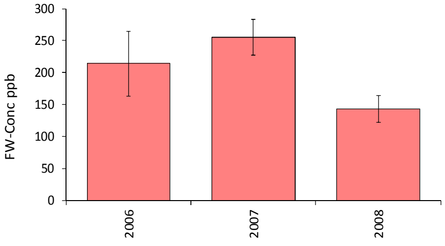










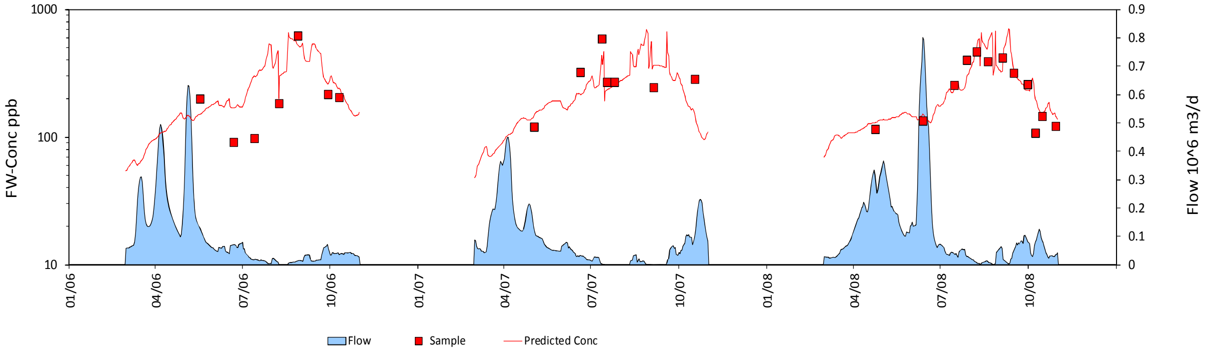


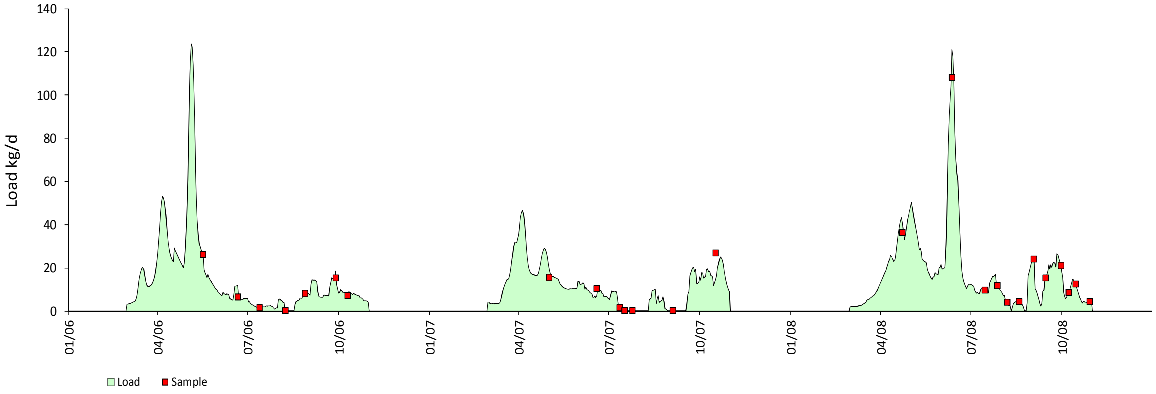
A-16 Flow & TP Load Time Series - Lake Outflow

Site: LAKE All Lake Sites Variable: TP Total P ppb Months: 3 10 Dates: 01/01/2006 - 12/31/2008

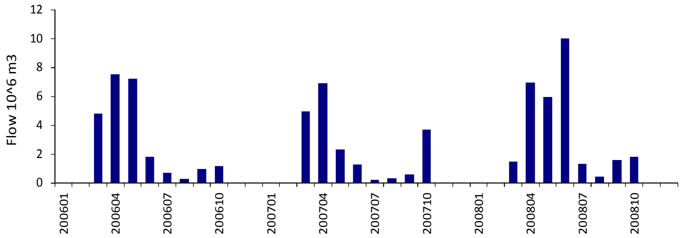
Flow Data: OUTFLOW_7 Flow 10^6 m3/yr = 24.699 Load kg/yr = 3637.7 FW-Conc ppb = 147.3 RSE = 9% Method Used= 5

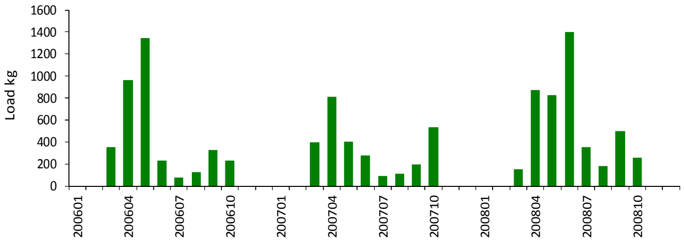
Daily Time Series:

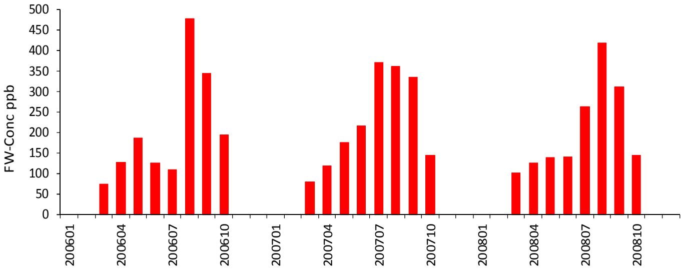




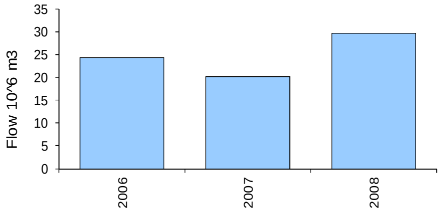
Monthly Time Series:

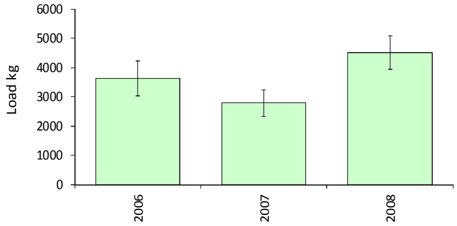


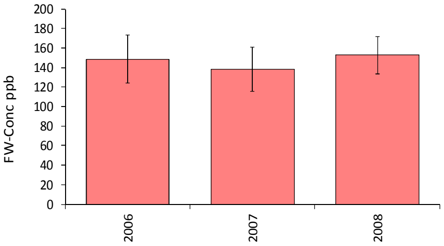




Yearly Time Series:

