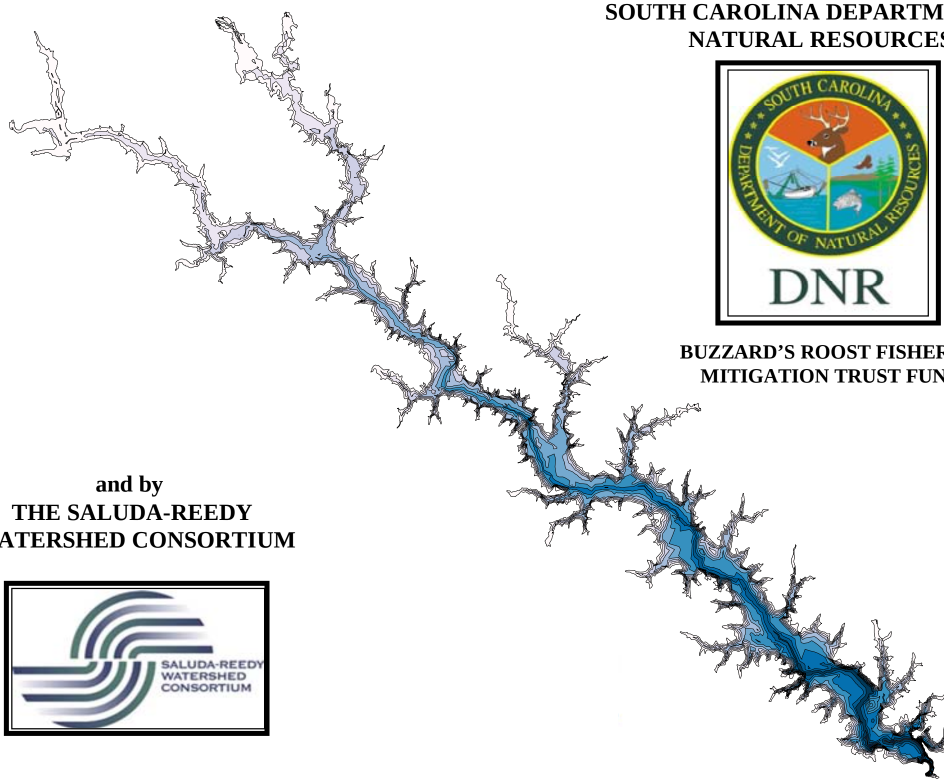
A Dynamic Water Quality Model of Lake Greenwood, SC

Development and Application toward Issues of Phosphorus Loading, Algal Dynamics, and Oxygen Depletion

Final Report

RESEARCH SUPPORTED BY SOUTH CAROLINA DEPARTMENT OF NATURAL RESOURCES

BUZZARD'S ROOST FISHERIES MITIGATION TRUST FUND



and by THE SALUDA-REEDY WATERSHED CONSORTIUM

Hank McKellar, Jim Bulak and Barbara Taylor
SC Dept. Natural Resources
Freshwater Fisheries Research Lab
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14 March, 2008

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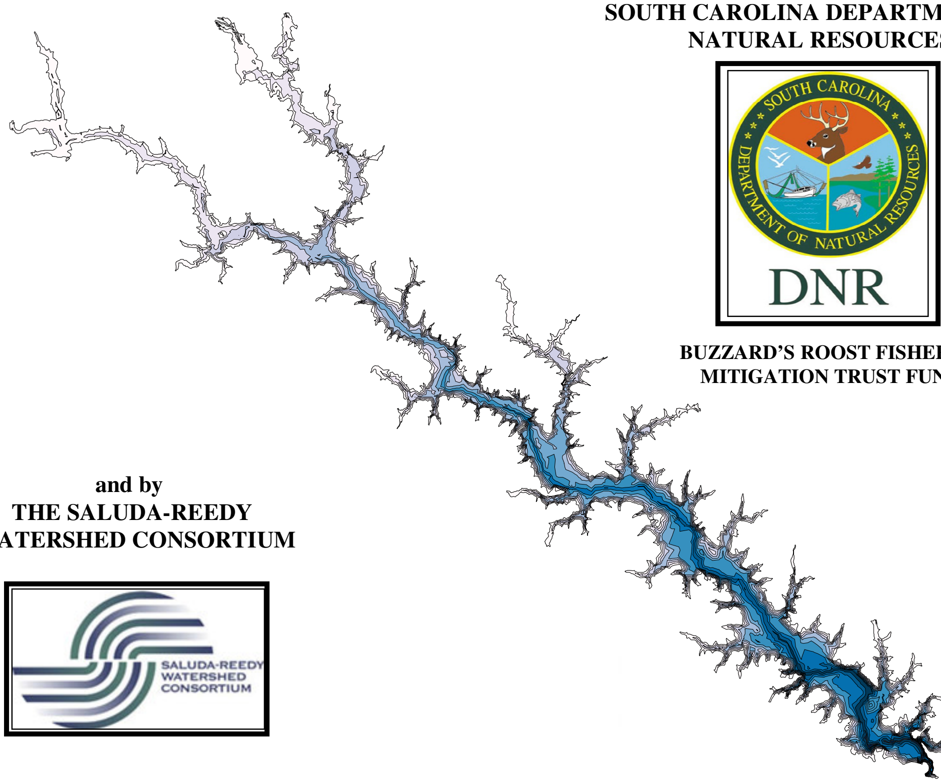

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The bathymetric analyses, GIS work, and initial model setup was accomplished through the efforts of several analysts. Gene Hayes provided the original 1989 bathymetric data for the lake. John Foster and Dawn Misura (SCDNR) scanned, rectified and digitized the 1989 bathymetric data, combined it with more recent data, and produced the final bathymetric map of the lake. Jason Bettinger (SCDNR) provided assistance for calculations of map properties. Steve Springs (North Wind, Inc.) developed the GIS contour extrapolations and determined area relationships for the contour polygons. Gary Hauser (Loginetics, Inc) provided initial assistance and consultation with the CE-QUAL-W2 model including the use of W2i/AGPM for processing input data, model output, and preliminary calibrations

ABSTRACT

Lake Greenwood is a 42 km^2 impoundment of the Saluda River in the piedmont ecoregion of South Carolina. The reservoir represents a valuable regional resource for flood control, water supply and recreation. Although the lake has a productive fishery, water quality and aquatic habitat is threatened by excessive nutrient loading, intense algal blooms and extensive oxygen depletion in the bottom waters. The main objectives of this research were to develop a database and simulation model of the lake to help predict implications of changing phosphorus loading. We collected a 2-year database (2004-05) on phosphorus, algae, and oxygen dynamics in the lake and used the data to set up a dynamic water quality model (CE-QUAL-W2). To calibrate the model, we used updated data for lake bathymetry, nutrient loading from the two main tributaries (Saluda and Reedy Rivers) and the corresponding water quality dynamics within the lake. The model was calibrated for particular focus on phosphorus distributions, algal biomass, and oxygen dynamics along the main axis of the upper tributaries to the forebay. The overall absolute mean errors for the two-year simulations were 1.1 mg/L for dissolved oxygen, 0.05 mg/L for total phosphorus, and 6.8 μg/L for chlorophyll-*a*.

For current conditions, the total phosphorus loading to the lake was dominated by the Saluda River (74%) although the mean concentration in the Reedy River (0.11 mg/L) was 22% higher than in the Saluda. Daily inflow concentrations exceeded the SC standard for piedmont lakes (0.06 mg/L) more than 58% of the time in the Saluda River and > 68 % of the time in the Reedy River. With this loading, the model simulated observed patterns of phosphorus distribution in the lake, with high and variable mean concentrations in the upper arms (0.07-0.09 mg/L annual means), declining to < 0.05 mg/L in the surface waters of the lower lake. The model also reproduced documented seasonal pulses of algal biomass (chlorophyll-*a*) with peak concentrations during the summer in the upper arms (40-60 μg/L). The model also reproduced observed patterns of hypolimnetic oxygen depletion, which began in early spring and resulted in 34 % of the lake volume being occupied by concentrations < 1 mg/L by late summer. Together with warm surface temperatures during the summer, these oxygen conditions offered limited habitat for cool water fish species such as striped bass. For current conditions, during the peak of summer stratification, the model estimated that < 1% of the main lake volume met the tolerable habitat requirements for striped bass.

We used the model to test the consequences of changes in the phosphorus load to water quality and habitat patterns in the lake. While reductions of phosphorus load in the Reedy River alone would lead to some improvement in water quality, particularly in the Reedy Arm, the model indicated that a substantial reduction of loading in both rivers would be required for lake-wide improvement in water quality. A 50% reduction in the phosphorus load in both rivers would reduce the annual mean phosphorus concentration and the risk of algal blooms, especially in the Upper Reedy Arm of the lake. This level of load reduction would also decrease the extent of extreme hypoxia by 31% throughout the year and would increase the tolerable habitat for striped bass by about 10 %.. Further reductions could increase tolerable habitat but would need further evaluation regarding impacts on algal biomass, forage fish production and the overall sport fishery.

As currently calibrated, the Lake Greenwood model simulated basic patterns of water and habitat quality in the lake and offered science-based predictions of some consequences of phosphorus load reduction. The model could form an integral link in the tools available for effective watershed/water quality management in the Saluda-Reedy watershed.

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INTRODUCTION

Lake Greenwood is the first major impoundment on the Saluda River, located approximately 100 km downstream from the Saluda headwaters in the SC Blue Ridge Mountains. With a total surface area of 41.9 km^2 (10,400 acres), the reservoir has a productive fishery although high levels of nutrient loading and eutrophication may threaten water quality and biotic habitat. Extended periods of oxygen depletion in waters below the thermocline in Lake Greenwood are widespread and may limit viable habitat for fish and benthic organisms (Snoots 1993). Although some oxygen depletion below the thermocline is a natural process in stratified lakes (Wetzel 1983, Kalff 2002), the rate of depletion, and the extent and duration of hypoxic conditions can be affected by nutrient loading, subsequent algal blooms, and oxygen consumption of excess organic matter (Lee and Jones 1984). A recent assessment of water quality in SC confirmed earlier designations of Lake Greenwood as "impaired" due, in part, to excessive phosphorus concentrations (SC DHEC 2006) in the upper and mid-sections of the lake. In response to these assessments, the SC Dept. of Health and Environmental Control (SCDHEC) is currently developing recommendations for a total maximum daily load (TMDL) of phosphorus inputs to the most eutrophic arm (Reedy River) of Lake Greenwood.

As a complement to these efforts, the SC Dept. of Natural Resources (SCDNR) has developed a dynamic water quality model of Lake Greenwood, aimed a predicting the consequences of phosphorus load reductions to patterns of water and habitat quality in the lake. The primary goals of the SCDNR study were

- to develop a comprehensive dataset for Lake Greenwood aimed at quantifying key interactions among lake hydrology, nutrient loading and water quality and
- to develop and test a computer simulation model of water quality dynamics in Lake Greenwood designed to predict implications of future TMDL's and to help formulate long-term plans for water quality enhancement and aquatic habitat protection.

The basic conceptual scope of the study (Fig. 1) was to link information on nutrient inputs from the larger watershed (point-source dischargers and nonpoint source runoff) to ecological/water quality patterns and interactions within the lake. The details of watershed-level interactions and modeling (left panel of Fig. 1) are the focus of continuing research on the Saluda-Reedy watershed. The focus of this report is on the "in-lake" response (right panel) as represented in interactions among nutrient distributions, algal production, and oxygen balances.

The total nutrient load to the lake interacts with the physical properties of the lake (such as morphology, hydrology, and related water residence time) to control the distributions of nutrient concentrations throughout the lake. Some of the nutrient load (especially in the case of phosphorus) adsorbs to suspended sediments and settles to the benthic sediments. The remaining bioavailable phosphorus supports algal production which is an important base of the food web leading to fishery production. However, excessive nutrient loads can lead to more eutrophic conditions with excessive algal production, potential changes in the algal community composition, and noxious algal blooms. A common consequence of eutrophication is increased oxygen depletion caused by decomposing algal biomass. Widespread episodes of oxygen depletion may reduce the volume and area of viable biotic habitat and may cause fish kills. Furthermore, whenever anaerobic conditions develop in the bottom waters, the benthic sediments release nutrients back to the water, representing an "internal" source of nutrient loading which can further enhance algal production.

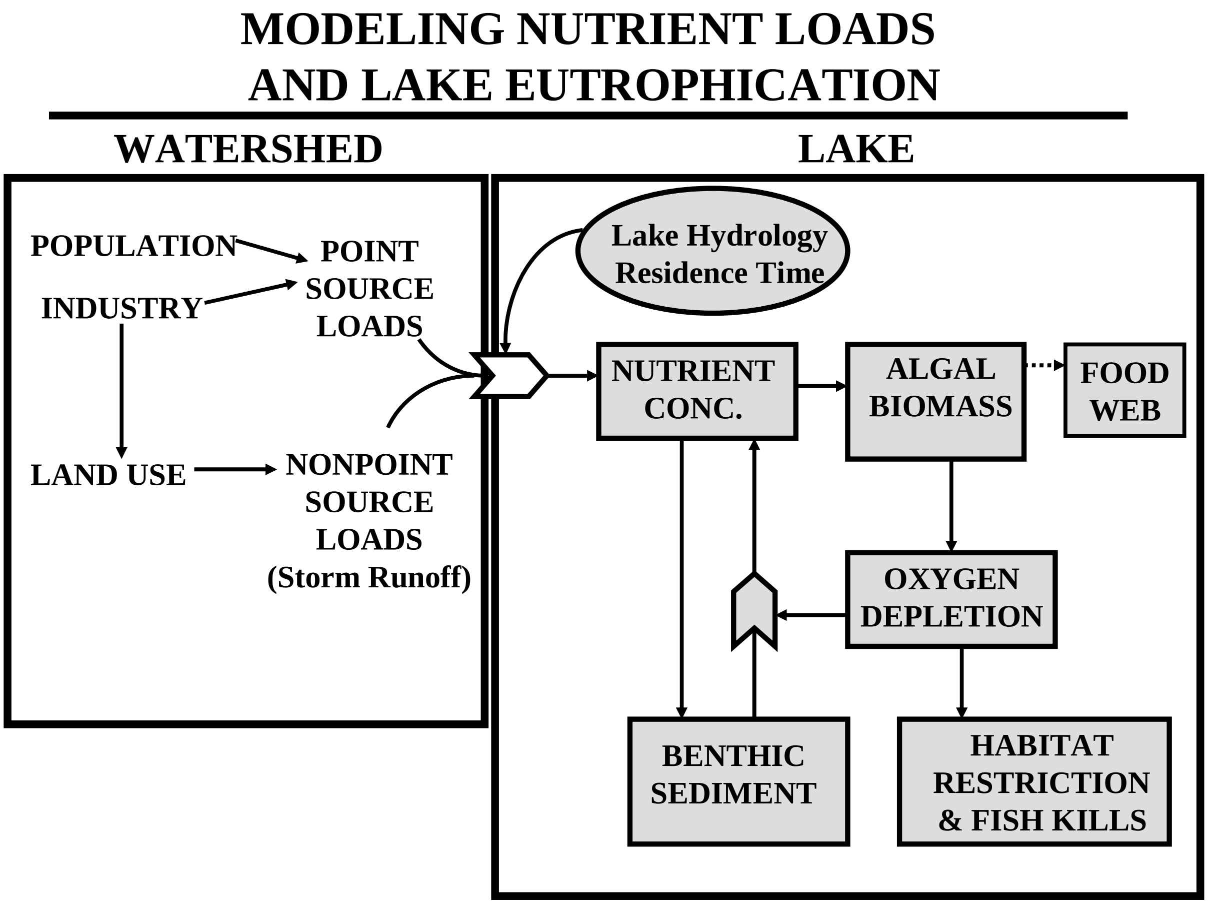

Fig. 1. Conceptual Diagram of Watershed/Water Quality Interactions

The main goal of this study was to enhance our understanding of these interactions in Lake Greenwood and to develop a simulation model (CE-QUAL-W2) to help predict consequences of nutrient management alternatives (i.e. TMDL development). As part of this effort, we collected an intensive 2-year data set (2004-2005) on phosphorus, phytoplankton, and oxygen distributions in the lake. These data were used to quantify some basic water quality dynamics in the lake and to provide a comprehensive dataset for model calibration. While the first year of data collection was reported earlier (McKellar and Bulak 2005), the full, 2-year dataset is documented here in Appendix A. The emphasis of this section of the report is to document the development, calibration, and initial application of the model. We used the model to predict the consequences of changing phosphorus loads on the distributions of phosphorus and algal biomass within the lake as well as the resulting degrees of oxygen depletion and viable aquatic habitat.

MODEL DEVELOPMENT

Model Selection and Description

The main sources of spatial variability in long, narrow reservoirs like Lake Greenwood include longitudinal (upstream-downstream) and vertical (surface-bottom) dynamics in the lake. Longitudinal variability arises largely from the transition between a riverine-like environment at the upstream end (typically the source of hydrologic inputs) and a lacustrine environment at the downstream end (typically near the dam with its hydrologic releases) (Thornton et al. 1990). The vertical variability arises mainly from seasonal thermal stratification between warm upper layers (epilimnion) and the cooler bottom layers (hypolimnion) (Wetzel 1983). To fully account for these sources of variability, we chose the US Army Corps of Engineers lake and reservoir model (CE-QUAL-W2) which has an established history of development (Cole and Wells 2002) and application in issues of hydrodynamic and water quality dynamics (Boegman et al. 2001, , Bowen and Hieronymus 2003, Moskus et al. 2003, Sawyer and Ruane 2003, Sullivan and Rounds 2005, Ruane and Hauser 2006). The model is a laterally-averaged, 2-dimensional (longitudinal and vertical) hydrodynamic and water quality model that is well suited for long, narrow water bodies (Cole and Wells 2002).

The bathymetry of the lake is represented in CE-QUALl-W2 by a series of cells representing longitudinal segments and vertical layers. Within each cell, the model simulates a wide range of physical and biochemical interactions (Fig. 2, Table 1) including the physical transport of constituents between adjacent cells and biotic exchanges involved with photosynthesis, excretion, mortality, respiration and decay.

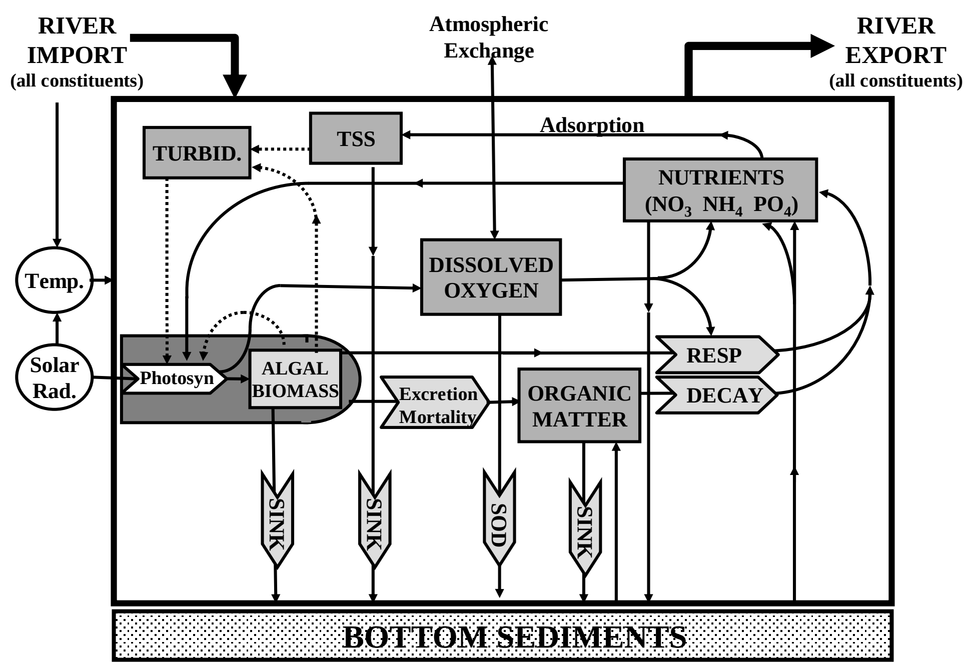

Fig. 2. Basic biochemical interactions in the CE-QUAL-W2 model. Turbid.= turbidity, TSS =total suspended solids (inorganic), RESP =respiration, SOD=Sediment oxygen demand. Solid lines indicate mass transfer from one compartment to another; dotted lines indicate an influence of one compartment on another, with no mass transfer

Table 1. List of major constituents and pathways of internal sources and sinks in CE-QUAL-W2 V3.1 (From Cole and Wells 2002).

CONSTITUENT	INTERNAL SOURCES	INTERNAL SINKS
Algae/Phytoplankton (3 phytoplankton groups)	Photosynthesis/Growth	Respiration, Mortality, Excretion, Sinking
Bioavailable Phosphorus [1] (dissolved and adsorbed)	Algal Respiration Organic Matter Decay Sediment Release	Algal Uptake/Growth Sedimentation
Ammonium	Algal Respiration Organic Matter Decay Sediment Release	Algal Uptake Nitrification
Nitrate+Nitrite	Nitrification	Algal Uptake Denitrification
Labile Dissolved Organic Matter (LDOM)	Algal Mortality/Excretion	Decay
Refractory Dissolved Organic Matter (RDOM)	Decay of LDOM	Decay
Labile Particulate Organic Matter (LPOM)	Algal Mortality	Sinking, Decay
Refractory Particulate Organic Matter (RPOM)	Decay of LPOM	Sinking, Decay
Dissolved Oxygen	Surface Exchange with Atmos. Algal Photosynthesis	Surface Exchange Algal Respiration Nitrification, Sediment Oxygen Demand Organic Matter Decay

[1] Total phosphorus is computed as the sum of bioavailable phosphorus and the phosphorus content of organic matter in the water (algal biomass, LDOM, RDOM, LPOM, and RPOM)

Bathymetry and Model Segmentation

The database for the bathymetric analysis included 11,307 geo-referenced depth points derived from 3 separate depth surveys. The SC DNR conducted a whole-lake survey in 1989 (1,810 points) where cross-lake transects were mapped using a depth recorder and aerial photography of the lake shoreline. The printed map from this survey (Hayes and Penney 1989) was scanned and rectified to the latest US Geological Survey, digital line graph (DLG) of the lake shoreline. Additional depth transects (155 points) were provided by the Natural Resources Conservation Service (Kroeger 2002) who conducted range pole transects with in-field GPS coordinates across the upper sections of the Saluda and Reedy arms. Finally, 9,342 additional points were added in the upper Reedy arm based on 3D sediment modeling utilizing data from Kroeger (2002) and North Wind (2006). This last study was designed specifically to quantify detailed volume loss in the upper arms due to recent sedimentation (Hargett 2004). All individual depth measurements were then digitized by SC DNR (John Foster and Dawn Misura) for additional GIS analysis.

Combining all data from the depth surveys, along with shoreline and stream centerline GIS files, North Wind, Inc. (Steve Springs) created a triangulated irregular network (TIN) to develop 1-m contour lines throughout the lake. For some areas where direct observations were sparse, additional depth values were added (largely along known channels) to improve the accuracy of the contouring results. Once a realistic contour map was created (Fig. 3), the lake map was partitioned into approximately 1-km long segments and 1-m depth layers for spatial representation in the water quality model (Appendix B). According to this combined bathymetry and GIS analysis, Lake Greenwood has a total surface area of 41.9 km^2 (10,357 Acres), a total volume at full stage of 259.1 x 10^6 m^3, and a mean depth of 6.2 m (Fig. 4). These values for surface area and volume were 9 and 19% lower, respectively, than earlier reported values (Inabinet and Pearse 1981, Stecker and Crocker 1991). This reduction may be due, in part, to recent, accelerated rates of sediment deposition in the upper lake (Hargett 2004) as reflected in recent bathymetric data used in this study.

MODEL CALIBRATION

The Lake Greenwood model was calibrated using data on hydrology, loading, and in-lake distributions of major water quality parameters for two full annual cycles, 2004-2005. During this 2-yr time period, we had access to detailed information on loading dynamics (Klaine and Smink 2004-05) as well as detailed data for in-lake distributions of water temperature, dissolved oxygen, total phosphorus and chlorophyll-a (Appendix A). These 2 years also provided a range of hydrologic conditions (Fig. 5) with corresponding potential effects on water quality. While the annual average flows in the Saluda River during 2004 (1018 cfs) was relatively normal (54^{th} percentile), seasonal variability included a low-flow spring (41% below normal) and a high-flow fall (66% above normal). The following year (2005) included a high-flow summer (70% above normal) and a relatively normal (low flow) fall. Overall, 2005 was a relatively high flow year for the Saluda, with a mean discharge (1282 cfs) in 88^{th} percentile of long-term records. This time period did not include an extended period of drought conditions, which could have considerable effect on water quality dynamics. An appropriate follow-up for model calibration and testing would be to apply drought conditions on hydrologic inputs in order to predict the implications on water quality dynamics.

