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Applications of Reservoir Limnology Theory and Steady-State Modeling to Eutrophication Management in Beaver Lake, Arkansas

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Applications of Reservoir Limnology Theory and Steady-State Modeling to Eutrophication Management in Beaver Lake, Arkansas

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ABSTRACT

Reservoir limnology theory predicts that phytoplankton biomass (PB) is greatest in riverine-transition zones and least in lacustrine zones leading to an inverse pattern in water clarity. These theoretical patterns were utilized to create a statistical model of chlorophyll-a (Chl-a), an indicator of PB, and Secchi transparency (ST), an indicator of Chl-a, in Beaver Lake, Arkansas, a 12,800-ha reservoir, in order to hindcast historical conditions. Sampling for Chl-a, ST, and photic depth occurred semimonthly at 12 locations along a 78-km transect from the river inflow to the dam during the 2015 growing season. The ratio of Chl-a and ST measured at each site to the Chl-a and ST measured at the dam ($FracDAM_{Chl-a}$ and $FracDAM_{ST}$, respectively) were computed for each sampling date, and regression models were developed to predict $FracDAM_{Chl-a}$ and $FracDAM_{ST}$ as a function of distance from reservoir inflow. The models were used to estimate Chl-a ($r^2 = 0.83$, $p = 0.0003$) and ST ($r^2 = 0.98$, $p < 0.0001$) at any location in the lake in years where spatially explicit data were not available. United States Geological Survey (USGS) monitoring data collected at the dam, and three additionally overlapping sites, from 2001 to 2015 were used to develop and test the hindcast models. Residuals of the modeled-measured USGS data suggested that variation in hydrology across years created predictable interannual variation in the spatial patterns in Chl-a and ST across the riverine-lacustrine continuum. Whole-lake averages of Chl-a and ST were related to whole-lake total phosphorus (TP) for modeled, measured, and target data sets between 2001-2015 for the purposes of estimating Vollenweider P loads. The most important finding of this study revealed that average of modeled and measured P loading required reductions of 18.9% and 33.3% to meet the newly adopted Beaver Lake Chl-a and ST standards, respectively.

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1. PROPOSAL AND LITERATURE REVIEW

1.1 Introduction

River impoundment reservoirs are generally constructed for flood control, irrigation, electrical power generation, aquaculture, recreation, water supply, or some combination of these uses. Reservoirs are relatively young compared to their natural lake counterparts, as most reservoirs in the US are less than 80 years old (Thornton et al. 1990), while many natural lakes may range in age from 100 – 10,000 years (Wetzel 2001). Unlike natural lakes, the explicit study of reservoir limnology theory (RLT) is also fairly new. Until 1990, when the seminal text on RLT was published (Thornton et al. 1990), much of what was known to the western world about the water quality of reservoirs had been garnered from classic natural lake studies. Before 1990 most research on reservoir limnology was contained in a three-volume text (Hrbáček 1966, Hrbáček and Straškraba 1973a and b) reporting the long-term study of Czech hydroelectric/drinking water reservoirs (Kalff 2002). In the United States, large reservoir construction began earnestly in the early 20th century and peaked in the 1960s (Cech 2010), with almost 74,000 large dams (> 2m tall) existing in the U.S. today (Figure 1). While natural lakes and reservoirs have biotic and abiotic factors in common, there are important distinctions that make them differ from one another. Reservoirs are often characterized by three distinct hydrologic zones that affect their chemistry and biology: the riverine zone, the transitional zone, and the lacustrine zone (Figure 2).

The riverine zone is a near-lotic environment at the most up-reservoir end of the continuum. The riverine zone is characterized by a narrow, channelized basin with relatively short water residence time, greater suspended solids, greater nutrient concentrations, lesser light availability, and lesser Secchi transparency (ST) when compared to the other zones. In the transitional zone, flow velocity slows as the basin becomes wider and deeper. Thus, suspended solids concentrations are less in this zone and light penetration through the water column is greater, allowing for greater primary production. The lacustrine zone is characterized by morphometric characteristics similar to natural lakes with a broad and deep basin. Here, the flow is dramatically reduced, suspended sediments have largely fallen out of the water column contributing to lesser nutrient content, less light extinction, and the greatest ST. The exact location of these zones differs among reservoirs and even within reservoirs, depending on weather and flow conditions (Brooks et al. 2009). A recent study of eight Texas reservoirs containing 85 sampling sites across these zones showed that chlorophyll-a (Chl-a; an indicator of phytoplankton biomass) concentrations were least at lacustrine sites (n=29), greatest at transitional sites (n=48), and near-median values at up-reservoir sites (n=43) with values of 14, 28, and 22 µg/L respectively (Forbes et al. 2012).

Eutrophication is caused by excess inputs of nitrogen (N) and phosphorus (P) from either point sources or non-point sources within the watershed. Point sources of nutrients are directly attributable to one influence and can be linked to sources such as industrial or municipal wastewater effluent. Non-point sources are more diffuse in origin, not easily attributable to one single source, and can be associated with varying land uses such as agricultural or urban runoff. Carlson (1977) developed a numerical trophic state index (TSI) that ranges from 0 to 100 where each factor of 10 represents an approximate doubling of phytoplankton biomass, which estimates the degree of eutrophication in lakes and reservoirs from simple water quality measurements. Carlson’s (1977) trophic state index states that Chl-a values in the 0-2.6 µg/L range are designated as oligotrophic (low primary productivity), ranges of >2.6 up to 20 µg/L are designated as mesotrophic (intermediate primary productivity), and >20 µg/L are designated as eutrophic (high primary productivity).

Worldwide, water quality (WQ) has been degraded in many freshwater reservoirs by human-induced eutrophication. Dodds et al. (2008) calculated potential annual value losses in recreational water usage, waterfront real estate, spending on recovery of endangered and threatened species, and drinking water were approximately $2.2 billion annually as a result of eutrophication in U.S. freshwaters. Eutrophication of surface waters can occur when inorganic nitrogen (ammonia plus nitrate nitrogen) is ≥ 300 µg/L of N (Sawyer et al. 1945) and when P concentrations of ≥ 10µg/L are present (Wetzel 2001).

Phosphorus and Eutrophication in Beaver Lake, Arkansas

Beaver Lake (BL), an artificial, man-made reservoir, is an impoundment of the White River that serves as a flood control structure, power generation source, and water supply for almost 0.5 million residents of Northwest Arkansas. The State of Arkansas has recently adopted effects-based WQ criteria to protect BL against accelerated eutrophication (APCEC 2012). This approach examines the effect that nutrient inputs into the catchment have on WQ conditions such as Chl-a and ST. The Arkansas Department of Environmental Quality (ADEQ), beginning in 2016, assessed WQ in BL at Hickory Creek (HC), which is just upstream of the first of four drinking water utility intake structures. The HC site is physically located near where the riverine and transitional zones converge (Figure 3). According to the State of Arkansas Regulation 2 (Reg. 2; APCEC 2012), the newly adopted effects-based WQ criteria for BL state that the growing season geometric mean for Chl-a (Chl-a_{GSGM}) shall not exceed 8 µg/L nor shall the annual average ST (ST_{AA}) be less than 1.1 meters at HC, respectively. The growing season in Arkansas is defined by Reg. 2 as the period between May 1 and October 31.

The effects-based WQ criteria that were adopted for BL by the State of Arkansas were based on recommendations of a working group that conducted a multi-tiered analysis (FTN 2008). According to FTN (2008), recommended target values for Chl-a were based on expected long-term averages of the Chl-a_{GSGM} and ST_{AA} at HC (Scott and Haggard 2015) derived by modeling typical reservoir patterns in Chl-a, ST and total phosphorus (TP). The typical assessment method by the ADEQ says that water bodies must meet standards in at least 80% over a five-year period (Scott and Haggard 2015). However, by defining the standards based on long-term averages, and assuming normal data distributions, half of all water quality assessments on BL would result in an impairment status. Nevertheless, even if the assessment method is made more appropriate to standards, BL is likely to exceed standards and be listed as impaired in future assessment.

Few studies of spatial and temporal aspects of WQ gradients have been addressed in BL, Arkansas. Those that have primarily isolated and inspected one or two of the reservoir's zones. In a study investigating the influence of rainfall in taste and odor production in the riverine/transitional area of BL, Winston et al. (2014) reported that annual average ST was reduced from 2.1 to 1.6 m, and average annual TP increased from 17 to 23µg/L between 2007 and 2008, which were relatively wet and dry years, respectively. Average annual ST values ranged from 0.3 – 3.3 m and average annual TP values ranged from 5 – 73 µg/L among the two-year study (Winston et al. 2014). When assessing bi-monthly WQ gradients of the headwater reaches of BL, Arkansas, Haggard (1999) found that mean annual Chl-a (7.4 to 3.6µg/L) and average annual ST (1.4 to 1.0 m) decreased along the riverine-transitional-lacustrine zones while average annual TP concentrations increased (36.1 to 45.2 µg/L) along the same continuum. Even though there have been sporadic efforts to quantify ST, Chl-a, and TP in various locations along the reservoir continuum, surprisingly, there have not been consistent efforts to capture spatially-explicit data along the BL riverine-transitional-lacustrine zone. This fact is surprising considering that Thornton et al. (1990) used BL as a basis by which they developed their reservoir limnology concepts.

Phosphorus, Chlorophyll-a, and Secchi Transparency Relationships in Reservoirs

A relationship for predicting the summer time levels of Chl-a and TP in 143 north-temperate lakes (Jones and Bachmann 1976) was first developed so that a regression line can be used to predict average Chl-a or TP when the other value is known:

Equation 1 $$\log TP_C = \frac{\log CHLA + 1.09}{1.46}$$

where TP_c is the average annual concentration of TP (µg/L) (TP_{AA}) and *Chl-a* is the Chl-a_{GSGM} in the upper mixed layer of a lake. The model of Jones and Bachmann (1976) assumes that P is the element controlling algal biomass in a broad geographic distribution of lakes.

Carlson (1977) showed that for lakes where ST is controlled primarily by phytoplankton biomass, a P – ST relationship (Carlson 1977) was also applicable based on the equation:

Equation 2 $$TP_c = \frac{48}{ST}$$

where TP_c is the TP_{AA} (µg/L) in the upper mixed layer of the lake and ST is the ST_{AA} (m) of the lake. These models can be applied effectively to lakes and reservoirs that conform to these relationships in order to identify the TP concentrations that result in desirable ST or Chl-a values. WQ lake managers and fisheries managers should find the value and effectiveness when making cost-benefit analysis of nutrient reduction programs to reduce algal densities (Jones and Bachmann 1976).

Steady State Models

Steady state limnological models can be used to link the average annual P concentration in lakes to P loading rates from the watershed (Vollenweider 1968, 1976). Limnological steady state models rely on the assumption that the capacity of a water body to gain or lose a certain nutrient over time is zero and the models do not take into account aspects such as losses by outflow or gains by sedimentation (Wetzel 2001). Steady state models assume that nutrient load is completely and instantaneously mixed throughout the water body. Although this doesn't occur in reality, lakes that receive relatively constant nutrient loads over many years can be viewed in a steady state condition (Wetzel 2001). Vollenweider and Kerekes (1982) developed a regression analysis that best describes how TP concentration, water residence time, and water discharge affect external P loading in low-humic lakes (Kalff 2002):

Equation 3
$$L_{Pc} = \left(\frac{TP_c}{1.55}\right)^{\frac{1}{0.82}} \left(1 + \sqrt{\tau_w}\right) q_s$$

where L_{Pc} is the critical P loading rate (mg/m^2/yr), TP_c is the TP_{AA} (µg/L) in the upper mixed layer of the lake, τ_w is the water residence time (yr), and q_s is the annual water loading rate (m/yr).

Alternative models can be used to calculate the critical P loading rate (L_{Pc}). Canfield and Bachmann (1981) developed a regression analysis that expanded on the work of others that predicts L_{Pc} with an empirical estimate for P sedimentation by examining the P input-output relationship in 704 artificial and natural lakes. Their refined model was:

Equation 4
$$L_{Pc} = \frac{TP_c(0.257z + z\rho)}{0.603}$$

where L_{Pc} is the annual P loading per unit of lake surface area (mg/m^2/yr), TP_c concentration of TP in the reservoir (mg/m^3), z is mean depth of any particular section of reservoir (m), and ρ is the hydraulic flushing rate (yr).Thus, if water quality targets exist for ST or Chl-a for a lake or reservoir, a critical TP concentration can be estimated using the Chl-a-TP-ST relationships presented above (Equations 1 and 2), and the steady state models (Equations 3 and 4) can then be used to estimate the annual P loading rate that will result in the critical TP values. Regional WQ targets and assessment methodologies may vary, but throughout the Mid-South and Southeast regions of the U.S., WQ criteria ranges from 1.5 to 27 µg/L (EPA 2014).

One important aspect that has been poorly studied in BL is the understanding of the horizontal, vertical, and temporal gradient patterns of Chl-a, ST, and TP along the riverine-transitional-lacustrine continuum. With this thesis I aimed to explore reservoir limnology patterns of Chl-a, ST, TP, photic depth (Z_{EU}), and reservoir morphometry in BR in order to derive whole-lake estimates relating these variables (Equations 1 and 2) and compute P loading estimates from steady state models (Equations 3 and 4). The goal was to recreate P loading estimates for 2001-2014, for which reliable Chl-a, ST, and TP data were available for Bl. Phosphorus loading values that relate to newly adopted effects-based WQ targets were also calculated. The difference between measured and target P load values generated a P load reduction estimate within the BL watershed.

Objectives

The objectives of this thesis were as follow:

1. Using the Reservoir limnology theories discussed in the introduction, I derived linear regressions of the spatial patterns in Chl-a, and ST, Z_{EU}, and reservoir morphometry across the riverine-transitional-lacustrine gradient in BL.

2. I used empirical models derived in objective 1 to predict Chl-a and ST along the riverine-transitional-lacustrine gradient using measured Chl-a and ST values at the dam for those years.

3. I used whole-lake Chl-a and ST, computed from work in objective 1 and 2 to relate whole-lake Chl-a and ST with TP using common limnologic models (Equations 1 and 2).

4. I computed whole-lake Chl-a and ST from empirical models that relate to the 8µg/L Chl-a and 1.1m ST WQ standards that apply to BL at HC.

5. Using common limnologic models, I compared historical P loads of historic modeled data from objective 3 and WQ target data from objective 4 to derive P load reduction estimates needed to meet Chl-a and ST WQ standards in BL.

1.2 Hypotheses

1. I hypothesized that the spatial patterns of Chl-a, ST, Z_{EU}, and reservoir morphology along the riverine-transitional-lacustrine gradient of BL would closely conform to RLT (Thornton et al. 1990) and would result in statistically-valid regression models.

2. I hypothesized that empirical models based on reservoir limnology theory could reasonably predict Chl-a and ST throughout the reservoir based on a Chl-a or ST value at the dam because spatial variation in Chl-a and ST in BL are stronger than inter-annual differences.

1.3 Methods

1.3.1 *Study Locations and Sampling Methods*

Beaver Lake is located in the Northwest Arkansas counties of Washington, Benton, and Carroll and is the most upstream in a series of three reservoirs along the White River in Arkansas and Missouri (BWD 2010). Beaver Lake Watershed includes seven major subwatersheds that encompass more than 3095 km^2 with land use dominated by forests (71%) and agriculture (22%). Construction on the dam began in 1959 and by 1966, the lake level reached conservation pool elevation (BWD 2010). Allocations of the conservation pool amount to ~79% for hydroelectric power, and ~21% for drinking water supply (BWD 2010). Beaver Reservoir serves nearly one in seven Arkansans (BWA 2015, USDOC 2015) with their drinking water through one of four public water utilities: Beaver Water District, Benton-Washington Regional Public Water Authority, Carroll Boone Water District, and Madison County Regional Water District (BWD 2010).

Sites were sampled along a ~80 kilometer transect that began at the White River and HWY 412 bridge and ended at the reservoir dam (Figure 3). Twelve sites were sampled evenly across the riverine, transitional, and lacustrine zones, respectively. The United States Geological Survey (USGS) has been sampling four of the twelve sites on an approximately month to bi-monthly basis since 2001. The site located at the dam has been sampled by the USGS monthly since 1973 and was sampled additionally throughout the 2015 growing season for this thesis. All sites were sampled twice monthly, around the 1st and 15th of the month between May 1 and October 31st, 2015, the defined growing season in Arkansas (APCEC 2012). Sites along the continuum ware sampled in the thalweg of the old White River channel to limit unwanted biological activity in low-velocity areas. Secchi transparency was measured using a standard 20 cm Secchi disk and calculated as the average of the two depth measurements as the Secchi disk was lowered and when the disk was raised (Lind 1985). The depth of the photic zone was calculated using a LI-COR® Li-193 Underwater Quantum Sensor attached to a LI-COR® 2009S lowering frame and connected to a LI-COR® Li-250A Light Meter by a 30m LI-COR® 2222UWB communications cable. Depth of photic zone was determined by lowering the sensor and recording illumination values in 0.5m increments. Z_{EU} was calculated as the depth of the water column receiving 1% of the surface illumination measured as photosynthetically active radiation. The photic zone was divided into equally spaced depths (Table 1) depending on depth, and each depth sampled using a Wildlife Supply Company model E-411-19XX-G62 horizontal water sampler. Samples were transferred into acid-washed and rinsed 1L UV-resistant amber HDPE bottles, stored on ice and returned to the lab.

1.3.2 *Chl-a and TNTP Laboratory Processing Methods*

Within 24 hours of collection, a 100mL sub-sample was preserved by freezing in a 125mL amber HDPE bottle for later Total Nitrogen (TN) and Total Phosphorus (TP) analysis. Volumetrically measured portions (100 – 750 mL) of a well-shaken sample were filtered under vacuum pressure using a 0.7µm pore size, 25 mm diameter Watmann® GF/F filters with enough site water so that color is evident on the filter. Filters were folded in half so that the plankton sides were touching, wrapped in aluminum foil, and freezer stored for Chl-a analysis. Chl-a was analyzed using a Turner Trilogy fluorometer following an acetone extraction using Method #10200H (APHA 2005).

Preserved TN and TP sub-samples were thawed, and digestions were performed in the Scott Lab following Method #4500-P J (APHA 2005). Briefly, TN and TP samples were digested using a potassium persulfate oxidizing solution prior to autoclaving at 120°C for 55 minutes. Following autoclaving, phenolphthalein dye indicator was added to aid in visual pH corrections of all samples to a neutral pH of 7. Following digestions and pH adjustment, TN and TP samples were transported to Arkansas Water Resource Center where TN and TP were analyzed using the Cadmium Reduction Method (4500-NO2-B) (APHA 2005) and Automated Ascorbic Acid Reduction Method (4500-P F) (APHA 2005) on a Skalar San ++ Continuous Flow Analyzer (Skalar 1995).

