Review of River Eutrophication Standards for the Minnesota River: Summary of findings

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Summary

Eutrophication data and associated relevant parameters were analyzed to determine if a site-specific River Eutrophication Standard (RES) is needed for the Minnesota River mainstem. Discharge and total phosphorus (TP) are the most important predictors of summer average algal concentrations in the Minnesota River. At high flows, residence time or abiotic turbidity limit the growth of algae in the Minnesota River despite a surplus of nutrients. At moderate to low flows, residence time and light are not limiting, and high levels of TP allow the growth of undesirable levels of algae. There is no evidence that other factors such as nitrogen, solar energy, temperature, internal loading, carp bioturbation, or external loading of chlorophyll-a (Chl-*a*) are important for estimating long-term summer averages of Chl-*a*. Analysis of TP-Chl-*a* relationships in the Minnesota River indicated that TP levels are not limiting for most or all of the summer season. As a result, data from other large rivers in Minnesota with a range of nutrient concentrations were analyzed to model nutrient-algal relationships. Analysis of these data indicated that the algal-nutrient relationship for the Minnesota River is consistent with those used to develop the RES (Heiskary et al. 2013). This review demonstrates that similar factors control algal growth in the Minnesota River and other rivers in Minnesota and that a site-specific standard (SSS) for nutrients is not needed in the Minnesota River. In order to meet the Chl-*a* standard, average summer concentrations of TP need to at least meet 150 µg/L. In order for the Minnesota River to meet the RES, reductions in TP loading are needed with a focus on reducing loading during periods of average to low flows. There are indications that TP is currently at or nearing limiting concentrations during low flows which demonstrates that further reductions, even if the standard is not met, are likely to result in reductions in algal blooms.

Introduction

Nutrients are part of the natural functioning of aquatic ecosystems, but excessive loading of nutrients can negatively impact aquatic biota and recreational uses (Miltner & Rankin 1998, Wang et al. 2007, Heiskary et al. 2013, Heiskary and Bouchard 2015). To establish standards to protect these beneficial uses, the Minnesota Pollution Control Agency (MPCA) adopted RES in 2015. As part of the southern nutrient region, the Minnesota River is assigned standards for TP of 150 µg/L and Chl-*a* of 40 µg/L[1]. An assessment of multiple reaches of the Minnesota River in 2016 demonstrated that this river does not meet the RES. The City of Mankato asked the MPCA to review the RES for the Minnesota River to determine if a site-specific standard (SSS) is warranted. This analysis had the following objectives:

1. Compile available RES data from the Minnesota River and similar systems;
2. Develop new models or refine existing models to describe algal dynamics in the Minnesota River;
3. Describe the uncertainty in models used to predict TP and Chl-*a* relationships; and
4. Determine if the existing TP standard (150 µg/L) will result in attainment of the Chl-*a* standard.

[1] 40 µg/L for Chl-*a* was adopted into rule language; however, that value was taken from an earlier version of MPCA's analysis. MPCA currently believes the correct value is 35 µg/L and plans to correct this error. Once the revised value is promulgated, the Minnesota River will need to meet 35 µg/L Chl-*a*.

Methods

Data

This study focuses on undredged and unimpounded reaches of the Minnesota River from the Lac qui Parle dam just upstream of Montevideo, Minnesota to Carver Creek near Carver, Minnesota[2]. Data from additional large rivers in Minnesota (Mississippi, St. Croix, Rainy, and Red rivers) were also included. For all rivers, data from impounded and dredged sections were excluded. The following parameters were compiled: TP (unfiltered), orthophosphate (filtered), Chl-*a* (pheophytin corrected), total suspended solids, volatile suspended solids, ammonia, nitrate, nitrite, total Kjeldahl nitrogen, and discharge. Sources for these datasets included the MPCA's water quality monitoring system (Environmental Quality Information System [EQuIS]), the Metropolitan Council Environmental Services (MCES), and the United States Geological Survey (USGS). The compiled dataset included data from the 1970s through 2018. Data were summarized at the Waterbody ID (WID) level and daily, annual, and decadal averages were calculated. These data are provided in Appendix A.

Analyses

The analyses for this review included methods similar to those used to develop the RES. Regressions of TP-Chl-*a* using large rivers (including the Minnesota River mainstem) and the RES development datasets were compared. As with the RES technical development, LOESS regressions ("loess" function in R; R Development Core Team 2019) were used with 95th and 5th percentile quantile smoothing splines regressions ("rq" in "quantreg" package; Koenker 2009 and "bs" in "splines" package; R Development Core Team 2019) to estimate uncertainty around LOESS models. Examination of the relationship between Chl-*a* and other factors was also performed using LOESS regressions ("loess" function in R; R Development Core Team 2019) and bubble charts (SigmaPlot ver. 14; Systat Software 2017). All analyses performed in R were in v. 3.6.1 (R Development Core Team 2019). For factors where sufficient datasets were not available (e.g., solar radiation, carp bioturbation), reviews of relevant literature were performed to determine their potential to be important predictors of summer-average algal populations.

Factors influencing algal growth in the Minnesota River

Multiple factors can affect the growth of algae in river systems. The most important factors identified in rivers are nutrients (largely phosphorus), shading, residence time, and temperature (Heiskary et al. 2013, Kleinteich et al. 2019). As part of this review, these factors are considered to determine if relationships among eutrophication parameters in the Minnesota River are different than the rivers used to develop the RES.

Phosphorus

The relationship of TP and Chl-*a* in the Minnesota River was first examined to determine if the relationships between these two factors was similar to other rivers and if a Chl-*a* model specific to the Minnesota River could be developed. Both daily and summer averages of TP and Chl-*a* had limited relationships (Figure 1) due to high TP concentrations in the Minnesota River through the summer season. However, insight can be gained from these figures regarding phosphorus limitation of algae in the Minnesota River. Specifically, TP concentrations do not become low enough to limit the growth of algae during the summer season. High algal concentrations were observed across the available TP gradient (Figure 1A). Although Chl-*a* concentrations are low for some sampling events, other factors are limiting at those times (see section "Residence Time and Shading").