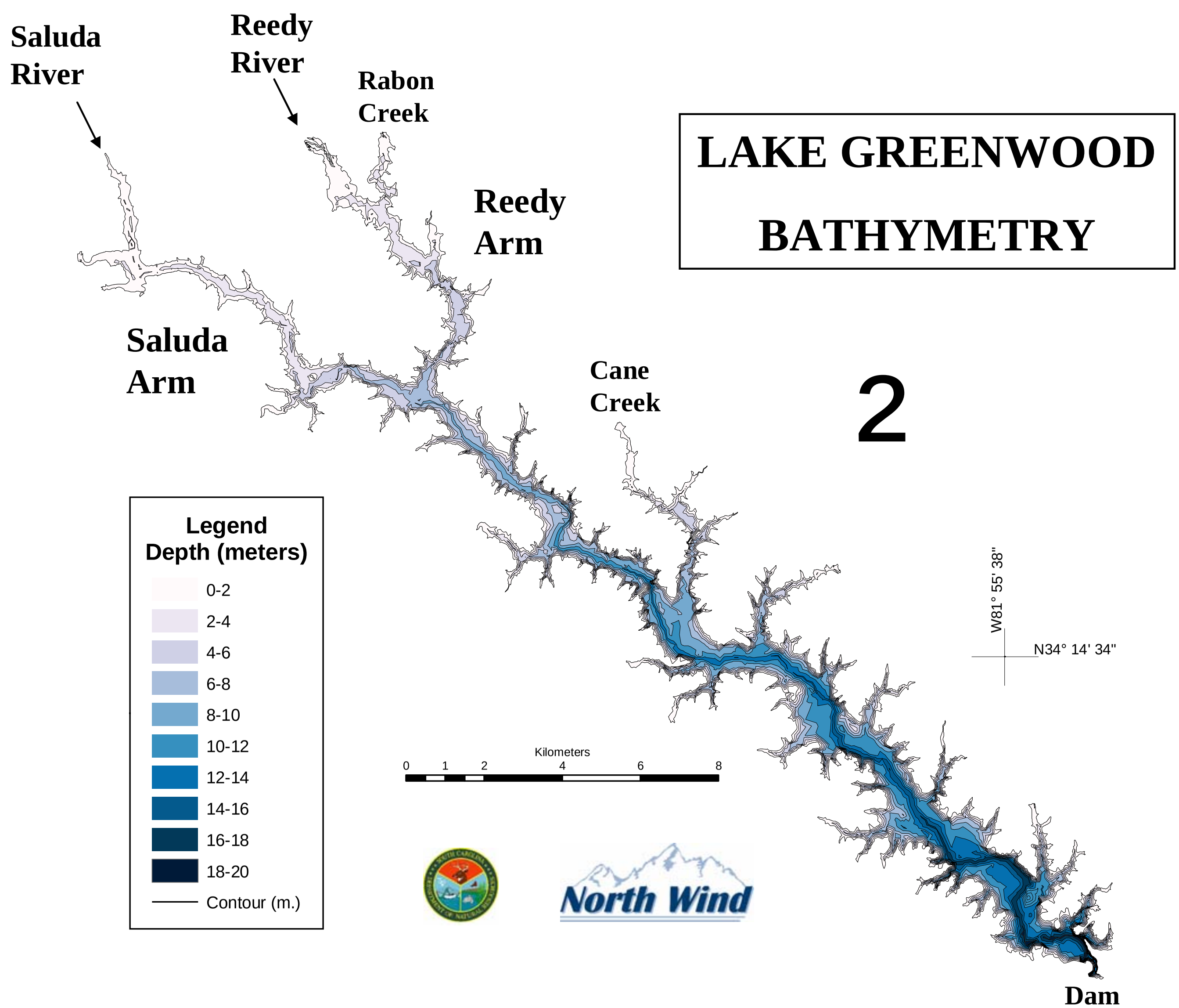

Fig. 3. Bathymetric map of Lake Greenwood, SC. The main axis of the lake begins with the inflow of the Saluda River to the Saluda Arm of the lake, 41.4 river-km (rkm) upstream from the dam. The Reedy Arm begins at the inflow of the Reedy River (r-km=36.3) and flows downstream to the conjunction of the 2 arms at rkm 27.1

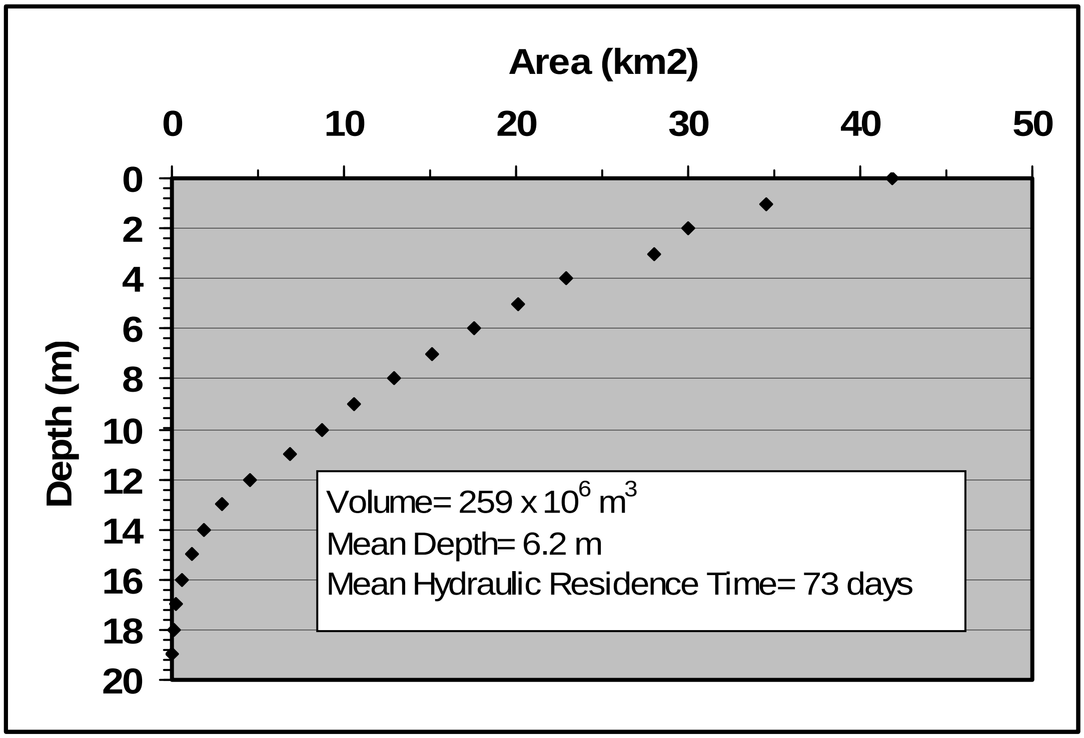

Fig. 4. Hypsographic curve for Lake Greenwood. Based on the compilation of all available bathymetry data (described above) and the resulting contours (Fig. 3). The mean hydraulic residence time was based on the 8-yr mean (1997-2005) for total discharge in the Lake Greenwood tailrace (41.3 m^3/s, Cooney et al. 2005)

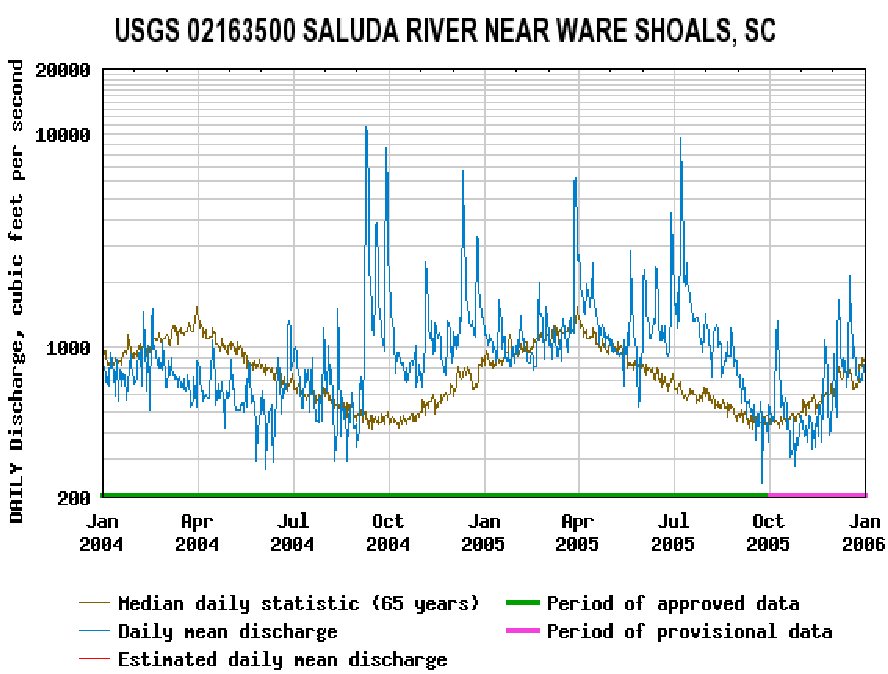

Fig. 5. Daily discharge in the Saluda River at Ware Shoals during 2004-2005 (blue line) compared to the long-term (65-yr) mean daily discharge at this site (brown line; http://waterdata.usgs.gov/nwis/)

Initial settings for coefficients and constants in the CE-QUAL-W2 model were based on default values and examples provided by Cole and Wells (2002) with recent modifications by Ruane and Hauser (2006). The complete set of model input files (including lake bathymetry and model segmentation, meteorology, input/output hydrology and water quality, and the model control files) are available from the authors. Appendix C in this report, provides a basic listing and description of the major calibrated coefficients and constants along with comparisons to default settings from Cole and Wells (2002)

Water Balance

The initial steps in model calibration were accomplished in collaboration with Loginetics, Inc.(Gary Hauser, Knoxville TN) who applied extensive experience with CE-QUAL-W2 development and recent applications (Ruane and Hauser 2006). Daily hydrologic inputs were based on USGS gaging stations on the lake and the major inflowing tributaries (Fig. 6). The water level in Lake Greenwood was maintained according to an operating "rule curve" approved by FERC and SC environmental agencies. The rule curve calls for water level reductions through the late fall and winter to a late January minimum of 434.5 ft (MSL). Beginning in February, the water level increased gradually to 439 ft by mid-April. This level was maintained through summer and early fall with a gradual drawdown beginning again in November. Some fluctuations in water level (+/- 1 ft) occurred in response to several major storm events in September and December, 2004. The rapid drawdown in August 2005 was in anticipation of Hurricane Katrina.

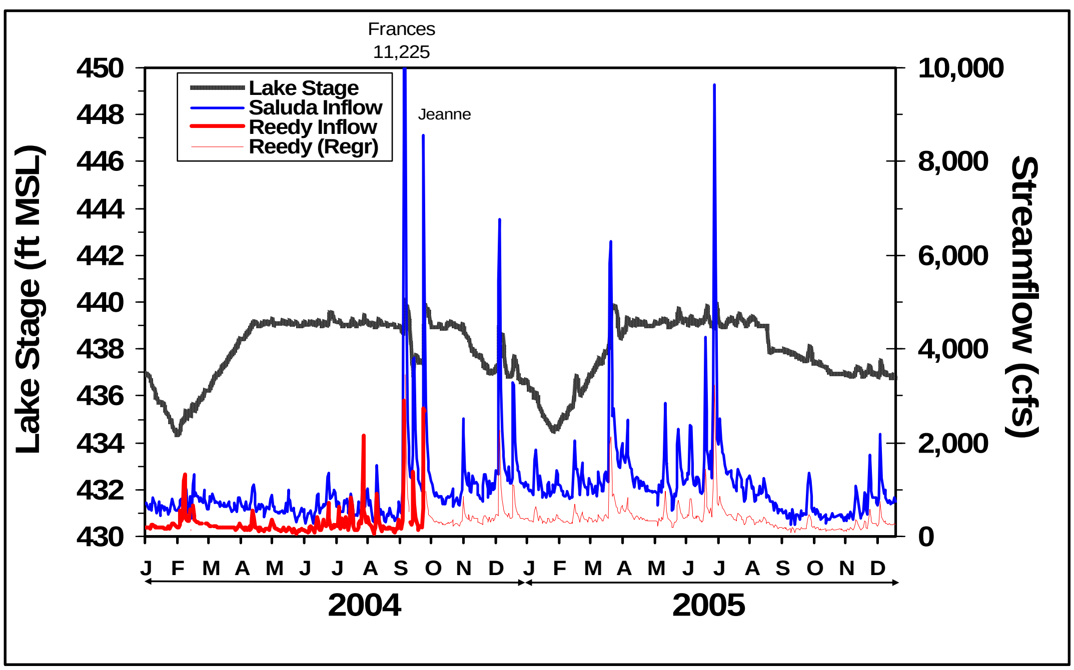

Fig. 6 Lake Greenwood water level (forebay of Lake Greenwood near Chappells, USGS 02166500) and major tributary inflows from the Saluda River at Ware Shoals (USGS 02163500) and the Reedy River at Ware Shoals (USGS 02165000). Extended gaps in gaging records for the Reedy River, after storm damage in September 2004, were extrapolated from a 2003 regression of Reedy River discharge with Saluda River discharge (Reedy_cfs = 0.3336 Saluda_cfs + 9.2892; r^2 = 0.7838;see Appendix D).

The Saluda and Reedy Rivers represented the major sources of inflow to Lake Greenwood. Long-term mean daily discharge (since 1939) in the Saluda (976 cfs, http://nwis.waterdata.usgs.gov) was about 2.8 times that in the Reedy (352 cfs). This relative relationship was roughly the same during this study period, except for a few short-term runoff events in February and July, when flow in the Reedy was approximately equivalent to those in the Saluda (Fig.6). The major hydrologic events of the year were a series of tropical storms in September (Frances, Ivan, and Jeanne). The peak discharge in the Saluda following Hurricane Frances (11,225 cfs) was more than 10 times the long-term annual mean and was about 70% of the highest daily flow on record (16,100 cfs, Aug 27, 1995). The mean flow for September 2004 (2,837 cfs) was almost 5 times higher than the average flow for this month (594 cfs) and about 52% higher than the long-term maximum flows for September (1,862 cfs). The gaging station on the Reedy was damaged during the September 2004 storms so the remaining time-series estimates for Reedy River discharge was based on a 2003 regression with the Saluda (see Fig. 6 and Appendix D).

Gauged outflows were based on discharge in the tailrace canal (USGS 02166501), which included flow through the hydroelectric turbines and release through the spillway gates. A minor outflow (19.9 ± 2.4 cfs) also occurred through the Greenwood County CPW drinking water withdrawal at mid-lake (W.R. Wise Water Treatment Plant, approx. 20 km upstream from the dam, David Tuck, pers. comm.). Hourly meteorological conditions (air temperature, dew point, wind speed and direction, and cloud cover) at Greenwood County airport were provided by Southeastern Regional Climate Center (Columbia SC).

The <u>total</u> water inflow to the lake included inputs from the gauged rivers (Saluda and Reedy Rivers), ungauged tributaries (Rabon, Turkey, and Cane Creeks) and lakeside drainage. This total inflow was estimated as the sum of daily total outflow (tailrace discharge plus drinking water withdrawal) plus change in lake water storage volume (Appendix E). After minor calibration adjustments, the model reproduced the observed changes in lake stage over the 2-year simulation period (Fig. 7). The model slightly under-predicted water levels during the spring 2004 recharge and briefly over-predicted spikes during major storm events. Overall, the absolute mean error in the simulated lake water level (A = 0.035 m) was small, only 2.1 % of the annual range in lake water levels.

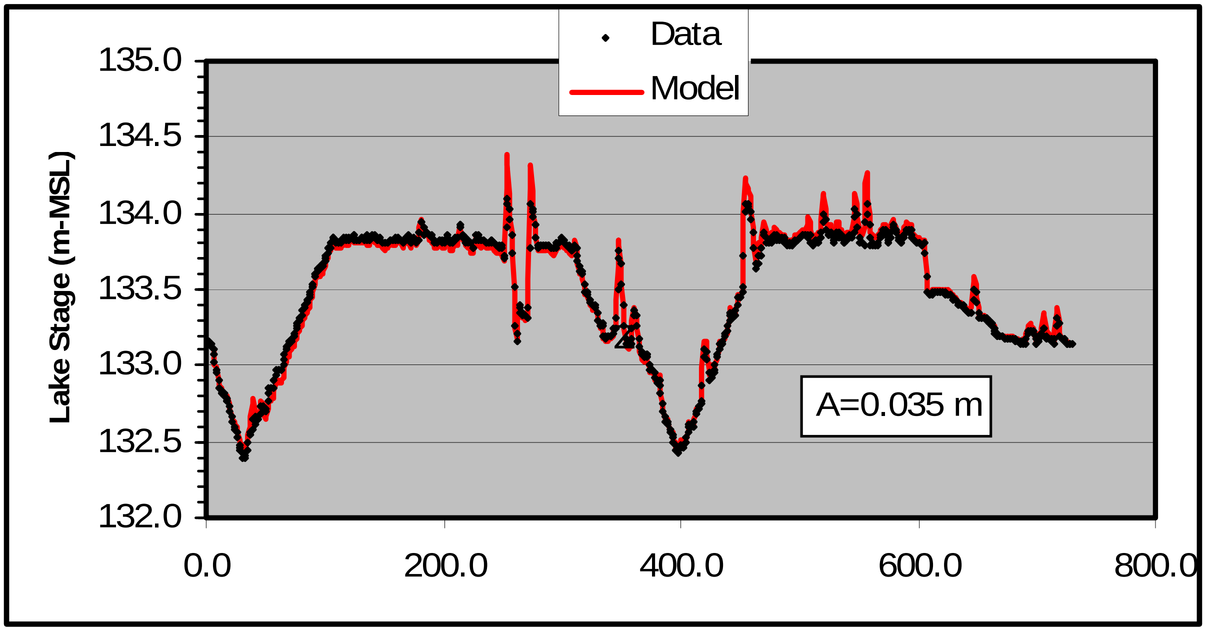

Fig. 7. Observed and simulated patterns of lake water elevation (Lake Stage) for 2004-2005.- "A" is the absolute mean error of the simulated 2-year time series, where A = ∑ |Model-Data|/n; n = total number of model-data pairs (729).

Water Quality Calibration

Daily inflow concentrations from the Saluda and Reedy Rivers near Ware Shoals (Fig. 8) were based on automated storm event sampling and base flow grab sampling results from Klaine and Smink (2004-05), combined with monthly grab sampling by SCDHEC at the same sites (Reedy River near Ware Shoals: Clemson RR3 and DHEC S-021; Saluda River near Ware Shoals: Clemson SR2 and DHEC S-125). Inflow concentrations on dates between sampling events were obtained by linear interpolation.

During initial calibrations, several correlations between hydrologic patterns and water quality constituents were evaluated for possible use in future work to fill in some data gaps in the loading record (Appendix D). Other notes on loading, calculations and assumptions are provided in Appendix F. Concentrations in the distributed inflow (nonpoint-source runoff) along the Saluda Arm and the main body of the lake were assumed to be the same as for inflowing concentrations in the Saluda River. Similarly, concentrations in distributed inflow along the Reedy Arm of the lake were assumed to be the same as for the inflowing Reedy River. Concentrations in the Cane Creek input and associated distributed flow along the Cane Creek embayment were based on SCDHEC monthly monitoring results on Cane Creek.

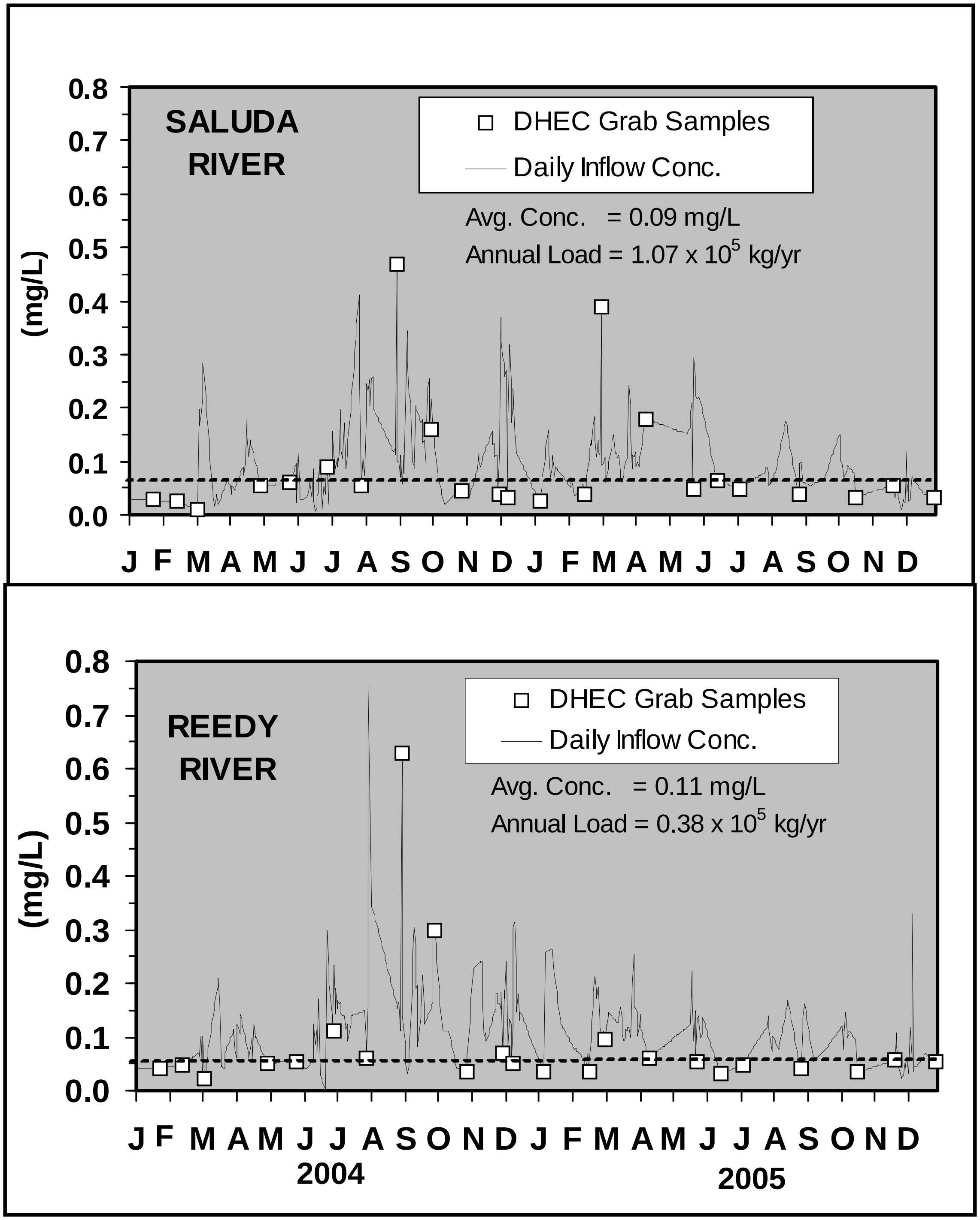

Fig. 8 Total phosphorus concentrations in the Saluda and Reedy Rivers near Ware Shoals. Daily inputs used in the Lake Greenwood model were based on the automated storm-flow sampling by Clemson University (Klaine and Smink 2004-05) in conjunction with SC DHEC monthly grab samples. Dashed lines indicate the water quality standard for Piedmont lakes (0.06 mg/L)

The calibration of model parameters for in-lake processes was based on 2-yr model runs representing meteorologic and hydrologic conditions during the 2004-2005 sampling period. Model output was calibrated to SC DNR data (SC DHEC certified) for water temperature, dissolved oxygen, total phosphorus, and chlorophyll-*a* at key locations along the major arms and lake proper (Table 2, Appendix A, Fig. A1, Table A1). Data from these sampling stations represented the main longitudinal, vertical and temporal variability observed in the lake during this 2-yr period. Additional calibration adjustments may be needed for specific applications of the model to semi-enclosed tributary embayments, such as Hidden Lake on the Cane Creek embayment (see "MODEL LIMITATIONS", p. 35).

Table 2. Lake Greenwood study sites and descriptions. A combination of all sampling sites except "Hidden Lake" was used in model calibration. Hidden Lake represented a special habitat (enclosed tributary embayment with low flushing rates) that should receive additional calibration work in future applications of the model to this type of habitat.

STATION NAME	DESCRIPTION *	CODE	Lat. N	Lon. W	Distance fr. Dam (km)	Depth** (m)
Open lake forebay	1.3 km NW of Dam	FBY	34.1792	81.9097	1.6	19
Greenwood State Prk	0.5 km NE of State Park Office	GSP	34.1967	81.9432	6.4	17
Random Lake Sta	2.5 km SE of Goat Island	RND	34.2193	81.9643	9.7	16
Irvines Point	150 m N of Irvine Camp Point	IRP	34.2430	82.0098	16.0	13
Hidden Lake***	2.7 km downstream of Cane Creek bridge	HDN	34.2743	82.0157	21.2	5
Highway 72 Bridge	0.2 km SE of Hwy 72/221 bridge	HW72	34.2782	82.0573	23.0	10
Lower Saluda Arm	0.2 km E of Hwy39 bridge	SBR	34.3082	82.1100	29.2	7
Upper Saluda Arm	2.7 km downstream from Souls Harbor	USAL	34.3257	82.1423	35.0	3
Lower Reedy Arm	0.2 km W of point at Twin Rivers Landing	LRDY	34.3098	82.0862	26.0	6
Middle Reedy Arm	0.5 km E of HW29 bridge	MRDY	34.3258	82.0812	28.1	6
Upper ReedyArm	0.3 km SE of mouth of Rabon Crk	RDY	34.3427	82.1005	31.5	2

* All stations were in the main channel of the lake and tributary arms
** At full pool elevation (440ft MSL; 134.1m)
*** Station not used in current calibration

Temperature and Dissolved Oxygen

After initial setup of the model using recommended "default" values for most parameters (Cole and Wells 2002), we first adjusted several sensitive coefficients for initial calibration of temperature and DO. For water temperature, the most sensitive parameters were associated with physical and hydraulic factors such as surface heat exchange (SLHTC=ET rather than TERM), bottom roughness (CHEZY roughness coefficient = 100 $m^2 s^{-1}$, i.e. rougher than default value of 70 $m^2 s^{-1}$), vertical eddy viscosity (AZMAX = 1×10^{-4} $m^2 s^{-1}$) and heat loss to sediments (TSEDF=0, i.e. no short-wave radiation transmitted from sediments to overlying water)(See Appendix C). Other sensitive calibration parameters represented the physical location of the turbine intake structures. Although, the actual centerline of the turbine intake structure (ESTR) was at elevation 125 m MSL, the upward slope of the draft tubes from the turbine to the intake structure probably caused the effective centerline of water intake to be somewhat higher so ESTR was calibrated upward to 128m.

Sediment oxygen demand (SOD) was an important calibration factor for dissolved oxygen profiles. CE-QUAL-W2 simulates SOD as 2 processes; (a) a 1st-order decomposition of recently settled particulate organic matter and dead algae (0.08 d^{-1}) and (b) a zero-order oxygen consumption by the sediments ($g\ m^{-2}\ d^{-1}$), adjustable by model segment. Direct measures of SOD ($1.8 \pm 0.2\ g\ m^{-2}\ d^{-1}$) were available for the Reedy Arm of upper Lake Greenwood, based on recent work by the USEPA (Parsons 2005). Although, this SOD rate was applied as the zero-order SOD rate in the Reedy Arm in the model, this value was adjusted downward for application to other areas of the lake. Our final calibrated levels of SOD were $1.8\ g\ m^{-2}\ d^{-1}$ for the Reedy Arm, $1.0\ g\ m^{-2}\ d^{-1}$ for the Saluda Arm and $0.5\ g\ m^{-2}\ d^{-1}$ for the lake proper. A more comprehensive study of lake-wide distributions of SOD is needed for more effective calibration of this parameter.

After calibration of model parameters for Lake Greenwood, the model accurately simulated the dominant trends in water temperature and dissolved oxygen along the main axis of the lake from the upper arms to the forebay. Fig .9 (a and b) shows representative patterns of temperature (a) and dissolved oxygen(b) at a selected site in the lower-lake. The absolute mean error for the simulated temperature in this series ranged from 0.180 ^{o}C (13 Sept) to 1.432 ^{o}C (14 Dec). Although the model under-predicted water temperatures during the autumn turnover (Nov-Dec), the model exhibited a good fit ($A < 1\ ^{o}C$) during the 8 months of thermal stratification (spring through early fall). The overall temperature calibration statistics (Table 3) indicated a good calibration ($A< 1\ ^{o}C$) with slightly better fit in the lower lake and forebay ($A= 0.7\text{-}0.8\ ^{o}C$) than in the upper arms and mid-lake areas ($A=0.9\text{-}1.1\ ^{o}C$)

Dissolved oxygen simulations reproduced the major patterns observed in Lake Greenwood, including DO depletion below the thermocline during the spring and summer and re-oxygenation during fall overturn (Fig. 9b). Similar to the temperature simulations, the simulated DO profiles were better (A<1 mg/L) during the April-Oct period of stratification. The model was less accurate (A = 1.1-1.9 mg/L) during more isothermal conditions in early spring and late fall when the model tended to over-predict DO values. Additional refinement of parameters related to heat exchange and reaeration may improve overall DO and temperature simulations. However, the current calibration is good for the critical period of stratification and oxygen depletion, particularly in the lower lake and forebay (A = 0.9-1 mg/L; Table 3), where oxygen depletion affects a larger volume of aquatic habitat. The model is less accurate in the middle Reedy arm (A=1.4 mg/L) where more focused calibration work may be needed in the future.

