Jordan Lake Responses to Reduced Nutrient Loading: Results from a New Three-Dimensional Mechanistic Water Quality Model

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Abstract

A new three-dimensional mass-balance-based water quality model implementation of EFDC (Environmental Fluid Dynamics Code) was developed for Jordan Lake, North Carolina. The model considered the time-varying inputs of water, nutrients, organic matter, and dissolved oxygen from atmospheric, surface waters (rivers and creeks), and benthic sediments for five calendar years (January 1, 2014 - December 31, 2018). Two separate two-year time periods (2014-2015, 2017-2018) were used to calibrate the physical, chemical, and biological rate processes used by the model. A third time period (2016 calendar year) was used to validate the model calibration. The hydrodynamic and water quality model simulated the temporal and spatial dynamics in lake circulation and water quality for the two calibrated time periods for a base case and a set of scenarios that considered nitrogen and/or phosphorus load reductions from zero to fifty percent. These load reduction scenarios were used to estimate potential reductions in phytoplankton abundance as measured by chlorophyll a concentration that might be expected for load reductions of inorganic and organic nitrogen and phosphorus. Separate hydrodynamic model runs that simulated releases of dye tracer into the lake were used to estimate the residence time of lake waters for different regions of the lake, and the relative contributions of surface water inputs from the Haw River and New Hope Creek arms of the lake. The model input files were also used to quantify the relative nutrient load contributions from various atmospheric, surface water, and benthic sources.

The nutrient loading analysis for Jordan Lake indicated that the majority of nutrients entered the lake in organic forms that were not immediately bioavailable. The majority of water and nutrients entered the lake from the Haw River arm, but on a long-term basis only a small fraction of these inputs moved up into the upper reaches of the New Hope Creek arm of the lake. Some high flow events, however, did transport Haw River water throughout the lake, but these high flow events did not contribute significantly to the flushing of the New Hope Creek arm of Jordan Lake. In general, local surface water sources (Morgan, New Hope, Northeast, and other smaller creeks) provided the majority of water and nutrients to the New Hope Creek arm of the lake. Atmospheric deposition was a relatively minor source of nutrients to the lake. Based upon water quality model results, benthic sediments acted as a significant sink for the particulate fraction of organic nutrients, nitrate, and dissolved oxygen. Benthic sediments were also the major source of bioavailable nutrients, providing more than 75% of phosphate and 90% of ammonia to the lake.

The relatively shallow waters and very limited flushing of the New Hope Creek arm of Jordan Lake provided highly favorable conditions for accumulation of algal biomass. For the five-year time period studied (2014-2018), the observed 90^{th} percentile photic-zone chlorophyll a concentration at eighteen monitoring stations across Jordan Lake was 72 μg/L, which is 44% above the North Carolina water quality criteria value of 40 μg/L. Three stations in the upper portion of the New Hope Creek arm of the lake had 90^{th} percentile photic-zone chlorophyll a concentrations that were more than twice the 40 μg/L criteria. In general the model did a good job in simulating the distribution of chlorophyll a concentrations within the lake, and the fraction of time that high chlorophyll a concentrations were present in Jordan Lake. For the base case simulations, more than forty percent of observed and corresponding model predictions of chlorophyll a concentration were above the North Carolina water quality criteria of 40 μg/L (10% is the allowable exceedance). This analysis indicates that a substantial (40-50%) decrease in phytoplankton biomass would be needed to meet the existing NC water quality criteria for chlorophyll a.

A range of nutrient reduction scenarios was simulated with the model, with reductions of nitrogen and/or phosphorus loading via surface waters ranging from zero to fifty percent. Algal abundances were sensitive to reductions in both nitrogen (N) and phosphorus (P). Likely as a result of the predominance of the benthic source of phosphate to the water column, algal abundances were more sensitive to N load reductions than P load reductions. Likely also as a consequence of benthic nutrient loading, percentage reductions in biomass were generally less than the corresponding reduction in surface water nutrient loading. For instance, N and P load reductions of 10%, 30%, and 50% reduced algal biomass by 3%, 13%, and 23%. Additional scenarios examined the consequences of removing the causeways or having different load reductions for the Haw River and New Hope Creek arms of the lake. Neither of these scenarios produced results significantly different than the corresponding base cases. Overall, none of the scenarios tested was able to produce sufficient biomass reductions to meet the water quality criteria value of 40 μg/L. Accordingly, exceedances of the 40 μg/L criteria value for chlorophyll a were well above the regulatory limit of 10% for all the scenarios tested. The 10%, 30%, and 50% nutrient load reductions decreased the percentage of chlorophyll a concentrations above 40 μg/L from 45% to 43%, 38%, and 32%, respectively. A scenario using the predictive sediment diagenesis model showed that sustained reductions in nutrient loading would eventually produce a larger positive effect on chlorophyll conditions, but it would take more than ten years to see a significant positive change in the water quality of the lake due to the relatively slow response time of changes in benthic sediment conditions.

Introduction

The North Carolina Policy Collaboratory was established by the state legislature to utilize and disseminate the environmental research expertise of the University of North Carolina for practical use by state and local governments. In 2016, the legislature approved a budget provision to develop a new, comprehensive nutrient management regulatory framework. The provision directed the Collaboratory to oversee a continuing study and analysis of nutrient management strategies and the compilation of existing water quality data for Jordan Lake. During 2019, a number of research projects, referred to collectively as the Jordan Lake Nutrient Management Study, were initiated under this provision. The resulting scientific findings have been integrated into a three-dimensional mass-balance-based simulation model of Jordan Lake. The Jordan Lake Nutrient Response Model, reported on herein, evaluates (1) the Lake's potential for eutrophication relative to nutrient loads, streamflow patterns, and climate, for both current conditions and future scenarios, and (2) the potential for nutrient mitigation by implementing best management practices, regulatory measures and restoration efforts.

The Jordan Lake watershed lies within the Cape Fear river basin in the Piedmont region of North Carolina (Figure 1). The Jordan Lake Nutrient Response Model is a numerical simulation of physical, chemical, and biological processes in the lake and underlying sediments. Water, nutrients, and organic matter constitute input loads to the lake at its inflow boundaries. Within the lake and its sediments, physical, chemical, and biological transformations occur under the prevailing conditions of heat and light. The quality of water in the lake and its outflow is transformed as a result.

This report describes the setup, calibration, and scenario testing with a newly developed Jordan Lake Nutrient Response Model that used a five year monitoring dataset (2014-2018). The model was developed to test how reductions in watershed loadings of nutrients, specifically nitrogen and phosphorus, would be expected to affect the water quality conditions in the lake. Of primary interest are the chlorophyll a concentrations in the lake for various load reduction scenarios. Chlorophyll a is an essential pigment in phytoplankton cells that is commonly used as quantitative measure of algal abundance. A second project objective was to better understand the interactions between the Jordan Lake watershed, the underlying benthic sediments of Jordan Lake, and the physical, chemical, and biological conditions in the water column of Jordan Lake.

The work follows an earlier study (Tetra Tech, 2002; Tetra Tech Inc., 2003) that used monitoring data from 1993-2001 to create a nutrient response model of the lake as a coupled EFDC/WASP application (Ambrose et al., 1993; Hamrick, 1992). The model described in this report takes advantage of a large amount of newly collected physical, chemical, and biological information on the lake, and reflects the latest conditions with respect to development within the Jordan Lake watershed. The newly developed model also takes advantage of advances in the capabilities of mass-balance based water quality models. The Jordan Lake model developed here utilizes a new computational model grid and a predictive sediment diagenesis submodel. These advances allow for a better accounting of the short and long-term responses that would be expected under a scenario that significantly reduces nutrient loading to the lake.

The following section provides a description of the numerical model (EFDC) that was used as the basis of the Jordan Lake nutrient response model. A literature review of similar EFDC modeling projects is also provided. Some background information on Jordan Lake and a summary of the observed chlorophyll a concentrations for the 2014-2018 model time period is provided in the system description section. Model setup and calibration describes the data sources used and the method for calibrating the model to the observed data on the physical, chemical, and biological conditions during the five-year model time period. The next two sections use the model to describe the functioning of the system. First, simulated releases of a non-reactive dye are used to determine the residence times of waters entering the lake, and the circulation and mixing of waters in various regions of the lake. A loading analysis is also presented that quantitatively compares the sources of water and inorganic and organic forms of phosphorus and nitrogen to the lake. Scenarios that look at the short and long-term impacts of various levels of reduction in watershed nutrient loading are then presented. Other scenarios consider the consequences of other changes to the system such as the removal of causeways that restrict circulation within the New Hope Creek arm of the lake. A discussion and conclusions section ends the report.

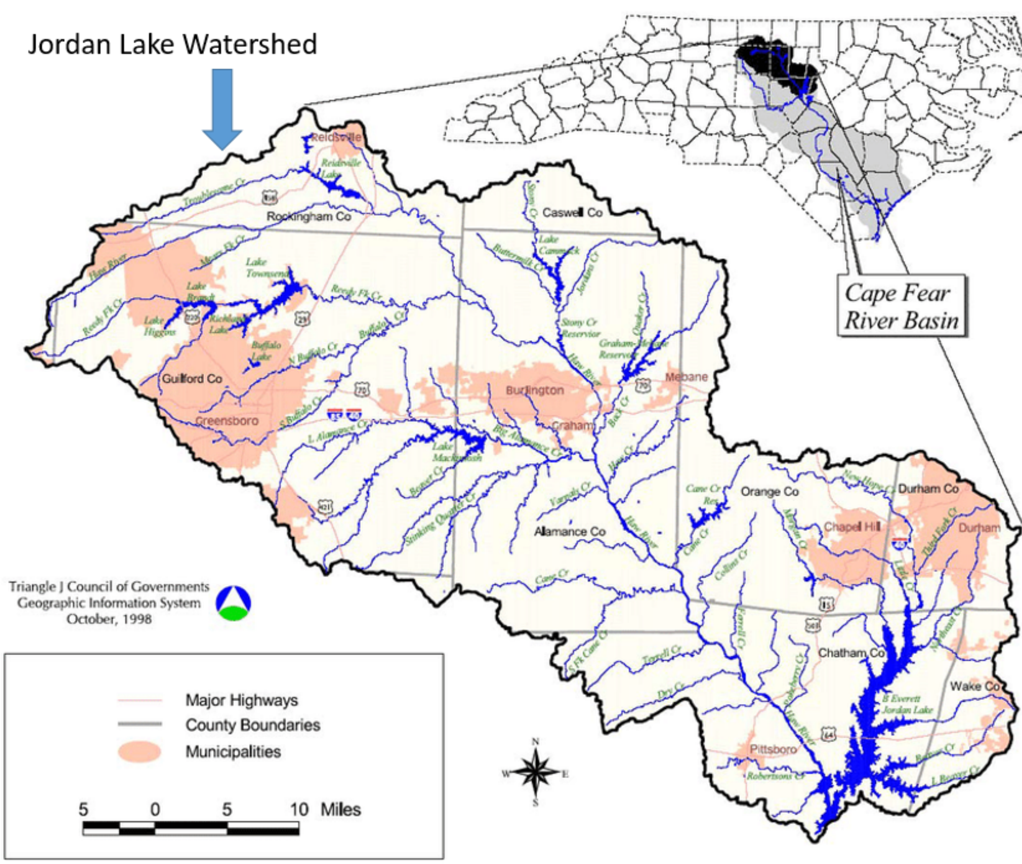


Figure 1. Jordan Lake watershed within the Cape Fear River basin of the piedmont region of North Carolina.