[2] Waterbody ID numbers: 07020004-747, 07020004-748, 07020004-749, 07020004-750, 07020007-720, 07020007-721, 07020007-722, 07020007-723, 07020012-799, 07020012-800

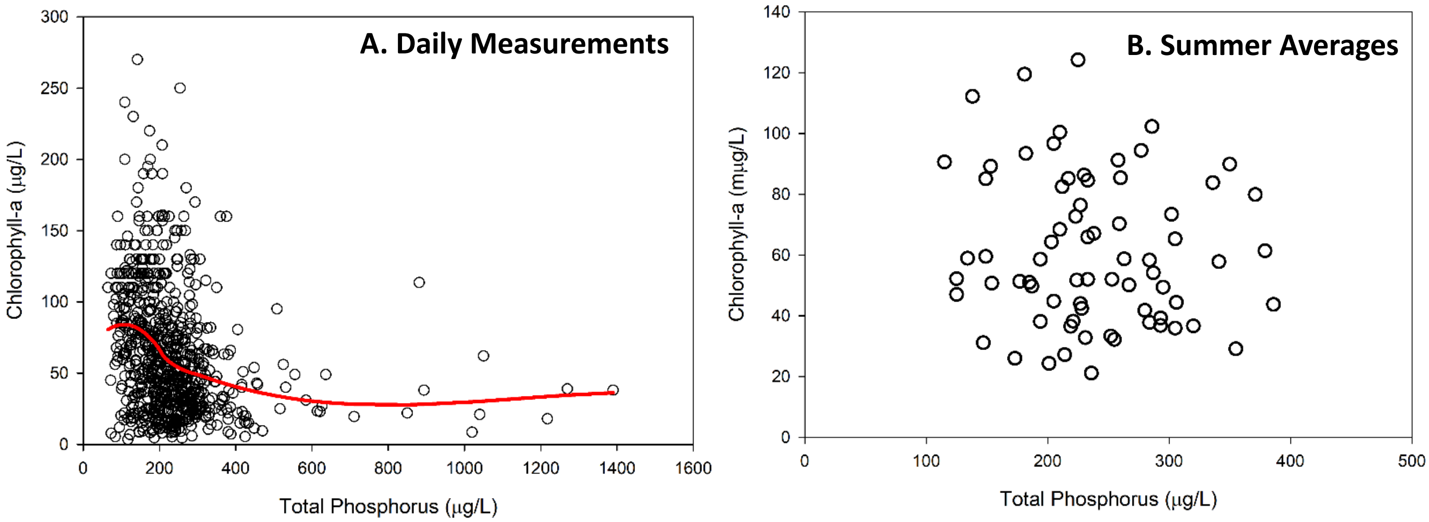

Figure 1: Relationship between total phosphorus and chlorophyll-a for undredged and unimpounded reaches of the Minnesota River during the summer season (June-September) for the years 1997-2018. A) Daily measurements (regression fit is a LOESS regression; span = 0.75, degree = 2) and B) summer average measurements.

A comparison of the Minnesota River with the St. Croix River further illustrates the lack of nutrient limitation in the Minnesota River. The St. Croix River (Figure 2) has a unimodal pattern with low Chl-*a* concentrations at low and high concentrations of TP. At low TP concentrations, algal growth (measured as Chl-*a*) is nutrient limited, but at high TP concentrations algal growth is limited by high flows (either low residence time or shading). In the Minnesota River (Figure 1A), TP never becomes limiting for algal growth, even at low flows. It is only during higher flows when algal growth is limited (see section "Residence Time and Shading").

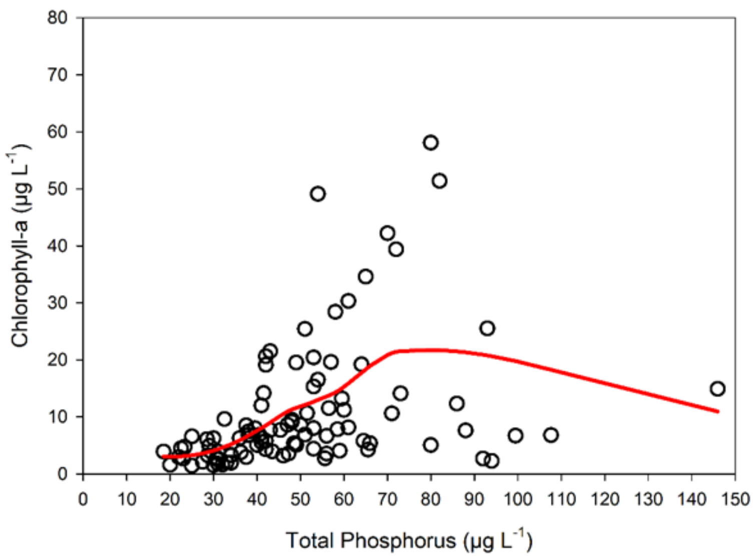

Figure 2: Relationship between daily summer measurements of total phosphorus and chlorophyll-a for the St. Croix River (07030001-619, 07030005-782, 07030005-783, 07030005-784, and 07030005-785) from 1996-2018. Regression fits are LOESS regressions (span = 0.75, degree = 2).

The lack of nutrient limitation in the Minnesota River dataset required the use of data from other large rivers to generate a more complete nutrient gradient and to model summer average Chl-*a* concentrations at lower TP concentrations in the Minnesota River. A new TP-Chl-*a* model (Figure 3) was developed using long-term (10 year), summer averages from unimpounded and undredged sections of Minnesota large rivers (Minnesota, Mississippi, St. Croix, Rainy, and Red rivers) following methods similar to those used to support the original RES rule. The new large river TP-Chl-*a* model is similar and is consistent with the model used to support the development of the RES. Specifically, the large river data points and the new large river model (blue lines; Figure 3) fall with within the prediction interval of the original model (grey lines; Figure 3; see Figure 21 in Heiskary et al. [2013]). The new model also has a better pseudo-R^2 value ($R^2 = 0.82$) compared the original model ($R^2 = 0.58$). The new large river model explicitly incorporates data from the Minnesota River demonstrating that the

TP-Chl-*a* relationship for the Minnesota River is predicted to be similar to other rivers in Minnesota. If the data points for the Minnesota River fell outside the prediction bands for the original model, it would be evidence that a SSS may be needed. The only large river with multiple points falling outside the original prediction bands was the Red River (Figure 3). As a result, the Red River was not included in the new large river model because non-nutrient factors (i.e., abiotic turbidity) are strongly limiting algal growth on the Red River.