TEMPERATURE: LOWER LAKE (RND) (2005)

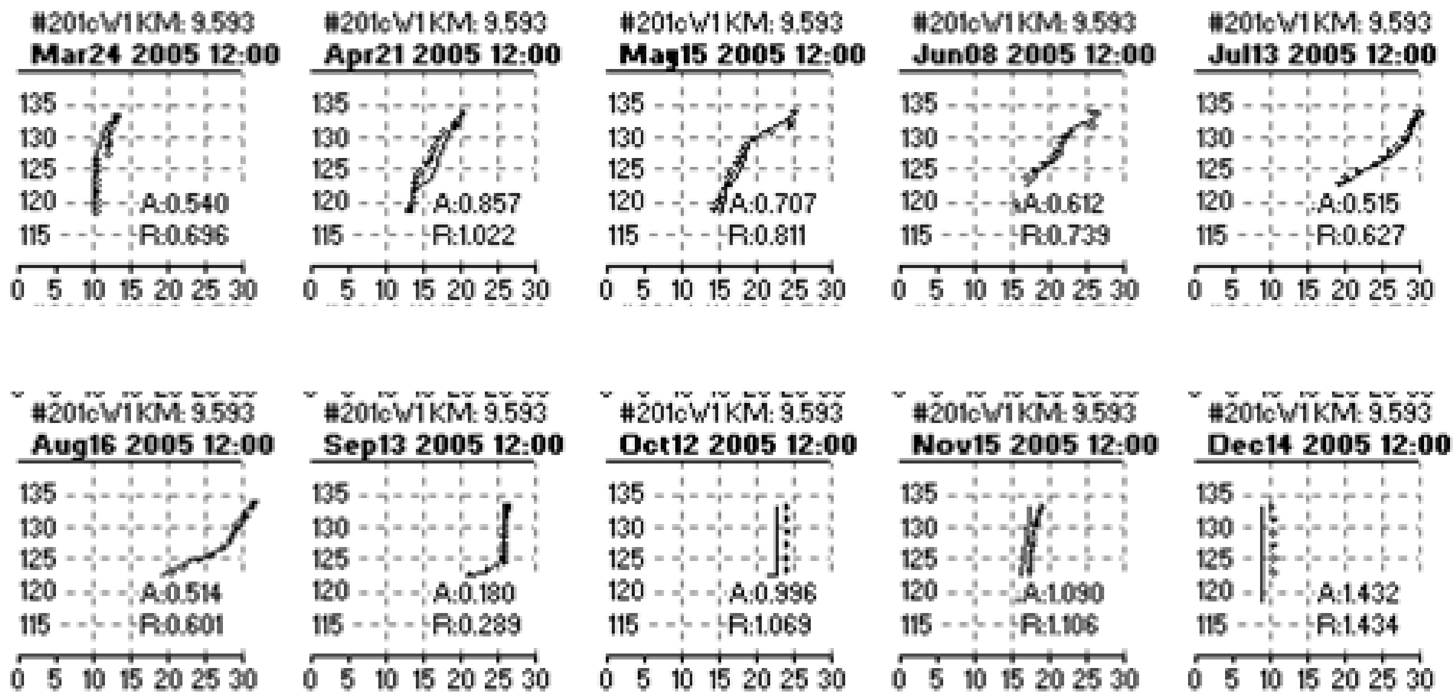

Fig. 9-a. Representative calibration plots for temperature profiles in the lower lake (Sta=RND, rkm 9.6). Model output is shown in solid lines while observed data are in open circles. Calibration statistics include absolute mean error (A) and the root mean square error (R)

DISSOLVED OXYGEN: LOWER LAKE (RND) 2005

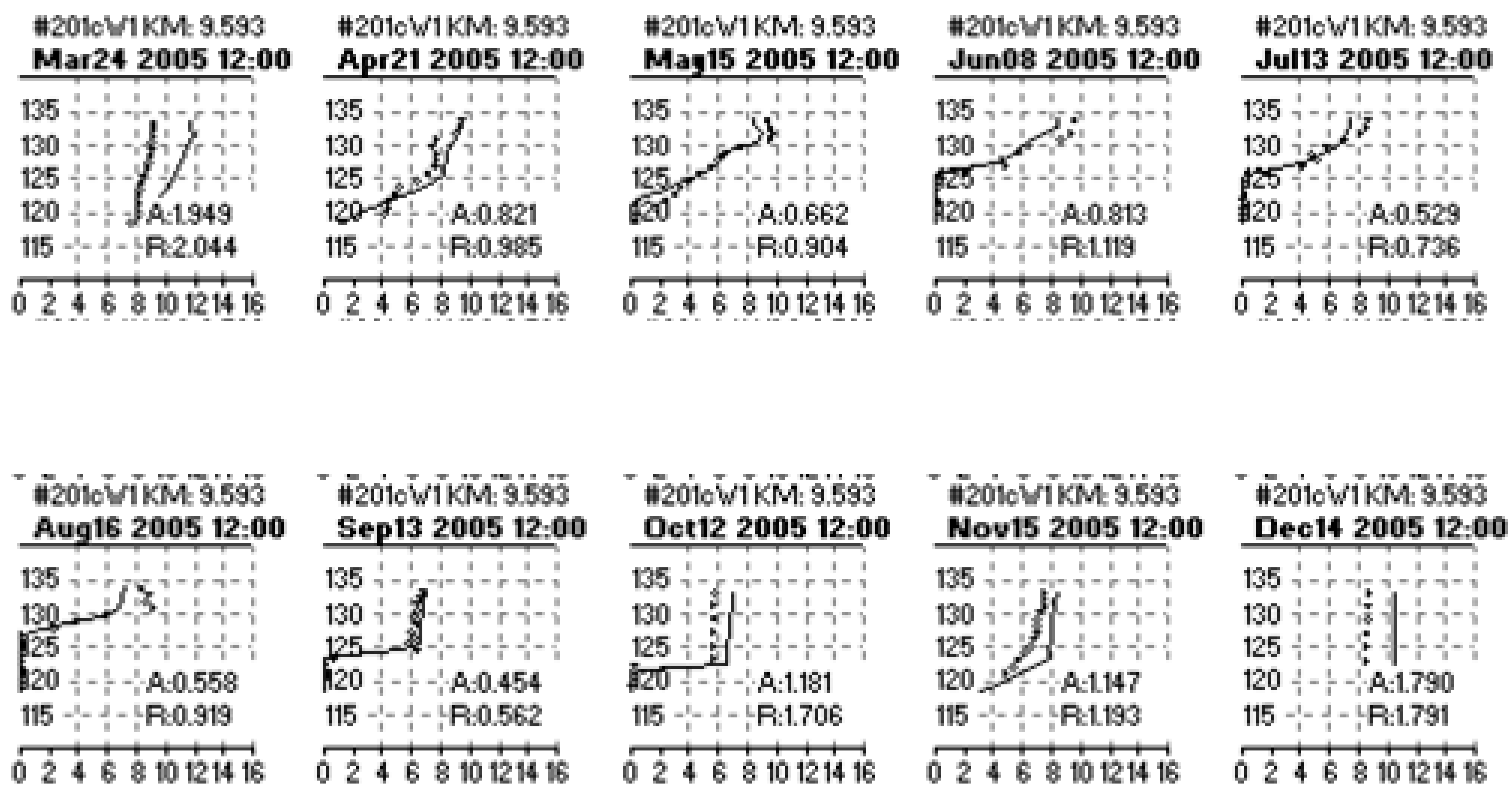

Fig. 9-b. Representative calibration plots for dissolved oxygen profiles in the lower lake (Sta=RND, rkm 9.6). Model output is shown in solid lines while observed data are in open circles. Calibration statistics include absolute mean error (A) and the root mean square error (R)

Table 3. Calibration statistics for temperature, oxygen, total phosphorus and chlorophyll-a at each calibration station.

Station	Distance from Dam (Km)	Absolute Mean Error* Temp. (degC)	D.O. (mg/L)	Surface TP (mg/L)	Bottom TP (mg/L)	Surface Chlorophyll-*a* (mg/L)
Upper Reedy Arm (RDY)	31.5	**	**	0.056	ns	0.0105
Middle Reedy (MRDY)	28.0	1.11	1.37	ns	ns	ns
Lower Reedy Arm (LRDY)	26.0	0.95	1.18	ns	ns	ns
Upper Saluda Arm (USAL)	35.0	**	**	0.045	ns	0.0061
Lower Saluda Arm (SBR)	29.2	0.95	1.08	0.049	0.041	0.0063
Mid-Lake (Hwy72)	23.0	0.87	1.11	0.024	0.037	0.0084
Mid-Lake (IRP)	16.0	0.85	1.13	ns	ns	ns
Lower Lake (RND)	9.7	0.75	0.93	0.024	0.086	0.0049
Lower Lake (GSP)	6.4	0.73	1.01	ns	ns	ns
Forebay (FBY)	1.6	0.82	0.96	0.020	0.066	0.0045
Overall		0.88	1.10	0.036	0.057	0.0068

* Absolute Mean Error (A)= $\sum$ | (Model-Data) | / n
** Stations not used for Temperature/DO calibrations
ns not sampled

Fig. 10 (a and b) illustrates seasonal snapshots of the combined spatial and temporal distributions of simulated DO distribution along the main axis of the lake for 2004 (a) and 2005 (b). Clearly, the extent of the spring-summer oxygen depletion in waters below the thermocline (red zones in Fig. 10) represents a significant portion of the habitat in Lake Greenwood. For baseline conditions, model computations for the average total annual volume and duration of extreme hypoxia (DO<1mg/L) in Lake Greenwood was 7.7 x 10^9 m^3-days, representing 9 % of the total annual volume. During the most severe periods of oxygen depletion (late August), 34% of the lake volume was hypoxic, representing a considerable portion of the lake with poor quality habitat for benthic invertebrates and fish.

DISSOLVED OXYGEN TRANSECTS
Baseline 2004

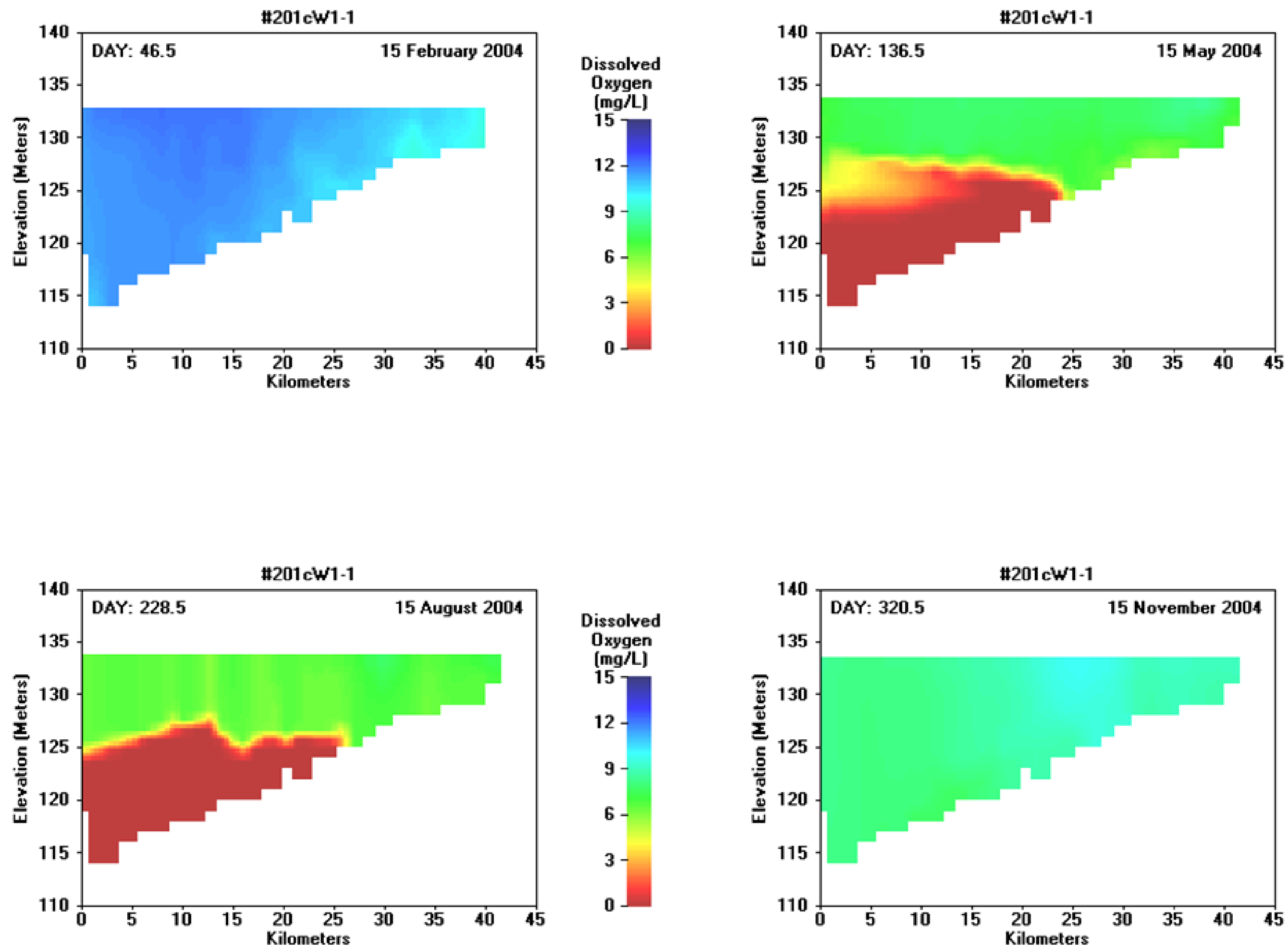

Fig. 10-a. Seasonal changes in spatial distributions of simulated dissolved oxygen in Lake Greenwood (2004)

DISSOLVED OXYGEN TRANSECTS
Baseline 2005

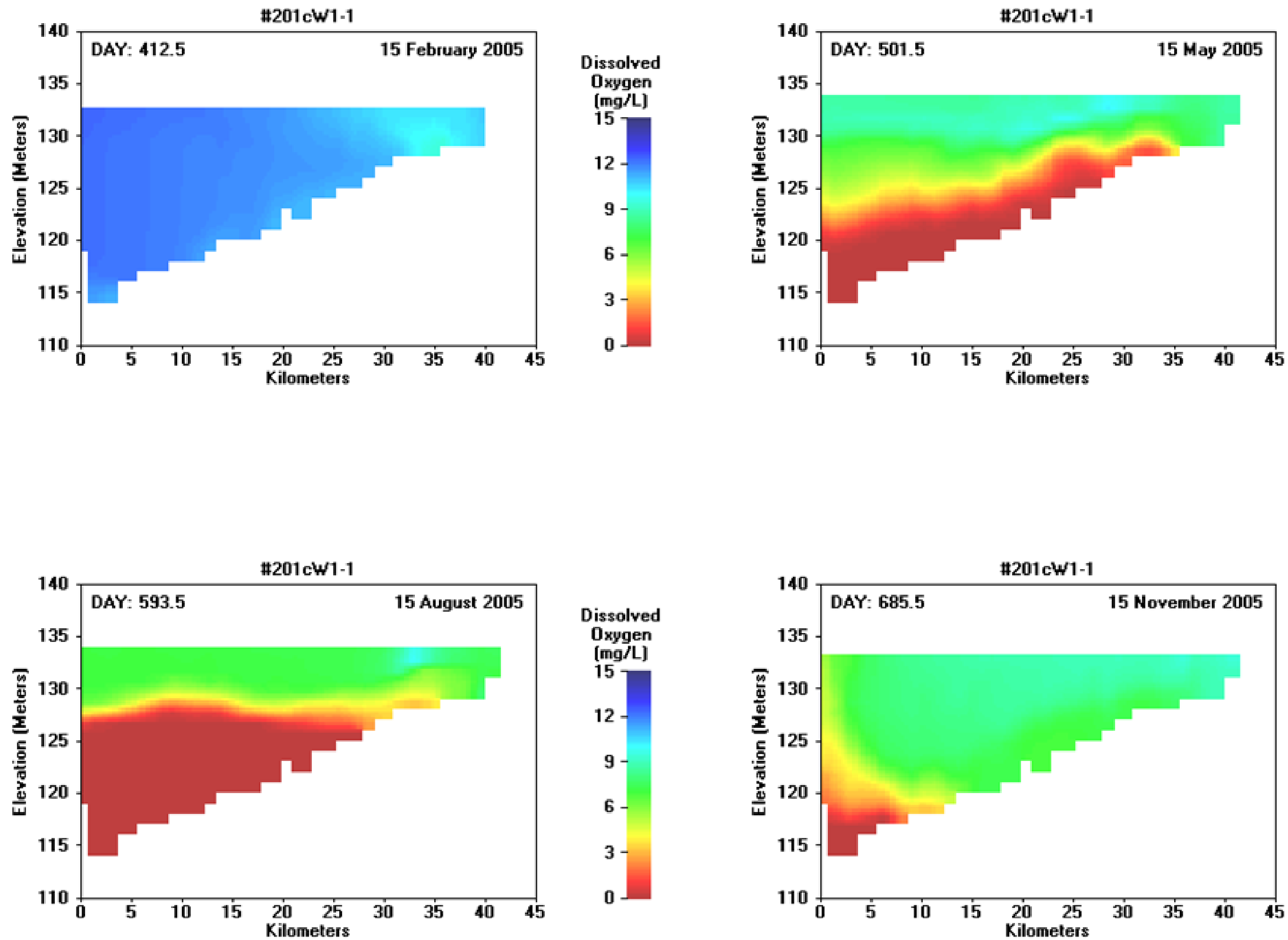

Fig. 10-b. Seasonal changes in spatial distributions of simulated dissolved oxygen in Lake Greenwood (2005)

Phosphorus

The calibrated model reproduced observed trends in total phosphorus (TP) with higher concentrations and greater variability in the upper and mid-lake sections than in the lower lake and forebay (Fig. 11). Concentrations in the upper and mid-lake sections exhibited annual means > 0.05 mg/L and frequently exceeded the 0.06 mg/L water quality standard for total phosphorus in piedmont lakes . The magnitude of the phosphorus peaks and the overall mean concentrations in the surface waters decreased with distance down lake from the upper arms to the lower-lake and forebay. Mean concentrations in the surface waters of the lower-lake and forebay (0.04 mg/L) were well below the water quality standard. The decreasing trend down lake was due in part to the sorption of bioavailable phosphorus onto suspended sediments and the subsequent settling of TP out of the surface waters. Recent modifications in CE-QUAL-W2 (Ruane and Hauser 2006) addressed this process specifically by including a phosphorus settling coefficient (PO4S), representing a settling velocity (m d^{-1}) of the adsorbed particulate phosphorus. For Lake Greenwood, a calibrated setting of PO4S=0.3 m d^{-1} was consistent with the observed decline in surface phosphorus from the upper to lower lake segments.

Another modification in recent applications of CE-QUALl-W2 allows variation in the phosphorus content between labile and refractory organic matter (Ruane and Hauser 2006). Since labile organic matter has a much higher rate of biological decay (0.08-0.10 d^{-1}, default values in Cole and Wells, 2002) than refractory organic matter (0.01 d^{-1}), variation in the phosphorus content of the two types of organic matter could have significant implications regarding phosphorus cycling in the lake. Whereas default values for the phosphorus content of organic matter is 0.5% (Cole and Wells 2002), we used 0.9% for labile organic matter and 0.09% for refractory organic matter, consistent with values used in recent applications on Catawba River reservoirs (Ruane and Hauser 2006).

The dominant trend in the vertical distribution of TP exhibited an accumulation of TP in the bottom waters at deeper stations (lower lake and forebay, Fig. 11). This pattern was due largely to the settling of TP from the upper layers (as described above) and to the release of phosphorus from anaerobic bottom sediments. From the onset of spring stratification, the model simulated a steady rise in bottom water TP to peak concentrations of 0.2-0.3 mg/L just before fall mixing. The data were more variable, with mid-to-late summer concentrations ranging from 0.1 to 0.5 mg/L. To simulate this process, the model included a sediment release rate (PO4R) that is modeled as a proportion of the sediment oxygen demand (SOD). For the Greenwood model, PO4R (2.5%) was estimated directly from in-lake measurements of sediment phosphorus release in the upper arms (0.046 ± 0.039 g m^{-2} d^{-1}, Deanhart 2007) and the documented SOD in the Reedy Arm (1.8 ± 0.2 g m^{-2} d^{-1}, Parsons 2005). Using this value throughout the lake, the model reproduced the general trend in bottom-water TP concentrations, although it did not match the extreme peaks observed during 2004 (Fig. 11). The absolute mean error in TP simulations for the 2-yr time period was 0.04 and 0.06 mg/L for surface and bottom water concentrations, respectively (Table 3).

TOTAL PHOSPHORUS

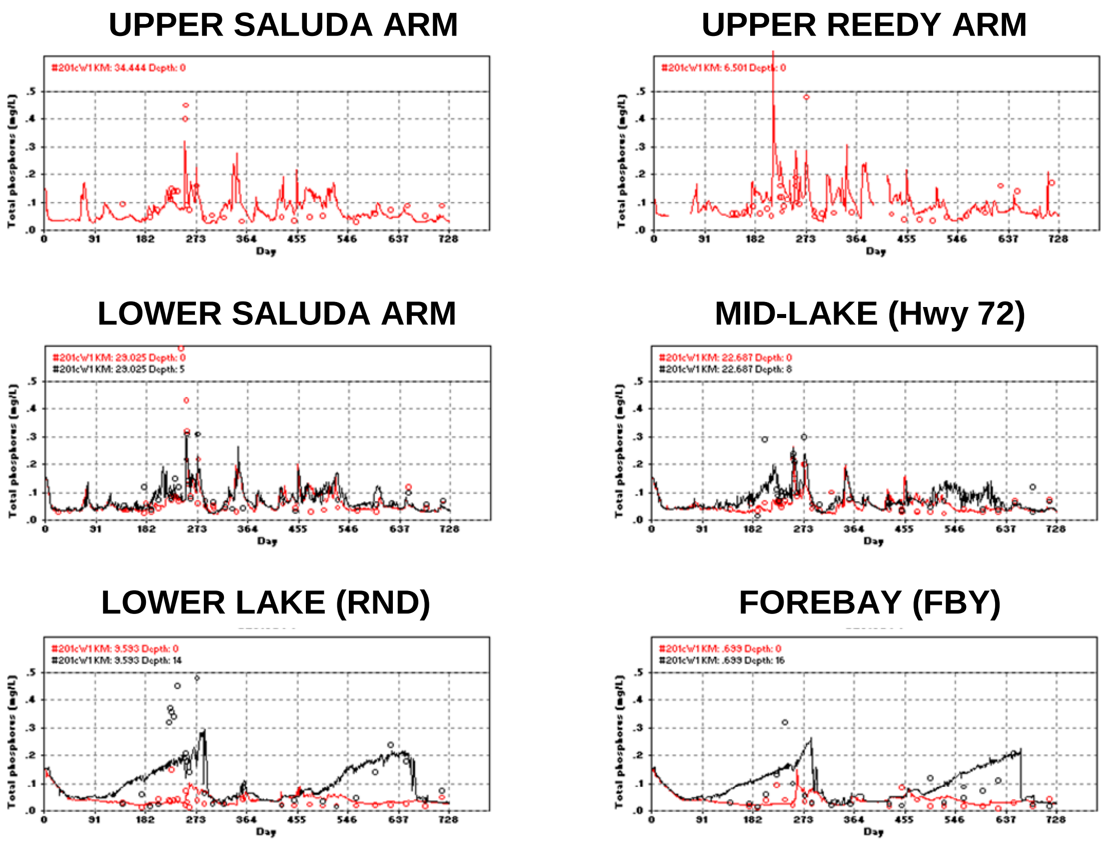

Fig. 11. Calibration plots for total phosphorus in Lake Greenwood. Model results are represented by solid lines (red for surface water and black for bottom water); data points are represented by open circles (red for surface and black for bottom).

Chlorophyll-*a*

The temperature ranges and growth rate coefficients for the 3 algal groups (diatoms, greens, and cyano-bacteria) represented sensitive parameters for chlorophyll-*a* calibration (Table 4). Although reported growth rates for algae range over 2 orders of magnitude (0.1-11.0 d^{-1}, Cole and Wells 2002), the calibrated value used in this model was near the median for all algal groups. The temperature ranges used in this model were the same as those used in recent modeling work on Lake Wateree, a similar sized reservoir on the Catawba chain of reservoirs (Ruane and Hauser 2006). With these settings, the model simulated the basic seasonal and spatial pattern of chlorophyll changes (Fig.12), with higher seasonal peaks (>0.03 mg/L) during the mid-to-late summer in the upper arms and mid-lake segments. The model provided a good match with chlorophyll along the main lake from the lower Saluda arm to the forebay (A=.005-.008 mg/L, Table 3)

Table 4. Growth rate and temperature ranges for algal productivity

		Algal Group		
Parameter	Units	Diatoms	Greens	Cyano-bacteria
Max. Growth Rate	(d^{-1})	2	2	2
Minimum Temp.	(^oC)	0	10	20
Optimum Temp. Range	(^oC)	15-22	20-35	28-35
Max. Temp	(^oC)	40	40	40

During initial calibrations, the model tended to under-predict mean concentrations in the upper arms , especially in the upper Saluda Arm. This model error was related, in part, to a potential under-estimate of upper lake volume in relation to the gauged river inflows. The inflow gauges were actually 8-10 km upstream from the lake headwaters and upper arms of the model, representing an additional volume and residence time that was not accounted for in the original model. This underestimate in upper lake volume effectively decreased the residence time in this part of the lake, causing rapid flushing and preventing a realistic algal growth response. To partially compensate for this effect, we adjusted the water volume in the Upper Saluda Arm by adding 2, 1-m surface layers to the first 8 segments. While this adjustment represented a negligible change in the total volume of the lake (no effect on the water budget and water level calibration, Fig. 7), the change effectively slowed the simulated water transport in this section, increased the water residence time, and resulted in more accurate simulated algal peaks in the upper Saluda arm. However, while this adjustment improved the simulated chlorophyll peaks, future applications of the model should specifically address this issue by including additional segments of river/reservoir environment between the gaging stations and the upper boundaries of the lake model (see “MODEL LIMITATIONS, p.34).

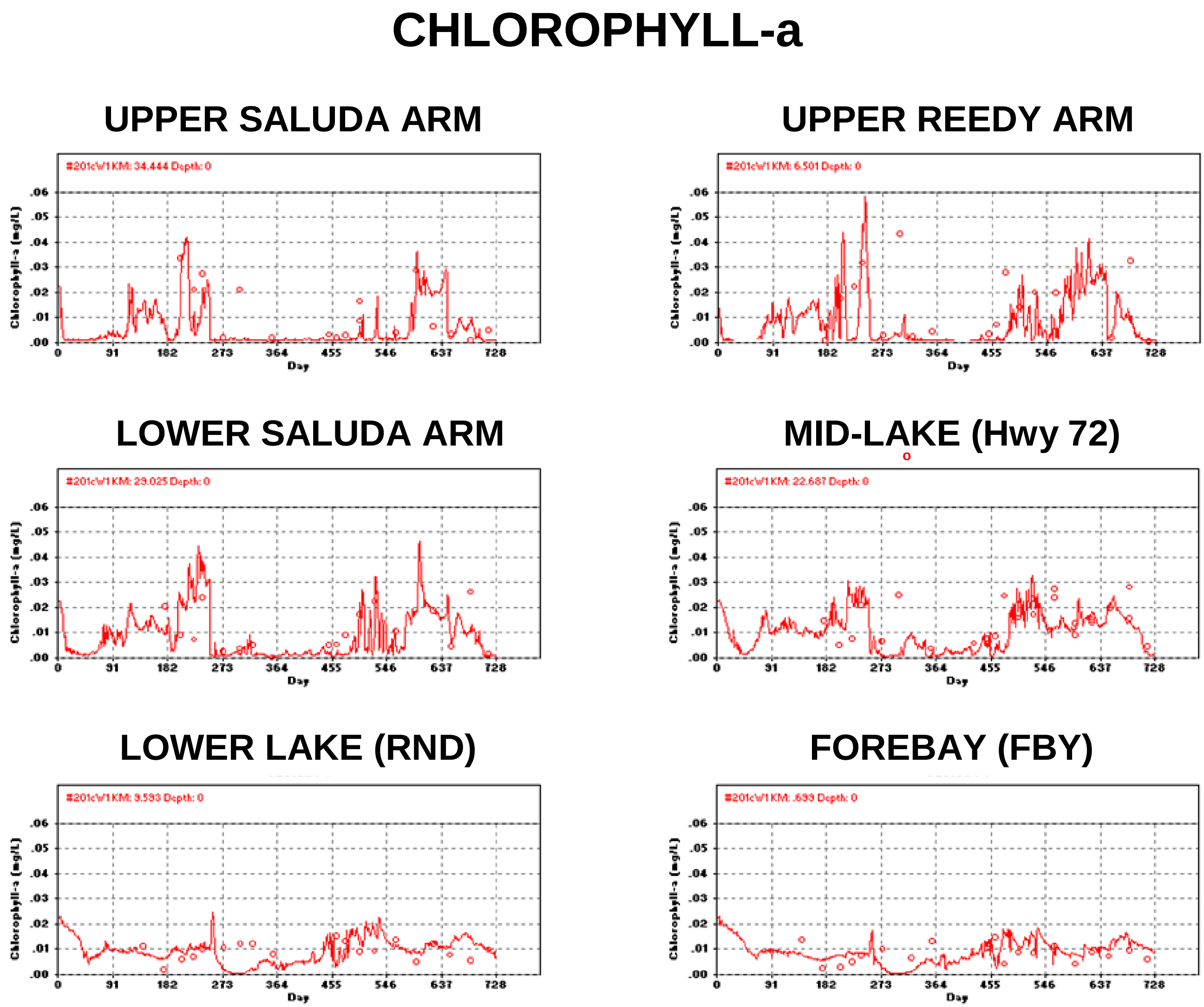

Fig. 12. Calibration plots for surface chlorophyll-*a* in Lake Greenwood. Model results are represented by solid lines; data points are represented by open circles.