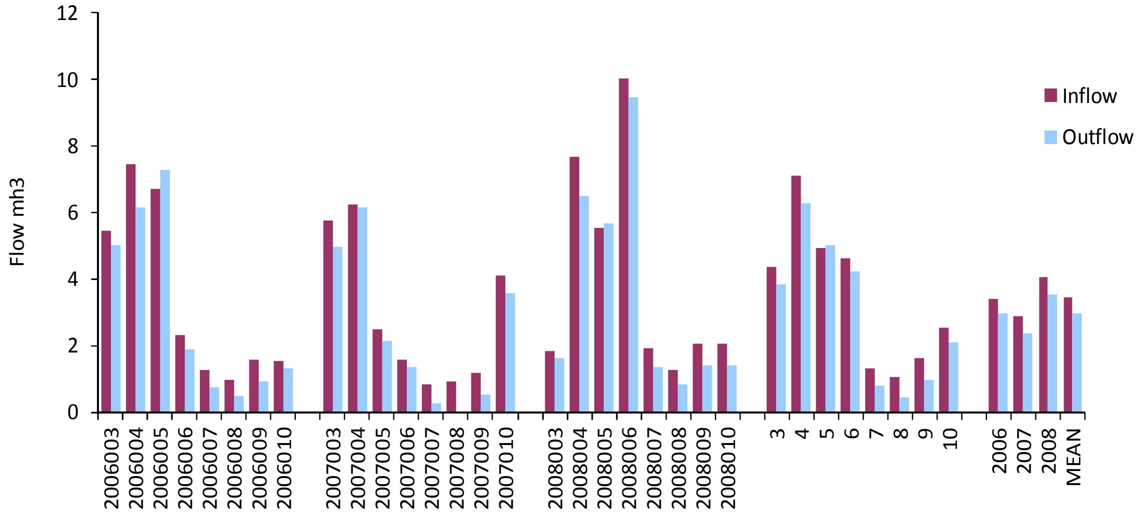


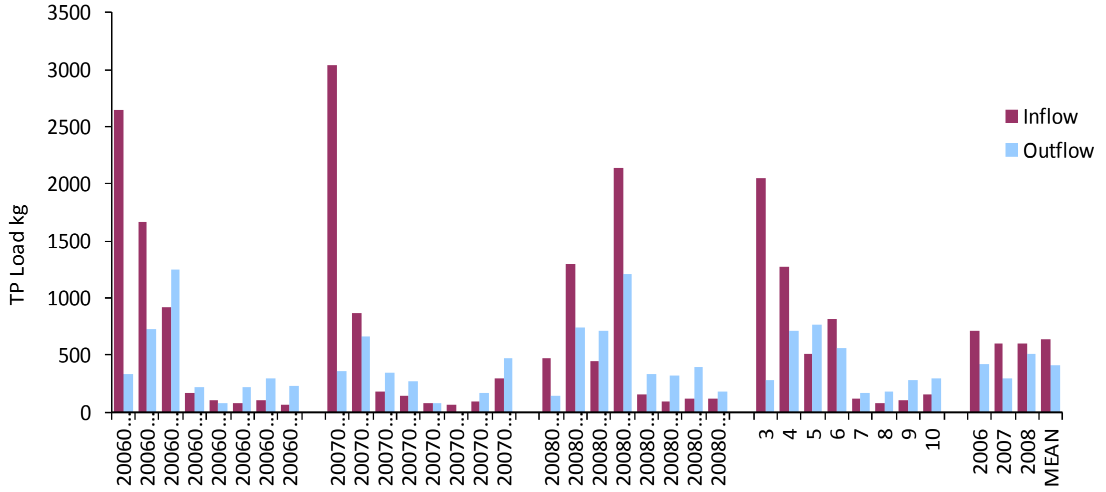


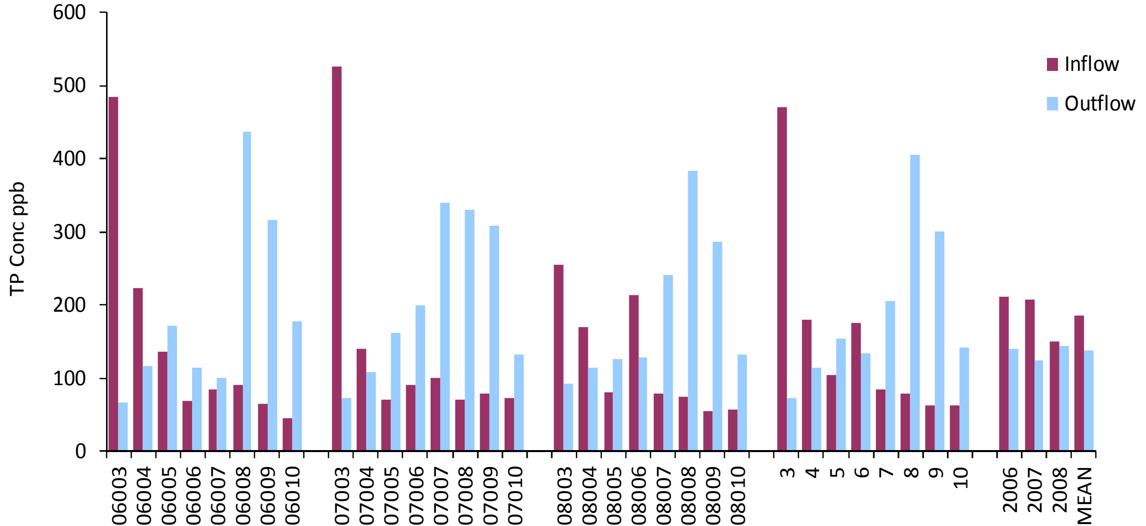


12/20/2010

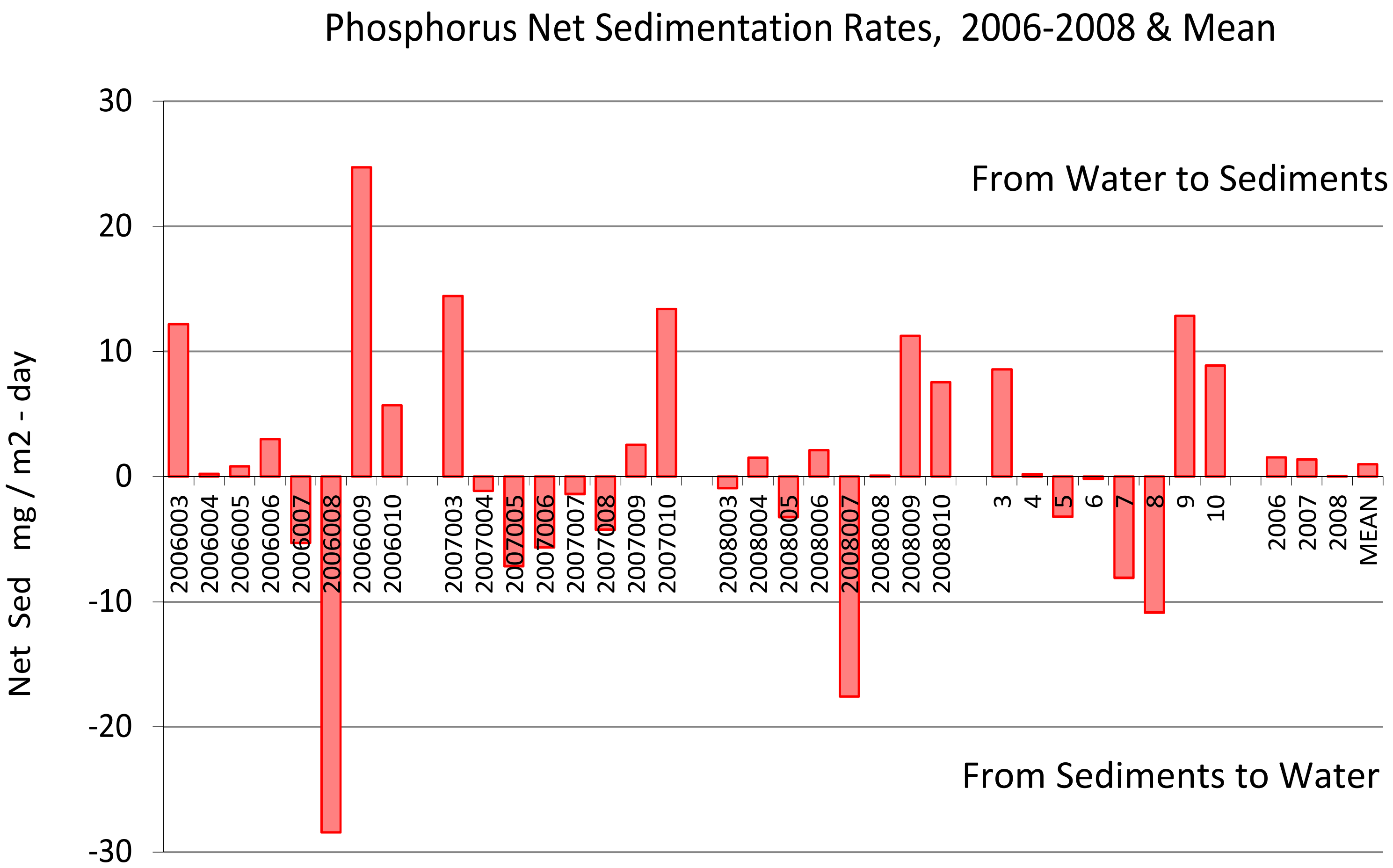
A-17 Monthly Inflows and Outflows







A-18 Monthly & Yearly Phosphorus Net Sedimentation Rates