The new large river model confirms the validity of the original RES model and is an improvement due to its narrower prediction interval compared to the original model. The model improvements can be attributed to the use of a dataset with more homogenous rivers because there is a positive relationship between catchment area and Chl-*a*:TP ratios (Van Nieuwenhuyse and Jones 1996). Using the new large river model provides greater predictive certainty for the determination of Chl-*a* concentrations under lower TP loading in the Minnesota River. The new model predicts that a TP concentration of 149 µg/L is needed to meet a Chl-*a* level of 40 µg/L[3]. The current TP standard of 150 µg/L is near this predicted concentration and is within the prediction interval. As a result, the application of the current standard is appropriate and does not require the adoption of a more stringent TP standard at this time.

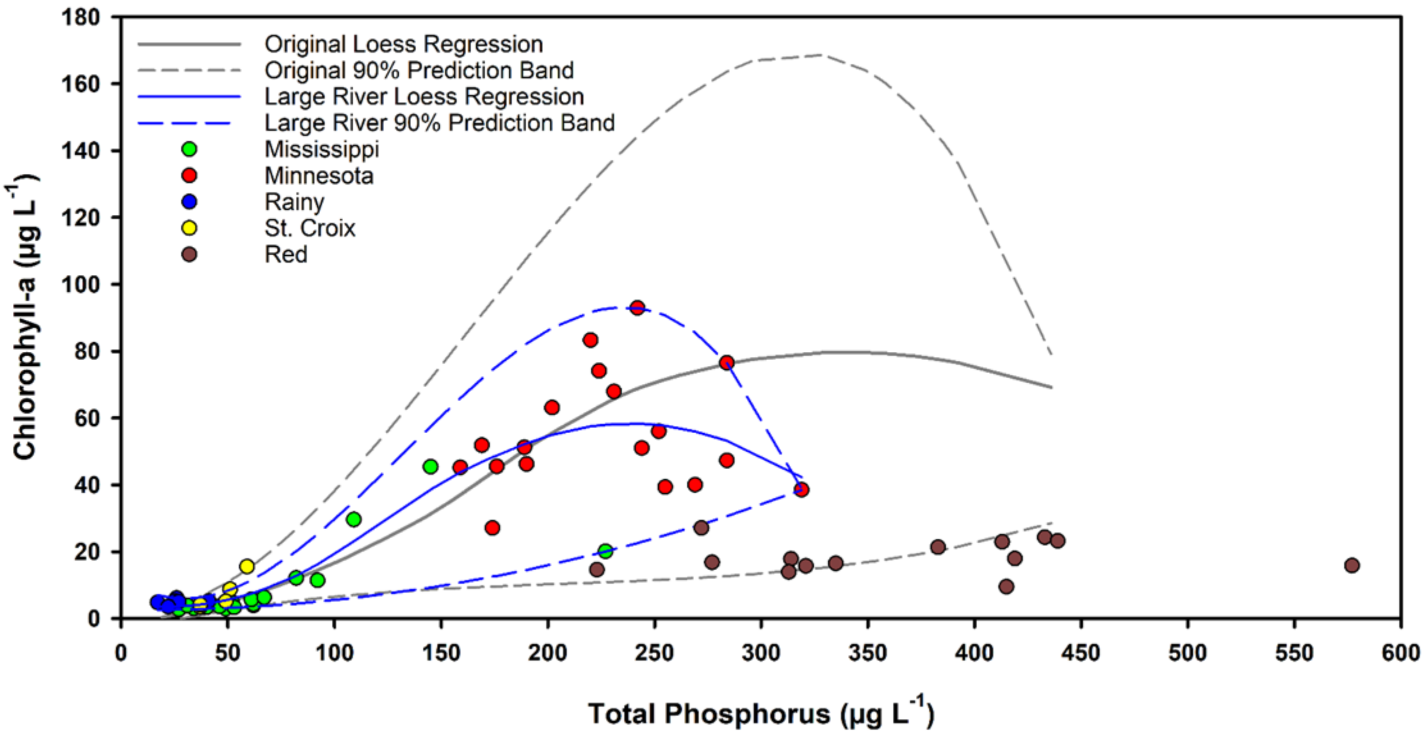

Figure 3: Relationship between total phosphorus and chlorophyll-a for Minnesota large rivers and the original RES development dataset. Data points show individual measurements used to develop the new large river model (blue lines) with exception of the Red River which was not included in the new large river model. Regression fits are LOESS regressions (span = 0.75, degree = 2) with 90% prediction intervals estimated using 95th and 5th quantile smoothing splines regressions (degree = 3, df = 3). Minnesota large river data points are decadal averages from river reaches with at least 4 measurements per summer and 2 years per decade. The original regressions (grey lines; see Figure 21 in Heiskary et al. [2013]) used non-wadeable rivers (watershed >500 mi^2).

In the Minnesota River, Chl-*a* is also correlated with orthophosphate concentrations although there is a negative relationship between these parameters (Figure 4A). This does not indicate that increasing orthophosphate results in the decrease of algal concentrations, rather orthophosphate is an indicator of algal concentration. Orthophosphate is readily utilized by plants; however, in the Minnesota River, high orthophosphate concentrations are an indication of algal limitation by some other non-nutrient factor. This is apparent in the relationship between orthophosphate and discharge on the Minnesota River. At low flows, when residence time and shading due to suspended solids are not important limiting factors for algal growth, orthophosphate is low (Figure 4B). This matches the pattern observed by Metropolitan Council (2010) and James (2007, 2008) where

[3] To meet Chl-*a* of 35 µg/L, the new large river model estimates a TP concentration of 137 µg/L is required.

soluble reactive phosphorus (SRP) and Chl-*a* were negatively correlated on the reach of the Minnesota River from Jordan to the mouth. Orthophosphate is highest during higher flows when algae is limited by residence time and shading (see section “Residence Time and Shading”), and there is surplus orthophosphate in the water column because algal populations are not large enough to utilize this form of phosphate. In the Minnesota River, orthophosphate concentrations essentially only provide an indication of algal concentrations and do not directly provide information regarding needed TP concentrations to meet the RES.