Model Description

The Environmental Fluid Dynamics Code (EFDC) model was used to model simulate the hydrodynamics and water quality of the lake. EFDC (Hamrick, 1992) is a general-purpose surface water modeling package for simulating three-dimensional (3-D) water circulation, mass transport, sediments and biogeochemical processes in surface waters. The graphical user interface EFDC Explorer 8.4 (Craig, 2018) was used for pre- and post-processing of data.

EFDC solves numerically the three-dimensional, vertically hydrostatic, free surface, Reynold's averaged momentum equations for a variable-density fluid (Hamrick, 1992). Turbulent kinetic energy, turbulent length scale, salinity and temperature transport equations are also solved. Wetting and drying of shallow areas is simulated using a mass conservation scheme (Hamrick, 1992).

A 16-state variable (Table 1) version EFDC water quality model was used for this study (Tetra Tech, 2007). Five variables found in the full 21-state variable model were not included for this study. (chemical oxygen demand (COD), total available metal (TAM), total suspended solids TSS), and bioavailable (SA) and non-bioavailable silicate (SU)). The state variables included

Table 1. EFDC Water Quality State Variables(Tetra Tech, 2007). Abbreviations refer to constituents as shown in Figure 2.

No.	Water Quality State Variable	Abbreviation	Unit
1	Cyanobacteria (blue-green algae)	Bc	g/m^3
2	Diatoms (algae)	Bd	g/m^3
3	Green algae (others)	Bg	g/m^3
4	Refractory particulate organic carbon	RPOC	g/m^3
5	Labile particulate organic carbon	LPOC	g/m^3
6	Dissolved organic carbon	DOC	g/m^3
7	Refractory particulate organic phosphorus	RPOP	g/m^3
8	Labile particulate organic phosphorous	LPOP	g/m^3
9	Dissolved organic Phosphorous	DOP	g/m^3
10	Total phosphate	TPO_4	g/m^3
11	Refractory particulate organic nitrogen	RPON	g/m^3
12	Labile particulate organic nitrogen	LPON	g/m^3
13	Dissolved organic nitrogen	DON	g/m^3
14	Ammonium	NH_4	g/m^3
15	Nitrate nitrogen	NO_3^-	g/m^3
16	Dissolved Oxygen	DO	g/m^3

were able to simulate the algal dynamics using three state variables (cyanobacteria, diatoms, green algae), nutrient dynamics using three inorganic (total phosphate, nitrate nitrogen, ammonium) and six organic state variables (refractory and labile particulate nitrogen and phosphorus, dissolved organic nitrogen), carbon cycling between algal and detrital fractions using three additional state variables (refractory and labile particulate carbon, dissolved organic carbon), and dissolved oxygen dynamics using one additional state variable. EFDC simulated the spatially and temporally varying mass balance of each of these state variables and the exchange of mass between the state variables to simulate processes in the water column such as nutrient uptake via photosynthesis, nutrient release via respiration and predation, and nutrient recycling between organic and inorganic forms (Figure 2).

Temporal and spatial variations in additional state variables (e.g. temperature, x-, y-, and z-direction velocity were simulated with the water-column hydrodynamic model. A predictive sediment diagenesis sub-model was also used to simulate the time-varying exchange of particulate organic matter settling from the water column and the benthic fluxes of inorganic nutrients and dissolved oxygen between the benthos and the water column (Craig, 2018; DiToro, 2001)

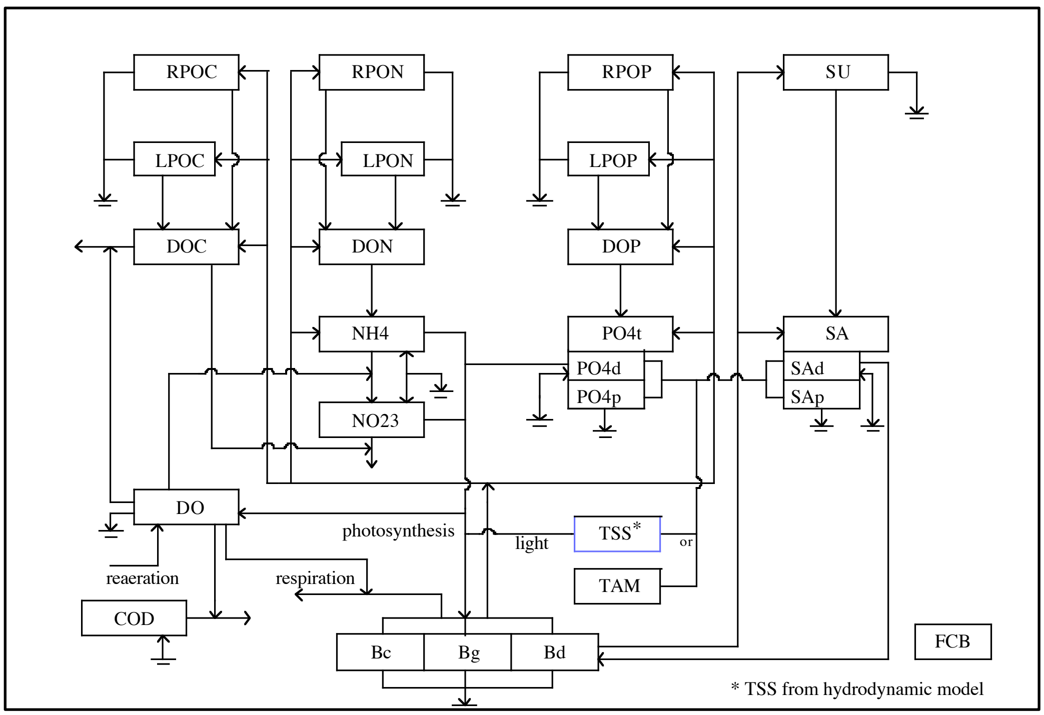


Figure 2. Box and arrow diagram showing the EFDC water quality state variables and the mass flows between them. See table 2 for the constituent names and abbreviations.

A version of the EFDC code was developed by Dynamic Solutions International, LLC (DSILLC) that simplifies the modeling process and provides links to a pre-processing and post-processing software called EFDC Explorer. Model setup, data input, and post-processing of model results can be performed with the EFDC Explorer graphical user interface . The model runs and post-processing of model results can also be done with programs such as MATLAB and Python. This project utilized the DSILLC version of the EFDC code. The pre-processing and post-processing was done with EFDC Explorer 8.4 and MATLAB scripts developed by the project team.

Several recent reports of modeling lake hydrodynamics and water quality were reviewed in preparation for developing the Jordan Lake model (Table 2). These reports provided technical support for the selection of the numerous model parameters, many of which are not explicitly

Table 2. Recent Water Quality Modeling Reports Used as a Basis for this Study

Report and Reference	Subject	Specific areas considered
Tenkiller Ferry Lake EFDC Water Quality Model (Michael Baker, 2015)	EFDC model of hydrodynamics and water quality	Water balance calibration, water quality model parameters, sediment diagenesis model setup
3-D Hydrodynamic and Water Quality Model of Lake Thunderbird, Oklahoma (Dynamic Solutions, 2013)	EFDC model of hydrodynamics and water quality	Water balance calibration, water quality model parameters, sediment diagenesis model setup
3-D Modeling of Hydrodynamics and Transport in Narragansett Bay (Abdelrhman, 2015)	EFDC model of hydrodynamics and water quality	Water balance calibration, water quality model parameters, sediment diagenesis model setup
Integration of a benthic sediment diagenesis module into the 2D hydrodynamic and water quality model – CE-QUAL-W2 (Zhang et al., 2015)	sediment diagenesis model integration into CE-QUAL-W2	Sediment diagenesis model theory and results for transient (seasonal) periods
Falls Lake Nutrient Response Model (Falls Lake Technical Advisory Committee, 2009; Lin and Li, 2011)	EFDC model of hydrodynamics and water quality	Consistency with local/regional data inputs and results
High Rock Lake Hydrodynamic and Nutrient Response Models (Tetra Tech, 2016)	EFDC model of hydrodynamics and water quality	Consistency with local/regional data inputs and results
Jordan Lake Nutrient Response Model (Tetra Tech, 2002; 2003)	EFDC model of hydrodynamics and water quality	Consistency with local/regional data inputs and results
Puget Sound Dissolved Oxygen Modeling Study: Development of an Intermediate Scale Water Quality Model (Khangaonkar et al., 2012)	FVCOM/ E-QUAL-ICM model of hydrodynamics and water quality	Water balance calibration and water quality model parameter selection, calibration
Total Maximum Daily Load Evaluation for Lake Lanier in the Chattahoochee River Basin for Chlorophyll a (Georgia Department of Natural Resources-Environmental Protection Division, 2017)	EFDC model of hydrodynamics and water quality	Water balance calibration, water quality model parameters, sediment diagenesis model setup

identifiable for Lake Jordan. They also provided guidance for conducting hydrodynamic and water quality model calibrations and selecting appropriate calibration targets. Local (in-state) model reports were also reviewed for consistency with previously reported input data and model results.

System Description

Jordan Lake is a physically unique lake with distinct characteristics. Some unique features include the sharp variations in depths across the lake area, a deep and narrow section along the Haw River Arm, and a shallow and broad section along the New Hope Arm. The Haw River contributes the most flow into the lake, accounting for about 70 to 90 percent of the total annual flow (NC DWQ, 2007). At normal operating conditions (216 feet MSL), Jordan Lake has an area of 13,940 acres. Another significant characteristic is the existence of a large area that alternates between wet and dry depending on the water level in the lake (Tetra Tech, 2002). As water level increases due to high inflows and precipitation, there is a significant increase in the wet area of the lake in comparison to the normal pool level. Additionally, the influence of causeways and natural constrictions restrict flow between sections of the lake. The influence of constrictions and causeways across the lake caused by the Mount Carmel Church Road, U.S. 64 Highway, NC 751 Road and Farrington Road are included in this project using EFDC's masking feature that simulates thin flow barriers between adjoining model cells (Craig, 2018; Tetra Tech, 2007).