1.3.3 *Data Manipulations and Statistical Analysis*

By having Chl-a and ST data sets at along the entire continuum, a set of values were determined which were the fraction of Chl-a and ST of all other sites that were relative to the fraction at the dam (FracDAM), respectively. I divided the Chl-a_{GSGM} computed for all sites by the Chl-a_{GSGM} at the dam (Chl-$a_{GSGM-DAM}$) to derive the fraction of Chl-a at the dam ($FracDAM_{Chl-a}$). The $FracDAM_{Chl-a}$ values for the dam were always 1. The same approach was used to calculate the fraction of ST at the dam ($FracDAM_{ST}$) from the ST_{AA} for all sites and ST_{AA} at the dam (ST_{AA-DAM}). A linear regression analysis was used to relate distance from the dam to the $FracDAM_{Chl-a}$ and $FracDAM_{ST}$ (Figure 4). I derived predicted Chl-a_{GSGM} and ST_{AA} for USGS monitoring sites from 2001-2015 by multiplying $FracDAM_{Chl-a}$ and $FracDAM_{ST}$ from this regression model by the Chl-a_{GSGM} and ST_{AA}, respectively, measured at the dam in these years. A linear regression analysis was conducted on predicted versus measured data to assess the model utility. Once the model was deemed acceptable (i.e., $p<0.05$ in test of slope $\neq 0$), a linear regression was derived to predict Z_{EU} and reservoir width from distance from the dam. Knowing the average annual Z_{EU} and reservoir width of all sites allowed me to compute a fractional photic volume (m^3) of each segment and a subsequent weighted average for whole-lake Chl-a (Chl-$a_{GSGM-WL}$) and ST (ST_{AA-WL}).

In order to test the first hypothesis of whether year-to-year WQ gradients in BL closely conformed to other regionally hypothesized reservoir productivity gradients, a repeated measures ANOVA was used where sampling date (temporal gradient) was the repeated measure and site (horizontal gradient) and depth (vertical gradient) were the main effects. The linear regression models discussed above provided a direct test of the second hypothesis regarding the control of RLT of spatial variation in Chl-a and ST in BL.

To address objective 3, Chl-$a_{GSGM\text{-}WL}$ and $ST_{AA\text{-}WL}$ were used to compute growing season geometric mean TP for the whole lake ($TP_{GSGM\text{-}WL}$) and annual average TP for the whole lake ($TP_{AA\text{-}WL}$), respectively, from Equations 1 and 2. To address objective 4, I divided the 8µg/L Chl-a standard value and the 1.1m ST standard value by the $FracDAM_{Chl\text{-}a}$ and $FracDAM_{ST}$ values, respectively, for the HC site in order to estimate the corresponding Chl-$a_{GSGM\text{-}DAM}$ and $ST_{AA\text{-}DAM}$. The linear regression models were then applied to derive estimates of Chl-$a_{GSGM\text{-}WL}$ and $ST_{AA\text{-}WL}$ that correspond to the WQ standards. To address objective 5, $TP_{GSGM\text{-}WL}$ and $TP_{AA\text{-}WL}$ estimated for the USGS data were used to compute annual P loads from the watershed based on the 2001-2015 USGS data and also the BL WQ standards. The difference in P loading estimated from the measured data and that estimated from WQ targets provided a direct estimate of load reductions needed to meet the WQ targets for BL.

1.4 Results

Based on the dendritic morphology of BL and the influence that morphometry has on reservoir functioning, I expected that there would be empirical models that could be applied to the reservoir that support spatial patterns of Chl-a, ST, Z_{EU}, and the morphometry along the riverine-transitional-lacustrine continuum (Figure 5 and 6). As shown in Figure 1, the riverine section is marked by the greatest flow and greatest suspended sediments within its narrow basin.

I expected that the greatest suspended sediments will limit light penetration into the water column therefore restricting Chl-a in the riverine zone compared to the transitional section. In the transitional section where reduced flows are caused by the widening of the basin and subsequently less suspended solids, I expected that Chl-a would be greatest of all three reservoir zones and that Chl-a would be least in the lacustrine zone due to low velocity and nutrient transport. I further expected that ST patterns would be least in the riverine because of greater suspended sediments, slightly elevated in the transitional zone due to reduced flows contributing to reduced suspended sediment, and the greatest ST wold be observed in the lacustrine sites where least suspended sediments within the system would allow for greatest light penetration into the water column.

I also expected that based on a steady state model for BL, that year-to-year patterns of Chl-a_{GSGM} and ST_{AA} would be similar. This was accomplished by designing an empirical model from fractional multipliers (Table 1) to test the year-to-year spatial variability of ST and Chl-a along the BR gradient. I expected that these empirical models would allow me to report that year-to-year patterns of Chl-a and ST in BR are similar enough for the purposes of multi-year comparisons as hypothesis 2 states.

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1.6 Tables

Table 1. Number of samples taken at varying photic depths (m)

Photic depth (m)	Number of samples taken
0 – 0.9	1
1 – 2.9	2
3 – 4.9	3
5 – 6.9	4
7 – 8.9	5
9 – 10.9	6
11+	7

1.7 Figure Legends

Figure 1. Large dams in the United States by completion date. Modified from NID 2015.

Figure 2. Horizontal zonation in environmental factors controlling primary productivity, phytoplankton biomass, and trophic state within reservoir basins. Modified from Kimmel and Groeger (1984).

Figure 3. Site number, distance from Inflow (km), latitude, and longitude of sampling locations in Beaver Lake, AR.

Figure 4. Conceptual flow diagram of objectives and equations for making annual phosphorus load reduction estimations.

Figure 5. Theoretical regression of the spatial patterns of chlorophyll-a within the reservoir continuum and how each site is relative to a location near the dam. All sites will be sampled in 2015 for this thesis while boxes indicate historic USGS sites.

Figure 6. Theoretical regression of the spatial patterns of Secchi transparency within the reservoir continuum and how each site is relative to a location near the dam. All sites will be sampled in 2015 for this thesis while boxes indicate historic USGS sites.

1.8 Figures

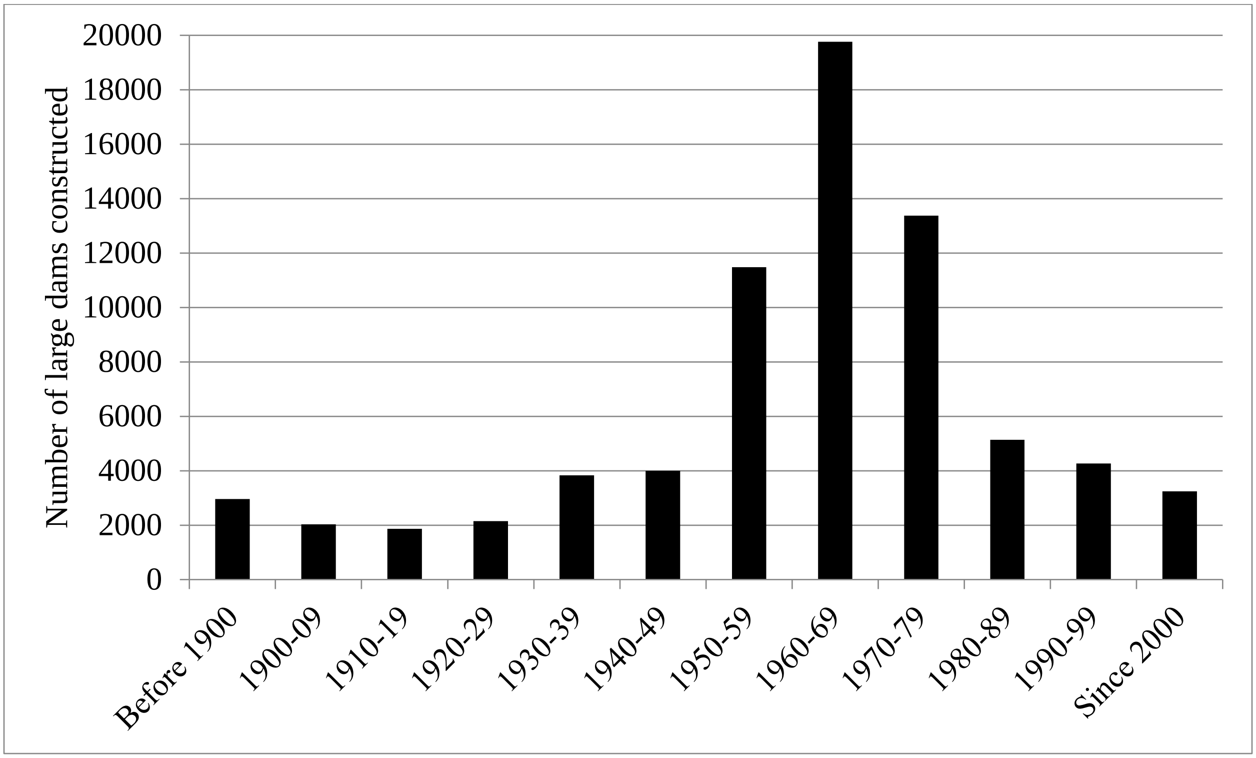
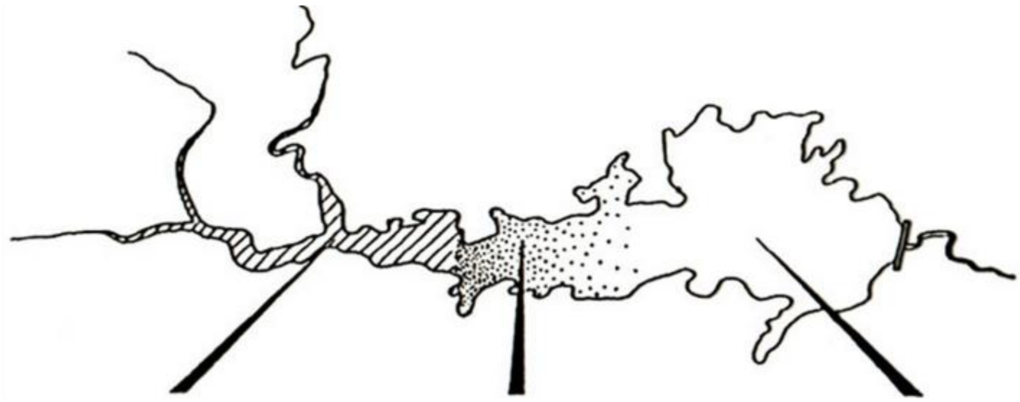
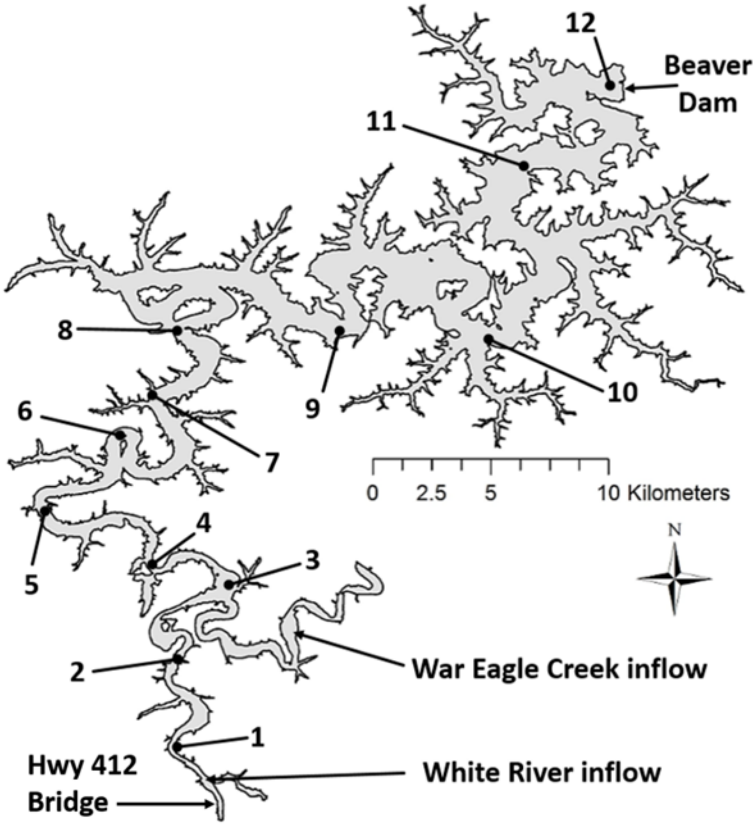


Figure 1



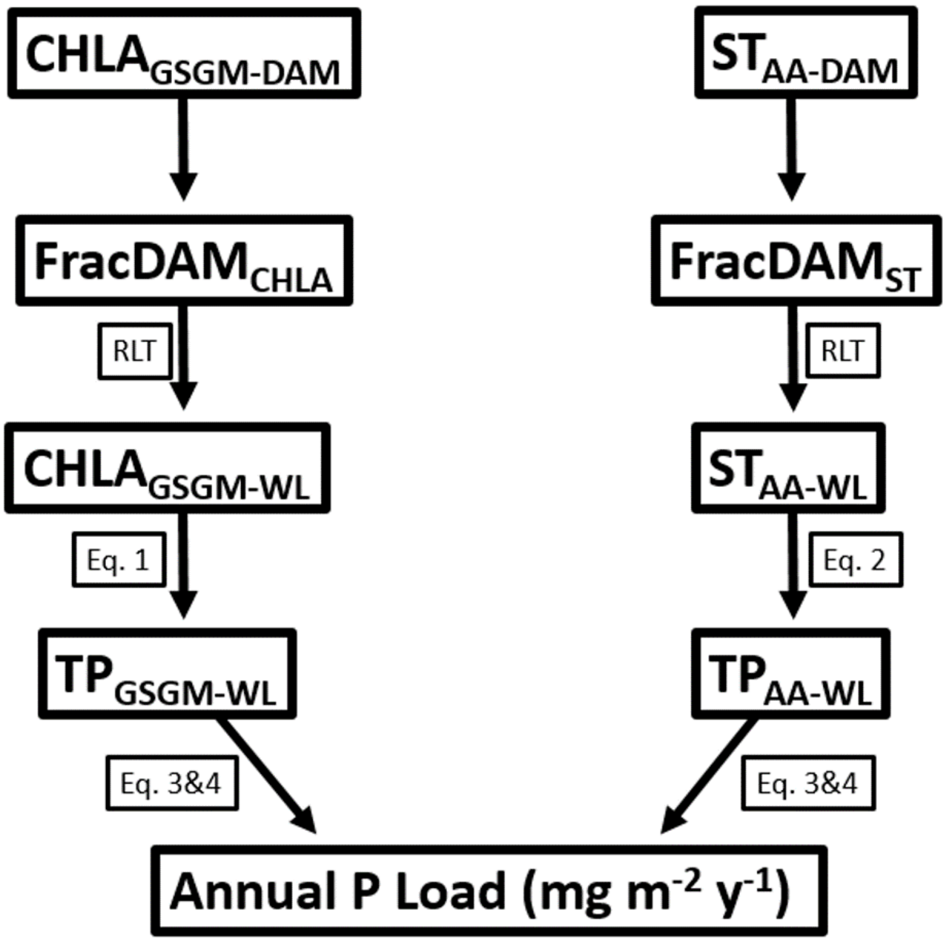
Riverine Zone	Transitional Zone	Lacustrine Zone
• Narrow basin	• Broader, deeper basin	• Lake-like basin
• High flow	• Reduced flow	• Little flow
• High susp. solids, low light	• Lower susp. solids, more light	• Lowest susp. Solids, most light
• High nutrients, advective supply	• Advective nutrient supply reduced	• Internal nutrient recycling, low nutrients
• Light limited photosynthesis	• High photosynthesis	• Nutrient limited photosynthesis
• Algal cell loss by sedimentation	• Algal cell loss by sedimentation, grazing	• Algal cell loss by grazing
• Allochthonous OM supply	• Intermediate	• Autochthanous OM supply
• More “eutrophic”	• Intermediate	• More “oligotrophic”

Figure 2



Site #	River km	Latitude	Longitude
12	78.2	36° 25' 28"	93° 51' 15"
11	68.9	36° 23' 17"	93° 53' 23"
10	59.9	36° 19' 51"	93° 54' 05"
9	46.5	36° 20' 02"	93° 57' 27"
8	37.7	36° 20' 06"	94° 01' 09"
7	32.7	36° 18' 28"	94° 01' 29"
6	26.4	36° 17' 36"	94° 02' 15"
5	21.1	36° 15' 37"	94° 04' 07"
4	15.0	36° 14' 29"	94° 01' 43"
3	11.6	36° 14' 23"	94° 00' 01"
2	5.0	36° 12' 31"	94° 00' 59"
1	0.0	36° 10' 34"	94° 01' 11"

Figure 3



$CHLA_{GSGM-DAM}$
ST_{AA-DAM}
$FracDAM_{CHLA}$
$FracDAM_{ST}$
RLT
RLT
$CHLA_{GSGM-WL}$
ST_{AA-WL}
Eq. 1
Eq. 2
$TP_{GSGM-WL}$
TP_{AA-WL}
Eq. 3&4
Eq. 3&4
Annual P Load (mg m^{-2} y^{-1})

Figure 4

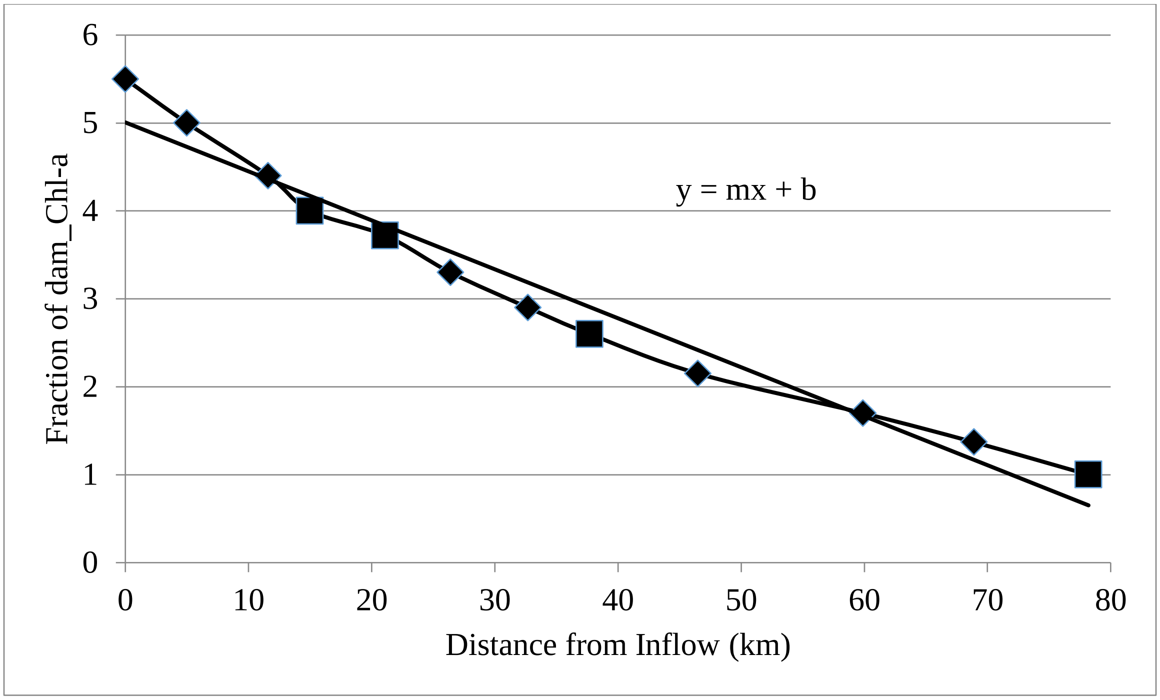

y = mx + b
Fraction of dam_Chl-a
Distance from Inflow (km)
0
1
2
3
4
5
6
0
10
20
30
40
50
60
70
80