Although orthophosphate is of limited use for reviewing the need for a SSS on the Minnesota River, some insight can be gained regarding progress toward limiting algal growth in the Minnesota River. Even under conditions where nutrients, shading, and residence time are not limiting algal growth, there are limitations to the carrying capacity of algae in the system. For example, even under ideal growing conditions (i.e., high nutrients, low turbidity, and high residence time), algal population size is still limited by biogenic turbidity (i.e., self-shading; Talling et al. 1973, Wetzel 2001). If under conditions where abiotic turbidity and residence time are not limiting and high concentrations of water column orthophosphate is present, it would indicate that the algal population has reached carrying capacity and that there is still excess available phosphorus. However, if under these conditions, there is very little orthophosphate in the water column, that indicates phosphorus is starting to become limited. This can be observed in Figure 4B where under low flow conditions (i.e., shading and residence time are not limiting), water column orthophosphate concentrations are low. Furthermore, there appears to be a decrease in orthophosphate in more recent years (2012-2018 versus 1998-2005) demonstrating that reductions in nutrient loading may be starting to impact algal growth. However, concentrations of phosphorus need to be reduced in the Minnesota River to increase nutrient limitation during periods when algal growth can occur.

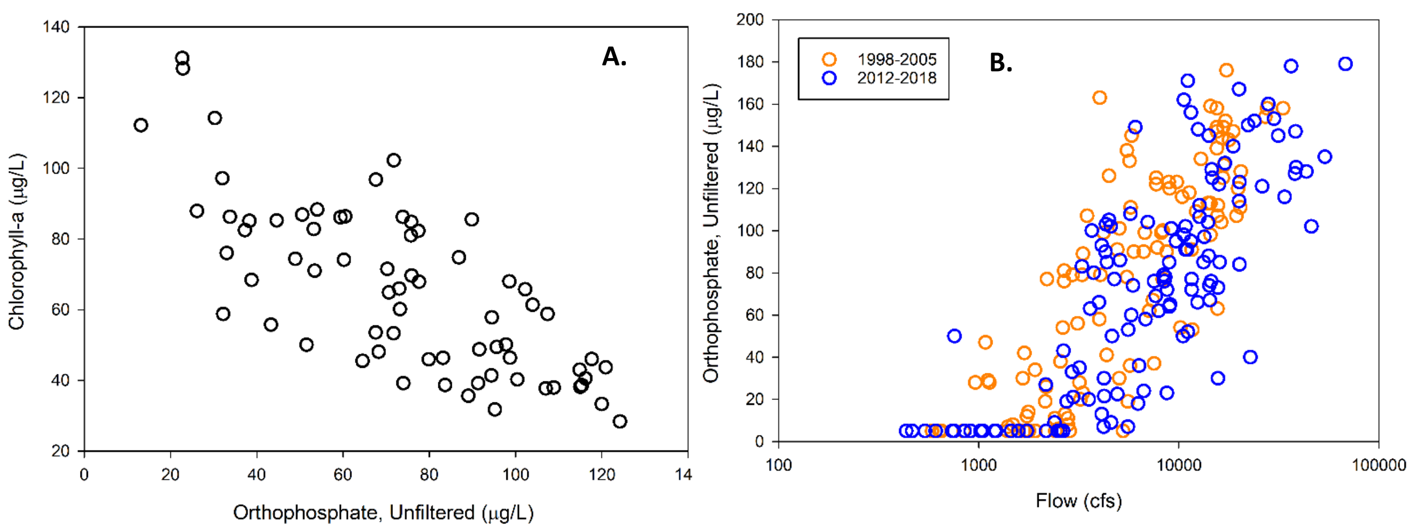

Figure 4: Relationships between orthophosphate (unfiltered), chlorophyll-a, and flow for the Minnesota River. A) Summer average (June through September) data collected by MCES at river miles 3.5, 8.5, 14.3, 25.1, 39.4, 89.7, and 120.0. B) Daily measurements during the summer index period (June through September) from 1998-2005 and 2012-2018 collected by MCES at river mile 39.4.

Residence Time and Shading

Residence time (or hydraulic flushing rate) and shading are important factors that can limit algal growth in rivers (Reynolds 2000, Heiskary et al. 2013). These two factors can be difficult to separate because in many systems the two factors are correlated. In the Minnesota River, limited suspended sediment data prevented a detailed analysis which could separate the individual impacts of these factors. However, the specific mechanism (i.e., residence time, shading, or a combination of the two) need not be determined as part of the SSS review because it is possible to document the impact of these factors together using discharge as a surrogate for their combined effects. As documented in Heiskary et al. (2013), the effects of shading and residence time, estimated using flow, on Chl-*a* concentrations can be observed in Minnesota’s rivers (Figure 5A). A repeat of this analysis using only large rivers, which includes the Minnesota River, documents a similar pattern and demonstrates that shading and residence time have similar impacts on Chl-*a* in the Minnesota River (Figure 5B). Figure 1A further demonstrates this relationship where high Chl-*a* concentrations are not observed at high flows when TP concentrations are highest. The combined impacts of residence time, shading, and TP are discussed in the "Multiple factors" section.