The model also simulated the basic pattern of seasonal change in algal community composition with diatoms dominating during cooler months and cyanobacteria dominating during the warmer months (Fig. 13). These general patterns were similar to those reported in other lakes (Wetzel 1983), and confirmed in Lake Greenwood by pigment marker analysis (Lewitus et al. 2005). The model predicted an overall annual mean dominance of diatoms (53%), followed by green algae (27%), and cyanobacteria (21%). The field data based on pigment-markers (Lewitus et al. 2005) suggested a more even distribution among the three algal groups with diatoms at 37%, cyanobacteria at 34%, and greens at 29%). Further calibration of algal production coefficients and temperature ranges may be useful for more detailed analysis of the algal community changes in Lake Greenwood.

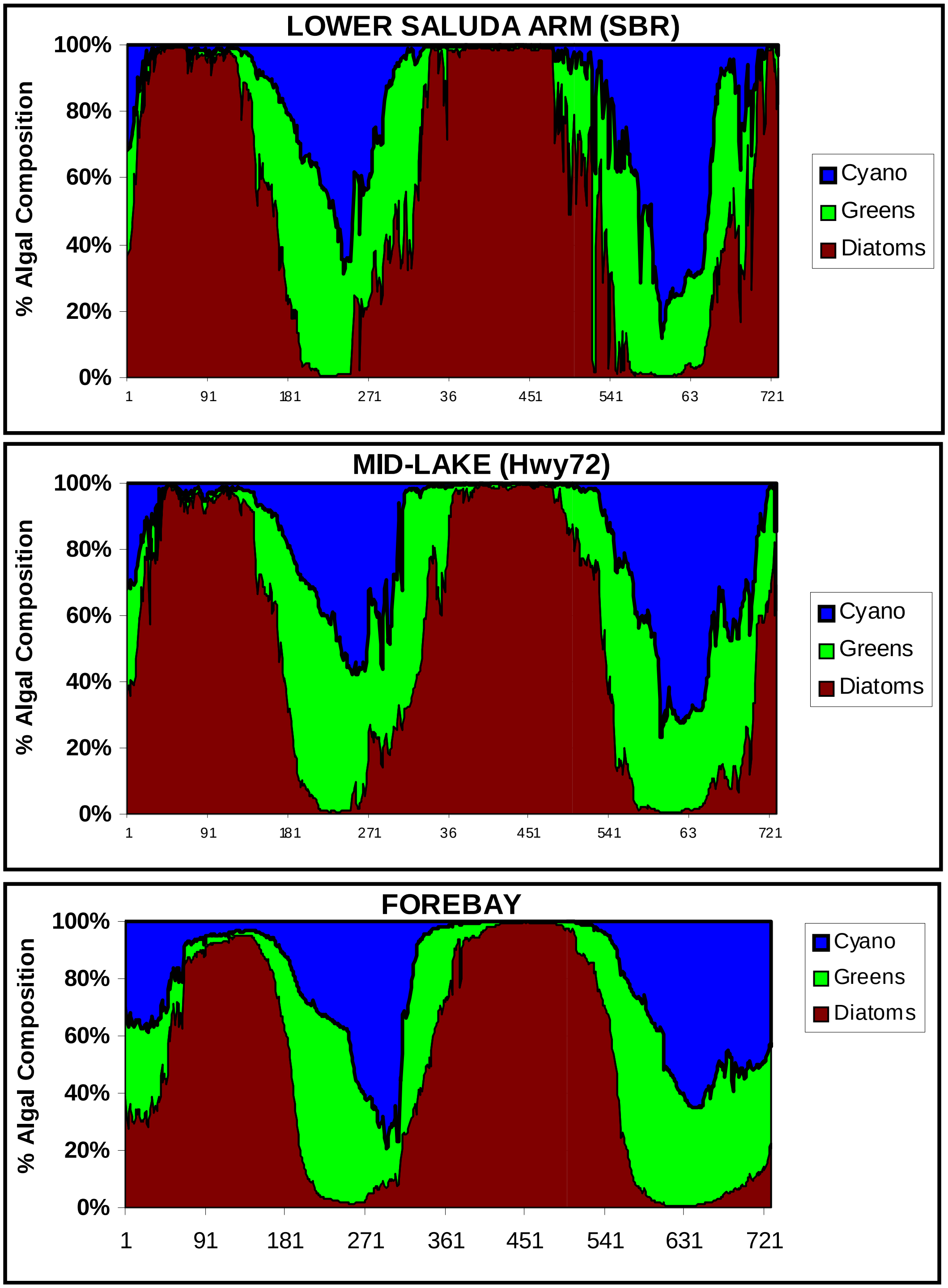

Fig. 13. Simulated changes in algal community composition in the Lower Saluda Arm, Mid-Lake, and the Forebay.

APPLICATIONS TO PHOSPHORUS LOADING

Phosphorus Loading Scenarios

After satisfactory calibration, we conducted a series of simulations aimed at predicting the consequences of changing total phosphorus (TP) inputs to Lake Greenwood. With the current efforts of SCDHEC to establish a total maximum daily load (TMDL) for phosphorus in the Reedy River, our first scenarios focused on phosphorus changes in the Reedy River only. In order to cover a wide range of conditions for phosphorus loading, we included 25% increase, followed by a 25%, 50%, 75%, and 90% reduction in the Reedy River inflow phosphorus concentrations (Table 5). This initial set of scenarios was then followed by a series of simulations representing similar proportional changes in both the Saluda and Reedy River inputs. While loading from minor tributaries and lake-side drainage (distributed flows) are potentially important to issues of total loading, they were not altered in these scenarios.

Total P in the in the model was represented by several functional components including bioavailable inorganic P, as well as the phosphorus contained in dissolved and particulate organic matter. For each scenario total P loading was changed by simultaneous proportional changes in the concentrations of all inflowing phosphorus components (Table 5). In addition to the direct effects of phosphorus loading on water quality in the lake, we also included the longer-term indirect effects on sediment oxygen demand (SOD) in each scenario. As described in the Model Calibration section, the effects of P loading on SOD are modeled as two related processes. The 1st process represents a direct 1st-order decomposition of recently settled organic matter. The second process addresses indirect effects of longer-term changes in SOD due to subsequent changes in sediment organics and phosphorus. This process is treated as a time-constant (zero-order SOD, g m^{-2} d^{-1}) that may vary spatially between model segments. Recent applications of the CE-QUAL-W2 to the Catawba River reservoirs (Ruane and Hauser 2006) used the basic framework in several related models which invoke a "square-root" relationship between SOD and the loading of organic matter and phosphorus (DiToro et al 1990, Chapra and Canale 1991, Chapra 1997). Following Ruane and Hauser (2006), we computed the zero-order SOD rate for each model segment as

$$SOD.= SOD_c \ (L/L_c)^{0.5}$$

where SOD was the segment-specific SOD (g m^{-2} d^{-1}) for the selected scenario, SOD_c was the SOD in the baseline calibration, L was phosphorus load for the selected scenario and L_c is the phosphorus load of the baseline calibration. For example, a 50% reduction in phosphorus loading would result in a 30% decrease in SOD. However, there is some uncertainty regarding the time scale of this response and some sensitivity analysis for this function is recommended for future applications of the model to phosphorus load reductions.

For baseline conditions, the total P load from the Saluda and Reedy Rivers (1.41 x 10^5 kg/yr) was dominated by the Saluda River (72%), although the mean concentration in the Reedy (0.11 mg/L) was 10% higher than in the Saluda (0.10 mg/L; Table 5). The larger flows and relatively high P concentrations in the Saluda combined to dominate the phosphorus load to Lake

Table 5. Phosphorus mass loading and mean inflow concentrations in loading scenarios for the Saluda and Reedy Rivers. Total P load was adjusted by changing the riverine concentrations of inorganic P (PO_4), labile and refractory dissolved organic matter (LDOM, RDOM), and labile and refractory particulate organic matter (LPOM, RPOM). [1, 2].

	TOTAL PHOSPHORUS LOADING						
	Mass Load (10^5kg/yr)			Mean Inflow Conc. (mg/L)		Percentile Inflow Conc. > 0.06 mg/L	
SCENARIO DESCRIPTION	Saluda	Reedy	Total	Saluda	Reedy	Saluda	Reedy
BASELINE CALIBRATION	1.02	0.39	1.41	0.10	0.11	58.8	73.7
REEDY LOADING							
25% Increase	1.02	0.48	1.50	0.10	0.14	58.8	94.1
25% Reduction	1.02	0.29	1.31	0.10	0.08	58.8	55.9
50% Reduction	1.02	0.20	1.22	0.10	0.06	58.8	30.6
75% Reduction	1.02	0.10	1.12	0.10	0.03	58.8	5.8
90 % Reduction	1.02	0.04	1.06	0.10	0.01	58.8	0.3
SALUDA AND REEDY LOADING							
25% Increase	1.27	0.48	1.75	0.12	0.14	73.9	94.1
25% Reduction	0.77	0.29	1.06	0.07	0.08	45.5	55.9
50% Reduction	0.51	0.20	0.71	0.05	0.06	27.9	30.6
75% Reduction	0.26	0.10	0.36	0.03	0.03	4.3	5.8
90 % Reduction	0.11	0.04	0.15	0.01	0.01	0.0	0.3

[1] The phosphorus content of riverine algae was also included in the total phosphorus load although it represented < 2% of the total input. Riverine algal biomass was not changed in this series of scenarios.

[2] Other input concentrations that could be potentially related to phosphorus loading (such as total suspended solids and nitrogen fractions) were not changed in these scenario runs.

Greenwood. For these conditions, the total P concentrations in the inflowing water from both rivers exceeded the water quality standard for piedmont lakes (0.06 mg/L) most of the time (59% and 74% in the Saluda and Reedy Rivers, respectively). A 25% increase in the inflowing phosphorus concentration in the Reedy would lead to water quality contraventions of the phosphorus standard 94% of the time (Table 5).

With these loading patterns, the model suggested that it would require > 50% reduction in phosphorus concentrations in the inflows to reduce mean concentrations below the 0.06 mg/L standard and to reduce water quality contraventions below 10%. A 50% reduction in both rivers would lower the mean inflow concentration to levels near the 0.06 mg/L standard, although 27-30% of the daily inflows would still exceed the phosphorus standard (Table 5). A 50% reduction in the Reedy alone would represent only a 14% reduction in total loading to the lake, suggesting that phosphorus reductions in both rivers will probably be necessary for lake-wide improvements in water quality. If nonpoint-source runoff controls (Best Management Practices) in both basins could be focused on storm flow peaks, the contravention rate for phosphorus could possibly be reduced to acceptable levels with a 50% reduction in loading from both rivers. Further reductions in the phosphorus loading in these piedmont streams would probably be impractical. For example, a 90% reduction would reduce the inflow concentrations near the lower limits of analytical detection (Table 5), which would be more characteristic of pristine mountain streams than piedmont rivers.

Phosphorus Distributions

As presented in the "Calibration" section, the model adequately reproduced baseline conditions of phosphorus distribution in the lake surface waters, with a steady decline from the upper arms (0.07-.09 mg/L), to mid-lake (0.05 mg/L), and forebay (0.04 mg/L, Table 6). The higher concentrations in the bottom waters at mid-lake and forebay reflected phosphorus sedimentation from the surface waters as well as phosphorus release from anaerobic sediments. The difference between surface and bottom water concentrations was more pronounced in the lower lake sections which was deeper and more stratified. The model predicted annual mean phosphorus concentration in the forebay bottom water to be 2.2 times the surface water concentration (Table 6). The overall significance of this release of phosphorus from anaerobic bottom sediments (internal loading) to the eutrophication of the lake is still unclear and could be the focus of future simulation tests for the model. As established in the earlier research (Larsen et al. 1981, Chapra and Canale 1991), phosphorus release from benthic sediments can significantly delay the response of lake algae and productivity to managed reductions in "external" loading. However, strong stratification in deeper lakes (as in the lower sections of Lake Greenwood) may inhibit the vertical transport of high bottom water concentrations to photic surface waters during the algal growing season, thereby diminishing the biological impacts of sediment phosphorus release(Welch and Cooke 2005).

Reductions in the simulated phosphorus load from the Reedy and Saluda Rivers resulted in roughly linear reductions in the predicted phosphorus distribution in the lake (Table 6) with a more sensitive response (steeper slope) in the upper arms of the lake. With changes in the Reedy River alone, a 50% reduction in load resulted in a 44% reduction in the annual mean concentration in the upper Reedy Arm. This level of load reduction resulted in a negligible (<0.01 mg/L) reduction in surface water concentrations at mid-lake and forebay, reflecting the major influence of the Saluda River load on the main body of the lake.

A substantial reduction in the phosphorus concentrations throughout the lake proper would require significant load reductions in both the Saluda and Reedy rivers. A 50% reduction in loading from both rivers would reduce annual mean concentrations to levels less than 0.06 mg/L throughout the lake surface waters. This level of load reduction would decrease mean concentrations in upper arms, mid-lake, and forebay by 44%, 40%, and 25% respectively (Table 6).

Bottom water concentrations were somewhat more sensitive to load reductions than the surface waters. The greater sensitivity in the bottom water was due to the combined effects of (1) reduced sedimentation/decomposition of organic matter from surface waters and (2) reduced SOD with a corresponding reduction in the release of phosphorus from the benthic sediments. Therefore, reductions in phosphorus loading from the Saluda and Reedy Rivers could have a secondary benefit by also reducing the internal phosphorus loading.

Table 6. Predicted distributions of annual mean total phosphorus concentration in Lake Greenwood. "S" and "B" refer to surface and bottom waters at the Mid-Lake and Forebay stations.

	TOTAL PHOSPHORUS DISTRIBUTION ANNUAL MEAN (mg/L)					
	Upper Saluda	Upper Reedy	Mid-Lake [1]		Forebay	
SCENARIO DESCRIPTION	Arm	Arm	(S)	(B)	(S)	(B)
BASELINE CALIBRATION	0.07	0.09	0.05	0.07	0.04	0.09
REEDY LOADING						
25% Increase	0.07	0.11	0.06	0.08	0.04	0.10
25% Reduction	0.07	0.07	0.05	0.07	0.04	0.09
50% Reduction	0.07	0.05	0.05	0.06	0.04	0.08
75% Reduction	0.07	0.03	0.05	0.06	0.04	0.08
90 % Reduction	0.07	0.02	0.04	0.06	0.03	0.08
SALUDA AND REEDY LOADING						
25% Increase	0.09	0.11	0.06	0.09	0.05	0.11
25% Reduction	0.05	0.07	0.04	0.06	0.03	0.08
50% Reduction	0.04	0.05	0.03	0.04	0.03	0.06
75% Reduction	0.02	0.03	0.02	0.03	0.02	0.04
90 % Reduction	0.01	0.02	0.02	0.02	0.02	0.03

[1] Station at Hwy72 bridge, 23 km upstream from dam

Chlorophyll-*a* Distributions

For baseline conditions, the model simulated the highest chlorophyll peaks in the Upper Reedy Arm, yielding a 13.2% exceedence rate of the 40 µg/L chlorophyll standard (Fig 14. Table 7), indicating a potential for nuisance algal blooms in this area of the lake. These predicted patterns for planktonic chlorophyll concentrations may also correlate to growth patterns for attached filamentous algae which caused nuisance blooms in the Reedy Arm during the extended drought of 1999.

A 25% increase in loading from the Saluda and Reedy would lead to a 20 % exceedence rate for chlorophyll in the upper Reedy and > 5 % exceedence rate in the Saluda Arm. (Fig. 14, Table 7). According to the model, it would require a 25% reduction in phosphorus loading in the Saluda and a 50% reduction in the Reedy to eliminate sustained chlorophyll exceedences in Lake Greenwood (Table 7).

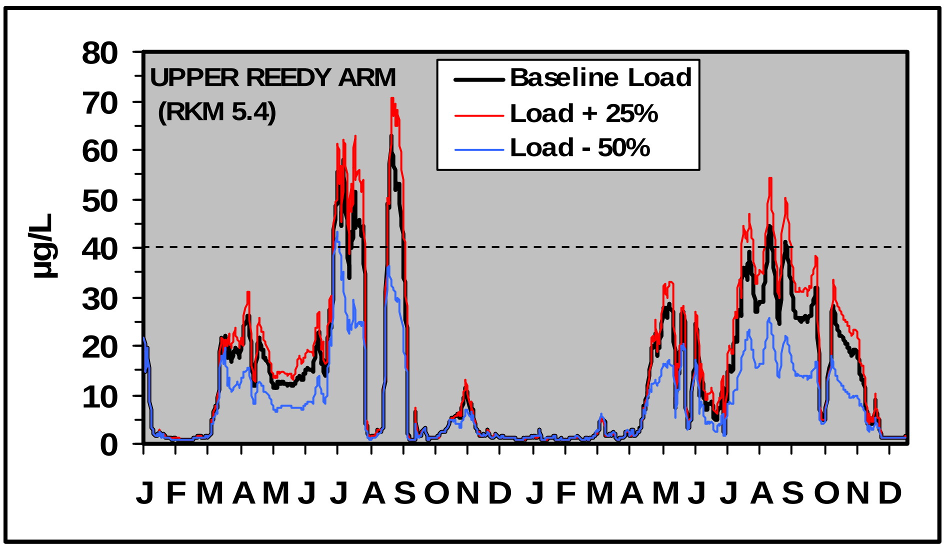

Fig. 14. Predicted chlorophyll-a distributions in the Upper Reedy Arm as a function of phosphorus loading (baseline load, baseline+25% for Saluda and Reedy, and baseline – 50% for the Saluda and Reedy)

Table 7. Predicted distributions of chlorophyll-*a* during the growing season in Lake Greenwood. Mean = average daily concentration during growing season (May--October), Max=maximum daily concentration, %>40=portion of daily concentrations exceeding the water quality standard (40 µg/L). Selected stations for the upper arms are those which exhibited consistent chlorophyll peaks in the model. Mid-Lake and Forebay stations are the same locations as in Table 2. Shaded cells indicate predicted maximum concentrations > 40 µg/L which are sustained for time periods > 5% of the growing season

	CHLOROPHYLL-a DISTRIBUTION (MAY-OCT); µg/L											
	Lower Saluda Arm (RKm 29.2)			Upper Reedy Arm (RKm 33.1)			Mid-Lake (RKm 22.7)			Forebay (RKm 0.7)		
SCENARIO DESCRIPTION	Mean	Max	%>40	Mean	Max	%>40	Mean	Max	%>40	Mean	Max	%>40
BASELINE CALIBRATION	13.8	46.4	1.7	20.6	62.6	13.2	14.2	32.9	0.0	8.9	18.5	0.0
REEDY LOADING												
25% Increase	14.2	46.8	1.8	23.7	69.8	20.2	15.0	33.7	0.0	9.3	19.1	0.0
25% Reduction	13.4	46.2	1.4	17.2	50.2	6.7	13.4	32.1	0.0	8.5	17.9	0.0
50% Reduction	12.9	46.1	0.9	13.4	43.7	1.2	12.5	31.0	0.0	8.0	17.5	0.0
75% Reduction	12.4	46.3	0.6	9.1	36.7	0.0	11.6	30.1	0.0	7.6	17.0	0.0
90 % Reduction	12.1	46.5	0.6	6.2	32.0	0.0	10.9	29.6	0.0	7.3	16.6	0.0
SALUDA AND REEDY LOADING												
25% Increase	16.1	52.5	5.2	24.1	70.5	20.7	16.8	37.3	0.0	10.4	21.4	0.0
25% Reduction	11.3	36.5	0.0	16.8	49.9	6.3	11.6	28.7	0.0	7.4	15.3	0.0
50% Reduction	8.6	25.9	0.0	12.6	43.2	0.5	8.9	24.6	0.0	5.8	12.6	0.0
75% Reduction	4.3	16.0	0.0	7.8	35.8	0.0	5.7	19.6	0.0	4.0	10.1	0.0
90 % Reduction	2.9	10.3	0.0	4.6	28.2	0.0	3.5	15.9	0.0	2.8	8.3	0.0

Oxygen Depletion and Biotic Habitat

A persistent feature in the water quality of Lake Greenwood water quality has been a rapid springtime depletion of dissolved oxygen below the thermocline, creating a volume of hypoxic/anaerobic bottom waters through the summer and into the fall. Although a summer hypolimnetic oxygen decline is a natural condition in stratified lakes, the extent (volume) and duration may be a function of high nutrient loading, algal blooms, and subsequent decomposition of excess organic matter. Therefore, changes in the phosphorus load to Lake Greenwood may affect the extent and/or duration of hypoxic conditions and viable biotic habitat for a large variety of aquatic organisms.

The stocking and management of striped bass in Lake Greenwood represents a special problem since these fish seek cool, well oxygenated waters (preferably $T < 20^{o}C$ and oxygen > 4 mg/L) which may be severely limited during the summer. However, these fish are able to tolerate somewhat more extreme conditions and may survive and propagate as long as there is sufficient duration and volume of water within tolerable limits ($T < 28^{o}C$ and oxygen levels > 2 mg/L). The reduction in suitable temperature and oxygen concentration is often referred to as the habitat "squeeze", where the depth range of preferred or tolerable conditions is narrowed because of high temperatures in the surface waters and low oxygen in the cooler, deeper waters. We used the model to predict the volume and duration of overall hypoxic conditions and tolerable striped bass habitat in Lake Greenwood as a function of phosphorus loading.

The model accounted for detailed changes in vertical distribution of dissolved oxygen along the entire longitudinal axis of the lake, clearly illustrating a reduction in hypoxic conditions with a 50% reduction in phosphorus loading (Fig. 15 a and b, Table 8). With a 50% load reduction in the Reedy River alone, the model predicted < 10% decrease in annual hypoxic conditions (vol-days/yr) and < 5% increase in the May-Oct habitat for striped bass (Table 8). However, a 50%-reduction in loading from both rivers would decrease hypoxia in the lake by 30%, and increase the May-Oct. volume of striped bass habitat by almost 10% (Fig. 16, Table 8). However, the suitability of the lake environment for striped bass may depend somewhat on the extent of tolerable habitat during the most severe periods of oxygen depletion. A 50% reduction in the phosphorus load from both rivers would result in a 14 % increase in tolerable striped bass habitat during the peak of oxygen depletion (Table 8).

The 50% level of load reduction appears to be a transition point in lake response from a relatively small impact (<5%) to considerable impact on hypoxia and habitat (Fig 16, Table 8). Further reductions in phosphorus loading could enhance striped bass habitat (in terms of suitable DO distributions) but would need to be evaluated in terms of the potential reduction in forage productivity and the total fishery. Several researchers have demonstrated a direct correlation between phosphorus concentration, algal biomass, forage fish production and sport fishery (Ney et al. 1990, Maceina and Bayne 2001) although the biomass of forage and sport fish tends to decline in hyper-eutrophic conditions (Kautz 1982). Clearly, the management of nutrient loading needs to be carefully evaluated in terms of the resulting balance between fishery production and habitat suitability.

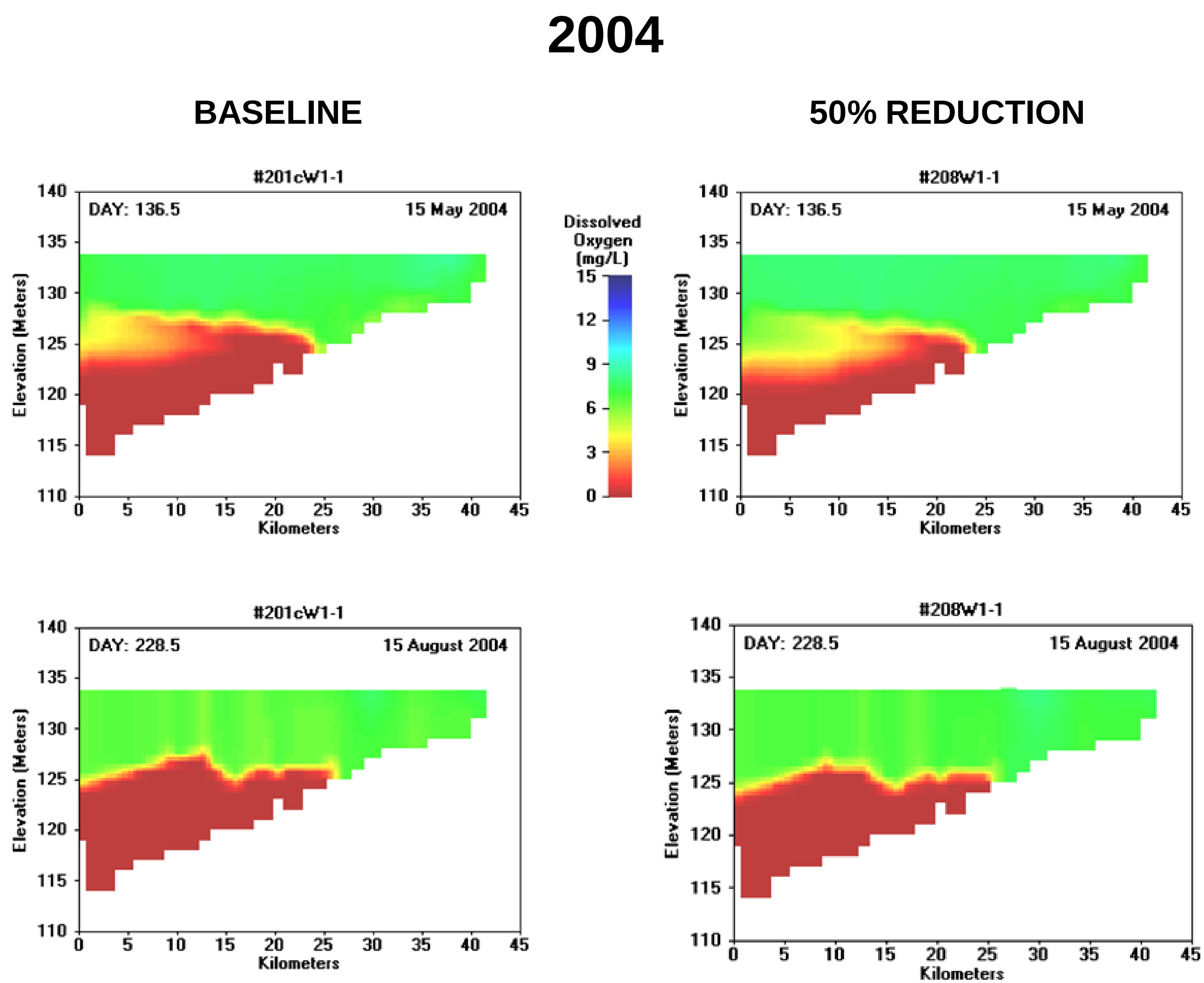

Fig. 15-a. Simulated vertical and longitudinal distributions of dissolved oxygen in Lake Greenwood in May and August during 2004 hydrologic conditions. The left-hand panels illustrate baseline conditions; the right-hand panels illustrate results of a 50% reduction in phosphorus load in the Saluda and Reedy Rivers.