An extensive water quality monitoring dataset was available to support the model. Water quality data were available at eighteen stations (Table 3) across the lake. The data were collected by the

Table 3. Monitoring Stations Used to Calibrate the Jordan Lake Model.

No.	Description	Station	Latitude in degrees	Longitude in degrees
1	Jordan Lake Dam	Dam	35.6548	-79.0672
2	Jordan Lake above Stinking Creek Near Pittsboro, NC	CPF055C	35.6913	-79.0791
3	Jordan Lake in Haw River Bay Arm Upstream	CPF055C1	35.6988	-79.0820
4	Jordan Lake in Haw River Bay Arm NE	CPF055C2	35.6955	-79.0761
5	Jordan Lake in Haw River Bay Arm NW	CPF055C3	35.6932	-79.0830
6	Jordan Lake in Haw River Bay Arm SE	CPF055C4	35.6899	-79.0756
7	Jordan Lake in Haw River Bay Arm SW	CPF055C5	35.6867	-79.0841
8	Jordan Lake in Haw River Arm Bay Downstream	CPF055C6	35.6822	-79.0780
9	Jordan Lake in Middle of Haw River Arm	CPF055D	35.6725	-79.0772
10	Jordan Lake above Dam Near Moncure, NC	CPF055E	35.6600	-79.0700
11	Jordan Lake Downstream Crooked Creek, New Hope Arm	CPF081A1B	35.8365	-78.9763
12	Jordan Lake @ Mouth of New Hope Creek	CPF081A1C	35.8162	-78.9868
13	Jordan Lake @ Mouth of Morgan Creek Near Farrington	CPF086C	35.8215	-78.9974
14	Jordan Lake In Upstream	CPF086CUPS	35.8382	-79.0014
15	Jordan Lake, Downstream Morgan, New Hope Creek Arm	CPF086D	35.8095	-78.9974
16	Jordan Lake Near Farrington, NC	CPF086F	35.7970	-79.0108
17	Jordan Lake at Buoy #9 Near Merry Oaks, NC	CPF087B3	35.7652	-79.0260
18	Jordan Lake @ Mouth White Oak Creek Near Seaforth, NC	CPF087D	35.7386	-79.0242
19	Jordan Lake Near Mouth Beaver Creek Near Merry Oaks, NC	CPF0880A	35.6965	-79.0436

NC Division of Water Resources and were made available to this study as a Microsoft Access database. Water quality parameters from the database that were used for this study included temperature profiles, and grab samples analyzed for nitrate, ammonia, total phosphorus, Kjeldahl nitrogen, chlorophyll a, and dissolved oxygen. Stations were present in all regions of the lake (Figure 3).

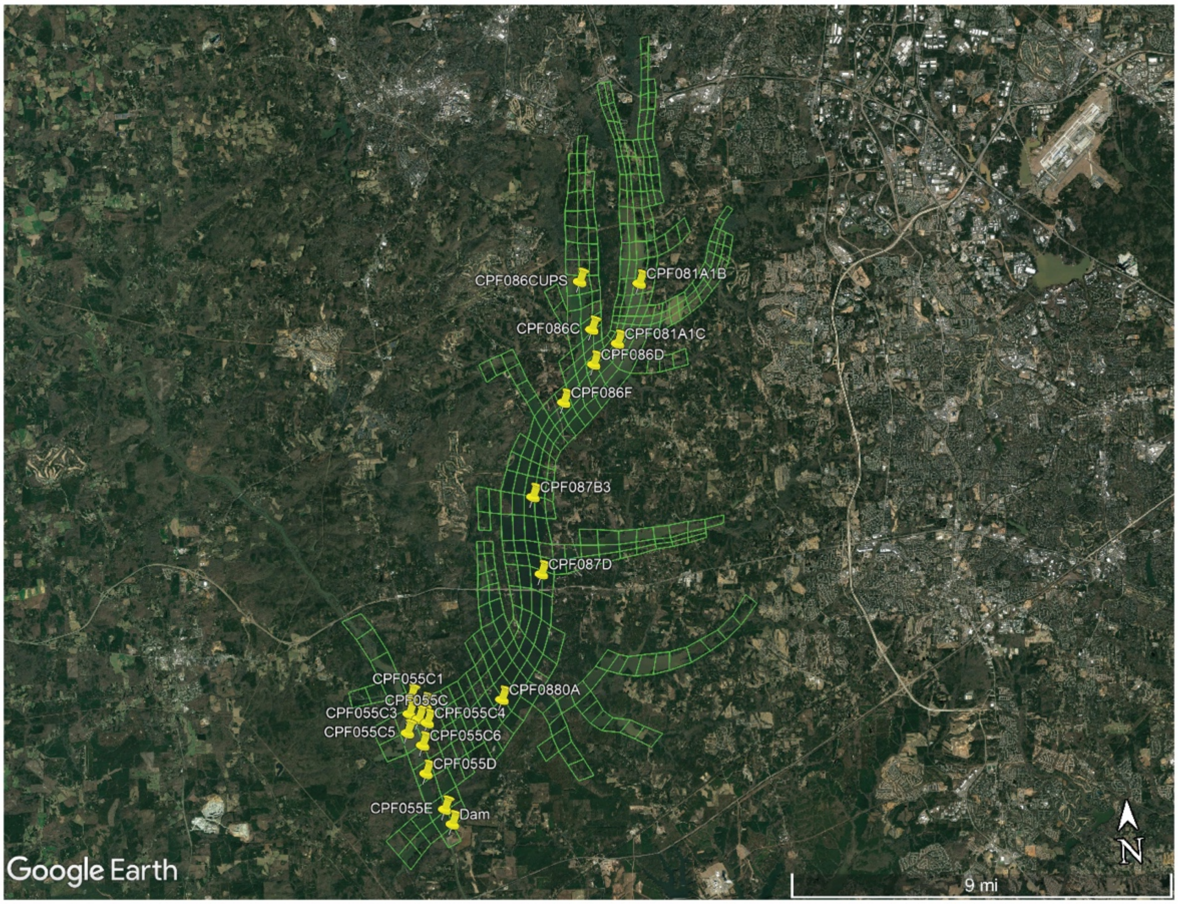


Figure 3. Locations of 18 Jordan Lake Monitoring Stations Sampled by the NC Division of Water Resources. The newly developed 407 cell EFDC model grid is also shown.

In the Jordan Lake nutrient response model, chlorophyll a data are used as a measure of the cumulative abundance of the three state variables (cyanobacteria, diatoms, and green algae) collectively representing the phytoplankton biomass. The spatial and temporal dynamics in the data are used to calibrate the algal growth kinetic parameters in the model. North Carolina also uses chlorophyll a as a numeric water quality criteria (NC Division of Water Resources, 2017). The current approved regulatory text for the State's chlorophyll a criteria, located at 15A NCAC 02B .0211(4), states:

> *Chlorophyll-a (corrected):* ***not greater than 40 ug/l for lakes, reservoirs, and other waters subject to growths of macroscopic or microscopic vegetation not designated as trout waters****, and not greater than 15 ug/l for lakes, reservoirs, and other waters subject*

to growths of macroscopic or microscopic vegetation designated as trout waters (not applicable to lakes or reservoirs less than 10 acres in surface area).

A waterbody is considered impaired if there is a 90% confidence that more than 10% of the photic zone average chlorophyll measurements area above the regulatory limit, in this case 40 µg/L (NCDWR, 2018).

Based upon a review of the chlorophyll a monitoring data collected over the five-year (2014-2018) model time period, every one of the eighteen monitoring stations exceeded the 40 µg/L more than 10% of the time. The 90th percentile (the value exceeded exactly 10% of the time) photic chlorophyll a concentration for all eighteen stations considered collectively (1004 measurements total) for the 2014-2018 time period was 72.0 µg/L (Table 4). A reduction of 44% would be needed to lower the 90th percentile chlorophyll a concentration to the regulatory limit of 40 µg/L.

Frequently sampled stations in each of the four regions of the lake also had 90th percentile chlorophyll a concentrations above the criteria value, but the magnitude of the exceedances varied significantly from region to region. The one station in the below causeways region of the New Hope Creek arm of the lake exceeded the 90th percentile by only 5% (74 samples total). In the above causeways region of the New Hope Creek arm of the lake, all three stations exceeded the 40 µg/L by more than 50% (Table 4). The other two regions of the lake (Haw River, between causeways had exceedance levels between these two extremes.

Table 4. Analysis of 2014 – 2018 Photic Zone Chl a Measurements at selected stations within four regions of Jordan Lake.

Lake Region	Station	Number of Chl a samples	Chl a median concentration (µg/L)	90th percentile Chla a concentration (µg/L)	Reduction needed for 90th percentile Chl a concentration at 40 µg/L
Haw River	CPF055C	74	29.0	63.7	37%
	CPF055D	72	25.0	44.9	11%
	CPF055E	73	28.0	44.0	9%
Above Causeways	CPF081A1C	74	57.5	90.4	56%
	CPF086C	74	58.5	89.0	55%
	CPF086F	74	52.5	81.7	51%
Between Causeways	CPF087B3	74	34.0	52.4	24%
	CPF087D	74	29.5	53.0	25%
Below Causeways	CPF0880A	74	28.0	42.0	5%
Jordan Lake	All 18 Stations	1004	36.0	72.0	44%

Model Setup and Calibration

The first step in the model setup is the definition of the model grid. The grid should provide a good approximation of the actual physical dimensions (morphometry) of the water body. EFDC is set up to use a curvilinear-orthogonal grid in the horizontal plane that is stretched to provide an approximate representation of the curvature of the actual water body. Vertical structure is represented by specifying a fixed or varying number of vertical subdivisions for each horizontal grid cell. CVLGrid, which is a grid generating preprocessor program alongside EFDC Explorer modeling package and Google Earth are used to construct the horizontal model grid. Bathymetric data obtained from sonar sampling in the lake by Collaboratory partners and LIDAR data obtained from North Carolina Flood Risk Information System (FRIS) were used to update the bottom elevation in the grid cells (Figure 4). Horizontal projection for the XY data used to define shoreline and grid coordinates is UTM Zone 17 as meters. The Jordan Lake model grid contains