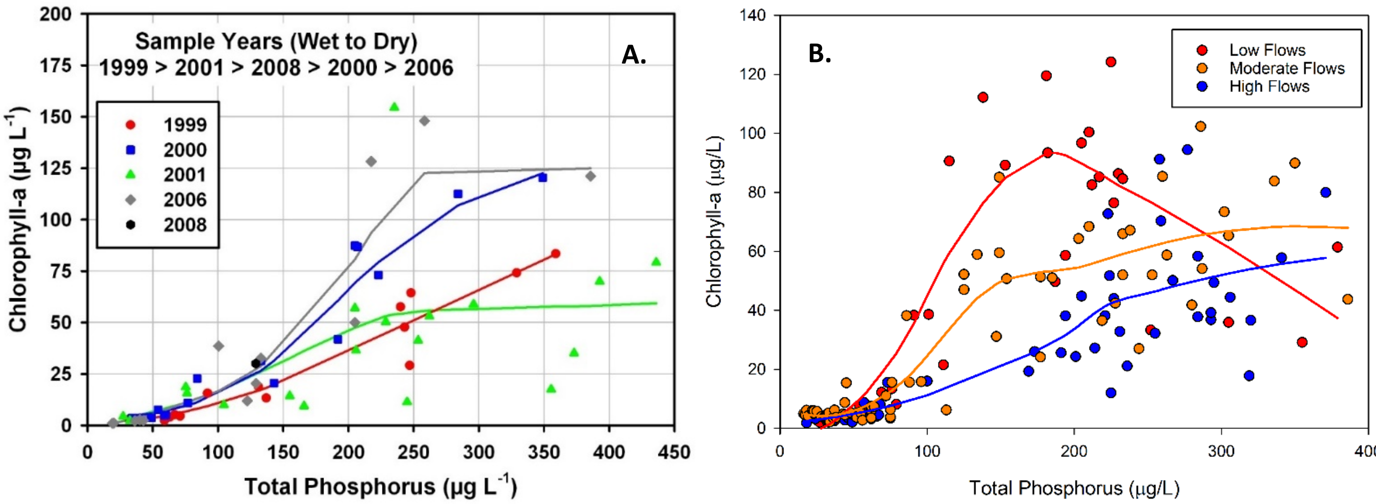

Figure 5: Relationship between total phosphorus and chlorophyll-a at different flows. A) From Heiskary et al. 2013 [Figure 23, p. 47]. B) Annual averages for Minnesota large rivers (Minnesota, Mississippi, Rainy, and St. Croix) with average summer flows divided into quartiles (data points are annual averages from river reaches with at least 4 measurements during the summer index period (June through September); regression fits are LOESS regressions (span = 0.75, degree = 2); low flows = average flow <25th quartile, R^2 = 0.78 [n = 36]; moderate flows = average flow 25-75th quartile, R^2 = 0.81 [n = 70]; low flows = average flow <25th quartile, R^2 = 0.67 [n = 53]).

Other factors

There are a number of other factors that likely play some role in the growth of algae in the Minnesota River, but a review of available data and the literature indicate that these factors are minor compared to TP, shading, and residence time. Some may be more important at smaller temporal or spatial scales in the Minnesota River, but when considered at the scale used to assess the RES (i.e., long-term summer averages), these factors are minimal.

Nitrogen. Several forms of nitrogen are correlated with Chl-*a* and TP, but phosphorus is more typically limiting in freshwater systems. In addition, reducing nitrogen without reductions in TP could promote the growth of cyanobacteria (Schindler et al. 2008). It is not likely that reductions in nitrogen alone would reduce algal production in the Minnesota River. Rather TP is the controlling nutrient (Schindler et al. 2016) or a combination of nitrogen and TP reductions are needed (Kleinteich et al. 2019, Paerl et al. 2016). As a result, nitrogen does not need to be considered as part of the review of the RES for the Minnesota River.

Temperature*.* Temperature impacts algal growth, but during the summer index period water temperature in the Minnesota River is sufficient for algal growth. The effects of temperature on algal growth is difficult to separate from other factors because it is correlated with discharge and therefore residence time and shading. However, the effects of temperature appear to be less important than other factors because high concentrations of Chl-*a* (e.g., >60 µg/L) are observed through the range of temperatures measured during the summer index period (Appendix A) and high algal concentrations can be observed in October, November, and December (Metropolitan Council 2010). Long-term summer average measurements also reduce the effects of temperature on estimates of Chl-*a* concentrations. Temperature affects algal growth rates on the Minnesota River, but these effects are likely to be relatively small during the summer index period and would not be expected to greatly differ from other rivers in Minnesota.

Solar energy. Solar energy impacts algal growth and there is some evidence that decreased solar energy in September may decrease algal growth on the Minnesota River (Metropolitan Council 2010). However, as with temperature, the impacts of solar energy would not be expected to differ greatly from other large rivers in Minnesota. The use of long-term summer averages for assessment further negates the effects of this factor.

Internal loading. Internal loading is not considered a major contributor to concentrations of TP in most lotic systems. However, under certain circumstances, where external loading and flow are low, internal release of legacy nutrients can occur (Smolders et al. 2017). Research in the lower Minnesota estimates that during low flows, internal release of TP is only 2.1% of total TP loading (James 2007, 2008). Part of the lower section of the Minnesota River is dredged, and in general it is deeper with lower water velocity than the upper and middle sections. As a result, it has greater deposition of sediments and would be expected to have greater internal release of TP compared to upstream reaches. Internal release has not been estimated for the middle and upper reaches of the Minnesota River so it is not known if the flows, dissolved oxygen levels, or sediment characteristics are such that conditions are suitable for internal release of TP. However, the upper and middle reaches of the Minnesota River are likely to have a lower potential for internal release than the lower section due to higher water velocities, relatively high dissolved oxygen, and less deposition of fine sediments.

Internal loading of phosphorus in lakes during late summer can often mask the impact of external loading reductions. The concentration of TP in the Mississippi River at Lock and Dam 3 was determined to be lower during low flows in summer after upstream point source reductions (MPCA 2014). Changes in TP loading in the Minnesota River also indicate that internal loading is not a major contributor of TP during periods when conditions are suitable for algal growth. Since 2006 there have been considerable reductions in TP loading from wastewater facilities in the Minnesota River basin (Figure 6A). Comparison of TP concentrations between the periods 1980-2006 and 2012-2018, indicates a decline in TP concentrations at all flows (Figure 6B). However, the relative decline is greatest at very low flows which can be attributed to decreases in loading from facilities at low flows when they are a greater proportion of loading. There is no indication that internal loading is maintaining TP concentrations during very low flows as might be expected if there was considerable release of TP from river sediments. Seasonal TP dynamics for the Minnesota River are similar to other rivers where high TP loading during high discharge is more attributable to nonpoint sources and loading during low flow periods is more attributable to point sources. This pattern coupled with low internal loading estimates for the lower Minnesota, indicates that internal loading does not need to be considered as part of a SSS review.

