

## **CHAPTER 8. PLANKTON**

Michael F. Coveney<sup>1</sup>  
John C. Hendrickson<sup>1</sup>  
Erich R. Marzolf<sup>1</sup>  
Rolland S. Fulton<sup>1</sup>  
Jian J. Di<sup>1</sup>  
Clifford P. Neubauer<sup>1</sup>  
Dean R. Dobberfuhl<sup>1</sup>  
Greeneville B. Hall<sup>1</sup>  
Hans W. Paerl<sup>2</sup>  
Edward J. Philips<sup>3</sup>

<sup>1</sup>St. Johns River Water Management District

<sup>2</sup>University of North Carolina

<sup>3</sup>University of Florida

St. Johns River Water Management District  
Palatka, Florida  
2011



*This page intentionally left blank.*



---

# TABLE OF CONTENTS

<b>TABLE OF CONTENTS .....</b>	<b>III</b>
<b>PLANKTON WORKING GROUP.....</b>	<b>V</b>
<b>LIST OF FIGURES.....</b>	<b>VI</b>
<b>LIST OF TABLES .....</b>	<b>XI</b>
<b>ACRONYMS, ABBREVIATIONS, AND CONVERSION FACTORS.....</b>	<b>XV</b>
 <b>1 ABSTRACT .....</b>	 <b>16</b>
 <b>2 INTRODUCTION .....</b>	 <b>17</b>
 <b>3 METHODS AND MATERIALS.....</b>	 <b>20</b>
3.1 Site Information .....	20
3.1.1 General Description .....	20
3.1.2 Nutrient Loading.....	24
3.1.3 Phytoplankton Communities.....	29
3.1.4 Zooplankton Communities.....	33
3.2 Data and Sample Collection and Analyses .....	34
3.2.1 Data Sets and Sampling Sites .....	34
3.2.2 Water Quality Sampling and Analyses .....	37
3.3 Hydrodynamic Model Output Data .....	39
3.4 Algal Bloom Metrics and Potential Effects .....	39
3.5 Empirical Regression Models .....	41
3.6 CE-QUAL-ICM Mechanistic Model .....	42
3.7 Assessment of Effects of Water Withdrawals .....	43
 <b>4 RESULTS AND DISCUSSION.....</b>	 <b>44</b>
4.1 Hydrology and Algal Blooms in the St. Johns River.....	44
4.1.1 Water Age .....	44
4.1.2 Phytoplankton, Nutrients, and Water Age .....	45

4.2	Algal Bloom Metrics.....	52
4.2.1	Potential Effects of Water Withdrawal .....	52
4.2.2	Magnitude of Freshwater Algal Blooms .....	55
4.2.3	Duration of Freshwater Algal Blooms .....	59
4.2.4	Change ( $\Delta$ ) in Nitrogen (N) Load .....	65
4.2.5	Marine Algal Blooms.....	66
4.2.6	Algal Bloom Metrics, Effects, and Thresholds.....	68
4.3	Assessment of Preexisting Impairment.....	71
4.4	Ecohydrological Models to Predict Algal Bloom Metrics.....	74
4.4.1	Empirical Regression Models .....	74
4.4.2	Multiple Linear Regression Model .....	76
4.4.3	Multiple Logistic Regression Model .....	78
4.4.4	CE-QUAL-ICM Mechanistic Model .....	79
4.5	Evaluation of the Effects of Water Withdrawal Scenarios on Algal Blooms.....	83
4.5.1	Ranking of Model Scenarios.....	83
4.5.2	Effects of Water Withdrawal Scenarios.....	85
4.5.3	Levels of Effects .....	94
4.5.4	Uncertainty in Assessment of Effects .....	96
4.5.5	Summary of Effects .....	98
<b>5</b>	<b>ACKNOWLEDGMENTS.....</b>	<b>103</b>
<b>6</b>	<b>REFERENCES .....</b>	<b>104</b>
<b>7</b>	<b>APPENDICES .....</b>	<b>108</b>
	Appendix 8.A. Relationships of Water Flows with Plankton and Water Quality.....	108
	Appendix 8.B. Effect of Water Withdrawal on Phytoplankton Biomass in Lake Poinsett. 108	
	Appendix 8.C. Empirical Regression Models and Goodness-of-Fit Diagnostics .....	108
	Appendix 8.D. CE-QUAL-ICM Setup, Calibration and Withdrawal Scenario Results .....	108
	Appendix 8.E. Nitrogen and Phosphorus Concentrations, Fluxes, and Loads in the St. Johns River .....	108
	Appendix 8.F. Dominant Phytoplankton Taxa at Lake George and Racy Point and Dinoflagellate Taxa in the Lower St. Johns River .....	108

## PLANKTON WORKING GROUP

Name (Organization)	Role
Michael F. Coveney (SJRWMD)	Working Group Leader, Author
John C. Hendrickson (SJRWMD)	Author
Erich R. Marzolf (SJRWMD)	Author
Rolland S. Fulton (SJRWMD)	Author
Jian J. Di (SJRWMD)	Author
Clifford P. Neubauer (SJRWMD)	Author
Dean R. Dobberfuhl (SJRWMD)	Author
Greeneville B. Hall (SJRWMD)	Author
Hans W. Paerl (University of North Carolina)	Author
Edward J. Philips (University of Florida)	Author
Edgar F. Lowe (SJRWMD)	Reviewer
Lawrence E. Battoe (SJRWMD)	Reviewer

## LIST OF FIGURES

Figure 2–1.	Processes affecting growth of phytoplankton and development of cyanobacterial blooms in river estuary systems. Most of the energy fluxes and nutrient cycling processes diagrammed in the foreground water column are occurring in all portions of the aquatic continuum from fresh water to ocean. OM = organic matter, Fe = iron, N = nitrogen, P = phosphorus (adapted from Paerl 2008). .....	19
Figure 2–2.	Conceptual diagram for analysis of potential effects of water withdrawal on plankton. Simulated retention time (water age) under various withdrawal scenarios was key input data, and a key effect was change in characteristics of phytoplankton blooms. ....	20
Figure 3–1.	Lower and middle St. Johns River and lower portion of the Ocklawaha River. Major lacustrine features of the fresh–tidal reach of the lower St. Johns River are the broad river reach downstream (north) of the confluence with the Ocklawaha River (LSJR Freshwater Reach) and Lake George. Major lakes in the middle St. Johns River are Lakes Monroe, Jesup, and Harney.....	23
Figure 3–2.	Water bodies in the St. Johns River basin impaired by excess nutrients. These water bodies have TMDLs or are listed as impaired by the Florida Department of Environmental Protection.....	25
Figure 3–3.	Average total phosphorus (TP) concentrations, TP annual load, and TP annual areal load along the St. Johns River from the upper river (south-S; Christmas) to the marine reach of the estuary (north-N; Jacksonville). Concentrations at Jacksonville reflect some dilution by seawater. Load calculations were net (i.e., concentration times discharge). Values are averages for 10 to 15 years starting in the mid 1990s; see Appendix 8.E for details. TMDL= estimated values if existing nutrient TMDLs were met. ....	27
Figure 3–4.	Average total nitrogen (TN) concentrations, TN annual load, and TN annual areal load along the St. Johns River from the upper river (south-S; Christmas) to the marine reach of the estuary (north-N; Jacksonville). Concentrations at Jacksonville reflect some dilution by seawater. Load calculations were net, (i.e., concentration times discharge). Values are averages for 10 to 15 years starting in the mid 1990s; see Appendix 8.E for details. TMDL= estimated values if existing nutrient TMDLs were met. ....	28
Figure 3–5.	Distribution of chlorophyll-a (chl-a) concentrations along the St. Johns River from the upper river (south; Lake Poinsett) to the marine reach of	



	the estuary (north; Fulton Point). Two offline lakes (Lake Jesup, Doctors Lake) are marked. Horizontal line indicates $40 \mu\text{g chl-a L}^{-1}$ , the working definition of an algal bloom for this report. Fulton Point is in segment 1, which was typically not analyzed by the Plankton Working Group; locations are identified in Table 3–3. Data set 1995 to 2005. ....	30
Figure 3–6.	(a) Mean monthly discharge as percent of annual total for the Susquehanna River at Marietta, Pennsylvania, (1932 to 2007), the Neuse River at Kinston, North Carolina, (1983 to 2007), and two locations in the lower St. Johns River basin (LSJRB): the St. Johns River at DeLand (1958 to 2006), and the North Fork of Black Creek at Middleburg (1958 to 2006). Data from USGS. (b) Mean monthly chlorophyll-a (chl-a) as percent of the annual mean concentration for Lake Apopka in the upper Ocklawaha River basin and for five locations in the LSJRB (two sites in Lake George and three in the LSJR Freshwater Reach)(1996 to 2007). ....	31
Figure 3–7.	Annual patterns in chlorophyll-a for water quality monitoring stations at the outlet of Lake George and in the LSJR Freshwater Reach (see Figure 3–1) at Racy Point, 1994 through 2006. Brown (dashed) lines indicate three years with the highest discharge, and green (solid) lines indicate three years with the lowest discharge. ....	32
Figure 3–8.	Data collection sites used in the plankton analyses. ....	36
Figure 3–9.	Averaging periods for water age variables used as independent variables in empirical regression models to predict algal bloom metrics. April to October is the entire bloom season. Periods A and B precede the development of blooms. ....	42
Figure 4–1.	Relationship between water age and inverse water age for values in the range commonly found for sites in the St. Johns River. ....	45
Figure 4–2.	Linear relationship between the inverse of monthly mean water age and monthly mean discharge. Inverse water age is positively related to discharge but without the negative values that occur for discharge during periods with reverse flows. Water age data are from Racy Point. Discharge data are from the nearby downstream site Shands Bridge (SJSR16). Both data sets were taken from calibration runs of the EFDC hydrodynamic model (1996 to 2005). ....	45
Figure 4–3.	Relationships among monthly mean values for chlorophyll-a (chl-a), water color, total phosphorus (TP), soluble reactive phosphorus (SRP), total nitrogen (TN), and dissolved inorganic nitrogen (DIN) and monthly mean water age for three locations in the freshwater lower St. Johns River: Lake George (LG), Palatka (Pal), Racy Point (Racy). Values are	

	for April to October, 1995 to 2005. Analytical uncertainty at low concentrations contributed to the increased variability in SRP and DIN at long water ages. Note logarithmic Y-axes. ....	47
Figure 4–4.	Relationships between monthly mean values for chlorophyll-a (chl-a), total phosphorus (TP), total nitrogen (TN), and water color and monthly mean water age for two locations in the middle St. Johns River: Lakes Monroe and Harney. Values are for April to October, (1995 to 2005). Note logarithmic Y-axes. ....	48
Figure 4–5.	Relationships between monthly mean values for chl-a and TP for (left) short retention times (monthly mean water age $\leq 80$ d), and (right) long retention times (monthly mean water age $> 80$ d). Data for three sites in the freshwater lower St. Johns River: Lake George (LG), Palatka (Pal), Racy Point (Racy). Values are for April to October, 1995 to 2005. Regressions were calculated as $\ln(y) = mx + b$ . Regressions for water age $\leq 80$ d had negative slopes and were borderline significant ( $p = 0.05$ ) or not significant ( $p > 0.05$ ). All regressions for water age $> 80$ d were significant with positive slopes ( $p < 0.05$ ). ....	49
Figure 4–6.	Time series for observed chlorophyll-a (chl-a) and water age for Racy Point 1995 to 2005. ....	50
Figure 4–7.	Chlorophyll-a (chl-a) measured at Lake George (LG), Palatka (Pal), and Racy Point (Racy) versus sample date (1993 to 2007). ....	51
Figure 4–8.	Percent cyanobacterial biovolume of total phytoplankton biovolume at three locations in the lower St. Johns River (i.e., Lake George, Palatka, Racy Point). Median values for growth season (April to October) samples from 1993 to 2007. Based on median values, cyanobacteria make up $> 50\%$ of biovolume at 20 to 30 $\mu\text{g chl-a L}^{-1}$ and $> 80\%$ above 40 $\mu\text{g L}^{-1}$ . ....	52
Figure 4–9.	Examples of low dissolved oxygen (DO) coincident with collapse of summer algal blooms (chlorophyll-a [chl-a]). River discharge is shown in relative units. Lower St. Johns River USGS station at Dancy Point (DO), SJRWMD station at Federal Point (chl-a), and USGS station at Buffalo Bluff (discharge).....	57
Figure 4–10.	Minimum dissolved oxygen (DO) measured using continuous recording following collapse of cyanobacterial blooms at three locations in the freshwater lower St. Johns River. Depression in DO affected the entire water column. Fifteen bloom collapse events were measured during 1999 to 2009.....	58
Figure 4–11.	Relationships among abundances of four zooplankton groups (individuals $\text{L}^{-1}$ ) and the duration of algal blooms for three locations in the lower St.	

Johns River: Lake George (LG), Palatka (Pal), and Racy Point (Racy). Samples taken during algal blooms ( $\text{chl-a} \geq 40 \mu\text{g L}^{-1}$ ), April to October (1997 to 2007).....	60
Figure 4–12. Seasonal variation in water temperature at three locations in the lower St. Johns River: Lake George, Palatka, and Racy Point (1997 to 2007). ....	61
Figure 4–13. Relationships among abundances of four zooplankton groups (individuals $\text{L}^{-1}$ ) and sampling date for three locations in the lower St. Johns River: Lake George, Palatka, and Racy Point. Filled symbols (No Bloom) indicate samples during periods with no or short duration ( $< 50$ days) cyanobacterial blooms. Open symbols (Long Bloom) indicate samples during extended blooms (bloom duration at time of sampling $\geq 50$ days) (1997 to 2007). ....	62
Figure 4–14. Relationship between abundance of cladocera (individuals $\text{L}^{-1}$ ) and the duration of algal blooms for three locations in the lower St. Johns River: Lake George (LG), Palatka (Pal), and Racy Point (Racy). Regression line is 85 <sup>th</sup> percentile fit to ln-transformed abundances using quantile regression. Samples taken during algal blooms ( $\text{chl-a} \geq 40 \mu\text{g L}^{-1}$ ), April to October (1997 to 2007). ....	64
Figure 4–15. Longitudinal pattern of dinoflagellate biovolume for monthly samples in the St. Johns River, 1998 to 2007. Dashed lines indicate salinity zones.....	67
Figure 4–16. Observed versus predicted $\text{chl-a}$ ( $\mu\text{g L}^{-1}$ ) for the multiple linear regression model used to predict freshwater bloom magnitude (maximum $\text{chl-a}$ ) from water age variables in segments 3 and 4. Line shows 1:1 relationship. Adjusted $R^2 = 0.88$ . ....	78
Figure 4–17. Prediction of the annual probability of long-duration ( $> 50$ d) bloom events for segments 3 and 4 by binomial logistic regression. Observations were coded (1 = event, 0 = no event) for regression. The coded values and predicted bloom probabilities were plotted in order of increasing probability. Two points are labeled for discussion (see text).....	79
Figure 4–18. Time series for observed $\text{chl-a}$ and continuously simulated $\text{chl-a}$ done with the mechanistic water quality model CE-QUAL-ICM for (a) Lake George and (b) Racy Point.....	81
Figure 4–19. Observed $\text{chl-a}$ versus simulated $\text{chl-a}$ based on (a) daily values, and (b) annual bloom maximum values. Values are shown for both Lake George and Racy Point. $\text{Chl-a}$ was simulated with the mechanistic water quality model CE-QUAL-ICM for 1996 to 2005. ....	82

Figure 4–20. Relationships between mean bloom season water age and observed and simulated maximum chlorophyll-a (chl-a; panels a and c) and mean chl-a:TP ratio (panels b and d). Chl-a and total phosphorus (TP) were simulated with the mechanistic water quality model CE-QUAL-ICM for 1996 to 2005. ....	83
Figure 4–21. Maximum annual bloom chl-a predicted for Lakes Harney, Jessup, and Monroe (segment 6) for baseline scenario Base1995NN and withdrawal scenario FwOR1995NN. Prediction was by linear regression model from water age variables. Because a variable for antecedent period A (see Figure 3–9) was used in the regression, no prediction could be made for 1995. FwOR1995NN ranked first by difference in water age from the baseline (see Table 4–12). Horizontal line indicates lower threshold (50 $\mu\text{g chl-a L}^{-1}$ ) for magnitude of freshwater algal bloom. ....	87
Figure 4–22. Probability of long-duration (> 50 d) algal blooms predicted for Lake George (LG12, segment 4) and Racy Point (SRP, segment 3) for baseline scenario Base1995NN and withdrawal scenario FwOR1995NN. Prediction was by logistic regression model from water age variables. FwOR1995NN ranked first by difference in water age from the baseline (Table 4–20). ....	88
Figure 4–23. Relationships between maximum bloom chlorophyll-a (chl-a) and mean April to August water age for CE-QUAL-ICM simulations of baseline (Base1995NN) and withdrawal scenarios .....	93

## LIST OF TABLES

Table 3–1.	Morphometric parameters for selected St. Johns River lacustrine reaches and lakes. Values calculated at water level 0 m for the lower St. Johns River and Lake George and at long-term mean water levels for Lakes Monroe, Jesup, and Harney. The lower St. Johns River (LSJR) Freshwater Reach is the portion of the river from Palatka (river km 128) to the city of Green Cove Springs (river km 77). .....	21
Table 3–2.	Primary plankton and water chemistry data sets available for the St. Johns River from SJRWMD monitoring programs. We used a subset of these data in the plankton analyses. ....	34
Table 3–3.	River segments, locations, and sites for the primary data sets used in analysis of potential effects of water withdrawals on plankton. Data from multiple sites per location were averaged. Site names in parentheses were sampled by Volusia County, Florida, and were used for chl-a data only. See Figure 3–8 for a map of the data collection sites. ....	35
Table 3–4.	Algal bloom metrics and the associated effects and measured variables used in the empirical regression models. See Table 4–6 for the thresholds set for each effect. ....	40
Table 4–1.	Preliminary list of potential effects of water withdrawal on plankton, nutrients, and TMDLs. ....	53
Table 4–2.	Results of screening the preliminary set of water withdrawals effects, aggregated under four algal bloom metrics for final consideration. Some effects were eliminated from further analyses for the reasons indicated. ....	55
Table 4–3.	ANOVA on ln-transformed zooplankton abundances for three locations in the lower St. Johns River: Lake George, Palatka, and Racy Point. Effects were Date (covariate) and Bloom (presence or absence of long-duration [ $\geq 50$ days] algal bloom at time of sampling). Shaded effects were significant at $p < 0.05$ . Data set June to October, 1997 to 2007. ....	63
Table 4–4.	$N_2$ -fixation in Lake George estimated from the annual N budget. We calculated a lower bound for $N_2$ -fixation as the net internal load (mass out – mass in). % $N_2$ = Estimated $N_2$ -fixation as percent of N mass leaving the lake. ....	66
Table 4–5.	Dominant taxa (all groups) during the four highest biomass dinoflagellate blooms in Doctors Lake in the St. Johns River oligohaline reach. ....	68

Table 4–6.	Summary of algal bloom metrics, the associated effects, measured variables, and the thresholds used for evaluation. Freshwater algal blooms in the St. Johns River generally were dominated by cyanobacteria, and the marine algal blooms of concern were dinoflagellates. ....	69
Table 4–7.	Filled panels indicate algal bloom metrics assessed in the St. Johns River system by river segment. Metrics include one or two (i.e., magnitude of freshwater algal blooms) potential effects of water withdrawals on plankton (see Table 4–6). We assessed both single and multiple locations depending on segment. See Section 4.2.6. for details on river segments assessed and metrics assessed in each segment. ....	71
Table 4–8.	Levels of effects used to assess preexisting impairment in the St. Johns River using the frequency of years that bloom thresholds (see Table 4–6) were exceeded in monitoring data sets for 1995 to 2005. We did not define a frequency for extreme effects. ....	73
Table 4–9.	Preexisting impairment in the St. Johns River system assessed by the same algal bloom metrics used to assess potential effects of water withdrawals (see Table 4–6). The overall assessment considered multiple locations and all bloom metrics. ....	74
Table 4–10	Eight of the nine empirical regression models to predict bloom variables (Table 4–6) from water age variables. For bloom variables freshwater maximum chl-a and freshwater bloom duration, three separate models were developed for segments 2, 3 + 4, and 6. One additional regression model was developed in Appendix 8.B for Lake Poinsett (segment 8). NA= Not applicable. ....	76
Table 4–11.	Independent variables selected for the linear multiple regression model to predict freshwater bloom magnitude (maximum chl-a) in segments 3 and 4. Standard partial regression coefficients indicate the direction (negative or positive) of the effect, and the absolute values show the relative weight of each independent variable in prediction of maximum chl-a. ....	77
Table 4–12	Summary of water withdrawal scenarios and the rank for assessment by the Plankton Working Group. We prioritized scenarios based on increase in water age at Racy Point from the baseline scenario (Base1995NN). Some scenarios were omitted from ranking; see text for details. Mgd= million gallons per day. ....	85
Table 4–13.	Predicted effects of water withdrawal scenario FwOR1995NN on algal bloom metrics. This scenario ranked first for analysis based on water age (see Table 4–12) but was an unrealistic scenario for water supply withdrawal. Effects were quantified as the percent change in annual	

	median value (%) and/or the change in annual frequency that thresholds were exceeded compared with the baseline (see Table 4–6 for thresholds).....	90
Table 4–14.	Predicted effects of water withdrawal scenario FwOR1995PN on algal bloom metrics. This scenario ranked second for analysis based on water age (see Table 4–12) but was an unrealistic scenario for water supply withdrawal. Effects were quantified as the percent change in annual median value (%) and/or the change in annual frequency that thresholds were exceeded compared with the baseline (see Table 4–6 for thresholds).....	91
Table 4–15.	Predicted effects of water withdrawal scenario FwOR2030PS on algal bloom metrics. This forecast scenario ranked third for analysis based on water age (see Table 4–12) and represented the largest modeled 2030 water withdrawal (262 mgd). Effects were quantified as the percent change in annual median value (%) and/or the change in annual frequency that thresholds were exceeded compared with the baseline (see Table 4–6 for thresholds).....	92
Table 4–16.	Effects of water withdrawal scenarios FwOR1995NN, FwOR1995PN, and FwOR2030PS on maximum algal bloom chl-a predicted using mechanistic model CE-QUAL-ICM. Effects were quantified as the percent change in annual median values compared with the baseline. Locations in segments 3 and 4 were assessed. ....	93
Table 4–17.	Levels used to characterize effects of water withdrawals based on the percent change in magnitude (where available) of a bloom metric or on the change in frequency of years that the bloom metric threshold was exceeded (see Table 4–13, Table 4–14, and Table 4–15). We did not define major or extreme effects.....	95
Table 4–18.	Uncertainty in assessment of effects of water withdrawals for each bloom metric where applied in each river segment. Ordinal scores rate the strength of three types of evidence for effects as 1 (strong) to 5 (weak) (see Chapter 2. Comprehensive Integrated Assessment). These scores are shown as follows: EFDC model output — Predictive Model   Supporting Evidence   Understanding of Mechanisms. The corresponding uncertainty codes (* very low to ***** very high) were based on the strength of evidence scores and followed definitions outlined in Chapter 2. See text for further details. ....	98
Table 4–19.	Predicted levels of effects of water withdrawal scenario FwOR1995NN on algal bloom metrics compared with the baseline (Base1995NN). This scenario ranked first for analysis based on water age (see Table 4–12) but was an unrealistic scenario for water supply withdrawal. Levels of uncertainty also are shown. ....	100

Table 4–20. Predicted levels of effects of water withdrawal scenario FwOR1995PN on algal bloom metrics compared with the baseline (Base1995NN). This scenario ranked second for analysis based on water age (see Table 4–12) but was an unrealistic scenario for water supply withdrawal. Levels of uncertainty also are shown. ....	101
Table 4–21. Predicted levels of effects of water withdrawal scenario FwOR2030PS on algal bloom metrics compared with the baseline (Base1995NN). This scenario ranked third for analysis based on water age (see Table 4–12) and represented the largest modeled 2030 water withdrawal (262 mgd). Levels of uncertainty also are shown. ....	102



## ACRONYMS, ABBREVIATIONS, AND CONVERSION FACTORS

CDFs	cumulative distribution functions
CE-QUAL-ICM model	Water quality model developed by the US Army Corps of Engineers
Chl-a	Chlorophyll-a
DIN	Dissolved inorganic nitrogen
DO	Dissolved oxygen
DOC	Dissolved organic carbon
EFDC model	Environmental Fluid Dynamics Code hydrodynamic model
FDEP	Florida Department of Environmental Protection
LSJR Freshwater Reach	Lower St. Johns River Freshwater Reach
LSJRB	Lower St. Johns River Basin
MCA	Marsh Conservation Area
MSJRB	Middle St. Johns River Basin
NAVD88	North American Vertical Datum of 1988
NO	Nitrogen oxide
PM	predictive model
PNCC	Palmer Maloney nannoplankton counting cell
RO	Reverse osmosis
SE	Supporting evidence
SJRWMD	St. Johns River Water Management District
SRP	Soluble reactive phosphorus
SR	State Road
TBD	To be determined
TKN	Total Kjeldahl nitrogen
TMDLs	Total maximum daily loads
TN	Total nitrogen
TP	Total phosphorus
TSS	Total suspended solids
UM	Understanding of mechanisms
USACE	U.S. Army Corps of Engineers
USGS	U.S. Geological Survey
USJRB	Upper St. Johns River Basin
WMA	Water Management Area
WSIS	Water Supply Impact Study

## 1 ABSTRACT

The St. Johns River Water Management District's Water Supply Impact Study evaluated potential effects of water supply withdrawals from the St. Johns River on biological and water resources. The Plankton Working Group was charged with the identification and quantification of possible environmental effects of water withdrawals on plankton communities and Total Maximum Daily Loads. Most of the potential effects that we investigated were causes or consequences of enhanced growth of phytoplankton. Consequently, phytoplankton blooms (algal blooms), primarily cyanobacteria in fresh water and dinoflagellates in brackish water, were a primary focus of our work. Important steps were to 1) determine the potential effects of water withdrawals on plankton communities, 2) determine ecological metrics to measure effects and thresholds for adverse effects, 3) develop hydroecological models to link changes in hydrology to changes in ecological metrics and other aspects of plankton communities, and 4) assess the effects of several scenarios for water withdrawal. We aggregated the effects of algal blooms into four algal bloom metrics: marine algal blooms (dinoflagellate biovolume), increase in nitrogen (N) load due to  $N_2$ -fixation, magnitude of freshwater algal blooms (maximum chlorophyll-a [chl-a] concentration), and duration of freshwater algal blooms. We set thresholds for adverse effects for each algal bloom metric using literature information and analyses of St. Johns River data. For example, we set two thresholds for maximum chlorophyll-a concentration in freshwater blooms: a lower threshold at  $50 \mu\text{g chl-a L}^{-1}$  based on dominance by cyanobacteria and risk for cyanotoxins and a higher threshold at  $138 \mu\text{g chl-a L}^{-1}$  based on depletion of dissolved oxygen from collapsing blooms. We derived a threshold for duration of freshwater blooms, 50 days continuous duration above  $40 \mu\text{g chl-a L}^{-1}$ , based on a finding of low abundance of cladocera (but not other zooplankton groups) during long-duration blooms. We used empirical multiple linear or multiple logistic regression models (empirical regression models) to predict algal bloom metrics from independent variables based on water age, a simulated measure of residence time. This step required nine empirical regression models to cover the four algal bloom metrics in five river segments. Models included two to seven independent variables, and adjusted coefficient of determination ( $R^2$ ) values for linear regressions varied from 0.74 to 0.97. We used the mechanistic water quality model CE-QUAL-ICM as an independent check on the empirical regression models. EFDC hydrodynamic model simulations were done for 19 water withdrawal scenarios that we ranked by their effects on water age. Scenarios included both unrealistic and realistic conditions. We assessed the three top-ranked withdrawal scenarios with the regression models to compare algal bloom metrics under baseline and withdrawal conditions. Using our algal bloom metrics and monitoring data, we found preexisting major impairment in four of the five segments of the St. Johns River that we assessed. When we compared withdrawal scenarios to baseline conditions, the two top-ranked but unrealistic scenarios (FwOR1995NN and FwOR1995PN) had minor or negligible effects on algal bloom metrics. The top-ranked realistic scenario (FwOR2030PS) showed negligible effects, with no changes in algal bloom metrics greater than + 3.7% magnitude or + 0.1 annual frequency of exceedence. Lower ranked scenarios would have similar, negligible effects. CE-QUAL-ICM model results confirmed the small effects of withdrawal scenarios on bloom magnitude. We judged the uncertainty associated with these assessments of effects to be low or very low.

## 2 INTRODUCTION

The St. Johns River Water Management District (SJRWMD) Water Supply Impact Study (WSIS) evaluated potential effects of water supply withdrawals from the St. Johns River on biological and water resources. Objectives were 1) to determine acceptable limits (thresholds) for those environmental conditions that would be affected by water withdrawals, and 2) to assess effects of different withdrawal scenarios to determine whether any would cause significant adverse impacts to important biological communities, species, or water resource values. Further information on the WSIS is found in Chapter 2. Comprehensive Integrated Assessment.

Plankton (phytoplankton and zooplankton) are of fundamental importance to the food web in aquatic ecosystems (Carpenter and Kitchell 1993, Scheffer 1998). Phytoplankton are the major primary producers and form the base of the food web in most pelagic waters. Virtually all fish are dependent on zooplankton as a food source during a period after hatching, and some fish rely heavily on zooplankton during other parts of their lifecycle.

Nutrient enrichment increases phytoplankton abundance and can lead to a phytoplankton community dominated by undesirable bloom-forming taxa, such as cyanobacteria and red-tide dinoflagellates, which can have deleterious effects on herbivorous zooplankton and on higher trophic levels (Scheffer 1998, Paerl et al. 2001a, Paerl and Fulton 2006). Furthermore, phytoplankton blooms (algal blooms) characteristic of enriched nutrient conditions can shade desirable submersed plants, contribute to low dissolved oxygen (DO), increase nitrogen (N) loading through N<sub>2</sub>-fixation, and produce toxins (Scheffer 1998, Paerl et al. 2001a, Paerl and Fulton 2006).

Hydrologic conditions also affect planktonic communities in lakes, rivers, and estuaries (see Appendix 8.A). Export of nutrients and dissolved organic matter from floodplains and watersheds increases with increased rainfall. Higher flows generally mean greater water velocities in rivers and shorter residence times through river reaches and riverine lakes. Conversely, lower flows result in longer water residence times and greater phytoplankton nutrient uptake, increasing the potential for blooms (Phlips et al. 2002). Freshwater inputs to estuaries help determine density stratification and, therefore, the resistance to wind-driven mixing. These interacting processes affect growth of phytoplankton and particularly development of cyanobacterial blooms in river estuary systems (see [Figure 2 – 1](#)). Because of the direct and indirect effects of water flows, withdrawals of water could affect nutrient concentrations and plankton communities in the St. Johns River.

The Plankton Working Group was charged with the identification and quantification of possible environmental impacts of water withdrawals on plankton communities and total maximum daily loads (TMDLs) in the St. Johns River.

Most of the potential impacts we investigated were causes or consequences of enhanced phytoplankton growth. Consequently, phytoplankton bloom dynamics were a primary focus of our work. The following were important steps:

- (1) Determine the suite of potential effects of water withdrawals on plankton communities

- (2) Ascertain ecological metrics to measure potentially important effects and set thresholds for adverse effects
- (3) Develop hydroecological models to link changes in hydrology to changes in the ecological metrics and other aspects of plankton communities
- (4) Assess the effects of several scenarios for water withdrawal.

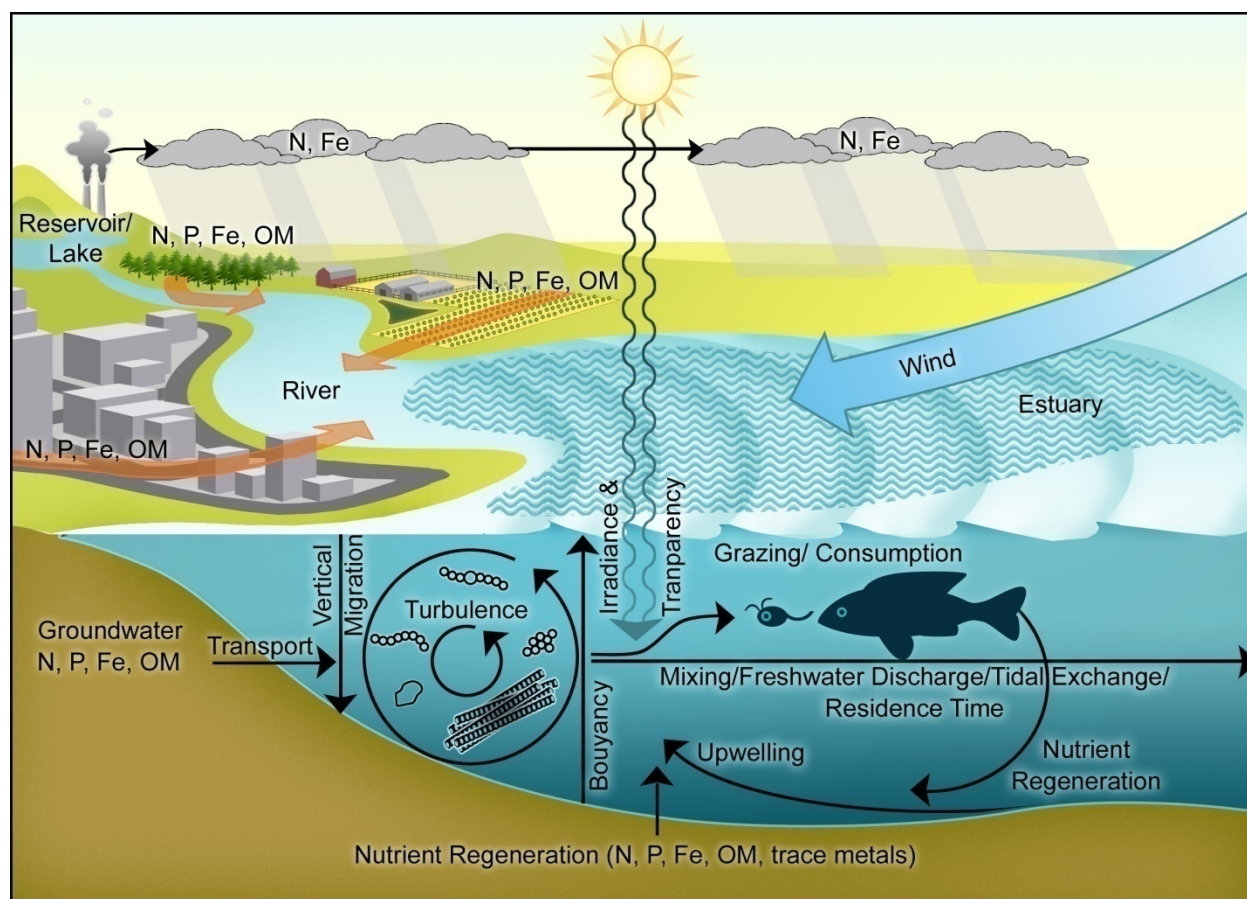


Figure 2–1. Processes affecting growth of phytoplankton and development of cyanobacterial blooms in river estuary systems. Most of the energy fluxes and nutrient cycling processes diagrammed in the foreground water column are occurring in all portions of the aquatic continuum from fresh water to ocean. OM = organic matter, Fe = iron, N = nitrogen, P = phosphorus (adapted from Paerl 2008).