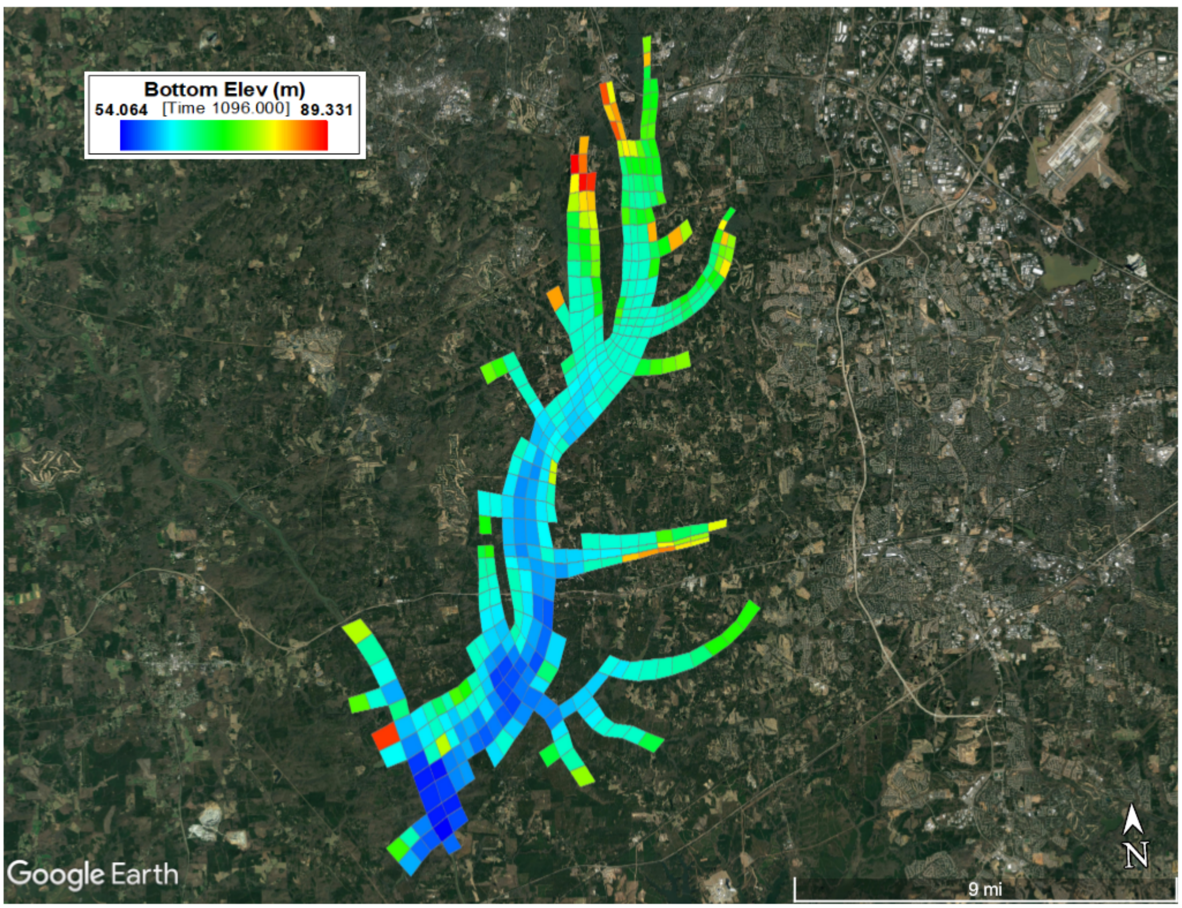


Figure 4. Color contours of bottom elevations (m) for the 407 cells in the Jordan Lake model grid.

407 horizontal grid cells, with cell sizes varying from 178 m to 1105 m. Depth of the water column was represented with vertical layers using the SGZ vertical layering option, with the model grid having seven minimum active vertical layers and 25 maximum vertical layers to

account for the effects of seasonal stratification. The SGZ (also known as a z-grid) vertical layering option was developed by DSI to deal with pressure gradient errors that occur in models that have steep changes in bed elevation (Craig, 2018). The developed model grid was validated using the Volume Elevation relationship reported by US Army Corps of Engineers.

Model input data sets and observed data sets used for calibration and validation, and model scenario testing were developed using observed data gathered from various agencies (Table 5). Time series data including flow and pollutant loading from the drainage areas, withdrawals from water supply intakes and releases at the dam, meteorological and wind forcing data, and atmospheric and benthic deposition of nutrients were obtained for the model period 2014 to 2018. Flow input data were obtained from US Geological Survey (USGS) gages and US Army Corps of Engineers, meteorological and wind forcing data were obtained from North Carolina

Table 5. EFDC Input Files and Data Sources

EFDC Input Filename	Description of Data Contained in File	Data Sources
QSER.INP	Flow time series data at flow specified model boundaries and point source locations	US Geological Survey (Haw River, creeks), US Army Corps of Engineers (water treatment plant, dam outflows)
ASER.INP	Meteorological time series data (air temp, dewpoint temp, relative humidity, short-wave solar radiation, precipitation, cloud cover)	North Carolina State Climate Office (NCSCO) and National Oceanic and Atmospheric Administration (NOAA)
TSER.INP	Temperature time series data at all model boundaries and point source inputs	North Carolina Department of Environmental Quality (NC DEQ) and US Geological Survey
WSER.INP	Wind time series data for magnitude and direction	North Carolina State Climate Office (NCSCO) and National Oceanic and Atmospheric Administration (NOAA)
DXDY.INP	Horizontal cell lengths, widths, depths, bottom roughness	Collaboratory Partners (Bathymetry), North Carolina Flood Risk Information System (FRIS) and US Army Corps of Engineers (Lake elevation at the start of each model phase)
LXLY.INP	Horizontal cell size location, orientation relative to E-W, N-S direction	Google Earth (UTM Zone 17) and CVLGrid (Craig, 2018)
WQBENMAP.INP, BENFLUX.INP	Map of benthic nutrient and DO flux zones, specification of NO3, NH4, PO4, and DO flux time histories by zone (flux specified runs only)	Calibrated values w/ information from Collaboratory Partners (NO3 and DO benthic flux measurements), and recent lake model studies (Table 2)
TEMP.INP	Initial condition for temperature for every model cell and layer	created with a model spinup run, using the EFDC restart option
CWQSRXX.INP (XX indicates constit. number)	Time series concentration boundary condition at flow specified boundaries	Collaboratory partners (TN and TP loading), DWR data on N speciation, MATLAB script used to create files as described in Cape Fear Model Report (Bowen et al., 2009)

State Climate Office (NCSCO) and National Oceanic and Atmospheric Administration (NOAA). Atmospheric deposition of nutrients were obtained from National Atmospheric Deposition Program (NADP) and Clean Air Status and Trends Network (CASTNET) for nitrogen with phosphorus estimated from annual average N/P ratios for atmospheric deposition of N and P (Willey and Kiefer, 1993). Benthic deposition of nutrients for the flux specified simulations were obtained using limited sampling data from Collaboratory partners and model results for sediment fluxes from other lake models that included sediment diagenesis (Abdelrhman, 2015; Dynamic Solutions, 2013; Michael Baker, 2015). Pollutant concentration at flow specified boundaries were obtained by nutrient fractionation estimates based upon total nitrogen and total phosphorus data (Collaboratory partners), and NC DWR data on nitrogen speciation. A transformation matrix approach used in a previous modeling study (Bowen et al., 2009) was used to estimate the time series pollutant concentrations.

Model calibration was divided into two phases: hydrodynamic and water-quality calibration. In both cases, model predictions were compared to observed data collected at 18 NC DWR monitoring stations (see Table 3 for list of station, Figure 3 for a map of station).

Observations of water surface elevations were available at the dam and were compared to model predictions over each model time period. These observed data were collected by the Army Corps of Engineers, who also provided outflow data at the dam and the Cary Water Treatment Plant. Over both the two time periods used for model calibration (2014-2015 and 2017-2018) and the one time period used for model validation (2016) the mean error of water surface elevations is low and does not suggest a trend (positive or negative) in the error with respect to the time period of the model (Table 6). The normalized mean error is also low, and a minor negative or positive trends in the error with respect to the time period are indicated. The root mean square error, which does not reveal positive or negative trends in the error, is low for all three model periods. The normalized root mean square error is also low for all three model

Table 6. Statistical comparison of modeled vs observed water surface elevations.

Parameter	Model Time Period			Units
	2014-2015	2016	2017-2018	
Mean Error (predicted – observed)	-0.09	0.08	-0.05	m
Normalized Mean Error	-0.10%	0.10%	-0.10%	%
Root Mean Square Error	0.38	0.26	0.44	m
Normalized Root Mean Square Error	0.60%	0.40%	0.70%	%
Correlation R^2	85%	95%	91%	%
Number of Model/Data Comparisons	779	365	717	-
Model Efficiency	82%	93%	88%	%

periods. The correlation coefficients indicate that measured and modeled water surface elevations are highly correlated for all model periods. Consideration of water surface elevation time histories (Figure 5) is essential to understanding the goodness of fit between measured and modeled water levels. The largest errors are associated with time periods when the lake water level apparently has been drawn down and when peak water level events occur due to high inflows. Accurate modeling of water level drawdowns is problematic as these periods were not directly related to gaged and estimated (ungaged) inflows, outflow at the dam, and water supply withdrawals used in the model. Accurate modeling of peak water level events is also problematic as they are very sensitive to the timing of inflows and outflows that are applied as daily averages in the model.

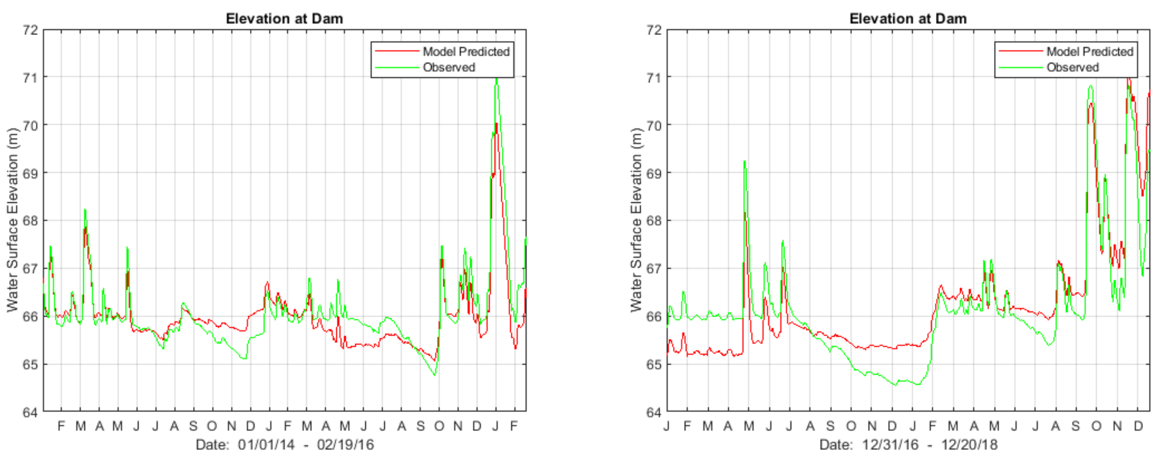


Figure 5. Time history comparisons of observed and model predicted water surface elevations at the Jordan Lake dam for the 2014-2015 model time period (left panel) and the 2017-2018 time period (right panel).