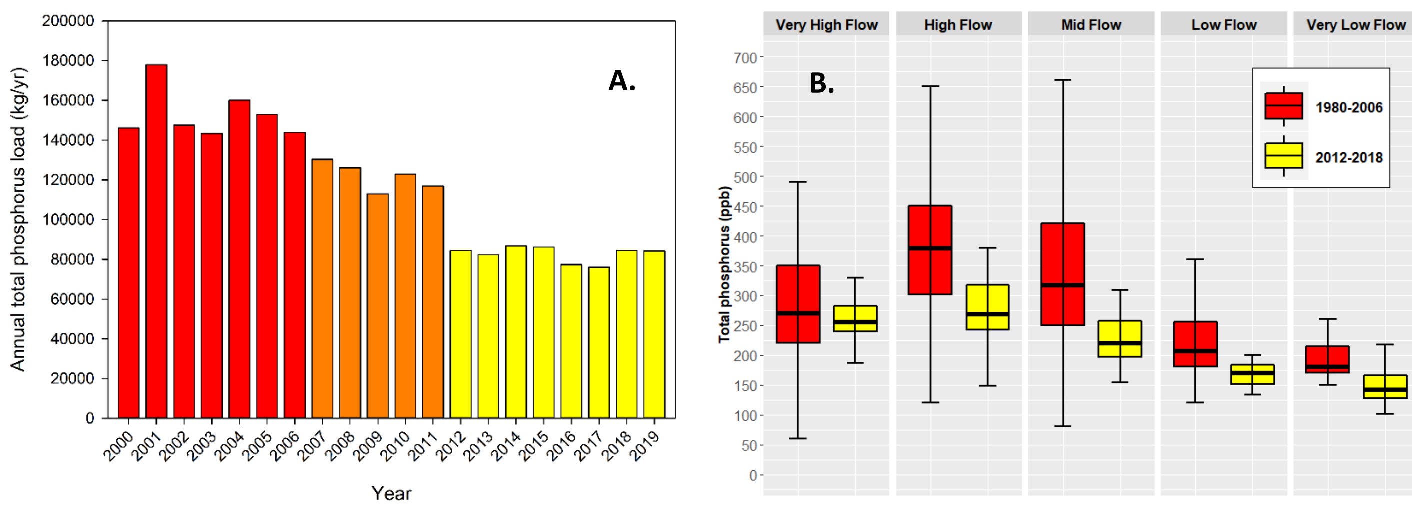

Figure 6: Changes in total phosphorus loading and concentrations in the Minnesota River. A) Total annual loading of total phosphorus by permitted facilities in the Minnesota River upstream of Jordan, MN. B) Box plots of total phosphorus concentrations at different flows for the time periods 1980-1996 and 2012-2018 collected by MCES at river mile 39.4. Description of box plots: solid black line = median; upper and lower bounds (i.e., hinges) = 75th and 25th percentiles; lower whisker = smallest observation greater than or equal to lower hinge - 1.5 * IQR; upper whisker = largest observation less than or equal to upper hinge + 1.5 * IQR; Very high flow = >90th percentile (>18,400 cfs); High flow = 90th-60th percentile (18,400-6,710 cfs); Mid flow = 60th-40th percentile (6,710-3,510 cfs); Low flow = 40th-10th percentile (3,510-783 cfs); Very low flow = <10th percentile (<783 cfs).

Carp bioturbation. Similar to internal loading, the effects of carp on nutrient dynamics are more important in lentic systems. A literature review of the effects of common carp (*Cyprinus carpio*) on nutrient release in lotic systems did not identify this as an important factor. In addition, common carp are also present in other central and southern Minnesota rivers so any potential impact would have been part of the development of the RES. Overall, there is no indication that carp bioturbation impacts nutrient and algal dynamics on the Minnesota

River or that any impact would be different compared to other rivers in Minnesota. Therefore, carp bioturbation does not need to be considered as part of a SSS review.

Transport of algae into the Minnesota River mainstem. In some systems, algae may grow in upstream water bodies and be transported to downstream waters that are not suitable for the growth of undesirable algal blooms. In cases where algae is exported from a productive waterbody to a less productive water body, a decline in the algal population is typically observed (Chandler 1937). To assess this potential in the Minnesota River mainstem, Chl-*a* data from Minnesota River tributaries were examined to determine if they were a major source of algae in the mainstem. There were 81 river and streams in the Minnesota River watershed with 12 or more Chl-*a* samples from 2009-2018. Only nine of these 81 waters exceeded the Chl-*a* standard, and on average Chl-*a* concentrations were 13 µg/L, well below the RES for Chl-*a* of 40 µg/L (Figure 7). This demonstrates that in general, Chl-*a* is lower in the Minnesota River tributaries compared to the mainstem. Based on this assessment, it is apparent that some algae enters the Minnesota River from tributaries, but conditions in the Minnesota River (e.g., increased nutrients, hydrology) result in continued algal growth. Declines in algae are observed downstream in the dredged portions of the Minnesota River (Metropolitan Council 2010), but the unimpounded and undredged sections of the Minnesota River are very productive. Furthermore it is important to note that upstream loading of Chl-*a* does not result in inappropriate application of the RES because regardless of the source of Chl-*a*, the presence of high levels of algae negatively impacts beneficial uses. The source of Chl-*a* does become relevant to the implementation total maximum daily loads (TMDL) and permits because the location of algal growth can be important for its control. However, the source of the algae is not relevant to the standard itself and as described above, external loading of algae is not a major source of Chl-*a* in the Minnesota River.