We summarized the plausible chain of causation underlying our work, our use of hydroecological models, and the primary linkages to other working groups in a conceptual model (Figure 2–2). The primary causal relationships for our investigation were increased withdrawal → lower discharge → increased retention time → increased algal blooms. Algal blooms could have several adverse effects on other biota, water quality, or water resource values (Figure 2–2).

We used two types of hydroecological models to relate changes in hydrology to changes in algal blooms: empirical multiple linear or multiple logistic regression models (empirical regression models) and a mechanistic water quality model (CE-QUAL-ICM model). Our primary tools were empirical regression models that related retention time (water age) variables to algal bloom metrics. We also used the CE-QUAL-ICM model linked to the EFDC hydrodynamic model to predict algal bloom density (Figure 2–2). EFDC hydrodynamic model simulations of water withdrawals, including modeled water age, were conducted by the River Hydrodynamics Working Group and Watershed Hydrology Working Group (see Chapter 6. River Hydrodynamics Results) and provided primary input data to our work. The Biogeochemistry Working Group assessed changes in nutrient loading resulting from water withdrawals (see

Chapter 7. Biogeochemistry). Finally, predicted changes in algal bloom metrics were considered by other working groups, especially Submersed Aquatic Vegetation (see Chapter 9. Submersed Aquatic Vegetation), Benthic Macroinvertebrates (see Chapter 11. Benthic Macroinvertebrates), and Fish (see Chapter 12. Fish) (Figure 2–2).

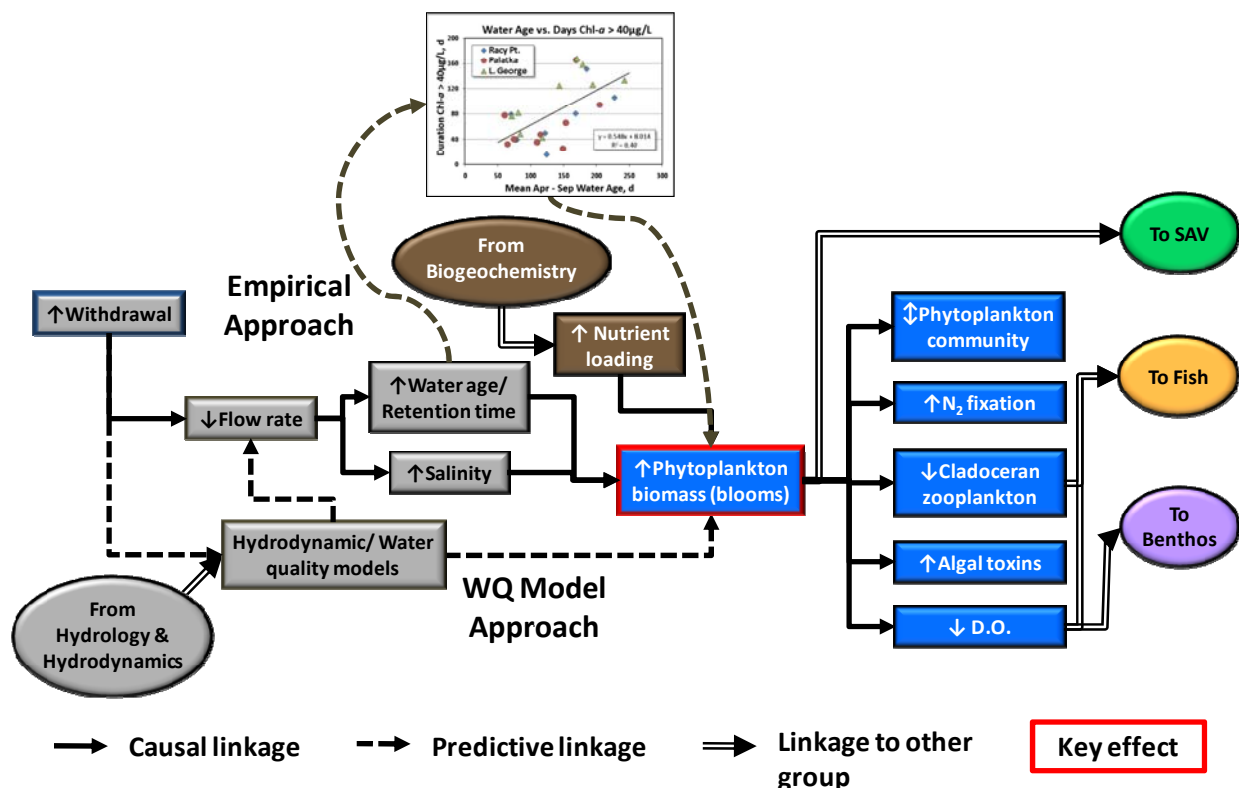


Figure 2–2. Conceptual diagram for analysis of potential effects of water withdrawal on plankton. Simulated retention time (water age) under various withdrawal scenarios was key input data, and a key effect was change in characteristics of phytoplankton blooms.

## 3 METHODS AND MATERIALS

### 3.1 SITE INFORMATION

#### 3.1.1 GENERAL DESCRIPTION

The primary study area for the Plankton Working Group was the St. Johns River, including the lower St. Johns River and Lake George and the middle St. Johns River upstream from Lake George (see Figure 3 – 1). The middle St. Johns River includes three lakes (Lakes Jesup, Monroe, and Harney) and two major tributaries. A part of our analyses was conducted in Lake Poinsett, the most downstream lake in the upper St. Johns River, which discharges to Lake Harney.

The lower St. Johns River is a river estuary with significant lacustrine characteristics due to the relatively low elevation of its headwaters and its low discharge relative to its depth and volume. The two predominant lacustrine features of the fresh tidal reach of the St. Johns River estuary are the broad river reach downstream (north) of the confluence of the large tributaries of the Ocklawaha River and Dunns Creek, referred to as the lower St. Johns River (LSJR) Freshwater Reach, and Lake George (Table 3–1, Figure 3–1).

Severe algal blooms in the fresh tidal lower St. Johns River have led to impaired water quality conditions, resulting in the establishment of a TMDL (USEPA 2008) to limit the amounts of nitrogen (N) and phosphorus (P) loading. No regulatory nutrient limit has yet been established for Lake George, although the lake is listed as impaired for nutrients (FDEP 2008) (Figure 3–2).

Table 3–1. Morphometric parameters for selected St. Johns River lacustrine reaches and lakes. Values calculated at water level 0 m for the lower St. Johns River and Lake George and at long-term mean water levels for Lakes Monroe, Jesup, and Harney. The lower St. Johns River (LSJR) Freshwater Reach is the portion of the river from Palatka (river km 128) to the city of Green Cove Springs (river km 77).

Reach/Lake	Surface Area km <sup>2</sup>	Mean Depth m	Volume 10 <sup>6</sup> m <sup>3</sup>	Reference Stage m NAVD88
LSJR freshwater reach	180	2.9	528	0
Lake George	210	2.8	581	0
Lake Monroe	45.6	1.8	83.3	0.34
Lake Jesup	42.6	1.3	53.7	0.39
Lake Harney	26.8	2.1	56.9	0.61
Lake Poinsett	13.6	2.0	27.4	3.28

The middle St. Johns River extends from the Econlockhatchee River in Osceola, Orange, and Seminole counties northward into Lake and Volusia counties—heavily urbanized areas in central Florida (Figure 3–1). The middle St. Johns River basin (MSJRB) covers approximately 3,120 km<sup>2</sup>, includes 60 km of the St. Johns River and comprises five watersheds: the Econlockhatchee River, the Wekiva River, Lake Jesup, Lake Monroe, and Lake Harney, including Deep Creek. Lakes Harney and Monroe were formed by a broadening of the St. Johns River.

Three water bodies in the middle St. Johns River had TMDLs adopted in 2006: Lake Jesup, Crane Strand (Little Econlockhatchee River watershed), and Long Branch (Econlockhatchee River watershed). TMDLs were developed in 2007 for the Wekiva River, Rock Springs Run, and other water bodies in the Wekiva River Study Area as a result of the Wekiva Parkway and Protection Act of 2004. Lakes Harney and Monroe and the St. Johns River reaches between them are listed as impaired for nutrients (FDEP 2008), and the Florida Department of Environmental Protection (FDEP) adopted a TMDL for the area in 2009 (Gao 2009) (Figure 3–2).

The upper St. Johns River basin (USJRB), extending from the headwaters of the St. Johns River in Indian River and Okeechobee counties to the confluence of the St. Johns and Econlockhatchee rivers in Seminole County, originally contained more than 1,600 km<sup>2</sup> of floodplain marsh. By the



early 1970s, 62 % of the marsh had been drained for agricultural and flood control purposes. Canals were constructed to divert floodwaters from the basin to the Indian River Lagoon. Impacts included a loss of water storage areas, diminished water quality, excessive freshwater discharges going into the Indian River Lagoon, and significant decreases in fish and wildlife populations. The marsh that remained was further degraded by hydrologic alterations and nutrients in agricultural runoff (Miller et al. 1998).

In response to flooding problems in the basin, a structural federal flood control project was started in the 1960s but was halted because of potential environmental harm. During the late 1980s, this project was transformed into the USJRB project, cosponsored by the SJRWMD and the U.S. Army Corps of Engineers (USACE). The upper basin project is a semi-structural system of water management areas (WMAs), marsh conservation areas (MCAs), and marsh restoration areas covering approximately 674 km<sup>2</sup> in Indian River and Brevard counties. The system is designed to reduce damage from floods, improve water quality, reduce freshwater discharges to the Indian River Lagoon, provide for water supply, and restore or enhance wetland habitat. The downstream portion of the upper basin includes a chain of medium-sized lakes, of which Lake Poinsett is the largest and furthest downstream. It was included in this evaluation due to the location of a potential water withdrawal site just downstream of the lake.



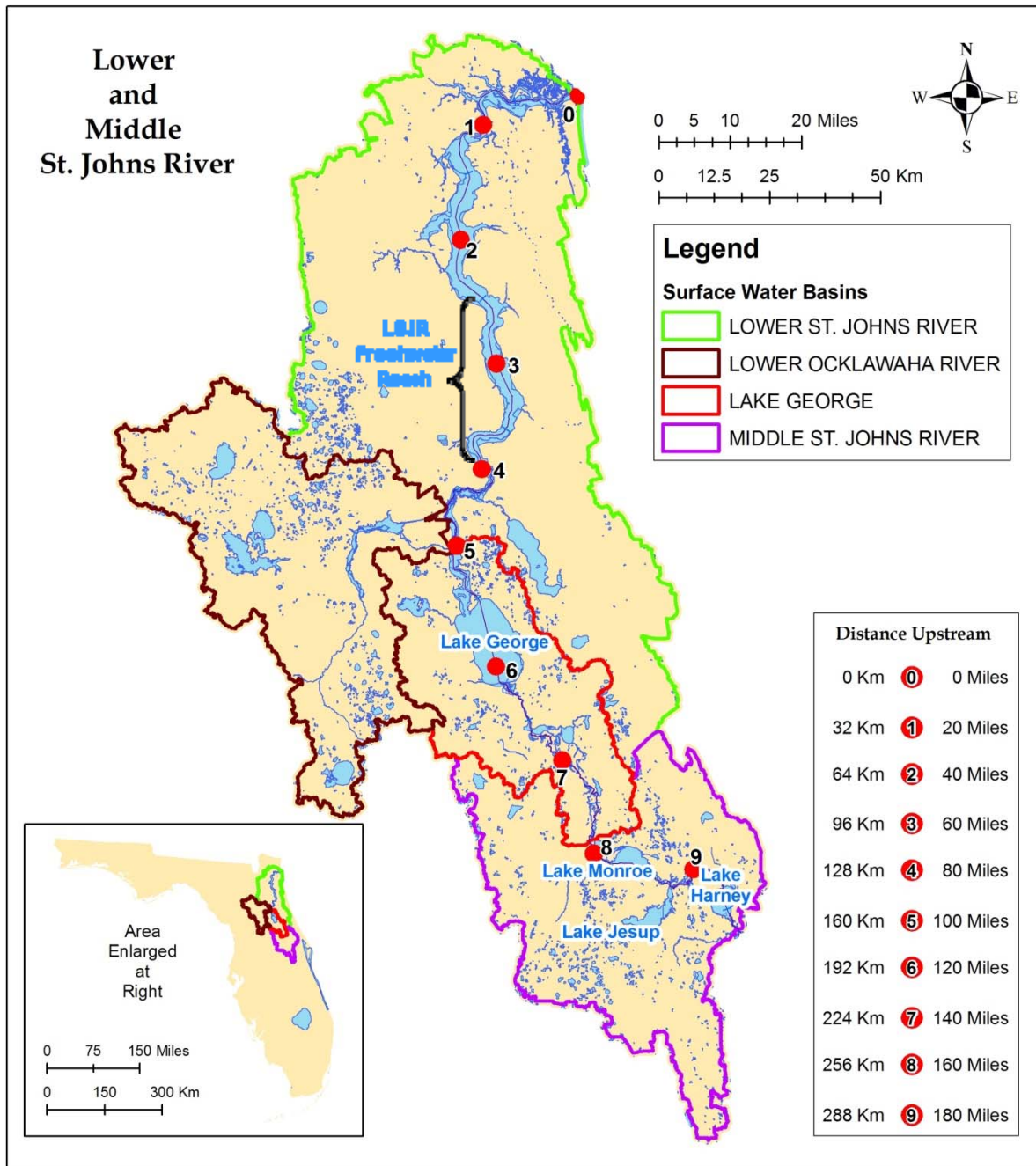


Figure 3–1. Lower and middle St. Johns River and lower portion of the Ocklawaha River. Major lacustrine features of the fresh–tidal reach of the lower St. Johns River are the broad river reach downstream (north) of the confluence with the Ocklawaha River (LSJR Freshwater Reach) and Lake George. Major lakes in the middle St. Johns River are Lakes Monroe, Jesup, and Harney.

### 3.1.2 NUTRIENT LOADING

Most reaches of the St. Johns River are subject to current nutrient TMDLs or are listed as impaired by the FDEP (Figure 3–2). Average concentrations of total phosphorus (TP) were high, with 70 to 100  $\mu\text{g L}^{-1}$  in the upper reaches and 140  $\mu\text{g L}^{-1}$  in the lower part of the estuary (Figure 3–3). Annual TP loads increased progressively downstream as watershed area and discharge increased. In contrast, areal TP loads were relatively constant until loads increased sharply in the Jacksonville area due to greater urban density and point sources (Figure 3–3). Total nitrogen (TN) concentrations had a different pattern with a gradual decline downstream (Figure 3–4). Although TN loads increased downstream because of increased discharge, areal loading of TN was relatively constant.

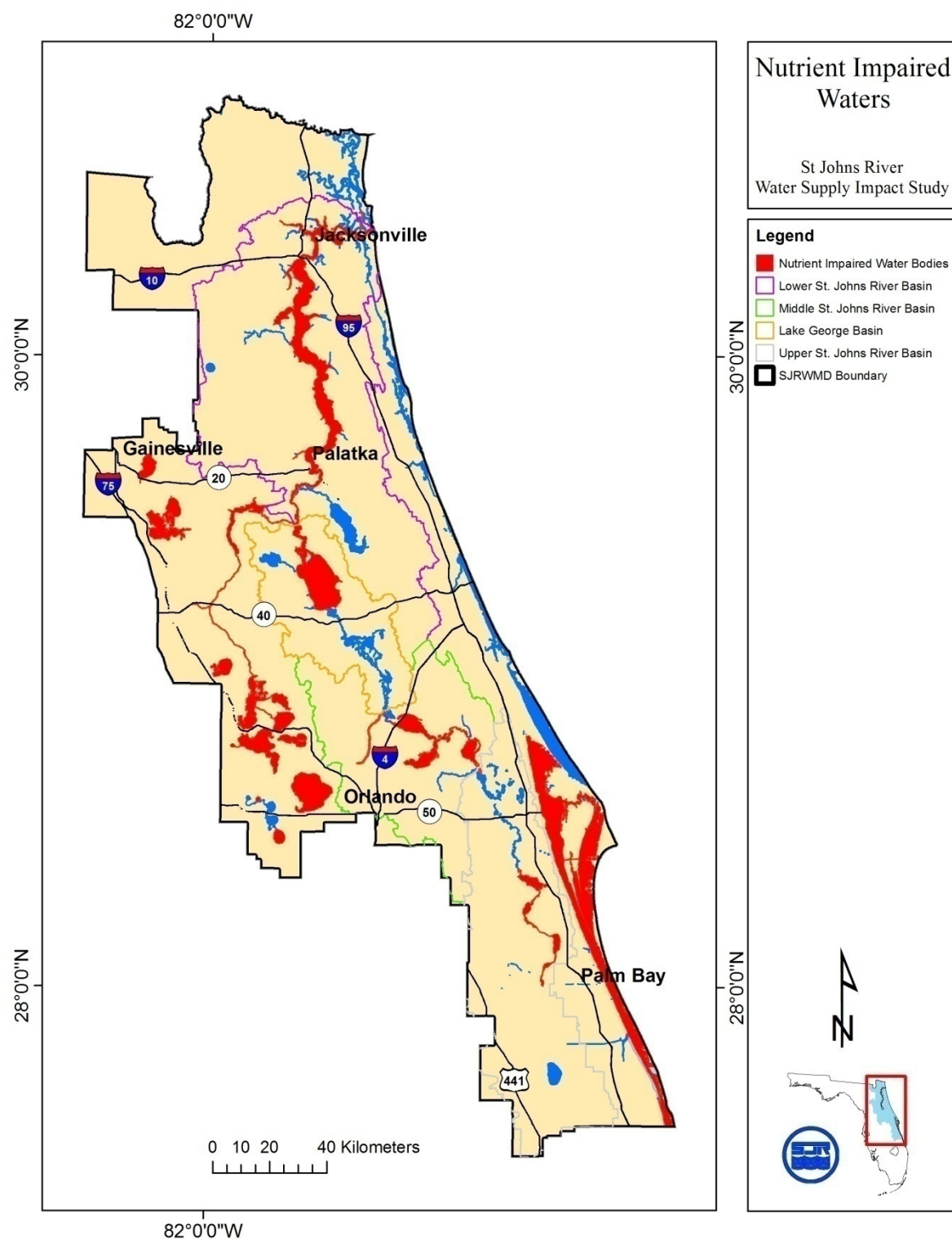


Figure 3–2. Water bodies in the St. Johns River basin impaired by excess nutrients. These water bodies have TMDLs or are listed as impaired by the Florida Department of Environmental Protection.

Primary sources of anthropogenic N and P in the lower St. Johns River basin (LSJRB) included domestic and industrial waste water and urban and agricultural nonpoint source runoff. Anthropogenic loads made up 53% of the average annual incoming TN load and 71% of the TP load (1995 to 1999). Anthropogenic load was higher in inorganic and labile organic nutrient forms that rendered its effect on eutrophication greater than that of natural background contributions (Hendrickson et al. 2003, 2007). Established TMDLs will require a 32% reduction in anthropogenic loading of both N and P in the freshwater reach of the lower St. Johns River, and a 47% decrease of anthropogenic TN in the marine reach.

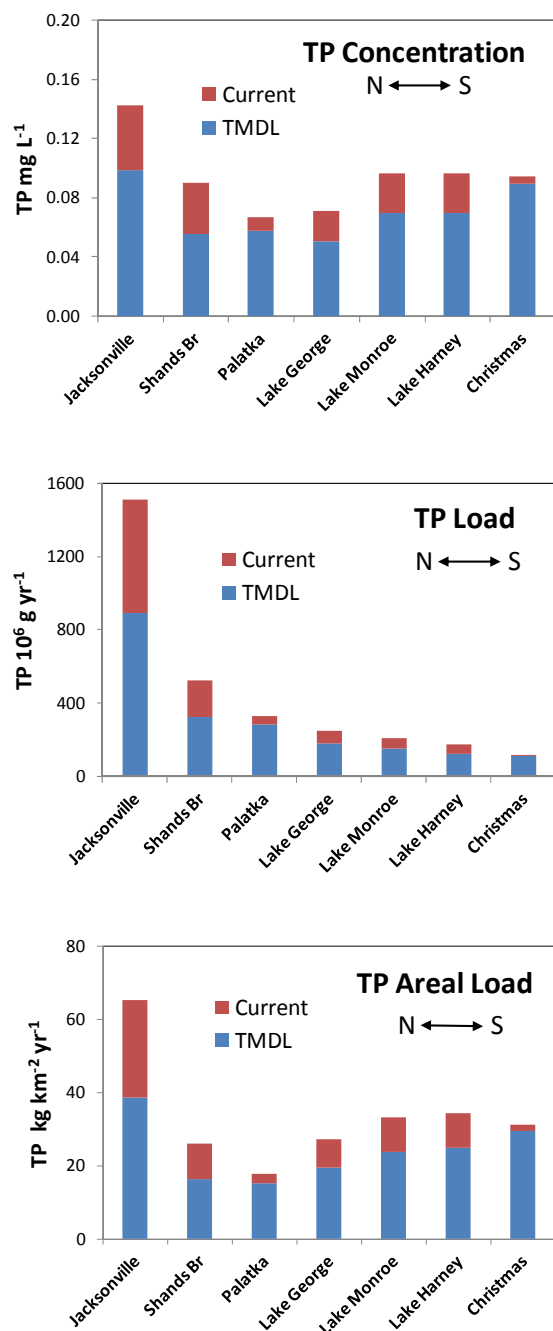


Figure 3–3. Average total phosphorus (TP) concentrations, TP annual load, and TP annual areal load along the St. Johns River from the upper river (south-S; Christmas) to the marine reach of the estuary (north-N; Jacksonville). Concentrations at Jacksonville reflect some dilution by seawater. Load calculations were net (i.e., concentration times discharge). Values are averages for 10 to 15 years starting in the mid 1990s; see Appendix 8.E for details. TMDL= estimated values if existing nutrient TMDLs were met.

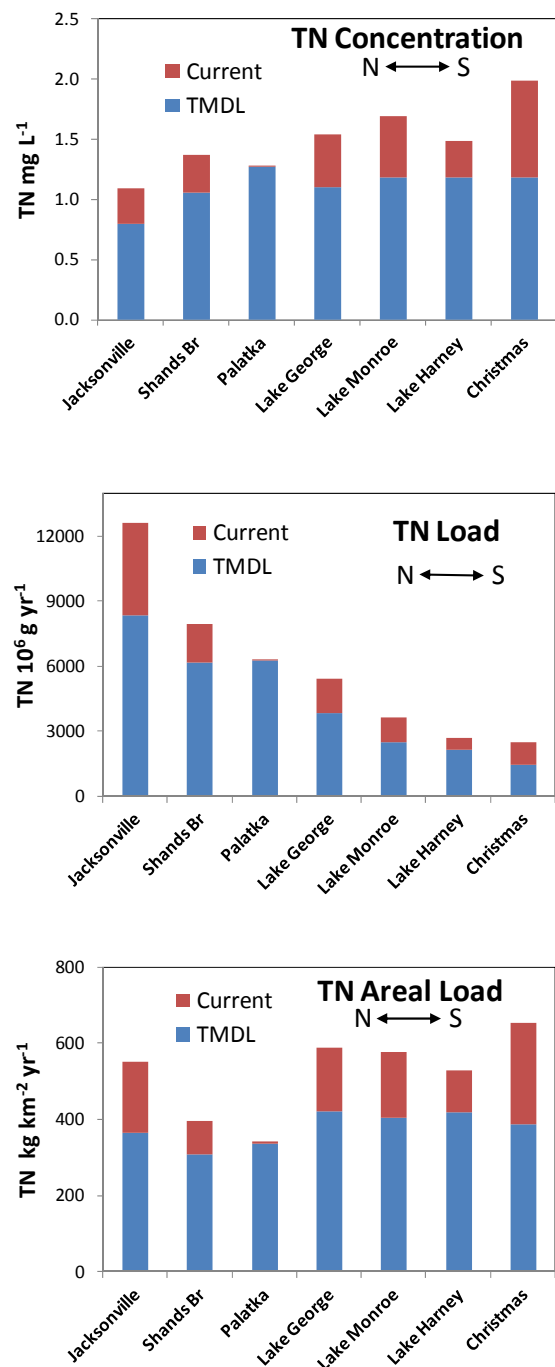


Figure 3–4. Average total nitrogen (TN) concentrations, TN annual load, and TN annual areal load along the St. Johns River from the upper river (south-S; Christmas) to the marine reach of the estuary (north-N; Jacksonville). Concentrations at Jacksonville reflect some dilution by seawater. Load calculations were net, (i.e., concentration times discharge). Values are averages for 10 to 15 years starting in the mid 1990s; see Appendix 8.E for details. TMDL= estimated values if existing nutrient TMDLs were met.

A combination of past, unregulated land use, waste disposal practices, and increased quantities of stormwater runoff have negatively impacted waterways in the middle St. Johns River. Lake Jesup is one of the most polluted lakes directly connected to the St. Johns River, as evidenced by high levels of chl-a (Figure 3–5). Since the early 1900s, two canals, multiple bridge spans, and the State Road (SR) 46 bridge restricted water flow between Lake Jesup and the St. Johns River. In 2010 a new high-span bridge was completed across the entire confluence floodplain. This project included the removal of filled causeways that had restricted flow paths between the lake and river for nearly a century and allowed for the reopening of other historic flow paths. In addition to agricultural runoff, Lake Jesup receives stormwater runoff from urban portions of the watershed, including portions of Orlando, Sanford, Winter Park, Winter Springs, Longwood, Oviedo, and Altamonte Springs. The Lake Monroe watershed is heavily developed, although wetlands exist on the east and west shores of the lake. Lake Monroe receives wet weather discharges of wastewater effluent from the city of Sanford. Lake Harney and the Deep Creek–Lake Ashby system make up the Deep Creek watershed. The Deep Creek watershed has the least impact from urbanization of any of the watersheds in the MSJRB; however, existing water quality does need to be improved.

### 3.1.3 PHYTOPLANKTON COMMUNITIES

The lower St. Johns River is the southernmost river estuary of the south Atlantic region. It is unique within this assemblage of rivers in that its entire drainage area is within the Atlantic coastal plain, unlike river estuaries to the north that drain portions of the piedmont and Appalachian provinces. This physiographic setting results in watershed inputs that are high in colored dissolved organic matter and relatively low in inorganic ions, including dissolved silicate. Nonetheless, St. Johns River water can exhibit substantial hardness and alkalinity due to the large contribution of artesian spring flows. Spring inflows also dilute colored organic matter, which can be important during low discharge periods.

The timing of the St. Johns River annual runoff pulse also departs from that typically associated with Virginian Province and south Atlantic river estuaries. High runoff in the St. Johns River occurs in late summer and early fall, as opposed to the late winter to early spring “freshet” pattern (Figure 3–6). The low flow phase occurs earlier in the year, typically from April through June. The interplay between the relative contribution of watershed runoff and artesian spring flow tends to accentuate the annual phytoplankton productivity cycle that is characteristic of the runoff pulse and subsequent dry-down pattern of river estuaries. Seasonal variability in phytoplankton biomass is greater for short-residence-time lower St. Johns River sites than for longer residence time upstream lakes in the Ocklawaha River basin, the major tributary to the lower St. Johns River (Figure 3–6). The pulsed pattern of water inflow to the St. Johns River contributes to a succession in phytoplankton species dominance (Phlips and Cichra 2002).

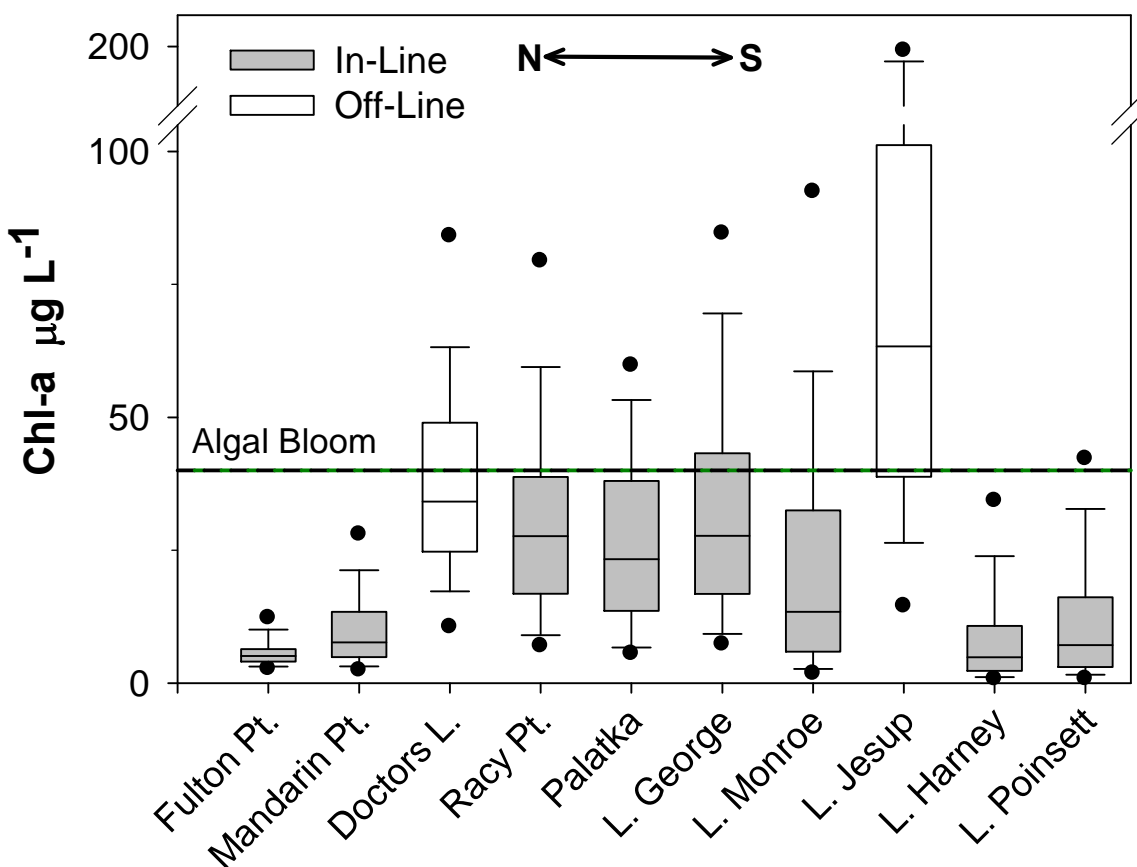


Figure 3–5. Distribution of chlorophyll-a (chl-a) concentrations along the St. Johns River from the upper river (south; Lake Poinsett) to the marine reach of the estuary (north; Fulton Point). Two offline lakes (Lake Jesup, Doctors Lake) are marked. Horizontal line indicates  $40 \mu\text{g chl-a L}^{-1}$ , the working definition of an algal bloom for this report. Fulton Point is in segment 1, which was typically not analyzed by the Plankton Working Group; locations are identified in Table 3–3. Data set 1995 to 2005.



The St. Johns River shows maximal phytoplankton biomass within the fresh tidal reaches, with maximal chlorophyll-a (chl-a) in excess of  $75 \mu\text{g L}^{-1}$  (see Figure 3–5). Algal blooms are common occurrences along the main stem of the St. Johns River from Lake Monroe through the fresh tidal reach north of Racy Point as well as in offline lakes such as Lakes Jesup and Doctors (see Figure 3–5).