Net Sedimentation Rate = (Inflow - Outflow - Storage Increase) / Lake Area

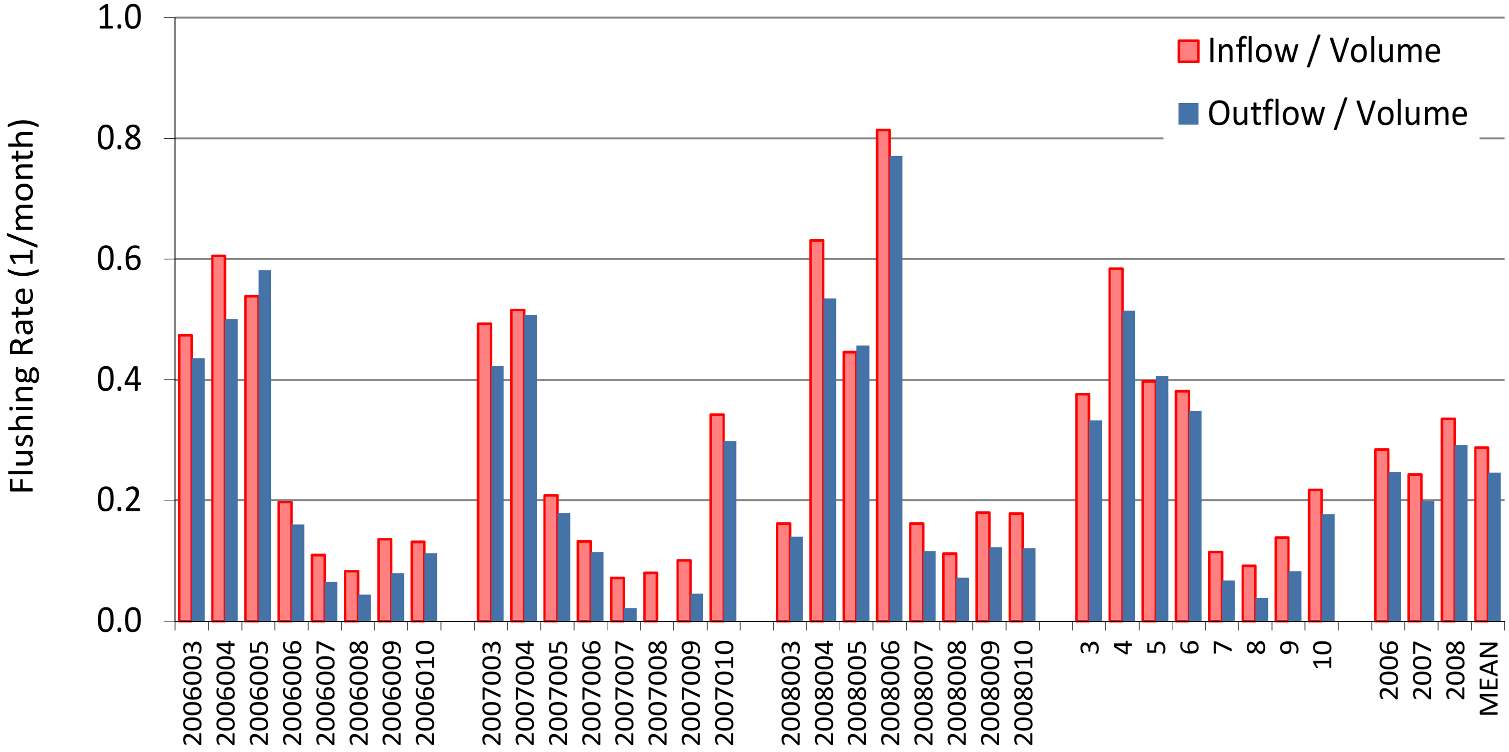
Positive values reflect deposition to sediments; negative values are releases from sediments to water column (internal load)