Temperature profile data from the eighteen DWR monitoring stations were compared to corresponding model predictions for each model time period. All statistical measures of calibration performance for temperature calibration (Table 7) such as mean error, root mean square error, and goodness of fit measures (correlation R^2 and model efficiency) indicate the agreement between measured and modeled water temperatures is good to very good. The time history comparisons for the surface and bottom water layers (Figure 6) are also in good agreement with the measured temperatures at a representative location (station CPF055E) for the two periods used for model calibration. The model results are consistent with observed water temperature for both well-mixed winter conditions and summer stratified conditions. Scatter plots of observed vs. model predicted temperature (Figure 7) indicate the limited scatter and low bias across the full range of predicted and observed temperatures.

Table 7. Statistical comparison of modeled vs. observed water temperatures

Parameter	Time Period			Units
	2014-2015	2016	2017-2018	
Mean Error (predicted – observed)	0.17	-0.3	-0.8	°C
Normalized Mean Error	1%	-2%	-4%	%
Root Mean Square Error	2.11	1.56	2.07	°C
Normalized Root Mean Square Error	11%	9%	11%	%
Correlation R^2	94%	97%	96%	%
Number of Model/Data Comparisons	1144	370	398	-
Model Efficiency	94%	97%	93%	%

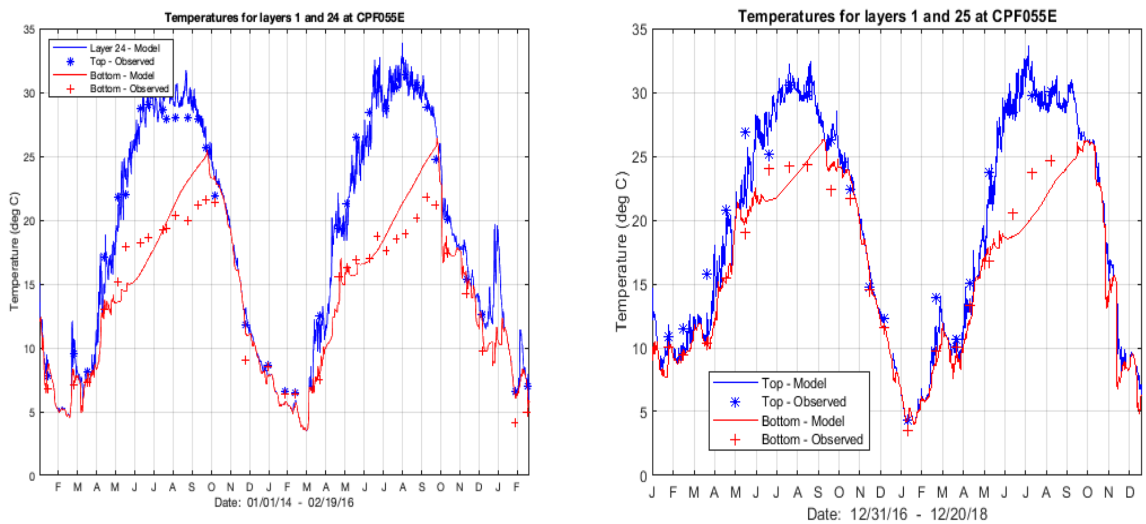
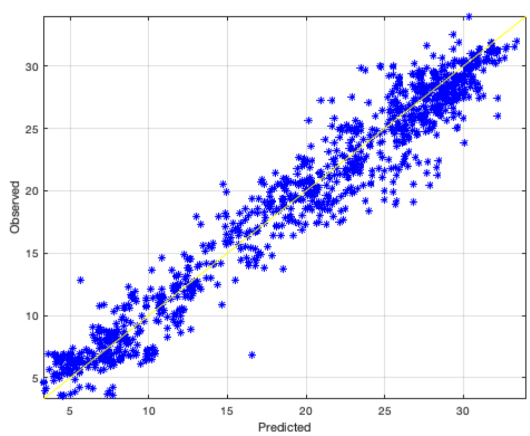


Figure 6. Time history comparisons of observed and model predicted surface and bottom temperatures at station CPF055E for the 2014-2015 model time period (left panel) and the 2017-2018 time period (right panel).

Since the numeric criteria for chlorophyll a examines the magnitude of infrequently occurring high chlorophyll a values (i.e. what is the chlorophyll a value that is exceeded exactly 10% of the time), it is important the distribution of model predicted chlorophyll a concentrations closely match that of the observed values. Said another way, it is more important the model accurately predicts the magnitude and frequency of high chlorophyll values than it accurately predicts exactly where and when those high values occur. For this reason, the primary objective of the chlorophyll a calibration was to match well the frequency distribution of observed values. Other quantitative calibration measures such as mean error, root mean square error and correlation R^2 were considered as well, but were given a lower importance during calibration.

During calibration, cumulative distribution functions (CDFs) of model predicted and observed chlorophyll a data were examined for approximately 200 separate model runs for both the 2014-2015 and 2017-2018 model time periods to select kinetic parameters related to algal growth and nutrient dynamics. Kinetic parameters that were considered during calibration include maximum algal growth rates for each algal group, temperature, nutrient, and light dependence of algal growth, carbon to chlorophyll ratio for algal organic matter, algal respiration and predation rates, benthic nutrient fluxes of nitrate, phosphate, and ammonia, background light attenuation, and chlorophyll and organic matter specific light attenuation. The CDFs were based on concurrent measured and modeled chlorophyll a concentrations for all eighteen monitoring stations taken over the two model time periods. The CDFs for the calibrated model matched well that of the observed values for both model time periods (Figure 8). The 90th percentile value was slightly overpredicted for both time periods by about 20%. Median values (50th percentile) matched



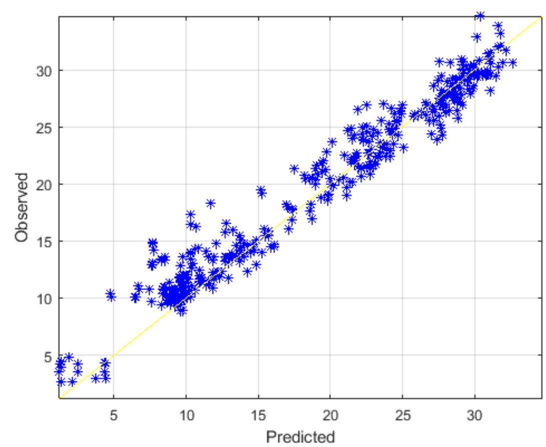


Figure 7. Scatter plot of observed and model predicted temperatures at all monitoring stations in Jordan Lake, NC for the 2014-2015 (left) and 2017-2018 (right) model time periods.

more closely the observed chlorophyll a concentration, while the model underpredicts the lower end of the chlorophyll a distributions (Figure 8). Time history comparisons of one representative station in the New Hope arm of the lake (station CPF087D) and one station in the Haw River arm of the lake (CPF055D) shows the large magnitude of seasonal variation in the chlorophyll a concentrations (Figure 9) observed in the lake during the 2014-2015 model time period. The 2017-2018 temporal patterns in chlorophyll a concentrations at these stations were qualitatively similar. The model does a good job overall in predicting the chlorophyll magnitudes but misses in some cases the timing of the minimum and maximum values. Model predicted chlorophyll a concentrations can vary widely at a station over a short time period, particularly in the Haw

River arm of the lake. This is likely due to variations in inflow that move the peak chlorophyll a concentrations upstream and downstream.

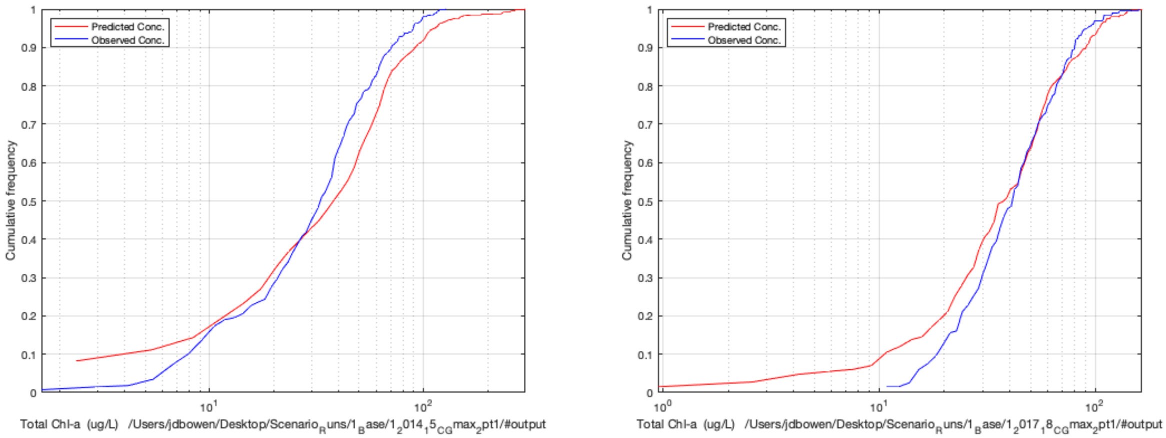


Figure 8. Cumulative distribution functions of observed and model predicted chlorophyll a concentration at all 18 monitoring stations in Jordan Lake, NC for the 2014-2015 (left) and 2017-2018 (right) model time periods.

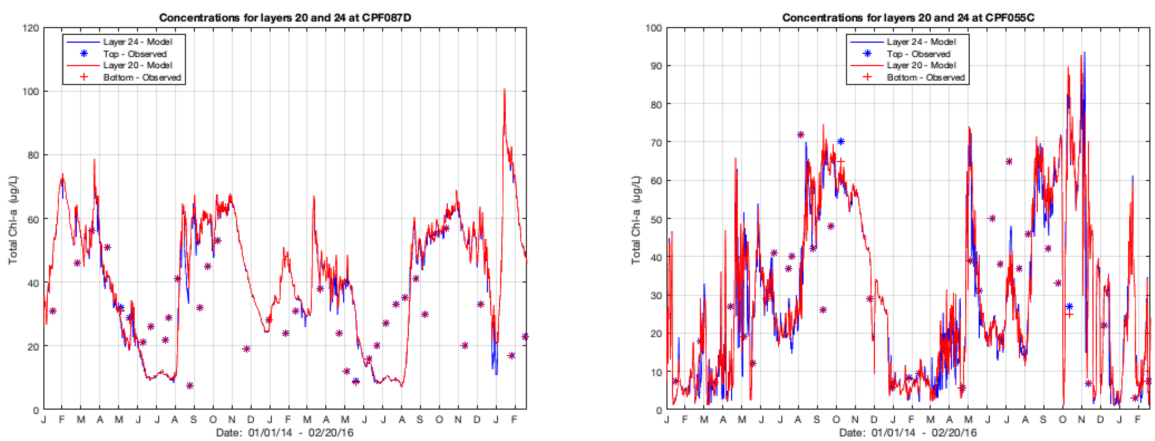


Figure 9. Time history comparison of observed and model predicted chlorophyll a concentrations at monitoring stations CPF087D and CPF055C for the 2014-2015 model time period.