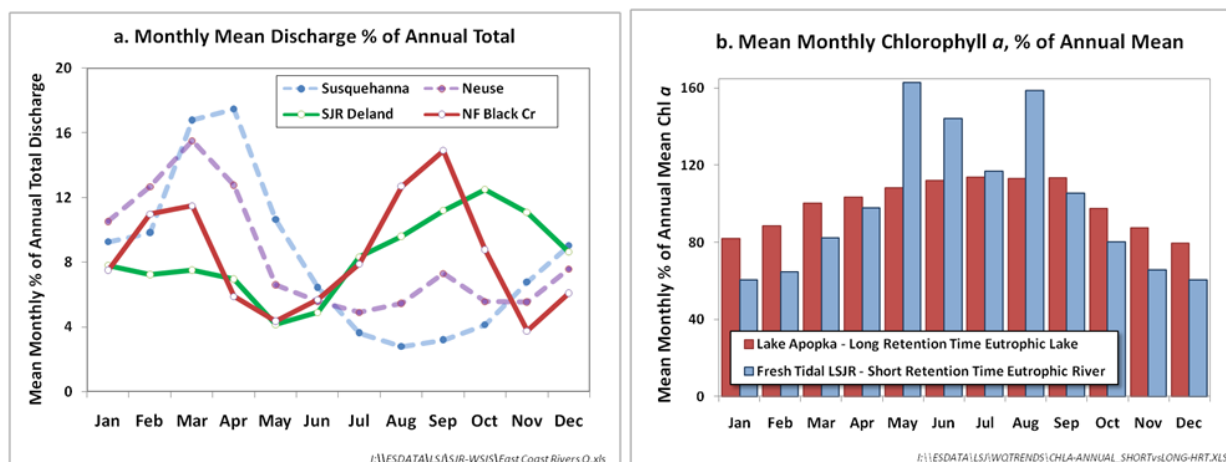


Figure 3–6. (a) Mean monthly discharge as percent of annual total for the Susquehanna River at Marietta, Pennsylvania, (1932 to 2007), the Neuse River at Kinston, North Carolina, (1983 to 2007), and two locations in the lower St. Johns River basin (LSJRB): the St. Johns River at DeLand (1958 to 2006), and the North Fork of Black Creek at Middleburg (1958 to 2006). Data from USGS. (b) Mean monthly chlorophyll-a (chl-a) as percent of the annual mean concentration for Lake Apopka in the upper Ocklawaha River basin and for five locations in the LSJRB (two sites in Lake George and three in the LSJR Freshwater Reach)(1996 to 2007).

The timing of annual chl-a maxima and change points in the sequence of community succession are related to both antecedent and concurrent hydrologic conditions. A contrast is evident between low discharge years and high discharge years, with low discharge years exhibiting earlier bloom initiation, higher maximum biomass, and longer sustained high biomass or secondary biomass peaks (Figure 3–7). During dry years, the spring bloom begins exponential growth in early April. In wet years, initiation of the bloom typically is delayed until June. Annual phytoplankton chl-a maxima in the lower St. Johns River are attained sometime between the months of May and August (Figure 3–7). Annual chl-a minima occur between the months of December and February and typically are an order of magnitude less than the annual maxima. During periods with generally low phytoplankton density, lakes in the middle St. Johns River and Lake George harbored higher concentrations of phytoplankton than the riverine portions and provided a ready inoculum to downstream reaches of the lower St. Johns River.

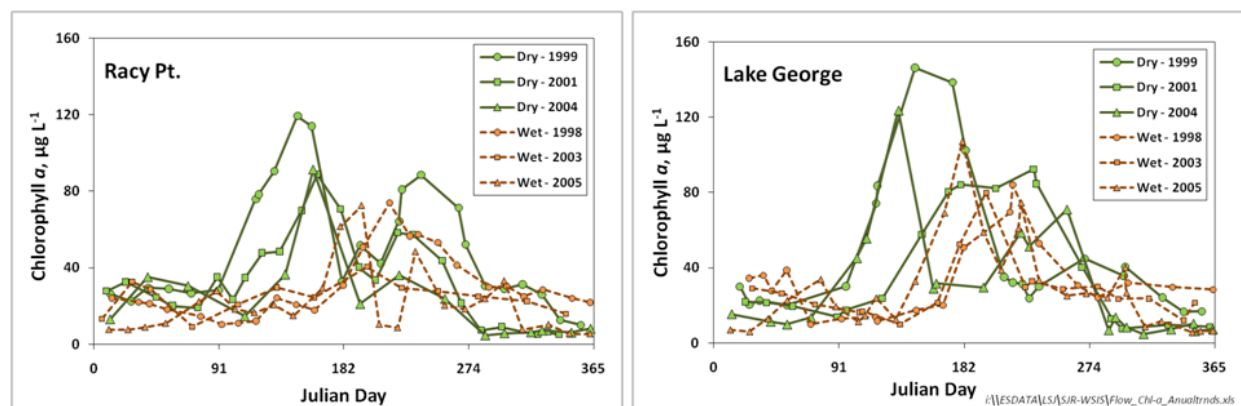


Figure 3–7. Annual patterns in chlorophyll-a for water quality monitoring stations at the outlet of Lake George and in the LSJR Freshwater Reach (see Figure 3–1) at Racy Point, 1994 through 2006. Brown (dashed) lines indicate three years with the highest discharge, and green (solid) lines indicate three years with the lowest discharge.

Phytoplankton enumeration and identification and nutrient enrichment assays (Phlips and Cichra 2002, Schelske and Aldridge 1994, Paerl et al. 2001b) performed in Lake George and the LSJR Freshwater Reach characterized a typical phytoplankton succession in which diatoms dominated in winter and early spring followed by increasing dominance by cyanobacteria. A nitrogen-fixing cyanobacterial community with one or two codominant species represented the culmination of this successional sequence. On average, the top three species constituted roughly 60% of the total phytoplankton biovolume from April through August. Following high flow fall and winter runoff pulses, the late winter phytoplankton community was dominated by diatoms, and to a lesser degree, green algae (Phlips et al. 2007). Conversely, low flow conditions in winter were associated with greater cyanobacterial dominance. Typical winter dominants included diatoms of the class Coscinodiscophyceae and *Aulacoseira* spp., with *Planktolyngbya* spp. being the most commonly observed cyanobacteria. The most commonly observed genera in the spring and summer included the cyanobacteria *Oscillatoria*-type spp. (including *Limnothrix* spp. and *Planktothrix* spp.), *Cylindrospermopsis*, *Microcystis*, *Lyngbya*, and *Anabaena*. A summary of the dominant phytoplankton at Lake George and Racy Point is found in Appendix 8.F.

In the middle St. Johns River, Lake Harney's phytoplankton assemblage was composed of non-diatom chrysophytes, diatoms, cryptophytes, and cyanobacteria, with blooms being dominated by cyanobacteria or non-diatom chrysophytes. Common genera include *Anabaena*, *Planktolyngbya*, *Rhodomonas*, *Mallomonas*, *Stephanocyclus*, and *Skeletonema*. Lake Monroe's phytoplankton was composed of diatoms, cryptophytes, and cyanobacteria, with blooms dominated by cyanobacteria of the genera *Cylindrospermopsis*, *Planktolyngbya*, *Anabaena*, and *Microcystis*. Common non-cyanobacteria genera included *Cryptomonas*, *Nitzschia*, *Staurosirella*, and *Aulacoseira*. The Lake Jesup phytoplankton community was dominated by cyanobacteria, particularly of the genera *Cylindrospermopsis*, *Planktolyngbya*, and *Limnothrix*. Green algae of the genera *Scenedesmus*, *Staurostrum*, and *Pediastrum*, and diatoms of the genera *Staurosirella* and *Nitzschia* were common also. Phytoplankton densities in all three lakes were negatively correlated with stage and discharge due to reduced residence time and washout at high discharge and the strong positive correlation between water color and discharge. Lake Jesup likely served

as a frequent source for phytoplankton species in Lake Monroe, as evidenced by dominance of *Cylindrospermopsis* and *Planktolyngbya* in both lakes.

### 3.1.4 ZOOPLANKTON COMMUNITIES

SJRWMD sampling of zooplankton in the lower St. Johns River began in 1994 and continues through the present. Zooplankton samples from the lower St. Johns River have been filtered through a 41  $\mu\text{m}$  mesh. Zooplankton abundances were generally highest in the LSJR Freshwater Reach and declined dramatically downstream in the oligohaline zone, although abundances were also high in offshore oligohaline Doctors Lake (Phlips and Cichra 2001, 2002, unpublished data). Zooplankton communities in the LSJR Freshwater Reach generally were numerically dominated by rotifers, with secondary dominance by nauplii (likely copepod nauplii, but may include some barnacle nauplii). Adult copepods and cladocera were generally less abundant, but these larger bodied groups likely were more important contributors to zooplankton biomass (not measured). Further downstream in the oligohaline zone, abundances of rotifers and cladocera decreased dramatically, although rotifers remained a prominent part of the community. Nauplii and adult copepods showed less decrease in the oligohaline zone, thus becoming more prominent components of the community. Larval stages of marine invertebrates also became more prominent in the oligohaline zooplankton community.

Highest zooplankton abundances in the lower St. Johns River were generally observed during the warm season, although there was substantial variability among years and sites (Phlips and Cichra 2002). Zooplankton abundances also tend to increase with increases in water age or decreases in flow (R. S. Fulton unpublished data).

Zooplankton communities in Lake Jesup (in the middle St. Johns River) and Lake Washington (in the upper St. Johns River) were sampled during 2001, using a 75  $\mu\text{m}$  mesh (Osborne 2002). Although this mesh likely is too large to quantitatively retain rotifers, the samples were generally numerically dominated by rotifers, with secondary dominance by copepods, and relatively few cladocera. Total zooplankton abundances were generally highest in the warmer months in Lake Jesup, but there was not as clear a seasonal pattern in Lake Washington.

During 2003 to 2005, Burnes et al. (2008) sampled zooplankton in the middle St. Johns River using an 80  $\mu\text{m}$  mesh filter. Rotifers likely were not sampled quantitatively with this coarse mesh. Unlike the previously cited reports, biomasses of crustacean zooplankton were determined by Burnes et al. (2008). Copepods (primarily nauplii) were the most abundant. Cladocera were less abundant, but some of the sites (Lakes Jesup and Monroe) had higher average biomass of cladocera than of copepods. Seasonally, copepods were most abundant during spring and summer, while cladocera were most abundant in the fall.

Overall, zooplankton communities in the St. Johns River system tend to be dominated numerically by rotifers and small crustaceans, which is typical for riverine systems (Appendix 8.A) and for Florida freshwater zooplankton communities (Burnes et al. 2008).

## 3.2 DATA AND SAMPLE COLLECTION AND ANALYSES

### 3.2.1 DATA SETS AND SAMPLING SITES

The SJRWMD has conducted monitoring programs in parts of the St. Johns River since at least the early 1980s, and comprehensive monitoring in all reaches since the 1990s (Table 3–2). Typical long-term monitoring programs included water chemistry, plankton (identification, enumeration, and biomass or biovolume estimation), and tributary and discharge monitoring for nutrients and other constituents.

Table 3–2. Primary plankton and water chemistry data sets available for the St. Johns River from SJRWMD monitoring programs. We used a subset of these data in the plankton analyses.

Area	Data	Number of Sites	Samples yr <sup>-1</sup>	Period
Lower St. Johns River	Water chemistry	~18	12 to 24	1984 to present; Expanded 1993
	Tributaries and point sources	72 tributaries 36 point sources	6 to 12	1993 to present
	Plankton identification, counts, biomass	14	12	1994 to present
	Probes (dissolved oxygen [DO], temperature, specific conductivity)	5	Continuous	1998 to present; Expanded 2009
Middle St. Johns River	Water chemistry	30	4 – 12	1991 to present; Expanded 1995, 2001, 2003
	Tributaries and point sources	18	12	1985 to present; Expanded 1995
	Plankton identification, counts, biomass	9	12	1995 to present; Expanded 2001
Upper St. Johns River	Water chemistry	15	12	1992 to present
	Tributaries	11	12	1992 to present
	Plankton identification, counts, biomass	15	12	1992 to present

The WSIS ecological working groups divided the St. Johns River into nine segments for analysis (see Chapter 2. Comprehensive Integrated Assessment) (Table 3–3). The Plankton Working Group focused on five of these segments: segments 2, 3, 4, 6, and 8. We analyzed data from multiple locations in three segments and a single location in the remaining two segments. Some of these locations were represented by single sampling sites. For other locations, we averaged

data from multiple sites in close proximity or from multiple sites in one lake. In these cases, we first averaged duplicate values for the same site, and then we averaged values for multiple sites sampled on the same date. The sampling sites used in our analyses are shown in Figure 3–8.

Table 3–3. River segments, locations, and sites for the primary data sets used in analysis of potential effects of water withdrawals on plankton. Data from multiple sites per location were averaged. Site names in parentheses were sampled by Volusia County, Florida, and were used for chl-a data only. See Figure 3–8 for a map of the data collection sites.

River Segment*	Location	Data Collection Sites Used in Plankton Analyses
1	—	—
2	Mandarin Point	MP72
	Doctors Lake	DTL
3	Racy Point	SRP, FP44
	Palatka	SJP, PA32
4	Lake George	LEO, LG12, 20030373, MSJLGN
5	—	—
6	Lake Monroe	LMAC, LM-OW-NE, LM-OW-S, (SJ09, SJ10, SJ11, SJ12)
	Lake Jesup	OW-2, OW-3, OW-4, OW-4, OW-5, OW-6, OW-7
	Lake Harney	CLH, LH-OW-NE, LH-OW-SW, (SJ03, SJ04)
7	—	—
8	Lake Poinsett	LPO, LPC
9	—	—

\* Segments 1, 5, 7, and 9 were not assessed by the Plankton Working Group.

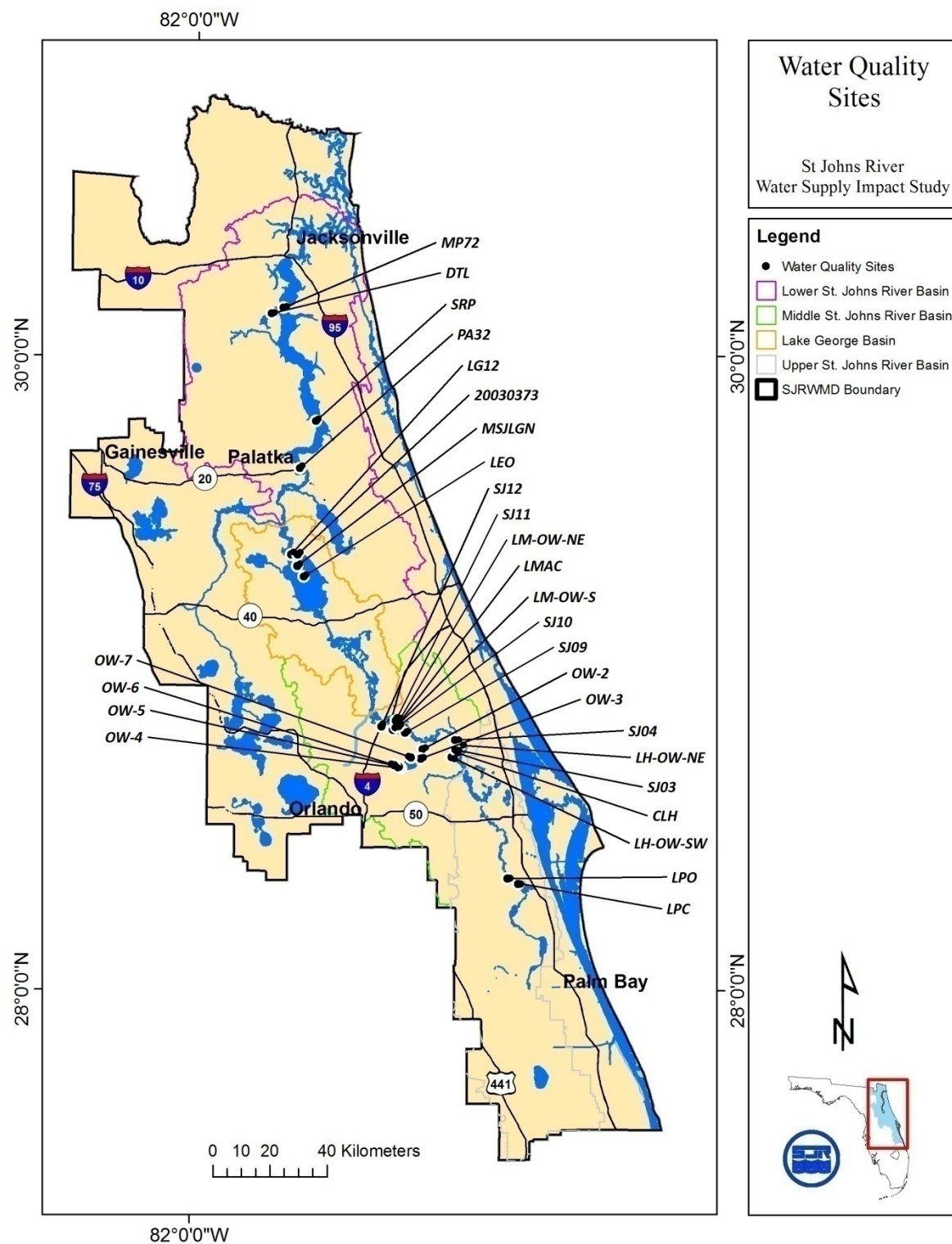


Figure 3–8. Data collection sites used in the plankton analyses.



### 3.2.2 WATER QUALITY SAMPLING AND ANALYSES

#### **Field Sampling**

In the lower St. Johns River, the majority of water samples for chemical and plankton analyses were collected as a 2.5-m vertical composite with a pre-cleaned, 3-m long, 3.8-cm inside diameter PVC pipe. At least five pipe collections per site were pooled in a sampling churn. A small number of samples were collected as grab samples at a depth of 0.5 m.

In the middle St. Johns River, water samples (i.e., chemistry and phytoplankton) collected prior to October 2002 were collected from a single depth of 0.5 m using a 5 L horizontal Van Dorn bottle. From October 2002 through December 2005, water chemistry and phytoplankton samples were collected using a horizontal 5 L Van Dorn bottle at three depths (0.25, 0.5, and 1 m), and then composited into a single sample. Zooplankton samples were collected using a 30-cm (12-in) conical zooplankton net (80  $\mu$ m mesh) towed horizontally for 3.0 m (10 ft) at a depth of 0.5 m. Since January 2006, samples for water chemistry and zooplankton have been collected using an integrating tube sampler from the surface to a maximum depth of 3 m (or 30 cm above the sediment). Zooplankton were collected by pouring a 3-L sample through a 42- $\mu$ m mesh. Phytoplankton samples were collected by integrated tube to a maximum depth of 1 m (or 30 cm above the sediment).

Samples for chlorophyll-a were collected in opaque bottles and placed on ice, and samples for total P and N were preserved with 1 N sulfuric acid. Phytoplankton and zooplankton samples were preserved with Lugol's solution (Clesceri et al. 1998). Secchi depth was measured with a 25-cm diameter disc with black and white quadrants. Additional analyses included dissolved oxygen (DO), pH, and water temperature by sondes (Hydrolab Surveyor or YSI 6000) with calibrated probes.

#### **Analyses**

Dissolved nutrients were analyzed on water passing through a 0.45- $\mu$ m Gelman Supor membrane filter. The following methods were used in analysis, with numbers in parentheses referenced from Clesceri et al. (1998):  $\text{NO}_{2+3}$  by copper–cadmium reduction and diazotization colorimetry (4500- $\text{NO}_3$ -F);  $\text{NH}_4$  by the alkaline phenol–colorimetric method (4500- $\text{NH}_3$ -G); total Kjeldahl nitrogen (TKN) by high temperature sulfuric acid digestion on whole water (unfiltered) sample followed by  $\text{NH}_4$  analysis (4500- $\text{N}_{\text{org}}$ -C);  $\text{PO}_4$  (soluble reactive phosphorus, [SRP]) by antimony–phospho–molybdate complex ascorbic acid–colorimetric method (4500-P-F); and total phosphorus (TP) by high-temperature sulfuric acid digestion on a whole water sample followed by  $\text{PO}_4$  analysis. Nutrient analyses were performed on an O-I Analytical Enviro-Flow FS 3000 auto-analyzer. Total suspended solids (TSS) were measured gravimetrically on a Whatman GF/C filter (2540-D). Chlorophyll-a (chl-a) was determined spectrophotometrically after filtration (Whatman GF/C) and extraction into 90% acetone (10200-H, not corrected for phaeopigments). TN was calculated from total Kjeldahl-N plus  $\text{NO}_{2+3}$ -N, and dissolved inorganic nitrogen (DIN) was calculated as  $\text{NH}_4$ -N plus  $\text{NO}_{2+3}$ -N.

Most surface water chemistry was performed by the SJRWMD laboratory, with a smaller number of samples analyzed by the FDEP Bureau of Laboratories and the Volusia County Health

Department's Environmental Health Laboratory. SJRWMD water chemistry data below detection limits were not censored.

In the lower St. Johns River, phytoplankton and zooplankton were settled and enumerated using an inverted microscope (Utermöhl 1958). For phytoplankton analysis, an aliquot of preserved sample was placed directly into settling chambers. Phytoplankton were classified and enumerated in random grids until a count of 100 counting units was attained or five grids were examined at 400X. Uncommon forms were enumerated in 30 grids. Large forms were enumerated at 100X utilizing the entire bottom of the chamber. Phytoplankton biovolumes were determined using the closest geometric shape(s) and the appropriate volumetric formula(e). Zooplankters were concentrated on a 41- $\mu$ m mesh nylon filter. A minimum of three subsamples were analyzed for each sample. Zooplankton were enumerated in settling chambers at 100X magnification, and counts were continued to a minimum of 100 for the most common taxa.

In the middle St. Johns River, different analytical methods were used for phytoplankton and zooplankton depending on sampling date, as follows:

- Middle St. Johns River Phytoplankton sampled from Nov 2002 to July 2007—Samples were quantified in a Palmer Maloney nannoplankton counting cell (PNCC) using a compound microscope with interference contrast microscopy at 400X. A sufficient number of random fields were analyzed to count 300 natural units. In samples with sparse phytoplankton, transects across the PNCC were enumerated. For rare and large taxa, one-half or an entire PNCC was examined at 100X magnification. Phytoplankton were identified to the lowest practical taxonomic level which usually was species.
- Middle St. Johns River Phytoplankton sampled from August 2007 to present—Phytoplankton analysis was performed by the Utermöhl (1958) method for inverted microscope examination following at least 4 hours settling per cm of sample height in the chamber. At least 400 algal units were enumerated at 1,000X magnification for each sample when possible by counting all organisms within random fields or along transects. The entire chamber was scanned at 100X for large or rare taxa. Cell biovolumes were determined by measuring the mean volume for up to 10 specimens of a taxon at a station in that month. All organisms were identified to species when possible using standard taxonomic references. Diatom identification was facilitated by acid cleaning or by making burn mounts.
- Middle St. Johns River Zooplankton sampled from Oct 2002 to Feb 2003—Three 1-ml aliquots of sample concentrate were examined at 100x using an inverted microscope equipped with phase contrast. If the total tally was less than 200, additional aliquots up to a maximum of 20 ml were examined. Biomass was estimated with established length-weight relationships.
- Middle St. Johns River Zooplankton sampled from March 2003 to present—Settling procedures for zooplankton were adapted from procedures described in Clesceri et al. (1998). Zooplankton were enumerated in settling chambers at 100X magnification. A minimum of three subsamples were viewed to reach two count targets: 1) a minimum



count of 100 individuals for each taxon, and/or 2) a total zooplankton count of at least 250 individuals. Biovolume calculations used 10 measurements of each taxon.

### **Continuous Monitoring of Dissolved Oxygen**

Continuous DO data were available from a monitoring network funded by the SJRWMD and operated and maintained by U.S. Geological Survey (USGS). Sites for the freshwater lower St. Johns River were Lake George (river km 183) from 2004 to the present, Buffalo Bluff (river km 145) from 2003 to present (data gap May 2004 to October 2005), and Dancy Point (river km 110) from 1998 to present (data gaps in 2003 and 2007). DO was measured by YSI Clark amperometric probes at hourly intervals.

### **3.3 HYDRODYNAMIC MODEL OUTPUT DATA**

We used water age (a measure of residence time) as the primary hydrologic variable to relate to algal community dynamics. Water age was calculated in the Environmental Fluids Dynamic Code (EFDC) hydrodynamic model and was the average time that water resided in the model domain before reaching a specific site (model grid cell) (see Chapter 6. River Hydrodynamics Results). We also coupled a mechanistic water quality model (CE-QUAL-ICM model) to the EFDC model to predict algal chl-a at select sites in the St. Johns River (Appendix 8.D). Water age output data sets consisted of temporally (daily) and vertically (six layers) averaged values for specific sites and specific withdrawal scenarios. Hydrodynamic simulations were run for the 11-yr period 1995 to 2005. Because time was required for variables to reach realistic values (spin-up), most uses of the simulated hydrodynamic data from the EFDC model omitted 1995. We examined water age values and found sufficient stability at three months to use water age data starting in April 1995. For details on hydrodynamic modeling, see Chapter 6. River Hydrodynamics Results.

### **3.4 ALGAL BLOOM METRICS AND POTENTIAL EFFECTS**

Based on previous analyses, literature, and professional experience, we compiled a preliminary set of potential ecological effects of water withdrawal on plankton, nutrients, and TMDLs. We refined this initial set (see Section 4), and our final approach used five potential adverse effects assessed with four algal bloom metrics (Table 3–4). For each adverse effect, we developed environmental thresholds (see Section 4).

For the marine algal blooms metric, the effect was the risk for toxic dinoflagellates to develop high population levels in oligohaline portions of the estuary, and the measured variable was maximum annual dinoflagellate biovolume (see Table 3–4). We examined phytoplankton data from locations Piney Point, Mandarin Point, and Doctors Lake from 1995 to 2005 and used best professional judgment to select a biovolume threshold for dinoflagellates beyond which bloom conditions exist. Although toxic dinoflagellates currently do not develop high densities in the St. Johns River, we used the total biomass for the class as a surrogate for potentially toxic blooms.

Table 3–4. Algal bloom metrics and the associated effects and measured variables used in the empirical regression models. See Table 4–6 for the thresholds set for each effect.

Algal Bloom Metric	Effect(s)	Measured Variable
Marine algal blooms	Potential toxic species	Maximum annual dinoflagellate biovolume
Change ( $\Delta$ ) in N load	Additional N loading	Annual mass N added via N <sub>2</sub> -fixation
Magnitude of freshwater algal blooms (two effects)	1) Altered phytoplankton community; cyanobacterial toxins  2) Depletion of dissolved oxygen (DO); effects on fish reproduction, growth, and mortality	Maximum annual bloom chl-a
Duration of freshwater algal blooms	Altered zooplankton community; reduction in fish production	Duration of longest annual bloom

For the change ( $\Delta$ ) in N load metric, the effect was additional N loading, and the measured variable was the annual N mass added through N<sub>2</sub>-fixation by cyanobacterial blooms. N<sub>2</sub>-fixation is a source of N to the St. Johns River where N loading needs to be decreased to achieve water quality standards (USEPA 2008). We used best professional judgment to choose a threshold above which additional N<sub>2</sub>-fixation would present an important problem in meeting the TMDL.

For the magnitude of freshwater algal blooms metric, we evaluated two effects (altered phytoplankton community and depletion of dissolved oxygen) and adopted different thresholds for each effect. The first effect was the shift in phytoplankton community structure toward increased dominance by cyanobacteria with increasing bloom density (biomass measured as chl-a). We analyzed this relationship with data on phytoplankton species composition, biovolume, and chl-a from Lake George, Palatka, and Racy Point from 1993 to 2007. We set this threshold for bloom magnitude based on both the shift in phytoplankton community structure and on the potential risk to humans presented by cyanotoxins (World Health Organization 2003).

The second ecological effect for the magnitude of freshwater algal blooms metric was the depression in water column dissolved oxygen (DO) following collapse of dense algal blooms. We matched continuous DO data collected during cyanobacterial blooms with chl-a data from adjacent or nearby chemistry monitoring sites at Lake George, Buffalo Bluff, and Racy Point. Values for maximum bloom chl-a were paired with the subsequent minima in daily mean DO concentration and percent saturation. We used ordinary least-squares regression to relate DO after collapse of blooms with maximal bloom chl-a. We set the second threshold for bloom magnitude based on potential effects of low DO on fish populations.

For the duration of freshwater algal blooms metric, the effects were altered zooplankton community structure and reduction in fish production, and the measured variable was the duration of the longest annual bloom. Persistent algal blooms generally were dominated by cyanobacteria. We analyzed data on zooplankton abundance and species composition, chl-a, and temperature from Lake George, Palatka, and Racy Point for 1997 to 2007. We measured duration of algal blooms by linear interpolation of chl-a values between sampling occasions and tabulation of number of continuous days with chl-a  $\geq 40 \mu\text{g L}^{-1}$ . We used quantile regression to examine relationships between algal bloom duration and abundances of four zooplankton groups:

rotifers, cladocera, copepod nauplii, and adult copepod stages. We used literature and best professional judgment to adopt a threshold for bloom duration based on shifts in zooplankton species composition and the potential for reduced fish production due to reduced food availability.

### 3.5 EMPIRICAL REGRESSION MODELS

We constructed empirical regression models to relate algal bloom metrics to water age variables. Regression models were based on water age output from a calibration run of the EFDC hydrodynamic model (see Chapter 5. River Hydrodynamics Calibration). The calibration run used measured discharges as input wherever possible instead of modeled discharges and should have been a close description of actual conditions. We used these regression models to assess scenarios for water withdrawals using water age output from scenario runs where all hydrologic input variables were modeled. We used water age output from the EFDC model for the baseline scenario and each withdrawal scenario with regression models and compared the predicted algal bloom metrics. This use of modeled water age for baseline versus withdrawal scenarios isolates the difference due to withdrawal and minimizes the effects of bias in the EFDC hydrodynamic model output.

The time period for data used in regression models was determined by the 11-year simulation period for the EFDC hydrodynamic model: January 1995 through December 2005. Because model initiation (spin-up) caused low water age values during the first three months, we did not use data from January to March 1995.

Dependent variables for regression models were the quantities measured for each of the four algal bloom metrics (Table 3–4). These variables were tabulated annually and represented either the maximum (biovolume, chl-a, bloom duration) or the total (mass N<sub>2</sub>-fixation) for that year.

The annual mass N added via N<sub>2</sub>-fixation was estimated from N mass balance calculations for Lake George where inflows, outflows, and tributary contributions of N were well characterized. We subtracted the annual mass of N in all inflows from the mass in the outflow to estimate N<sub>2</sub>-fixation in Lake George. N<sub>2</sub>-fixing cyanobacterial blooms were not limited to Lake George, but we could not apply the same mass-balance approach downstream, where inputs were measured less accurately. Instead, we multiplied N<sub>2</sub>-fixation estimated for Lake George by an areal factor 1.37 ([area of Lake George + area downstream] ÷ [area of Lake George]) to account for additional N<sub>2</sub>-fixation in downstream reaches where N<sub>2</sub>-fixing cyanobacterial blooms typically occurred.

Independent variables for regression models were minimum, mean, and maximum water age and the inverse of each water age calculated for several time periods: five quarters (A to E, starting with the last quarter of the previous calendar year) and two growth-season periods (April to August and April to October) (Figure 3–9). For example, we regressed algal bloom metrics for 2002 on water age variables for seven periods beginning in October 2001. We included this wide range of water age variables as potential prediction variables in regression models because relationships between algal bloom patterns and hydrology can be both positive and negative and show complex seasonality (see section 4).

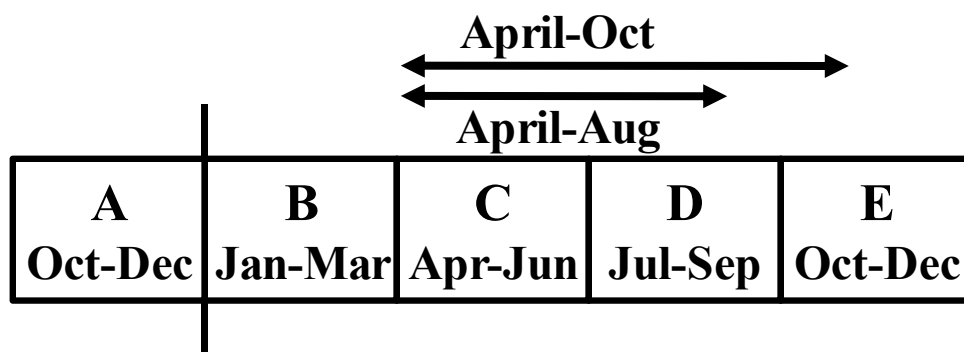


Figure 3–9. Averaging periods for water age variables used as independent variables in empirical regression models to predict algal bloom metrics. April to October is the entire bloom season. Periods A and B precede the development of blooms.

For some river segments, we used data from a single location in regression models. In other segments, we pooled data sets from up to three locations (see [Table 3–3](#)). For regression models where no water age variables from period A or B (see Figure 3–9) were selected, the number of observations in the data set was 11 (one for each year from 1995 to 2005) for each location. For models that included a variable from period A or B, the number of observations per location was 10 (1996 to 2005).

No variables were transformed except for maximum annual dinoflagellate biovolume, where logarithmic transformation appeared to be necessary. Best fit regression models were constructed using either stepwise or  $R^2$  selection of independent variables for linear regression and by highest likelihood score selection for logistic regression. We used SAS<sup>®</sup> 9.2 to calculate regression models.

### 3.6 CE-QUAL-ICM MECHANISTIC MODEL

CE-QUAL-ICM is a mechanistic water quality model designed for the study of eutrophication in aquatic ecosystems (Cercio and Cole 1995). The model was applied in WSIS to provide a second assessment of water withdrawal effects on bloom maximum chl-a in segment 3, independent of the empirical regression model.