Figure 5

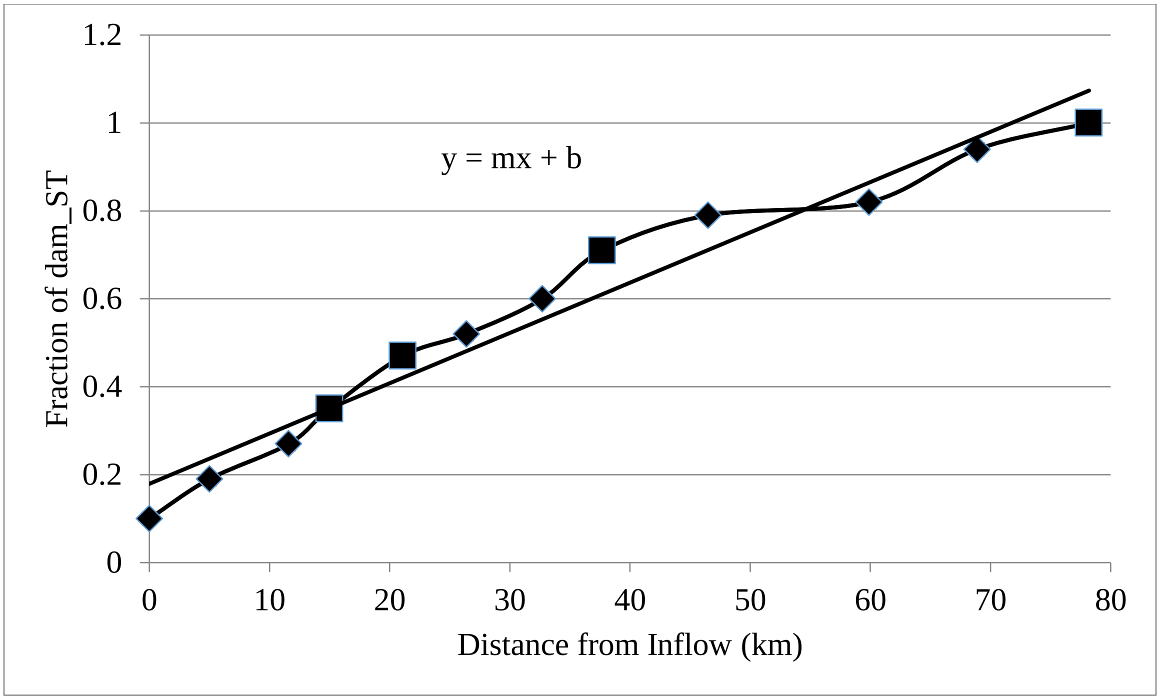

1.2
1
0.8
0.6
0.4
0.2
0
y = mx + b
Fraction of dam_ST
0
10
20
30
40
50
60
70
80
Distance from Inflow (km)

Figure 6

2. RECONSTRUCTING SPATIOTEMPORAL PHYTOPLANKTON BIOMASS TRENDS USING RESERVOIR LIMNOLOGY THEORY

2.1 Introduction

Limnology expanded its scope to man-made lakes (i.e., reservoirs) during the 1970s and 1980s, which resulted in the characterization of reservoirs as unique ecosystems with predictable ecological patterns (Walker 1981; Soballe and Kimmel 1987; Kimmel and Groeger 1984). Before 1990, most research on reservoir limnology was contained in a three-volume text (Hrbáček 1966; Hrbáček and Straškraba 1973a, b) reporting the long-term study of Czech hydroelectric/drinking water reservoirs (Kalff 2002). A more comprehensive text on reservoir limnology was published in the United States (Thornton et al. 1990).

Reservoirs are often characterized by three distinct zones along the upstream to downstream gradient (Kimmel and Groeger 1984): the riverine, transitional, and lacustrine zones. The riverine zone is typically a near-lotic environment characterized by a narrow, channelized basin with relatively short water residence time, greater suspended solids, greater nutrient concentrations, reduced light availability, and reduced Secchi transparency (ST) compared to the other zones. The transitional zone is typically characterized by slower water velocity as the basin becomes wider and deeper and water residence time increases. Thus, suspended solid concentrations are reduced and light penetration is greater in the transition zone compared to the riverine zone, which allows for greater primary production in the transitional zone. The lacustrine zone is characterized by morphometric characteristics most similar to natural lakes. Consequently, the water velocity is dramatically reduced, and nutrients have been removed from the upper water column through phytoplankton uptake and sedimentation. Thus, the lacustrine zone typically has the greatest light penetration and ST in reservoirs.

Reservoir zonation has been shown to be a reliable predictor of spatial patterns of phytoplankton biomass and productivity in reservoirs (Forbes et al. 2012). In a tributary embayment of the Three Gorges Reservoir in China, Mao (2015) reported a strong spatial dependence of algal bloom events, where > 73% of algal blooms were observed within a 10-km reach within the transitional and lacustrine zones. Wang (2011) used cluster analysis based on Chl-a, ST, and water column stability to identify longitudinal patterns along the main stem of the Yangtze River in China during different water level fluctuation periods. These patterns in phytoplankton biomass and productivity appear to be driven by physical factors that control biogeochemical phenomenon, such as light availability, nutrient uptake, and nitrogen fixation (Scott et al. 2008, 2009; Forbes et al. 2008).

Due to the high degree of spatial and temporal heterogeneity (Perkins et. al 2000), traditional trophic-state classification (Carlson 1977) has been discouraged in managing reservoirs (Lind et Al. 1993). Reservoir primary production and phytoplankton biomass are dependent upon several interconnecting chemical, biological, and physical factors, such as light availability, temperature, dam operation (i.e., water retention time), and macro- and micronutrient availability. For example, primary production can be driven by water residence time in some reservoirs (Kimmel and Groeger 1984; Soares et Al. 2008), while primary production in other reservoirs may be more strongly controlled by fluctuating inflows that modify zonation patterns (Soballe and Bachmann, 1984). Thus, hydrologic-controlled patterns in reservoir zonation may be a useful tool for predicting water quality patterns in reservoirs. However, simply adopting and/or implementing nutrient criteria from natural lakes may not be appropriate for reservoirs.

Kennedy (2001) outlined some major considerations for developing nutrient criteria for reservoirs, including considerations about within-reservoir differences due to spatial heterogeneity. Because reservoir zonation creates predictable spatial variation, location in a reservoir may be a useful way to model whole-reservoir conditions with monitoring data from a limited number of sites.

The U.S. Environmental Protection Agency (EPA), under the direction of The Clean Water Act (40 CFR 131), has tasked states with establishing numeric water quality (WQ) standards for nutrients that protect the designated use for different waterbodies (EPA 2010a). The EPA suggests that a weight-of-evidence approach be used to establish WQ criteria and recommended that data be used from either: (1) reference conditions, (2) mechanistic modeling applications, and/or (3) stressor-response relationships (EPA 2010a, b). The empirical stressor-response method can be used when estimating relationships between nitrogen (N) or phosphorus (P) to a response measure, such as Chl-a concentrations or ST measurements (EPA 2010a). The State of Arkansas recently adopted its first effects-based WQ criteria for Beaver Lake, which states that the growing season (May – October) geometric mean Chl-a (GMChl-a) concentration shall not exceed 8µg/L and the average annual ST (AAST) shall not be less than 1.1 m at Hickory Creek (HC). These criteria were based on a set of recommendations by a working group that analyzed reservoir data from within the region as well as from Beaver Lake (FTN 2008) and were followed by a set of recommended evaluation conditions (Scott and Haggard 2015). The HC location was chosen because it is upstream of all municipal water facility intake structures; however, the HC sampling location is in the riverine-transition area of Beaver Lake, which may not represent the broader water quality conditions in Beaver Lake because of HC's location.

The objective of this study was to characterize the spatial and temporal patterns of Chl-a and ST in Beaver Lake and derive a prediction tool to reconstruct whole-lake estimates of these variables based on measurements at the dam. It was hypothesized that the variation in Chl-a and ST across hydrologic zones would be strongly predictable (i.e., Chl-a decreases and ST increases from upstream to downstream locations) and greater in magnitude than interannual variation in these variables. Thus, a prediction model generated from new data could be combined with long-term data from a single location to hindcast the Chl-a and ST at any location in the reservoir in years where spatially explicit data were not available. To test this hypothesis, regression models were developed for Chl-a and ST with spatial location using data collected during the 2015 growing season and applied these statistical models to long-term data sets for the purposes of hindcasting. The residuals of these spatial models were evaluated against a variety of hydrologic variables to test the effect of variable hydrology in controlling interannual variation in spatial patterns of Chl-a and ST along the riverine-lacustrine continuum.

2.2 Materials and Methods

2.2.1 *Site Description*

Beaver Lake is located in the Northwest Arkansas counties of Washington, Benton, and Carroll, and is the most upstream in a series of three reservoirs along the White River in Arkansas and Missouri (BWD 2010). Beaver Lake watershed includes seven major subwatersheds that encompass more than 3095 km^2 with land use dominated by forests (71%) and agriculture (22%) (Beaver Water District, 2010). Beaver Lake was among the earliest man-made impoundments where reservoir limnology hypotheses were tested (Walker 1981). However, there has been limited data collected in a spatially explicit way so as to capture reservoir zonation along the entire length of the reservoir since the early studies. There has been little effort put toward a systematic evaluation of long-term variability of spatially explicit patterns in nutrients and the trophic state of the lake (Haggard 1999; Galloway and Green 2006; Koller 2007), particularly as it relates to State of Arkansas assessments of water quality in Beaver Lake (APCEC 2012).

2.2.2 *Sampling Locations, Dates, and Methods*

Water quality samples were collected from May 12 through October 14, 2015 at 12 sites (Figure 1) along an ~78-km transect that began near the White River at the Highway 412 bridge and ended at the reservoir dam. Seven sites were spaced approximately equidistant apart along the first half of the linear distance, and specifically included sampling sites at HC and Lowell at the Beaver Water District (BWD) intake structure. Another five sites were located approximately equidistant apart along the second half of the linear distance to the dam. The United States Geological Survey (USGS) has sampled four of these 12 sites (i.e., sites 4, 5, 8 and 12) on an approximately monthly to bi-monthly basis since 2001. For the present study, all sites were sampled semimonthly, around the 1st and 15th of the month, for the defined growing season in Arkansas (APCEC 2012), for a total of 11 times throughout the season. Sites were located in the thalweg. Secchi transparency was measured with a 20-cm-diameter Secchi disk and water column euphotic depth (Z_{EU} = depth at 1% surface irradiance) was measured using a Li-COR spherical quantum sensor (model LI-193SA, LI-COR Biosciences Inc., Lincoln, NE) on a vertical lowering frame. The photic zone was divided into two to seven equally spaced depths that varied by site and date, according to the variation in Z_{EU}, and each depth was sampled using a horizontal water sampler (model E-411-19XX—G62, Wildlife Supply Co., Yulee, FL). Samples were transferred into acid-washed and rinsed, 1-L ultraviolet-resistant, amber high-density polyethylene bottles, stored on ice, and returned to the laboratory at the University of Arkansas, Fayetteville, AR.

2.2.3 *Laboratory Processing Methods*

Within 24 hours of collection, volumetrically measured portions (100 to 750 mL) of a well-mixed sample were filtered under vacuum pressure using a 0.7-μm-pore-size, 25-mm-diameter Whatmann® GF/F filter (GE Healthcare, Buckinghamshire, UK) with enough site water so that color was evident on the filter. Filters were folded in half so that the plankton-containing sides were touching, wrapped in aluminum foil, and freezer-stored for Chl-a analysis. Chl-a was analyzed using a Turner Trilogy fluorometer (model #7200-000, Sunnyvale, CA) following an overnight acetone extraction using Method #10200H (APHA 2005; #10200 H).

2.2.4 *Data Manipulations and Statistical Analysis*

Although Chl-a samples were collected at multiple depths, for this study, these data were averaged into a single photic-zone Chl-a value for all site/date combinations. A geometric mean Chl-a value (GMChl-a) and arithmetic average ST (AAST) were then calculated for each site. These measures of central tendency were chosen because they correspond to those chosen by the State of Arkansas for water quality assessment purposes. The spatial patterns in GMChl-a and AAST were evaluated with linear regression models assuming a monotonic change with distance from reservoir inflow. Preliminary data indicated that Chl-a values at the two upstream-most sites were generally less than Chl-a at the third upstream-most site. Thus, these sites were excluded from the linear regression models to meet the assumption of monotonic change across reservoir zonation. Linear regression analysis was conducted on the GMChl-a and AAST data using SigmaPlot (version 12.5, San Jose, CA) to test the null hypothesis that no change in either parameter occurred along the riverine to lacustrine gradient.

The GMChl-a for all sites was divided by the GMChl-a at the dam to derive a ratio for each site ($FracDAM_{Chl\text{-}a}$). This ratio represented a multiplier for each location in the reservoir such that the $FracDAM_{Chl\text{-}a}$ at the dam was always equal to 1. The same approach was used to calculate the fraction of ST at the dam ($FracDAM_{ST}$). The GMChl-a and AAST calculated from USGS monitoring data collected from the dam location between 2001 to 2015 were then multiplied by the FracDAMChl-a and FracDAMST, respectively, to hindcast the GMChl-a and AAST values at the other USGS sites in the same years. Measured USGS GMChl-a and AAST data were plotted against these calculated values and linear regression analysis was applied (SigmaPlot) to test if the slope of the line differed from 1 (i.e., perfect prediction). Significance for all statistical analyses was judged at $p < 0.05$. The residuals of the measured x calculated GMChl-a and AAST analyses were then evaluated against 2001 - 2015 United States Corps of Engineers hydrologic variables (i.e., mean elevation [m], mean volume [m^3], mean inflow [km^3/month], and mean water residence time [years]) in order to determine the correlation between hydrologic parameter variation and spatial patterns in Chl-a and ST.

2.3 Results

2.3.1 *Spatial and Temporal Trends in 2015 Phytoplankton Biomass*

Water quality parameters varied widely among the 11 temporal measurements during the 2015 growing season. Chlorophyll-a and ST ranged between 1.67 and 22.40 µg/L and 0.1 and 8.5 m, respectively (Tables 1 and 2), among the 12 sites along the longitudinal gradient in Beaver Lake. The average between the maximum and minimum Chl-a concentration at any one site across the entire photic zone was generally less than 5 µg/L. These data supported the decision to average Chl-a concentrations across depths for each site. Throughout the 2015 growing season, Chl-a concentration was generally greatest in the riverine/transition zones and lowest in the lacustrine zone (Table 1, Figure 2). Similarly, ST was generally least in the riverine/transition zone and greatest in the lacustrine zone (Table 2, Figure 2). Between May 12 and July 1, 2015, Chl-a concentration was lower in the riverine section just upstream of the War Eagle Creek confluence and greater in the transitional zone (Table 1). Secchi transparency increased with distance from the inflow at all times sampled throughout the growing season (Table 2). When the two upper-most sites were excluded from the analysis, the GMChl-a concentration exhibited a monotonic decrease, and AAST exhibited a monotonic increase, as the distance from the inflow increased (Figure 2). Linear regression analysis indicated a statistically significant ($p = 0.0003$) negative relationship between distance from inflow and GMChl-a concentration with a slope of -0.0937 mg/L/km. The R^2 value indicated that 83% of the variation in GMChl-a along the riverine-lacustrine continuum could be explained by distance from the inflow. Thus, according to the model developed from the 2015 monitoring data, Beaver Lake exhibited a 0.09 mg L^{-1} decrease in Chl-a concentration for every kilometer of linear distance from the riverine to the lacustrine zone. Linear regression analyses also indicated a statistically significant ($p < 0.0001$) positive relationship between distance from the inflow and AAST, with a slope of 0.661 m/km, where the R^2 value indicated that 98% of the variation in AAST along the riverine-lacustrine continuum could be explained by distance from the inflow. Thus, according to the model developed from the 2015 monitoring data, Beaver Lake exhibited a 0.066 m increase in ST for every kilometer of linear distance from the riverine to the lacustrine zone.

2.3.2 *Hindcasting Models*

As a result of the monotonic patterns observed in the 2015 GMChl-a and AAST monitoring values, FracDAMChl-a consistently decreased and FracDAMST consistently increased as distance from the inflow increased (Table 3). Geometric mean chlorophyll-a at the upstream-most location was more than four times greater than GMChl-a at the dam, and AAST at the upstream-most location was only 20% of AAST at the dam. The application of the FracDAM estimates for both Chl-a and ST resulted in ecologically relevant estimates of GMChl-a and AAST computed from USGS measurements at the dam from 2001 to 2015. Linear regression analyses of measured vs. calculated data indicated that the hindcasting model tended to underestimate GMChl-a (Figure 3), with the average prediction being approximately one-half of the average measured value. However, 31% of the variation in measured values was explained by the model. Similar to GMChl-a, linear regression analyses of measured vs. calculated data indicated that the hindcasting model also tended to underestimate AAST (Figure 4), with the average prediction being approximately 70% of the average measured value. However, 40% of the variation in measure values was explained by the AAST hindcasting model.

Hydrologic measurements across years were useful predictors of the measured-model residuals from the GMChl-a hindcasting model, but not the AAST hindcasting model (Table 4). Mean annual inflow ($p = 0.0003$), mean annual elevation ($p = 0.0215$), and mean annual volume ($p = 0.0163$) were all positively correlated with the measured-model residuals, indicating that the difference between the measured and calculated values were greater at greater values of inflow, elevation, or volume. For example (Figure 5), when mean inflow was low (~ < 0.12 km^3/month), the predicted-measured Chl-a residuals tended to be negative and thus showed that the model tended to underestimate the calculated Chl-a values. As inflow increased (~ > 0.12 km^3/month), the predicted-measured Chl-a residuals shifted to more positive values, which indicated that the model tended to overestimate the calculated Chl-a values.

2.4 Discussion

The overarching goal of this study was to characterize the spatial patterns of Chl-a and ST in 2015 to derive different models that predict the spatial patterns of Chl-a and ST in other years by using whole-lake estimates of Chl-a and ST based on a measurement at a single location (e.g., the dam). Results from 2015 revealed that predictable patterns allowed for the combination of new and historic data from the dam to accurately estimate Chl-a and ST at any location in the lake where spatially explicit data were not previously available. Because three overlapping sites existed between USGS sampling efforts between 2001 and 2015 and the 2015 sampling, a unique opportunity existed to test the utility of hindcasting models for reconstructing spatial patterns in phytoplankton biomass in reservoirs. Results indicated that spatial patterns predicted using reservoir limnology theory were useful in reconstructing recent patterns in phytoplankton biomass, but were limited in accuracy due to interannual variation in hydrology. The most important finding of this study revealed that the Chl-a and ST predictive models tended to underestimate USGS-measured GMChl-a and AAST.