Positive sedimentation rates in fall reflect dieoff and settling of algal blooms.

Sedimentation rates in March-April are relatively uncertain because of limited watershed and lake data.

Internal loading rates in July-August comparable to values measured by James (2008) under anaerobic conditions.

A-19 Monthly Flushing Rates



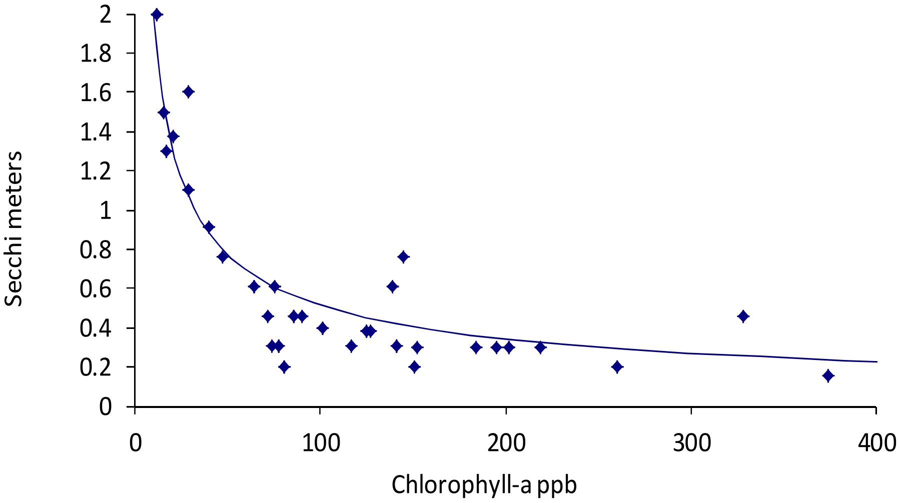
Flushing Rate = Flow / Volume = 1 / Hydraulic Residence Time

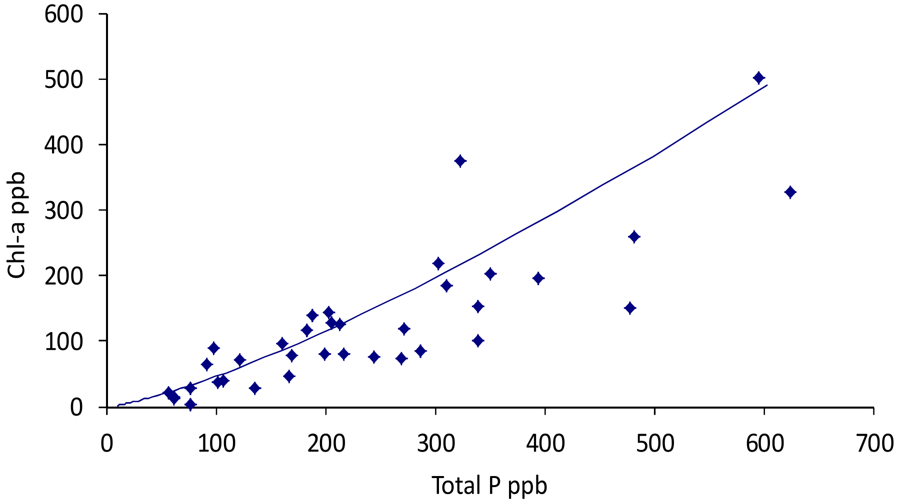
Stagnant conditions (no outflow) occurred in July-August 2007, when toxic bluegreen blooms were observed.

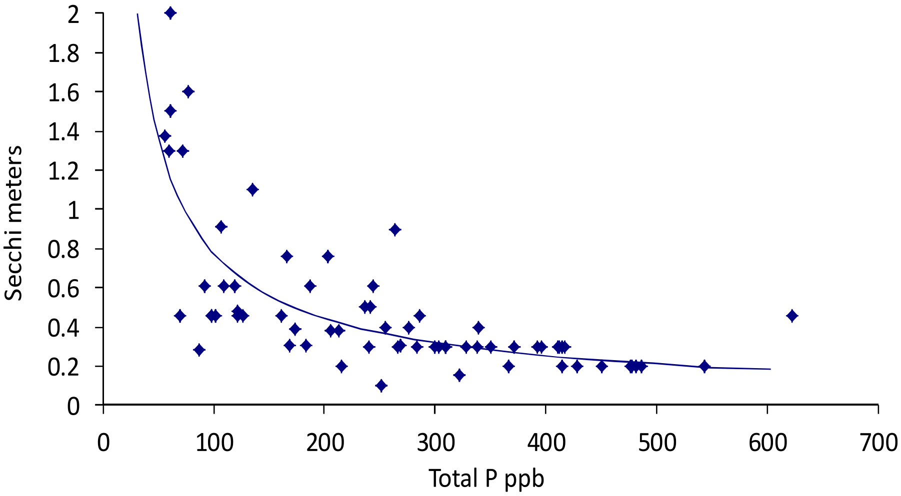
Lake TP concentration increases from ~100 to >300 ppb in summer due to low flushing rate and high sediment P release rates.

Lake TP buildup is enhanced by drought and depletion of stream base flows due to groundwater pumping for irrigation.