CE-QUAL-ICM originally was applied to the lower St. Johns River to evaluate external nutrient load scenarios to determine nutrient TMDL targets (Sucsy and Hendrickson 2004, Tillman et al 2004, Limno-Tech 2005). Subsequent expansions to the model domain and time period, enhancements to model algorithms, and scenario simulations for WSIS were done by SJRWMD staff members with assistance from Limno-Tech and the U.S. Army Corps of Engineers.

CE-QUAL-ICM is a finite-difference, time-varying, eutrophication model built on mechanistic relationships among underwater irradiance, primary production, nutrient assimilation, grazing, decomposition, and settling. For WSIS, this model used hydrodynamic output from the EFDC hydrodynamic model. CE-QUAL-ICM included three functional phytoplankton groups set up to represent cyanobacteria, freshwater diatoms, and marine diatoms. The model separated labile and refractory organic carbon (C), N, and P pools to account for the high natural background pool of largely refractory colored dissolved organic matter in the St. Johns River.

The model domain for the CE-QUAL-ICM model application to WSIS was the tidal freshwater and marine portions of the St. Johns River estuary, from the inlet to Lake George to the Atlantic inner shelf at the St. Johns River mouth, and included the major offline lakes—Crescent and Doctors lakes. The CE-QUAL-ICM model was set up with input boundary water chemistry concentrations to match the 1996 to 2005 period of EFDC-simulated hydrodynamic output (see Chapter 6. River Hydrodynamics Results; 1995 EFDC output was not used because of spin-up artifacts). Observed time series for nutrient loading were used for the two major inflows (St. Johns River inflow to Lake George and Ocklawaha River inflow). Nutrient loading from other watersheds was constant based on 1995 land use, and mean values (1995 to 1999) provided constant loading from point sources. (See Appendix 8.D for details about the structure, set-up, and operation of the CE-QUAL-ICM model for WSIS.)

### 3.7 ASSESSMENT OF EFFECTS OF WATER WITHDRAWALS

We used the empirical regression models appropriate to each river segment to predict algal bloom metrics (magnitude of freshwater algal blooms, duration of freshwater algal blooms, etc.; see Table 3–4) from water age output from the EFDC hydrodynamic model for each simulation year from 1995 to 2005. In each case, we predicted algal bloom metrics for both a baseline scenario and the withdrawal scenario. Where feasible, we expressed the change from baseline to withdrawal in two ways as follows:

- (1) The percent change in the median response (withdrawal minus baseline) of the algal bloom metric

$$\frac{(\text{median of withdrawal values} - \text{median of baseline values})}{\text{median of baseline values}} \times 100$$

- (2) The change in annual frequency (withdrawal minus baseline) that thresholds were exceeded (e.g., two in 11 years (0.18) minus one in 11 years (0.09) = +0.09)

In those cases where both the percent change in median response and the change in annual frequency of exceedence were calculated, we used the percent change in median values to assess overall effect (see section 4).

Because water age was a key hydrologic variable in our regression models, we ranked the various water withdrawal scenarios by their effects on water age compared with the baseline scenario. Our assessment of effects of withdrawals on plankton started with the scenario with greatest effect on water age (top-ranked scenario) and continued with scenarios in ranked order until we found negligible effects on algal bloom metrics.

We used the CE-QUAL-ICM mechanistic water quality model in parallel with empirical regression models to assess effects of water withdrawals on the magnitude of algal blooms in segment 3. Output data from the EFDC hydrodynamic model for simulation years 1996 to 2005 was input data for the water quality model along with measured water quality boundary conditions. We tabulated annual maximum chl-a for each simulation year, and compared the baseline scenario with several withdrawal scenarios prioritized by effects on water age. Simulation year 1995 was omitted from this work to allow sufficient model iterations for complete initiation (spin-up) of all model variables.

Potential effects of water withdrawals on plankton in the most upstream river segment that we assessed (segment 8, Lake Poinsett) were analyzed using different methodology and are described in a separate report (Appendix 8.B).

## **4 RESULTS AND DISCUSSION**

### **4.1 HYDROLOGY AND ALGAL BLOOMS IN THE ST. JOHNS RIVER**

#### **4.1.1 WATER AGE**

We chose water age (a measure of residence time) as the primary hydrologic variable to relate to algal community dynamics. Residence time plays a central role in affecting nutrient supply, nutrient uptake, and the balance between algal production and loss processes and, therefore, algal bloom development (see Appendix 8.A).

Water age is the average time that water resides in the model domain before reaching a specific model grid cell. Water age has several conceptual and practical benefits in evaluating hydrologic effects on environmental factors compared to the standard calculation of residence time as  $\text{volume} \div \text{flow rate}$ :

- Water age is calculated for each cell in the model grid without the necessity to define a cross section for volume and flow rate.
- Water age incorporates the antecedent variable flow conditions. Antecedent conditions can be considered in calculation of residence time through use of long averaging periods or lag times, but these approaches have less resolution than continuous simulation of water age.
- Water age accounts for the dynamic circulation of water within water bodies, whereas calculation of residence time from volumes and discharges assumes instantaneous mixing (Shen and Wang 2007).
- Water age does not take negative values during periods of reverse flow that occur during low flow conditions in the lower and middle St. Johns River.

In addition to water age, we used inverse water age as a potential explanatory variable in regressions with algal bloom metrics. Use of both variables expands scales at both ends of the flow continuum (Figure 4–1). Whereas water age variables are sensitive to low flow conditions in regression equations, inverse water age variables are sensitive to high flow conditions. Inverse water age is a proxy for discharge; it is linearly related to discharge (Figure 4–2) but does not become negative during reverse flows.

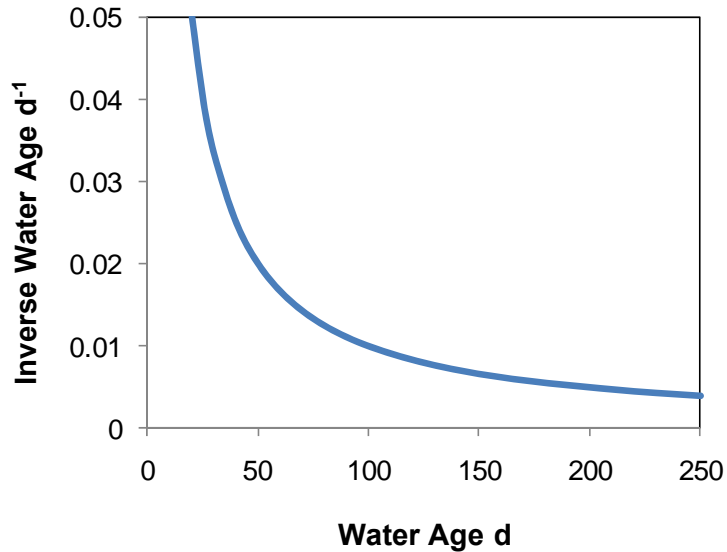


Figure 4–1. Relationship between water age and inverse water age for values in the range commonly found for sites in the St. Johns River.

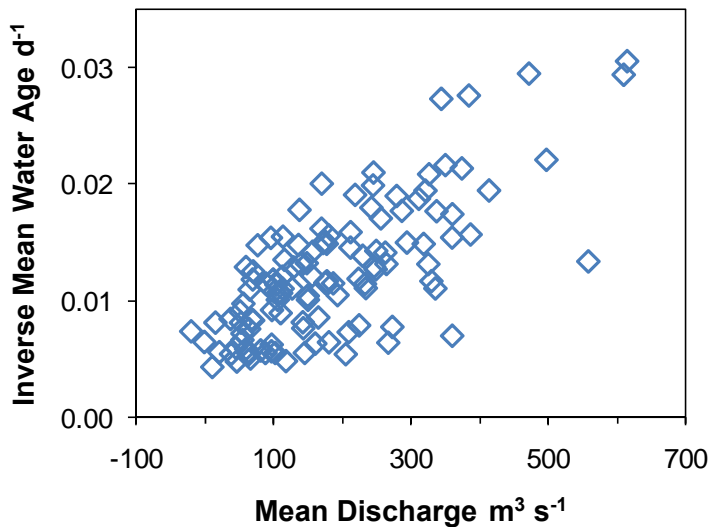


Figure 4–2. Linear relationship between the inverse of monthly mean water age and monthly mean discharge. Inverse water age is positively related to discharge but without the negative values that occur for discharge during periods with reverse flows. Water age data are from Racy Point. Discharge data are from the nearby downstream site Shands Bridge (SJSR16). Both data sets were taken from calibration runs of the EFDC hydrodynamic model (1996 to 2005).

#### 4.1.2 PHYTOPLANKTON, NUTRIENTS, AND WATER AGE

During the April to October algal bloom period, measured chl-a showed a saturation relationship with monthly mean water age for locations in the freshwater lower St. Johns River (Figure 4–3).

Chl-a increased until water age reached approximately 80 d, but further increases in monthly mean water age corresponded to little or no change in chl-a. In contrast, TP and water color concentrations decreased with increasing water age, with almost a four-fold decline in TP and a 10-fold decline in water color between the extremes in water age. Similarly, soluble reactive phosphorus (SRP) concentrations were highest at short water ages and declined as water age (and chl-a) increased (Figure 4–3). Concentrations of TN showed a different pattern with little change in average values across the range in water age. In contrast, dissolved inorganic nitrogen (DIN) concentrations declined with increasing water age in tandem with SRP. Sustained high TN values at long water age, especially in Lake George, likely stemmed from N<sub>2</sub>-fixation (Figure 4–3).



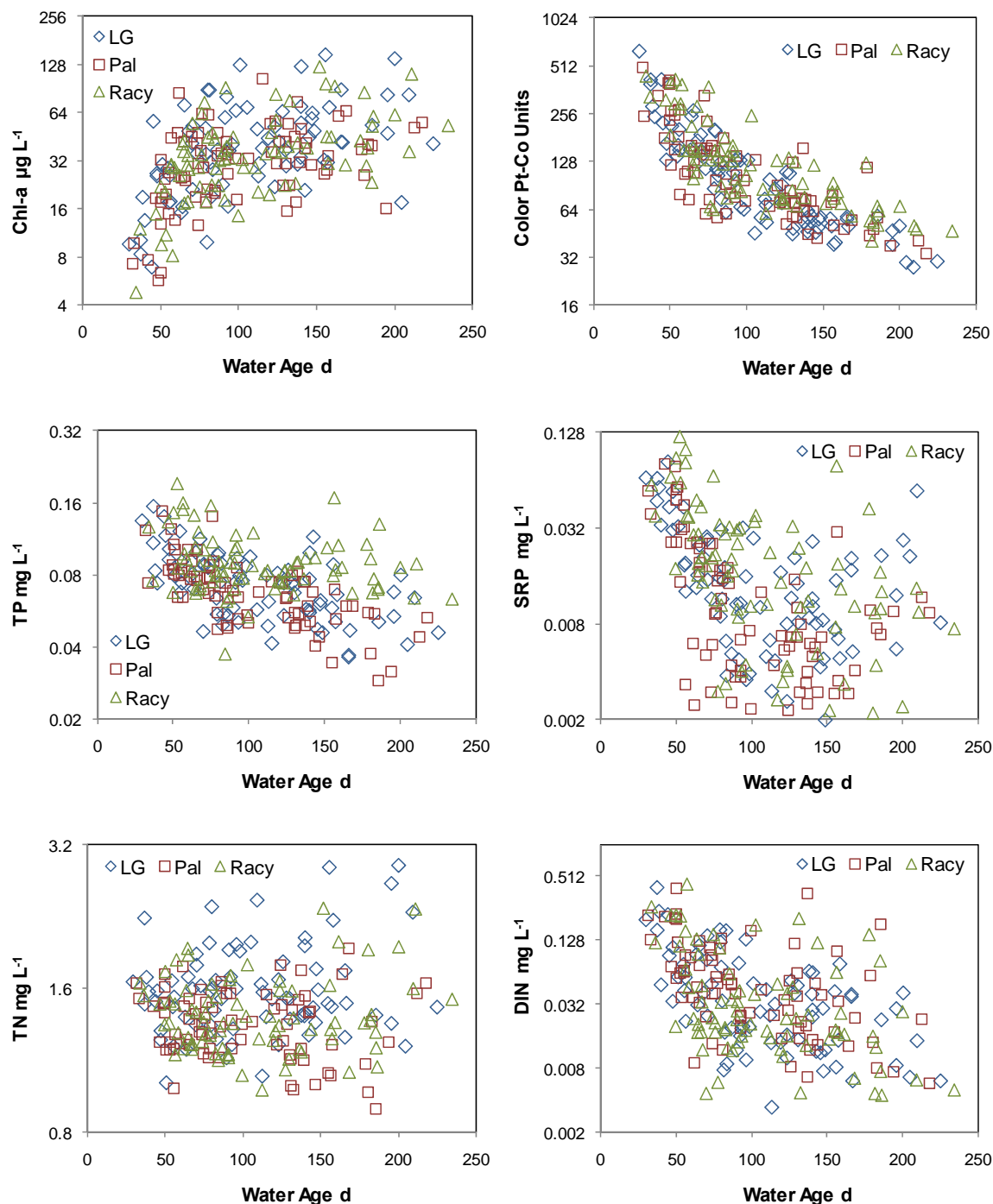


Figure 4–3. Relationships among monthly mean values for chlorophyll-a (chl-a), water color, total phosphorus (TP), soluble reactive phosphorus (SRP), total nitrogen (TN), and dissolved inorganic nitrogen (DIN) and monthly mean water age for three locations in the freshwater lower St. Johns River: Lake George (LG), Palatka (Pal), Racy Point (Racy). Values are for April to October, 1995 to 2005. Analytical uncertainty at low concentrations contributed to the increased variability in SRP and DIN at long water ages. Note logarithmic Y-axes.

Relationships between mean monthly chl-a, TP, TN, and color and water age for two main-stem locations in the middle St. Johns River (Figure 4–4) were very similar to patterns in the lower St. Johns River (Figure 4–3). Chl-a showed a similar saturation response to water age, and reached maximum values at about 45 d. The different water age (x-axis) scales for middle and lower St. Johns River locations reflected the closer proximity of middle St. Johns River lakes to the boundary of the model domain (water age = 0 d) among other factors. Color showed about 10-fold variation with water age. One difference from lower St. Johns River locations was that TP changed little with water age in Lake Monroe (Figure 4–4). This pattern may have stemmed from an increased proportion of high nutrient inflow to Lake Monroe from eutrophic Lake Jesup under low discharge conditions in the river.

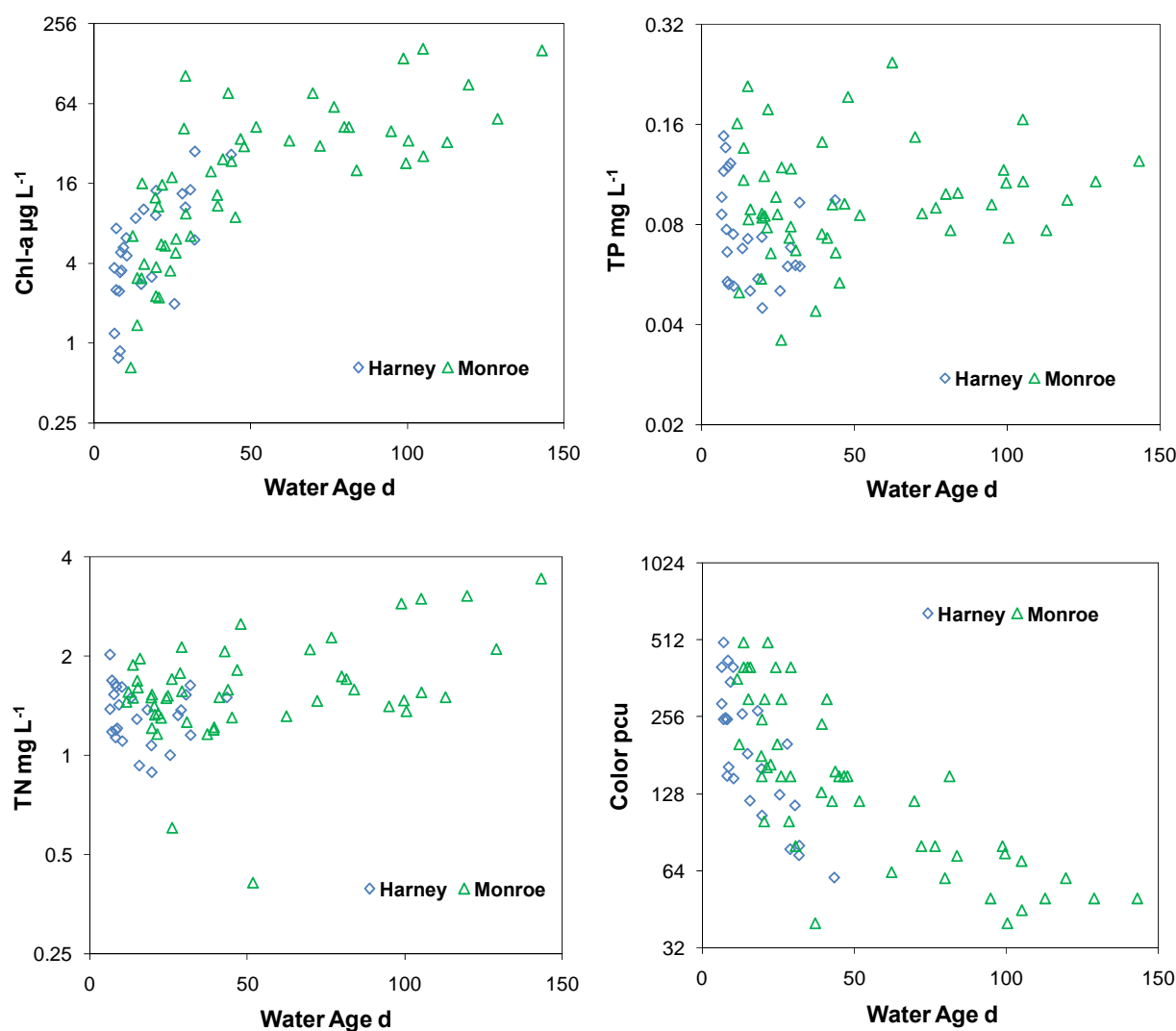


Figure 4–4. Relationships between monthly mean values for chlorophyll-a (chl-a), total phosphorus (TP), total nitrogen (TN), and water color and monthly mean water age for two locations in the middle St. Johns River: Lakes Monroe and Harney. Values are for April to October, (1995 to 2005). Note logarithmic Y-axes.

Algal biomass (chl-a) was low under short water age–high flow conditions and was high under long water age–low flow conditions, a pattern opposite that shown by algal nutrients (N, P) (see Figure 4–3). This apparent disconnect between nutrients and chl-a could be understood better by examination of short and long water age conditions separately. Under short water age–high flow conditions (water age  $\leq 80$  d), chl-a either was not significantly related to TP (Racy Point) or showed borderline significant negative relationships (Lake George and Palatka) (Figure 4–5). When only months with long water age–low flow (water age  $> 80$  d) were considered, there were significant, positive relationships between chl-a and TP at all locations (Figure 4–5).

The apparent explanation for this dichotomy was that, under long retention time and low color conditions, phytoplankton in the St. Johns River more fully utilized nutrients; and algal blooms occurred. Furthermore, long retention times favored recycling of nutrients from sediments and  $N_2$ -fixation, if N was limiting. In contrast, during short retention time–high flow periods, loading of nutrients in runoff was maximal (see Figure 4–3) but other constraints, such as light limitation due to high color (see Figure 4–3 and Figure 4–4) and high flushing rate, suppressed the attainment of maximal algal standing stock even in the presence of sufficient nutrients (Figure 4–5).

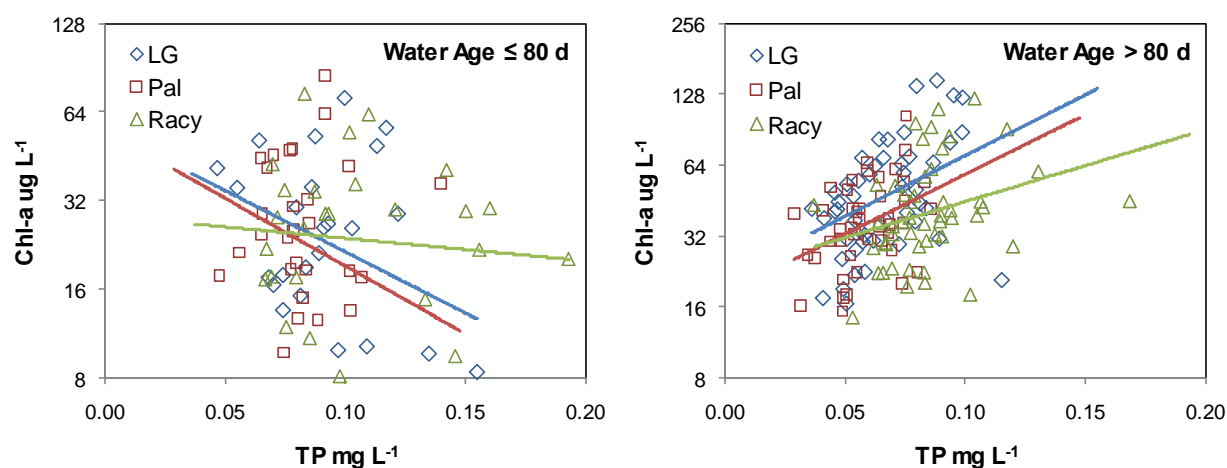


Figure 4–5. Relationships between monthly mean values for chl-a and TP for (left) short retention times (monthly mean water age  $\leq 80$  d), and (right) long retention times (monthly mean water age  $> 80$  d). Data for three sites in the freshwater lower St. Johns River: Lake George (LG), Palatka (Pal), Racy Point (Racy). Values are for April to October, 1995 to 2005. Regressions were calculated as  $\ln(y) = mx + b$ . Regressions for water age  $\leq 80$  d had negative slopes and were borderline significant ( $p = 0.05$ ) or not significant ( $p > 0.05$ ). All regressions for water age  $> 80$  d were significant with positive slopes ( $p < 0.05$ ).

Relationships between algal biomass and water age could be complex because both coincident and antecedent flow conditions could affect algal growth. Algal blooms occurred annually during summer in the lower St. Johns River, although the magnitude and the exact timing varied (Figure 4–6). In contrast with this annual pattern in chl-a, time series for water age showed both annual increases and high values sustained over several years, for example during droughts in 1996 to 1997 and 2000 to 2001 (Figure 4–6). We expected these patterns due to combined effects of annual and longer term oscillations in rainfall. The magnitude (and likely the duration) of

summer algal blooms was related to summer water age (see Figure 4–3, Figure 4–4), and discharge conditions the previous winter could be important as well. High winter flows meant higher nutrient levels in the lower St. Johns River

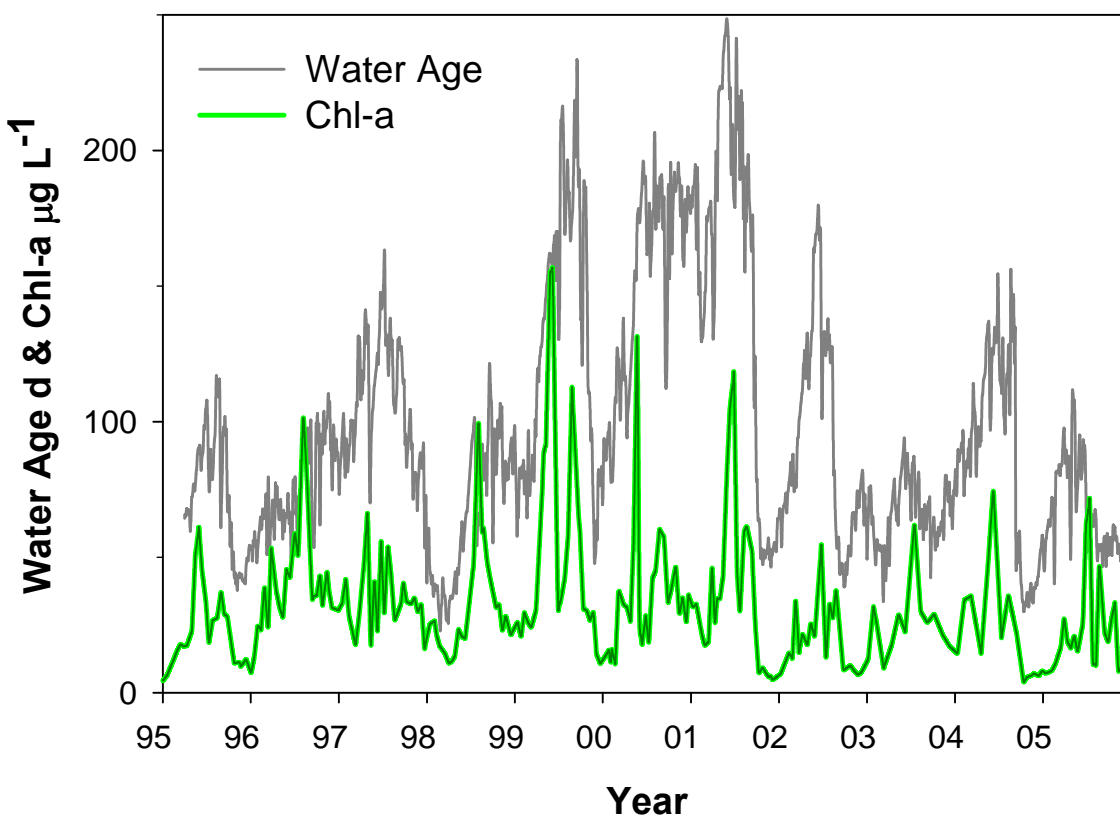


Figure 4–6. Time series for observed chlorophyll-a (chl-a) and water age for Racy Point 1995 to 2005.

Our working definition of an algal bloom was a chl-a concentration  $\geq 40 \mu\text{g L}^{-1}$ . We chose this definition for several reasons. Seasonal patterns for chl-a at several sites in the lower St. Johns River revealed that values varied up to approximately  $40 \mu\text{g L}^{-1}$  in all seasons but exceeded that value almost exclusively during algal blooms occurring from April to October (Figure 4–7). The value  $40 \mu\text{g L}^{-1}$  separated the typical range from summertime excursions. Second,  $40 \mu\text{g L}^{-1}$  is the threshold chl-a concentration for eutrophic (poor) conditions in low water color lakes under the Florida trophic state index (Gao et al. 2006) and is the standard for lakes in some other states (NCDWQ 2007). This definition was consistent with that used in analyses to develop TMDLs for the freshwater lower St. Johns River (Hendrickson et al. 2003). Last, when appreciable levels of the cyanobacterial toxin microcystin were detected in nutrient enrichment assays, chl-a concentration usually was greater than  $40 \mu\text{g L}^{-1}$  (Paerl et al. 2001b).

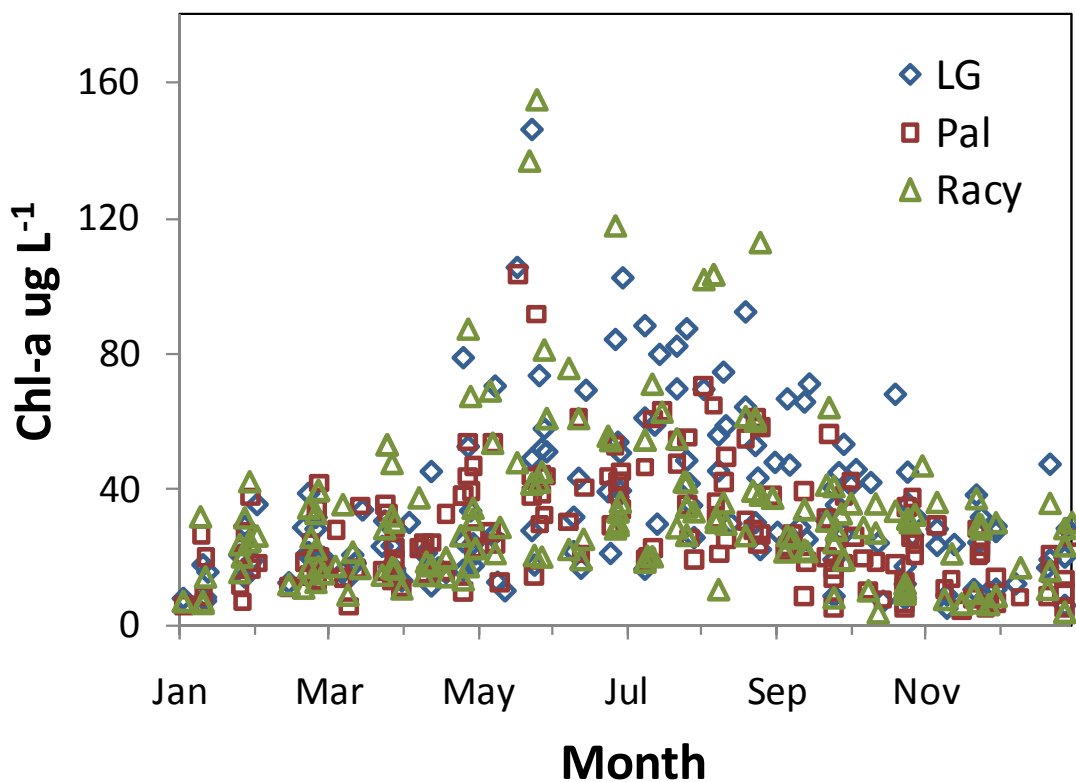


Figure 4–7. Chlorophyll-a (chl-a) measured at Lake George (LG), Palatka (Pal), and Racy Point (Racy) versus sample date (1993 to 2007).

During the growing season, cyanobacteria showed increasing dominance as chl-a values increased during blooms (Figure 4–8). At  $\text{chl-a} \geq 40 \mu\text{g L}^{-1}$ , cyanobacteria made up  $> 80\%$  of the total phytoplankton biomass based on median values from three sites during April to October (1993 to 2007).

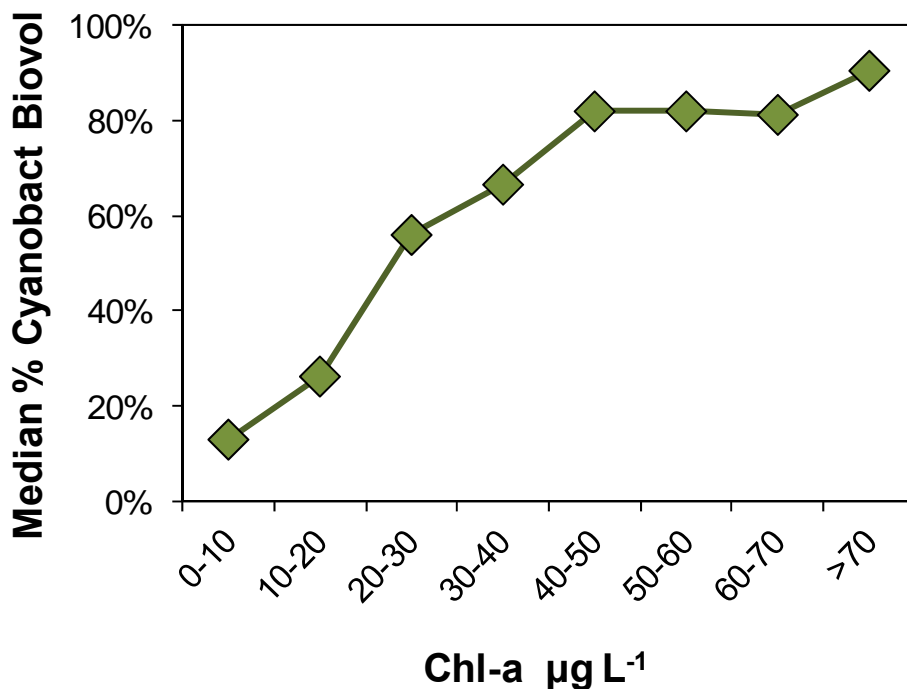


Figure 4–8. Percent cyanobacterial biovolume of total phytoplankton biovolume at three locations in the lower St. Johns River (i.e., Lake George, Palatka, Racy Point). Median values for growth season (April to October) samples from 1993 to 2007. Based on median values, cyanobacteria make up > 50% of biovolume at 20 to 30  $\mu\text{g chl-a L}^{-1}$  and > 80% above 40  $\mu\text{g L}^{-1}$ .

## 4.2 ALGAL BLOOM METRICS

### 4.2.1 POTENTIAL EFFECTS OF WATER WITHDRAWAL

Based on previous analyses, literature, and professional experience, we compiled a preliminary list of sixteen potential effects of water withdrawal on plankton, nutrients, and TMDLs (Table 4–1). Many of these effects were interrelated, and some effects were direct consequences of other effects (e.g., increased nutrient supply can cause more intense algal blooms; algal blooms can cause reduction in zooplankton abundance).

Table 4–1. Preliminary list of potential effects of water withdrawal on plankton, nutrients, and TMDLs.