Results from the 2015 data set suggest comparable within-reservoir Chl-a and ST patterns to those hypothesized and measured in previous studies on Beaver Lake (Walker 1981; Thornton et al 1990). Findings from previous studies also postulated that phytoplankton biomass would be greatest in the transitional zone and least in the lacustrine zone (Sthapit et al. 2008) leading to inverse relationships with Secchi depth (Havel et al. 2009). As hypothesized, 2015 GMChl-a concentrations exhibited maximum values in the transition zone, with reduced Chl-a concentrations in the riverine zone, and the lowest Chl-a concentrations in the lacustrine zone. The patterns in Chl-a and ST suggest that phytoplankton may have been light-limited due to increased inorganic turbidity in the two upper-most sites (riverine zone) of the study (Koller 2007; Bryant 1992).

2.4.1 *Limitations and Strengths of the Models*

Although the Chl-a and ST models tended to underestimate actual measured values, interannual variations in inflow volume into the reservoir can, at least partially, account for the model results. In 2015, inflow into Beaver Lake was 2.01^6 km^3/year, the greatest among the 15-year USGS data set, whereas the average inflow over the 15-year USGS data set was 1.02^6 km^3/year. This high inflow volume results in nearly a doubling of inflow volume in 2015 compared to the average inflow associated with the 15-year data set.

The predictive models have the potential to be useful but were also biased. Two sources of model bias were identified:

1. Interannual variation in hydrology. The magnitude of the interannual variability in Chl-a and ST measurements appear to be controlled by hydrologic variation into the reservoir, as evidenced by Figure 5.
2. Variation in sampling methodology between historical USGS data and data garnered in 2105. The USGS has historically sampled at the 1.83m (6') depth, whereas, in 2015, samples were collected throughout the photic zone and results were reported as photic zone Chl-a concentration. By sampling only at the 1.83m depth, the USGS targeted a depth that biases the analysis by sampling at a location that is the optimal depth for phytoplankton growth. The methodology used in 2015 by sampling throughout the photic zone and presenting data that were a composite of the photic zone is likely a more accurate and depth-integrated representation of the true conditions in the lake.

Although the interannual variability in Chl-a and ST appear to be controlled by hydrologic variation, this limitation could become a strength by repeating this experiment in multiple years. By having a larger data set over multiple years with which to develop the model, finer acuity can be achieved. A multi-year analysis is needed to develop a spatially explicit model that can more accurately capture interannual variability in phytoplankton biomass patterns along the riverine-lacustrine continuum. Beginning in 2018, sampling along the same longitudinal gradient annually is planned in an effort to improve the relationship between calculated and measured Chl-a and ST.

In this study, strong spatial patterns of Chl-a and ST have been demonstrated along the riverine-to-lacustrine gradient in a large southeastern United States reservoir during the 2015 growing season. Linear regression analysis has also been demonstrated to be suitable for explaining spatial patterns of Chl-a and ST because the models are simple, their slopes are significant, and the equations describe the data well. This project represents the initial steps in a process to develop a simplistic model that can be implemented to better inform reservoir managers of spatial patterns in phytoplankton biomass and productivity. However, more effort needs to be applied in order to understand the interannual variabilities that occur in man-made reservoir systems.

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2.6 Tables

Table 1. Beaver Lake chlorophyll-a concentration (μg/L) across all sites and sampling dates expressed as mean (μ) chlorophyll-a ± standard deviation (SD) and number of samples (n).

Date	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Site 9	Site 10	Site 11	Site 12	μ±SD
5/12/15		5.1±0.0 (1)	13.9±0.3 (2)	11.6±0.5 (3)	5.1±1.3 (3)	5.1±2.1 (4)	3.4±1.5 (6)	3.4±0.7 (3)	3.2±1.4 (6)	6.0±2.0 (6)	4.5±1.0 (6)	2.5±0.9 (6)	5.1±3.6 (11)
6/2/15	6.0±4.3 (2)	5.6±6.1 (2)	22.4±2.3 (2)	16.9±3.9 (2)	15.5±11.0 (3)	16.7±9.4 (3)	14.6±9.6 (4)	6.2±1.9 (4)	5.1±2.2 (5)	2.9±1.3 (6)	3.2±1.1 (6)	3.1±1.1 (6)	7.7±6.8 (12)
6/16/15	6.1±0.2 (3)	7.8±1.4 (3)	11.7±5.7 (3)	11.4±9.6 (3)	16.1±4.9 (3)	16.0±5.4 (4)	12.1±7.8 (4)	3.2±3.8 (5)	5.3±1.8 (5)	3.3±1.5 (6)	2.7±1.1 (6)	2.7±1.5 (6)	6.7±5.1 (12)
7/1/15	9.2±5.6 (3)	6.2±3.1 (3)	14.5±4.9 (4)	20.4±9.5 (4)	15.6±6.6 (4)	12.4±5.0 (4)	11.5±3.1 (5)	8.2±1.6 (4)	6.2±4.2 (6)	5.4±3.5 (6)	2.6±0.5 (6)	1.7±0.9 (6)	7.7±5.6 (12)
7/15/15	17.5±11.9 (2)	15.2±5.7 (2)	10.0±5.9 (2)	9.5±4.6 (4)	7.0±1.9 (4)	9.1±2.2 (4)	7.7±2.0 (5)	6.0±2.6 (5)	4.6±0.8 (6)	3.9±2.2 (6)	2.9±1.5 (6)	2.2±0.8 (6)	6.7±4.7 (12)
8/3/15	10.8±0.1 (2)	8.5±5.5 (4)	10.2±2.9 (4)	7.7±1.9 (4)	9.9±2.1 (5)	6.5±3.3 (5)	6.6±5.2 (5)	7.2±4.8 (5)	4.6±4.0 (6)	3.3±2.5 (6)	3.9±2.8 (6)	2.1±1.7 (6)	6.0±2.7 (12)
8/17/15	11.2±2.7 (4)	9.8±2.0 (4)	8.8±1.2 (4)	7.3±2.2 (4)	6.0±2.0 (5)	5.9±1.7 (5)	4.9±1.2 (6)	5.4±1.5 (5)	6.8±6.1 (6)	10.0±9.5 (6)	7.7±6.2 (6)	3.4±1.2 (6)	6.9±2.3 (12)
9/2/15	10.7±1.9 (4)	8.5±2.6 (4)	7.6±2.0 (4)	6.4±1.7 (5)	5.7±1.9 (5)	4.4±2.3 (6)	3.9±1.5 (6)	4.6±0.5 (4)	3.9±1.2 (6)	6.0±4.3 (6)	5.9±3.6 (6)	3.3±1.1 (6)	5.6±2.2 (12)
9/16/15	8.8±0.9 (3)	9.2±0.4 (3)	7.5±0.6 (4)	6.0±0.4 (4)	4.6±0.5 (5)	5.4±0.5 (5)	3.5±0.5 (5)	2.5±1.2 (6)	2.4±1.1 (6)	3.1±0.3 (6)	3.6±0.7 (6)	2.7±1.0 (6)	4.4±2.5 (12)
9/30/15	9.1±0.5 (4)	5.5±3.1 (4)	6.0±0.4 (5)	4.0±0.1 (5)	3.7±0.3 (5)	2.8±1.0 (6)	1.9±0.6 (6)	2.4±0.1 (5)	2.1±0.8 (6)	2.8±1.0 (6)	3.5±0.7 (6)	3.6±0.6 (6)	3.6±2.1 (12)
10/14/15	11.3±0.7 (3)	6.8±0.5 (3)	6.4±0.6 (3)	5.7±1.1 (4)	4.3±0.8 (4)	3.3±0.3 (5)	3.4±0.4 (5)	3.0±0.5 (5)	3.0±0.4 (6)	3.1±0.6 (6)	3.7±0.2 (6)	4.4±0.3 (6)	4.5±2.4 (12)
μ±SD	9.6±3.2 (10)	7.6±2.9 (11)	10.0±4.8 (11)	8.7±5.1 (11)	7.2±4.9 (11)	6.7±4.9 (11)	5.6±4.3 (11)	4.4±2.0 (11)	4.0±1.5 (11)	4.2±2.2 (11)	3.8±1.5 (11)	2.8±0.8 (11)	

Table 2. Beaver Lake Secchi transparency (m) across all sites and sampling dates expressed as mean (μ) Secchi ± standard deviation (SD) and number of samples (n).

Date	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Site 9	Site 10	Site 11	Site 12	μ±SD
5/12/15		0.1	0.5	0.3	0.4	2.1	3.1	3.4	4.1	4.7	5.6	8.3	3.0 ± 2.6 (11)
6/2/15	0.5	0.6	0.7	0.8	1.2	1.5	1.8	2.5	3.5	5.0	5.7	7.6	2.6 ± 2.4 (12)
6/16/15	1.7	1.8	1.6	1.7	1.5	1.6	2.1	2.9	4.6	5.8	6.1	7.3	3.3 ± 2.1 (12)
7/1/15	1.5	1.3	1.4	1.5	1.6	1.9	2.1	2.7	3.7	5.1	5.4	6.4	2.9 ± 1.8 (12)
7/15/15	0.5	0.6	1.6	1.7	2.7	2.2	2.5	2.9	3.1	4.4	4.8	6.2	2.8 ± 1.7 (12)
8/3/15	1.5	1.6	2.1	2.2	2.7	2.9	3.2	3.7	4.4	5.1	6.2	8.5	3.7 ± 2.1 (12)
8/17/15	1.6	2.0	2.2	2.4	2.7	2.9	3.3	2.8	3.8	3.9	4.3	6.4	3.2 ± 1.3 (12)
9/2/15	2.1	2.4	2.6	2.8	3.5	3.9	4.0	4.1	3.7	4.6	4.3	6.2	3.7 ± 1.1 (12)
9/16/15	1.3	1.7	2.3	2.0	2.8	2.6	3.7	3.5	3.7	4.3	4.4	5.9	3.2 ± 1.3 (12)
9/30/15	1.4	1.8	2.4	2.6	2.9	3.6	3.6	3.3	3.5	3.3	3.5	4.4	3.0 ± 0.9 (12)
10/14/15	1.2	1.1	1.5	1.7	1.7	3.0	3.8	3.5	4.3	4.2	4.0	4.1	2.8 ± 1.3 (12)
μ±SD	1.2±0.5 (10)	1.1±0.7 (11)	1.6±0.7 (11)	1.6±0.8 (11)	1.9±0.9 (11)	2.5±0.8 (11)	2.9±0.8 (11)	3.2±0.5 (11)	3.8±0.5 (11)	4.5±0.7 (11)	4.9±0.9 (11)	6.3±1.4 (11)	3.0 ± 1.2

Table 3. Fraction (Frac) of chlorophyll-a (Chl-a) and Secchi transparency (ST) at each site relative to the value at the dam (DAM).

Distance (km)	Site	$FracDAM_{Chl\text{-}a}$	$FracDAM_{ST}$
0.0	1	4.13	0.22
5.0	2	2.93	0.23
11.6	3	4.82	0.29
15.0	4	4.27	0.30
21.1	5	3.54	0.36
26.4	6	3.34	0.43
32.7	7	2.68	0.50
37.7	8	2.05	0.52
46.5	9	1.65	0.62
59.9	10	1.45	0.73
68.9	11	1.23	0.76
78.2	12	1.00	1.00

Table 4. Residuals from measured vs calculated chlorophyll-a (Chl-a) and Secchi transparency (ST)scatter plots regressed against four hydrologic parameters.

Predicted-measured parameter	Parameter	Slope	R^2	p-value
Chl-a residual	Elevation	0.19	0.377	0.0215
Chl-a residual	Volume	0.000007	0.393	0.0163
Chl-a residual	Inflow	0.00003	0.563	0.0003
Chl-a residual	WRT	-0.0013	0.183	0.2785
ST residual	Elevation	-0.027	0.196	0.2448
ST residual	Volume	-0.000001	0.205	0.2234
ST residual	Inflow	-0.000002	0.187	0.2681
ST residual	WRT	-0.0002	0.109	0.5190

2.7 Figure legends

Figure 1. Site number, distance from inflow (km), latitude, and longitude of sampling locations in Beaver Lake, AR. Sites 4, 5, 8, and 12 had historically been sampled by the United States Geological Survey between 2001 and 2015.

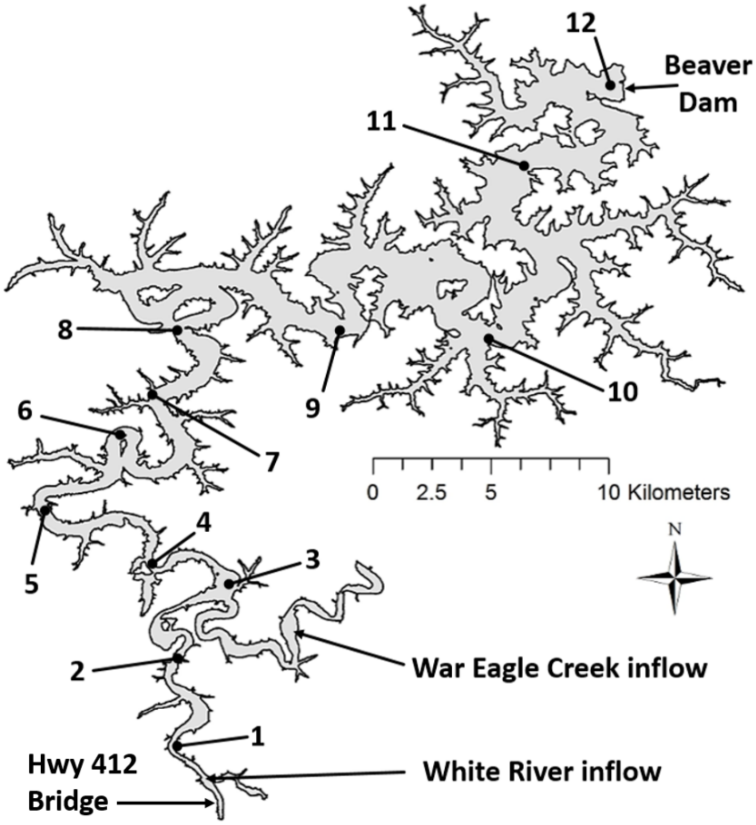
Figure 2. Linear regression models derived from averaged geometric mean chlorophyll-a (in gray) and arithmetic average Secchi transparency (in black) values across the 2015 growing season to evaluate the change in chlorophyll-a and Secchi transparency from inflow to dam. The vertical reference line at 15.5 km denotes the Hickory Creek location. The gray X and black X represent the chlorophyll-a and Secchi transparency water quality standards at Hickory Creek of 8 μg/L and 1.1 m, respectively.

Figure 3. United States Geological Survey measured values of chlorophyll-a concentration (μg/L) were plotted against their calculated values (that were determined by multiplying the ratio of chlorophyll-a measured at each site to the chlorophyll-a measured at the dam) by the geometric mean chlorophyll-a data collected by USGS at the dam of Beaver Lake from 2001 to 2015.

Figure 4. United States Geological Survey measured values of Secchi transparency (m) were plotted against their calculated values (that were determined by multiplying the ratio of Secchi transparency measured at each site to the Secchi transparency measured at the dam) by the arithmetic average Secchi transparency data collected by USGS at the dam of Beaver Lake from 2001 to 2015.

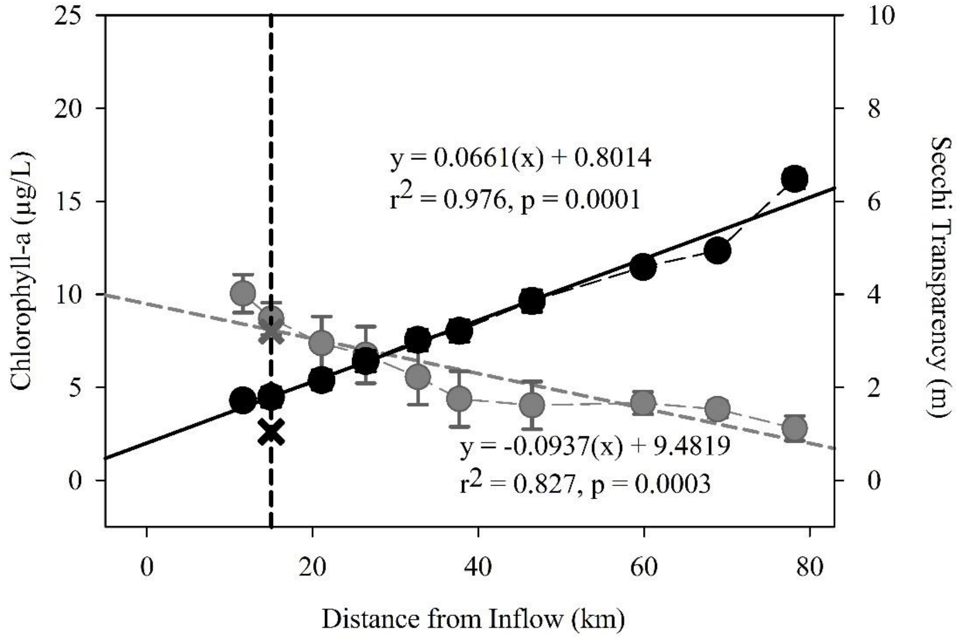
Figure 5. Mean inflow (km^3/month) plotted against the measured-calculated chlorophyll-a concentration residuals in an effort to reveal possible hydrologic drivers of interannual variability in Chl-a response.