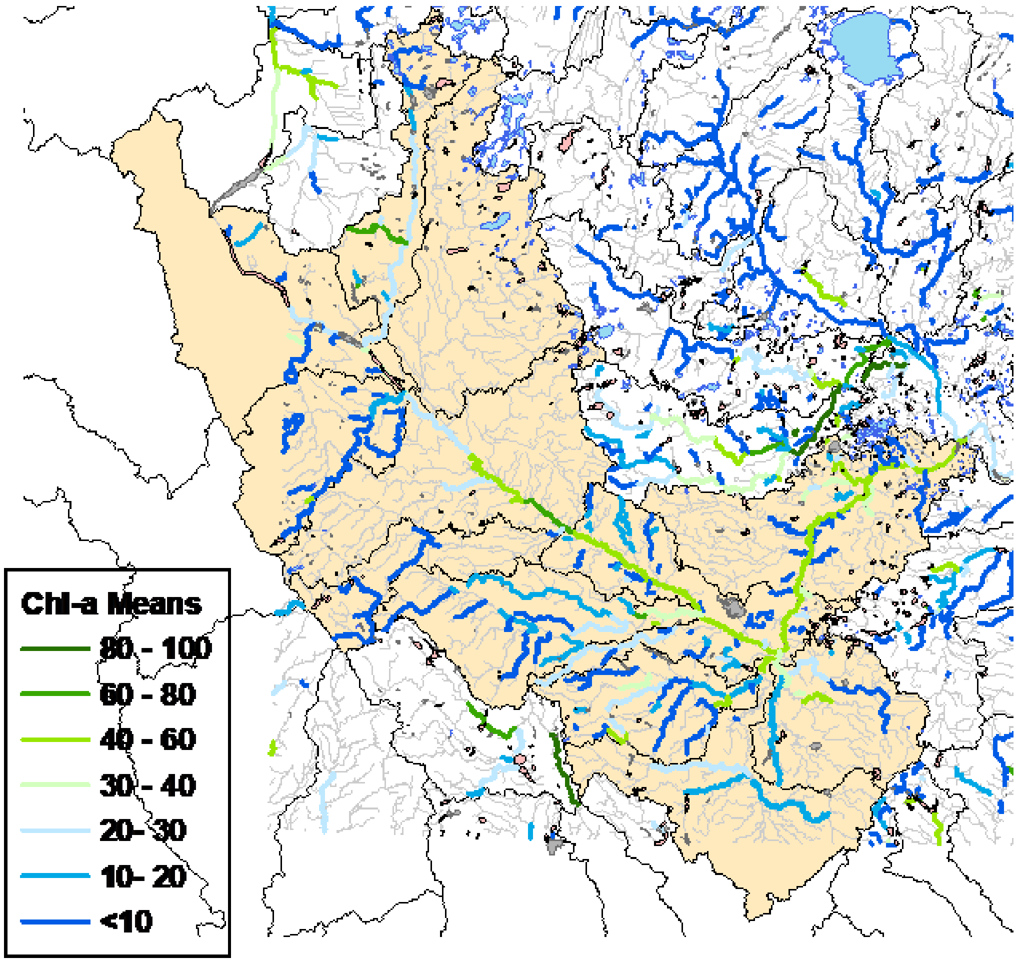

Figure 7: Summer average chlorophyll-a in the Minnesota River watershed (shaded) from 2009-2018.

Multiple factors

Multiple factors affect algal growth in the Minnesota River and it is important to consider these together. As in other Minnesota rivers, phosphorus, residence time, and shading are the most important factors affecting algal concentrations during the summer index period. Other factors such as temperature and solar radiation also affect algal growth, but there is limited anthropogenic control over these factors and they are already considered as part of the RES framework (i.e., regionalization of standards). When the RES was developed, these multiple factors were considered as part of standard development and the development of procedures for implementing the standard. Specifically, the RES is based on an assessment of annual averages of data from the summer index period over multiple years. This recognizes the fact that individual daily measurements of eutrophication parameters are not sufficient to characterize the trophic condition of a river and analysis of data at this scale can lead to misinterpretations of trophic relationships. Depending on the time of sampling, different factors may be limiting the growth of algae, and by estimating a long-term average trophic condition, this reduces variability and accounts for the effects of multiple factors.

To document how the Minnesota River is similar to the other rivers in Minnesota used to develop the RES, it is useful to examine relationships between eutrophication parameters at different temporal scales. As described in the “Phosphorus” section, TP does not or only rarely becomes limiting for algal growth. This relationship is further illustrated in Figure 8. Chl-*a* concentrations remain high in the Minnesota River (Figure 8A) at low flows when TP concentrations are lower. The relationship observed in the Minnesota River may appear to be counterintuitive (i.e., greater algal growth during periods with lower TP concentrations), but it is fully consistent with the ecological underpinnings of the RES. The interpretation that there is a negative relationship between long-term summer average measurements of TP and Chl-*a* in the Minnesota River is incorrect and ignores the effects of other factors that control algal growth. In the Minnesota River, residence time or shading are important for limiting algal growth during high flows and these factors can confound the relationship between daily measurements of TP and Chl-*a*. The use of long-term summer averages reduces seasonal variability and the effects of these other factors. This is why many of the data analyses which support the RES and the data requirements for assessing attainment of the RES are based on long-term summer averages. When long-term summer averages are calculated for the Minnesota River, these data are consistent with the statewide relationship (see Figure 3). Since there is currently no indication that the Minnesota River behaves differently compared to other rivers in terms of eutrophication dynamics, the original RES model and the large river-only model (Figure 3) are sufficient to demonstrate that the current RES applied to the Minnesota River is appropriate.

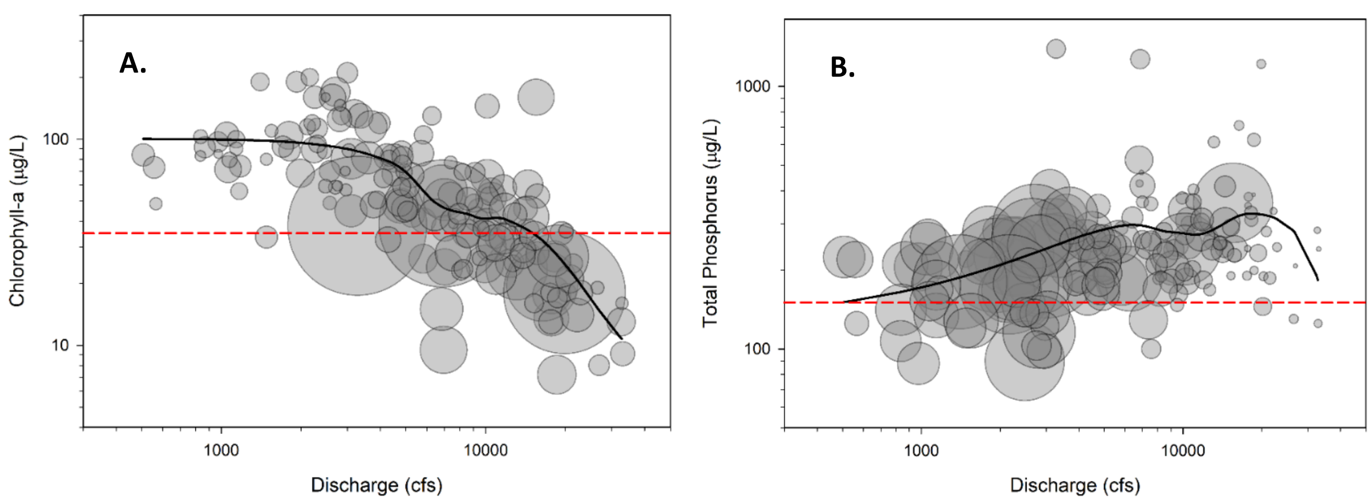

Figure 8: Bubble charts of daily measurements of discharge, chlorophyll-a, and total phosphorus for the Minnesota River (-723) River. Bubble size indicates magnitude of the third parameter: A) total phosphorus and B) chlorophyll-a. Regression fits are LOESS regressions (span = 0.75, degree = 2). Red dotted line is the chlorophyll-a standard.