No.	Category	Potential Effect
1	Phytoplankton	Change in phytoplankton community composition. Effects on food webs and energy flow
2	Phytoplankton	Increased phytoplankton biomass. Harmful algal blooms. Cyanobacterial toxins
3	Phytoplankton	Reduction due to entrainment
4	Zooplankton	Change in zooplankton community composition
5	Zooplankton	Decreased crustacean zooplankton abundance
6	Zooplankton	Reduction due to entrainment
7	Water quality and TMDLs	Increased nutrients due to reduced dilution or greater regeneration Increased N <sub>2</sub> -fixation
8	Water quality and TMDLs	Lower stage leading to increased resuspension, TSS, and nutrient concentrations
9	Water quality and TMDLs	Lower dissolved oxygen (DO) at longer residence times
10	Water quality and TMDLs	Change in water color (increased or decreased). Photolysis of DOC at longer residence time. Export of DOC if wetlands drained
11	Water quality and TMDLs	Increased pH and/or NH <sub>4</sub> <sup>+</sup> with toxic levels of un-ionized ammonia NH <sub>3</sub> for fish
12	Reverse osmosis (RO) concentrate	Increased nutrients in RO reject water
13	RO concentrate	Increased cyanobacterial toxins in RO reject water
14	Other	Influx of saline water into Doctors Lake
15	Other	Oxidation of flood plain soils. Nutrient release
16	Other	Pycnocline shifts upstream into areas with less tidal mixing. Stable density stratification could exacerbate low DO

We screened this preliminary set of effects using the following considerations:

- Some potential effects would be assessed by other working groups
- Some potential effects would be small or have very low chances for occurrence
- Remaining potential effects were causal to or were effects of enhanced growth of phytoplankton and should be aggregated and assessed together

Our final approach used the following four algal bloom metrics to assess the aggregated potential effects of water withdrawals: magnitude of freshwater algal blooms, duration of freshwater algal blooms, change ( $\Delta$ ) in N load, and marine algal blooms (Table 4–2). Freshwater algal blooms in the St. Johns River generally were dominated by cyanobacteria, and the marine algal blooms of concern were dominated by dinoflagellates. The following sections describe thresholds for adverse effects associated with each of these metrics.

Some of our preliminary effects dealt with possible consequences of entrainment of planktonic organisms or of discharge of reject water from reverse osmosis (RO) treatment systems (Table 4–2). We considered these preliminary effects to be small and to have a low probability of occurrence. First, we considered that the ecological effects (both negative and positive) of plankton largely are density dependent. If raw water were withdrawn from the St. Johns River, and no treated water returned, then the effects of withdrawal would be a decrease in flow and stage and an increase in retention time but no change in density of plankton. Details about proposed water treatment have yet to be developed. However, we understood that not all uses or all potential withdrawal sites would require membrane filtration (e.g., RO), and that return flow of reject water likely would be minimized in any case. Finally, we judged that plankton organisms would recover quickly from dilution by RO reject water due to their relatively high production rates.

Overall effects of water treatment on nutrient concentrations likely would be reduction rather than increase if RO reject water was returned. This is because initial treatment of water will include flocculation, coagulation, and settling of particulates prior to membrane filtration. Similarly, we assigned low probability to the return water increasing concentrations of cyanobacterial toxins in the St. Johns River because pretreatment of water should remove intracellular toxins effectively.

Another preliminary effect of water withdrawal dealt with the potential for the pycnocline to shift upstream. Such a shift might increase density stratification and exacerbate low DO. We judged this effect to have a low probability of occurrence. First, low DO is not a general problem in the oligohaline segment 2, where we expect the greatest potential change in salinity. Low DO is a chronic problem further downstream, but is not generally associated with density stratification. Second, the primary driver of DO consumption in segment 2 is excessive algal biomass, an effect that we consider in other algal bloom metrics (Table 4–2). Last, modeled surface salinities at the downstream limit of segment 2 were projected to change very little for water withdrawal scenarios based on 7-day maximum values (see Chapter 9. Submersed Aquatic Vegetation).

In initial discussions of potential effects of water withdrawals, we included increases in fecal coliform bacteria. We subsequently removed this effect from consideration because we could not identify a likely mechanism whereby water withdrawals might increase fecal coliforms.



Table 4–2. Results of screening the preliminary set of water withdrawal effects aggregated under four algal bloom metrics for final consideration. Some effects were eliminated from further analyses for the reasons indicated.

Bloom Metric	Potential Effects
Magnitude freshwater algal blooms	Change in phytoplankton community composition Increased phytoplankton biomass Harmful algal blooms Cyanobacterial toxins Increased nutrients due to reduced dilution or greater regeneration Photolysis of dissolved organic carbon (DOC) at longer residence time (improved light climate for algal production) Lower dissolved oxygen (DO) at longer residence times Increased pH and/or $\text{NH}_4^+$ with toxic levels of un-ionized ammonia $\text{NH}_3$ for fish
Duration of freshwater algal blooms	Effects on food webs and energy flow Change in zooplankton community composition Decreased crustacean zooplankton abundance Increased nutrients due to reduced dilution or greater regeneration Photolysis of DOC at longer residence time (improved light climate for algal production)
Change in N load	Increased $\text{N}_2$ -fixation
Marine algal blooms	Influx of saline water into Doctors Lake
No metric– Addressed by other working groups	Lower stage leading to increased resuspension, total suspended solids (TSS), and nutrient concentrations Change in water color (increased or decreased). Export of DOC if wetlands drained Oxidation of flood plain soils. Nutrient release
No metric– Small effect or low probability of occurrence	Reduction in phytoplankton due to entrainment Reduction in zooplankton due to entrainment Increased nutrients in reverse osmosis (RO) reject water Increased cyanobacterial toxins in RO reject water Pycnocline shifts upstream into areas with less tidal mixing. Stable density stratification could exacerbate low DO

#### 4.2.2 MAGNITUDE OF FRESHWATER ALGAL BLOOMS

We set thresholds for two types of adverse effects that can result from high density algal blooms.

The first threshold was based on shifts in composition of the phytoplankton community toward dominance by cyanobacteria in high biomass blooms and on health risks from cyanobacterial toxins. Based on median values for biovolume at several lower St. Johns River locations,

cyanobacteria accounted for > 50% of total phytoplankton at chl-a concentrations of 20 to 30  $\mu\text{g L}^{-1}$  and > 80% of phytoplankton at chl-a >40  $\mu\text{g L}^{-1}$  (Figure 4–8). Furthermore, high density algal blooms dominated by cyanobacteria present a risk for adverse health effects from cyanobacterial toxins. Neither the USA or Florida currently has regulatory standards for cyanobacterial toxins. Guidelines developed by the World Health Organization show a moderate probability of adverse human health effects at chl-a 50  $\mu\text{g L}^{-1}$  and a high probability of health effects where there are surface algal scums dominated by cyanobacteria (World Health Organization 2003). We weighed both these potential consequences of high-density algal blooms (altered phytoplankton community, cyanobacterial toxins) and set one threshold for adverse effects at a chl-a concentration 50  $\mu\text{g L}^{-1}$ .

The second adverse effect potentially associated with high density algal blooms was the depression of dissolved oxygen (DO) when blooms decline. The rapid decline (crash) of high density cyanobacterial blooms was a common phenomenon in the fresh tidal St. Johns River (Hendrickson et al. 2003) (Figure 4–9). The triggering events and mechanisms to explain these bloom crashes presently are unknown. Presumably, low DO levels during these crashes resulted from heterotrophic utilization of the dead phytoplankton biomass. Steep declines in DO appeared virtually coincident with declines in algal biomass, although the exact timing of changes in chl-a could not be ascertained as samples were collected at intervals (Figure 4–9).

We used continuous DO data to identify bloom crash DO minima. Two exceptions were DO minima on 1 June 2006, at Racy Point, and 10 June 2004, at Buffalo Bluff, when discrete monitoring data were substituted for periods of outages in the USGS DO stations. We compared the timing of bloom declines with patterns of river discharge and excluded DO minima associated with periods of rising water because these minima may have been caused by pulses of low DO water or allochthonous labile organic carbon. DO minima recorded at Buffalo Bluff on 12 June 2006 and 15 June 2009 were excluded for this reason. Fifteen DO minima associated with blue–green bloom declines remained.

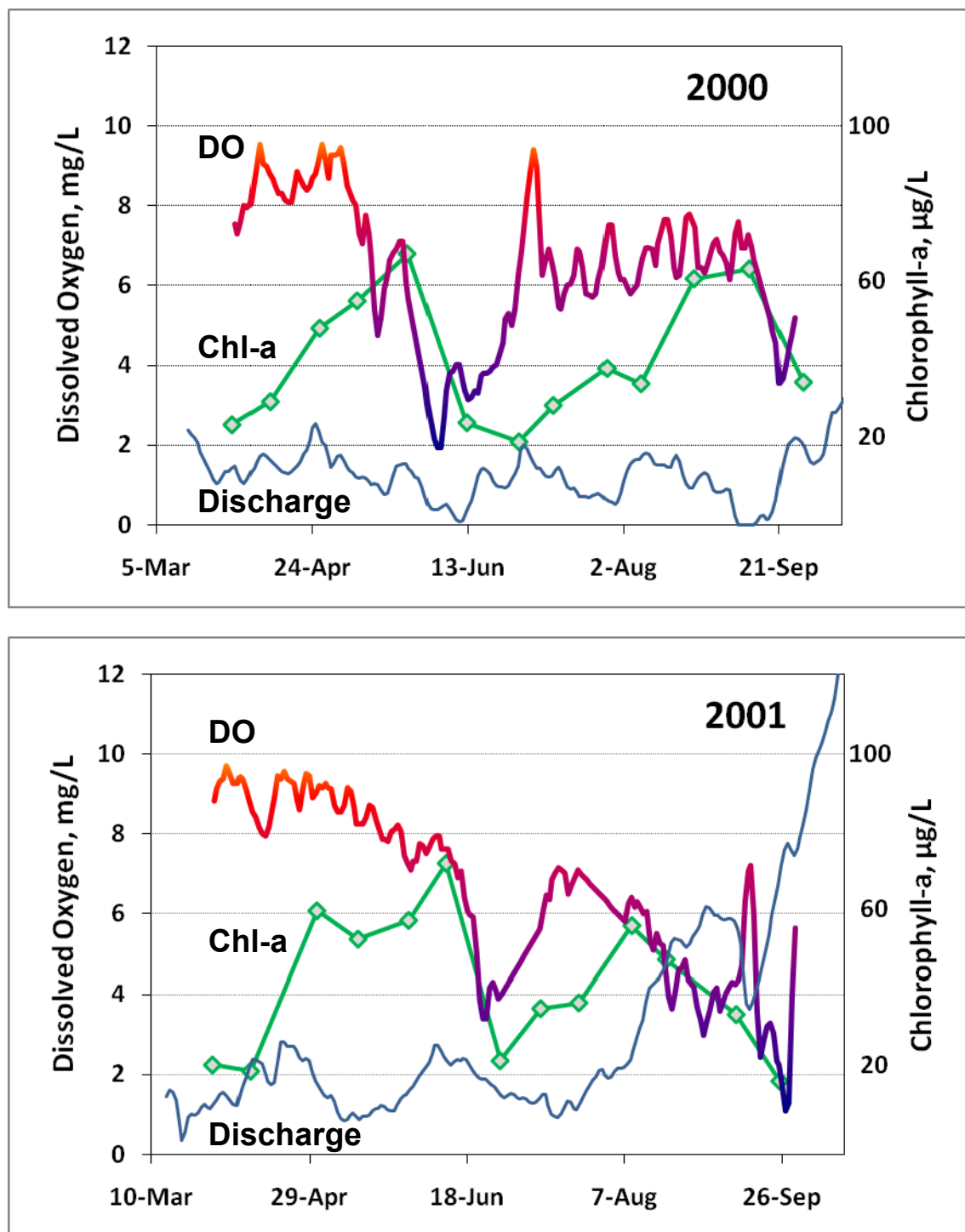


Figure 4–9. Examples of low dissolved oxygen (DO) coincident with collapse of summer algal blooms (chlorophyll-a [chl-a]). River discharge is shown in relative units. Lower St. Johns River USGS station at Dancy Point (DO), SJRWMD station at Federal Point (chl-a), and USGS station at Buffalo Bluff (discharge).

Low DO over large areas and for extended time can affect the feeding and reproduction of fish as well as cause mortality. Although sensitivity to low DO varies with species and life stage, the Fish Working Group recommended a general DO threshold of  $2.5 \text{ mg L}^{-1}$  for these adverse effects (Steven J. Miller, pers. comm.) (see Chapter 12. Fish). We found a linear relationship between water column DO measured after a bloom collapse and the magnitude (expressed as chl-a) of the preceding bloom (Figure 4–10). We used this relationship to translate the low DO threshold ( $2.5 \text{ mg L}^{-1}$ ) to an equivalent threshold for chl-a (bloom magnitude) of  $138 \mu\text{g L}^{-1}$ .

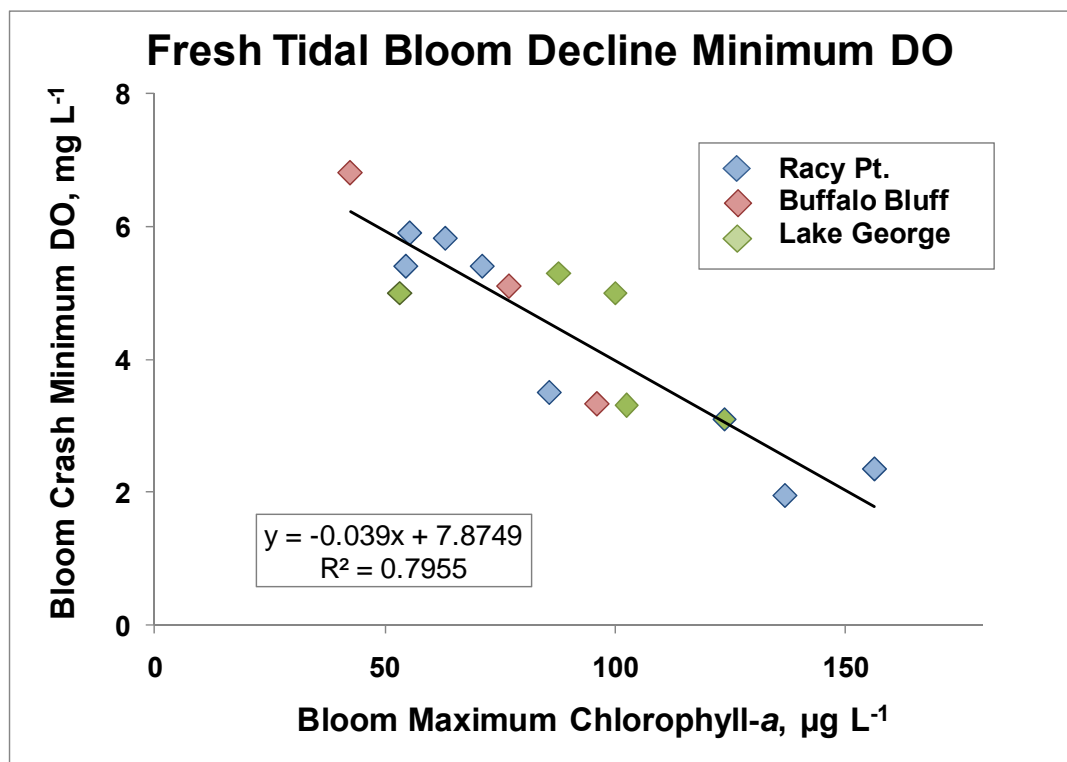


Figure 4–10. Minimum dissolved oxygen (DO) measured using continuous recording following collapse of cyanobacterial blooms at three locations in the freshwater lower St. Johns River. Depression in DO affected the entire water column. Fifteen bloom collapse events were measured during 1999 to 2009.

It is important to note that this threshold is based on chl-a and DO measurements in the river channel. However, cyanobacteria contain gas vacuoles and can be buoyant, and algal biomass can accumulate along downwind shorelines and in coves. This wind-driven accumulation could result in DO depletion in limited areas that is greater than the depletion predicted for the river channel. We did not have data to predict wind-driven accumulation of algal biomass from chl-a levels in the river channel, so we could not evaluate risks for low DO along shorelines. We are confident that adverse effects can occur over large areas at algal bloom magnitudes above  $138 \mu\text{g chl-a L}^{-1}$ , but we acknowledge that adverse effects could be present in smaller areas at chl-a levels in the river channel less than  $138 \mu\text{g chl-a L}^{-1}$ .

Low DO events associated with crashes of algal blooms were not the only occurrences of low DO in the fresh tidal lower St. Johns River. Low DO was also associated with late summer and early fall increases in discharge (see Figure 4–9). These periods with low DO likely resulted

from different mechanisms than bloom–crash events. The most likely explanation was the inflow of allochthonous organic matter and low DO water from recently saturated near stream source areas as well as the overall senescence of phytoplankton throughout the river system. Because these low DO events were associated with increasing rather than decreasing discharges, water withdrawals should not directly exacerbate these low DO events. One possible indirect effect of water withdrawals is increased transport of organic matter from upstream wetland areas and increased oxygen demand if withdrawals extend the time that wetland soils are exposed and oxidized between flood events. This effect was examined by the Biogeochemistry Working Group and was found to be negligible (see Chapter 7. Biogeochemistry).

#### **4.2.3 DURATION OF FRESHWATER ALGAL BLOOMS**

Zooplankton are primary consumers of phytoplankton and provide trophic transfer of primary production to higher consumers (e.g., fish). Up to a point, zooplankton will thrive on increased phytoplankton biomass. However, as phytoplankton biomass continues to increase, the proportion of algal species that are less desirable as food (i.e., cyanobacteria) tends to increase as well.

Inhibitory effects of cyanobacterial blooms on zooplankton have been described from numerous experiments and field studies (Benndorf and Henning 1989; Paerl et al. 2001a; Ghadouani et al. 2003; Leonard and Paerl 2005; Paerl and Fulton 2006). These effects on zooplankton have been attributed to poor digestibility or poor food quality of cyanobacteria, to physical interference with feeding by cyanobacterial trichomes and colonies, and to cyanobacterial toxins (Paerl et al. 2001a, Paerl and Fulton 2006). Earlier work in the lower St. Johns River (Phlips and Cichra 2002) found negative effects of blooms on crustacean zooplankton, and a conclusion that chlorophyll-a should not exceed  $40 \mu\text{g L}^{-1}$  for more than 40 days was part of development of nutrient TMDLs for the freshwater reaches of the lower St. Johns River (Hendrickson et al. 2003).

We examined relationships between the duration of algal blooms ( $\text{chl-a} > 40 \mu\text{g L}^{-1}$ ) and abundance of zooplankton at several locations in the lower St. Johns River. Peak abundances of each zooplankton group examined (cladocera, adult copepods, copepod nauplii, and rotifers) appeared to decline with increasing duration of algal blooms (Figure 4–11). High abundances were frequent during short periods of blooms, but were infrequent or absent as blooms extended in time. Low abundances could occur under all bloom conditions.

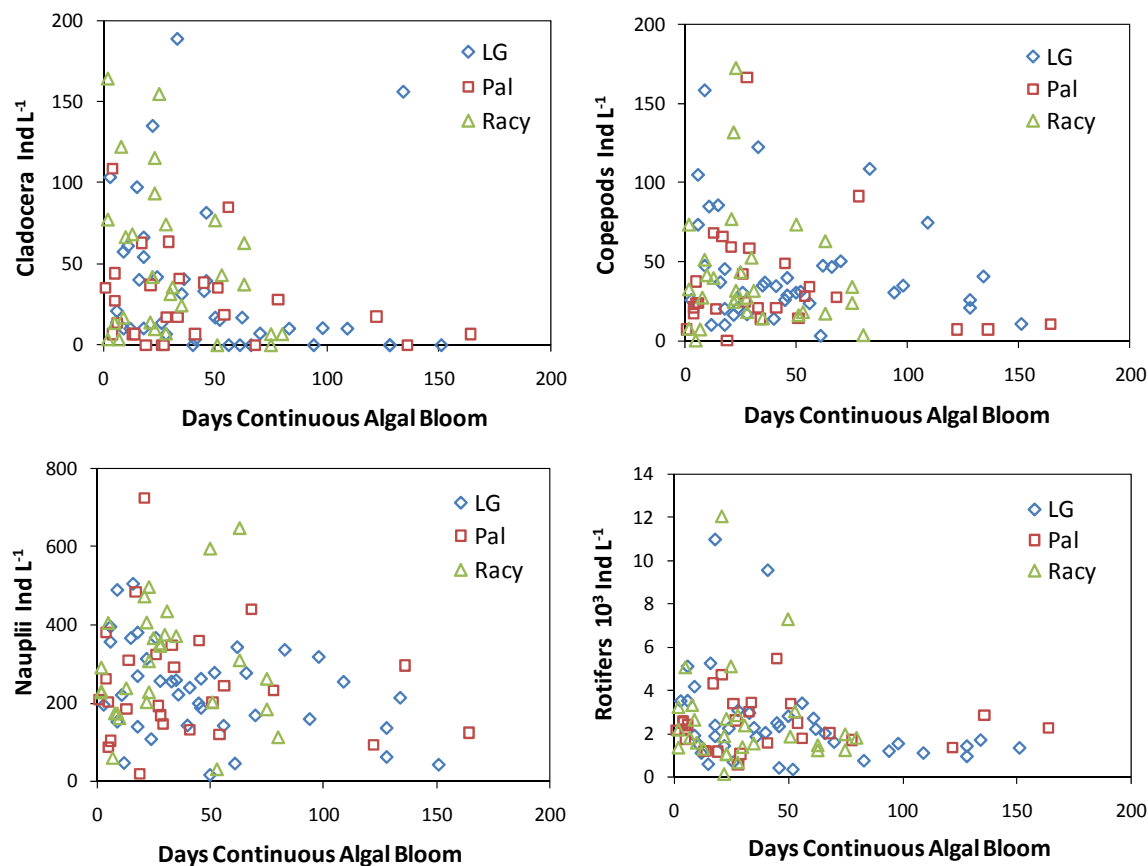


Figure 4–11. Relationships among abundances of four zooplankton groups (individuals L<sup>-1</sup>) and the duration of algal blooms for three locations in the lower St. Johns River: Lake George (LG), Palatka (Pal), and Racy Point (Racy). Samples taken during algal blooms ( $\text{chl-a} \geq 40 \mu\text{g L}^{-1}$ ), April to October (1997 to 2007).

Other factors besides inhibition by cyanobacteria could explain the lack of high zooplankton abundances during extended algal blooms. Long duration blooms extended over several months during which seasonal changes in factors, such as temperature (Figure 4–12), life-cycle patterns, or predation could have caused the apparent lower numbers of zooplankton during long blooms. If coincidental seasonal changes were responsible for lower zooplankton abundances during extended cyanobacterial blooms, then zooplankton numbers would be equivalent at similar times of year whether or not extended blooms occur; there would be no “bloom effect” on numbers.

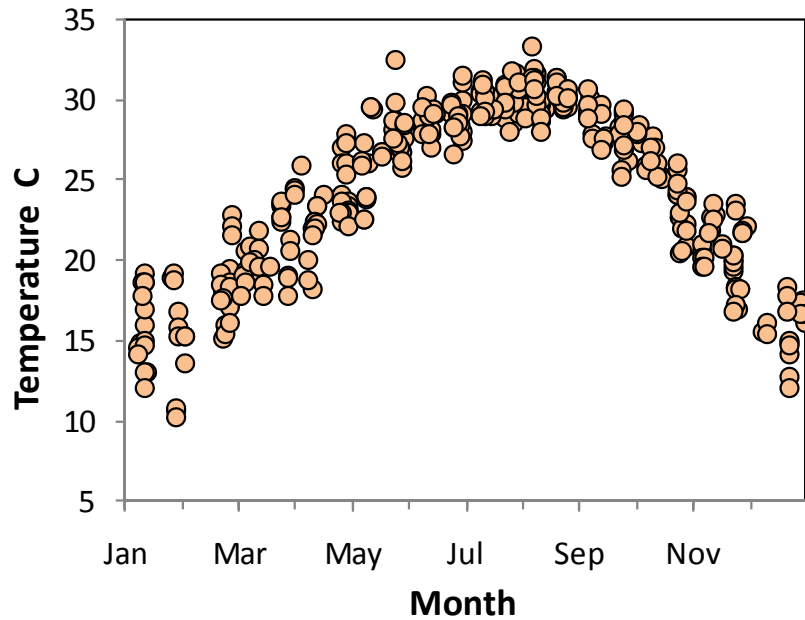


Figure 4–12. Seasonal variation in water temperature at three locations in the lower St. Johns River: Lake George, Palatka, and Racy Point (1997 to 2007).

We examined the alternative hypothesis that reduced zooplankton numbers were caused by seasonal changes by comparing zooplankton abundances by date with and without extended blooms. We defined extended blooms for this comparison as a bloom duration  $\geq 50$  days at time of sampling, because inspection of zooplankton abundances versus duration of blooms indicated that peak numbers largely were absent after that time (see Figure 4–11).

Abundances of cladocera during long duration blooms appeared to be lower than abundances on similar dates without blooms (Figure 4–13). Differences were not so apparent for the other three zooplankton groups. Clearly, one or more seasonal factors were present, because abundances of all groups declined from mid-summer to fall regardless of the presence of extended blooms (Figure 4–13).

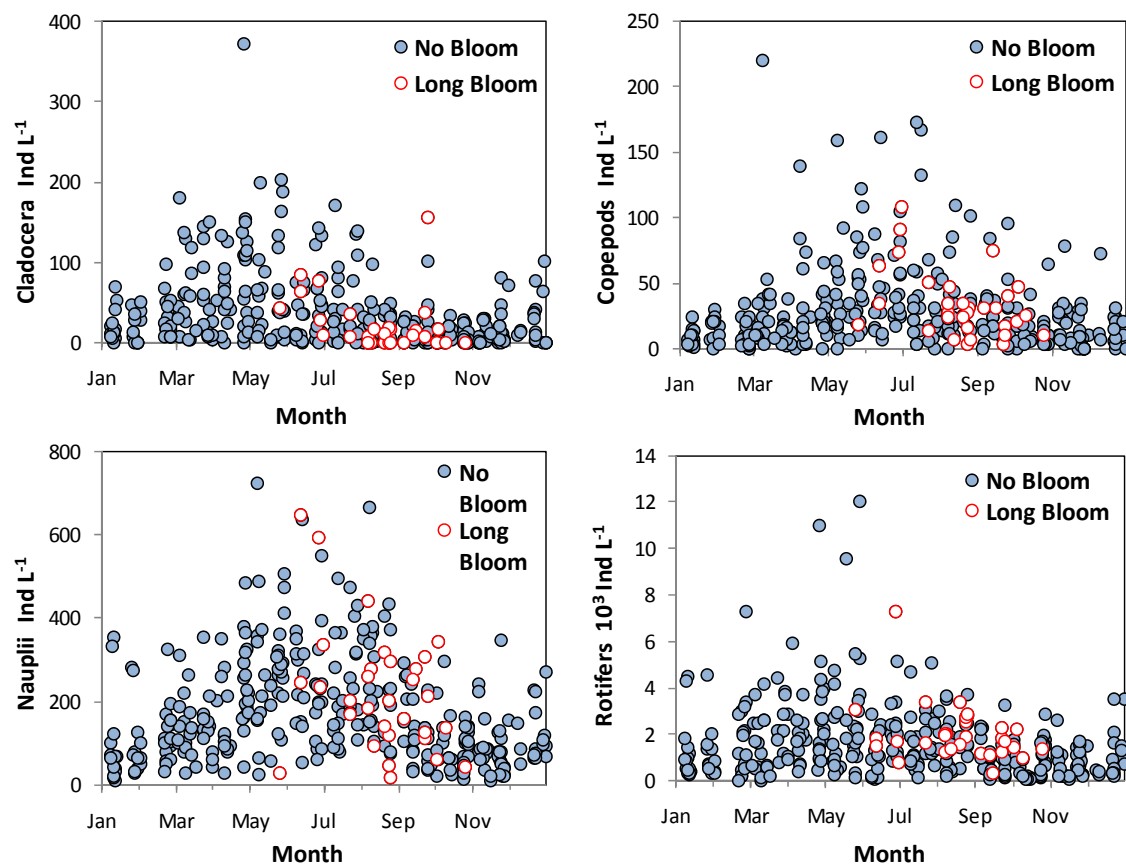


Figure 4-13. Relationships among abundances of four zooplankton groups (individuals  $L^{-1}$ ) and sampling date for three locations in the lower St. Johns River: Lake George, Palatka, and Racy Point. Filled symbols (No Bloom) indicate samples during periods with no or short duration ( $< 50$  days) cyanobacterial blooms. Open symbols (Long Bloom) indicate samples during extended blooms (bloom duration at time of sampling  $\geq 50$  days) (1997 to 2007).



We used ANOVA to test whether abundances of zooplankton differed significantly under bloom versus non-bloom conditions. Effects of seasonal factors were considered by inclusion of date as a covariate. We limited the data set to June to October, because all long duration bloom samples but one were taken after 1 June (Figure 4–13). Zooplankton abundances generally declined during June to October, which enabled a monotonic function to fit the relationship of zooplankton on the date covariate. We used the  $\ln(y+1)$  transformation on abundances to help equalize variances.

The date covariate was significant ( $p < 0.05$ ) for each zooplankton group (Table 4–3), which was evident from declining abundances after mid-summer (see Figure 4–13). The effects of long duration blooms were significant for cladocera and rotifers, but not for either life-stage of copepods (Table 4–3). The bloom effect was negative for cladocera and positive for rotifers; cladocera abundance was lower during extended blooms, whereas rotifer abundance was higher.

This test (ANOVA with date covariate) provided a stringent (worst-case) test of bloom effects, because a bloom effect present in the data set was reduced if samples affected by blooms were grouped with the “no bloom” samples. This situation was possible since the zooplankton community did not recover immediately after the collapse of long-duration blooms. Samples taken during a recovery period may still have shown bloom effects but would have been scored “no bloom” since no bloom was present during sampling. We cannot estimate the extent of this problem, but it would tend to mask a part of the real effect of blooms.

Table 4–3. ANOVA on  $\ln$ -transformed zooplankton abundances for three locations in the lower St. Johns River: Lake George, Palatka, and Racy Point. Effects were Date (covariate) and Bloom (presence or absence of long-duration  $\geq 50$  days] algal bloom at time of sampling). Shaded effects were significant at  $p < 0.05$ . Data set June to October, 1997 to 2007.

	Source	DF	Seq SS	Adj SS	Adj MS	F	P
<b><math>\ln(\text{Cladocera}+1)</math></b>	Date	1	49.860	49.311	49.311	26.99	0.000
	Bloom	1	8.482	8.482	8.482	4.64	0.033
	Error	169	308.722	308.722	1.827		
	<b>Total</b>	<b>171</b>	<b>367.064</b>				
<b><math>\ln(\text{Copepods}+1)</math></b>	Date	1	29.127	29.317	29.317	34.81	0.000
	Bloom	1	1.883	1.883	1.883	2.24	0.137
	Error	169	142.329	142.329	0.842		
	<b>Total</b>	<b>171</b>	<b>173.339</b>				
<b><math>\ln(\text{Nauplii}+1)</math></b>	Date	1	23.889	24.013	24.013	58.62	0.000
	Bloom	1	0.987	0.987	0.987	2.41	0.123
	Error	169	69.233	69.233	0.410		
	<b>Total</b>	<b>171</b>	<b>94.108</b>				
<b><math>\ln(\text{Rotifers}+1)</math></b>	Date	1	10.0014	10.2530	10.2530	13.23	0.000
	Bloom	1	9.1388	9.1388	9.1388	11.79	0.001
	Error	169	131.0076	131.0076	0.7752		
	<b>Total</b>	<b>171</b>	<b>150.1478</b>				

The significant decline in cladocera abundance during long-duration blooms was similar to a factor-ceiling distribution (Karr and Chu 1999, Carter and Fend 2005). This distribution can occur when the x-variable controls the maximum expression (ceiling) of the y-variable, but other (generally unknown) factors depress some y-values below the maximum. Our interest was the upper range of the distribution, or the ceiling on abundance apparently imposed by long-duration algal blooms.

Various methods have been proposed to define the upper bounds of factor-ceiling distributions, and quantile regression is one approach that has strong statistical support (Scharf et al. 1998, Cade and Noon 2003). We used quantile regression to fit a relationship to the upper region of cladocera abundance after ln-transformation to help equalize variances and to fit the curvilinear relationship (Figure 4–14). The slope of an 85<sup>th</sup> percentile line was significantly different from zero ( $p = 0.004$ ), whereas 90<sup>th</sup> and 95<sup>th</sup> percentile lines were not significant ( $p > 0.05$ ). This relationship was one tool in setting a threshold for adverse effects of blooms on the zooplankton community.

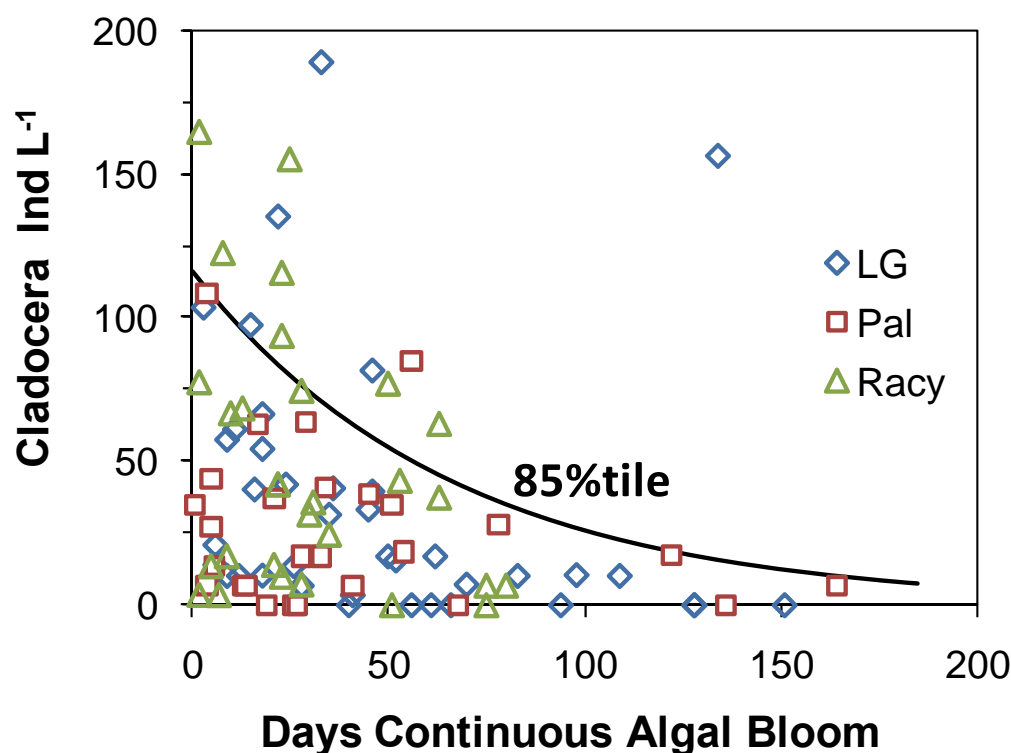


Figure 4–14. Relationship between abundance of cladocera (individuals  $L^{-1}$ ) and the duration of algal blooms for three locations in the lower St. Johns River: Lake George (LG), Palatka (Pal), and Racy Point (Racy). Regression line is 85<sup>th</sup> percentile fit to ln-transformed abundances using quantile regression. Samples taken during algal blooms ( $chl-a \geq 40 \mu g L^{-1}$ ), April to October (1997 to 2007).