2.8 Figures



Site #	River km	Latitude	Longitude
12	78.2	36° 25' 28"	93° 51' 15"
11	68.9	36° 23' 17"	93° 53' 23"
10	59.9	36° 19' 51"	93° 54' 05"
9	46.5	36° 20' 02"	93° 57' 27"
8	37.7	36° 20' 06"	94° 01' 09"
7	32.7	36° 18' 28"	94° 01' 29"
6	26.4	36° 17' 36"	94° 02' 15"
5	21.1	36° 15' 37"	94° 04' 07"
4	15.0	36° 14' 29"	94° 01' 43"
3	11.6	36° 14' 23"	94° 00' 01"
2	5.0	36° 12' 31"	94° 00' 59"
1	0.0	36° 10' 34"	94° 01' 11"

Figure 1



25
20
15
10
5
0
10
8
6
4
2
0
0
20
40
60
80
Chlorophyll-a (µg/L)
Secchi Transparency (m)
Distance from Inflow (km)
y = 0.0661(x) + 0.8014
r2 = 0.976, p = 0.0001
y = -0.0937(x) + 9.4819
r2 = 0.827, p = 0.0003

Figure 2

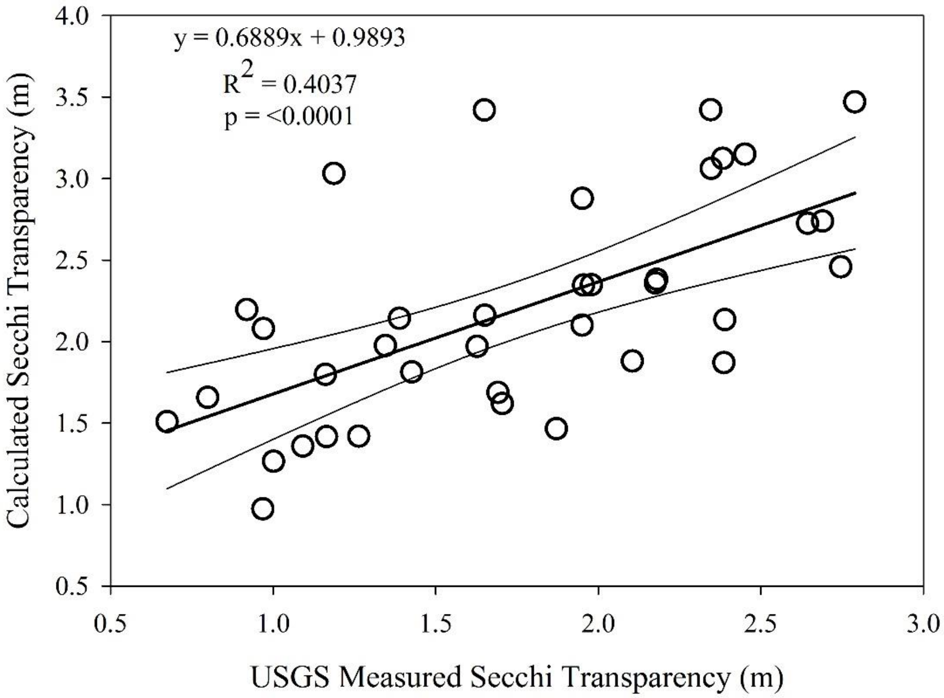
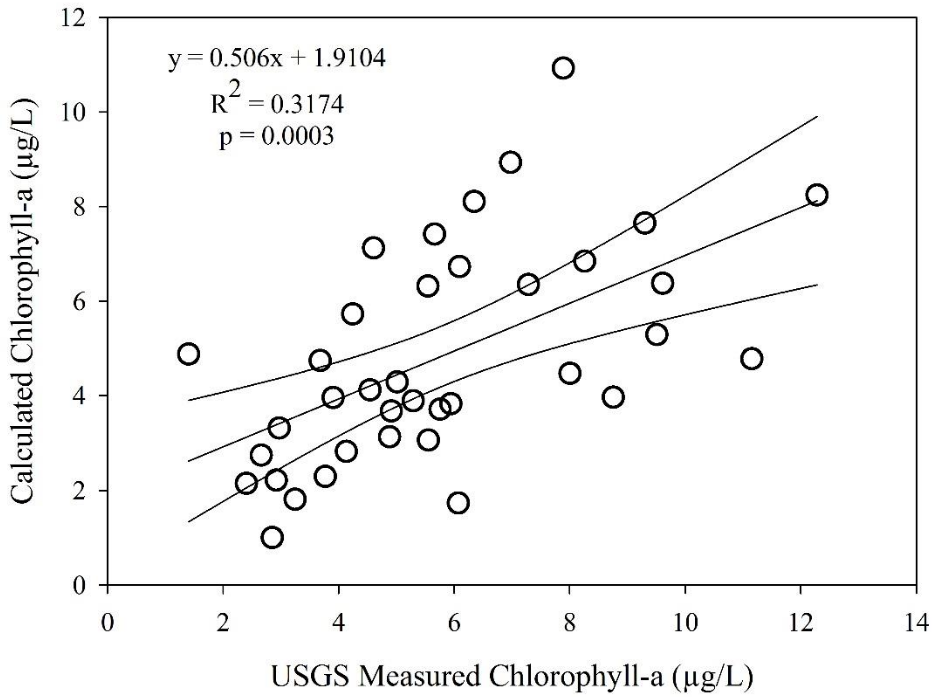

y = 0.6889x + 0.9893
R2 = 0.4037
p = <0.0001
Calculated Secchi Transparency (m)
USGS Measured Secchi Transparency (m)
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.5
1.0
1.5
2.0
2.5
3.0

Figure 3



y = 0.506x + 1.9104
R2 = 0.3174
p = 0.0003
Calculated Chlorophyll-a (µg/L)
USGS Measured Chlorophyll-a (µg/L)
0
2
4
6
8
10
12
14

Figure 4

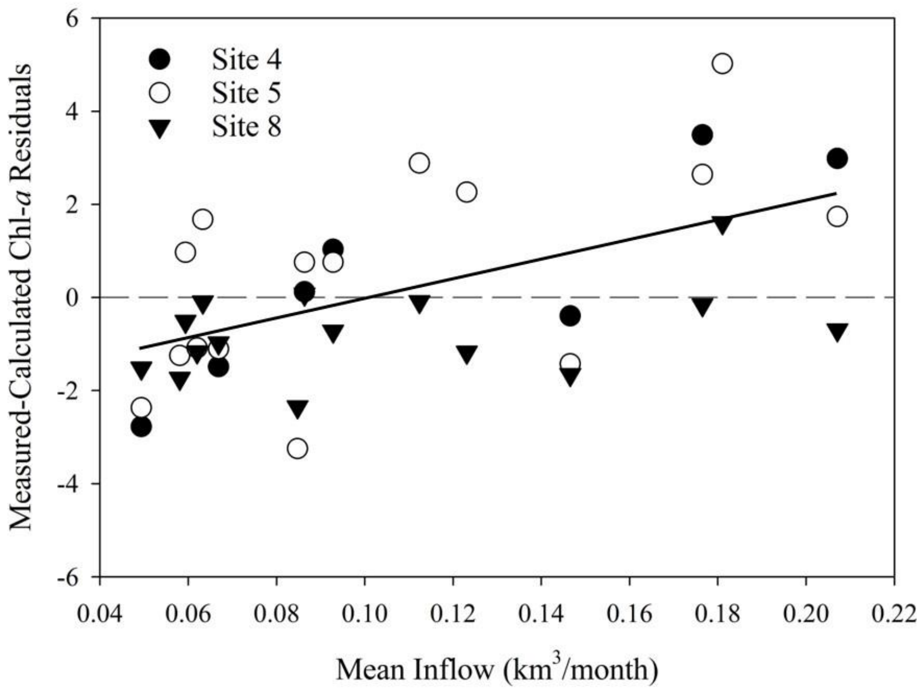

Site 4
Site 5
Site 8
Measured-Calculated Chl-*a* Residuals
6
4
2
0
-2
-4
-6
0.04
0.06
0.08
0.10
0.12
0.14
0.16
0.18
0.20
0.22
Mean Inflow (km^3/month)

Figure 5

3. DERIVING PHOSPHOROUS LOAD REDUCTION ESTIMATIONS FOR EUTROPHICATION MANAGEMENT BY COMBINING RESERVOIR LIMNOLOGY THEORY WITH STEADY-STATE RESERVOIR MODELING

3.1 Introduction

Assessing the natural distribution of nutrient concentrations in reservoirs is imperative to understanding the contributions of anthropogenic nutrient inputs to downstream loading. Excessive nutrient additions are the main driver of cultural eutrophication in surface waters of the United States (US) (USEPA, 2015). In comparison to other macronutrients required for biological metabolism, phosphorus (P) is usually the least abundant and also the first to limit productivity (Wetzel, 2001) in made-made, freshwater reservoirs in the Southeastern US. Thus, controlling P inputs, and in many cases in combination with nitrogen (N), is necessary for reducing the harmful effects of eutrophication such as oxygen depletion, harmful algal blooms, taste and odor compounds, and disinfection byproduct precursors (Paerl et al. 2016a, 2016b).

The nutrient-loading concept (Rawson, 1939; Edmundson, 1961) implies that a relationship exists between nutrient inputs into a water body and the response to those nutrient inputs. The resulting effect from nutrient inputs can be quantified by a measurable index of primary productivity, such as Chl-a or ST (Canfield and Bachmann, 1981), where an empirical equation can be formed that predicts the response. Empirical equations are simple mathematical models that provide generalized predictions. Empirical lake response models have been developed to quantify the relationships between nutrient concentrations, such as for total P (TP), and response variables such as s Chl-a (Dillon and Rigler, 1974; Jones and Bachmann, 1976) and ST (Carlson, 1977).

Many simple, steady-state, nutrient-response models have been defined (Vollenweider 1968, 1976; Nürnberg, 1984) and refined (Vollenweider and Kerekes, 1980; Nürnberg, 1998) over the years, leading to the development of more complex predictive relationship models like BATHTUB (Walker, 1981), CEQUAL-W2 (Cole and Wells, 2003), EFDC (Tetra Tech, Inc. 2007), and CAEDYM (Hipsey et al. 2013). In general, these nutrient-response models provide a method to predict lake P concentrations based on morphometric and hydrologic data and subsequently estimate lake productivity. One of the central assumptions of steady-state models is that storage gains (sources) and losses (sinks) of nutrients over time equal zero. Another assumption is that nutrient loadings into the water body are considered instantaneous and completely mixed throughout the lake, while ignoring shorter-term fluctuations. While smaller, naturally formed lakes are more apt to meet these assumptions, larger, dendritic, man-made reservoirs generally have greater spatial variation caused by the interaction of more complex hydrology and morphometry (Thornton et al. 1990). Kerekes (1982) advocated for the segmentation of long and narrow reservoirs by treating each segment as a series of connecting individual bodies of water. Although this approach has become widely used, it can be computationally intensive and can be too coarse when spatial variation is drastic.

Beaver Lake in Northwest Arkansas is experiencing excessive algal growth due to cultural eutrophication from P loading, as indicated by the violation of Secchi transparency (ST) and chlorophyll-a (Chl-a) water quality (WQ) standards (Scott and Haggard, 2015). The WQ standard for Chl-a dictates that the growing season geometric mean Chl-a concentration shall not exceed 8 μg/L, with a minimum of five evenly distributed, discrete samples required throughout the growing season in order to calculate a geometric mean. The WQ standard for ST dictates that the annual average Secchi depth shall not be less than 1.1 m, with a minimum of 10 discrete samples evenly distributed over 12 calendar months in order to calculate an annual average. These WQ standards (ADEQ, 2018) were assigned to the Hickory Creek (HC) location on

Beaver Lake because the site is physically located just upstream of the first of four municipal water intake structures on Beaver Lake, and because this location integrates the loadings from all three major tributaries – White River, Richland and War Eagle Creeks (FTN, 2008). However, the HC location is problematic for modeling trophic conditions in Beaver Lake because it is located in the middle of the riverine-transition zone of the lake.

While notable research has been conducted that models nutrient loading into Beaver Lake (Walker, 1981), few studies exist that have examined Chl-a, ST, and TP concentrations along the entire riverine-lacustrine continuum. In addition, loading models previously developed for Beaver Lake (Walker, 1981; Bolyard et al., 2010) require complex, multiple-input parameters, which can be expensive and cost-prohibitive for WQ managers to obtain and/or generate. Therefore, an opportunity exists to test whether a simple, minimal-input loading model can produce results that are similar in magnitude and variation to results from more complex models.

The main objective of this study was to apply the spatial productivity model derived in Chapter 2 with the Vollenweider and Kerekes (1980) steady-state model to estimate P-load reductions necessary to meet the Chl-a and ST water quality standards that apply to Beaver Lake at HC. To apply the Vollenweider steady-state model to estimate historic, current, and target P loads to Beaver Lake, we first estimated the annual whole-lake average Chl-a and ST for the lake. It was hypothesized that Vollenweider-modeled estimates would be similar in magnitude and variation to those measured in other studies; therefore, validating the use of a simple, minimal-input model for use in the reservoir. To test this hypothesis, modeled P-loading estimates were compared to those of Bolyard et al. (2010) and Scott et al. (2016). Modeled P-load estimates were also compared to loading estimates calculated from 2015 TP grab samples.

3.2 Materials and Methods

3.2.1 *Site Description*

Beaver Lake in Northwest Arkansas was created in 1966 when a concrete and earthen dam was constructed across the White River that created a reservoir for flood control, hydroelectric power, and a local residential water supply. Currently, nearly 420,000 Arkansans get their drinking water from Beaver Lake, which has a nearly 114-km^2 surface area when the reservoir is at the top of the flood-control pool (1130 ft MSL). Beaver Lake is a monomictic waterbody with three distinct zone classifications within the reservoir (i.e., riverine, transitional, and lacustrine) that exhibits a trophic gradient from eutrophic in the riverine zone to oligotrophic in the lacustrine zone (Haggard et al., 1999). Although Beaver exhibits all three reservoir zones, varying inflow volumes greatly control the hydrologic metrics that define zonation classification; therefore, defining the exact location where one zone ends and next begins is difficult because the exact location is dynamic and can vary from day to day.

Beginning in the 1990s, residents around and consumers of drinking water from Beaver Lake began experiencing annual taste and odor events caused by the accumulation of excess algae (i.e., eutrophication). In an effort to monitor eutrophication in Beaver Lake, the US Geological Survey (USGS) has been sampling multiple sites along the riverine-lacustrine continuum associated with Beaver Lake; however, the majority of USGS sites are spatially imbalanced along the continuum to favor more up-reservoir sites. These up-reservoir sites are primarily in the riverine and transitional zones, where, because of greater nutrient input from inflows, primary production is typically greatest. Regardless, researchers have previously utilized data collected by the USGS at up-reservoir sites to explain WQ conditions in down-reservoir locations.

3.2.2 *Sampling dates, locations, and methods*

Between May 12 and October 14, 2015, water quality samples were collected, approximately every two weeks, at 12 fixed sites along the riverine-to-lacustrine continuum (~78 km) on Beaver Lake. Detailed sampling procedures are discussed in the previous chapter including systematic depth sampling throughout the photic zone at each site. Briefly, 1000 mL individual samples (n = 607) were collected from multiple depths (0.1 – 16 m) for determination of photic zone Chl-a concentration, for the previous chapter, as well as TP concentration for the current chapter. Samples were transferred into acid-washed and rinsed, 1-L, ultraviolet-resistant, amber, high-density polyethylene bottles, stored on ice, and returned to the laboratory at the University of Arkansas, Fayetteville, AR.

3.2.3 *Laboratory processing methods*

Within 24 hours of collection, approximately 200 mL of a well-mixed subsample from every site/depth combination was transferred into 250-mL ultraviolet-resistant, amber high-density polyethylene bottles and preserved by freezing. Following persulfate digestion (APHA, 2005; #4500-PJ), water samples were analyzed for TP on a Skalar San++ Continuous Flow Analyzer (Skalar Inc., The Netherlands) (APHA, 2005; #4500-P F) at the Arkansas Water Resource Center's Certified Laboratory in Fayetteville, Arkansas.

3.2.4 *Whole-lake calculations and P-load estimates*

3.2.4.1 *Calculation of whole-lake Chl-a and ST*

The Vollenweider steady-state model requires that whole-lake averages be calculated and used as model inputs. As detailed in the previous chapter, in order to calculate whole-lake averages of Chl-a and ST, the geometric mean Chl-a (GMChl-a) and arithmetic average ST (AAST) first needed to be calculated at each site for 2001 through 2015. In brief, 2015 site averages of Chl-a and ST were divided by the average Chl-a and ST measured at the dam in order to calculate ratios for each site, which were termed FracDAMChl-a and FracDAMST, respectively. These individual site ratios were then multiplied by the respective USGS-measured averages at the dam between 2001 and 2015 in order to hindcast the GMChl-a and AAST values at all other sites in the same years. Whole-lake averages of Chl-a and ST for 2001 through 2015 were computed for both modeled and measured data. Whole-lake modeled data for 2001 through 2015 were calculated at 1-km intervals using regression analysis garnered from data obtained from our own sampling at 12 sites on Beaver Lake in 2015. Whole-lake measured data for 2001 through 2015 were calculated using regression analysis garnered from data collected by the USGS at four sites on Beaver Lake from 2001 through 2015. In order to identify the target whole-lake Chl-a and ST values that correspond to WQ standards assigned to Beaver Lake at HC, the recently adopted Chl-a and ST WQ targets of 8 μg/L and 1.1 m, respectively, were assigned to the HC location and regression models garnered from data obtained from our own sampling at 12 sites on Beaver Lake in 2015 were applied by rearranging the equation with HC WQ standards as the endpoints.

3.2.4.2 *Calculation of whole-lake TP*

Once whole-lake averages of Chl-a and ST for measured and modeled data for 2001 through 2015, as well as target data sets, were calculated, whole-lake averages of TP could be calculated using widely used limnological models that quantify the relationship between measurable lake responses (i.e., Chl-a and ST) to eutrophication. Carlson's (1977) equation (Equation 1) that quantifies the relationship between ST and TP was used to derive whole-lake values for TP, where $P_{c\text{-}ST}$ is the average total P concentration (mg/m^3) in the upper mixed layer of the lake and ST is the average annual Secchi transparency of the lake. Next, Dillon and

Rigler's (1974) equation (Equation 2) that quantifies the relationship between Chl-a and TP was used to derive whole-lake values for TP where $P_{c\text{-}Chl\text{-}a}$ is the average total P concentration (mg/m^3) in the upper mixed layer of the lake and *Chl-a* is the GMChl-a concentration (mg/m^3) in the upper mixed layer of the lake. Whole-lake averages of TP are critical to understanding and calculating P-load estimations using the Vollenweider and Kerekes (1980) equation.

3.2.4.3 *Calculation of Vollenweider P-loading*

The annual P load necessary to achieve the whole-lake TP concentrations required to fuel the measured and modeled whole-lake GMChl-a and AAST was calculated using the Vollenweider and Kerekes (1980) equation. This equation model was modified from that previously issued by Vollenweider (1968, 1976). In order to calculate external P loading using the Vollenweider and Kerekes (1980) equation (Equation 3), water residence time (τ_w, in years) and the annual water loading rate (q_s, in m/yr) had to be determined for Beaver Lake. Using publicly available data, τ_w and q_s were derived by combining data on average annual volume (m^3) and surface area (m^2) of Beaver Lake from the US Army Corps of Engineers (USACE) and the average annual inflow (m^3/yr) to Beaver Lake from the US Geological Survey (USGS). Beaver Lake q_s (m/yr) was calculated by dividing average annual inflow (m^3/yr) by the surface area (m^2), and Beaver Lake τ_w (years) was calculated by dividing average annual volume (m^3) by average annual inflow (m^3/yr). Vollenweider and Kerekes (1980) P-loading was calculated for each year from 2001 through 2015 using both whole-lake averages of Chl-a and ST in both measured and modeled data sets (n = 60), as well as target values of whole lake Chl-a and ST associated with WQ standards (n = 2) assigned at HC. As an added check of the accuracy of the Vollenweider P-load estimations calculated from whole-lake averages of Chl-a and ST, P-load estimations were also calculated from photic zone TP samples collected throughout the 2015 sampling season.

Phosphorous-load reduction estimations are the difference between the estimations of P load going into Beaver Lake and the estimations of P load that support the WQ standards at HC. The annual P load necessary to achieve the whole-lake TP concentrations required to fuel the measured and Modeled whole-lake GMChl-a and AAST estimates were computed using Vollenweider's and Kerekes (1980) equation, where L_{Pc} is the critical P-loading rate (mg/m/yr), P_c is the critical whole-lake P concentrations identified in Equations 1 or 2, τ_w is the water residence time (years), and q_s is the annual water loading rate (m/yr). An annual average reduction was calculated for each year from 2001 through 2015, for both Chl-a and ST whole-lake averages, for both modeled and measured data sets. Averages of 2001 through 2015 P loading were also calculated to reveal an ultimate 15-year average P load, mass and percentage needed to attain WQ standards.