TSI Correlations vs. MPCA Statewide Regressions







All LRL Data, May-Sept, 1990-2008; Each point represents a paired sample.
Lines are regressions of summer-mean values; state-wide data (Heiskary & Wilson, 2008)

A-21 LRL Water Quality vs. MINLEAP Predictions

LAKE AND RESERVOIR MANAGEMENT, 1989 5(2): 11-22

Development of Lake Assessment Methods Based Upon the Aquatic Ecoregion Concept

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ABSTRACT

The development of practical lake management strategies in Minnesota has been greatly facilitated by using the aquatic ecoregion approach and standard assessment methodologies (models). Previous studies have shown the significance of the aquatic ecoregion in determining lake water quality patterns, water quality attainability, and development of nutrient criteria (Heiskary et al. 1987; Heiskary and Walker, 1988). This paper focuses upon the use of ecoregion data for modeling purposes. The Minnesota Lake Eutrophication Analysis Procedure (MINLEAP) is a computer program designed to predict eutrophication indices in Minnesota lakes based upon area watershed, depth, and ecoregion. Ecoregion is used to predict runoff and average stream phosphorus concentration. The program formulates water and phosphorus balances and uses a network of empirical models to predict lake phosphorus, chlorophyll a, and transparency values. The program is intended primarily as a screening tool for estimating lake conditions with minimal input data and for identifying "problem" lakes. Included in the program output are: (1) statistical comparisons of observed and predicted phosphorus, chlorophyll a, and transparency values; (2) uncertainty estimates; and (3) estimates of chlorophyll a interval frequencies (nuisance frequencies), for observed and predicted conditions. These expressions of lake condition may be calibrated to citizen preferences using observer surveys (Heiskary and Walker, 1988) to define swimmable and nonswimmable conditions in a locally meaningful manner. The model should be used to approximate lake water quality expectations acknowledging that individual lakes may deviate greatly from regionally defined patterns.

▌ LRL Historical Range

MINLEAP Forecasts 90% Conf Intervals

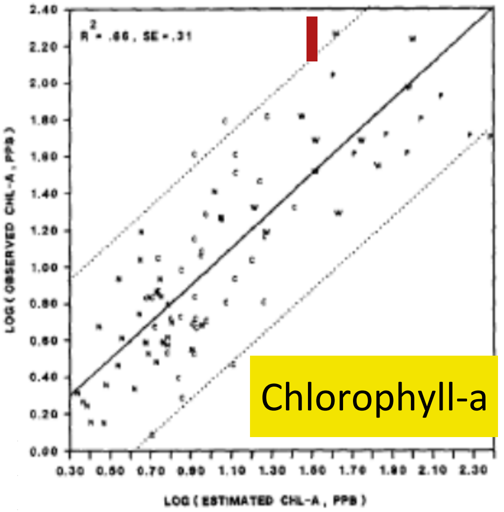


Figure 4.—Observed verus predicted chlorophyll a mean + 2 standard errors by ecoregion. Legend: base-10 logarithmic scales. Symbols: N = Northern Lakes and Forests, C = Northern Central Hardwood Forests, P = Northern Glaciated Plains, W = Western Corn Belt Plains.

With one exception, MINLEAP provides unbiased predictions (mean residual not significantly different from zero) for each ecoregion and lake response variable. The average chlorophyll a residual for Northern

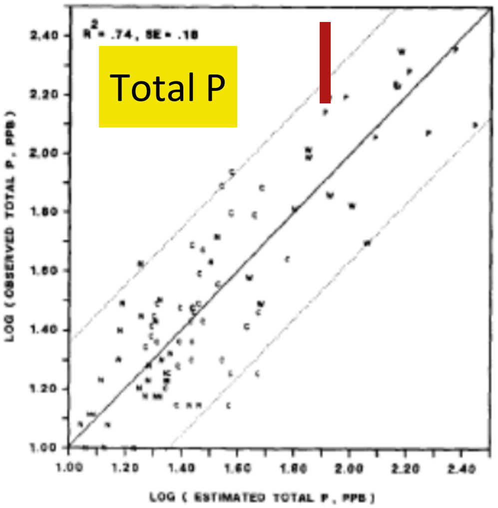


Figure 3.—Observed versus predicted total phosphorus mean + 2 standard errors by ecoregion. Legend: Base-10 logarithmic scales. Symbols: N = Northern Lakes and Forests, C = Northern Central Hardwood Forests, P = Northern Glaciated Plains, W = Western Corn Belt Plains.

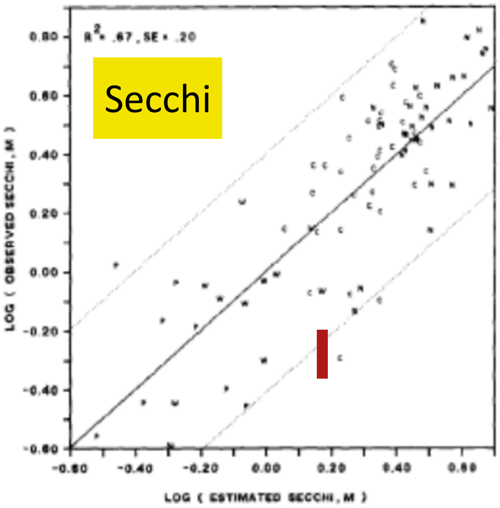
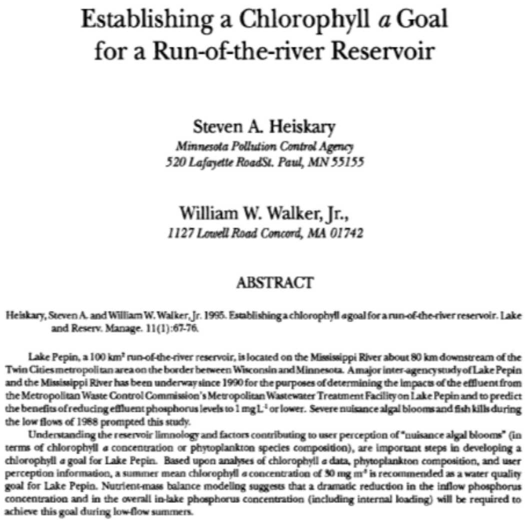


Figure 5.—Observed versus predicted Secchi transparency mean + 2 standard errors by ecoregion. Legend: base-10 logarithmic scales. Symbols: N = Northern Lakes and Forests, C = Northern Central Hardwood Forests, P = Northern Glaciated Plains, W = Western Corn Belt Plains.

MINLEAP predicted Trophic State Indicators in Minimially impacted lakes.
TP values exceeding predicted values likely to reflect excessive P loads and/or limited assimilative capacity due to internal P cycling mechanisms.