We found two consequences of long-duration algal blooms for zooplankton and higher trophic organisms, the altered zooplankton community (i.e., decreased cladocera and increased rotifers) and the decline in high quality prey (cladocera) for small fish and other zooplanktivores.

Changes in abundance of both cladocera and rotifers were significant at 50-d duration blooms (see Table 4–3). The 85th percentile level for cladocera abundance was reduced from initial values of approximately 120 individuals  $L^{-1}$  to about 55  $L^{-1}$  at algal bloom duration 50 d (see Figure 4–14). Survival and growth of larval fishes may be limited at crustacean zooplankton abundance in this same range of 50 to 100  $L^{-1}$  (Breck 1993; Garvey and Stein 1998; Sammons et al. 2001; Hoxmeier et al. 2004; Dockendorf and Allen 2005). Furthermore, some cladocera have larger body mass than copepods and likely represent a better prey item.

Using the quantile regression, and employing professional judgment to weigh these consequences of blooms on zooplankton, we set a bloom duration threshold of 50-d for adverse effects.

#### 4.2.4 CHANGE ( $\Delta$ ) IN NITROGEN (N) LOAD

Most spring and summer phytoplankton blooms (algal blooms) in the fresh tidal St. Johns River progress to the point of dominance by  $N_2$ -fixing cyanobacteria (Phlips and Cichra 2002; Schelske and Aldridge 1994; Paerl et al. 2001b). These blooms could significantly increase TN concentration. For example, during a cyanobacterial bloom in Lake George in 1999, TN increased from 1.3  $mg L^{-1}$  on 23 February to 3.6  $mg L^{-1}$  on 22 June (data not shown).

We used mass–balance calculations for Lake George to estimate  $N_2$ -fixation as the net internal load of N (outputs minus inputs, Table 4–4). We chose Lake George because inputs and outputs were well characterized through direct monitoring, and algal biomass increased substantially from inflow to outflow.

Estimated annual  $N_2$ -fixation in Lake George from 1996 to 2005 varied from - 259 to 1780  $\times 10^6$  g and averaged 806  $\times 10^6$  g. By these estimates,  $N_2$ -fixation contributed 20% or more of the downstream flux of N in 1996 to 1998 (Table 4–4). To estimate total  $N_2$ -fixation in the lower St. Johns River, we multiplied values for Lake George by 1.37 to account for the additional areal extent of  $N_2$ -fixing cyanobacterial blooms downstream of Lake George.

We set a threshold for adverse effects of  $N_2$ -fixation, because high  $N_2$ -fixation represents an additional load of N to the freshwater lower St. Johns River where a TMDL is in place to reduce the N load by approximately 1540  $\times 10^6$  g  $N yr^{-1}$ . On this basis, we set a threshold for  $N_2$ -fixation where the additional N load (change from baseline to withdrawal scenario) exceeded 20% of the required reduction or 308  $\times 10^6$  g  $N yr^{-1}$  ( $0.2 \times 1540 \times 10^6$  g).

Other N-cycle processes would be included in our mass–balance estimates of  $N_2$ -fixation. These processes included sedimentation and resuspension, release of ammonia from sediments through decomposition, and denitrification. The net internal load actually was the sum of these processes plus  $N_2$ -fixation. Totaled over long periods, the sediments should be a net sink for N. Therefore, it is likely that our use of net internal load provided a lower bound for the true  $N_2$ -fixation value.

Table 4–4.  $N_2$ -fixation in Lake George estimated from the annual N budget. We calculated a lower bound for  $N_2$ -fixation as the net internal load (mass out – mass in). %  $N_2$  = Estimated  $N_2$ -fixation as percent of N mass leaving the lake.

Year	N inflow ( $N_{in}$ ) $10^6$ g	N outflow ( $N_{out}$ ) $10^6$ g	$N_2$ -fixation ( $N_{out}-N_{in}$ ) $10^6$ g	% $N_2$
1996	4485	5850	1365	23
1997	3756	5536	1780	32
1998	6344	7968	1625	20
1999	5638	6328	690	11
2000	3145	3244	99	3
2001	5672	6649	977	15
2002	6279	6020	(-259)	(-4)
2003	5411	5837	426	7
2004	6880	7578	698	9
2005	6332	6994	662	9

Acetylene reduction assays performed in three diel experiments in Lake George in 2004 by Paerl et al. (2005) found significant levels of  $N_2$ -fixation associated with cyanobacterial blooms. The mean light (total – dark) acetylene reduction rates, converted to nitrogen (4:1 Molar  $C_2H_2$  to  $N_2$ ), would produce a daily increase of  $4.4 \mu\text{g N L}^{-1}$ , or a total May through August increase of  $0.58 \text{ mg N L}^{-1}$ . This estimate was similar to the measured across-lake increase of  $0.67 \text{ mg N L}^{-1}$  from April through July for that same year, providing support for the estimation of  $N_2$ -fixation by mass-balance.

Piehl et al. (2009) also estimated net internal load of N in Lake George by mass-balance and demonstrated the potential significance of  $N_2$ -fixation. Our estimates differed somewhat from those of Piehl (2009) owing to our refinements in hydrologic calculations. These refinements better accounted for effects of reverse flows, which are common during low discharge conditions.

#### 4.2.5 MARINE ALGAL BLOOMS

The dinoflagellate plankton are a diverse group of photoautotrophs, mixotrophs, and heterotrophs, many of which exhibit fundamentally different adaptive strategies and environmental niches than those of cyanobacteria and the chlorophyte and bacillariophyte phytoplankton (Smayda, 2002). Dinoflagellates are of particular interest in estuarine ecosystem management, as many species produce toxins that can lead to the mortality of aquatic animals, contaminate seafood, and induce allergic and respiratory reactions in humans. Toxin production

capability is widespread among the dinoflagellates, but environmental triggers for the expression of toxin production are not well understood.

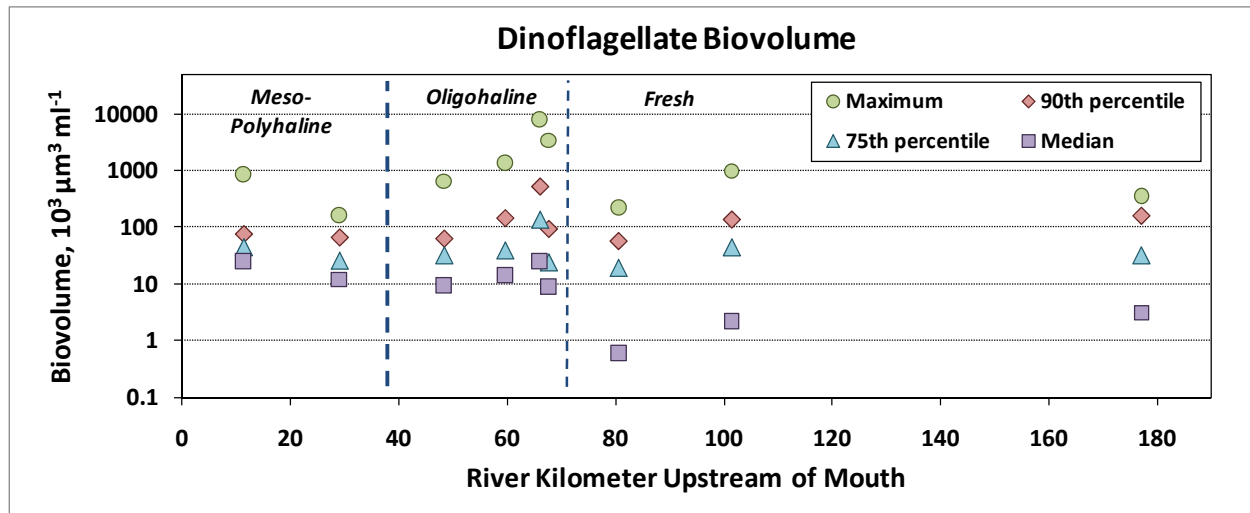


Figure 4–15. Longitudinal pattern of dinoflagellate biovolume for monthly samples in the St. Johns River, 1998 to 2007. Dashed lines indicate salinity zones.

Sample sites in the oligohaline reach of the St. Johns River consistently showed the highest dinoflagellate biovolume (see Figure 4–15). Seven percent of samples from offshore, oligohaline Doctors Lake exhibited what we considered bloom biovolume levels of  $> 1,000 \times 10^3 \mu\text{m}^3 \text{ml}^{-1}$ . High dinoflagellate biomass rarely occurred in the downstream meso- and polyhaline sections of the lower St. Johns River (see Figure 4–15). When high biomass did occur, it apparently stemmed from an Atlantic inner shelf bloom that entered the river with inflowing tides. The occasional appearance of *Karenia brevis* in the St. Johns River is an example of this mechanism.

Although dinoflagellate blooms in the oligohaline reach of the lower St. Johns River are not a regular occurrence, these blooms are of concern, because when they do occur they are often composed of genera or species that have been associated with toxin production elsewhere. These include species of the orders Gymnodiniales (*Akashiwo sanguinea*, *Katodinium spp.*), Peridinales (*Protoperidinium spp.*), and Prorocentrales (*Prorocentrum minimum*) (Table 4–5). Documented effects of harmful dinoflagellate blooms in northeast Florida include bitter crab outbreaks caused by the parasitic *Hermatodinium spp.* (Messick and Shields, 2000) and the presence of saxitoxins from *Pyrodinium bahamense* in puffer fish (Abbott et al. 2009). Recent analysis of blue crab (*Callinectes sapidus*) tissue from the St. Johns River indicated persistent, low to moderate level concentrations of saxitoxin in tissues (J. Landsberg, Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute, pers. comm.).

Table 4–5. Dominant taxa (all groups) during the four highest biomass dinoflagellate blooms in Doctors Lake in the St. Johns River oligohaline reach.

Date	Percent of Total Phytoplankton Biovolume	Species
27 May 1999	29.7	Prorocentrum minimum
	9.3	Unidentified Pyrrophycomphyta
	1.7	Gymnodinium sp.
23 August 2003	49.9	Akashiwo sanguinea
	36.1	Protoperidinium spp.
	3.8	Gymnodinium spp.
9 July 2007	28.4	Protoperidinium spp.
	24.6	Akashiwo sanguinea
	22.2	Oscillatoria-type species (including Limnithrix spp. and Planktothrix spp.)
7 August 2007	88.2	Protoperidinium spp.
	2.5	Oscillatoria-type species (including Limnithrix spp. and Planktothrix spp.)
	2.2	Unidentified centric diatom < 25µm

We set a threshold for adverse effects of marine algal blooms by consideration of the potential risk from toxic species if blooms of dinoflagellates should become more frequent or more intense. We considered the current occasional blooms of dinoflagellates to be surrogates for potential toxic blooms. We used professional judgment to set a dinoflagellate biovolume threshold of  $1,000 \times 10^3 \mu\text{m}^3 \text{ml}^{-1}$  for marine algal blooms. Although regulatory standards for single dinoflagellate taxa usually are expressed in cells  $\text{ml}^{-1}$ , we used biovolume because in an aggregated dinoflagellate assemblage, cell sizes vary greatly.

#### 4.2.6 ALGAL BLOOM METRICS, EFFECTS, AND THRESHOLDS

Table 4–6 summarizes the algal bloom metrics that we developed, the potential effects of water withdrawals associated with each metric, the variable measured to assess each metric, and the threshold set for adverse effects.

Table 4–6. Summary of algal bloom metrics, the associated effects, measured variables, and the thresholds used for evaluation. Freshwater algal blooms in the St. Johns River generally were dominated by cyanobacteria, and the marine algal blooms of concern were dinoflagellates.

Algal Bloom Metric	Effect(s)	Measured Variable	Threshold(s)
Marine algal blooms	Potential toxic species	Maximum annual dinoflagellate biovolume	$1,000 \times 10^3 \mu\text{m}^3 \text{ml}^{-1}$
Change ( $\Delta$ ) in N load	Additional N loading	Annual mass N added via $\text{N}_2$ -fixation	20% of TMDL reduction for LSJR Freshwater Reach or $308 \times 10^6 \text{ g N yr}^{-1}$
Magnitude of freshwater algal blooms (two effects)	1) Altered phytoplankton community; cyanobacterial toxins  2) Depletion of dissolved oxygen (DO); effects on fish reproduction, growth, and mortality	Maximum annual bloom chl-a	1) $50 \mu\text{g chl-a L}^{-1}$  2) $138 \mu\text{g chl-a L}^{-1}$
Duration of freshwater algal blooms	Altered zooplankton community; reduction in fish production	Duration of longest annual bloom	50 d

We did not assess each algal bloom metric in each segment of the river. We focused most of our examination on segments 3, 4, and 6 (freshwater lower St. Johns River and middle St. Johns River lakes) for two reasons:

- Segments 3, 4, and 6 already showed degraded water quality and suffered from algal blooms, periodic hypoxia, and risk for cyanotoxins (see Figure 3–5).
- Residence time (water age) in the lakes and lacustrine areas of segments 3, 4, and 6 should be more sensitive to reduced freshwater flows than would residence time in lotic areas or in estuarine areas further downstream that are flushed by tidal flows.

Despite our focus on segments 3, 4, and 6, we also examined potential effects of water withdrawals on dinoflagellate blooms in the oligohaline portion of the lower St. Johns River (segment 2) and on freshwater blooms in offline Doctors Lake (segment 2) because of development of cyanobacteria during periods of low salinity (see Figure 3–5). We also assessed effects of water withdrawals on freshwater phytoplankton blooms in segment 8 (Lake Poinsett) but used a slightly different methodology from that developed here (Appendix 8.B).

We did not assess effects on algal blooms in segments 1, 5, or 7. In the most saline portion of the estuary (segment 1), the strongest effects of freshwater flows on algal blooms likely involve nutrient loading. An excessive transport of nutrients downstream is a high flow rather than low flow phenomenon and would not be exacerbated directly by water withdrawals. A possible

indirect effect is increased transport of nutrients from upstream wetland areas if water withdrawals extend the time that wetland soils are exposed and oxidized between flood events. This effect was found to be negligible and is discussed in section [4.5.2](#).

We did not assess effects on algal blooms in segments 5 and 7 because phytoplankton communities in these riverine sections of the river are controlled by conditions in upstream lakes, and chl-a typically decreases as the river traverses these segments.

Further, we could not apply every bloom metric to every river segment even in segments 2 – 4, 6, and 8 where we made assessments. Marine (dinoflagellate) blooms could be assessed only where they occurred in oligohaline segment 2. We assessed the change in N load only for segments 3 and 4 where we had well-characterized mass balances for N and corroborating acetylene reduction assays (see above). The magnitude of freshwater algal blooms metric could not be used in segment 2 except for Doctors Lake because freshwater algal blooms were uncommon at other sites. We applied the magnitude of freshwater algal blooms (DO) metric only in segments 3 and 4, where our analysis of monitoring data linked collapse of blooms to low DO, and in Doctor's Lake, where we judged that similar relationships would apply. We did not assess the duration of freshwater algal blooms metric in Lakes Harney or Poinsett because the frequencies of long-duration blooms were too low to build useful hydroecological models.

Table 4–7 summarizes the algal bloom metrics that we assessed by river segment. We assessed some segments at a single location; other segments were assessed at several locations.



Table 4–7. Filled panels indicate algal bloom metrics assessed in the St. Johns River system by river segment. Metrics include one or two (i.e., magnitude of freshwater algal blooms) potential effects of water withdrawals on plankton (see Table 4–6). We assessed both single and multiple locations depending on segment. See Section 4.2.6. for details on river segments assessed and metrics assessed in each segment.

River Segment <sup>*</sup>	Location	Algal Bloom Metric				
		Marine Algal Blooms	Change in N Load	Magnitude of Freshwater Algal Blooms	Magnitude of Freshwater Algal Blooms (DO)	Duration of Freshwater Algal Blooms
2	Mandarin Point					
	Doctors Lake					
3	Racy Point					
4	Lake George					
6	Lake Monroe					
	Lake Jesup					
	Lake Harney					
8	Lake Poinsett					

\* Segments 1, 5, 7, and 9 were not assessed by the Plankton Working Group.

### 4.3 ASSESSMENT OF PREEXISTING IMPAIRMENT

Most reaches of the St. Johns River have nutrient TMDLs or are listed as impaired (due to nutrient loading) by the Florida Department of Environmental Protection (FDEP) (see Figure 3–2). The state listing is based on a water body not meeting state water quality standards. Our assessment of effects of water withdrawals on plankton was separate and fundamentally different from the state water quality assessment. We considered potential ecological effects of water withdrawals and developed four algal bloom metrics with thresholds for adverse effects (see Figure 2–1 and Table 4–6).

An important first step in predicting the change in algal bloom metrics due to water withdrawals was to assess preexisting impairment using the same methodology. An understanding of pre-existing impairment provided context for assessment of the predicted changes.

We assessed preexisting impairment using the frequency of years that bloom thresholds (see Table 4–6) were exceeded in monitoring data sets for 1995 to 2005 at the various locations and

river segments. We chose that period to correspond with the modeling period for evaluation of water withdrawals.

The WSIS Working Groups developed a common terminology for levels of environmental effects (see Chapter 2. Comprehensive Integrated Assessment). These levels ranged from extreme to negligible, based on considerations of three characteristics of effects: persistence, strength, and diversity. To assess preexisting impairment, we considered these characteristics as well as the definition of each effect level. We used professional judgment to assign levels of effects based on the frequency of years that bloom thresholds were exceeded (Table 4–8).

For example, a major effect was defined as an alteration of ecosystem components that significantly reduces natural resource values but does not alter ecosystem character (Chapter 2). In our view exceedence of thresholds, for example, magnitude of algal blooms (dominance of cyanobacteria, risk from cyanotoxins, low DO) or duration of algal blooms (altered zooplankton community, risk to fish production) in more than 50% of years (frequency > 0.5) met that definition. Ecosystem components (e.g., phytoplankton and zooplankton communities) would be altered, resource values (e.g., recreation, fish production) would be reduced, and the reduction would be significant because it occurs in more than 50% of years.

As a further example, a moderate effect was defined as an alteration of ecosystem components that does not significantly reduce natural resource values (Chapter 2). Similar to the case above, we judged that exceedence of algal bloom thresholds in more than 30% of years (frequency > 0.3) met that definition. Ecosystem components would be altered, but with a frequency less than 50% of years, we judged that resource values would not be significantly reduced.

We used analogous reasoning to assign a frequency for exceedence of algal bloom thresholds to minor and negligible levels (Table 4–8). We did not define a frequency for extreme effects. Our consensus was that several of the bloom thresholds that we developed could be exceeded every year (frequency = 1) without producing conditions that met the definition of extreme effects (effects that alter ecosystem character and significantly change natural resource values) (see Chapter 2. Comprehensive Integrated Assessment). Other algal bloom metrics would need to be developed to distinguish extreme from major effects.

Table 4–8. Levels of effects used to assess preexisting impairment in the St. Johns River using the frequency of years that bloom thresholds (see Table 4–6) were exceeded in monitoring data sets for 1995 to 2005. We did not define a frequency for extreme effects.

Level	Frequency of Years that Threshold Exceeded
Extreme	Not defined
Major	> 0.5
Moderate	> 0.3
Minor	> 0.1
Negligible	≤ 0.1

We determined preexisting impairment for each location in the St. Johns River where we assessed effects of water withdrawals (Table 4–9). To determine overall preexisting impairment for each segment, we gave greater weight to the magnitude of freshwater algal blooms metric (lower threshold 50 µg chl-a L<sup>-1</sup>) and the duration of freshwater algal blooms metric, than remaining metrics. These two metrics dealt with direct effects of algal blooms. In contrast, the marine algal bloom metric dealt with the risk for toxic dinoflagellate blooms where they did not occur, and the change in N load metric dealt with an indirect effect of cyanobacterial blooms. Further, where multiple locations were assessed in a single segment of the river, we used the worst-case location to determine overall impairment.

We found a preexisting major overall impairment in all segments of the St. Johns River that we assessed except for segment 8 (Lake Poinsett), in which the preexisting impairment was minor (Table 4–9).

Table 4–9. Preexisting impairment in the St. Johns River system assessed by the same algal bloom metrics used to assess potential effects of water withdrawals (see Table 4–6). The overall assessment considered multiple locations and all bloom metrics.

River Segment*	Location	Algal Bloom Metric					Overall
		Marine Algal Blooms	Change in N Load	Magnitude of Freshwater Algal Blooms	Magnitude of Freshwater Algal Blooms (DO)	Duration of Freshwater Algal Blooms	
2	Mandarin Point	Negligible					Major
	Doctors Lake	Minor		Major	Negligible	Major	
3	Racy Point		Moderate	Major	Minor	Moderate	Major
4	Lake George		Moderate	Major	Negligible	Major	Major
6	Lake Monroe			Major		Moderate	Major
	Lake Jesup			Major		Major	
	Lake Harney			Minor		Minor	
8	Lake Poinsett			Minor		Minor	Minor

\*Segments 1, 5, 7, and 9 were not assessed by the Plankton Working Group.

## 4.4 ECOHYDROLOGICAL MODELS TO PREDICT ALGAL BLOOM METRICS

### 4.4.1 EMPIRICAL REGRESSION MODELS

We developed regression models to predict algal bloom metrics from water age. Dependent variables for regression models were the quantities measured for each of the four algal bloom metrics (see Table 3–4). These variables were tabulated annually and represented either the maximum (biovolume, chl-a, bloom duration) or the total (N mass) for that year.

Independent variables for regression models were minimum, mean, and maximum water age and the inverse of each water age calculated for seven time periods, for a total of 42 possible variables (see Figure 3–9). We included this wide range of water age variables as potential prediction variables in regression models because relationships between algal bloom patterns and hydrology could be both positive and negative and show complex seasonality. For example, chl-a in the lower St. Johns River and middle St. Johns River tended to increase to saturation levels with greater water age despite the fact that TP decreased (in the lower St. Johns River) or was

level (in the middle St. Johns River), perhaps because available light increased as water color declined with water age (Figure 4–3, Figure 4–4). Relationships between chl-a and TP could be either negative or positive depending on the range in water age considered (Figure 4–5). Further, although chl-a had a strong annual pattern, water age showed both annual and multi-year changes (Figure 4–6), and antecedent flow conditions likely affected subsequent algal blooms (see section 4.1.2).

One implicit assumption in our use of these empirical regression models is that nutrient loading remains unchanged. If this were not the case, then models developed with current data might not represent future conditions accurately. This assumption was appropriate for the following reasons. Current levels of nutrients are excessive in most of the St. Johns River system (see Figure 3–2), and substantial reductions in loads of both N and P are required under existing TMDLs (see Figure 3–3 and Figure 3–4). However, the projected 2030 hydrologic scenarios evaluated in this report predicted increased land development and attendant overall greater flows of water in the St. Johns River (see Chapter 3. Watershed Hydrology). Increased development and more stormwater runoff might increase nutrient loads, but TMDLs require that loads should be reduced. The assumption of current nutrient levels was reasonable given this uncertainty in future nutrient conditions and that current nutrient levels are intermediate between the other possible outcomes (e.g., lower nutrient levels if TMDLs are met or higher nutrient levels because of greater development). Further, an assumption of constant nutrient levels was consistent with the focus of WSIS on the effects of water withdrawals on ecological conditions rather than the effects of nutrient loading.

We constructed a total of nine different regression models using both multiple linear and multiple logistic regression techniques (Table 4–10, Appendix 8.B). These regression models covered the four algal bloom metrics in the five river segments that we assessed (Lake Poinsett in segment 8 was assessed in Appendix 8.B). Not all metrics were assessed in all segments (see Table 4–7). For segment 6 we used only data from Lakes Monroe and Jesup to develop the regression equation for freshwater bloom duration. We omitted Lake Harney because chl-a concentrations rarely met our working definition of an algal bloom (chl-a concentration  $\geq 40 \mu\text{g L}^{-1}$ ). Blooms occurred only three of 11 years in Lake Harney. Because of that sparse data set, we omitted Lake Harney both from development of the regression model and from subsequent assessments using that model.

Table 4–10 Eight of the nine empirical regression models to predict bloom variables (Table 4–6) from water age variables. For bloom variables freshwater maximum chl-a and freshwater bloom duration, three separate models were developed for segments 2, 3 + 4, and 6. One additional regression model was developed in Appendix 8.B for Lake Poinsett (segment 8). NA= Not applicable

Bloom Variable	Segment	Type	Number of Independent Variables	Adjusted R <sup>2</sup> Value
Ln (Dinoflagellate Biovolume)	2	Linear	5	0.81
N <sub>2</sub> -Fixation	3 + 4	Linear	3	0.96
Freshwater Max Chl-a	2	Linear	4	0.91
Freshwater Max Chl-a	3 + 4	Linear	7	0.88
Freshwater Max Chl-a	6	Linear	4	0.80
Freshwater Bloom Duration	2	Linear	4	0.97
Freshwater Bloom Duration Event	3 + 4	Logistic	2	NA
Freshwater Bloom Duration	6	Linear	4	0.91

Eight of the regression models were linear regressions with adjusted R<sup>2</sup> values that varied from 0.80 to 0.97. The number of independent variables selected for these models varied from two to seven (see Table 4–10). In each case, both the overall model and the individual independent variables were significant ( $p < 0.05$ ). The remaining model (freshwater bloom duration in segments 3 and 4) was a logistic regression with two independent variables (Table 4–10). In this case, we chose logistic regression because the best fit linear regression model achieved only a modest R<sup>2</sup> value.

We used linear regression models to evaluate both the frequency that bloom thresholds were exceeded over the simulation period and the median value of the algal bloom metric. The logistic model predicted only the frequency of exceedence.

Below are examples of two of the nine regression models: one multiple linear regression model and the multiple logistic regression model. Appendix 8.C includes definitions of all water age variables, all regression model equations, and goodness-of-fit diagnostics for each model.

#### 4.4.2 MULTIPLE LINEAR REGRESSION MODEL

We used a multiple linear regression model to predict freshwater bloom magnitude (maximum chl-a) in segments 3 and 4. Seven independent variables were selected for the model (Table 4–11). Water age variables from periods D and E (July through December; see Figure 3–9) dominated the model, although variables from the entire bloom season (April through October;

see Figure 3–9) and from the antecedent period A (see Figure 3–9) also were selected. We used data from two locations (Lake George and Racy Point), and because period A was represented in the regression, the data set consisted of 20 observations.

This model for maximum chl-a had an adjusted  $R^2$  of 0.88 (see Table 4–10). Standard partial regression coefficients, which measure the relative contribution of each variable in the regression, showed three tiers of effect strengths. Variable MinAgeD had the greatest effect, followed by three variables at similar but lower weights and two variables with the lowest weights (Table 4–11). Observed and predicted chl-a values were similar throughout the range, and residuals appeared normal with no evidence of bias (Figure 4–16).

Table 4–11. Independent variables selected for the linear multiple regression model to predict freshwater bloom magnitude (maximum chl-a) in segments 3 and 4. Standard partial regression coefficients indicate the direction (negative or positive) of the effect, and the absolute values show the relative weight of each independent variable in prediction of maximum chl-a.

Variable	Standard Partial Regression Coefficient
MinAgeD*	-2.318
invMinAgeD	-1.764
MeanAgeD	1.367
MaxAgeE*	1.297
invMeanAgeE	0.755
invMean_Age_Apr_Oct	0.636
invMaxAgeA*	0.540

\*A, D, and E are defined in Figure 3–9.

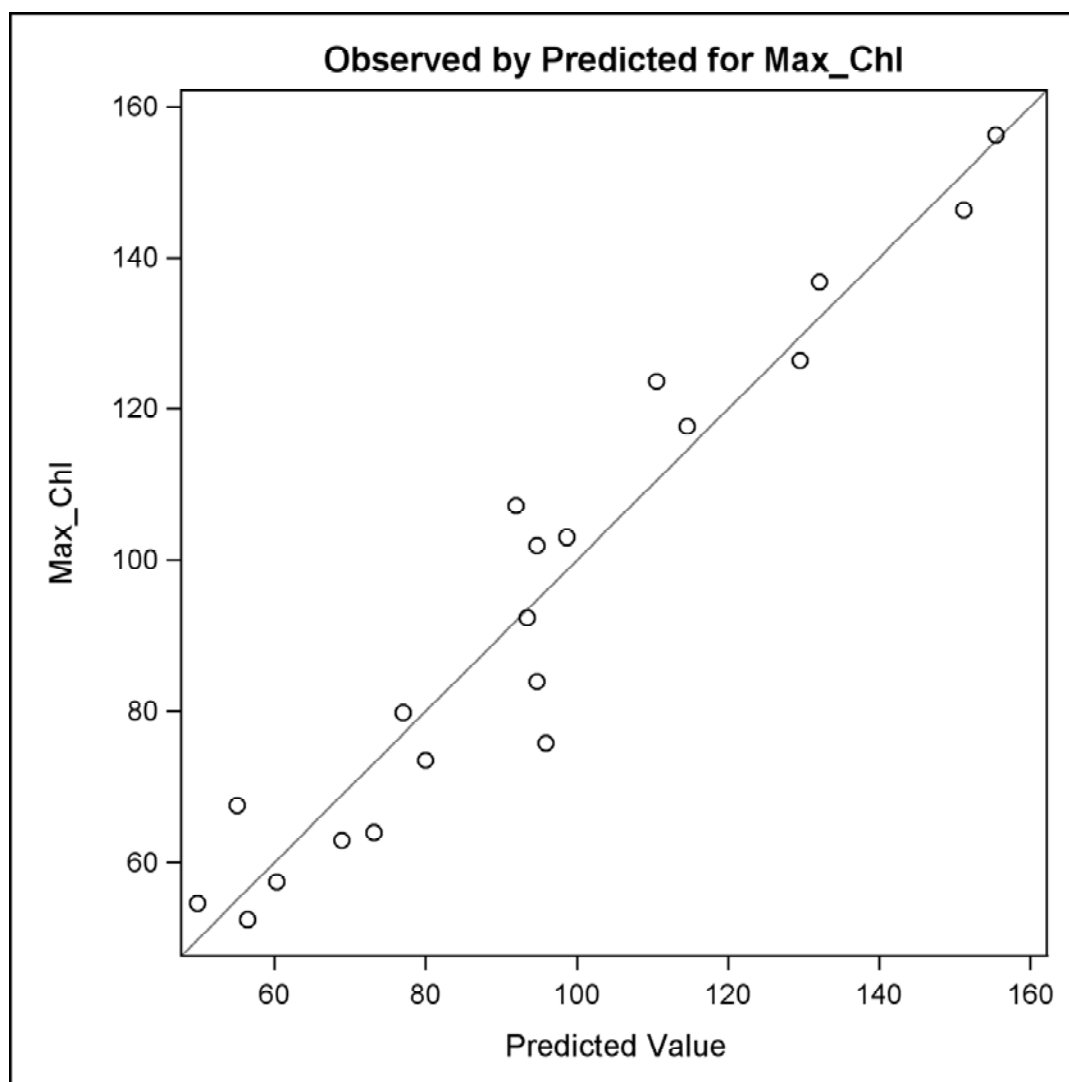


Figure 4–16. Observed versus predicted chl-a ( $\mu\text{g L}^{-1}$ ) for the multiple linear regression model used to predict freshwater bloom magnitude (maximum chl-a) from water age variables in segments 3 and 4. Line shows 1:1 relationship. Adjusted  $R^2 = 0.88$ .

#### 4.4.3 MULTIPLE LOGISTIC REGRESSION MODEL

We used a multiple logistic regression model to predict the annual probability of a long bloom event (maximum bloom duration  $> 50$  d) in segments 3 and 4. We chose this alternative because the analysis did not require an estimate of the duration of blooms but rather an estimate of the likelihood of bloom duration exceeding the 50-d threshold. Whether the predicted bloom duration was, e.g. 60 d, 100 d, or 120 d, was less important.

Binomial logistic regression does not fit an equation for a continuous response variable but fits an equation for the probability of an event based on values of one or more prediction variables. In our case, the event was annual exceedence of the bloom duration threshold. The response variable (annual maximum bloom duration) was coded as 1 = event (maximum bloom duration  $> 50$  d), and 0 = no event (maximum bloom duration  $\leq 50$  d); and a logistic equation was fit to the



coded values. We used data from three locations (Lake George, Palatka, and Racy Point), so the data set consisted of 33 observations. The best fit set of independent (prediction) water age variables was selected by highest likelihood score.