3.3 Results

3.3.1 *Spatial and temporal trends in 2015 total phosphorous concentrations*

Sample TP concentrations (n = 607) from individual date/site/depth combinations varied greatly among the 11 temporal measurements during the 2015 growing season. Total P concentrations from the individual date/site/depth combinations ranged between 0.5 and 180.9 µg/L (Appendix A) among the 12 sites along the longitudinal gradient in Beaver Lake. When individual site photic zone TP concentrations were arithmetically averaged by date, TP concentrations ranged between 3.7 and 171.3 µg/L. When averaged photic zone TP concentrations from every date were averaged by site, TP concentrations ranged between 25.4 and 73.8 µg/L (Table 3.1). Throughout the 2015 growing season, TP concentrations were greatest at the most-upstream site and least at the dam. Linear regression analysis indicated a highly significant ($p < 0.001$) negative relationship (Figure 3.1) between distance from inflow and average annual TP concentration with a slope of -0.67 µg/L/km. The R^2 value indicated that 97.8% of the variation in growing season photic zone TP along the riverine-lacustrine continuum was explained by distance from the inflow (Figure 3.1).

3.3.2 *Whole-lake chlorophyll-a and Secchi transparency*

Modeled whole-lake Chl-a and ST values for 2001 through 2015 were computed using the spatially explicit estimates of Chl-a and ST at 1-km intervals using the respective FracDAM models and the values of Chl-a and ST measured at the dam by the USGS. Modeled whole-lake Chl-a and ST values from 2001 through 2015 ranged from 1.6 to 9.8 µg/L and from 1.7 to 3.3 m, respectively (Table 3.2). Measured whole-lake Chl-a and ST values for 2001 through 2015 were also computed by averaging the GMChl-a and AAST for each site within a given year. Measured whole-lake Chl-a and ST values from 2001 through 2015 ranged from 2.5 to 8.5 µg/L and from 1.8 to 3.1 m, respectively (Table 3.3). When the models were applied inversely, whole-lake average Chl-a and ST values were estimated that correspond to the 8.0 µg/L and 1.1 m target water quality values that were promulgated for the HC location. Based on the results of the spatially explicit model, the 8.0 µg/L Chl-a standard for Beaver Lake translated into a 5.2-µg/L standard for the whole lake and the 1.1 m ST standard translated into a 3.2-m standard for the whole lake (Table 3.2 and 3.3). Interestingly, when whole-lake values from USGS measured data set were regressed against whole-lake values from 2015 modeled data, results indicated justification in performing the 2015 sampling across the entire riverine-lacustrine continuum with sites that were relatively equally spaced. Whole-lake USGS measured ST values were less than whole-lake 2015 modeled ST values (Figure 3.2), while the majority of whole-lake USGS measured Chl-a values were greater than whole-lake 2015 modeled Chl-a values (Figure 3.3).

3.3.3 *Whole-lake total phosphorus*

Modeled, measured, and target whole-lake TP values for 2001 through 2015 were computed for whole-lake Chl-a and ST values using Dillon and Rigler (1974) and Carlson (1977) equations, respectively. Modeled whole-lake TP values based on whole-lake Chl-a and ST values ranged from 14.4 to 27.8 µg/L and from 8.3 to 29.4 µg/L, respectively (Table 3.2). Measured whole-lake TP values based on whole-lake Chl-a and ST values ranged from 15.5 to 34.3 µg/L and from 11.3 to 27.1 µg/L, respectively (Table 3.3). Target whole-lake TP values associated with the 8.0 µg/L Chl-a and 1.1 m ST standard at HC were 19 and 15 µg/L, respectively (Table 3.2 and 3.3).

3.3.4 *Annual phosphorus load values*

Annual TP loading values for both modeled and measured data sets for 2001 through 2015 were computed using the Vollenweider and Kerekes (1980) equation by applying modeled and measured estimates of whole-lake Chl-a and ST. Vollenweider-estimated TP loads for all four scenarios are shown in Table 3.3. The critical P load needed to produce the modeled whole-lake Chl-a and ST from 2001 through 2015 ranged from 123 to 1117 mg/m^2/yr and from 190 to 1042 mg/m^2/yr, respectively (Table 3.2). The critical P load needed to produce the measured whole-lake Chl-a and ST from 2001 through 2015 ranged from 195 to 897 mg/m^2/yr and from 213 to 1348 mg/m^2/yr, respectively (Table 3.3). The critical P load needed to meet the target whole-lake Chl-a and ST standard, based on the average hydrologic variability observed from 2001 through 2015, was 361 and 481 mg/m^2/yr, respectively. Thus, based on these results, P loading into Beaver Lake should not exceed 361 mg/m^2/yr in order to comply with the 8.0-µg/L Chl-a standard at HC, or 481 mg/m^2/yr in order to comply with the 1.1-m ST standard at HC (Table 3.2 and 3).

Annual Vollenweider-modeled P loads estimated from this exercise were compared to load estimations calculated by Bolyard et al. (2010) and Scott et al. (2016) for the purposes of determining whether the Vollenweider model could be used to make reasonable estimations of P loading into Beaver Lake (Figure 3.4). The 15-year average Vollenweider-measured P-load estimates were also compared to estimates calculated from photic zone TP concentrations from throughout the 2015 season. Results of this comparison revealed that Chl-a and ST Vollenweider-measured, 15-year averages were 495 and 565 mg/m^2/yr, respectively, while the 15-year average Vollenweider P load associated with 2015 TP concentrations was 509 mg/m^2/yr.

3.3.5 *Phosphorus load reductions needed to attain water quality standards*

Based on the modeled Chl-a and ST inputs, the Vollenweider model indicated that there was a surplus P load into Beaver Lake between 2001 and 2015. Modeled Chl-a and ST estimates indicated P surplus in 5 of 15 and 9 of 15 years, respectively (Table 3.2). Averaged across the 15-year data set, the P load into Beaver Lake would have needed an annual average reduction of 92 mg/m^2/yr, or an annual reduction of 20.2%, in order to support the Chl-a standard. In order for Beaver Lake to have supported the ST standard, P load into the lake would have needed an annual average reduction of 115 mg/m^2/yr, or an annual average reduction of 26.3%.

Based on the measured Chl-a and ST inputs, the Vollenweider model also indicated that there was a surplus P load into Beaver Lake between 2001 and 2015. Measured Chl-a and ST estimates indicated P surplus in 6 of 15 and 12 in 15 years, respectively. Averaged across the 15-year data set, the P load into Beaver Lake would have needed an annual average reduction of 86 mg/m^2/yr, or an annual reduction of 17.5%, in order to support the Chl-a standard. In order for Beaver Lake to have supported the ST standard, P loading into the lake would have needed an annual average reduction of 227 mg/m^2/yr, or an annual average reduction of 40.2%.

3.4 Discussion

The main objective of this study was to derive P load-reduction estimates needed to meet water quality standards at HC on Beaver Lake. The Vollenweider and Kerekes (1980) steady-state model was used along with modeled and measured whole-lake Chl-a and ST estimates for each year from 2001 through 2015 coupled with USACE and USGS hydrologic data. The most important finding of this study revealed that average of modeled and measured P loading required reductions of 18.9% and 33.3% to meet the Chl-a and ST standards, respectively. Based on long-term average conditions these load reduction percentages translate to a decreased load of 89 $mg/m^2/yr$ and 171 $mg/m^2/yr$, respectively. Because historically sampled USGS sites were spatially imbalanced to favor upstream locations, resulting TP values based on measured Chl-a and ST on a whole-lake scale were likely inflated. In other words, since ST typically increases as Chl-a decreases, these findings support the need for estimating P loading from relatively equally spaced locations, as opposed to estimating P loading from USGS locations, which heavily favored up-reservoir locations that were more biologically productive. Additionally, historic long-term TP concentrations were not always available for Beaver Lake, thus the unique opportunity existed to develop a better methodology to estimate TP along the entire riverine-lacustrine continuum using long-term data measured at the dam.

Using a complex loading model, Bolyard et al. (2010) estimated P loading into Beaver Lake from 1999 to 2008 using monitoring data collected from up-reservoir USGS gauges on the major inflowing rivers to the lake. Bolyard et al. (2010) reported that, during generally wetter years, increased runoff contributed to an increase in streamflow, where fluctuations in P loads followed the annual streamflow patterns. In identifying trophic conditions of Beaver Lake and its headwaters, Haggard et al. (1999) reported a negative relationship between mean TP concentrations and site location (i.e. headwater sites generally had greater TP concentrations than downstream sites); however, the most downstream site was in the transitional zone of the reservoir. In examining P loads delivered to Beaver Lake, Scott et al. (2016) used the software program LOADEST coupled with samples collected at multiple sites in the upper White River basin, along with streamflow data to estimate P loads. However, none of the sites were located in the riverine-lacustrine continuum, but rather farther upstream in headwaters sections of the basin (Scott et al., 2016). Much debate exists whether nutrient loading can be attributed to overland flows from a combination of land use gradients coupled with storm/base flow events (Giovannetti et al., 2013), or from internal nutrient releases from sediment-water column equilibrium dynamics (Sen et al., 2007). Regardless, understanding the role of predictive limnology (Peters, 1986), using simple input/output models, in assessing the cumulative effect of P loading into Beaver Lake is an important step in eutrophication management. These simple-input, predictive models usually require that whole-lake averages of the model input parameters be calculated and used.

Vollenweider modeled P-load estimates derived in this study were generally lower, although similar in magnitude and variation, than the measured values reported by Bolyard et al. (2010) and Scott et al. (2016). Cumulative TP loads from Bolyard et al. (2010) and Scott et al. (2016) were calculated as a sum of three reported TP load values from the three major inflows into Beaver Lake. These TP load values were reported using data from USGS stream gaging stations that were many kilometers upstream from the 2015 sampling area used in this study. Two of the three inflows were heavily dominated by agricultural production and have been previously identified as areas of concern for nonpoint source runoff. Because sites used in this study were in the main stem of Beaver Lake, far downstream of USGS gauging sites used by

Bolyard et al. (2010) and Scott et al. (2016), settling of P-bound suspended sediments could account for lower TP load values in the Vollenweider-modeled P-load estimates.

Empirical equations such as that in Vollenweider and Kerekes (1980) provide general predictions with minimal input at minimal cost and provide a “big picture” of the response to perturbations and can provide useful diagnostics to WQ managers. Because Vollenweider is a simple mathematical equation that can be calculated using simple spreadsheets, one does not need powerful computing resources to run complex-input models, nor costly program packages. A well-known modeler’s credo that values simplicity over complexity can be attributed to 15th century philosopher William of Ockham. The principle, known as Occam’s Razor, states “All things being equal, the simplest solution tends to be best.” For these reasons, results of this study indicate that the simplicity of the Vollenweider model may be its greatest strength. Beginning in 2018, Beaver Water District plans to initiate a multi-year sampling campaign at 10 of the downstream-most sites along the riverine-lacustrine continuum. By continuing these sampling methods, the Vollenweider P-loading model can be better calibrated to include year-to-year variability.

In this study, strong spatial patterns of TP concentrations have been demonstrated along the riverine-lacustrine gradient in a large southeastern US reservoir during the 2015 growing season. Using suitable regression analyses from the previous chapter, whole-lake averages of Chl-a and ST were calculated and applied to a Vollenweider steady-state model revealing that a simple, minimal-input model can be used to make reasonable estimations of P loading into Beaver Lake. Since the P-loading estimates generated in this study are similar to results from complex, multiple-input models, it appears that the Vollenweider steady state model is reasonably adequate for estimating P-load reductions into Beaver Lake. While results from this study seem robust enough for the purposes of P-load estimation, more effort needs to be applied to better understand seasonal and morphometric variabilities in Beaver Lake.

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3.6 Tables

Table 3.1. Average 2015 grab sample photic zone total phosphorus (TP) values from every date averaged by site.

Site Number	Distance from Inflow (km)	Mean TP (µg/L)
1	0	73.8
2	5	71.5
3	12	65.9
4	15	61.9
5	21	57.0
6	26	52.6
7	33	49.1
8	38	48.3
9	47	39.9
10	60	27.6
11	69	25.9
12	78	25.4

Table 3.2. Modeled whole-lake (WL) chlorophyll-a (Chl-a) and Secchi transparency (ST) parameters along with whole-lake total phosphorus (TP), and critical phosphorus loads for both parameters.

Year	WL Chl-a (µg/L)	WL ST (m)	WL TP_Chl-a (µg/L)	WL TP_ST (µg/L)	P-load_Chl-a ($mg/m^2/year$)	P-load_ST ($mg/m^2/year$)
2001	1.6	3.2	8.3	14.8	123	251
2002	6.4	2.4	21.9	20.2	611	554
2003	2.8	2.6	12.4	18.6	167	273
2004	4.4	2.5	16.9	19.5	432	516
2005	4.3	2.5	16.5	19.0	355	422
2006	5.1	3.0	18.8	16.2	228	190
2007	3.4	3.3	14.3	14.7	290	300
2008	9.8	1.7	29.4	27.8	1117	1042
2009	4.8	2.5	17.8	19.2	451	493
2010	6.1	2.6	21.3	18.2	631	521
2011	6.7	2.7	22.5	17.6	733	542
2012	3.6	2.6	14.6	18.5	288	384
2013	5.7	3.1	20.2	15.4	412	297
2014	3.3	3.3	14.0	14.4	263	272
2015	6.0	2.9	21.0	16.3	685	503
Target	5.2	3.2	19.0	15.0	481	361

Table 3.3. Measured whole-lake (WL) chlorophyll-a (Chl-a) and Secchi transparency (ST) parameters along with whole-lake total phosphorus (TP), and critical phosphorus loads for both parameters.

Year	WL Chl-a (µg/L)	WL ST (m)	WL TP_Chl-a (µg/L)	WL TP_ST (µg/L)	P-load_Chl-a (mg/m^2/year)	P-load_ST (mg/m^2/year)
2001	5.0	3.1	18.4	15.5	326	265
2002	4.3	2.2	16.7	21.8	438	608
2003	5.6	2.7	20.1	17.8	300	259
2004	2.5	2.1	11.3	22.9	266	625
2005	4.7	2.6	17.6	18.5	384	406
2006	4.3	2.7	16.5	17.8	195	213
2007	4.9	2.7	18.1	17.8	387	379
2008	7.6	1.4	24.6	34.3	897	1348
2009	8.5	2.0	26.6	24.0	734	647
2010	6.7	2.3	22.5	20.9	675	615
2011	6.5	1.5	22.2	32.0	722	1126
2012	8.7	2.3	27.1	20.9	612	445
2013	6.7	2.4	22.7	20.0	474	407
2014	6.7	2.6	22.5	18.5	471	370
2015	4.6	2.1	17.3	22.9	541	757
Target	5.2	3.2	19.0	15.0	481	361

3.7 Figure legends

Figure 3.1. Linear regression analysis between distance from inflow (km) and average photic zone total phosphorous (TP) concentrations across the 2015 season to evaluate the change in total phosphorous from inflow to dam.

Figure 3.2. Scatter plot analysis of 2001 through 2015 whole-lake (WL) values of Secchi transparency (ST) depths (m) calculated from measured US Geological Survey (USGS) data plotted against whole-lake Secchi transparency depth values calculated from 2015 modeled data from relatively equally spaced sites throughout the riverine-lacustrine continuum.

Figure 3.3. Scatter plot analysis of 2001 through 2015 whole-lake (WL) values of chlorophyll-a (Chl-a) concentrations (µg/L) calculated from measured US Geological Survey (USGS) data plotted against whole-lake chlorophyll-a concentration values calculated from 2015 modeled data from relatively equally spaced sites throughout the riverine-lacustrine continuum.

Figure 3.4. Scatter plot analysis of 2001 through 2015 measured phosphorus (P) loads (mg/m^2/year) reported by Bolyard et al. (2010) in blue, and Scott et al. (2016) in orange, plotted against Vollenweider modeled estimates of P loads in order to evaluate the magnitude and variation for the purposes of determining if the Vollenweider model can be used to make reasonable estimations of P loads into Beaver Lake.