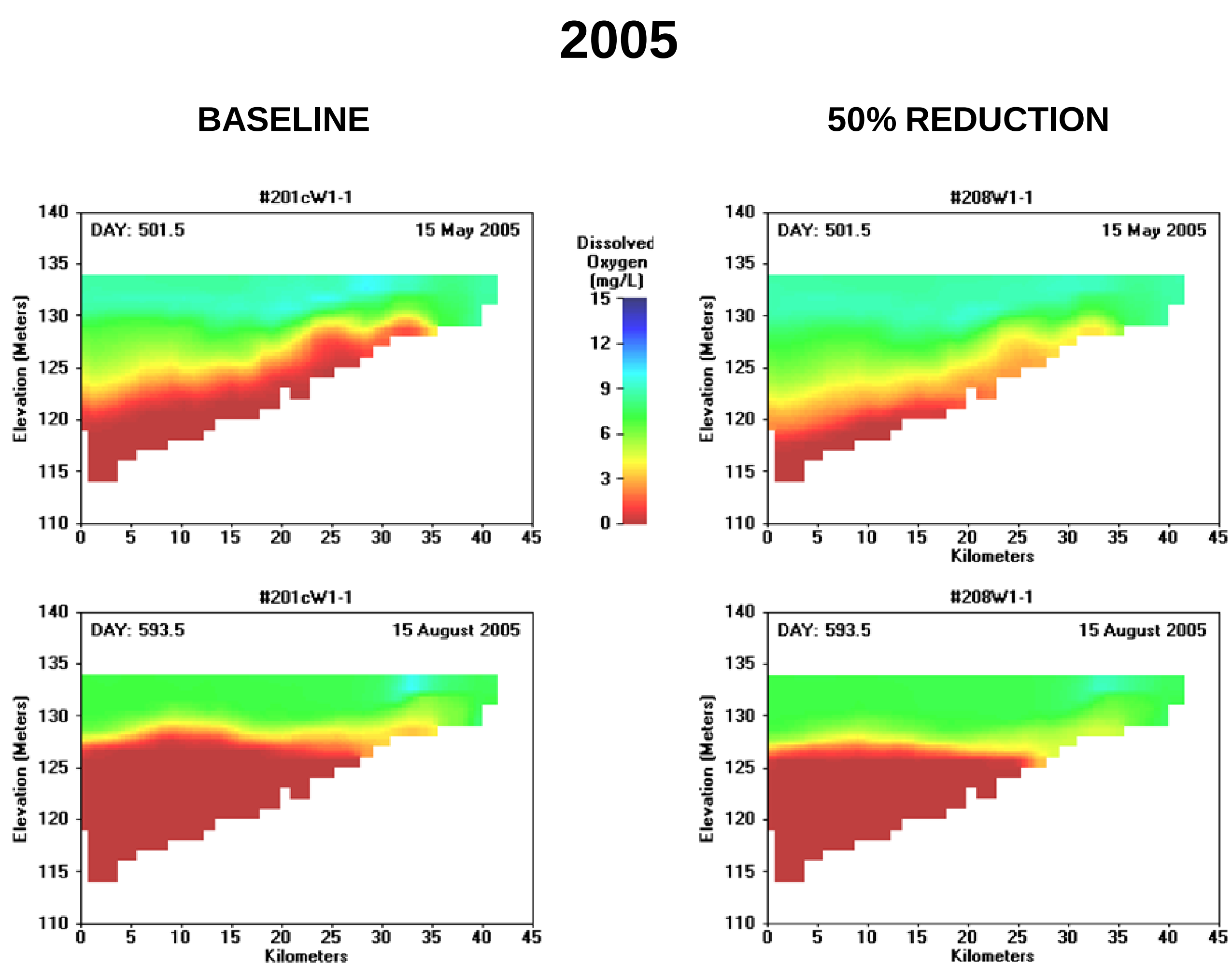

Fig. 15-b. Simulated vertical and longitudinal distributions of dissolved oxygen in Lake Greenwood in May and August during 2005 hydrologic conditions. The left-hand panels illustrate baseline conditions; the right-hand panels illustrate results of a 50% reduction in phosphorus load.

Table 8. Extent of extreme hypoxia and tolerable habitat for striped bass as a function of phosphorus loading in Lake Greenwood. "Total Vol-Day" values indicate the integrated volume and duration of these conditions. The "One-Day Max or Min" values indicate the volume and % of the total lake volume of hypoxic water and habitat during extreme conditions of oxygen depletion.

	EXTREME HYPOXIA (DO < 1 mg/L)				STRIPED BASS HABITAT (May-Oct) (Temp<28°C and DO>2mg/L)			
SCENARIO DESCRIPTION	Total Vol-Day/Yr ($10^9 m^3 da$)[a]	Total Vol-Day/Yr (%Tot)	One-Day Max Vol (10^6 m^3)	One-Day Max Vol (%Tot)	Total Vol-Day ($10^9 m^3 da$)[b]	Total Vol-Day (%Tot)	One-Day Min Vol (10^6 m^3)	One-Day Min Vol (%Tot)
BASELINE CALIBRATION	7.7	8.9	85.6	33.6	25.3	55.1	1.4	0.6
REEDY LOADING								
25% Increase	8.0	9.2	87.9	34.5	25.1	54.5	1.4	0.6
25% Reduction	7.4	8.5	83.1	32.6	25.7	55.8	1.4	0.6
50% Reduction	7.0	8.1	78.5	30.6	26.1	56.6	1.4	0.6
75% Reduction	6.6	7.6	76.2	29.9	26.5	57.5	1.4	0.6
90 % Reduction	6.4	7.3	73.5	28.8	26.8	58.3	1.5	0.6
SALUDA AND REEDY LOADING								
25% Increase	8.7	10.0	93.3	36.6	24.5	53.3	1.0	0.4
25% Reduction	6.6	7.6	75.5	29.6	26.4	57.4	1.5	0.6
50% Reduction	5.3	6.1	60.8	24.3	27.8	60.4	1.6	0.6
75% Reduction	3.4	3.9	44.4	17.5	29.9	65.1	7.2	2.9
90 % Reduction	1.7	1.9	29.6	11.8	32.1	69.7	17	6.8

[a] Computed as the sum of the total integrated volume-days for the entire 2-yr simulation divided by 2 to yield the average vol-days/yr

[b] Computed as the sum of the total integrated volume-days for the entire 2-yr simulation divided by 2 to yield the average vol-days for the season (May-Oct)

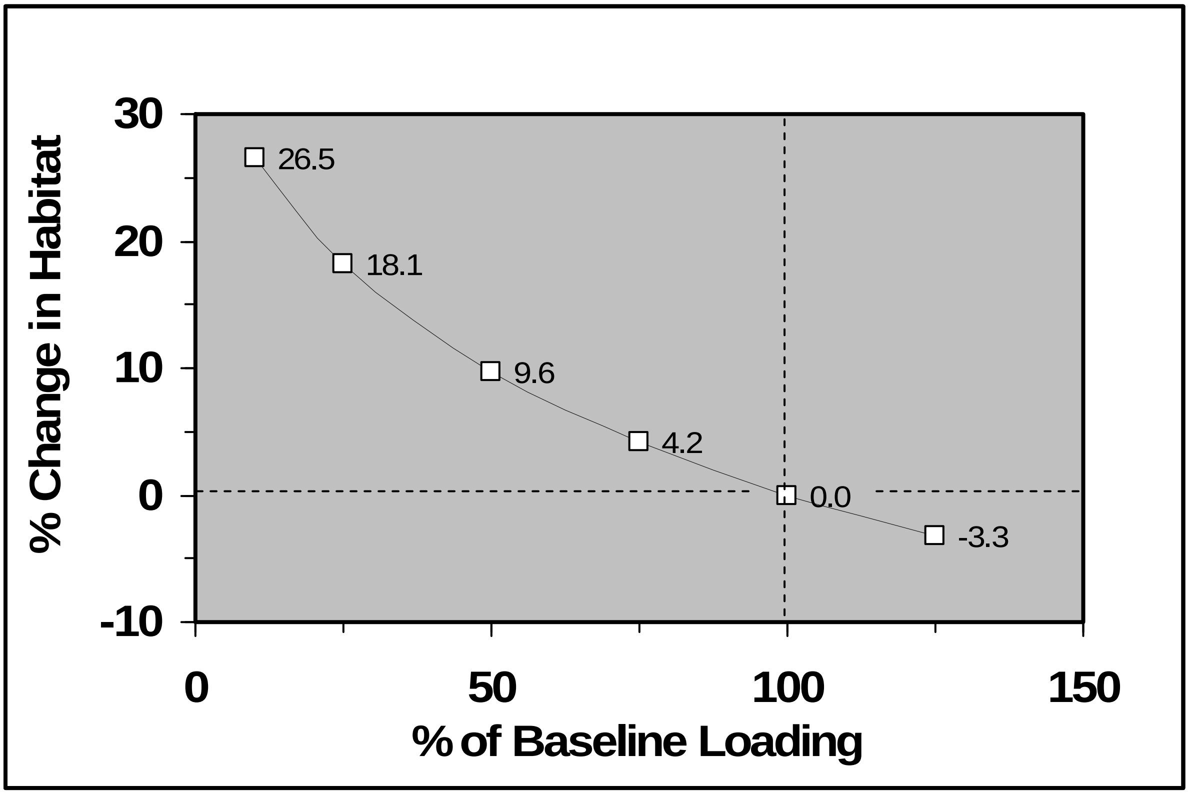

Fig. 16. Relative changes in tolerable habitat for striped bass (May-Oct) as a function of changes in the phosphorus loading from the Saluda and Reedy Rivers.

MODEL LIMITATIONS

As documented in this report, the model is calibrated to simulate phosphorus, chlorophyll, and oxygen dynamics in the main axis of the lake from the upper arms to the forebay. However, additional calibration and development would be needed for confident application in a few special locations within the lake and during periods of extended drought.

Upper Saluda Arm. As described in the calibration section (p. 21), the model under-predicted algal growth and chlorophyll concentrations in the upper sections of the Saluda arm of the lake, although the model performed well for the lower Saluda Arm, the Reedy Arm, and the main lake proper (Fig. 12). After experimental adjustment of the model bathymetry in this section of the lake, we inferred that this limitation was related to a possible under-estimate of depth and/or volume in this section of the lake. An under-estimate of storage volume in this shallow section of the lake could decrease the simulated residence time, preventing adequate response time for the simulated algal growth in the upper lake. For more accurate simulations in he upper Saluda Arm, we recommend additional focus on the bathymetry of this part of the lake. We also recommend the inclusion of additional model segments representing the portion of the Saluda River between the upper Saluda Arm of the lake and the position on the river where river inflow to the lake is gauged (Ware Shoals, 8 km upstream). This additional volume would decrease the simulated flushing rate in the upper lake and perhaps provide adequate residence time in the model to reproduce the observed algal growth in this area.

Isolated Tributary Embayments. Similar uncertainties exist about the accuracy of the current model calibration in isolated tributary embayments such as in Hidden Lake on the Cane Creek arm. The Cane Creek arm remains on SCDHEC's list of impaired waters because of problems with phosphorus loading and oxygen depletion,(scdhec.net/environment/water/docs/06_303d.pdf), and the model could be very useful for future management plans for this sub-basin and similar habitats. However, as in the case of the upper Saluda arm, water quality dynamics in these habitats is affected considerably by water flow, storage volume, and flushing rates as well as contaminant loading. Although there is adequate information on input water quality (SCDHEC monitoring) and in-lake water quality dynamics for Cane Creek arm (Appendix A), there is little detailed information on embayment bathymetry and water flow in the Cane Creek tributary. For confident applications of the model in these habitats, we recommend the acquisition of this additional information and additional focus on model calibration for these parts of the lake.

Forebay Channel with Oxygen Injection., The most downstream sampling station that was used for the current model calibration was the open-lake forebay site,1.6 km upstream from the dam (Table 2, FBY). The most downstream segment of Lake Greenwood includes a restricted channel (150 m wide) extending 0.9 km from the open-lake forebay to the dam and power house. Duke Energy equipped this channel with three oxygen diffuser lines along the channel bottom; these diffusers were activated whenever dissolved oxygen in the tailrace dropped below 5 mg/L. Water quality data from sites in this channel were not used in the current model calibration. To apply the model in this environment, and to provide accurate predictions of tailrace oxygen concentrations, we recommend development and addition of oxygen injection kinetics (Hauser et al. 2000) for the downstream segment of the lake model.

Extended Drought. Hydrologic conditions during the calibration period (2004-05) included a wide range of stream flows and water residence times (Figs. 5 and 6) that were representative of most variations in stream flow and hydraulic turnover in the lake. However, this time period did not include an extended drought which could impose critical conditions for water quality dynamics in the lake. In future applications of the model for potential worst-case scenarios, we recommend additional model calibration and testing for extended drought conditions such as those that occurred from 1998-2001 in the Saluda-Reedy watershed.

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APPENDIX A

PHOSPHORUS, ALGAE, AND OXYGEN DYNAMICS IN LAKE GREENWOOD:

(H. McKellar and J. Bulak)

Lake Greenwood is the first major impoundment on the Saluda River, located approximately 100 km downstream from the Saluda headwaters in the SC Blue Ridge Mountains. The reservoir has a productive fishery although excess nutrient loading and eutrophication may threaten the quality of water and biotic habitat. This Appendix analyzes information from two years of intensive sampling (2004-05) to help quantify key interactions among nutrient distributions, algal productivity and oxygen depletion in the lake. This information was used to develop and calibrate a dynamic water quality model (CE-QUAL-W2) which was used to simulate key scenarios related to nutrient loading and effects on water and habitat quality in Lake Greenwood.

STUDY SITES AND METHODOLOGY

Field Sampling

A total of 11 sampling sites were established (Fig. A1) to quantify the spatial detail in the reservoir from the input tributaries (Saluda and Reedy River Arms) to the downstream forebay. Most of the sampling sites were located along the main axis of the lake, with one site in a mid-lake embayment (Mid-Cane Creek Embayment, Fig. A1). Table A1 indicates the study components at each of the stations. In addition to these stations in the lake, Clemson University (Klaine and Smink 2004-05) sampled water quality upstream on the Saluda and Reedy Rivers to quantify nutrient loading to Lake Greenwood.

Sampling Schedule and Analysis

The initial protocol included monthly sampling for all study components (Table A1). We increased sampling frequency to twice monthly through the active growing season of 2004 (May-Oct) with additional sampling during major storm events. During this time we also collected additional samples for the Algal Ecology Lab in Charleston to quantify algal community structure (Lewitus et al. 2005) During 2004, we sampled the lake for distributions of oxygen, phosphorus, algae distributions and productivity on 35 days, including 9 days of sampling before and after 3 major storm events (Tropical storm Bonnie and Hurricane Frances, and Hurricane Jeanne). During 2005, we continued monthly sampling for phosphorus distributions, algal biomass (chlorophyll-*a*), and oxygen profiles.

Field Measurements and Laboratory Analyses

<u>Temperature/Oxygen Profiles.</u> At all 11 sampling sites (Fig. A1, Table A1), a detailed vertical profile of water temperature and dissolved oxygen (YSI-58 DO Meter) was examined at 1-m depth intervals from the surface to the bottom. The oxygen sensor was air-calibrated and checked daily; the YSI-58 thermistor was calibrated against a certified, NIST-traceable thermometer (FisherBrand, SN:1295). To insure a high level of quality control in field data collection, our laboratory secured SC certification for field measures of temperature and oxygen profiles (Lab.ID 40570, 21 May 2004).

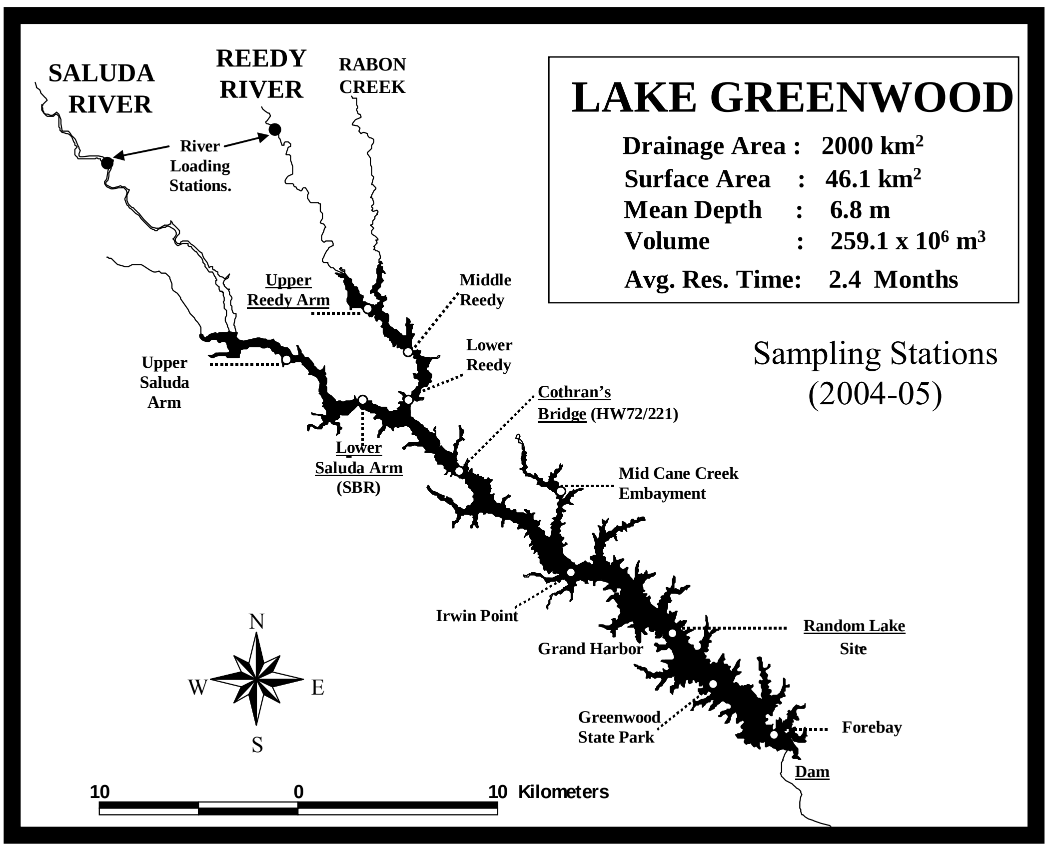

Fig. A1. Sampling Sites. Stations on the lake were were sampled by SCDNR; upstream sites ions on the Saluda and Reedy Rivers were sampled by Klaine and Smink (2004-05) to determine loading to the lake. Underlined station names indicate additional sampling by the SCDHEC monitoring program. Residence time was computed for the 8-yr average discharge in the Lake Greenwood tailrace (1997-2005; 41.3 m^3/s, Cooney et al. 2005)

Table A1. Site Locations and Study Components

Site Locations	Temperature/ Oxygen Profiles (2004-2005)	Phosphorus/ Chlorophyll Concentrations (2004-2005)	Plankton Productivity (2004)
Upper Saluda Arm	*	*	
Saluda Bridge	*	*	
Upper Reedy Arm	*	*	*
Middle Reedy Arm	*		
Lower Reedy Arm	*		
Highway 72 Bridge	*	*	*
Irvin's Point	*		
Cane Creek Embayment	*	*	
Irwin's Point	*		
Random Lake Station	*	*	*
Greenwood State Park	*		
Forebay	*	*	*

Phosphorus and Chlorophyll Concentrations. At 7 of the sampling sites (Table A1), we collected -water samples for analysis of algal biomass (chlorophyll-a) and phosphorus. Surface water samples for chlorophyll-a were placed in opaque HDPE bottles, labeled and placed immediately on ice. Phosphorus samples were collected from both surface and bottom waters and were partitioned into 3 HDPE bottles designated for analysis of total phosphorus, total soluble phosphorus, and soluble ortho-phosphate. Samples for total phosphorus were preserved with 1 ml H_2SO_4. Only surface water samples were collected at the upper tributary arms (Upper Saluda and Upper Reedy) which were < 5m total depth. All sample bottles were immediately labeled, placed on ice and transported to a certified analytical laboratory within 24 hrs of sample collection. Shealy Environmental Services (Lab. ID32010) performed the phosphorus analyses, using acid-persulfate digestion and ascorbic acid reduction (EPA Method 365.2). During 2004, two sets of samples for chlorophyll-*a* analyses were collected and placed on ice in amber or foil-covered HDPE bottles. The first set was analyzed by SEAUS, Inc (Cert. Lab ID 36001) using acetone extraction and fluorometric analysis (APHA 1998, Standard Method 10200H). The second set was analyzed by the SCDNR Freshwater Fisheries Research Lab using acetone extraction and a modified, non-acidification fluorometric analysis (Welschmeyer 1994, Arar and Collins 1997, APHA 1998,); the DNR lab was subsequently certified (Cert. ID 4057) for continued studies in 2005.

Algal Productivity. The vertical distribution of algal productivity was quantified monthly, based on oxygen changes in a vertical array of light and dark bottles incubated *in situ* at 4 of the sampling sites (Table A1, Fig. A3). The 4 stations were selected to provide a wide range of nutrient and light conditions for robust estimates of production coefficients in the model. At each station and time, a 15-L sample of surface water was collected, stirred vigorously

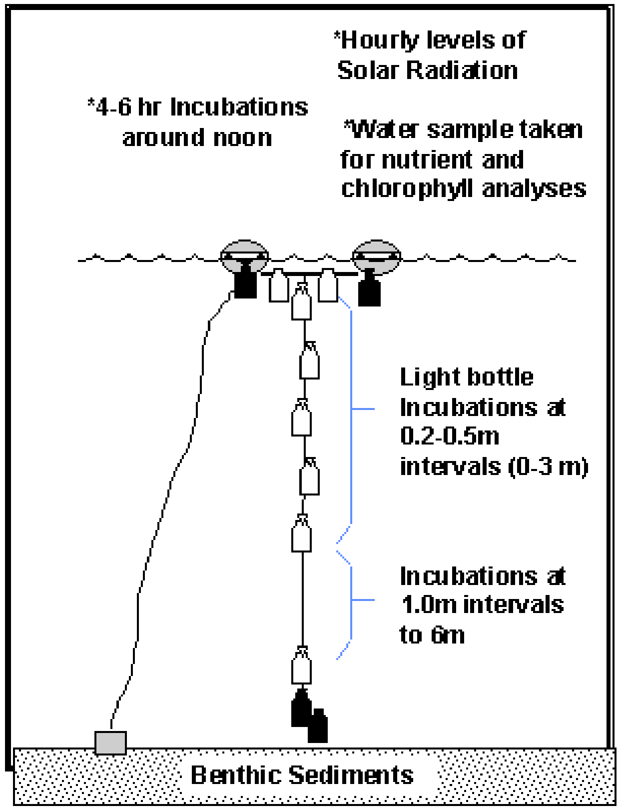

Fig. A3. Schematic view of the vertical array of light and dark bottles for evaluating algal production and water column respiration. Stations at the Forebay, Lower-Lake, and Mid-Lake were > 6m deep, so the "Benthic Sediments" were well below the vertical array. (The station in the upper Reedy River Arm was < 3 m deep so the vertical array extended only to 2.1 m).

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to insure homogenous conditions, and then used to fill 14 light bottles (300 ml BOD bottles) and 4 dark bottles. After the initial oxygen concentration was determined in 2 of the bottles (using a aYSI-58 DO meter and 5905 bottle probe) the remaining light bottles were suspended at 0.1, 0.3, 0.6, 1.1, 1.6, 2.1, 2.6, 3.1, 4.1, 5.1. and 6.1 m depths, with duplicate bottles at the 0.1m level. The depth range from 0 to 6.1 m was usually within the photic zone at these stations. Dark bottles were suspended in duplicate near the surface (0.1m) and just below the lower light bottles. After a 4 to 6-hr *in situ* incubation, the bottles were retrieved and the change in oxygen concentration determined. Net productivity (Pn, mg L^{-1} h^{-1}) for each depth in the vertical array was calculated as (L-I)/t, where L was the final oxygen concentration in each light bottle, I was the initial oxygen concentration, and t was the time of incubation (h). Respiration (R, mg L^{-1} h^{-1}) was calculated as (I-D)/t where D was the final oxygen concentration in the dark bottles. Gross productivity (Pg, mg L^{-1} h^{-1}) was then calculated for each depth in the vertical array as Pn + R. During winter and fall, R was evenly distributed between the upper and lower levels of the vertical array and the mean R from all 4 dark bottles was used to calculate Pg for each depth in the vertical array. However, during periods of thermal stratification in the top 6 m (May-August), the deeper dark bottles were often cooler (2-7 ^{o}C) than surface dark bottles and exhibited correspondingly lower rates of respiration. During these periods, R for each level in the vertical array was computed as an exponential function of the observed temperature profile as follows:

$$R = (k_1)\, \mathbf{e}^{(k2)T}$$

Where k1 and k2 were determined from an exponential regression (EXCEL) of R vs T for the shallow and deeper dark bottles. Hourly levels of gross productivity were extrapolated to daily rates (mg L^{-1} d^{-1}) by the following calculation:

$$Pg(mg\ L^{-1}\ d^{-1}) = Pg(mg\ L^{-1}\ h^{-1})(t)(Ld)/Li$$

where t = the duration (h) of incubation, Ld = total solar radiation for the day (µmol m^{-2} da^{-1}), and Li = total solar radiation during the incubation (µmol m^{-2}). Ld and Li were derived from continuous recordings of photosynthetically active radiation (PAR) using a LiCor Li-190SA quantum sensor and Campbell CR21X data logger, deployed on a dock at a mid-lake location near Station HW72 (Fig. 2). Additional information on light distribution through the water column was gained by vertical profiles of PAR at 0.5 to 1.0m intervals throughout the photic zone at each station (LiCor 250A underwater quantum meter). Secchi disk observations were also recorded as an additional indication of water clarity at each station and time.

Phytoplankton Taxonomy. To determine the succession of algal dominants during 2004 and to assess the potential for harmful algal blooms, additional samples were collected for taxonomic analysis by the SC Algal Ecology Lab in Charleston SC (Hollings Marine Lab and SCDNR Marine Resources Research Inst). This analysis included microscopic screening of preserved water samples and High Performance Liquid Chromatography (HPLC) analysis of extracted pigments of known taxonomic importance to algal identification. The details of methodology and results are provided in Lewitus et al. 2005.

RESULTS AND DISCUSSION

Basic Hydrology

Water quality in reservoirs responded to changes in basic hydrology such as tributary inflows, outflows and resultant residence time and water level. The water level in Lake Greenwood was maintained according to an operating “rule curve” approved by FERC (Fig. A4) The rule curve calls for water level reductions through the late fall and winter to a late January minimum of 434.5 ft (MSL). Beginning in February, the water level increased gradually to 439 ft by mid-April. This level was maintained through summer and early fall with a gradual drawdown beginning again in November. Some fluctuations in water level (+/- 1 ft) occurred in response to several major storm events in September and December.

The Saluda and Reedy Rivers represent the major sources of inflow to Lake Greenwood. Long-term mean daily discharge (since 1939) in the Saluda (976 cfs, http://nwis.waterdata.usgs.gov) was about 2.8 times that in the Reedy (352 cfs). This relative relationship was roughly the same during 2004-05, except for a few short-term runoff events in

February and July, when flow in the Reedy was approximately equivalent to those in the Saluda (Fig. A4).