Two independent variables were selected for the model: Mean\_Age\_Apr\_Aug and invMeanAgeC, both with positive signs). A useful binomial logistic regression should predict low probabilities for observations where no event occurred and high probabilities where the event occurred. The model fit this pattern for long-duration (> 50 d) bloom events with the exception of two observations: Lake George 2005 and Palatka 2000 (Figure 4–17). The overall regression was significant (likelihood ratio test of global null hypothesis,  $p < 0.001$ ), and both regression coefficients were significant (Wald maximum likelihood chi-square,  $p < 0.05$ ). The Hosmer-Lemeshow goodness-of-fit test confirmed that the observed frequency of events did not differ significantly from that predicted by the regression (chi-square,  $p > 0.05$ ).

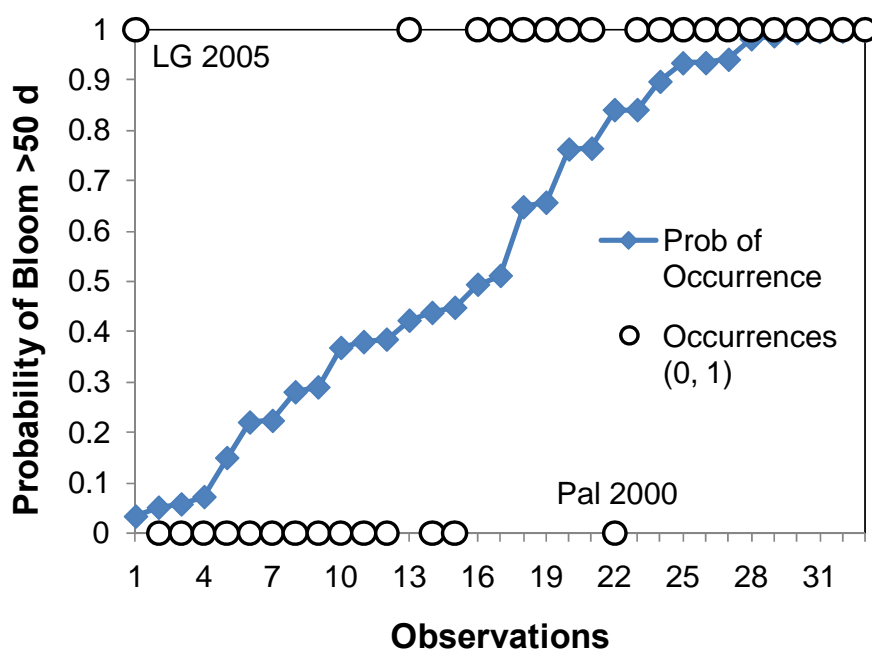


Figure 4–17. Prediction of the annual probability of long-duration (> 50 d) bloom events for segments 3 and 4 by binomial logistic regression. Observations were coded (1 = event, 0 = no event) for regression. The coded values and predicted bloom probabilities were plotted in order of increasing probability. Two points are labeled for discussion (see text).

#### 4.4.4 CE-QUAL-ICM MECHANISTIC MODEL

We evaluated the mechanistic water quality model CE-QUAL-ICM as a second approach to predict changes in algal bloom metrics from changes in hydrology that was independent from regressions between algal bloom metrics and water age. We calibrated the CE-QUAL-ICM model through iterative adjustment of the algal kinetic control parameters within bounds defined by the literature and lower St. Johns River data. Because this model was applied to the lower St. Johns River previously for TMDL analyses (Sucsy and Hendrickson 2004, Tillman et al. 2004,

Limno-Tech 2005), these further changes were relatively minor. Adjustments were made to the cyanobacteria predation rate, settling rate, and salinity toxicity terms to reflect the low level of grazing, low net settling, and low salinity tolerance derived from studies in the lower St. Johns River (Paerl et al. 2001b). Adjustments also were made to settling terms in the model to match the net settling of phosphorus calculated from the Lake George input – output mass balance (see Appendix 8.D for details).

We calibrated the CE-QUAL-ICM model using data for the outlet of Lake George and the freshwater lower St. Johns River at Racy Point. The principle objective in calibration was to capture the seasonal and inter-annual pattern in phytoplankton biomass measured as chl-a. Model performance was gauged primarily on its ability to reflect annual peak biomass. We also examined how observed and modeled chl-a:TP ratios varied with regard to residence time.

Simulated chl-a values had a seasonal pattern similar to that of the empirical data. Simulated bloom biomass, however, tended to remain high longer than observed levels at both Lake George and Racy Point (Figure 4–18). We investigated a bloom crash algorithm for the model; however, our poor understanding of the triggers for bloom crashes led us to abandon this modification. In many years, simulated maximum chl-a considerably exceeded the observed data. CE-QUAL-ICM predicted similar chl-a maxima year to year, whereas observed maxima varied considerably among years (Figure 4–18). The use of constant, mean nutrient loads from point sources and from some watersheds may have contributed to the lower interannual variation in simulated chl-a.

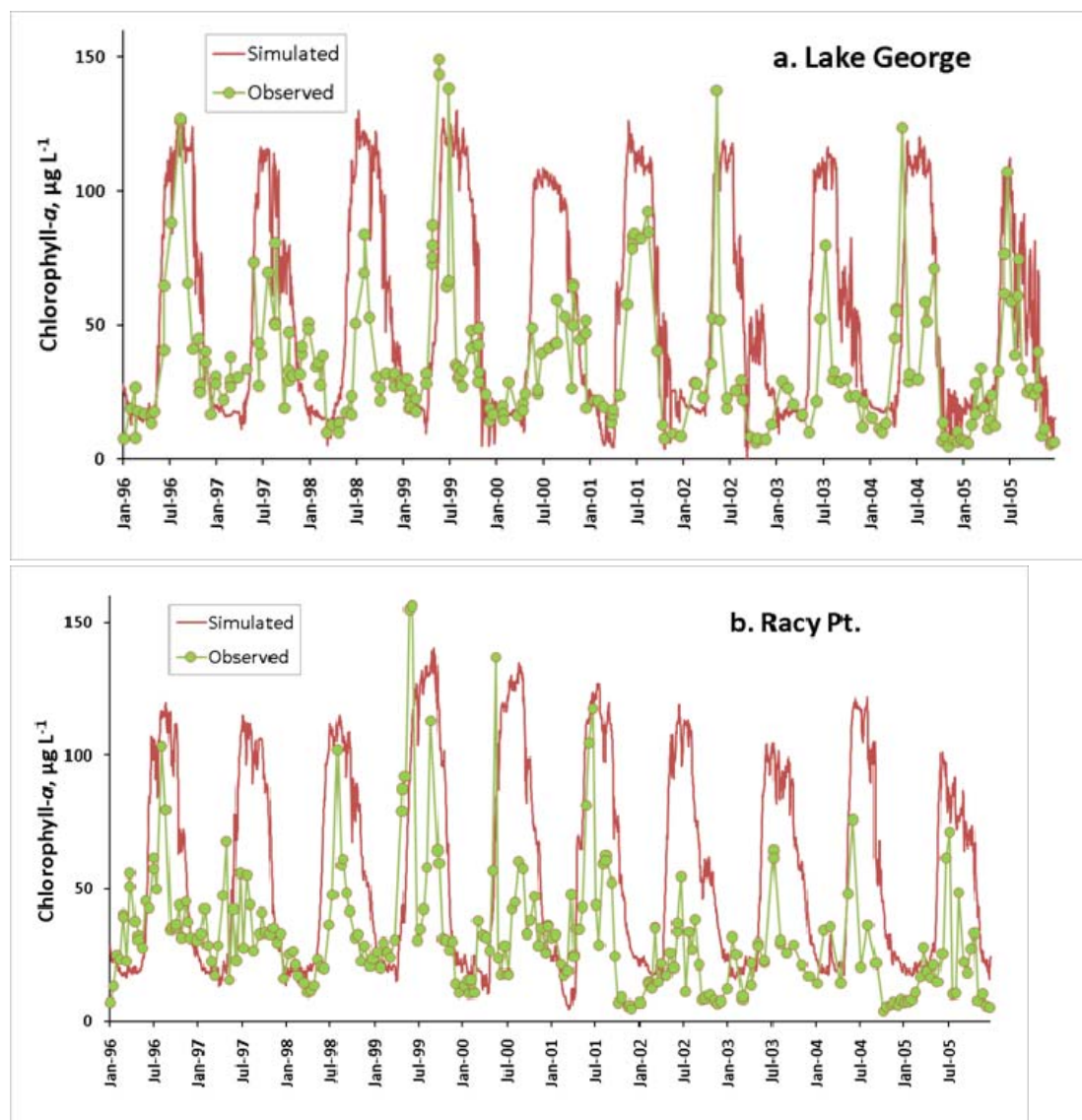


Figure 4–18. Time series for observed chl-a and continuously simulated chl-a done with the mechanistic water quality model CE-QUAL-ICM for (a) Lake George and (b) Racy Point

Daily values for simulated and observed chl-a had a significant positive relationship, but simulated values were biased upwards (Figure 4–19). The ability of the CE-QUAL-ICM model to predict annual bloom maximum chl-a was relevant to our use of this model in evaluation of water withdrawals. Observed and predicted maximum chl-a showed a significant linear relationship (Figure 4–19). However, at both Lake George and Racy Point, the simulated maxima were greater than observed for years when observed bloom peaks were less than  $100 \mu\text{g L}^{-1}$ . We expected some bias in the model predictions toward higher bloom peaks because empirical data, which were based on intermittent sampling, may miss peak biomass whereas the continuous model output would not. However, the consistent overestimation of lower bloom peaks by the model likely had other causes.

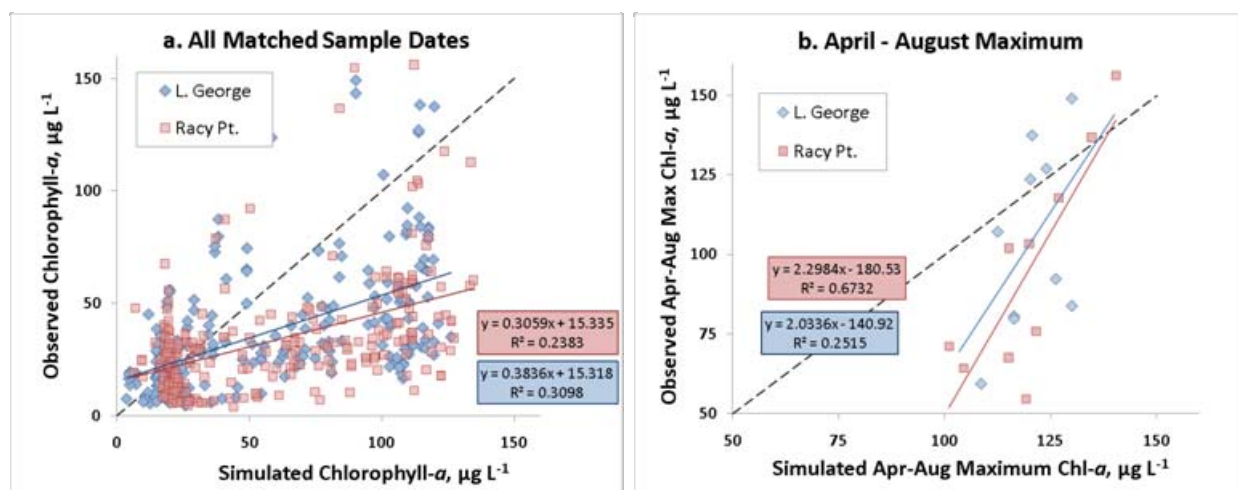


Figure 4–19. Observed chl-a versus simulated chl-a based on (a) daily values, and (b) annual bloom maximum values. Values are shown for both Lake George and Racy Point. Chl-a was simulated with the mechanistic water quality model CE-QUAL-ICM for 1996 to 2005.

CE-QUAL-ICM model simulations captured some of the relationships of peak bloom chl-a and phytoplankton utilization of nutrients with water age that were found in observed data. For both observed and simulated data, bloom maximum chl-a did not vary with mean water age in Lake George and showed only slight variation at Racy Point (Figure 4–20a, b). We interpreted these relationships to reflect the ultimate role of nutrient supply to determine bloom maxima. Because  $\text{N}_2$ -fixing cyanobacteria were abundant in the St. Johns River, phosphorus availability likely was the primary limiting factor for bloom expansion (Schindler 1977). Residence time was longer, and TP lower, in Lake George than at Racy Point. Consequently, we hypothesized that TP limited (or colimited with TN) phytoplankton growth more frequently in Lake George than at Racy Point. This tendency was confirmed in nutrient enrichment and dilution assays (Paerl et al. 2005 [Federal Point is adjacent to Racy Point]).

The more frequent condition of phosphorus limitation in Lake George led to greater exploitation of the available supply as water age increased. The mean seasonal chl-a:TP ratio (a coarse index of phytoplankton utilization of phosphorus) for Lake George was closely related to water age in both the observed and simulated data, although simulated ratio values were displaced upwards because of overestimation of chl-a by the model (Figure 4–20). A similar relationship was present in observed data at Racy Point, although with the lower slope and  $R^2$  value that would be consistent with a smaller influence of P over maximum phytoplankton biomass. Simulated values for chl-a:TP at Racy Point were similar to observed values but did not show a significant relationship with water age (Figure 4–20).

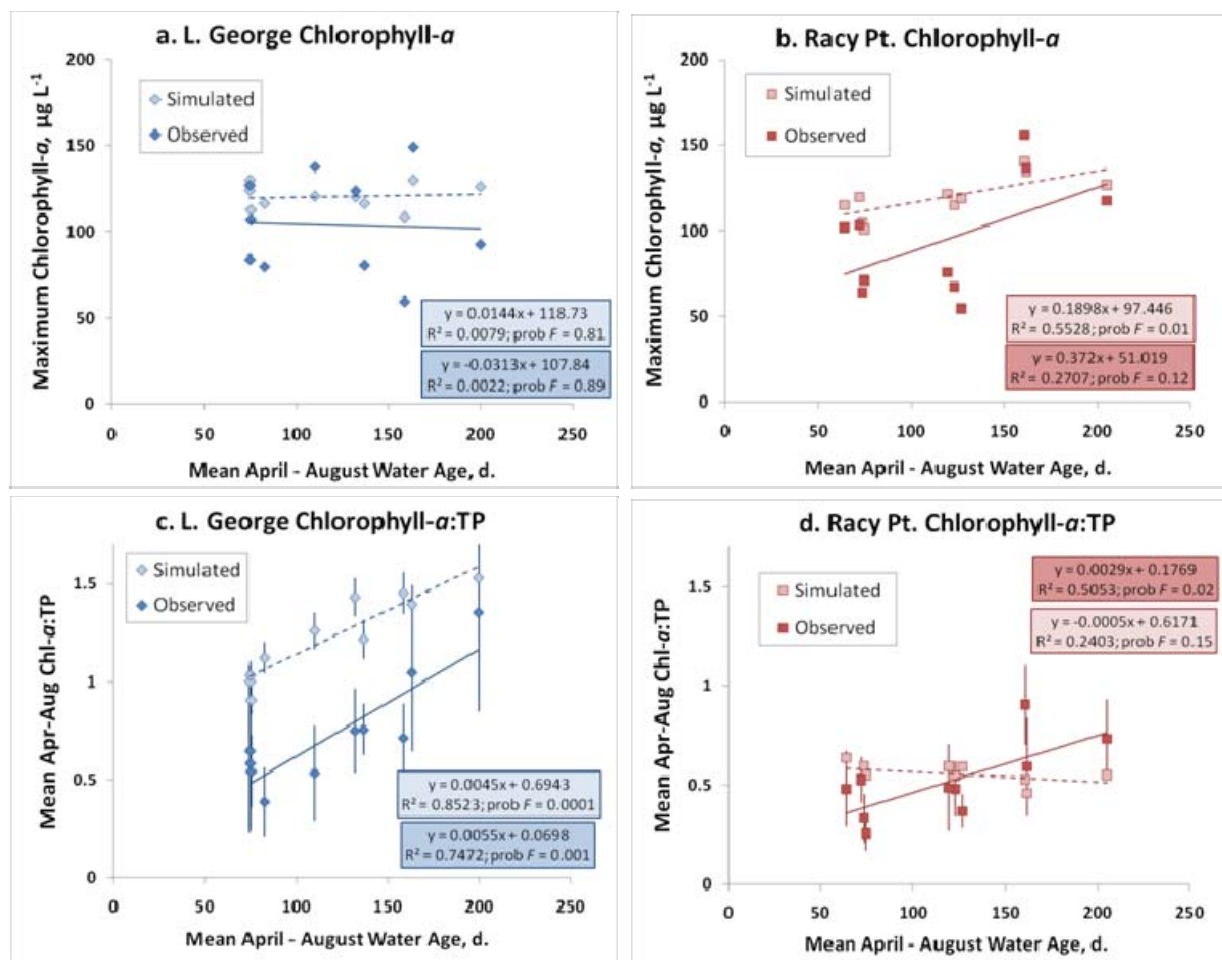


Figure 4–20. Relationships between mean bloom season water age and observed and simulated maximum chlorophyll-a (chl-a; panels a and c) and mean chl-a:TP ratio (panels b and d). Chl-a and total phosphorus (TP) were simulated with the mechanistic water quality model CE-QUAL-ICM for 1996 to 2005.

In consideration of the general similarity between observed data and the output of the CE-QUAL-ICM model with respect to the seasonality of blooms (see Figure 4–18), bloom maxima (see Figure 4–19), and bloom relationships to water age (see Figure 4–20), we concluded that the model was a useful adjunct to the empirical hydroecological models in predicting phytoplankton responses to changes in hydrology.

## 4.5 EVALUATION OF THE EFFECTS OF WATER WITHDRAWAL SCENARIOS ON ALGAL BLOOMS

### 4.5.1 RANKING OF MODEL SCENARIOS

Nineteen scenarios were modeled by the River Hydrodynamics Working Group (see Chapter 6). We ranked these scenarios for use in the plankton analyses (Table 4–12). Scenario Base1995NN was the scenario most representative of recent conditions and was the baseline for assessment of effects on algal blooms. Many of the remaining scenarios were included for test purposes or for

comparison and do not realistically represent conditions likely to be present when withdrawals could occur. We omitted six scenarios from further consideration: four had no water withdrawals so could have no withdrawal effects, and two paired projected 2030 sea level with 1995 land use, a condition that represents neither past or likely future conditions. We prioritized the remaining 12 scenarios for assessment according to the strength of their effects on water age. We compared cumulative distribution functions (CDFs) for daily mean water age at Racy Point for the baseline scenario Base1995NN and each of the other model scenarios (see Chapter 6. River Hydrodynamics Results for an example of CDFs). Two withdrawal scenarios showed no increase in water age and were omitted. We ranked the remaining 10 scenarios for assessment starting with the greatest increase in water age as the highest rank. Not unexpectedly, the top-ranked three scenarios all included the maximum modeled water withdrawal of 262 mgd.

Table 4–12 Summary of water withdrawal scenarios and the rank for assessment by the Plankton Working Group. We prioritized scenarios based on increase in water age at Racy Point from the baseline scenario (Base1995NN). Some scenarios were omitted from ranking; see text for details. Mgd= million gallons per day.

Scenario*	Variable				Category	Rank
	Land Use	USJRB Project Implemented	Water Withdrawal (mgd)	Sea Level†		
Base1995NN	1995	No	0	1995	Hindcast	Baseline
FwOR1995NN	1995	No	262	1995	Hindcast	1
FwOR1995PN	1995	Yes	262	1995	Unrealistic	2
FwOR2030PS	2030	Yes	262	2030	Forecast	3
FwOR2030PN	2030	Yes	262	1995	Unrealistic	4
Full1995NN	1995	No	155	1995	Hindcast	5
Full1995PN	1995	Yes	155	1995	Unrealistic	6
Full2030PS	2030	Yes	155	2030	Forecast	7
Half1995NN	1995	No	77.5	1995	Hindcast	8
Half1995PN	1995	Yes	77.5	1995	Unrealistic	9
Half2030PS	2030	Yes	77.5	2030	Forecast	10
Full2030PN	2030	Yes	155	1995	Unrealistic	Omitted – No Increase in Water Age
Half2030PN	2030	Yes	77.5	1995	Unrealistic	
Full1995PS	1995	Yes	155	2030	Unrealistic	Omitted – 2030 Sea Level
FwOR1995PS	1995	Yes	262	2030	Unrealistic	
Base1995PN	1995	Yes	0	1995	Unrealistic	Omitted – No Withdrawal
Base1995PS	1995	Yes	0	2030	Unrealistic	
Base2030PN	2030	Yes	0	1995	Unrealistic	
Base2030PS	2030	Yes	0	2030	Forecast	

\*See Chapter 6, River Hydrodynamics Results, for descriptions of scenarios and categories and for details on EFDC hydrodynamic modeling.

†1995 Sea Level = actual sea levels 1995 to 2005; 2030 Sea Level = 14 cm added to sea level values for 1995 to 2005.

#### 4.5.2 EFFECTS OF WATER WITHDRAWAL SCENARIOS

We assessed potential effects of water withdrawals in segments 2, 3, 4, 6, and 8 of the St. Johns River (see Table 4–7) using the empirical regression models to predict algal bloom variables from water age variables (see Table 4–10). We used water age output from the EFDC model results (see Chapter 6. River Hydrodynamics Results) for the baseline scenario and each withdrawal scenario, and compared the predicted bloom variables. By focusing on the difference



between predicted results for the baseline scenario and withdrawal scenarios, we minimized the effects of modeling bias because the bias would be present in both sets of results. Below are examples of prediction results for one scenario using two regression models, the linear model used to predict maximum annual chl-a in segment 6 and the logistic model used to predict annual probability of a long-duration ( $> 50$  d) bloom event in segments 3 and 4.

We used withdrawal scenario FwOR1995NN in these examples. This scenario generated the largest difference in water age from the baseline scenario (see Table 4–12). FwOR1995NN was a hindcast scenario that applied the highest future water withdrawal (262 mgd) to past (1995) land use and river conditions. It reflects the potential effects of water withdrawals alone and does not consider the effects of other changes likely to occur before additional withdrawals, such as changes in land use and completion of the upper basin project.

For withdrawal scenario FwOR1995NN, maximum annual bloom chl-a predicted by linear regression for Lakes Harney, Jesup, and Monroe (segment 6) did not differ greatly from the baseline scenario (Base1995NN) (Figure 4–21). The higher chl-a predicted for both baseline and withdrawal scenarios in Lakes Harney and Monroe around 1996 to 1997 and 1999 to 2001 corresponded to drier periods with increased water age (Figure 4–6). The chronic algal blooms in hypereutrophic Lake Jesup made patterns such as these more difficult to discern. The lower threshold ( $50 \mu\text{g chl-a L}^{-1}$ ) for freshwater algal bloom magnitude was exceeded every year in Lake Jesup and every year but one in Lake Monroe, under both the baseline and withdrawal scenarios (Figure 4–21). The modeled effect of water withdrawal was a slight decline in magnitude of freshwater algal blooms compared to baseline in offline Lake Jesup (Figure 4–21). This effect likely stemmed from modeled withdrawals at Lake Jesup that increased the inflow of “younger” water from the St. Johns River. In reality, this water also would contain lower concentrations of chl-a.

In Lake George (segment 4) and at Racy Point (segment 3), logistic regression predicted that the probabilities of long-duration blooms ( $> 50$  d) were also similar to those predicted for the baseline scenario (Base1995NN) (Figure 4–22).



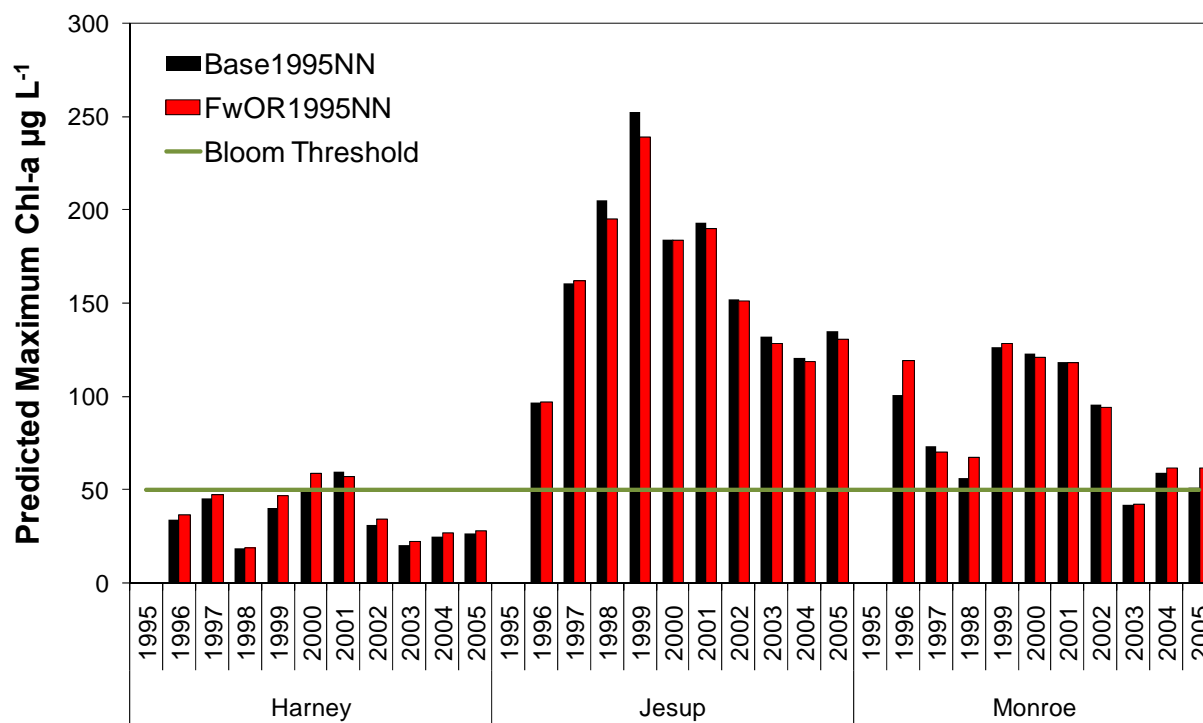


Figure 4–21. Maximum annual bloom chl-a predicted for Lakes Harney, Jessup, and Monroe (segment 6) for baseline scenario Base1995NN and withdrawal scenario FwOR1995NN. Prediction was by linear regression model from water age variables. Because a variable for antecedent period A (see Figure 3–9) was used in the regression, no prediction could be made for 1995. FwOR1995NN ranked first by difference in water age from the baseline (see Table 4–12). Horizontal line indicates lower threshold ( $50 \mu\text{g chl-a L}^{-1}$ ) for magnitude of freshwater algal bloom.

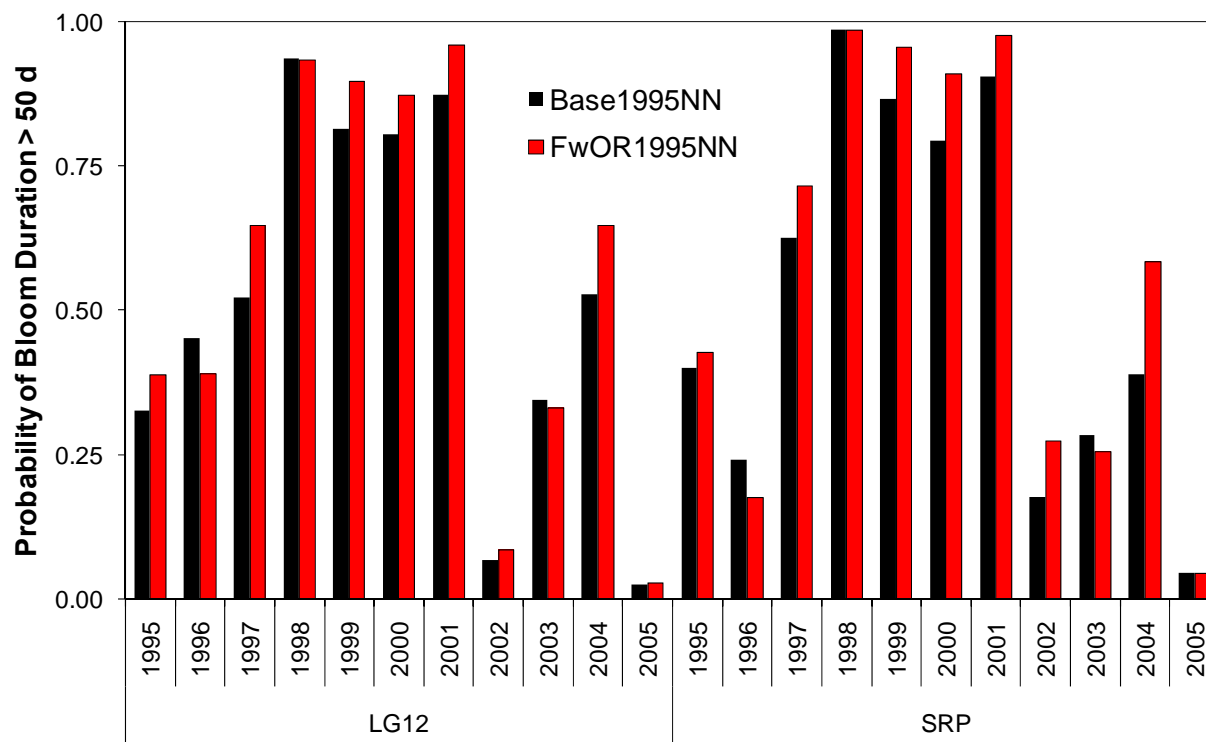


Figure 4–22. Probability of long-duration (> 50 d) algal blooms predicted for Lake George (LG12, segment 4) and Racy Point (SRP, segment 3) for baseline scenario Base1995NN and withdrawal scenario FwOR1995NN. Prediction was by logistic regression model from water age variables. FwOR1995NN ranked first by difference in water age from the baseline (Table 4–20).

We made analogous calculations for each combination of algal bloom metric, river segment, and withdrawal scenario that we assessed. In addition to top-ranked scenario FwOR1995NN, we assessed withdrawal scenarios ranked second and third by difference in water age from the baseline (see Table 4–12). The second ranked scenario FwOR1995PN also was a hindcast scenario that applied the maximum water withdrawal to 1995 land use, an unrealistic condition for water supply withdrawals. The third ranked scenario, FwOR2030PS, combines conditions that may be extant when water withdrawals occur (2030 land use, completed upper basin project, and higher sea level).

For the magnitude of freshwater algal blooms metric (e.g., Figure 4–21) in segments 2, 3, 4, 6, and 8, and the duration of freshwater algal blooms metric in segments 2 and 6, we summarized the effect of the withdrawal both as the percent change in the median annual value (withdrawal minus baseline) of the metric and as the change in annual frequency that thresholds were exceeded (see Section 3). Duration of freshwater algal blooms in segments 3 and 4 (e.g., Figure 4–22) was predicted by logistic regression where the predicted value was the probability of a bloom duration exceeding the 50-d threshold. Here, the effect of withdrawal could be summarized only as the change in annual frequency of exceedence.

The marine algal blooms metric (dinoflagellate biovolume) and the change ( $\Delta$ ) in N load metric also were quantified as the change in annual frequency that thresholds were exceeded. In the case of  $\Delta$  N load, this approach was necessary because the metric was based on the additional N added through  $N_2$ -fixation (withdrawal minus baseline) rather than on the individual values. In the case of marine algal blooms, we felt that the change in annual frequency of exceedence was a better measure of effect than the percent change in predicted biovolumes. Preexisting impairment was low for this metric (Table 4–9), so a worsening effect of a withdrawal would be measureable as an increase in exceedence of the threshold. This situation contrasted with other metrics where pre-existing impairment was high in one or more segments.

The predicted effects of water withdrawal scenarios FwOR1995NN, FwOR1995PN, and FwOR2030PS are summarized by river segment and algal bloom metric in Table 4–13, Table 4–14, and Table 4–15. In keeping with the fact that scenario FwOR1995NN showed the greatest increase in water age compared with the baseline (see Table 4–12), this scenario also showed the greatest effects on algal bloom metrics (see Table 4–13). Although effects on algal blooms were greatest with FwOR1995NN, they still were modest. For Lake Harney, the magnitude of blooms increased by 7.6%, and the fractional increase in the annual frequency of blooms  $> 50 \mu\text{g chl-a L}^{-1}$  was 0.1. Changes from the baseline for all other locations and metrics did not exceed + 3.6% for bloom magnitude or + 0.1 for annual frequency (see Table 4–13).

Table 4–13. Predicted effects of water withdrawal scenario FwOR1995NN on algal bloom metrics. This scenario ranked first for analysis based on water age (see Table 4–12) but was an unrealistic scenario for water supply withdrawal. Effects were quantified as the percent change in annual median value (%) and/or the change in annual frequency that thresholds were exceeded compared with the baseline (see Table 4–6 for thresholds).

River Segment <sup>*</sup>	Location	Algal Bloom Metric				
		Marine Algal Blooms	Change in N Load	Magnitude of Freshwater Algal Blooms	Magnitude of Freshwater Algal Blooms (DO)	Duration of Freshwater Algal Blooms
2	Mandarin Point	± 0.0				
	Doctors Lake	+ 0.09		+ 3.1%	+ 3.1%	+ 3.6%
3	Racy Point		+ 0.1	- 0.1	± 0.0	± 0.0
				+ 1.0%	+1.0 %	+ 0.06
4	Lake George		+ 0.1	+ 0.1	± 0.0	
				- 1.8%	- 1.8%	+ 0.04
6	Lake Monroe			± 0.0	± 0.0	- 2.9%
				+ 0.6%		± 0.0
	Lake Jesup			- 1.7%		- 2.4%
				± 0.0		- 0.1
	Lake Harney			+ 7.6%		
8	Lake Poinsett			+ 0.1		
				+ 1.3%		
				± 0.0		

\* Segments 1, 5, 7, and 9 were not assessed by the Plankton Working Group.