Net P Retention vs. Flow in Lake Pepin



Establishing a Chlorophyll *a* Goal for a Run-of-the-river Reservoir

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520 Lafayette Road St. Paul, MN 55155

William W. Walker, Jr.,
1127 Lowell Road Concord, MA 01742

ABSTRACT

Heiskary, Steven A. and William W. Walker, Jr. 1995. Establishing a chlorophyll *a* goal for a run-of-the-river reservoir. Lake and Reserv. Manage. 11(1):67-76.

Lake Pepin, a 100 km² run-of-the-river reservoir, is located on the Mississippi River about 80 km downstream of the Twin Cities metropolitan area on the border between Wisconsin and Minnesota. A major inter-agency study of Lake Pepin and the Mississippi River has been underway since 1990 for the purposes of determining the impacts of the effluent from the Metropolitan Waste Control Commission's Metropolitan Wastewater Treatment Facility on Lake Pepin and to predict the benefits of reducing effluent phosphorus levels to 1 mg L^{-1} or lower. Severe nuisance algal blooms and fish kills during the low flows of 1988 prompted this study.

Understanding the reservoir limnology and factors contributing to user perception of "nuisance algal blooms" (in terms of chlorophyll *a* concentration or phytoplankton species composition), are important steps in developing a chlorophyll *a* goal for Lake Pepin. Based upon analyses of chlorophyll *a* data, phytoplankton composition, and user perception information, a summer mean chlorophyll *a* concentration of 30 mg m^{-3} is recommended as a water quality goal for Lake Pepin. Nutrient-mass balance modelling suggests that a dramatic reduction in the inflow phosphorus concentration and in the overall in-lake phosphorus concentration (including internal loading) will be required to achieve this goal during low-flow summers.

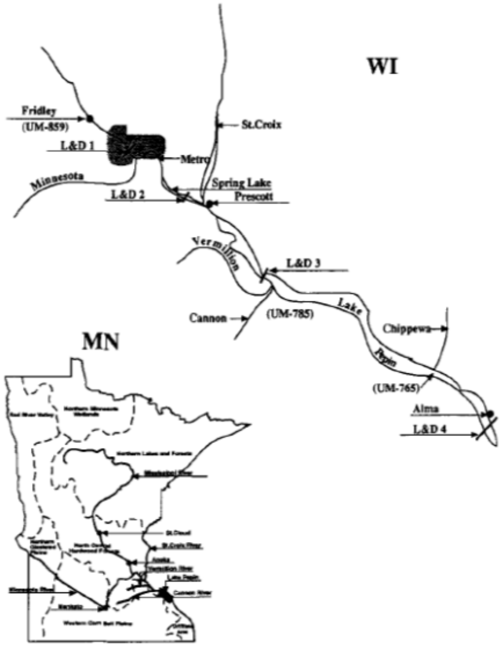
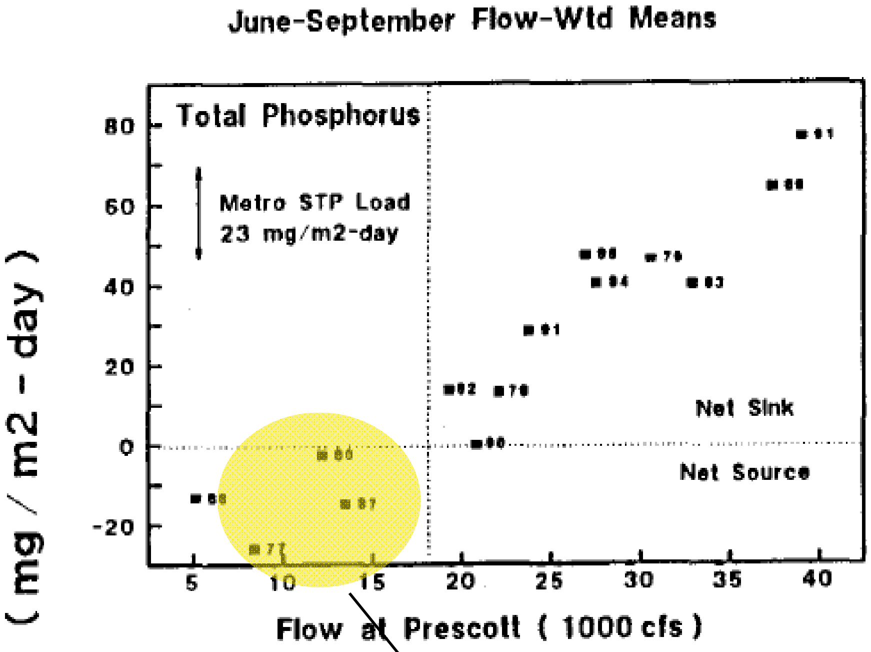
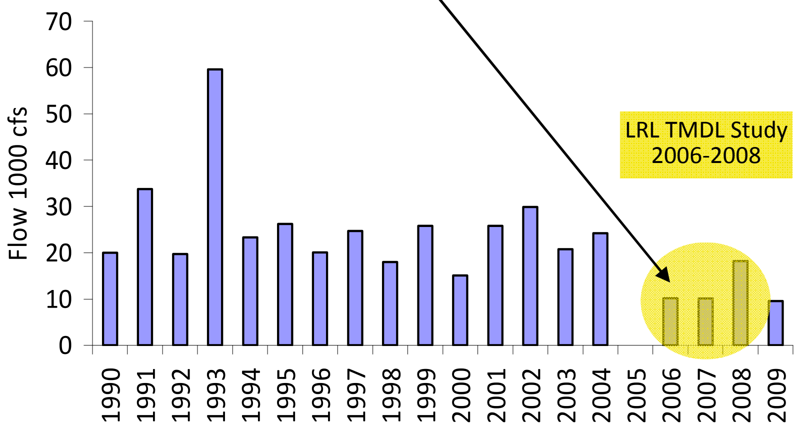


Figure 1.—Mississippi River phosphorus study area.