The statistics for chlorophyll a modeling calibration in Table 8 (mean error, normalized mean error, normalized mean error, correlation coefficient) should be considered along with the time histories in Figure 9 in order to understand the magnitude of some of the statistical errors which may seem high, but are also a reflection of the short-term transient nature of the measured and modeled concentrations. The validation time period (2016) had similar calibration performance to the two time periods used for model calibration. The level of calibration performance achieved is typical of models of algal growth in highly dynamic, long residence time systems. The model was considered to be sufficiently calibrated to the observed chlorophyll a for the purposes of investigating the effects of nutrient load reductions on chlorophyll a concentrations.

Table 8. Statistical comparison of modeled vs. observed chlorophyll a concentration

Parameter	Time Period			Units
	2014-2015	2016	2017-2018	
Mean Error (predicted – observed)	0.06	0.10	-0.03	log μg/L
Normalized Mean Error	4.1%	6.8%	-1.9%	%
Root Mean Square Error	0.41	0.40	0.27	log μg/L
Normalized Root Mean Square Error	29%	28%	17%	%
Correlation R^2	27%	26%	28%	%
Number of Model/Data Comparisons	898	366	334	-

The statistics for total phosphorus and nitrate nitrogen for the calibrated water quality are also presented as cumulative density functions (CDFs) for the 2014-2015 model time period (Figure 10). Total phosphorus was used for model vs. data comparison since no orthophosphate data were available. For calibrating algal growth models, orthophosphate is preferred because of its bioavailability to phytoplankton. Total phosphorus is a cumulative measure of inorganic, organic, and particle associate forms of phosphorus, and is therefore less desirable for model calibration. Like the chlorophyll a concentration, the CDFs were based on all concurrent measured and modeled total P and nitrate concentrations. The CDFs are indirect measures of both the magnitude and timing of the calibrated models for these nutrients. They were considered to be acceptable for both periods. The modeled trends for both nutrients are adequate. Total P is underpredicted at the lower range concentration and overpredicted at the higher range, while nitrate N was underpredicted throughout the range of data, but was not grossly overestimated. The 2017-2018 model calibration run produced total P and nitrate CDFs that were qualitatively similar to that shown for the 2014-2015 model time period.

Analysis of Simulated Dye Releases

The water age, or residence time, of water in Jordan Lake was calculated based upon the simulation of a dye study release using the hydrodynamic model. Qi et al. (2016) describes this approach in which a conservative tracer is discharged to the lake. Two advection-dispersion equations for concentration are derived: one for the tracer and another for water age. The residence time at any specific location and time is defined as the ratio of water age concentration to tracer concentration. The residence time distribution at any given model time represents the effects of flow boundary locations (inflow and outflow), structures within the lake (i.e.,

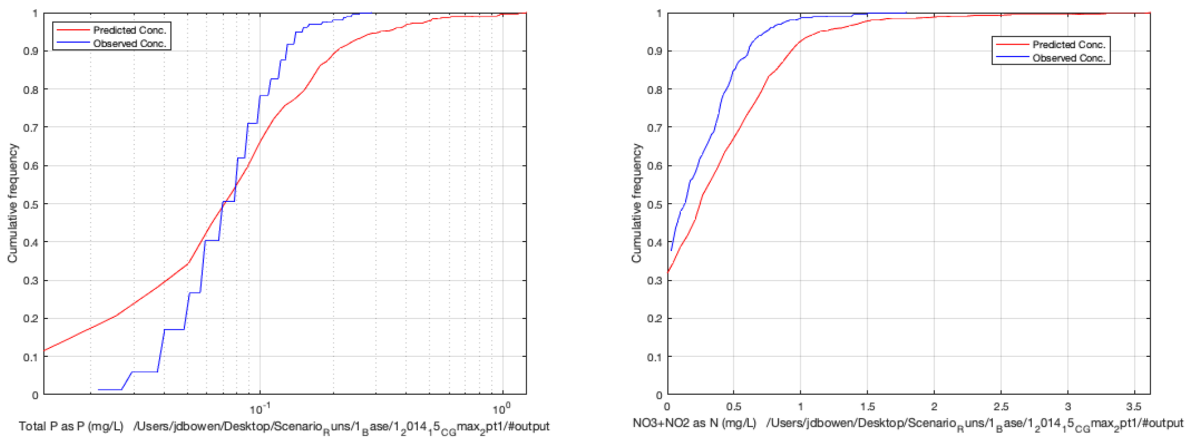


Figure 10. Cumulative frequency distributions of observed and model predicted total phosphorus (mg/L, left) and nitrate nitrogen (mg/L, right) concentrations at all 18 monitoring stations in Jordan Lake, NC for the 2014-2015 model time period.

causeways), and water quality zones, which are defined by local flow conditions (water depth and width) and water temperature. Residence times were found to vary significantly for different regions of Jordan Lake (Table 9). Residence time above the causeways was the shortest relative to other zones. This is a consequence of the average flow cross section, which is relatively narrow and shallow, and the proximity of inflows from Morgan Creek and New Hope Creek. This effect is most pronounced above Morgan Creek where the flow cross section is further reduced. Between the causeways, the available flow cross section is greater, and residence times were longer. Locally, this effect is reduced due to inflows from White Oak Creek and unnamed tributaries which discharge to this zone. Below the causeways, residence time is determined by the large flow cross section and inflow from the Haw River, which reduces residence time locally below its outfall to the lake. Residence time distributions for summer 2014 and winter 2015 were similar, while that for summer 2015 was longer throughout the lake. This implies that seasonal and event driven flow regimes may be partially offsetting, at least in this comparison.

Table 9. Estimates of Water Residence Time for Different Regions of Jordan Lake, NC for the 2014-2015 Model Time Period.

Jordan Lake Region	Residence time range (days)		
	Summer 2014	Winter 2015	Summer 2015
Haw River	<30	<20	40-80
Above causeways	< 80	< 90	< 100
Between causeways	80 - 150	80 - 150	100 - 170
Below causeways	20 - 200	20 - 220	20 - 200

A second simulated dye release was done to reveal the extent to which inflow from the Haw River mixes with waters in each of the four water quality regions of the lake (Table 10). Three of the regions are in New Hope Creek arm of the lake (above causeways, between causeways, below causeways), while the fourth region is the western arm of the lake along the

Table 10. Time-average simulated dye concentrations at locations across four Jordan Lake regions. Higher concentrations indicate a higher contribution from Haw River inflow. The average concentration for a region is based upon all stations within that region.

Jordan Lake Region	Station	Time-Average Contribution from Haw River Water (%)	
		2014-2015	2017-2018
Haw River	CPF055C	100%	100%
	CPF055D	100%	100%
	CPF055E	100%	100%
	Average	**93.5%**	**93.1%**
Above Causeways	CPF086C	0.0%	1.0%
	CPF086D	0.8%	2.8%
	CPF086F	1.0%	3.2%
	Average	**0.0%**	**1.2%**
Between Causeways	CPF087B3	12.0%	20.1%
	CPF087D	20.1%	30.0%
	Average	**16.0%**	**25.0%**
Below Causeways	CPF0880A	59.2%	70.4%
	Average	**59.2%**	**70.4%**

old Haw river bed. As expected, the contributions from the Haw decreased significantly moving uplake (upstream) from the Haw River region through each region in the New Hope Creek arm of the lake and eventually into to the Morgan Creek and New Hope arms of the lake (Table 10). At the same time, the results indicated that the Haw contribution was still potentially measurable throughout each region of the lake, but makes up only a very small fraction of the input to the upper and between causeways regions (< 2% and 25% Haw River water respectively). These results are also consistent with the residence time model described previously, which showed much longer residence times, and hence lower amounts of flushing in the New Hope Creek arm of the lake.

Nutrient Loading Analysis

A nutrient loading analysis based on the model's input files was used to quantify the relative nutrient load contributions from the external and internal sources of pollutants to the lake for the model period 2014 - 2018. The external sources include the surface water inputs, and wet and dry atmospheric deposition, while the internal sources include the benthic fluxes of inorganic nutrients across the sediment-water interface of the lake. A flow analysis was also used to quantify the inflow and outflow contributions to the lake for the model period as rates (m^3/day). The Haw River accounts for over 75% of the inflows while the dam outflow accounts for over 95% of the outflows for the model period (Figures 11 a & b). The nutrient loading analysis

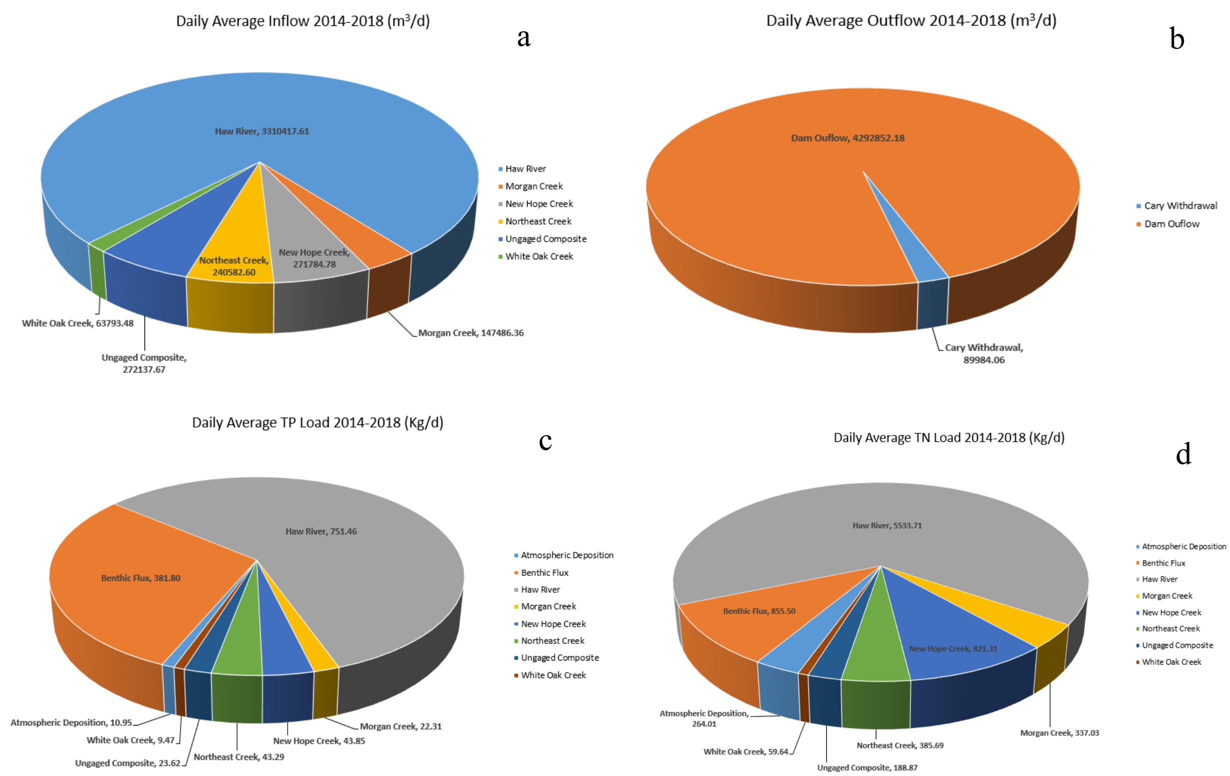


Figure 11. Pie charts showing 2014-2018 daily average inflows (a), daily average outflow (b), daily average total phosphorus (TP) load (c), and daily average total nitrogen (TN) load for Jordan Lake, NC.