3.8 Figures

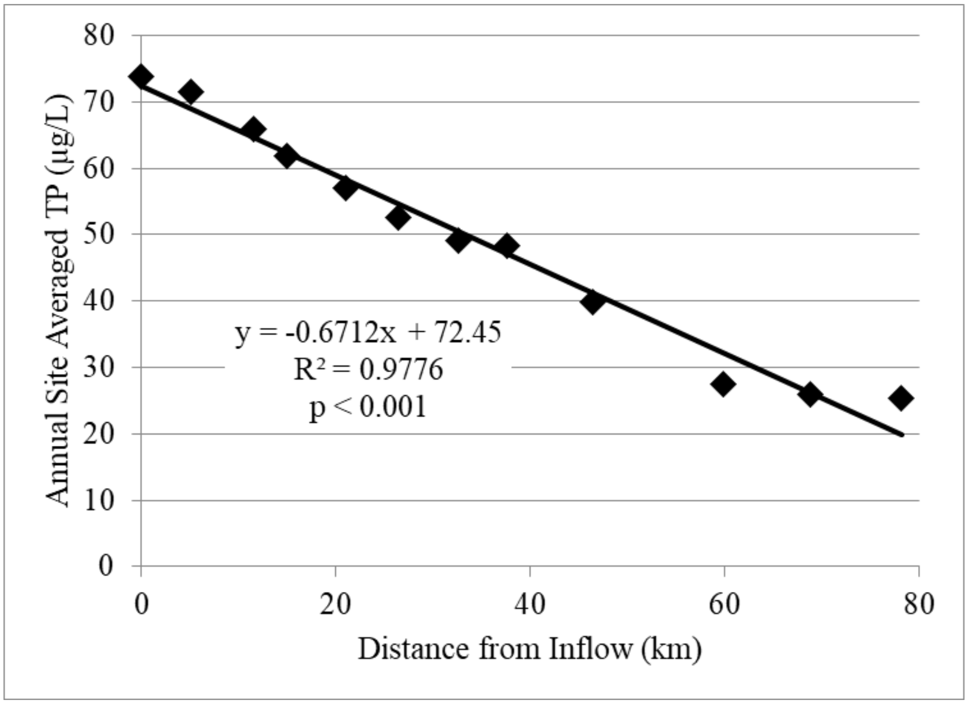

Figure 3.1

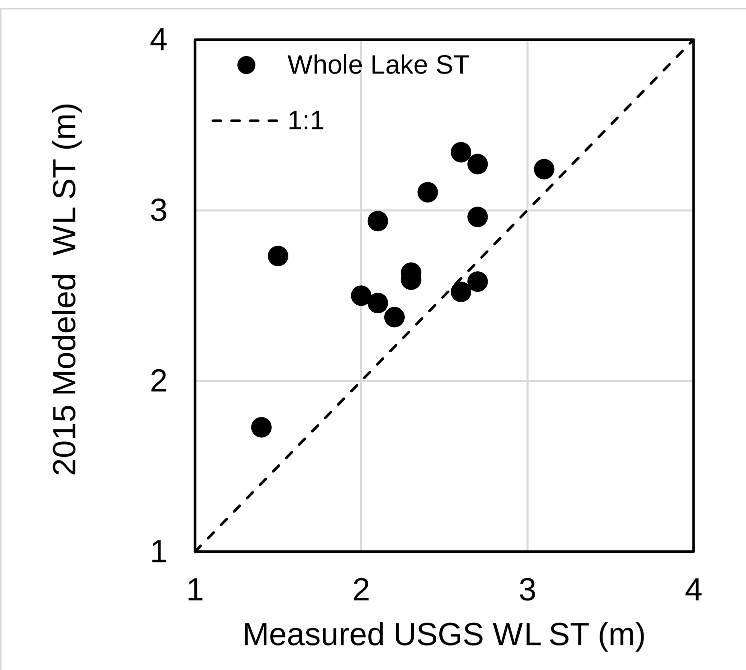

Figure 3.2

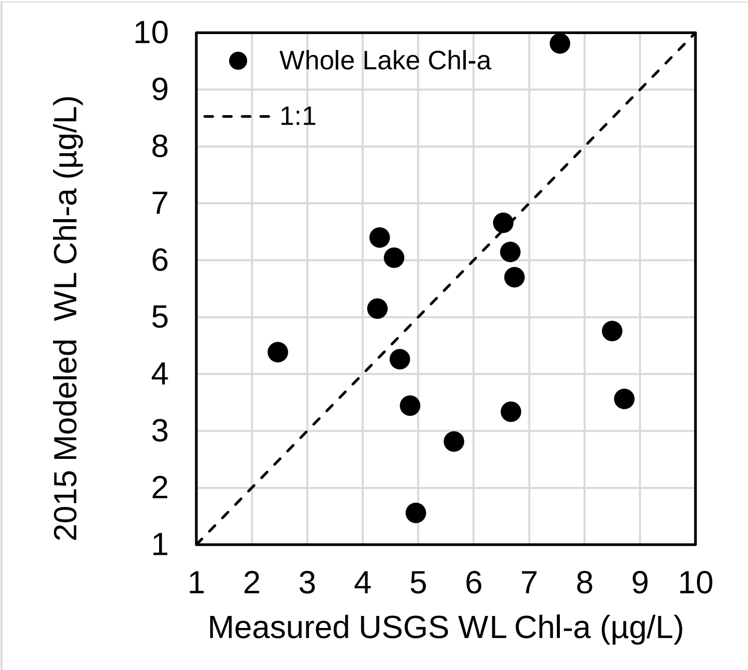

Figure 3.3

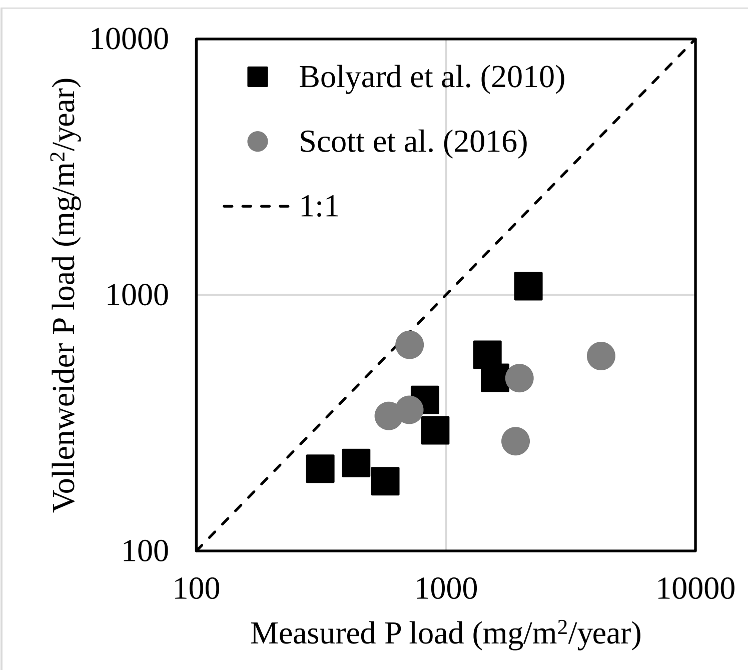

Figure 3.4

4. CONCLUSIONS

Water quality models can be defined as a set of formulas or algorithms that can be used to simulate aquatic systems and predict outcomes based on defined input data (Wagner, 2013). Models can be as simple as a single mathematical equation or complex enough to model the interactions between multiple-input parameters throughout space and time. Although water quality models can be highly beneficial in identifying best management practices (BMPs) that could potentially counteract perturbations within the watershed, the associated lag time between implementation of BMPs and the water quality response can be decadal in scale (Meals et al., 2010), thus adding uncertainty. Additionally, much debate surrounds the identification of the roles of certain nutrients in cultural eutrophication. Previous efforts focused solely on the reduction of external phosphorous (P) loading (Schindler et al. 2016), whereas, much research has emerged recently that investigates the colimitation of P and nitrogen (N) (McCarthy et al. 2017) in reducing the harmful effects of eutrophication. Therefore, proactive selection and implementation of monitoring programs is crucial to watershed characterization – a key component to source water protection programs (SWPP) that appears to be gaining favor throughout the U.S. by water resource managers (Sham et al., 2010). One aspect of a local SWPP is the implementation of a simple, minimal-input model that estimates phosphorus load into Beaver Lake. The main objectives of this thesis were to identify the spatial and temporal patterns of water quality assessment metrics and apply these patterns to steady-state modeling to estimate the phosphorous (P) load reductions necessary to meet the chlorophyll-a (Chl-a) and Secchi transparency (ST) standards that apply to Beaver Lake.

In 2015, the opportunity arose to test whether a simple, minimal-input phosphorous loading model could produce results comparable to multiple-input, complex models previously implemented on Beaver Lake, Arkansas (Walker, 1981; Bolyard et al., 2010; Scott et al., 2016). Prior to the implementation of a minimal-input model, the longitudinal and temporal patterns of Secchi transparency (ST), chlorophyll-a concentration (Chl-a), and total phosphorus (TP) within Beaver Lake were determined. Although Chl-a and ST parameters varied among the 11 temporal and 12 spatial measurements during the 2015 growing season, site-averaged Chl-a was generally greatest in the riverine/transitional zones and lowest in the lacustrine zone. Inversely, ST was generally least in the riverine/transitional zone and greatest in the lacustrine zone. When the two upper-most sites were excluded, the pattern of geometric mean Chl-a exhibited a monotonic decrease, and the pattern of arithmetic average ST exhibited a monotonic increase, as the distance from inflow increased. These annual site values of Chl-a and ST were regressed against values measured by the USGS in three overlapping sites and revealed that the Chl-a and ST models tended to underestimate USGS-measured Chl-a and ST values. However, regression analysis of the residuals from measured and calculated Chl-a data suggest that interannual variations in inflow volume into Beaver Lake can account for the model results. Additionally, the USGS historically sampled at a depth two times greater than was sampled for this research, thus, targeting a depth optimal for phytoplankton growth by the USGS. Nevertheless, the spatial productivity models of Chl-a and ST were used in conjunction with a steady-state model to estimate P-load reductions necessary to meet water quality standard adopted on Beaver Lake. First, whole lake averages of Chl-a and ST were calculated for every year between 2001 and 2015 for both modeled and measured data, as well as target whole lake averages that correspond to Beaver Lake water quality standards. Next, whole lake averages of Chl-a and ST were then converted to whole lake total P (TP) values using widely-accepted limnological models that quantify the relationship between measurable lake responses to eutrophication. Then, whole lake

TP values were used to estimate P loading into Beaver Lake using the Vollenweider and Kerekes (1980) equation. Lastly, the difference between whole lake target values of P load and P load values associated with both measured and modeled data were calculated revealing annual surplus P loads into Beaver Lake for each year.

This research incorporated widely-accepted models that are normally used in northern steady-state systems, by treating individual sections along the riverine-lacustrine gradient, as distinct parts of the whole. Research like this could be invaluable to water quality managers in Arkansas as well as other southern states in addressing cultural eutrophication by providing alternative methods to cumbersome, complex, multiple-input models. However, for a more complete comprehension of parameters that affect eutrophication in southern reservoirs, more research is needed that investigates the role that hydrologic and morphometric variables play in assessing and managing cultural eutrophication.

4.1 References

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Appendix

This appendix contains the spreadsheet of total phosphorus data throughout the 2015 season.

Date	River km	Site number	Depth (#)	Final TP (µg/L)
6/1/2015	0.0	1	1	67.0
6/1/2015	0.0	1	2	59.9
6/15/2015	0.0	1	1	31.6
6/15/2015	0.0	1	2	33.8
6/15/2015	0.0	1	3	35.5
7/1/2015	0.0	1	1	37.6
7/1/2015	0.0	1	2	48.1
7/1/2015	0.0	1	3	59.4
7/15/2015	0.0	1	1	59.4
7/15/2015	0.0	1	2	55.3
8/1/2015	0.0	1	1	32.0
8/1/2015	0.0	1	2	32.2
8/15/2015	0.0	1	1	38.6
8/15/2015	0.0	1	2	39.8
8/15/2015	0.0	1	3	40.2
8/15/2015	0.0	1	4	34.2
9/2/2015	0.0	1	1	140.2
9/2/2015	0.0	1	2	146.9
9/2/2015	0.0	1	3	147.0
9/2/2015	0.0	1	4	156.3
9/16/2015	0.0	1	1	180.9
9/16/2015	0.0	1	2	173.3
9/16/2015	0.0	1	3	159.8
9/30/2015	0.0	1	1	110.1
9/30/2015	0.0	1	2	106.4
9/30/2015	0.0	1	3	100.7
9/30/2015	0.0	1	4	104.5
10/14/2015	0.0	1	1	39.5
10/14/2015	0.0	1	2	39.4
10/14/2015	0.0	1	3	42.0
5/15/2015	5.0	2	1	96.9
6/1/2015	5.0	2	1	46.6
6/1/2015	5.0	2	2	57.8
6/15/2015	5.0	2	1	32.4
6/15/2015	5.0	2	2	32.3
6/15/2015	5.0	2	3	34.1
7/1/2015	5.0	2	1	38.2
7/1/2015	5.0	2	2	41.9
7/1/2015	5.0	2	3	38.0

Appendix (Cont.)

This appendix contains the spreadsheet of total phosphorus data throughout the 2015 season.

Date	River km	Site number	Depth (#)	Final TP (µg/L)
7/15/2015	5.0	2	1	54.5
7/15/2015	5.0	2	2	41.2
8/1/2015	5.0	2	1	27.5
8/1/2015	5.0	2	2	29.0
8/1/2015	5.0	2	3	30.4
8/1/2015	5.0	2	4	33.4
8/15/2015	5.0	2	1	28.5
8/15/2015	5.0	2	2	31.5
8/15/2015	5.0	2	3	31.6
8/15/2015	5.0	2	4	31.9
9/2/2015	5.0	2	1	149.0
9/2/2015	5.0	2	2	150.9
9/2/2015	5.0	2	3	153.9
9/2/2015	5.0	2	4	154.0
9/16/2015	5.0	2	1	161.9
9/16/2015	5.0	2	2	175.7
9/16/2015	5.0	2	3	158.7
9/30/2015	5.0	2	1	99.0
9/30/2015	5.0	2	2	106.4
9/30/2015	5.0	2	3	98.4
9/30/2015	5.0	2	4	102.9
10/14/2015	5.0	2	1	36.0
10/14/2015	5.0	2	2	36.7
10/14/2015	5.0	2	3	38.1
5/15/2015	11.6	3	1	35.0
5/15/2015	11.6	3	2	41.6
6/1/2015	11.6	3	1	63.8
6/1/2015	11.6	3	2	58.7
6/15/2015	11.6	3	1	31.6
6/15/2015	11.6	3	2	39.9
6/15/2015	11.6	3	3	29.3
7/1/2015	11.6	3	1	40.3
7/1/2015	11.6	3	2	37.2
7/1/2015	11.6	3	3	35.2
7/1/2015	11.6	3	4	39.3
7/15/2015	11.6	3	1	31.4
7/15/2015	11.6	3	2	56.0
7/15/2015	11.6	3	3	34.4
8/1/2015	11.6	3	1	24.0
8/1/2015	11.6	3	2	23.8
8/1/2015	11.6	3	3	27.3

Appendix (Cont.)

This appendix contains the spreadsheet of total phosphorus data throughout the 2015 season.

Date	River km	Site number	Depth (#)	Final TP (µg/L)
8/1/2015	11.6	3	4	34.0
8/15/2015	11.6	3	1	23.7
8/15/2015	11.6	3	2	24.4
8/15/2015	11.6	3	3	24.3
8/15/2015	11.6	3	4	27.6
9/2/2015	11.6	3	1	137.9
9/2/2015	11.6	3	2	150.7
9/2/2015	11.6	3	3	151.1
9/2/2015	11.6	3	4	155.0
9/16/2015	11.6	3	1	149.4
9/16/2015	11.6	3	2	174.3
9/16/2015	11.6	3	3	161.1
9/16/2015	11.6	3	4	159.8
9/30/2015	11.6	3	1	91.2
9/30/2015	11.6	3	2	102.7
9/30/2015	11.6	3	3	97.9
9/30/2015	11.6	3	4	160.0
9/30/2015	11.6	3	5	137.7
10/14/2015	11.6	3	1	33.0
10/14/2015	11.6	3	2	30.8
10/14/2015	11.6	3	3	34.6
5/15/2015	15.0	4	1	40.9
5/15/2015	15.0	4	2	44.0
6/1/2015	15.0	4	1	58.8
6/1/2015	15.0	4	2	51.4
6/15/2015	15.0	4	1	32.0
6/15/2015	15.0	4	2	31.7
6/15/2015	15.0	4	3	27.1
7/1/2015	15.0	4	1	37.5
7/1/2015	15.0	4	2	41.2
7/1/2015	15.0	4	3	32.4
7/1/2015	15.0	4	4	36.3
7/15/2015	15.0	4	1	29.3
7/15/2015	15.0	4	2	42.2
7/15/2015	15.0	4	3	31.3
7/15/2015	15.0	4	4	25.6
8/1/2015	15.0	4	1	22.9
8/1/2015	15.0	4	2	23.8
8/1/2015	15.0	4	3	24.2
8/1/2015	15.0	4	4	29.5
8/15/2015	15.0	4	1	20.8

Appendix (Cont.)

This appendix contains the spreadsheet of total phosphorus data throughout the 2015 season.

Date	River km	Site number	Depth (#)	Final TP (µg/L)
8/15/2015	15.0	4	2	21.6
8/15/2015	15.0	4	3	22.8
8/15/2015	15.0	4	4	26.0
9/2/2015	15.0	4	1	155.3
9/2/2015	15.0	4	2	152.8
9/2/2015	15.0	4	3	159.3
9/2/2015	15.0	4	4	155.4
9/2/2015	15.0	4	5	156.9
9/16/2015	15.0	4	1	155.7
9/16/2015	15.0	4	2	171.1
9/16/2015	15.0	4	3	149.3
9/16/2015	15.0	4	4	134.9
9/30/2015	15.0	4	1	92.7
9/30/2015	15.0	4	2	101.1
9/30/2015	15.0	4	3	90.7
9/30/2015	15.0	4	4	88.2
9/30/2015	15.0	4	5	99.6
10/14/2015	15.0	4	1	30.0
10/14/2015	15.0	4	2	31.3
10/14/2015	15.0	4	3	32.2
10/14/2015	15.0	4	4	41.1
5/15/2015	21.1	5	1	37.5
5/15/2015	21.1	5	2	34.5
6/1/2015	21.1	5	1	51.3
6/1/2015	21.1	5	2	51.4
6/1/2015	21.1	5	3	41.5
6/15/2015	21.1	5	1	32.9
6/15/2015	21.1	5	2	33.2
6/15/2015	21.1	5	3	28.1
7/1/2015	21.1	5	1	25.2
7/1/2015	21.1	5	2	32.6
7/1/2015	21.1	5	3	36.8
7/1/2015	21.1	5	4	30.2
7/15/2015	21.1	5	1	17.3
7/15/2015	21.1	5	2	26.7
7/15/2015	21.1	5	3	25.6
7/15/2015	21.1	5	4	21.6
8/1/2015	21.1	5	1	22.2
8/1/2015	21.1	5	2	27.8
8/1/2015	21.1	5	3	26.3
8/1/2015	21.1	5	4	29.6

Appendix (Cont.)

This appendix contains the spreadsheet of total phosphorus data throughout the 2015 season.

Date	River km	Site number	Depth (#)	Final TP (μg/L)
8/1/2015	21.1	5	5	29.8
8/15/2015	21.1	5	1	19.1
8/15/2015	21.1	5	2	20.0
8/15/2015	21.1	5	3	22.0
8/15/2015	21.1	5	4	23.4
8/15/2015	21.1	5	5	25.6
9/2/2015	21.1	5	1	136.6
9/2/2015	21.1	5	2	129.4
9/2/2015	21.1	5	3	150.8
9/2/2015	21.1	5	4	147.9
9/2/2015	21.1	5	5	158.9
9/16/2015	21.1	5	1	139.2
9/16/2015	21.1	5	2	142.8
9/16/2015	21.1	5	3	142.1
9/16/2015	21.1	5	4	140.6
9/16/2015	21.1	5	5	133.7
9/30/2015	21.1	5	1	84.7
9/30/2015	21.1	5	2	85.8
9/30/2015	21.1	5	3	85.2
9/30/2015	21.1	5	4	85.0
9/30/2015	21.1	5	5	134.0
10/14/2015	21.1	5	1	29.9
10/14/2015	21.1	5	2	29.0
10/14/2015	21.1	5	3	28.1
10/14/2015	21.1	5	4	29.2
5/15/2015	26.4	6	1	17.0
5/15/2015	26.4	6	2	21.7
5/15/2015	26.4	6	3	23.0
5/15/2015	26.4	6	4	22.3
6/1/2015	26.4	6	1	47.9
6/1/2015	26.4	6	2	50.0
6/1/2015	26.4	6	3	49.4
6/15/2015	26.4	6	1	35.8
6/15/2015	26.4	6	2	35.6
6/15/2015	26.4	6	3	29.8
6/15/2015	26.4	6	4	28.6
7/1/2015	26.4	6	1	22.6
7/1/2015	26.4	6	2	24.5
7/1/2015	26.4	6	3	28.8
7/1/2015	26.4	6	4	28.1
7/15/2015	26.4	6	1	26.8

Appendix (Cont.)

This appendix contains the spreadsheet of total phosphorus data throughout the 2015 season.

Date	River km	Site number	Depth (#)	Final TP (µg/L)
7/15/2015	26.4	6	2	23.5
7/15/2015	26.4	6	3	22.8
7/15/2015	26.4	6	4	22.1
8/1/2015	26.4	6	1	24.6
8/1/2015	26.4	6	2	25.5
8/1/2015	26.4	6	3	27.0
8/1/2015	26.4	6	4	26.8
8/1/2015	26.4	6	5	27.6
8/15/2015	26.4	6	1	19.5
8/15/2015	26.4	6	2	19.4
8/15/2015	26.4	6	3	19.9
8/15/2015	26.4	6	4	25.0
8/15/2015	26.4	6	5	27.8
9/2/2015	26.4	6	1	156.7
9/2/2015	26.4	6	2	143.6
9/2/2015	26.4	6	3	132.3
9/2/2015	26.4	6	4	149.3
9/2/2015	26.4	6	5	141.9
9/2/2015	26.4	6	6	166.9
9/16/2015	26.4	6	1	130.8
9/16/2015	26.4	6	2	121.8
9/16/2015	26.4	6	3	126.3
9/16/2015	26.4	6	4	133.5
9/16/2015	26.4	6	5	128.9
9/30/2015	26.4	6	1	77.1
9/30/2015	26.4	6	2	71.4
9/30/2015	26.4	6	3	76.3
9/30/2015	26.4	6	4	79.5
9/30/2015	26.4	6	5	76.7
9/30/2015	26.4	6	6	79.3
10/14/2015	26.4	6	1	23.4
10/14/2015	26.4	6	2	21.0
10/14/2015	26.4	6	3	26.3
10/14/2015	26.4	6	4	25.4
10/14/2015	26.4	6	5	24.2
5/15/2015	32.7	7	1	13.2
5/15/2015	32.7	7	2	14.9
5/15/2015	32.7	7	3	14.8
5/15/2015	32.7	7	4	16.4
5/15/2015	32.7	7	5	13.3
5/15/2015	32.7	7	6	11.7

Appendix (Cont.)