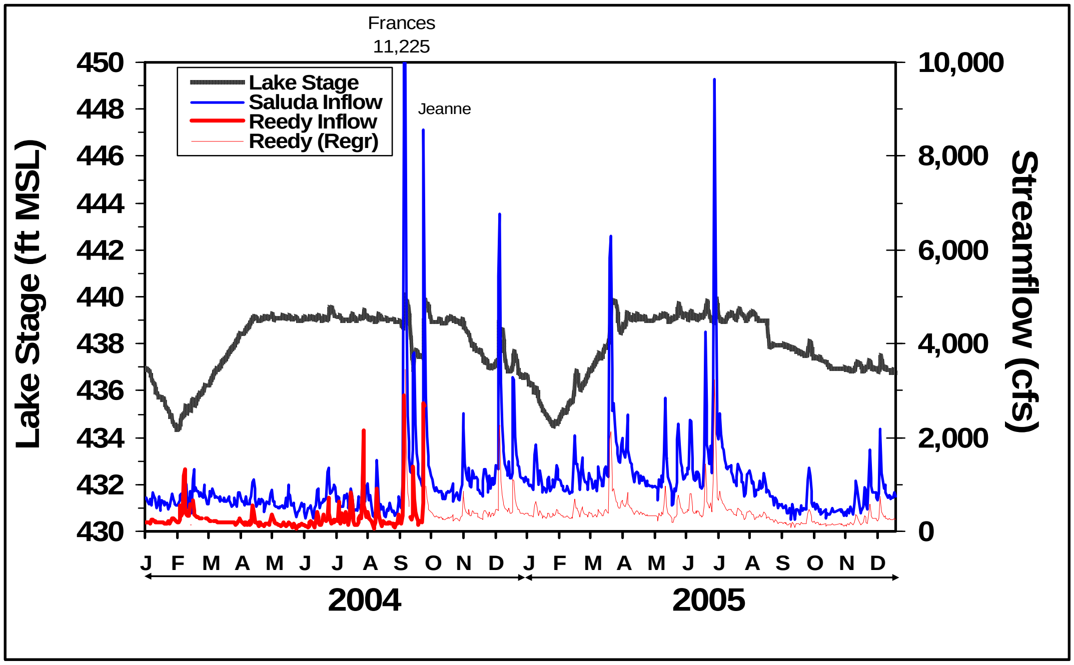

Fig. A4. Lake Greenwood water level (Lake Greenwood near Chappells, USGS 02166500) and major tributary stream flow inputs from the Saluda River at Ware Shoals (USGS 02163500) and the Reedy River at Ware Shoals (USGS 02165000). Extended gaps in gaging records for the Reedy River, after storm damage in September 2004, were extrapolated from a 2003 regression of Reedy River discharge with Saluda River discharge (Reedy_cfs = 0.3336 Saluda_cfs + 9.2892; r^2 = 0.7838;see Appendix D).

The highest flow for the study period resulted from a major storm in September (2004, Hurricane Frances) followed soon by several other hydrologic events (Ivan and Jeanne). Other hydrologic peaks > 6000 cfs occurred in December 04, March and June 05. The peak discharge in the Saluda following Hurricane Frances (11,225 cfs) was more than 10 times the long-term annual mean and was about 70% of the highest daily flow on record (16,100 cfs, Aug 27, 1995). The mean flow for September 2004 (2,837 cfs) was almost 5 times higher than the average flow for this month (594 cfs) and about 52% higher than the long-term maximum flows for September (1,862 cfs). Similar statistics for the Reedy were not available because of stream gauge damage during these storms.

Phosphorus and Chlorophyll Distributions

Phosphorus concentrations in the upper reaches of the lake (both the Saluda and Reedy River Arms, > 30 km upstream from the dam) were typically elevated above the SC water quality standard of 0.06 mg/L (Fig. A5). This observation was consistent with SCDHEC placement of Lake Greenwood on the State list of impaired waters due to high phosphorus concentrations. The overall mean concentration of total phosphorus in the Upper Saluda Arm (0.10 ± 0.02 mg $L^{-1;}$ Mean ± Std.Err) was similar to the Upper Reedy (0.10± 0.01 mg L^{-1}), but with 69 % exceedence of the 0.06 mg L^{-1} standard in the Reedy Arm and 57% exceedence in the

Saluda Arm. The highest concentrations in the surface waters (0.40-0.48 mg L^{-1}) were observed in both arms during discharge peaks related to Hurricanes Frances (Sep 9) and Jeanne

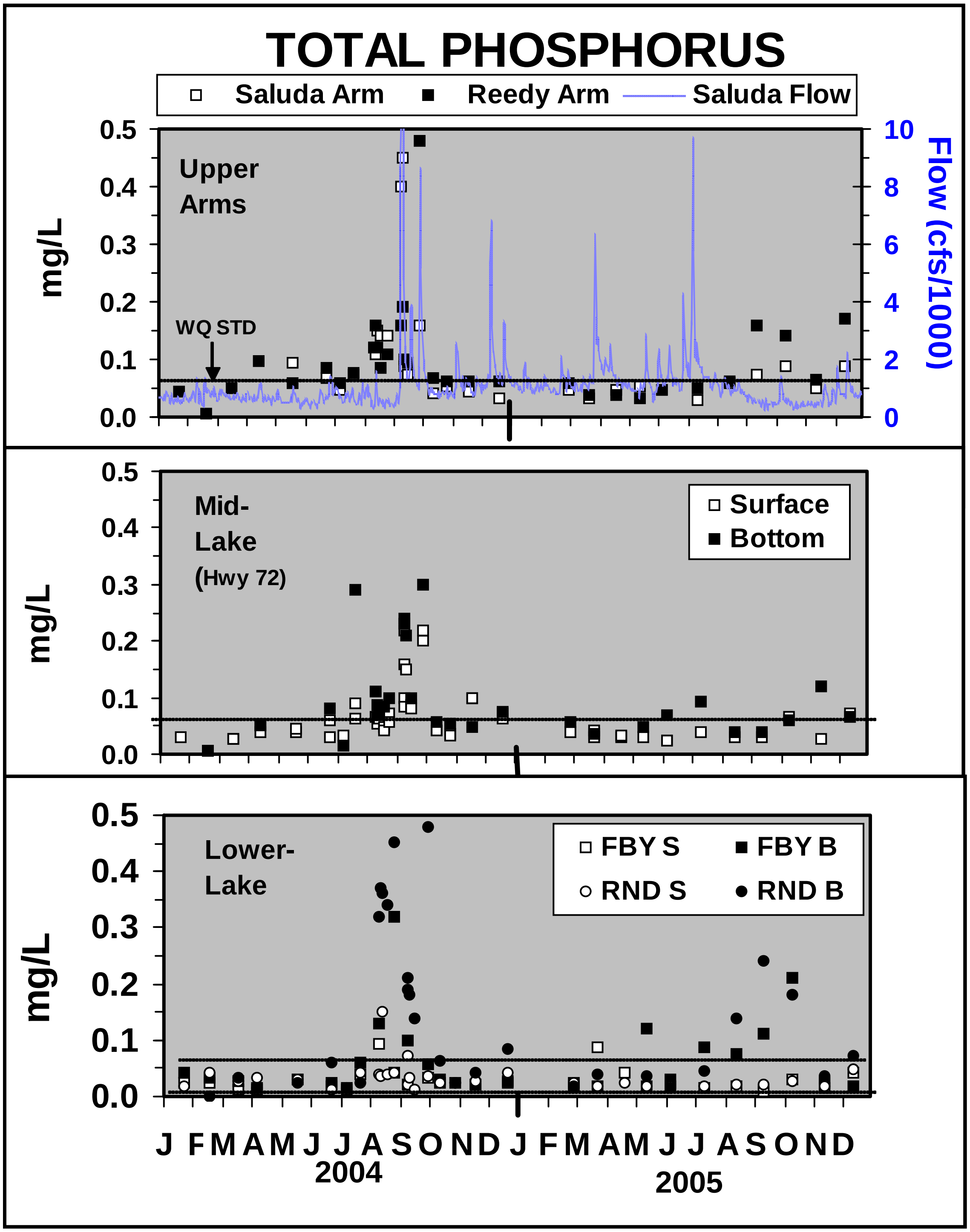

Fig. A5. Distributions of total phosphorus in surface (S) and bottom waters (B) of Lake Greenwood. The dashed line at 0.06 mg/L represents the SC water quality standard for total phosphorus.

(Sep 28, Fig. 4 in main report). These results suggest the importance of nonpoint sources of phosphorus in both tributaries during storm events. During the low flow period of the fall of 2005, the higher concentrations in the Reedy Arm suggest the influence of upstream point-source discharges of wastewater with high phosphorus concentrations.

Further downstream, surface water concentrations typically declined substantially (Fig. A5), due, in part, to the sedimentation of particulate phosphorus. However, bottom water concentrations at the downstream, deeper stations were much higher than in the surface waters during the late summer of both years. At the Lower Lake stations (15-20 m deep), average bottom water concentrations in Aug and Sep (0.15-0.30 mg L^{-1}) were 3-6 times higher than in the surface waters (0.05 mg L^{-1}). This increase of bottom water phosphorus was probably due to the accumulation of settling particulate matter and phosphorus release from anaerobic benthic sediments (see section on Oxygen Depletion). These data helped quantify key parameters in the model for sedimentation rates and release rates from the sediments (see Model Calibration).

Algal biomass (chlorophyll-*a*) also exhibited higher concentrations in the upper reaches of the lake (Fig.A6), extending downstream to the middle sections (20 km upstream from the dam) including the mid-lake tributary embayment (Cane Creek). Algal biomass reached moderate levels (10-20 µg L^{-1} chlorophyll *a*) throughout the lake during summer, with clear domination by cyanobacteria (blue-green algae) during August and September (Lewitus et al. 2005). Cyanobacteria are nitrogen-fixing species that commonly bloom in nutrient rich conditions with limited hydrodynamic flushing, particularly during the warmer months. Some genera of known toxin-producing species of Cyanophytes were identified (*Microcystis*, *Anabaena*, *Nitzchia*, *Aphanizamenon*, and *Anabaeneopsis*) although none of these genera was found in high concentrations (Lewitus et al. 2005). On the other hand, total algal biomass (as indicated by chlorophyll *a* concentrations) occasionally exceeded state standards (40 µg L^{-1}) in the upper and mid-lake sections. The pronounced bloom at the Mid-Lake station in late fall (60-80 µg L^{-1}, Fig.A1.6) was confirmed by both laboratories and was dominated by alloxanthin pigments (Cryptophytes, Lewitus et al. 2005). Cryptophytes are motile, protozoan-like algae that are not typically associated with harmful, toxin-producing algae. The highest persistent chlorophyll *a* concentrations occurred in the mid-lake tributary embayment, suggesting more pronounced blooms in semi-enclosed embayments with low flushing. Cryptophytes also dominated the algal community in the embayment during these fall blooms.

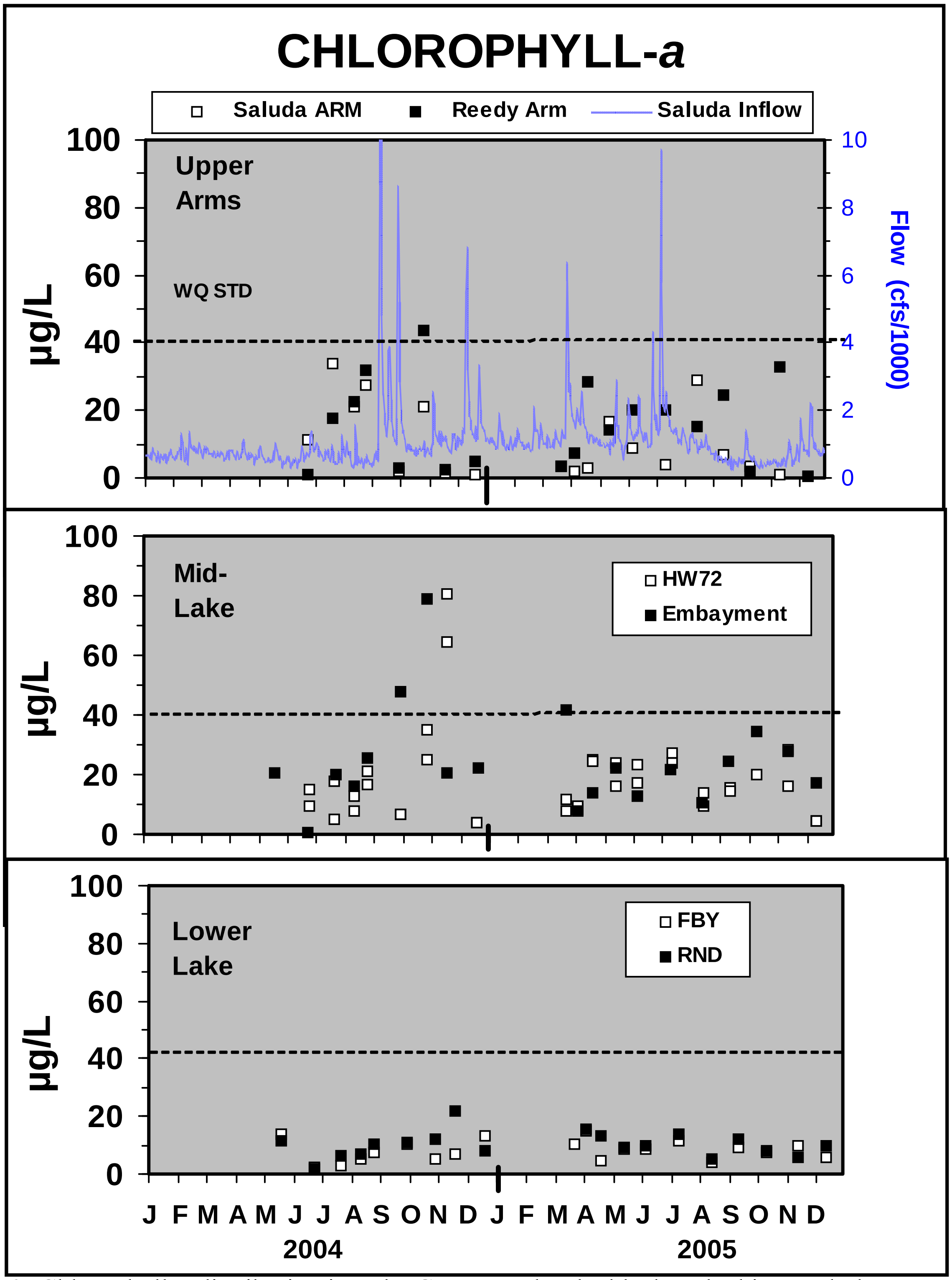

Fig. A6. Chlorophyll-*a* distribution in Lake Greenwood. The black and white symbols represent separate stations in each lake zone. The dashed lines indicate the water quality standard of 40 µg L^{-1}.

Algal Productivity

Vertical patterns and the spatial variability of algal productivity typically exhibited some general correlations with phosphorus concentration, algal biomass, and turbidity. For example, during mid-summer conditions (Fig. A7, Table A2), the Upper Reedy Arm exhibited the highest levels of surface productivity (Pg(max), perhaps in response to higher phosphorus concentration and algal biomass. However, productivity in the Upper Reedy attenuated rapidly with depth due to more turbid conditions in the upper lake. The Mid-Lake station had similar phosphorus levels, although this station displayed somewhat less surface production, due in part to less algal biomass. However, at this station, water clarity was higher, light attenuation was lower and productivity extended through deeper levels of the water column, yielding the highest level of total water column production (Pg(int), Table A2). At the downstream regions of the lake (Lower Lake and Forebay), phosphorus and chlorophyll levels were much lower, corresponding to considerably less algal production at the surface and through the water column.

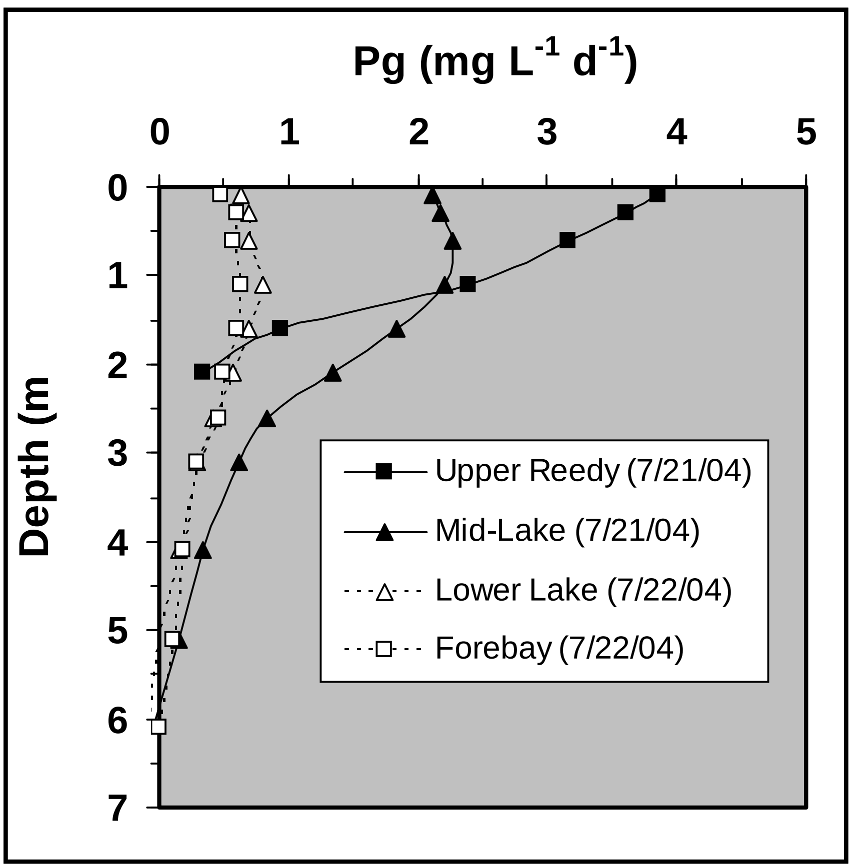

Fig. A7. Example of typical vertical patterns of gross oxygen production at 4 stations in Lake Greenwood (7/21-22/04).

Table. A2. Daily Oxygen Production, Phosphorus Concentration, Chlorophyll-*a*, and Light Attenuation In Lake Greenwood (7/21-22/04). Pg(max) is the maximum volumetric production rate in the water column, Pg(int) is the vertically integrated, area-based production through the water column, Kext is the light extinction coefficient.

Station	Pg(max) (mg L^{-1} d^{-1})	Pg (int) (g m^{-2} d^{-1})	Chlorophyll (µg L^{-1})	Tot. P (mg L^{-1})	Secchi (m)	Kext (m^{-1})
Upper Reedy	3.86	4.31	17.7	0.077	0.5	3.05
Mid-Lake	2.27	5.71	5.2	0.076	1.3	1.12
Lower Lake	0.80	2.07	6.1	0.042	2.7	0.78
Forebay	0.64	2.04	2.9	0.038	>3.0	0.82

The maximum productivity in the surface waters (Pg(max)) exhibited a distinct seasonal patterns with a rapid increase at all stations in early spring (Fig. A8, upper panel). Throughout the rest of the growing season (May-Oct), Pg(max) was typically higher in the Upper Reedy and decreased from upper lake to lower lake stations (Fig. 8, Table 3), similar to patterns for total phosphorus and chlorophyll. The major deviation from this growing season pattern was in late September, when runoff from Hurricane Jeanne (Fig. A4) produced extremely high turbidity in the upper and mid-lake stations (secchi disk observations ≅ 0.1 m). This high turbidity greatly inhibited algal productivity at the upper and mid-lake stations. During this same time, surface water turbidity in the lower lake remained relatively low (Secchi disk values > 1 m) and productivity exhibited a moderate fall peak.

The daily integrated area-based productivity (Pg(int)), displayed a similar seasonal pattern (Fig. A8, lower panel). However, the spatial pattern of Pg(int) during growing season was more variable (Fig. A8, Table A3), reflecting combined influences of nutrients and turbidity. While phosphorus concentrations (and algal biomass) were typically higher in the upper and mid-lake stations, the lower turbidity in the lower lake stations allowed productivity to extend to deeper levels often resulting in higher integrated productivity. This was particularly evident in early spring and fall, when the vertically integrated productivity at both the Lower-Lake and Forebay stations was higher than in the Upper and Mid-Lake stations (Fig. A8). Respiration rates (R(int)) indicated higher rates of oxygen demand and general heterotrophic conditions (P:R ratio < 1) in the photic zone of the Upper to Mid-Lake areas (Table A3). In contrast, the lower lake stations indicated a more autotrophic photic zone (P:R ratio > 1) with a net production of organic matter through the growing season.

Oxygen Depletion

Lake Greenwood has a history of hypolimnetic oxygen depletion that affects habitat availability (Snoots 1993). During 2004-05, the distribution of dissolved oxygen in Lake Greenwood exhibited a rapid depletion in the bottom waters throughout the lake from well-mixed conditions in March and early April to highly stratified conditions from May through

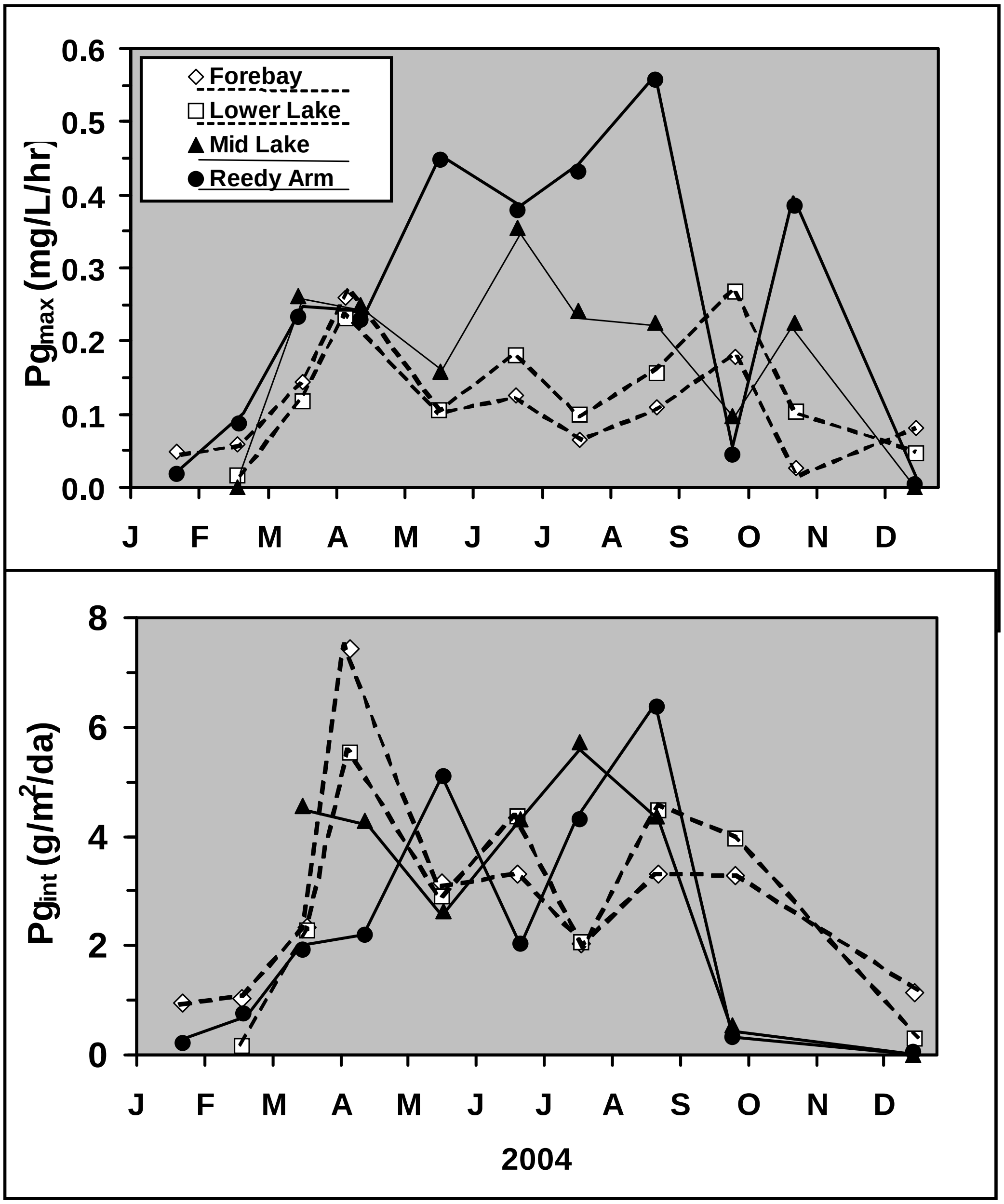

Fig. A8. Seasonal patterns of maximum surface productivity (Pg(max)) and vertically integrated productivity (Pg(int)) in Lake Greenwood, 2004.

Table A3. Growing Season Means (± standard error) for Pg(max), Pg(int), R(int), and P:R Ratios; May-Oct, 2004

Stations	Pg(max) (mg L^{-1} h^{-1})	Pg(int) (g m^{-2} d^{-1})	R(int) (g m^{-2} d^{-1})	P:R Ratio
Upper Reedy	0.375 ± 0.071	3.63 ± 1.09	3.80 ± 0.67	0.96
Mid-Lake	0.217 ± 0.035	3.51 ± 0.89	6.96 ± 2.20	0.50
Lower Lake	0.152 ± 0.027	3.55 ± 0.47	2.65 ± 0.84	1.34
Forebay	0.101 ± 0.021	3.01 ± 0.25	2.36 ± 1.09	1.27

October (Figs.A9,a-c). The lower river discharge and lower hydrologic flushing rates during the spring of 2004 (see Fig. 5 main report) probably led to an earlier development of a thermocline and earlier onset of oxygen depletion than in the more normal spring flows in 2005 (Fig. A9,a-b). A regression of the mean oxygen concentration below 10m depth (Feb-June) yielded a linear rate of DO depletion with a 2004 slope of 4.2 mg L^{-1} mo^{-1} (r^2> 0.94), compared to a 2005 slope of 3.1 mg L^{-1} mo^{-1} (r^2> 0.98), further quantifying the higher rate of depletion during 2004. Hypolimnetic oxygen was essentially depleted (< 1 mg/L) by July of both years (Fig. A9b). Water column mixing and reaeration of deep waters began with a deepening thermocline from August to October leading to extensive reaeration of the water column by November. The high flows in the September of 2004 probably accelerated reoxygenation during fall of this year (Fig A9c). This pattern in 2004 was in contrast to the delayed oxygenation of the bottom waters the following year, probably in response to the normal, low autumn flows in 2005.

During the 2-year study, there were few contraventions of the state DO standard for surface waters (4 mg/L). The lowest surface water DO concentrations were typically observed in the enclosed tributary embayment (Hidden Lake on Cane Creek) where DO contraventions were documented only once (mid-August, 2004, 3.8 mg/L). However, the observed oxygen depletion in deeper waters throughout the lake, as illustrated in Fig.A9, could represent an important factor in the distribution of viable habitat for many fish and benthic organisms. For cool water species, such as striped bass, the issue of sub-surface oxygen depletion becomes particularly important since these fish seek cooler, deeper waters during the warmest months. When deeper water oxygen declines, the tolerable habitat is "squeezed" to a narrow zone between the warm surface waters and the deeper oxygen-depleted waters.

Using detailed profiles for temperature and DO, Fig. A10 (a and b) maps the longitudinal distribution of habitat quality for striped bass in Lake Greenwood during 2004-05. "Good" habitat (dark blue) is defined as the preferred zone where water temperature is < 25^{o}C and DO is > 4 mg/L. "Poor" habitat is where water temperature is > 27^{o}C and/or DO is < 2 mg/L. "Marginal" or tolerable habitat is the zone between good and poor, indicated by light blue areas on Fig A10. As early as May 2004, good habitat began to diminish due to warming surface waters and oxygen depletion in the deeper cooler waters. This habitat "squeeze" was delayed in 2005 due to higher flows, cooler temperatures, and more mixing during the spring; zones of good habitat lingered into June 2005. From July through September of both years, high quality habitat was clearly limited (Fig. A10b). High quality habitat returned by October due to cooling surface waters; by November, only the deeper waters near the dam retained oxygen depleted, poor-quality habitat.

Although oxygen decline in the hypolimnion is a natural process in stratified lakes, the intensity, duration, and spatial extent of hypoxic conditions in Lake Greenwood could be related, in part, to the high rates of nutrient loading and eutrophic conditions in the upper regions of the lake. These data were used to calibrate the model for temperature and oxygen distributions in Lake Greenwood. Once calibrated the model was used to compute the extent and duration of predicted patterns of oxygen depletion and habitat quality as a function of nutrient loading.