indicated that majority of nutrients entered the lake from the Haw River arm, and these nutrients entered majorly in organic forms that were not immediately bioavailable. The Haw River arm accounted for over 65% of total nitrogen and 55% of total phosphorus loads into the lake (Figures 11 c & d). Benthic sediments were a significant source of bioavailable nutrients, providing more than 40% of phosphate and 85% of ammonia to the lake (Figures 12 a & b). Benthic sediments also acted as a major sink for the particulate fraction of organic nutrients, nitrate (Figure 12 c & d), and dissolved oxygen. Atmospheric deposition was a relatively minor source of nutrients to the lake, accounting for less than 5% of the total nitrogen. Through dye tracer studies, it was observed that only a small fraction of nutrient inputs from the Haw River arm moved up into the upper reaches of the New Hope Creek arm of the lake on a long-term basis. However, some high flow events did transport Haw River water throughout the lake, but

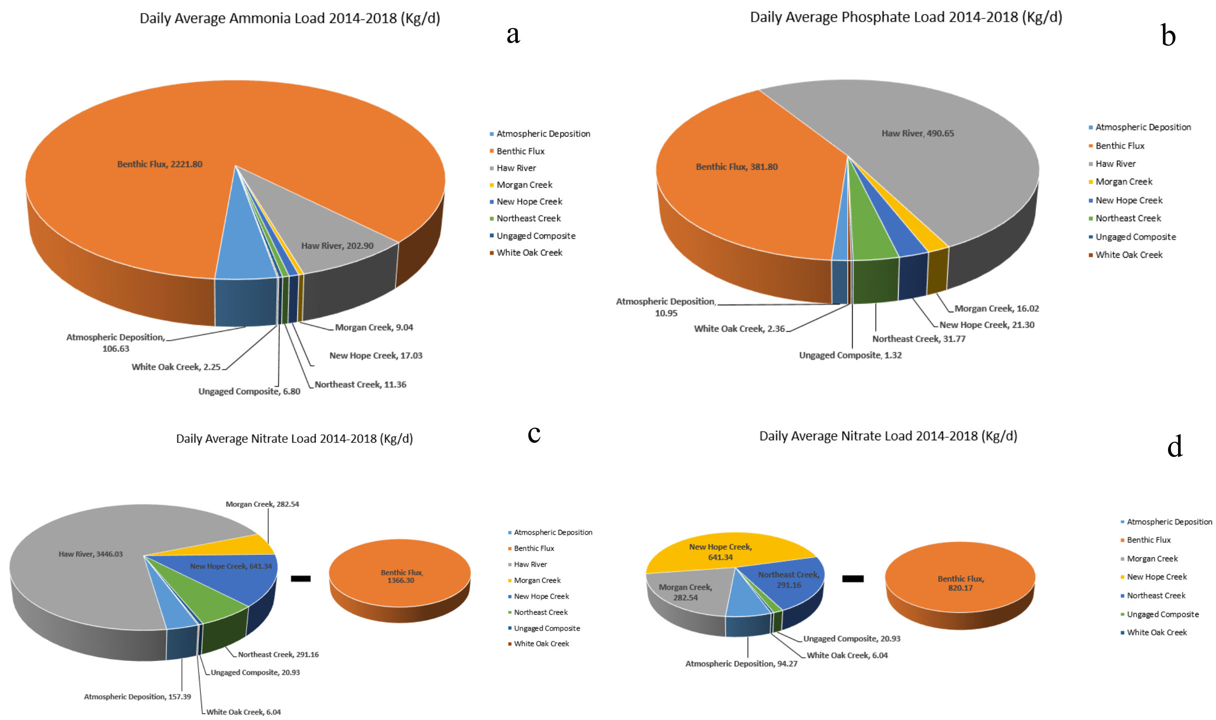


Figure 12. Pie charts showing 2014-2018 daily average ammonia load (a), daily average phosphate load (b), and the daily average nitrate load (c) for Jordan Lake NC. Panel d shows the daily average nitrate load for the New Hope Creek arm of the lake.

these high flow events did not contribute significantly to the flushing of the New Hope Creek arm or to the nutrient loading in these areas. The majority of water and nutrients to the New Hope Creek arm of the lake (Figure 13) are provided by the local surface water sources (Morgan, New Hope, Northeast, and other smaller creeks). For certain inorganic forms of nitrogen and phosphorus, benthic sediments were the major supply source to the water column in the New Hope arm of the lake, providing more than 75% of the phosphate and 90% of ammonia (Figures 13 a & b).

Model Scenario Testing

To test the sensitivity of the Jordan Lake system to nutrient load reduction, incoming loads of both organic and inorganic forms of nitrogen and phosphorus were reduced independently over a range of values from 10% to 50%. The load reductions were accomplished by reducing concentrations of the appropriate model state variables at the inflow boundaries by the given percentages. For the organic fractions, reductions were made in the concentrations of the labile particulate and dissolved nitrogen and phosphorus state variables, but not in the refractory particulate fraction. These relatively inert state variables were held constant because it was assumed that treatment operations would be less effective on the inert matter compared

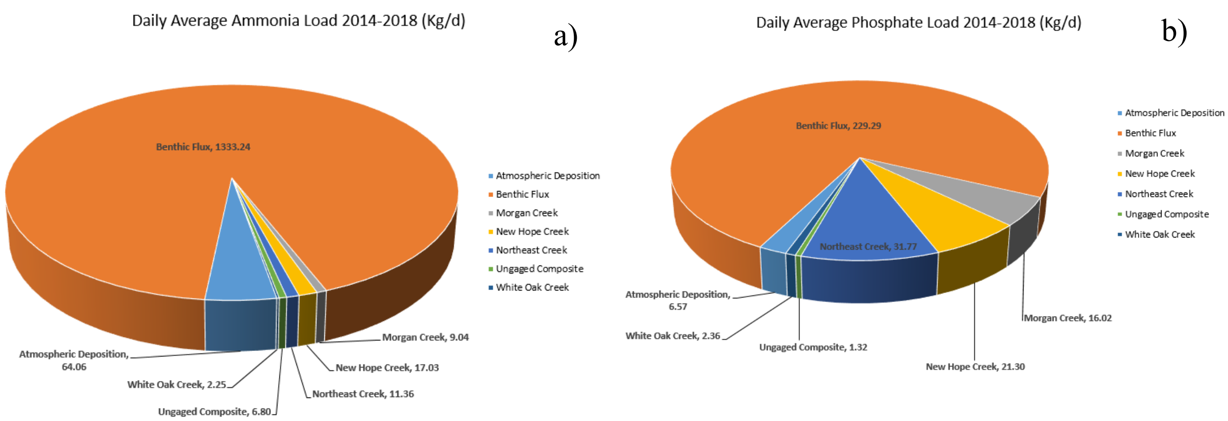


Figure 13. Pie charts showing the 2014-2018 daily average ammonia (a) and phosphate (b) load into the New Hope Creek arm of Jordan Lake, NC.

to the more reactive labile particulate and dissolve fractions. Wet and dry deposition of nutrients and benthic flux rates were held constant for the first set of nutrient load reductions (Table 11). A total of twenty-five load reduction cases were considered, with N and P reductions from zero to forty percent. For each load reduction case, an alternate set of the flow boundary concentration files (CWSRXX.INP files) were developed and run with the calibrated model for the 2014-2015 and 2017-2018 model time periods. The model results were analyzed to determine the reduction in the 25th, 50th, 75th, and 90th percentile chlorophyll a concentration for

Table 11. Average reduction in model predicted chlorophyll a concentration at 18 locations from the 2014-2015 and 2017-2018 base case runs for various nutrient load reduction scenarios (upper table) and the fraction of model predicted values exceeding the chlorophyll a water quality criteria value of 40 μg/L (lower table).

Change in N loading					**Chl a Reduction (μg/L)**	
-40%	**-30%**	**-20%**	**-10%**	**0%**		
-12%	-9%	-6%	-3%	0%	**0%**	**Change**
-13%	-10%	-7%	-3%	-1%	**-10%**	**in**
-15%	-12%	-9%	-5%	-3%	**-20%**	**P**
-16%	-13%	-10%	-7%	-5%	**-30%**	**Loading**
-17%	-14%	-12%	-10%	-7%	**-40%**	

Change in N loading					**% Chl a Above 40 μg/L**	
-40%	**-30%**	**-20%**	**-10%**	**0%**		
39%	41%	43%	44%	45%	**0%**	**Change**
38%	40%	42%	43%	44%	**-10%**	**in**
37%	39%	41%	43%	44%	**-20%**	**P**
36%	38%	40%	42%	43%	**-30%**	**Loading**
35%	37%	39%	41%	42%	**-40%**	

all eighteen monitoring stations as compared to the corresponding base case. For each case the fraction of chlorophyll a concentrations above the regulatory limit of 40 ug/L was also determined.

Both nitrogen and phosphorus load reductions reduced chlorophyll a concentrations, but to a different extent. Over the range of N and P reduction scenarios from 0% to 40%, chlorophyll a concentrations decreased by 0% to 17% (Table 11). In general nitrogen reductions of a particular percentage were more effective in reducing chlorophyll concentrations than the corresponding phosphorus load reduction. Since the chlorophyll reductions were modest in all cases (0% - 17%), not surprisingly, the frequency of exceedance of the 40 ug/L water quality criteria value were always significantly above 10% (Table 11). Even the case with largest decrease in loading (-40% N, -40% P) produced a chlorophyll a reduction (-17%) that was far from the 40-50% needed to bring the lake into compliance with the water quality criteria for chlorophyll a (see Table 4).