This appendix contains the spreadsheet of total phosphorus data throughout the 2015 season.

Date	River km	Site number	Depth (#)	Final TP (µg/L)
6/1/2015	32.7	7	1	26.7
6/1/2015	32.7	7	2	45.0
6/1/2015	32.7	7	3	34.4
6/1/2015	32.7	7	4	45.3
6/15/2015	32.7	7	1	26.7
6/15/2015	32.7	7	2	25.9
6/15/2015	32.7	7	3	29.4
6/15/2015	32.7	7	4	29.0
7/1/2015	32.7	7	1	19.0
7/1/2015	32.7	7	2	21.2
7/1/2015	32.7	7	3	24.3
7/1/2015	32.7	7	4	22.4
7/1/2015	32.7	7	5	26.3
7/15/2015	32.7	7	1	19.5
7/15/2015	32.7	7	2	22.8
7/15/2015	32.7	7	3	22.3
7/15/2015	32.7	7	4	19.7
7/15/2015	32.7	7	5	10.4
8/1/2015	32.7	7	1	20.5
8/1/2015	32.7	7	2	20.1
8/1/2015	32.7	7	3	19.1
8/1/2015	32.7	7	4	27.3
8/1/2015	32.7	7	5	21.5
8/15/2015	32.7	7	1	19.9
8/15/2015	32.7	7	2	19.7
8/15/2015	32.7	7	3	24.1
8/15/2015	32.7	7	4	20.0
8/15/2015	32.7	7	5	22.6
8/15/2015	32.7	7	6	29.1
9/2/2015	32.7	7	1	143.0
9/2/2015	32.7	7	2	156.4
9/2/2015	32.7	7	3	151.8
9/2/2015	32.7	7	4	136.5
9/2/2015	32.7	7	5	131.7
9/2/2015	32.7	7	6	152.4
9/16/2015	32.7	7	1	120.1
9/16/2015	32.7	7	2	134.7
9/16/2015	32.7	7	3	129.2
9/16/2015	32.7	7	4	139.1
9/16/2015	32.7	7	5	124.6
9/30/2015	32.7	7	1	70.9

Appendix (Cont.)

This appendix contains the spreadsheet of total phosphorus data throughout the 2015 season.

Date	River km	Site number	Depth (#)	Final TP (µg/L)
9/30/2015	32.7	7	2	78.4
9/30/2015	32.7	7	3	75.4
9/30/2015	32.7	7	4	82.6
9/30/2015	32.7	7	5	74.7
9/30/2015	32.7	7	6	79.2
10/14/2015	32.7	7	1	21.5
10/14/2015	32.7	7	2	22.2
10/14/2015	32.7	7	3	21.2
10/14/2015	32.7	7	4	26.3
10/14/2015	32.7	7	5	24.8
5/15/2015	37.7	8	1	15.6
5/15/2015	37.7	8	2	14.0
5/15/2015	37.7	8	3	13.8
6/1/2015	37.7	8	1	27.3
6/1/2015	37.7	8	2	26.6
6/1/2015	37.7	8	3	26.1
6/1/2015	37.7	8	4	66.4
6/15/2015	37.7	8	1	26.2
6/15/2015	37.7	8	2	23.8
6/15/2015	37.7	8	3	24.3
6/15/2015	37.7	8	4	22.8
6/15/2015	37.7	8	5	32.7
7/1/2015	37.7	8	1	15.3
7/1/2015	37.7	8	2	18.2
7/1/2015	37.7	8	3	16.8
7/1/2015	37.7	8	4	20.8
7/15/2015	37.7	8	1	18.8
7/15/2015	37.7	8	2	18.7
7/15/2015	37.7	8	3	17.5
7/15/2015	37.7	8	4	18.2
7/15/2015	37.7	8	5	21.8
8/1/2015	37.7	8	1	17.6
8/1/2015	37.7	8	2	18.1
8/1/2015	37.7	8	3	18.8
8/1/2015	37.7	8	4	20.3
8/1/2015	37.7	8	5	16.6
8/15/2015	37.7	8	1	20.2
8/15/2015	37.7	8	2	23.0
8/15/2015	37.7	8	3	34.6
8/15/2015	37.7	8	4	28.1
8/15/2015	37.7	8	5	30.4

Appendix (Cont.)

This appendix contains the spreadsheet of total phosphorus data throughout the 2015 season.

Date	River km	Site number	Depth (#)	Final TP (µg/L)
9/2/2015	37.7	8	1	136.3
9/2/2015	37.7	8	2	132.3
9/2/2015	37.7	8	3	140.5
9/2/2015	37.7	8	4	139.2
9/16/2015	37.7	8	1	126.6
9/16/2015	37.7	8	2	143.0
9/16/2015	37.7	8	3	123.9
9/16/2015	37.7	8	4	134.7
9/16/2015	37.7	8	5	141.7
9/16/2015	37.7	8	6	132.5
9/30/2015	37.7	8	1	74.7
9/30/2015	37.7	8	2	83.1
9/30/2015	37.7	8	3	74.2
9/30/2015	37.7	8	4	79.4
9/30/2015	37.7	8	5	82.7
10/14/2015	37.7	8	1	23.0
10/14/2015	37.7	8	2	23.1
10/14/2015	37.7	8	3	24.6
10/14/2015	37.7	8	4	24.0
10/14/2015	37.7	8	5	23.7
5/15/2015	46.5	9	1	9.5
5/15/2015	46.5	9	2	12.5
5/15/2015	46.5	9	3	10.3
5/15/2015	46.5	9	4	13.6
5/15/2015	46.5	9	5	13.2
5/15/2015	46.5	9	6	18.2
6/1/2015	46.5	9	1	29.4
6/1/2015	46.5	9	2	62.3
6/1/2015	46.5	9	3	49.3
6/1/2015	46.5	9	4	38.8
6/1/2015	46.5	9	5	33.0
6/15/2015	46.5	9	1	17.2
6/15/2015	46.5	9	2	17.7
6/15/2015	46.5	9	3	15.8
6/15/2015	46.5	9	4	17.8
6/15/2015	46.5	9	5	21.2
7/1/2015	46.5	9	1	13.9
7/1/2015	46.5	9	2	15.1
7/1/2015	46.5	9	3	14.0
7/1/2015	46.5	9	4	25.0
7/1/2015	46.5	9	5	22.7

Appendix (Cont.)

This appendix contains the spreadsheet of total phosphorus data throughout the 2015 season.

Date	River km	Site number	Depth (#)	Final TP (µg/L)
7/1/2015	46.5	9	6	21.8
7/15/2015	46.5	9	1	14.5
7/15/2015	46.5	9	2	21.8
7/15/2015	46.5	9	3	18.5
7/15/2015	46.5	9	4	14.1
7/15/2015	46.5	9	5	17.2
7/15/2015	46.5	9	6	20.5
8/1/2015	46.5	9	1	13.2
8/1/2015	46.5	9	2	13.1
8/1/2015	46.5	9	3	12.3
8/1/2015	46.5	9	4	16.2
8/1/2015	46.5	9	5	21.0
8/1/2015	46.5	9	6	22.7
8/15/2015	46.5	9	1	18.7
8/15/2015	46.5	9	2	17.9
8/15/2015	46.5	9	3	18.0
8/15/2015	46.5	9	4	25.4
8/15/2015	46.5	9	5	27.6
8/15/2015	46.5	9	6	29.9
9/2/2015	46.5	9	1	130.1
9/2/2015	46.5	9	2	133.8
9/2/2015	46.5	9	3	124.6
9/2/2015	46.5	9	4	131.9
9/2/2015	46.5	9	5	127.8
9/2/2015	46.5	9	6	158.2
9/16/2015	46.5	9	1	135.0
9/16/2015	46.5	9	2	121.5
9/16/2015	46.5	9	3	135.7
9/16/2015	46.5	9	4	19.2
9/16/2015	46.5	9	5	24.1
9/16/2015	46.5	9	6	23.1
9/30/2015	46.5	9	1	88.1
9/30/2015	46.5	9	2	71.5
9/30/2015	46.5	9	3	77.1
9/30/2015	46.5	9	4	20.1
9/30/2015	46.5	9	5	22.4
9/30/2015	46.5	9	6	40.4
10/14/2015	46.5	9	1	41.0
10/14/2015	46.5	9	2	21.6
10/14/2015	46.5	9	3	18.4
10/14/2015	46.5	9	4	21.0

Appendix (Cont.)

This appendix contains the spreadsheet of total phosphorus data throughout the 2015 season.

Date	River km	Site number	Depth (#)	Final TP (μg/L)
10/14/2015	46.5	9	5	20.7
10/14/2015	46.5	9	6	32.9
5/15/2015	59.9	10	1	9.3
5/15/2015	59.9	10	2	11.0
5/15/2015	59.9	10	3	9.4
5/15/2015	59.9	10	4	8.4
5/15/2015	59.9	10	5	9.1
5/15/2015	59.9	10	6	7.0
6/1/2015	59.9	10	1	45.4
6/1/2015	59.9	10	2	16.7
6/1/2015	59.9	10	3	17.2
6/1/2015	59.9	10	4	18.4
6/1/2015	59.9	10	5	19.9
6/1/2015	59.9	10	6	29.6
6/15/2015	59.9	10	1	14.5
6/15/2015	59.9	10	2	15.0
6/15/2015	59.9	10	3	15.6
6/15/2015	59.9	10	4	15.5
6/15/2015	59.9	10	5	14.3
6/15/2015	59.9	10	6	16.0
7/1/2015	59.9	10	1	12.1
7/1/2015	59.9	10	2	12.1
7/1/2015	59.9	10	3	11.8
7/1/2015	59.9	10	4	16.6
7/1/2015	59.9	10	5	18.7
7/1/2015	59.9	10	6	24.5
7/15/2015	59.9	10	1	12.1
7/15/2015	59.9	10	2	12.7
7/15/2015	59.9	10	3	11.8
7/15/2015	59.9	10	4	12.4
7/15/2015	59.9	10	5	15.7
7/15/2015	59.9	10	6	16.5
8/1/2015	59.9	10	1	14.3
8/1/2015	59.9	10	2	12.9
8/1/2015	59.9	10	3	14.5
8/1/2015	59.9	10	4	22.6
8/1/2015	59.9	10	5	24.3
8/1/2015	59.9	10	6	27.1
8/15/2015	59.9	10	1	19.1
8/15/2015	59.9	10	2	19.6
8/15/2015	59.9	10	3	18.7

Appendix (Cont.)

This appendix contains the spreadsheet of total phosphorus data throughout the 2015 season.

Date	River km	Site number	Depth (#)	Final TP (µg/L)
8/15/2015	59.9	10	4	18.7
8/15/2015	59.9	10	5	31.2
8/15/2015	59.9	10	6	26.2
9/2/2015	59.9	10	1	114.9
9/2/2015	59.9	10	2	119.9
9/2/2015	59.9	10	3	124.2
9/2/2015	59.9	10	4	120.8
9/2/2015	59.9	10	5	132.4
9/2/2015	59.9	10	6	150.8
9/16/2015	59.9	10	1	17.9
9/16/2015	59.9	10	2	18.0
9/16/2015	59.9	10	3	17.6
9/16/2015	59.9	10	4	17.7
9/16/2015	59.9	10	5	16.5
9/16/2015	59.9	10	6	24.6
9/30/2015	59.9	10	1	18.2
9/30/2015	59.9	10	2	19.6
9/30/2015	59.9	10	3	17.6
9/30/2015	59.9	10	4	18.9
9/30/2015	59.9	10	5	16.7
9/30/2015	59.9	10	6	21.1
10/14/2015	59.9	10	1	18.5
10/14/2015	59.9	10	2	21.3
10/14/2015	59.9	10	3	17.7
10/14/2015	59.9	10	4	20.3
10/14/2015	59.9	10	5	17.0
10/14/2015	59.9	10	6	17.9
5/15/2015	68.9	11	1	3.9
5/15/2015	68.9	11	2	4.7
5/15/2015	68.9	11	3	5.0
5/15/2015	68.9	11	4	3.5
5/15/2015	68.9	11	5	4.8
5/15/2015	68.9	11	6	0.5
6/1/2015	68.9	11	1	28.6
6/1/2015	68.9	11	2	19.6
6/1/2015	68.9	11	3	15.0
6/1/2015	68.9	11	4	14.6
6/1/2015	68.9	11	5	15.4
6/1/2015	68.9	11	6	16.5
6/15/2015	68.9	11	1	19.8
6/15/2015	68.9	11	2	14.6

Appendix (Cont.)

This appendix contains the spreadsheet of total phosphorus data throughout the 2015 season.

Date	River km	Site number	Depth (#)	Final TP (μg/L)
6/15/2015	68.9	11	3	13.9
6/15/2015	68.9	11	4	19.7
6/15/2015	68.9	11	5	17.6
6/15/2015	68.9	11	6	14.0
7/1/2015	68.9	11	1	10.0
7/1/2015	68.9	11	2	11.7
7/1/2015	68.9	11	3	10.9
7/1/2015	68.9	11	4	11.2
7/1/2015	68.9	11	5	19.4
7/1/2015	68.9	11	6	16.2
7/15/2015	68.9	11	1	10.0
7/15/2015	68.9	11	2	8.5
7/15/2015	68.9	11	3	9.9
7/15/2015	68.9	11	4	10.6
7/15/2015	68.9	11	5	14.7
7/15/2015	68.9	11	6	17.9
8/1/2015	68.9	11	1	15.3
8/1/2015	68.9	11	2	20.6
8/1/2015	68.9	11	3	20.5
8/1/2015	68.9	11	4	19.8
8/1/2015	68.9	11	5	23.2
8/1/2015	68.9	11	6	21.1
8/15/2015	68.9	11	1	15.0
8/15/2015	68.9	11	2	15.7
8/15/2015	68.9	11	3	14.0
8/15/2015	68.9	11	4	17.4
8/15/2015	68.9	11	5	19.1
8/15/2015	68.9	11	6	18.3
9/2/2015	68.9	11	1	125.1
9/2/2015	68.9	11	2	122.6
9/2/2015	68.9	11	3	128.5
9/2/2015	68.9	11	4	130.1
9/2/2015	68.9	11	5	130.1
9/2/2015	68.9	11	6	150.2
9/16/2015	68.9	11	1	16.7
9/16/2015	68.9	11	2	18.0
9/16/2015	68.9	11	3	16.5
9/16/2015	68.9	11	4	18.1
9/16/2015	68.9	11	5	17.8
9/16/2015	68.9	11	6	24.4
9/30/2015	68.9	11	1	17.9

Appendix (Cont.)

This appendix contains the spreadsheet of total phosphorus data throughout the 2015 season.

Date	River km	Site number	Depth (#)	Final TP (µg/L)
9/30/2015	68.9	11	2	17.2
9/30/2015	68.9	11	3	17.1
9/30/2015	68.9	11	4	17.9
9/30/2015	68.9	11	5	16.8
9/30/2015	68.9	11	6	19.6
10/14/2015	68.9	11	1	19.1
10/14/2015	68.9	11	2	16.4
10/14/2015	68.9	11	3	17.7
10/14/2015	68.9	11	4	17.7
10/14/2015	68.9	11	5	15.7
10/14/2015	68.9	11	6	14.8
5/15/2015	78.2	12	1	2.2
5/15/2015	78.2	12	2	3.1
5/15/2015	78.2	12	3	4.7
5/15/2015	78.2	12	4	11.5
5/15/2015	78.2	12	5	3.5
5/15/2015	78.2	12	6	5.2
6/1/2015	78.2	12	1	14.8
6/1/2015	78.2	12	2	15.6
6/1/2015	78.2	12	3	15.6
6/1/2015	78.2	12	4	14.7
6/1/2015	78.2	12	5	14.1
6/1/2015	78.2	12	6	17.8
6/15/2015	78.2	12	1	11.1
6/15/2015	78.2	12	2	12.1
6/15/2015	78.2	12	3	12.0
6/15/2015	78.2	12	4	12.6
6/15/2015	78.2	12	5	13.7
6/15/2015	78.2	12	6	12.7
7/1/2015	78.2	12	1	10.6
7/1/2015	78.2	12	2	9.9
7/1/2015	78.2	12	3	14.7
7/1/2015	78.2	12	4	12.1
7/1/2015	78.2	12	5	18.6
7/1/2015	78.2	12	6	12.4
7/15/2015	78.2	12	1	9.5
7/15/2015	78.2	12	2	13.1
7/15/2015	78.2	12	3	12.0
7/15/2015	78.2	12	4	11.7
7/15/2015	78.2	12	5	13.5
7/15/2015	78.2	12	6	13.4

Appendix (Cont.)

This appendix contains the spreadsheet of total phosphorus data throughout the 2015 season.

Date	River km	Site number	Depth (#)	Final TP (µg/L)
8/1/2015	78.2	12	1	12.1
8/1/2015	78.2	12	2	16.8
8/1/2015	78.2	12	3	19.6
8/1/2015	78.2	12	4	16.3
8/1/2015	78.2	12	5	17.7
8/15/2015	78.2	12	1	12.1
8/15/2015	78.2	12	2	12.2
8/15/2015	78.2	12	3	17.3
8/15/2015	78.2	12	4	16.9
8/15/2015	78.2	12	5	16.9
8/15/2015	78.2	12	6	16.4
9/2/2015	78.2	12	1	127.7
9/2/2015	78.2	12	2	139.8
9/2/2015	78.2	12	3	133.0
9/2/2015	78.2	12	4	129.2
9/2/2015	78.2	12	5	136.7
9/2/2015	78.2	12	6	155.7
9/16/2015	78.2	12	1	16.0
9/16/2015	78.2	12	2	16.4
9/16/2015	78.2	12	3	18.4
9/16/2015	78.2	12	4	19.5
9/16/2015	78.2	12	5	21.9
9/16/2015	78.2	12	6	19.3
9/30/2015	78.2	12	1	15.8
9/30/2015	78.2	12	2	17.0
9/30/2015	78.2	12	3	17.1
9/30/2015	78.2	12	4	18.2
9/30/2015	78.2	12	5	19.7
9/30/2015	78.2	12	6	18.1
10/14/2015	78.2	12	1	15.6
10/14/2015	78.2	12	2	17.7
10/14/2015	78.2	12	3	15.9
10/14/2015	78.2	12	4	17.0
10/14/2015	78.2	12	5	17.5
10/14/2015	78.2	12	6	16.8