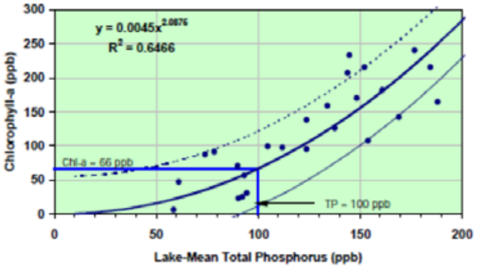
Mississippi River Flow @ Prescott - LRL Period of Record

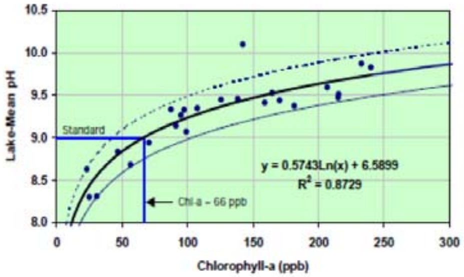


LRL TMDL Study
2006-2008

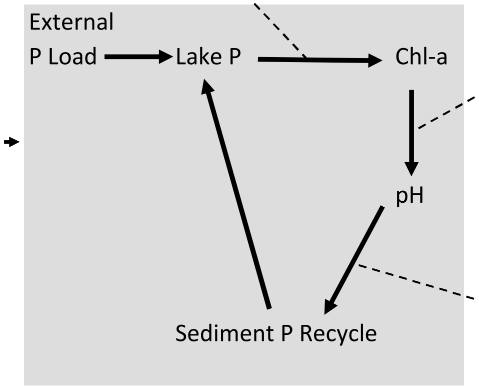
Lake Pepin is a TP Source in dry years similar to LRL Study Period (2006-2008) & Net Sink in Average-Wet Years
The LRL TP Mass Balances in the 2006-2008 drought are not likely to reflect of Long-Term Average Conditions

Feedback Loops Accelerating Internal P Recycling in Upper Klamath Lake, Oregon





Perturbations Accelerating Cycle
- High Temperature
- Low Flow / Dilution
- Wind
- Depth Fluctuations

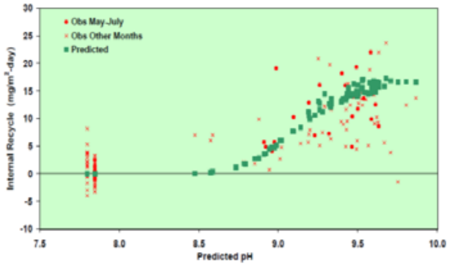

External P Load

Lake P

Chl-a

pH

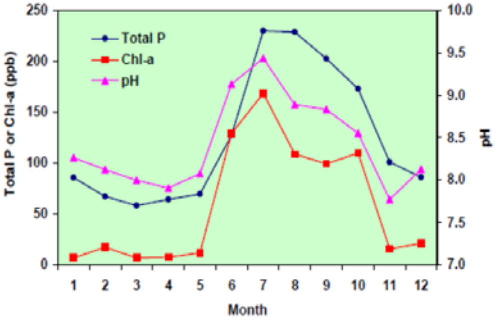
Sediment P Recycle



served Internal Load = Predicted Gross Sedimentation - Observed Net Sedimentation

dicted recycle rates (squares, bottom panel) vary with pH because they are also dependent on sediment P content