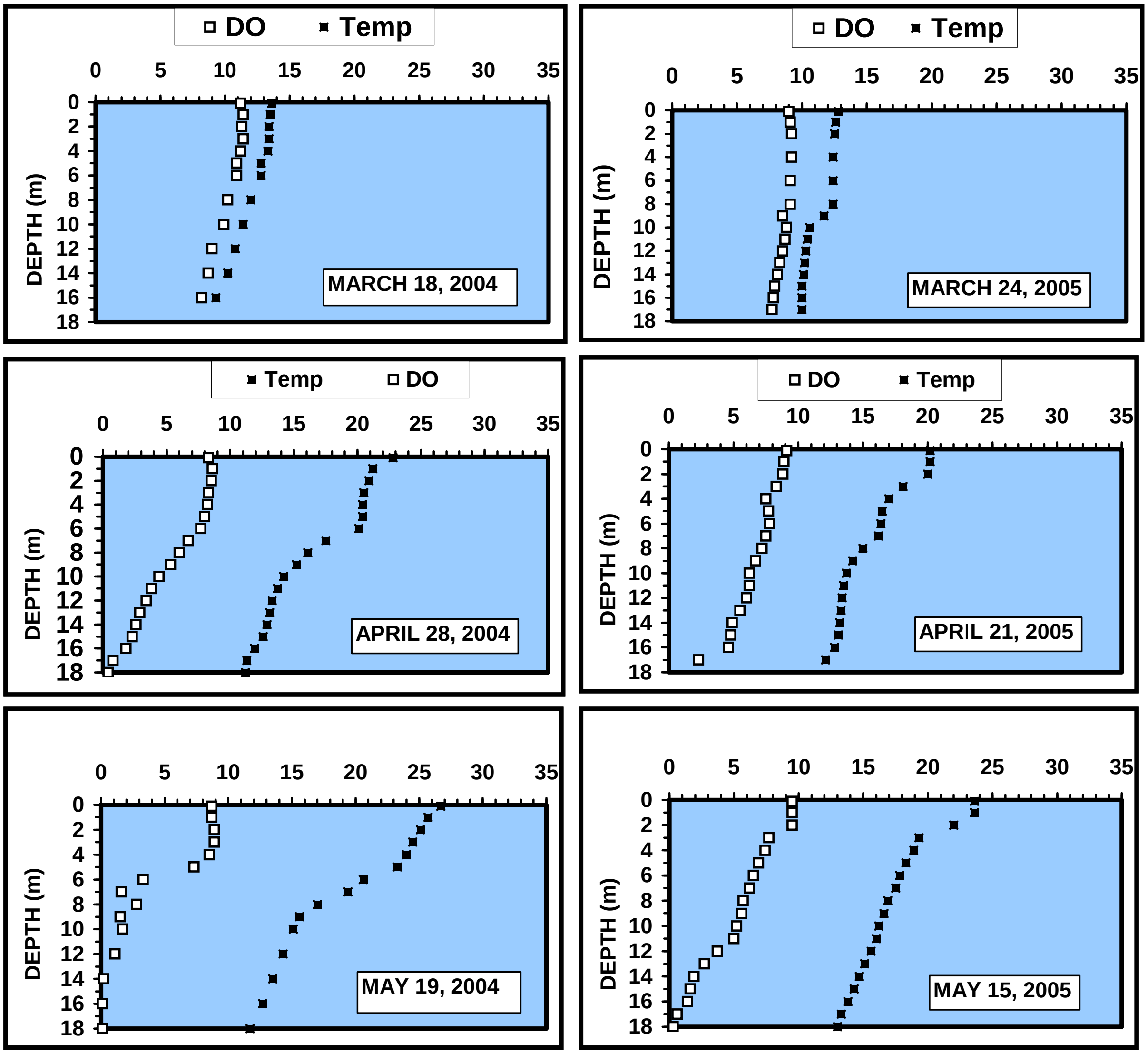

Fig. A9a. Vertical patterns of dissolved oxygen (DO) and water temperature in the forebay of Lake Greenwood; March-May: 2004 (left panels) and 2005 (right panels).

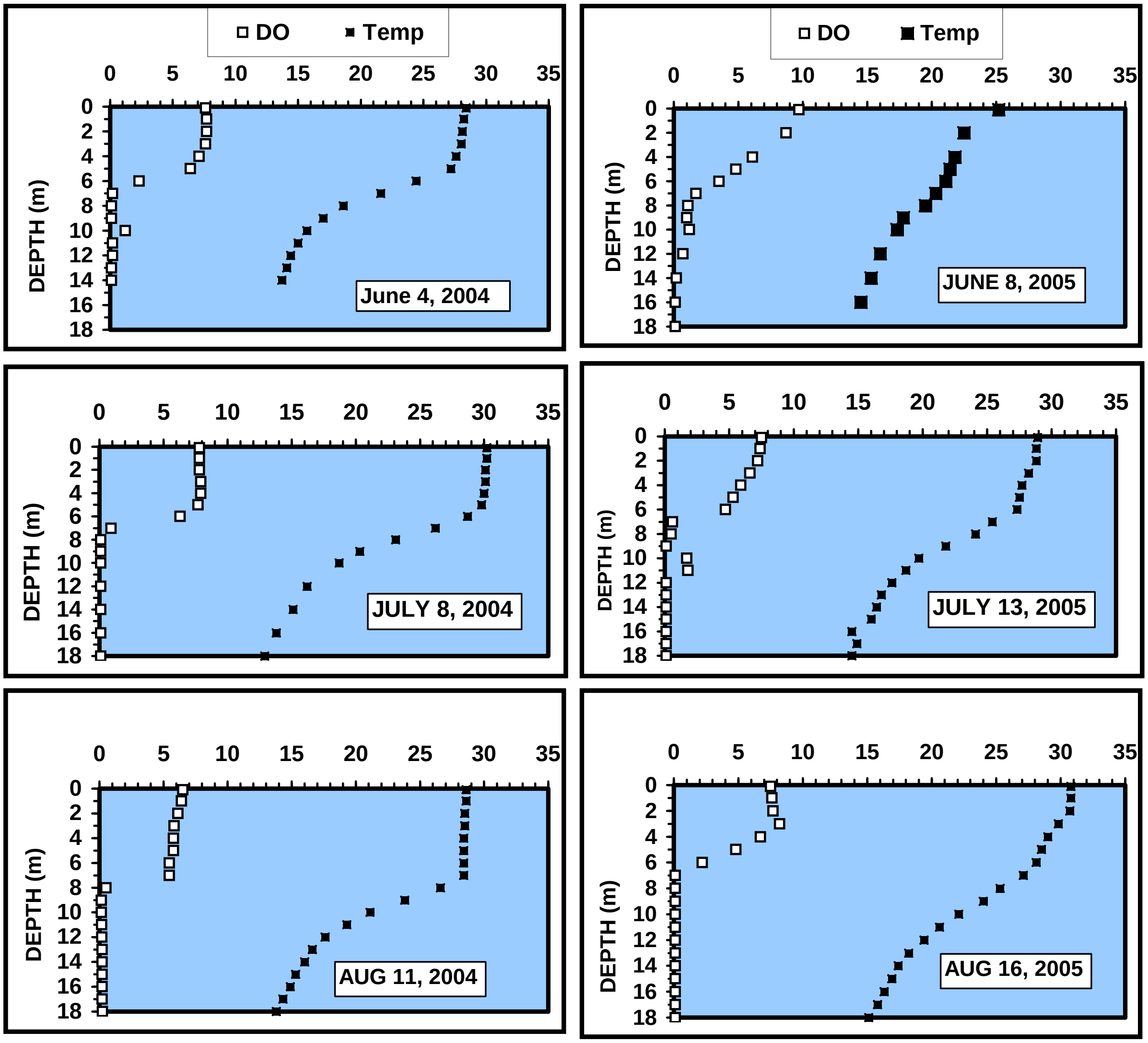

Fig. A9b. Vertical patterns of dissolved oxygen (DO) and water temperature in the forebay of Lake Greenwood; June-Aug: 2004 (left panels) and 2005 (right panels).

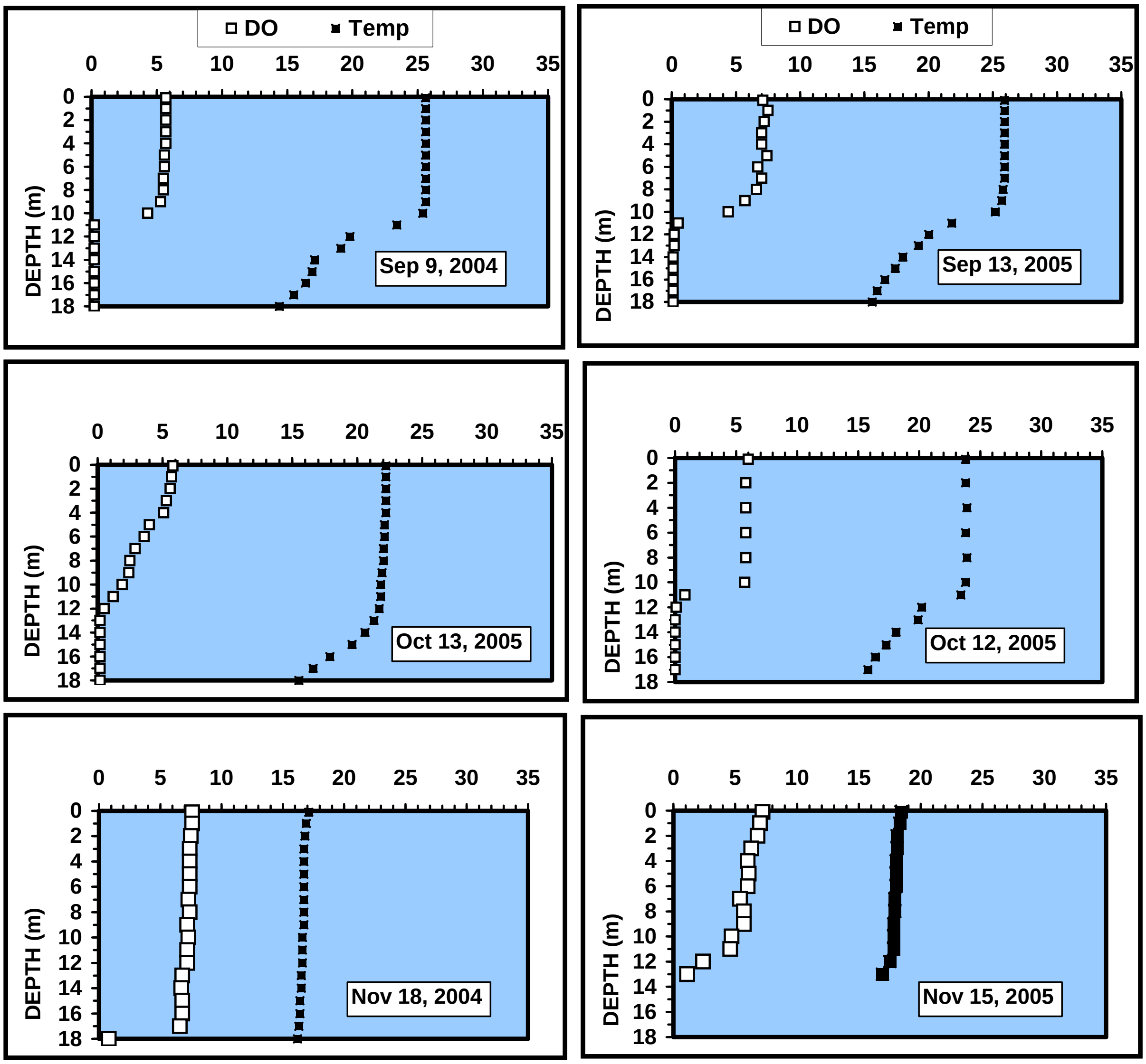

Fig. A9c. Vertical patterns of dissolved oxygen (DO) and water temperature in the forebay of Lake Greenwood; Sept.-Nov: 2004 (left panels) and 2005 (right panels).

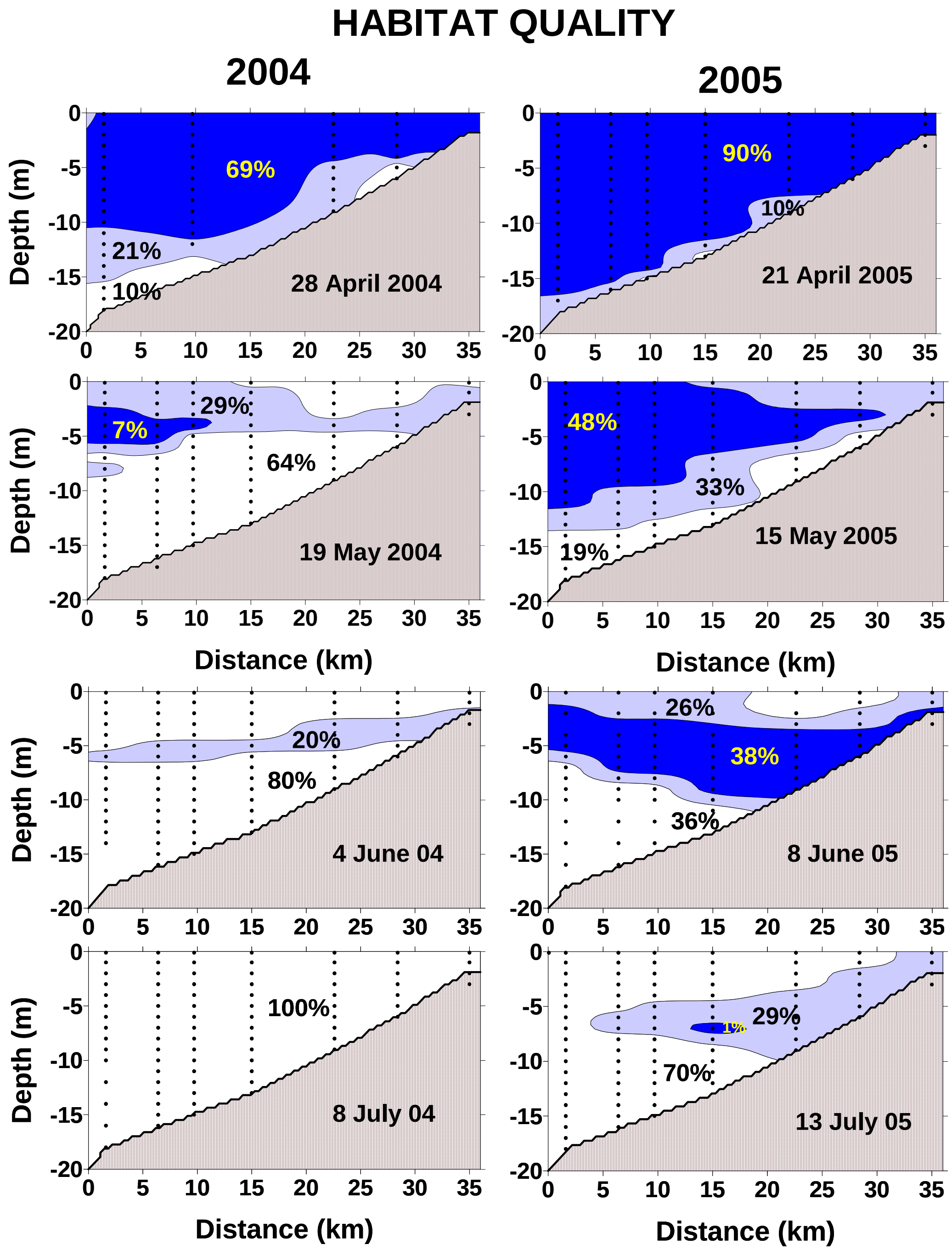

Fig. A10a. Spatial patterns of habitat quality in Lake Greenwood for spring and summer 2004 (left panels) and 2005 (right panels). Dark blue indicates "Good" habitat where DO>4 mg/L and Temperature is < 25°C). White areas indicate "Poor" quality habitat where DO < 2 mg/L or temperature is > 27°C. Light blue indicates a "Marginal" or tolerable habitat. Percent values indicate the proportion of the transect area occupied by each zone.

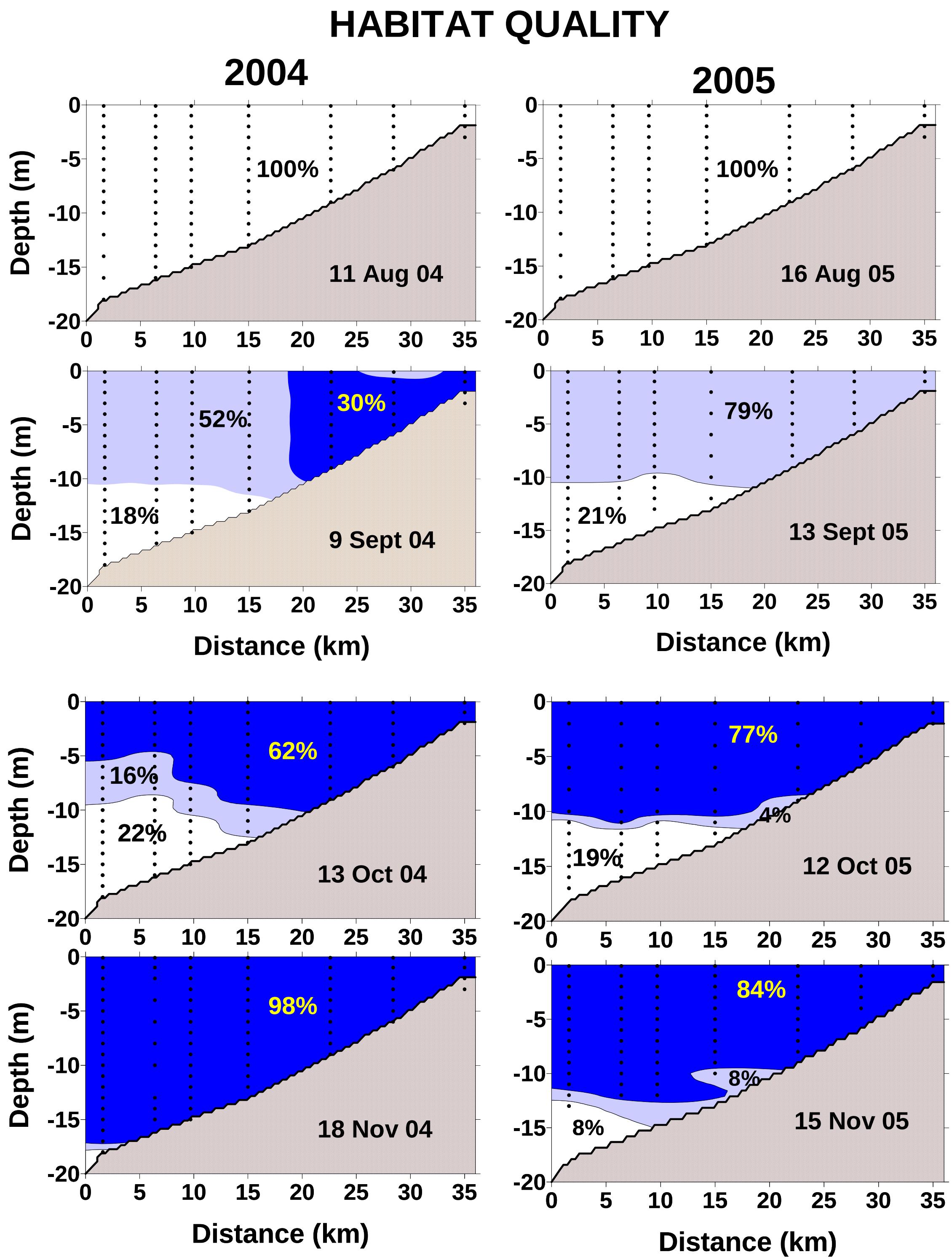

Fig. A12b. Spatial patterns of habitat quality in Lake Greenwood through the summer and fall for 2004 (left panels) and 2005 (right panels. See legend for Fig A1.12a..

APPENDIX B

MODEL SEGMENTATION

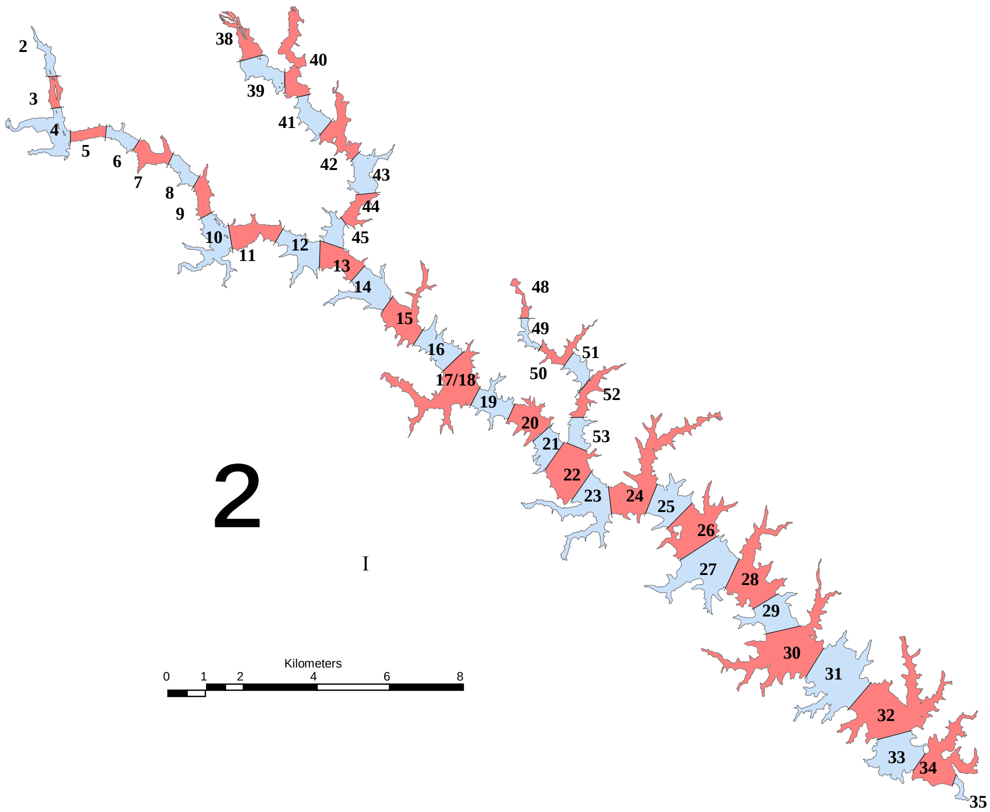

Fig. B1 Model segmentation for the Lake Greenwood model. The average length of each segment is 1.2 km, numbered sequentially down the main axis of the lake from the headwaters of the Saluda Arm (segment 2) to the forebay channel at the dam (segment 35). The Reedy Arm is represented by 8 segments (38-45) and the Cane Creek Arm is represented by 6segments (48-53). The missing segment numbers (1, 36,37, 46, 47, and 54) represent the upstream and downstream boundary conditions (phantom segments) for each branch in the model.

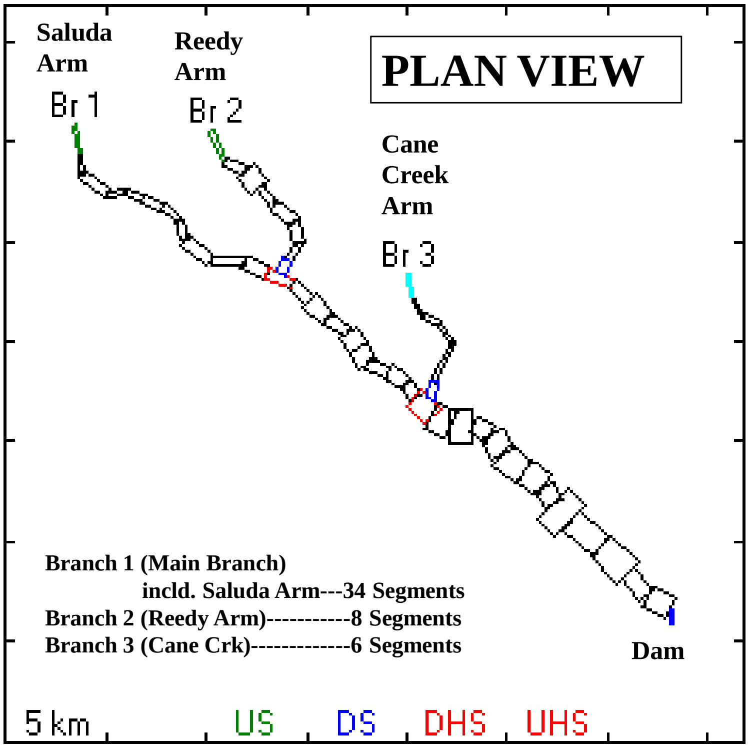

Fig. B2. Plan view of model schematic for Lake Greenwood segmentation; length, orientation and mean width of the surface layer of all segments. Details for the three branches are in Figs B3, B4, and B5. This geometric data (length, mean width, depth, and geographic orientation of each layer) were tabulated for direct input to the bathymetry files of CE-QUAL-W2

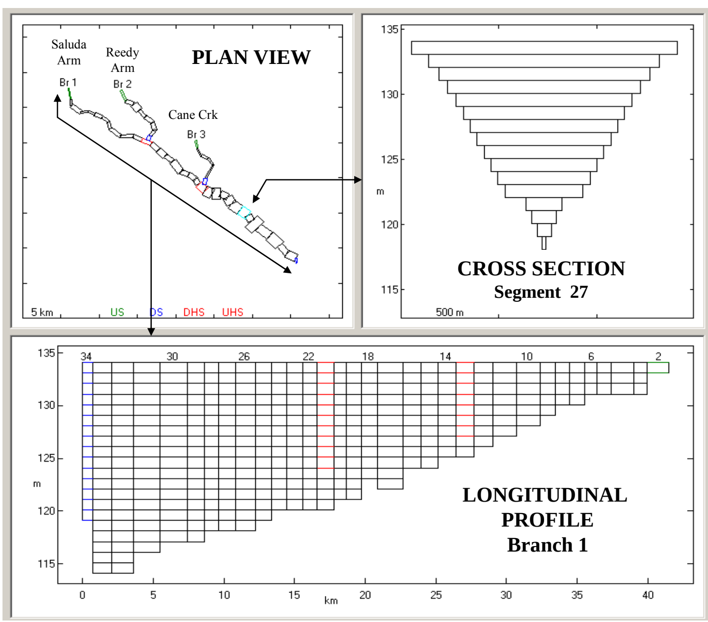

Fig.B3. Lake Greenwood model segmentation; Branch 1 longitudinal profile and cross-section profile at segment 27 of the main lake body. Y axis of the cross section and longitudinal profile represents the elevation (m) of each 1m vertical segment

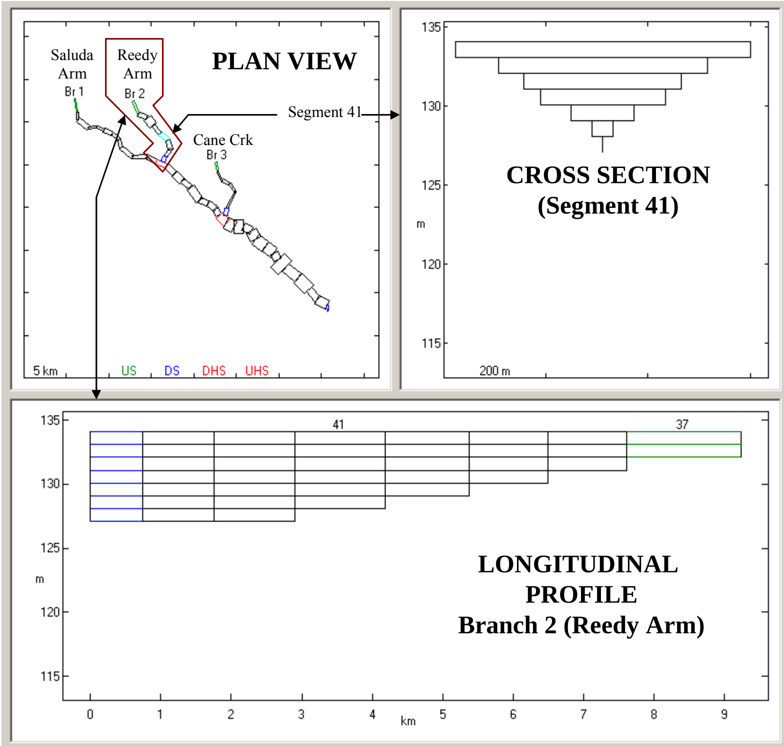

Fig. B4. Lake Greenwood model segmentation; Branch 2 longitudinal profile and cross-section profile at segment 41 of the Reedy River Arm.