The earlier analysis of simulated dye releases showed that very little of the water that enters the lake via the Haw River makes it up into the upper part of the New Hope Creek arm of the lake. This characteristic of Jordan Lake suggests that reductions in nutrient loading from the Haw River arm are likely to be minimally effective in reducing chlorophyll a concentrations outside of the Haw River arm of the lake. A set of load reductions considered this lake characteristic. Five runs that reduced N and P loading from 10% to 50% in all incoming surface waters were compared to five corresponding cases without reductions in the N and P loading from the Haw River (Table 12). As expected, eliminating the Haw River N & P load reductions lead to only a minor change in the reduction in chlorophyll a concentrations. Eliminating the Haw River loading had more of an effect when all eighteen monitoring stations were considered. When the analysis was limited to stations in the New Hope Creek arm of the lake, there was very little difference between cases with and without reductions in Haw River loading (Table 12).

Table 12. Average reduction in model predicted chlorophyll a concentration at 18 locations from the 2014-2015 and 2017-2018 base case runs for various nutrient load reduction scenarios with and without reductions in Haw River inputs (upper table) and the fraction of model predicted values exceeding the chlorophyll a water quality criteria value of 40 μg/L (lower table).

Change in N and P Loading					**Chl a Reduction**
-50%	**-40%**	**-30%**	**-20%**	**-10%**	
-23%	-17%	-13%	-9%	-3%	**With Haw Reduction, all stations**
-12%	-8%	-8%	-4%	-3%	**w/o Haw Reduction, all stations**
-22%	-17%	-13%	-9%	-4%	**With Haw Reductions, New Hope Creek arm only**
-19%	-16%	-12%	-8%	-4%	**w/o Haw Reductions, New Hope Creek arm only**

Change in N and P Loading					**% Chl a Above 40 ug/L**
-50%	**-40%**	**-30%**	**-20%**	**-10%**	
32%	35%	38%	41%	43%	**With Haw Reduction, all stations**
40%	41%	42%	43%	44%	**w/o Haw Reductions, all stations**

Another distinctive characteristic of Jordan Lake is the presence of four causeway bridges over the New Hope Creek arm of the lake. The four bridges (Hwy 64, Farrington Road (N & S), and Beaver Creek Road) greatly diminish the cross-sectional area available for up and downstream flow in the lake. Removal of one or more these bridges could potentially increase the flushing in the New Hope Creek arm of the lake, and therefore improve water quality. The hydrodynamic model used for this research (EFDC) is ideally suited to considering this effect, as it has a "masking" feature that allows for thin "no-flow" boundary between adjoining model cells. Masks were used to represent the four causeways in the calibrated model. Removal of the masks removes the no-flow boundary, and therefore allows for a simulation of the effect of eliminating the causeways. To study the "causeway removal" effect, five additional model runs, with N & P load reductions from 10% to 50% were run and analyzed as described previously. There was a measurable, but slight difference seen between the with and without causeways cases (Table 13). Removal of the causeways only very slightly improved water quality in the lake. Nitrogen and phosphorus load reductions from 0% to 50% reduced chlorophyll a concentrations by 3% to 23% for the with causeway cases, and 4% to 24% for the without causeway cases. These results suggest that the limited flushing and very high chlorophyll a concentrations in the New Hope Creek arm of the lake are not a result of the flow restrictions posed by the presence of the four causeways.

Table 13. Average reduction in model predicted chlorophyll a concentration at 18 locations from the 2014-2015 and 2017-2018 base case runs for various nutrient load reduction scenarios with and without removal of causeways (upper table) and the fraction of model predicted values exceeding the chlorophyll a water quality criteria value of 40 μg/L (lower table).

Change in N and P Loading					Chl a Reduction (% from base case)
-50%	**-40%**	**-30%**	**-20%**	**-10%**	
-24%	-18%	-13%	-8%	-4%	Cases w/o Causeways
-23%	-17%	-13%	-9%	-3%	Cases w/ causeways

Change in N and P Loading					% Chl a Above 40 ug/L	
-50%	**-40%**	**-30%**	**-20%**	**-10%**	**0%**	
33%	37%	41%	45%	48%	49%	w/o Causeways

One result of the loading analysis was that benthic cycling of nutrients was a major contributor to the water column's supply of bioavailable nitrogen and phosphorus. Particularly in the New Hope Creek arm of the lake, the loading of inorganic nutrients to the water column was principally from the benthic sediments (75% of phosphate, 90% of ammonia, see Figure 13). With this fact in mind, it is not surprising that the reductions in chlorophyll concentrations were relatively insensitive to nutrient load reductions (e.g. 40% N & P load reduction reduced chlorophyll a by 17%, see Table 11). It is possible, however, that long-term reductions in nutrient loading could eventually produce a change in the benthic flux of inorganic nutrients. As phytoplankton biomass is reduced with load reduction, settling of that organic matter would be reduced, which might eventually feedback to a reduced benthic flux of nutrients. To investigate this possibility, long-term simulations of nutrient load reductions were run using the predictive sediment diagenesis model available in the particular version of the model (EFDC+) used for this study (Craig, 2018) . EFDC+ is an updated and enhanced version of the original EFDC model (Tetra Tech, 2007) that includes among other improvements the predictive sediment diagenesis model originally developed by Di Toro and Fitzpatrick for the Chesapeake Bay water quality model (Di Toro and Fitzpatrick, 1993). To accomplish the long-term simulation, the 2014-2015 calibrated model was run repeatedly with a 50% N and P load reduction. At the end of the two-year simulation, the ending sediment conditions were saved to a file and used to initialize the sediment conditions for the next two-year simulation. The model was run this way a total of nine times, for a total simulation time of eighteen years. For each run having the time varying sediment organic matter composition and the 50% N & P load reduction, the reduction in chlorophyll a concentrations was determined as in the previous scenario test runs. Over the eighteen year time period, the reduction in chlorophyll concentrations did increase from 23% initially to 45% in the last two years of the simulation (Table 14) The change in chlorophyll a concentration reductions was most pronounced in the early years, and seem to plateau after approximately ten years (Table 14).

Table 14. Average reduction in model predicted chlorophyll a concentration at 18 locations for the 2014-2015 model time period for a 50% load reduction in N & P for various time periods using time-varying sediment organic matter compositions.

Effect of 50% N & P load reductions for the following time periods									Chl a Reduction (% from base case)
0-2 years	2-4 years	4-8 years	6-8 years	8-10 years	10-12 years	12-14 years	14-16 years	16-18 years	
-23%	-27%	-33%	-41%	-43%	-44%	-45%	-45%	-45%	

Discussion and Conclusions

The nutrient load reduction scenario analysis was designed to simulate the effects of a wide range of possible reductions in both nitrogen and phosphorus loading. Considering this broad range of possibilities with their resulting reductions in chlorophyll a concentrations, a few conclusions emerge. The first has to do with the magnitude of the decreases in chlorophyll a concentrations. Overall, the decreases in chlorophyll a were relatively modest considering the significant reductions in nutrient loading that were considered. For instance, a 50% N and P load reduction initially produced only an initial 23% decrease in chlorophyll a concentrations (Table 12 or 13), which increased to 45% eventually (Table 14), but this level was produced only after more than a decade of sustained load reductions. A second conclusion relates to the magnitude of the needed algal biomass reductions to meet numeric water quality criteria. The analysis of 2014-2018 chlorophyll a monitoring data in Jordan Lake (Table 4) showed that at least a 45% biomass reduction would be needed to lower chlorophyll a concentrations enough so that Jordan Lake meets current numeric water quality criteria for chlorophyll a. Taken together, these two results, one based on the analysis of monitoring data, and the second based upon results from the model, suggest that Jordan Lake needs significant load reductions (approximately 50%) in nitrogen and phosphorus over a long period time to meet water quality criteria. The model also suggests that reductions in loading to the Haw River arm of the lake will not improve water quality conditions in the New Hope Creek arm of the lake.

Water quality calibration relied upon existing data for the most part. Collaboratory partners provided many important new pieces of information on the lake that greatly aided the modeling project. Unfortunately there was some missing information in the water quality monitoring dataset provided by NC DWR that would have been useful for this modeling effort. In particular, there was no information available in either the watershed data or the lake data on phosphorus fractionation. Datasets used to support model efforts of this sort usually include orthophosphate measurements since this is the bioavailable fraction taken up phytoplankton during photosynthesis. Nitrogen fractionation into ammonia, Kjeldahl, and nitrate nitrogen was available and was helpful in creating model input files and for calibration purposes. Future modeling efforts in Jordan Lake would benefit greatly from adding orthophosphate measurements in the lake and the watershed.

A surprising challenge in this modeling effort was the simulation of water surface elevations. Considerable effort was expended during the project on the estimation of the flow hydrograph for the ungaged portion of the Jordan Lake watershed. The simple scaling approach based upon watershed area for estimating watershed hydrographs produced unacceptably large errors in the water surface elevation time histories at the dam. A review of previous modeling work on Jordan Lake (Tetra Tech, 2002; 2003) and other recent modeling work using the EFDC+ model for lake water quality simulation (Dynamic Solutions, 2013; Michael Baker, 2015) indicated that the problems we experienced are not unique. The earlier Jordan Lake model and the more recent Tenkiller reservoir model significantly improved the fit to observed water surface elevations by increasing the outflows at the dam. We chose not to make any adjustment to the measured outflow hydrodraph at the dam, and instead only adjusted the inflow hydrograph to the lake. This decision limited our ability to fit the model predicted elevations to the observed data. It is our judgement that this was a reasonable approach given the focus of this work on eutrophication issues rather than concerns more directly related to lake water depths.

Another challenging aspect of the modeling effort related to the predictive sediment submodel. The version of the Di Toro and Fitpatrick (Di Toro and Fitzpatrick, 1993) sediment diagenesis model used for this study (Craig, 2018) proved difficult to run, primarily due to numerical instability issues. To prevent numerical instabilities, the maximum model time step was decreased to five seconds, rather than the 100 second values that could be used for a model using the standard EFDC specified spatially and temporally varying benthic fluxes. Shortening the model time step by a factor of twenty increased computer model run times by a similar amount so that a two year run took closer to forty hours rather than two hours. This greatly complicated and lengthened model calibration and scenario testing in cases using the predictive sediment model. To get the work done within the available time, we therefore followed an approach that used both modeling approaches for the benthic sediment submodel, depending upon the available time and the modeling needs. We ended up not being able to do some model analyses that we had planned and have used in previous work (Bowen and Harrigan, 2018; Bowen and Harrigan, 2017). In future work we plan to continue to explore approaches to calibration and model testing in circumstances where a predictive sediment diagenesis model is needed.

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