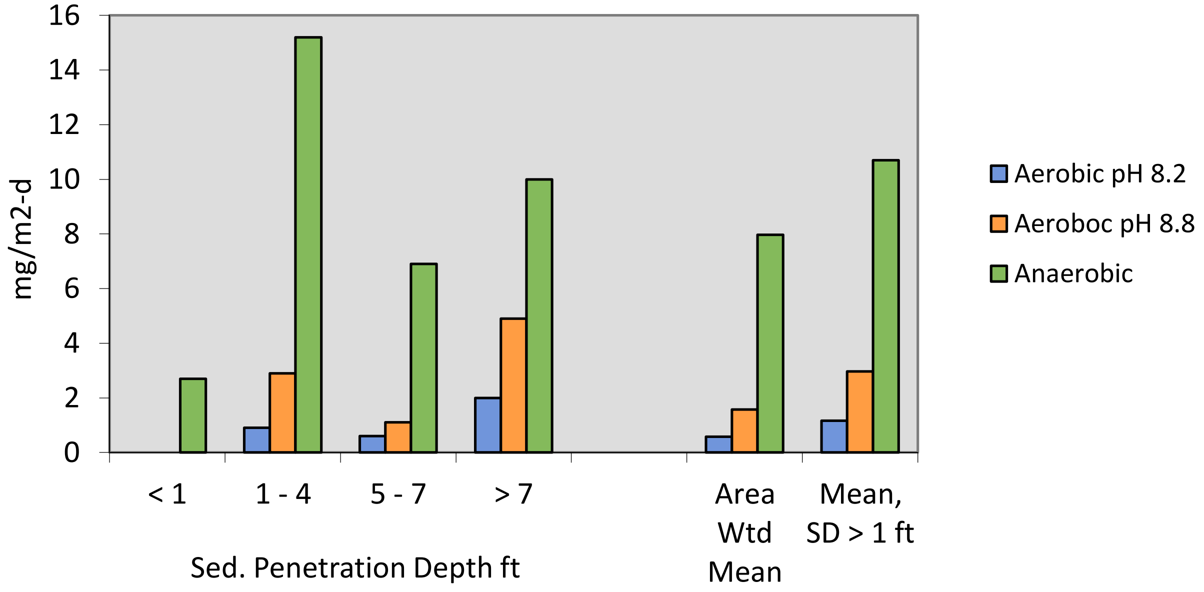
Monthly Variations - UKL

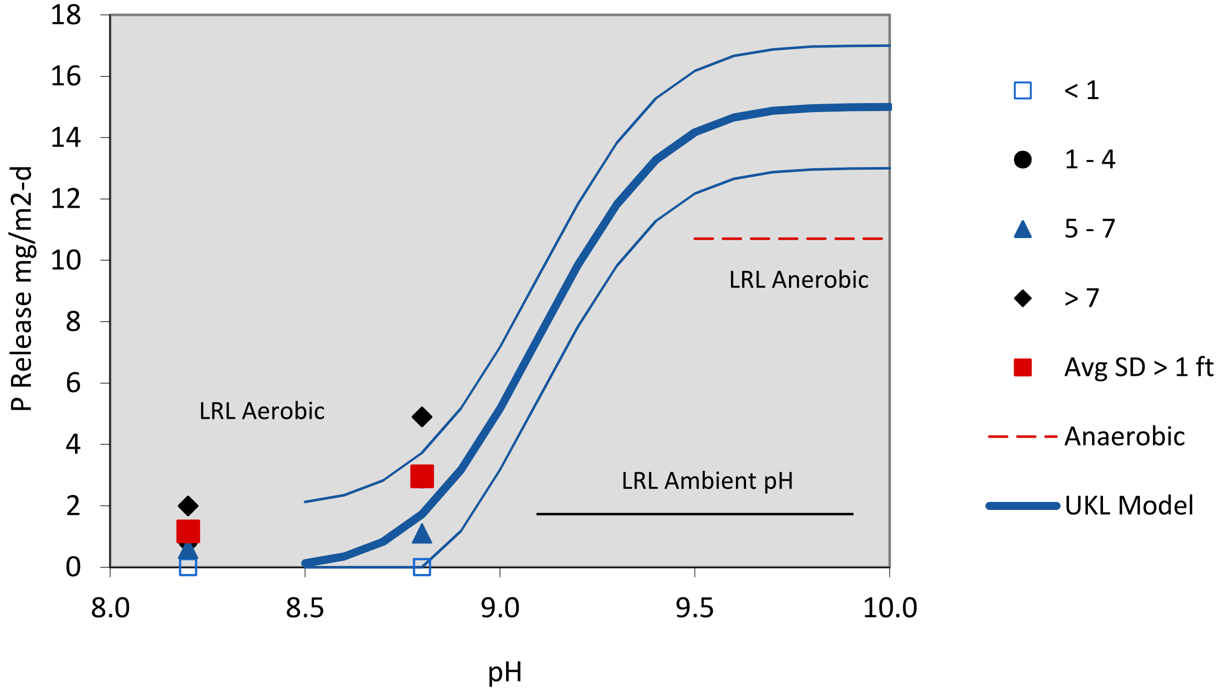


Biweekly Mean Values

Period of Peak Algal Growth Rates (June-July)
Upper Klamath Lake TMDL Model
Ref: Walker, 2001

A-24 Sediment P Release Rates in LRL vs. Upper Klamath Lake Model



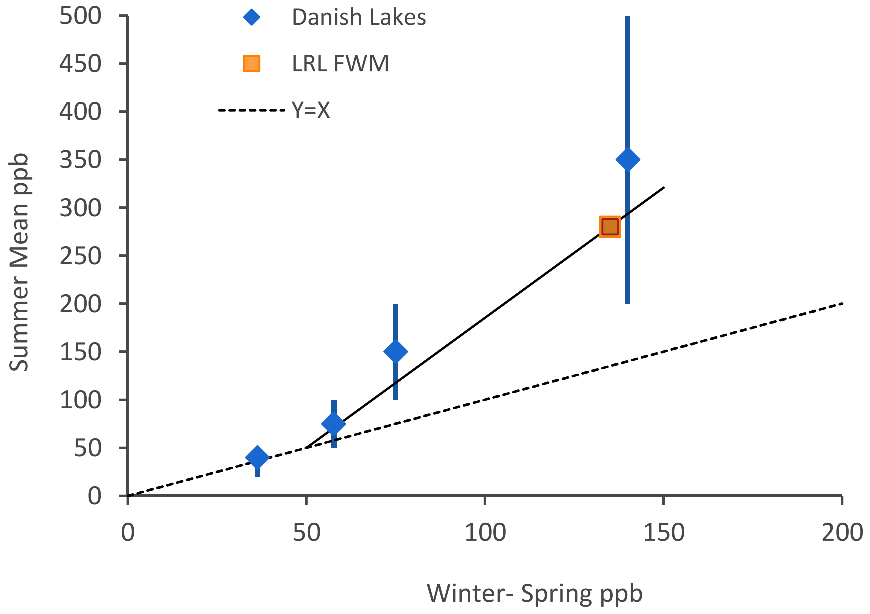


LRL Sediment P Release Rates Measured by James (2008)

UKL = Upper Klamath Lake Model (Walker, 2001)

A-25

Summer TP vs. Spring TP for Shallow Lakes in Denmark



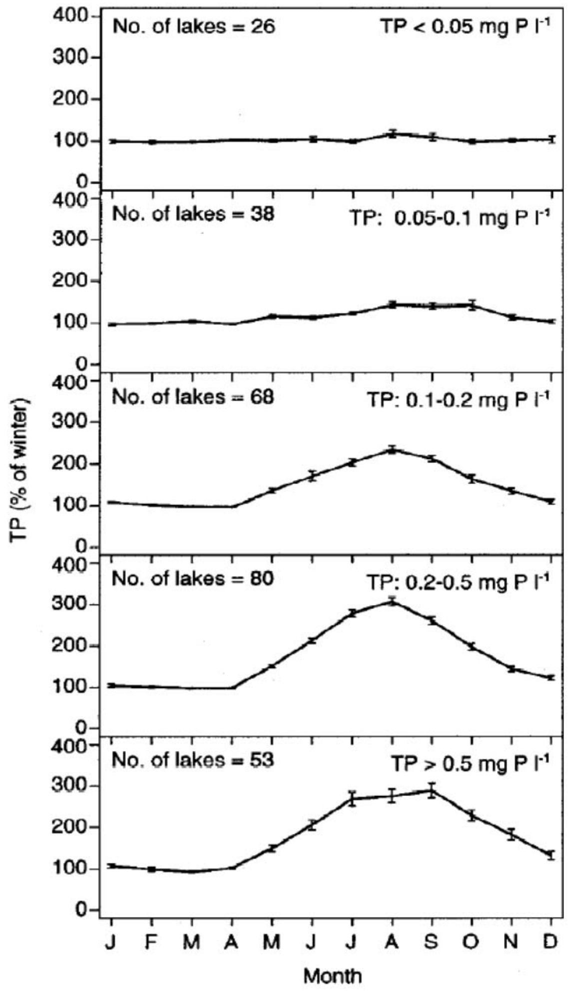


Figure 1. Seasonal variation in TP (monthly mean ± SD) as p
cent of winter values (1 Jan. – 31 March) in different categori
of TP_{sum} (number of lakes = 265). Modified from Jeppesen et a
(1997).

Values for Danish Lakes Inferred from Figure 1 (Sondergaard et al, 1999), Shown on Right

X- Axis: Values Approximating TP Concentration at Start of Growing Season
December-April Means, Danish Lakes

Y Axis: Summer Mean (June - Sept)

Hypothesis: Internal Recycling Feedback Loops Triggered at Spring TP > 50 ppb: (TP -> Chla -> High pH/Anoxic Sediments - > Internal Load -> TP)