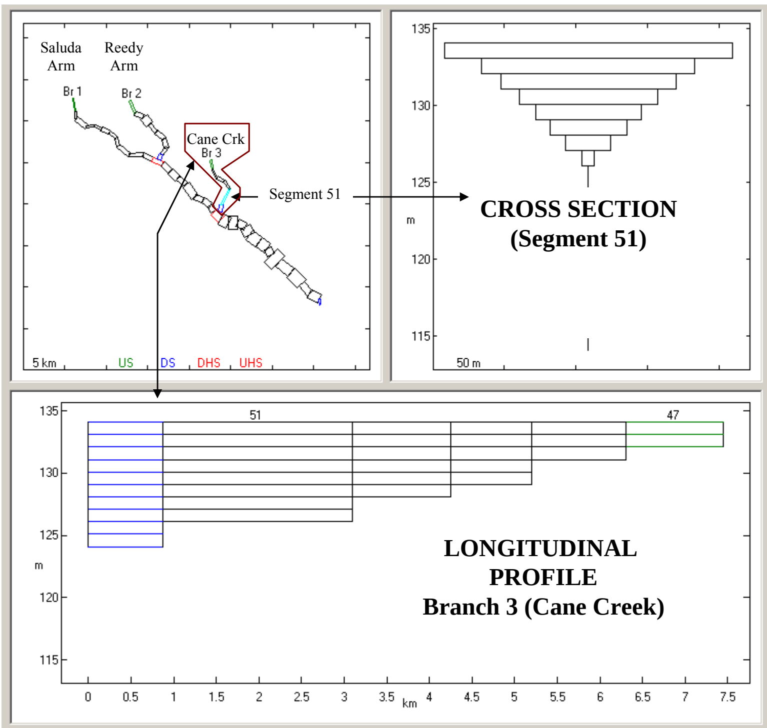

Fig.B5.. Lake Greenwood model segmentation; Branch 3 longitudinal profile and cross section profile for segment 51 on the Cane Creek Arm.

APPENDIX C

MAJOR COEFFICIENTS AND CONSTANTS USED IN THE CE-QUAL-W2 MODEL of LAKE GREENWOOD

Default values and/or representative example values provided by Cole and Wells (2002) are listed for comparison where applicable. NA= Not Applicable. A complete set of input files for the model are available from the authors.

MODEL GRID SETUP; INFLOW/OUTFLOW STRUCTURES

Model Symbol	Description	Value	Default Value
NWB	Number of water bodies	1	NA
NBR	Number of branches	3	NA
IMX	Number of segments in the computational grid	54	NA
KMX	Number of layers in the computational grid	22	NA
NTR	Number of tributaries (minor tributaries are treated as distributed flow)	0	NA
NST	Number structures (a single discharge structure at the dam)	1	NA
NWD	Number of withdrawals (a single drinking water withdrawal)	1	NA

TIME FACTORS

Model Symbol	Description	Value	Default Value
TMSTRT	Start time (1 Jan 2004)	1	NA
TMEND	End time (31 Dec 2005)	730	NA
DLTMAX	Maximum time step (seconds)	3600	NA

HEAT EXCHANGE

Model Symbol	Description	Value	Default Value
SLHTC	Equilibrium temperature computation (ET) for surface exchange	ET	NA
AFW	Intercept for wind-driven heat exchange function	9.2	9.2
BFW	Slope for wind-driven heat exchange function	0.46	0.46
CFW	Exponent of wind-driven heat exchange function	2.00	2.0
WINDH	Height of wind speed measurement (m)	10	NA
SLTRC	Transport solution scheme; ULTIMATE algorithm eliminates physically unrealistic over/undershoots due to longitudinal transport	ULTIMATE	ULTIMATE
THETA	Time-weighting for vertical advection	0.55	0.55

HYDRAULICS

Model Symbol	Description	Value	Default Value
AX	Longitudinal Eddy Viscosity (m^2 sec^{-1})	1	1
DX	Longitudinal Eddy Diffusivity (m^2 sec^{-1})	1	1
CBHE	Coeff. of bottom heat exchange (W m^{-2} sec^{-1})	7.0E-8	7.0E-8
TSED	Temperature of the sediment (oC)	14	11.5
FI	Interfacial friction factor	0	0.01
TSEDF	Heat from sediments added back to water	0	(0-1)
FRICC	Bottom friction	CHEZY	
WDRAGC	Wind drag; recent modification by Ruane and Hauser (2006) provides for the WUEST algorithm which produces a higher level of mixing under low wind speed conditions	WUEST	NA
AZC	Form of vertical turbulence closure algorithm		W2
AZSLC	Implicit (IMP) or Explicit (EXP) treatment of vertical eddy viscosity		EXP
AZMAX	Maximum value for vertical eddy viscosity, *m2 s-1*	1E-4	1E-3

HYDRAULIC STRUCTURE CHARACTERISTICS

Model Symbol	Description	Value	Default Value
KTSTR	Top water layer above which selective withdrawal will not occur through the intake structure of the power house draft tubes	2	NA
KBSTR	Bottom layer below which selective withdrawal will not occur through the intake structure of the power house draft tubes	10	NA
SINKC	Selective withdrawal algorithm for the intake structure of the power house draft tubes	POINT	NA
ESTR	Centerline elevation of intake structure for the power house draft tubes (m)	128	NA
IWD	Drinking water withdrawal structure; lake segment number	20	NA
EWD	Drinking water withdrawal structure centerline elevation (m)	130.45	NA
KTWD	Top water layer above which withdrawal will not occur through the drinking water intake structure	2	NA
KBWD	Bottom water layer below withdrawal will not occur occur through the drinking water intake structure	14	NA

LIGHT EXTINCTION and SUSPENDED SOLIDS

Model Symbol	Description	Value	Default Value
EXH2O	Extinction for pure water (m^{-1})	0.25	0.25-0.45
EXSS	Extinction due to inorganic suspended solids, *m-*	0.1	0.1
EXOM	Extinction due to organic suspended solids, *m-1*	0.1	0.1
BETA	Fraction of solar radiation absorbed at water surface	0.45	0.45
EXA	Extinction due to algal biomass, m-1	0.2	0.2
SSS	Suspended Solids Settling rate (m d^{-1}) *	0.5	0.5-1.5

* The default/recommended range of 0.5-1.5 in Cole and Wells (2002) accommodated settling rates for 3 size fractions of TSS (small, medium, and large, respectively). The Lake Greenwood model uses only one TSS compartment, assuming the lowest settling velocity corresponding to the domination of smaller-sized particles.

ALGAL METABOLISM

Model Symbol	Description	Group 1 Diatoms	Group 2 Greens	Group 3 Cyano-bact.	Default Value
AG	Maximum Growth Rate (da^{-1})	1.8	1.5	1.6	2.0
AR	Respiration (da^{-1})	0.04	0.04	0.04	0.04
AE	Excretion (da^{-1})	0.04	0.04	0.04	0.04
AM	Mortality (da^{-1})	0.08	0.1	0.1	0.1
AS	Sinking Rate (m da^{-1})	0.08	0.05	0.02	0.1
AHSP	Half-saturation const for P (mg/L)	.003	.003	.003	0.003
AHSN	Half-saturation const for N (mg/L)	.01	.01	.01	0.014
AHSS1	Half-saturation const for Si (mg/L)	.003	.003	.003	0
ASAT	Light Saturation (W m^{-2})	150	150	150	75
Temperature					
AT1	Min temperature for growth (oC)	0	10	20	5
AT2	Lower temp for max growth (oC)	15	20	28	25
AT3	Upper temp for max growth (oC)	22	35	35	35
AT4	Max temp for growth (oC)	40	40	40	40
Stoichiometry					
ALGP	Algal P: Biomass ratio *	0.009	0.009	0.009	.005
ALGN	Algal N:Biomass ratio	0.06	0.06	0.06	0.08
ALGC	Algal C:Biomass ratio	0.45	0.45	0.45	0.45
ALGSI	Algal Si:Biomass ratio	0.12	0.01	0.01	0.18
ALCHLA	Algal Biomass:Chlorophyll ratio	175	125	125	145
ALPOM	Fraction of biomass mortality converted to particulate org.matt.	0.8	0.8	0.8	0.8
ANEQN	Ammonium preference factor**	2	2	2	
ANPR	Half-saturation preference for ammonium,	0.005	0.005	0.005	0.001

* Recent modifications (Ruane and Hauser 2006) allow higher P:Biomass ratios for algal organic matter (original default value (0.005) in Cole and Wells (2002) assumed a constant ratio for all organic matter)

** Equation # 2 = Thomann and Fitzpatrick formulation (1982) as described in Cole and Wells (2002)

ORGANIC MATTER PROCESSING

Model Symbol	Description	Value	Default Value
LDOMDK	Labile dissolved organic matter decay rate (d^{-1})	0.1	0.1
RDOMDK	Refractory dissolved organic matter decay rate(d^{-1})	0.001	0.001
LRDDK	Labile to refractory DOM decay rate (d^{-1})	0.01	0.01
LDOMR	Labile dissolved organic matter release from anaerobic sediment as a stoichiometric fraction of SOD *	0.6	NA
LPOMDK	Labile particulate organic matter decay rate (d^{-1})	0.08	0.08
RPOMDK	Refractory particulate organic matter decay rate (d^{-1})	0.001	0.001
LRPDK	Labile to refractory POM decay rate (d^{-1})	0.001	0.01
POMS	Particulate organic matter settling rate (m d^{-1})	0.1	0.1
ORGPL	P:OrgMatt ratio for labile organic matter *	0.009	0.005
ORGNL	N:OrgMatt ratio for labile organic matter *	0.06	0.08
ORGC	C:OrgMatt ratio for dissolved and particulate organic matter	0.45	0.45
ORGSI	Si:OrgMatt ratio for dissolved and particulate organic matter	0.18	0.18
ORGPR	P:OrgMatt ratio for refractory organic matter *	0.0009	NA
ORGNR	N:OrgMatt ratio for refractory organic matter *	0.006	NA

OMT1	Lower temperature for organic matter decay (oC)	5	4
OMT2	Upper temperature for organic matter decay (oC)	30	25
OMK1	Fraction of organic matter decay rate at OMT1	0.1	0.1
OMK2	Fraction of organic matter decay rate at OMT2	0.99	0.99

* Recent modifications (Ruane and Hauser 2006) allow higher P:Organic matter ratios for labile organic matter. (original default value in Cole and Wells (2002) assumes a constant ratio for labile and refractory organic matter)

NUTRIENT CYCLING

Model Symbol	Description	Value	Default Value
PO4R	Phosphorus release from anaerobic sediments (fraction of SOD)	0.025	0.001
PO4S	Phosphorus settling rate (m d^{-1}) *	0.30	NA
NH4R	Ammonium release from anaerobic sediments (fraction of SOD)	0.08	0.001
NH4DK	Ammonium decay rate (d^{-1})	0.12	0.12
NH4T1	Lower temperature for ammonia decay, °*C*	5.0	5.0
NH4T2	Lower temperature for maximum ammonia decay, °*C*	25.0	25.0
NH4K1	Fraction of nitrification rate at NH4T1	0.1	0.1
NH4K2	Fraction of nitrification rate at NH4T2	0.99	0.99
NO3DK	Nitrate decay rate	0.05	0.03
NO3S	Nitrate loss to sediments due to sediment denitrification (m d^{-1})	0	1.0
NO3T1	Lower temperature for nitrate decay, □*C*	5.0	5.0
NO3T2	Lower temperature for maximum nitrate decay; □*C*	25.0	25.0
NO3K1	Fraction of denitrification rate at NO3T1	0.1	0.1
NO3K2	Fraction of denitrification rate at NO3T2	0.99	0.99
DSIR	Dissolved silica sediment release rate, fraction of SOD	0.1	0.1
PSIS	Particulate Si settling rate (m d^{-1})	0.0	1.0
PSIDK	Particulate Si decay rate	0.3	0.3
PARTSI	Dissolved Si partitioning coeff	0.2	0.0
FER	Fe release from anaerobic sediments (fraction of SOD)	0.5	0.5
FES	Fe settling velocity (m d^{-1})	2.0	2.0

* Recent modification (Ruane and Hauser 2006) include a settling rate for bioavailable P which represents the adsorption of P onto suspended sediments and the subsequent sinking out of the water column

CARBON DIOXIDE AND OXYGEN

CO2R	CO2 release from sediments (fraction of SOD)	0.1	0.1
O2NH4	Oxygen stoichiometry for nitrification	4.57	4.57
O2OM	Oxygen stoichiometry organic matter decay	1.4	1.4
O2AR	Oxygen stoichiometry for algal respiration	1.1	1.1
O2AG	Oxygen stoichiometry for algal primary production	1.4	1.4
O2LIM	O2 concentration below which anaerobic processes begin	0.1	0.1

SEDIMENT DYNAMICS

SEDC	Implements 1st-order sediment organic matter decay	ON	
SEDPRC	Records sediment organic matter to snapshot file	ON	
SEDCI	Initial sediment organic matter concentration (g m^{-2})	5.0	0.0
SEDK	Sediment organic matter decay rate (d^{-1})	0.08	0.1
FSOD	Fraction of the zero-order SOD rate used	1.0	1.0
FSED	Fraction of the first-order sediment rate used	1.0	1.0
SODT1	Lower temperature for sediment organic matter decay (oC)	4.0	4.0
SODT2	Upper temperature for sediment organic matter decay (oC)	30.0	25.0
SODK1	Fraction of sediment organic matter decay rate at SODT1	0.1	0.1
SODK2	Fraction of sediment organic matter decay rate at SODT2	0.99	0.99

REAERATION

TYPE	RIVER, LAKE, OR ESTUARY	LAKE	NA
EQN#	$Ka = 7.62U/H^{1.33}$ (Langbien and Durum 1967)	6	NA

APPENDIX D

HYDROLOGIC AND WATER QUALITY CORRELATIONS IN LAKE GREENWOOD

During the late summer and fall of 2004, several major storm events affected the hydrology and water quality of Lake Greenwood (Fig. 2, main report). The magnitude the precipitation, stream flow, and water levels in the Reedy River near Ware Shoals caused significant damage to gaging stations used to measure stream discharge (USGS 02165000) and water quality (Clemson storm water study; Klaine and Smink). In the CE-QUAL-W2 model, the concentrations during the data gaps caused by this storm damage were estimated using several correlations for stream discharge (Fig. B1), between water quality parameters (Fig. B2, B4), and between hydrology and water quality parameters (Fig. B3).

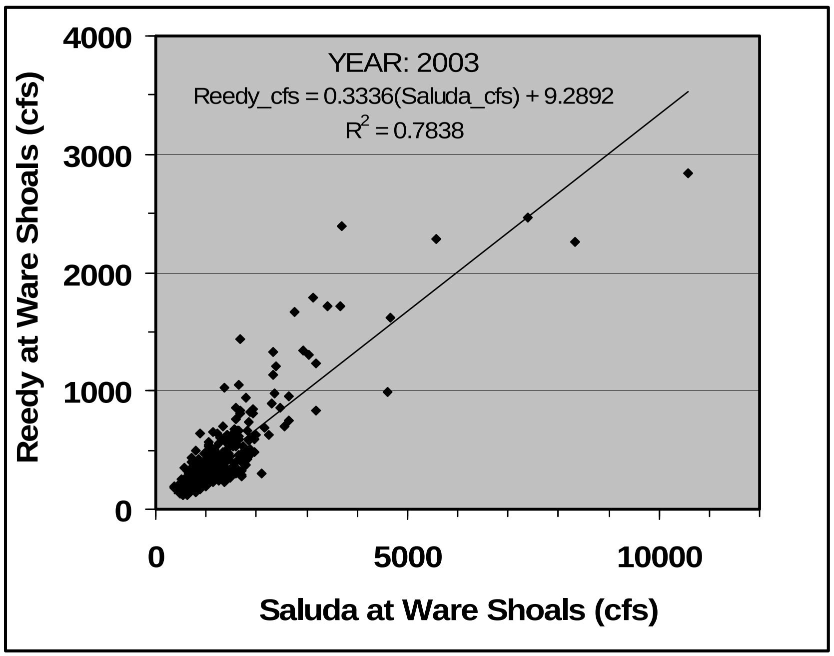

Fig. B1. Correlation between stream discharge at the Reedy River near Ware Shoals (USGS 02165000) and the Saluda River near Ware Shoals (USGS 02163500) for 2003.

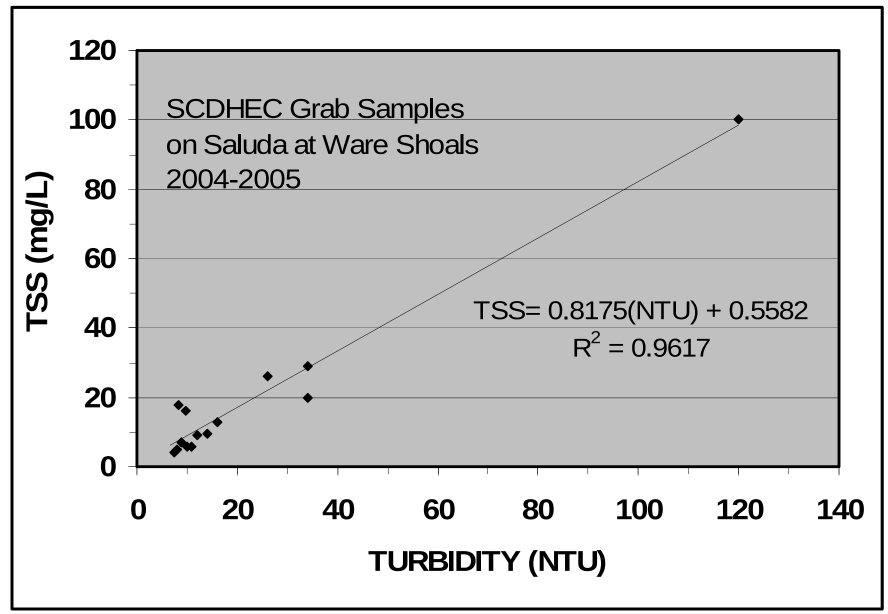

Fig. B2. Correlation between total suspended solids (TSS) and Turbidity (NTU) on Saluda at Ware Shoals (SCDHEC grab samples, 2004-2005). This function was used to estimate missing TSS concentrations on the Reedy River where NTUs were measured.

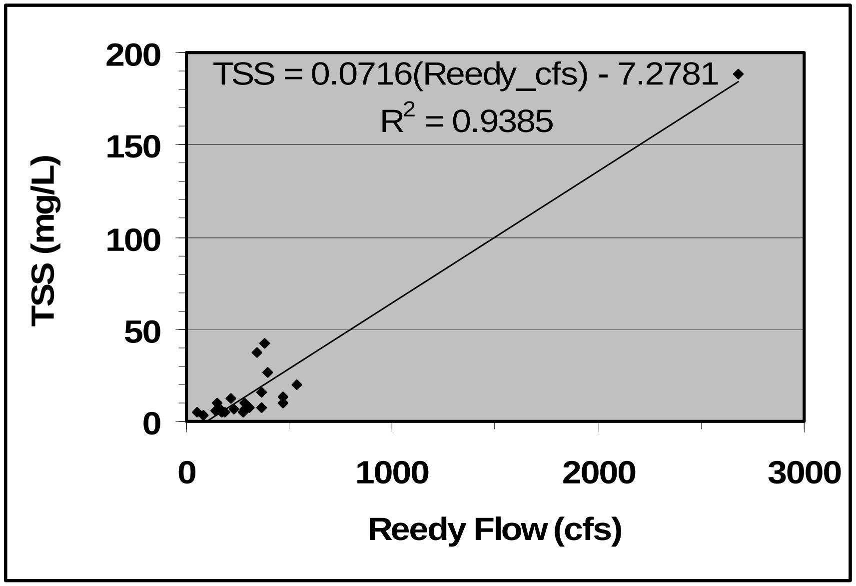

Fig. B3. Correlation between TSS and stream discharge on the Reedy River (TSS on the Reedy was based on the TSS vs NTU correlation found on the Saluda, Fig A2)

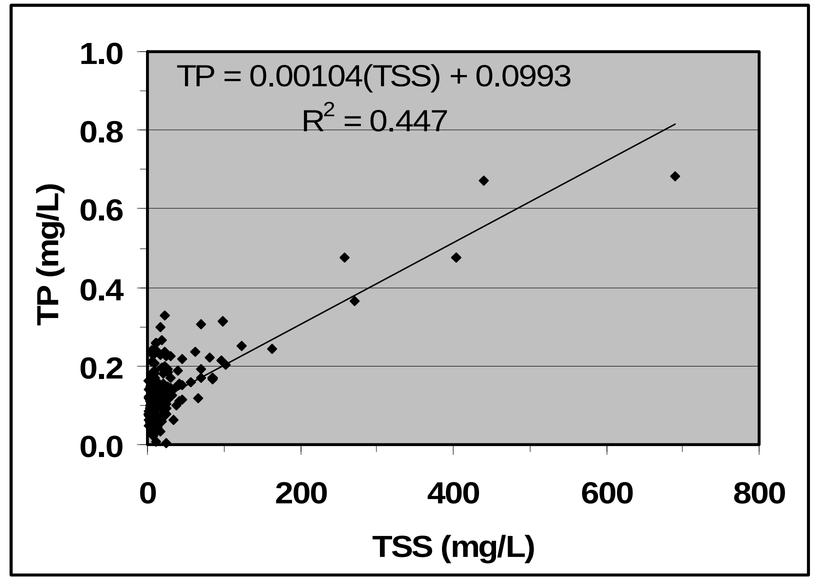

Fig. B4. Correlation between TP and TSS on the Reedy River. Daily TSS values were based on the correlation established in Fig A3 and TP concentrations were from the Clemson storm sampling project.

APPENDIX E

WATER BUDGET CALIBRATION
Gary Hauser (Loginetics, Inc.) and Hank McKellar (SC DNR)

Changes in lake water storage were calculated each day from midnight headwater elevations and the elevation-volume relationship developed from W2 input bathymetry. Ungauged inflow was then calculated as total inflow less gauged flow (Saluda River near Ware Shoals and for Reedy River near Ware Shoals)

Daily total outflow was first determined as

$$O_t = O_{tur} + O_{spill} + O_{ws}$$

where O_t = total outflow, O_{tur}= turbine flow, O_{spill}= spill flow, and O_{ws}= water supply withdrawal flow. Daily total inflow was then back-calculated as

$$I_t = O_t + \frac{dS}{dt}$$

where I_t = total inflow, O_t= total outflow, and *dS/dt*=daily storage change from midnight headwater elevations. In the above continuity equation, evaporation from the lake surface was inherently captured in the back-calculated total inflow. Thus, this total inflow was slightly lower than the actual inflow by an amount equal to the actual evaporation each day. In the W2 model, evaporative heat loss was allowed to occur, but mass loss due to evaporation was turned off when using back-calculated inflows (i.e., EVC=OFF in W2 control file).

Total ungauged inflow (minor tributaries and lakeside drainage) was then calculated as

$$I_{ug} = I_t - I_g$$

where I_{ug}= total ungauged inflow, I_t= total inflow, and I_g= total gauged inflow.
Ungauged inflow was then distributed by drainage area proportion for the remaining model inflows.

Lake Water Elevation Calibration. The back-calculation described above resulted in a daily time series of ungauged inflows that contained occasional negative flows due to minor errors in gauged inflows and headwater elevation data. Negative inflows cannot be used for W2 inputs, so ungauged inflows were adjusted by setting negative flows equal to zero. Setting negative flows to zero artificially adds inflow, which must be compensated by reducing the gauged inflows by an equal amount. Thus, we spread the error of back-calculation over all inflows, not just ungauged inflows. As a final step in the hydrologic calibration, this total ungauged flow was allocated by catchment area as “distributed” inflow representing major tributaries (Rabon, Turkey, and Cane Creeks) and lakeside drainage. A final adjustment of 0.6 m^3/s was applied during Jan-Apr (period of rapid rise in lake level), distributed as lakeside inflow along the main branch. This adjustment amounted to a mean annual correction of 0.249 m^3/s, representing only 3.1 % of the estimated total ungauged flow.

APPENDIX F

LOADING NOTES, CALCULATIONS AND ASSUMPTIONS

Concentrations of key constituent in the input loading files for the Lake Greenwood model were derived from several sources and involved several assumptions regarding constituent ratios and chemical stoichiometry. Daily loading concentrations for many constituents were derived from the ongoing Clemson storm and base-flow sampling on the Saluda and Reedy Rivers (S. Klaine and J. Smink data from 2004-2005). Other values were based on SCDHEC monthly grab samples at the same locations (DHEC 2004-05). A few additional constituents were required to run the model, but were not considered critical in the main objectives of this model, These input concentrations were assumed to be constant at levels similar to recent calibration settings for Lake Wateree and other Catawba Basin reservoirs (Ruane and Hauser, 2006;(total dissolved solids, 200 mg/L; dissolved Si, 5 mg/L; particulate Si, 1 mg/L; total iron, 1 mg/L))

MODEL SYMBOL	DESCRIPTION	DATA SOURCE	NOTES/CALCULATIONS/ASSUPTIONS
PO4	Bioavailable phosphorus	Klaine and Smink (04-05) combined with SC DHEC monitoring data (2004-05)	PO4=dissolved inorg.P + inorganic particulate-bound P; PO4=TP-organic P; where organic P=.009(sum of algal biomass, LDOM, LPOM) plus .0009(RDOM+RPOM). For Jan-June 2004, Klaine and Smink (04-04) measured total P on filtered samples (total dissolved P; TDP). For these dates, TP was estimated from TDP as TP=2.5(TDP), where the 2.5 ratio was derived from the 2-yr median ratio (TP/TDP) observed in the upper Saluda and Reedy arms (Appendix A).
NH4	NH4-N	DHEC (2004-05)	Daily concentrations based on averaged monthly grabs and assumed constant between months
NOx	(NO2 +NO3)-N	Klaine and Smink (04-05)	Based on storm event and base-flow sampling
LDOM	Labile dissolved organic matter	DHEC (2004-05) monthly grab samples for CBOD5	Assumed LDOM is indicated by the 5-day biochemical oxygen demand (CBOD-5) LDOM=2.2(CBOD-5)/1.2 where 2.2 mg/L ultimate BOD/mg/L CBOD-5 1.2 mg O2/g organic matter For cases where CBOD-5 was < detection limit of 2 mg/L), we assumed CBOD-5= 1 mg/L
RDOM	Refractory dissolved organic matter	Based on DOC values from Klaine and Smink (2004-05)	RDOM=(DOC/0.45)-LDOM, where 0.45 mg C/mg organic matter
LPOM	Labile particulate organic matter	Based on DOC (Klaine and Smink (2004-05), and TOC values from DHEC (2004-05)	POC=TOC-DOC; there were no apparent seasonal patterns or correlations with other parameters so we assumed POC=constant at the 2-yr median of 1.0 mg/L.; POM=POC/0.45 (see RDOM, above) Assumed that LPOM=0.33(POM)=0.73 mg/L
RPOM	Refractory particulate organic matter	Based on DOC (Klaine and Smink (2004-05), and TOC values from DHEC (2004-05)	RPOM=POM-LPOM (see LPOM above)
ALG2	Algal biomass Type 2 (Green algae)	(see above)	(see above)
ALG3	Algal biomass Type 3 (Cyano-bacteria)	(see above)	(see above)
DO	Dissolved oxygen	DHEC (2004-05) monthly grab samples	
ALK	Total Alkalinity	Klaine and Smink (04-05)	