Conclusions

Residence time/shading and TP are the most important factors controlling the growth of suspended algae in the Minnesota River. The RES is based on long-term, summer averages, so individual daily or even summer averages from single years are not necessarily predictive of attainment of the RES. In most Minnesota rivers, TP concentrations are greatest at high flows while Chl-*a* concentrations are low under these conditions. In enriched river systems, algal growth is low at high flows because residence time or shading limits the growth of algae despite surplus nutrients. At lower discharge, residence time or shading is no longer limiting, and although TP concentrations often decline, undesirable algal blooms will occur if TP concentrations are still sufficient to support high algal growth. In some systems, such as the Red River, shading due to turbidity rarely becomes low enough to support high algal growth even at lower flows. Based on a review of these factors, the current RES applicable to the Minnesota River is appropriate and a SSS is not needed. Furthermore, the new large river TP-Chl-*a* model developed as part of this review can be used to improve certainty regarding the outcomes of reductions of TP in the Minnesota River mainstem. Attainment of the RES in the Minnesota River requires a reduction of TP concentrations with the most important reductions at moderate to low flows. Nutrient management strategies for the Minnesota River will also need to consider other relevant standards. For example, the Mississippi River and Lake Pepin eutrophication standards (long-term summer averages) and the Minnesota River low dissolved oxygen TMDL (lowest flows in August and September) require the Minnesota River to be at or below 150 µg/L TP (Heiskary and Wasley 2012).

References

Chandler, D.C. 1937. Fate of typical lake plankton in streams. Ecological Monographs. 7:445-479.

Heiskary, S., and R.W. Bouchard, Jr. 2015. Development of eutrophication criteria for Minnesota streams and rivers using multiple lines of evidence. Freshwater Science. 34:574-592.

Heiskary, S., R.W. Bouchard Jr. and H. Markus. 2013. Minnesota Nutrient Criteria Development for Rivers. Environmental Analysis and Outcomes, MPCA, Saint Paul, Minnesota.

Heiskary, S. and D. Wasley. 2012. Mississippi River Pools 1 through 8: Developing River, Pool, and Lake Pepin Eutrophication Criteria, MPCA, Saint Paul, Minnesota.

James, W.F. 2007. Nutrient Dynamics and Budgetary Analysis of the Lower Minnesota River: 2003-2006; Eau Galle Aquatic Ecology Laboratory, U.S. Army Engineer Research and Development Center; Spring Valley, Wisconsin.

James, W.F. and C.E. Larson. 2008. Phosphorus dynamics and loading in the turbid Minnesota River (USA): controls and recycling potential. Biogeochemistry. 90: 75-92.

Kleinteich, J., S. Hilt, A. Hoppe, and C. Zarfl. 2019. Structural changes of the microplankton community following a pulse of inorganic nitrogen in a eutrophic river. Limnology and Oceanography.

Koenker, R. 2009. quantreg: Quantile Regression. R package.

Metropolitan Council. 2010. Lower Minnesota River Study: Monitoring and Modeling Water Quality From Jordan, Minnesota, to the Mouth. Metropolitan Council, Saint Paul, MN.

Metropolitan Council Environmental Services (MCES). 2018. Regional Assessment of River Water Quality in the Twin Cities Metropolitan Area (1976 – 2015). Metropolitan Council, Saint Paul, MN.

Miltner, R.J., and E.T. Rankin. 1998. Primary nutrients and the biotic integrity of rivers and streams. Freshwater Biology. 40:145-158.

MPCA 2014. Minnesota Nutrient Reduction Strategy. Minnesota Pollution Control Agency, St. Paul, MN.

Paerl, H.W., J.T. Scott, M.J. McCarthy, S.E. Newell, W.S. Gardner, K.E. Havens, D.K. Hoffman, S.W. Wilhelm, and W.A. Wurtsbaugh. 2016. It takes two to tango: When and where dual nutrient (N & P) reductions are needed to protect lakes and downstream ecosystems. Environmental Science & Technology. 50(20): 10805-10813.

R Development Core Team. 2019. R: A language and environment for statistical computing. Vienna, Austria.

Reynolds, C.S. 2000. Hydroecology of river plankton: the role of variability in channel flow. Hydrological Processes. 14(16-17): 3119-3132.

Schindler, D.W., R.E. Hecky, D.L. Findlay, M.P. Stainton, B.R. Parker, M.J. Paterson, K.G. Beaty, M. Lyng, and S.E.M. Kasian. 2008. Eutrophication of lakes cannot be controlled by reducing nitrogen input: Results of a 37-year whole-ecosystem experiment. Proceedings of the National Academy of Sciences of the United States of America. 105: 11254–11258.

Schindler, D.W., S.R. Carpenter, S.C. Chapra, R.E. Hecky, and D.M. Orihel. 2016. Reducing phosphorus to curb lake eutrophication is a success. Environmental Science and Technology. 50: 8923-8929.

Smolders, E., E. Baetens, M. Verbeeck, S. Nawara, J. Diels, M. Verdievel, B. Peeters, W. De Cooman and S. Baken. 2017. Internal loading and redox cycling of sediment iron explain reactive phosphorus concentrations in lowland rivers. Environmental science & technology. 51(5): 2584-2592.

State of Minnesota. 2014. The Minnesota Nutrient Reduction Strategy. Minnesota Pollution Control Agency, Saint Paul, MN.

Talling, J., R. Wood, M. Prosser, and R. Baxter. 1973. The upper limit of photosynthetic productivity by phytoplankton: evidence from Ethiopian soda lakes. Freshwater Biology. 3:53-76.

Van Nieuwenhuyse, E.E. and J.R. Jones. 1996. Phosphorus-chlorophyll relationship in temperate streams and its variation with stream catchment area. Canadian Journal of Fisheries and Aquatic Sciences. 53: 99-105.

Wang, L., D.M. Robertson, and P.J. Garrison. 2007. Linkages between nutrients and assemblages of macroinvertebrates and fish in wadeable streams: implication to nutrient criteria development. Environmental Management. 39:194-212.

Wetzel, R.W. 2001. Limnology: lake and river ecosystems. Academic Press, San Diego, CA.