Table 4–14. Predicted effects of water withdrawal scenario FwOR1995PN on algal bloom metrics. This scenario ranked second for analysis based on water age (see Table 4–12) but was an unrealistic scenario for water supply withdrawal. Effects were quantified as the percent change in annual median value (%) and/or the change in annual frequency that thresholds were exceeded compared with the baseline (see Table 4–6 for thresholds).

River Segment*	Location	Algal Bloom Metric				
		Marine Algal Blooms	Change in N Load	Magnitude of Freshwater Algal Blooms	Magnitude of Freshwater Algal Blooms (DO)	Duration of Freshwater Algal Blooms
2	Mandarin Point	± 0.0				
	Doctors Lake	+ 0.09		+ 3.1% - 0.1	+ 3.1% ± 0.0	+ 3.6% ± 0.0
3	Racy Point		+ 0.1	+ 1.0% + 0.1	+1.0 % ± 0.0	+ 0.06
4	Lake George		+ 0.1	- 1.8% ± 0.0	- 1.8% ± 0.0	+ 0.04
6	Lake Monroe			+ 0.6% ± 0.0		- 2.9% ± 0.0
	Lake Jesup			- 1.7% ± 0.0		- 2.4% - 0.1
	Lake Harney			+ 7.6% + 0.1		
				- 0.3% ± 0.0		
8	Lake Poinsett					

\* Segments 1, 5, 7, and 9 were not assessed by the Plankton Working Group.

Table 4–15. Predicted effects of water withdrawal scenario FwOR2030PS on algal bloom metrics. This forecast scenario ranked third for analysis based on water age (see Table 4–12) and represented the largest modeled 2030 water withdrawal (262 mgd). Effects were quantified as the percent change in annual median value (%) and/or the change in annual frequency that thresholds were exceeded compared with the baseline (see Table 4–6 for thresholds).

River Segment*	Location	Algal Bloom Metric				
		Marine Algal Blooms	Change in N Load	Magnitude of Freshwater Algal Blooms	Magnitude of Freshwater Algal Blooms (DO)	Duration of Freshwater Algal Blooms
2	Mandarin Point	± 0.0				
	Doctors Lake	± 0.0		- 0.4% ± 0.0	- 0.4% ± 0.0	- 2.8% + 0.1
3	Racy Point		± 0.0	- 3.2% - 0.1	- 3.2% - 0.1	+ 0.04
4	Lake George		± 0.0	- 7.6% ± 0.0	- 7.6% - 0.1	+ 0.02
6	Lake Monroe			- 3.0% - 0.1		- 12.4% - 0.3
	Lake Jesup			+ 3.7% ± 0.0		- 0.4% ± 0.0
	Lake Harney			+ 3.0% - 0.1		
				+ 1.6% ± 0.0		
8	Lake Poinsett					

\*Segments 1, 5, 7, and 9 were not assessed by the Plankton Working Group.

For scenario FwOR2030PS, the most extreme realistic forecast scenario based on water age (Table 4–12), predicted effects were small and in many cases negative, indicating a reduction in algal bloom metrics compared to baseline (see Table 4–15). These reductions likely reflected the increased discharge in the river predicted under 2030 land use conditions (see Chapter 3. Watershed Hydrology).

We used the CE-QUAL-ICM model as a secondary approach to predict changes in algal bloom metrics due to changes in hydrology. This model is mechanistic and independent from the empirical regressions that relate algal bloom metrics to water age. We calibrated the mechanistic

model with data from Racy Point (segment 3) and Lake George (segment 4) and used the calibrated model to predict freshwater bloom magnitude (annual maximum chl-a) for those two segments. We evaluated the predicted deviations from baseline scenario Base1995NN for the same three withdrawal scenarios that were evaluated by the regression models: FwOR1995NN, FwOR1995PN, and FwOR2030PS.

Predicted chl-a maxima were almost identical for the baseline scenario and withdrawal scenarios for both Lake George and Racy Point (Figure 4–23). For the withdrawal scenarios, CE-QUAL-ICM predicted median changes in bloom maximum chl-a from +1.7% to -0.1% (see Table 4–16). These independent predictions of small changes provided support for the similarly small changes predicted for segments 3 and 4 using the empirical regression model (see Table 4–13, Table 4–14, and Table 4–15). Similar to the comparison between observed and simulated data for model calibration (see Figure 4–20), predicted bloom maximum chl-a was not related to seasonal mean water age in Lake George, but these two variables were significantly related at Racy Point (Figure 4–23).

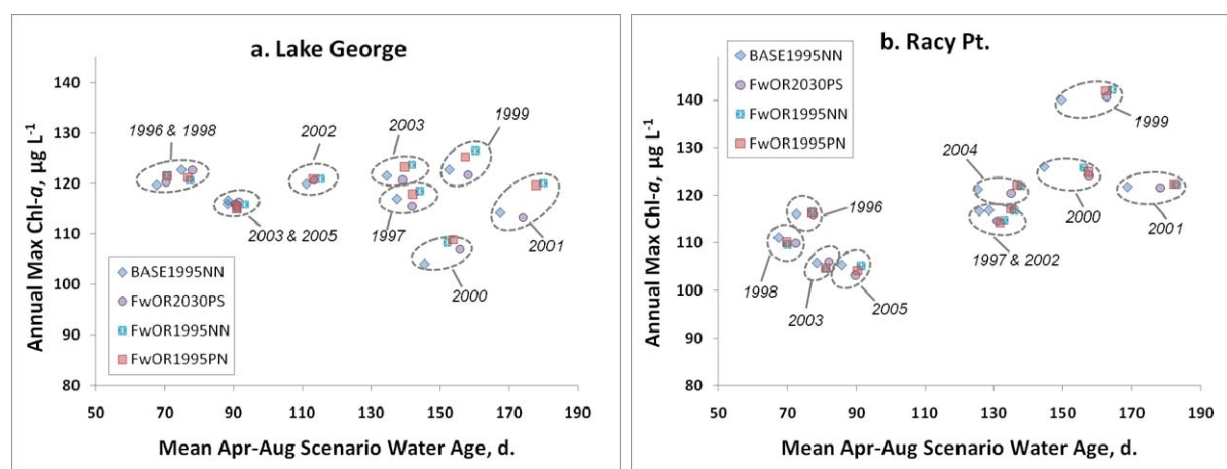


Figure 4–23. Relationships between maximum bloom chlorophyll-a (chl-a) and mean April to August water age for CE-QUAL-ICM simulations of baseline (Base1995NN) and withdrawal scenarios

Table 4–16. Effects of water withdrawal scenarios FwOR1995NN, FwOR1995PN, and FwOR2030PS on maximum algal bloom chl-a predicted using mechanistic model CE-QUAL-ICM. Effects were quantified as the percent change in annual median values compared with the baseline. Locations in segments 3 and 4 were assessed.

Segment	FwOR1995NN	FwOR1995PN	FwOR2030PS
Racy Point Segment 3	- 0.1%	±0.0%	- 0.1 %
Lake George Segment 4	+ 1.7%	+ 1.7%	±0.0%

In those cases where we quantified effects both as the percent change in median response and the change in annual frequency of exceedence (see Table 4–13, Table 4–14, and Table 4–15), we gave precedence to the percent change values in assessment of effects. We adopted this approach to solve two problems in assessment caused by preexisting impairment (see Table 4–9). First, preexisting impairment meant that thresholds could be exceeded almost every year even for modeled baseline conditions (see Figure 4–21, Lakes Jesup and Monroe). In that case, reliance on frequency of exceedence would not detect a worsening effect of water withdrawal even where one existed. Second, preexisting impairment could result in algal bloom metrics poised right at the threshold for modeled baseline conditions (see Figure 4–21, Lake Harney). In that case, even slight increases due to withdrawal could increase the frequency of exceedence and exaggerate the apparent effects. Use of the percent change in median response, where possible, minimized these problems.

Potential effects of water withdrawals in segment 8 (Lake Poinsett) were assessed with a slightly different methodology from that applied for other segments (Appendix 8.B). In summary, residence time (rather than water age) was the hydrologic variable, and the algal bloom metric was maximum annual chl-a. A three-variable linear regression model was used to predict maximum chl-a from residence time (adjusted  $R^2 = 0.85$ ) for the baseline and withdrawal scenarios. Both the percent change in median response and the change in annual frequency of exceedence fell within the negligible effects category for Lake Poinsett (Appendix 8.B).

A possible indirect effect of water withdrawals is increased release of nutrients from riparian wetlands if water withdrawals extend the time that wetland soils are exposed and oxidized between flood events. Greater nutrient loading likely would exacerbate algal blooms (Figure 4–5). Most reaches of the St. Johns River have nutrient TMDLs or have been listed as impaired for nutrients by the FDEP (Figure 3–2). Furthermore, with the algal bloom metrics developed here, we found major overall impairment in all segments of the St. Johns River that we evaluated but for segment 8 (Table 4–9).

The increased loading of P and N from extended drawdowns under the withdrawal scenario (Full1995NN) with the most extreme effects on water level in the upper St. Johns River was estimated to be  $0.29 \times 10^6$  g P yr<sup>-1</sup> and  $2.1 \times 10^6$  g N yr<sup>-1</sup> in segment 8 (median values; Chapter 7. Biogeochemistry). We doubled these values, i.e.  $0.58 \times 10^6$  g P yr<sup>-1</sup> and  $4.2 \times 10^6$  g N yr<sup>-1</sup>, to make upper-bound estimates for increased loading to segments 7 and 8 combined (L. Keenan, pers. comm.).

When this increased loading was expressed as a concentration change in Lake Poinsett (segment 8), the equivalent increase in median concentrations was negligible: 2% for TP and 0.6% for TN (Appendix 8.B). We also compared the upper-bound estimate for increased loading to combined segments 7 and 8 to loading at the inflow to Lake Harney (segment 6). With average net fluxes at Lake Harney of  $178 \times 10^6$  g P yr<sup>-1</sup> and  $2731 \times 10^6$  g N yr<sup>-1</sup> (Appendix 8.E), the increased loads were negligible (0.3% for P and 0.2% for N).

### 4.5.3 LEVELS OF EFFECTS

The WSIS working groups developed a common terminology and definitions of levels of environmental effects that ranged from extreme to negligible (see Chapter 2. Comprehensive Integrated Assessment). We considered these definitions and used professional judgment to



characterize effects of water withdrawals measured either as the change in median response of an algal bloom metric or as the change in frequency of years that the threshold for a metric was exceeded (Table 4–17). As explained above, we gave precedence to the percent change in median over the change in annual frequency for algal bloom metrics where both were measured. Our characterization considered the incremental effects of water withdrawals in addition to the preexisting impairment that we found in most segments of the St. Johns River (see Table 4–9).

For example, a moderate effect was defined as an alteration of ecosystem components that does not significantly reduce natural resource values (Chapter 2). We judged that a moderate effect should be measureable in a monitoring program even with preexisting impairment, and a 20% worsening in an algal bloom metric met that definition. Similarly, we judged a moderate effect to be an increase in annual frequency of exceedence of 0.18 (Table 4–17). This increase likely would raise a minor preexisting impairment to moderate and a moderate preexisting impairment to major. No changes in algal bloom metrics of this size were found for water withdrawal scenarios.

A minor effect was defined as an effect that does not significantly alter any ecosystem component (Chapter 2). We assigned a 10% to 20% worsening in an algal bloom metric or an increase in annual frequency of exceedence by 0.09 to the minor effect level (Table 4–17). Resolution of changes of this size in a monitoring program likely would be difficult.

Negligible effects were less than 10% increase in median response of the algal bloom metric or an increase in annual frequency  $\leq 0.09$ . We judged that changes of this size would not appreciably alter any ecosystem component.

Because the predicted effects of water withdrawals on plankton communities were modest, we did not find it necessary to define major or extreme levels of effects (Table 4–17).

Table 4–17. Levels used to characterize effects of water withdrawals based on the percent change in magnitude (where available) of a bloom metric or on the change in frequency of years that the bloom metric threshold was exceeded (see Table 4–13, Table 4–14, and Table 4–15). We did not define major or extreme effects.

Change in Magnitude		Or	Change in Frequency	
Level	Effect		Level	Effect
Extreme	Not defined		Extreme	Not defined
Major	Not defined		Major	Not defined
Moderate	> 20%		Moderate	+ 2 yr ( $\geq + 0.18$ )
Minor	10% to 20%		Minor	+ 1 yr ( $\geq + 0.09$ )
Negligible	< 10%		Negligible	+ 0 yr ( $< + 0.09$ )

#### 4.5.4 UNCERTAINTY IN ASSESSMENT OF EFFECTS

The WSIS working groups developed a common set of criteria to evaluate the level of uncertainty associated with potential environmental effects (see Chapter 2. Comprehensive Integrated Assessment). Uncertainty levels for effects could range from very low (very strong quantitative evidence) to medium (quantitative or strong qualitative evidence) to very high uncertainty (weak qualitative evidence). Three types of evidence were considered: predictive model evidence, supporting evidence, and understanding of mechanisms.

Our assessments of each of the three types of evidence for effects follow below. This approach to evaluate uncertainty first rated the strength of the evidence, and then used the strength assessment to express uncertainty. Because uncertainty was expressed with five categories (Chapter 2) we similarly expressed the strength of evidence with five ordinal categories (1 = strong and 5 = weak). Table 4–18 summarizes our ratings of the strength of evidence plus the final uncertainty scores for each river segment evaluated and each algal bloom metric used.

Predictive model evidence (PM) included both the EFDC hydrodynamic model used to describe the water withdrawal scenarios and the ecohydrological models used to relate hydrologic variables to ecological variables. We judged the ecohydrological models to be strong (PM = 1) because all regression models showed good fit (see Table 4–10, Appendix 8.C). Strength ratings for the EFDC hydrodynamic model (PM = 1 or 2, depending on segment) were taken from uncertainty ratings for water age in Chapter 6. River Hydrodynamics Results. The ordinal uncertainty scale used in Chapter 6 was inverted to strength to be equivalent to ratings of hydroecological models, i.e. very low uncertainty = strength 1, low uncertainty = strength 2, etc. The exceptions to the base uncertainty ratings noted in Chapter 6 did not apply at locations evaluated by the Plankton Working Group. We include both of the ratings of strength (i.e., EFDC hydrodynamic model and hydroecological models) for the predictive model evidence (PM) in Table 4–18. In evaluation of uncertainty, we chose the lesser strength (higher number) of the two values to be the overall rating of the PM.

We rated the supporting evidence (SE) as strong (SE = 1) for relationships between discharge or retention time and characteristics of phytoplankton blooms. This evidence stemmed both from analyses reported here (see sections 3.1.3 and 4.1) and from published literature (Appendix 8.A, Philips et al. 2007).

Similarly, we judged understanding of mechanisms (UM) to be strong (UM = 1) whereby longer retention times in the St. Johns River tend to promote blooms and shorter retention times tend to limit blooms (see section 4.1; Appendix 8.A). Mechanisms whereby longer retention times promote blooms can include greater utilization of nutrient pools by algae, higher N<sub>2</sub>-fixation, greater production due to a more-transparent water column (lower water color), and increased stratification that favors bloom-forming cyanobacteria.

We concluded that the basic strength of hydroecological model components was 1|1|1 for PM|SE|UM. However, we also found the following exceptions:

- The marine algal blooms metric was rated SE=3, because current taxa were not toxin-producing, and UM=2, because water age may have functioned in part as a surrogate for salinity.

- The change ( $\Delta$ ) in N load metric was rated PM=2 and SE=2, because prediction was based on a constant areal factor, and the N mass balance may under-predict N<sub>2</sub>-fixation.
- The magnitude of freshwater algal blooms metrics were rated SE=2 and UM=2 in Doctors Lake, because blooms were not always cyanobacteria, and water age may have been a surrogate for salinity. These metrics were rated UM=2 in segment 6, because differences in water age may have been confounded with differences in nutrient concentrations among lakes.
- The duration of freshwater algal blooms metric was rated UM=2 in segment 6 because differences in water age may have been confounded with differences in nutrient concentrations among lakes.

Assignments of level of uncertainty based on the strength of components followed the definitions in Chapter 2. Comprehensive Integrated Assessment. Because SE and UM were always present (and usually were strong), we gave priority to the strength of the PM to determine level of uncertainty.

We used the uncertainty associated with the most highly weighted effects (see section 4.5.5) to determine the uncertainty for the overall level of effect in each river segment (Table 4-18).

Table 4–18. Uncertainty in assessment of effects of water withdrawals for each bloom metric where applied in each river segment. Ordinal scores rate the strength of three types of evidence for effects as 1 (strong) to 5 (weak) (see Chapter 2. Comprehensive Integrated Assessment). These scores are shown as follows: EFDC model output — Predictive Model | Supporting Evidence | Understanding of Mechanisms. The corresponding uncertainty codes (\* very low to \*\*\*\*\* very high) were based on the strength of evidence scores and followed definitions outlined in Chapter 2. See text for further details.

River Segment <sup>§</sup>	Location	Algal Bloom Metric				
		Marine Algal Blooms	Change in N Load	Magnitude of Freshwater Algal Blooms	Magnitude of Freshwater Algal Blooms (DO) <sup>†</sup>	Duration of Freshwater Algal Blooms
2	Mandarin Point	2-1 3 2 ** <sup>‡</sup>				
	Doctors Lake	2-1 3 2 **		2-1 2 2 **		2-1 1 1 **
3	Racy Point		2-2 2 1 **	2-1 1 1 **		2-1 1 1 **
4	Lake George		1-2 2 1 **	1-1 1 1 *		1-1 1 1 *
6	Lake Monroe			1-1 1 2 *		1-1 1 2 *
	Lake Jesup			1-1 1 2 *		1-1 1 2 *
	Lake Harney			1-1 1 2 *		
8	Lake Poinsett			2-1 1 1 **		

<sup>§</sup> Segments 1, 5, 7, and 9 were not assessed by the Plankton Working Group.

<sup>†</sup> Uncertainty in the Magnitude of Freshwater Algal Blooms (DO) is rated the same as the Magnitude of Freshwater Algal Blooms metric.

<sup>‡</sup>Uncertainty: \* = very low; \*\* = low; \*\*\* = medium; \*\*\*\* = high; \*\*\*\*\* = very high

#### 4.5.5 SUMMARY OF EFFECTS

To determine the overall level of effect for each river segment, we gave two metrics, magnitude of freshwater algal blooms (lower threshold 50 µg chl-a L<sup>-1</sup>) and duration of freshwater algal blooms, greater weight than remaining metrics. These two metrics dealt with direct effects of algal blooms. In contrast, the marine algal blooms metric dealt with the potential risk for toxic dinoflagellate blooms where they did not occur, and the change in N load metric dealt with an

indirect effect of cyanobacterial blooms. Where multiple locations were assessed in a single segment of the river, we used the worst case location to determine the overall level of effect.

The predicted levels of effects for water withdrawal scenarios FwOR1995NN, FwOR1995PN, and FwOR2030PS are summarized by river segment and algal bloom metric in Table 4–19, Table 4–20, and Table 4–21. Levels of uncertainty are summarized as well. Predicted effect values that formed the basis for these levels are shown in Table 4–13, Table 4–14, and Table 4–15.

Scenarios FwOR1995NN (Table 4–19) and FwOR1995PN (Table 4–20) showed mixed negligible and minor effects. For both scenarios, overall effects were negligible in each segment that was assessed. Both FwOR1995NN and FwOR1995PN were based on 1995 land use and were unrealistic representations of future water withdrawal conditions (see Table 4–12).

Scenario FwOR2030PS was the most extreme (based on water withdrawal and water age) of the realistic forecast scenarios (see Table 4–12) and showed negligible effects for all algal bloom metrics in all segments (Table 4–21 ). We concluded that the remaining forecast scenarios with lesser effects on water age also would show negligible effects on algal bloom metrics.

Table 4–19. Predicted levels of effects of water withdrawal scenario FwOR1995NN on algal bloom metrics compared with the baseline (Base1995NN). This scenario ranked first for analysis based on water age (see Table 4–12) but was an unrealistic scenario for water supply withdrawal. Levels of uncertainty also are shown.

River Segment <sup>§</sup>	Location	Algal Bloom Metric					Overall
		Marine Algal Blooms	Change in N Load	Magnitude of Freshwater Algal Blooms	Magnitude of Freshwater Algal Blooms (DO)	Duration of Freshwater Algal Blooms	
2	Mandarin Point	**† Negligible					**
	Doctors Lake	** Minor		** Negligible	** Negligible	** Negligible	Negligible
3	Racy Point		** Minor	** Negligible	** Negligible	** Negligible	** Negligible
			** Minor	* Negligible	* Negligible	* Negligible	* Negligible
4	Lake George						
6	Lake Monroe			* Negligible		* Negligible	* Negligible
	Lake Jesup			* Negligible		* Negligible	
	Lake Harney			* Negligible			
8	Lake Poinsett			** Negligible			** Negligible

<sup>§</sup> Segments 1, 5, 7, and 9 were not assessed by the Plankton Working Group.

<sup>†</sup> Uncertainty: \* = very low; \*\* = low; \*\*\* = medium; \*\*\*\* = high; \*\*\*\*\* = very high

Table 4–20. Predicted levels of effects of water withdrawal scenario FwOR1995PN on algal bloom metrics compared with the baseline (Base1995NN). This scenario ranked second for analysis based on water age (see Table 4–12) but was an unrealistic scenario for water supply withdrawal. Levels of uncertainty also are shown.

River Segment <sup>§</sup>	Location	Algal Bloom Metric					Overall
		Marine Algal Blooms	Change in N Load	Magnitude of Freshwater Algal Blooms	Magnitude of Freshwater Algal Blooms (DO)	Duration of Freshwater Algal Blooms	
2	Mandarin Point	<b>**†</b> Negligible					<b>**</b>
	Doctors Lake	<b>**</b> Minor		<b>**</b> Negligible	<b>**</b> Negligible	<b>**</b> Negligible	Negligible
3	Racy Point		<b>**</b> Minor	<b>**</b> Negligible	<b>**</b> Negligible	<b>**</b> Negligible	<b>**</b> Negligible
4	Lake George		<b>**</b> Minor	<b>*</b> Negligible	<b>*</b> Negligible	<b>*</b> Negligible	<b>*</b> Negligible
6	Lake Monroe			<b>*</b> Negligible		<b>*</b> Negligible	<b>*</b> Negligible
	Lake Jesup			<b>*</b> Negligible		<b>*</b> Negligible	<b>*</b> Negligible
	Lake Harney			<b>*</b> Negligible			<b>*</b> Negligible
8	Lake Poinsett			<b>**</b> Negligible			<b>**</b> Negligible

<sup>§</sup> Segments 1, 5, 7, and 9 were not assessed by the Plankton Working Group.

<sup>†</sup> Uncertainty: \* = very low; \*\* = low; \*\*\* = medium; \*\*\*\* = high; \*\*\*\*\* = very high

Table 4–21. Predicted levels of effects of water withdrawal scenario FwOR2030PS on algal bloom metrics compared with the baseline (Base1995NN). This scenario ranked third for analysis based on water age (see Table 4–12) and represented the largest modeled 2030 water withdrawal (262 mgd). Levels of uncertainty also are shown.

River Segment <sup>§</sup>	Location	Algal Bloom Metric					Overall
		Marine Algal Blooms	Change in N Load	Magnitude of Freshwater Algal Blooms	Magnitude of Freshwater Algal Blooms (DO)	Duration of Freshwater Algal Blooms	
2	Mandarin Point	**†					**
	Doctors Lake	Negligible **		**	**	**	Negligible
3	Racy Point	Negligible	**	Negligible **	Negligible **	Negligible **	**
			Negligible	Negligible	Negligible	Negligible	Negligible
4	Lake George		**	*	*	*	*
			Negligible	Negligible	Negligible	Negligible	Negligible
6	Lake Monroe			*		*	*
				Negligible		Negligible	
	Lake Jesup			*		*	
				Negligible		Negligible	
8	Lake Poinsett			*			**
				Negligible			
				**			**
				Negligible			Negligible

<sup>§</sup> Segments 1, 5, 7, and 9 were not assessed by the Plankton Working Group.

<sup>†</sup> Uncertainty: \* = very low; \*\* = low; \*\*\* = medium; \*\*\*\* = high; \*\*\*\*\* = very high



## **5 ACKNOWLEDGMENTS**

The members and staff of the National Research Council Committee to Review the St. Johns River Water Supply Impact Study (see Chapter 2. Comprehensive Integrated Assessment) provided important and beneficial review and comments at several stages in this work. Alan Joyner (University of North Carolina at Chapel Hill, Institute of Marine Sciences) prepared Figure 2-1. Robert Godfrey (SJRWMD) prepared Figures 3-1, 3-2, and 3-8.

## 6 REFERENCES

- Abbott, J.P., L.J. Flewelling and J.H. Landsberg. 2009. Saxitoxin monitoring in three species of Florida puffer fish. *Harmful Algae* 8: 343-348.
- Benndorf, J. and M. Henning. 1989. Daphnia and toxic blooms of *Microcystis aeruginosa* in Bautzen Reservoir (GDR). *Internationale Revue Gesamten Hydrobiologie* 74:233-248.
- Breck, J. E. 1993. Hurry up and wait: growth of young bluegills in ponds and in simulations with an individual-based model. *Transactions of the American Fisheries Society* 122:467-480.
- Burnes, R. M., E. J. Phlips, M. F. Cichra and M. Lehmenseik. 2008. Zooplankton community structure in the Middle St. Johns River, Florida, 2003-2005. *Florida Scientist* 71:173-187.
- Cade, B.S. and B.R. Noon. 2003. A gentle introduction to quantile regression for ecologists. *Front. Ecol. Environ.* 1(8): 412–420.
- Carpenter, S. R. and J. F. Kitchell. 1993. *The Trophic Cascade in Lakes*. Cambridge University Press, Cambridge.
- Carter, J.L. and S.V. Fend. 2005. Setting limits: The development and use of factor-ceiling distributions for an urban assessment using macroinvertebrates. *American Fisheries Society Symposium* 47:179-191.
- Cerco, C. F. and T. Cole. 1995. User's Guide to the CE-QUAL-ICM Three-Dimensional Eutrophication Model. Tech. Rept. EL-95-15, March 1995. U.S. Army Corps of Engineers, Vicksburg, MS.
- Clesceri, L. S., A.E. Greenberg and A. D. Eaton. 1998. *Standard Methods for the Examination of Water and Wastewater*, (20th Edition). American Public Health Association, Washington, District of Columbia.
- Dockendorf, K. J. and M. S. Allen. 2005. Age-0 black crappie abundance and size in relation to zooplankton density, stock abundance, and water clarity in three Florida lakes. *Transactions of the American Fisheries Society* 134:172-183.
- FDEP. 2008. Integrated Water Quality Assessment for Florida: 2008 305(b) Report and 303(d) List Update. Florida Department of Environmental Protection, Division of Environmental Assessment and Restoration, Bureau of Watershed Management. Tallahassee, FL. 142 pp.
- Gao, X., 2009. TMDL Report: Nutrient and Dissolved Oxygen TMDLs for the Middle St. Johns River Segments between the Inlet of Lake Harney (2964) and the St. Johns River above the Wekiva River (2893C). Florida Department of Environmental Protection, Tallahassee, Florida.

- Gao, X., D. Gilbert, and W. Magley. 2006. Nutrient TMDL for Alachua Sink, WBID 2720A. Florida Department of Environmental Protection.  
(<http://www.dep.state.fl.us/WATER/tmdl/docs/tmdls/final/gp1/alachuasink-nutr-tmdl.pdf>)
- Garvey, J. E. and R. A. Stein. 1998. Competition between larval fishes in reservoirs: The role of relative timing of appearance. *Transactions of the American Fisheries Society* 127:1021-1039.
- Ghadouani, A., B. Pinel-Alloul, and E. E. Prepas. 2003. Effects of experimentally induced cyanobacterial blooms on crustacean zooplankton communities. *Freshwater Biology* 48:363–381.
- Hendrickson, J.C., E.F. Lowe, D. Dobberfuhl, P. Sucsy and D. Campbell. 2003. Characteristics of Accelerated Eutrophication In the Lower St. Johns River Estuary and Recommended Targets to Achieve Water Quality Goals for the Fulfillment of TMDL and PLRG Objectives. Draft Water Resources Department Technical Memorandum, St. Johns River Water Management District, Palatka, FL. 89 pp.
- Hendrickson, J., N. Trahan, E. Gordon and Y. Ouyang. 2007. Estimating the relevance of organic carbon, nitrogen, and phosphorus loads to a blackwater river estuary. *J. American Water Resources Assoc.* 43(1): 264-279.
- Hoxmeier, R. J. H., D. H. Wahl, M. L. Hooe, and C. L. Pierce. 2004. Growth and survival of larval walleyes in response to prey availability. *Transactions of the American Fisheries Society* 133:45-54.
- Karr, J. R. and E. W. Chu. 1999. *Restoring Life in Running Waters: Better Biological Monitoring*. Island Press, Washington D.C. USA.
- Leonard, J.A. and H. W. Paerl. 2005. Zooplankton community structure, microzooplankton grazing impact, and seston energy content in the St. Johns River system, FL as influenced by the toxic cyanobacterium *Cylindrospermopsis raciborskii*. *Hydrobiologia* 537:89–97.
- Limno-Tech, Inc. 2005. Studies in Support of Sediment and Eutrophication Modeling in the Lower St. Johns River, Florida. Final Report, Contract SD174AA, St. Johns River Water Management District, Palatka, FL.
- Messick, G.A. and J.D. Shields. 2000. Epizootiology of the parasitic dinoflagellate *Hematodinium* sp. In the American blue crab *Callinectes sapidus*. *Diseases of Aquatic Organisms* 43:139-152.
- Miller, S. J., M. A. Lee, and E. F. Lowe. 1998. The Upper St. Johns River Basin Project: Merging Flood Control with Aquatic Ecosystem Restoration and Preservation. *Trans. 63<sup>rd</sup> No. Am Wildl. And Natur. Resour. Conf.*: 156 – 170.
- NCDWQ. 2007. Surface Water Quality Standards. North Carolina Division of Water Quality [http://h2o.enr.state.nc.us/csu/documents/ncactable290807\\_000.pdf](http://h2o.enr.state.nc.us/csu/documents/ncactable290807_000.pdf)

- Osborne, J.A. 2002. Zooplankton populations dynamics in Lakes Jesup and Washington. Special Publication SJ-2002-SP7. St. Johns River Water Management District.
- Paerl, H. W. 2008. Nutrient and other environmental controls of harmful cyanobacterial blooms along the freshwater – marine continuum. In H. K. Hudnell, editor. *Cyanobacterial Harmful Algal Blooms. Advances in Experimental Medicine and Biology* 619. Springer Science + Business Media, New York.
- Paerl, H. W. and R. S. Fulton, III. 2006. Ecology of harmful cyanobacteria. Pages 95-109 in E. Granéli and J. T. Turner, editors. *Ecology of Harmful Algae*. Springer-Verlag, Berlin.
- Paerl, H. W., R. S. Fulton, III, P. H. Moisander, and J. Dyble. 2001a. Harmful freshwater algal blooms, with an emphasis on cyanobacteria. *TheScientificWorld* 1:76-113.
- Paerl, H. W., J. J. Joyner and M. F. Piehler. 2005. Relationships between nutrient conditions and phytoplankton community structure and function in the Lower St. Johns River. Final Report for Contract No. 99B170, St. Johns River Water Management District. Palatka, FL. 57 pp.
- Paerl, H.W., M. F. Piehler, J. Dyble, P. H. Moisander, J. Leonard, A. Waggener and W. W. Carmichael. 2001b. Phytoplankton and zooplankton in the St. John's River System: Factors affecting community structure and function. Final report to St. Johns River Water Management District, Contract SD 154 RA.
- Phlips, E.J. and M.F. Cichra. 2001. Biological monitoring of the Lower St. Johns River: Temporal and spatial trends in plankton November 1998 through October 1999. Annual Report to the St. Johns River Water Management District.
- Phlips, E. J. and M. Cichra 2002. Spatial and Temporal Patterns in the Plankton Community of the Lower St. Johns River. Final Report to St. Johns River Water Management District, Contract 97W165. Department of Fisheries and Aquatic Sciences, University of Florida, Gainesville, FL.
- Phlips, E.J., J. Hendrickson, E.L. Quinlana and M. Cichra. 2007. Meteorological influences on algal bloom potential in a nutrient-rich blackwater river. *Freshwater Biology* 52(11): 2141 - 2155.
- Piehler, M.F., J. Dyble, P.H. Moisander, A.D. Chapman, J. Hendrickson and H.W. Paerl. 2009. Interactions between nitrogen dynamics and the phytoplankton community in Lake George, Florida, USA. *Lake and Reservoir Management* 25:1-14.
- Sammons, S. M., P. W. Bettoli, and V. A. Greear. 2001. Early life history characteristics of age-0 white crappies in response to hydrology and zooplankton densities in Normandy Reservoir, Tennessee. *Transactions of the American Fisheries Society* 130:442-449.
- Scharf, F.S., F. Juanes, and M. Sutherland. 1998. Inferring ecological relationships from the edges of scatter diagrams: comparison of regression techniques. *Ecology* 79(2):448–460.

- Scheffer, M. 1998. Ecology of Shallow Lakes. Chapman and Hall, London.
- Schelske, C. L. and F. J. Aldridge. 1994. Unpublished studies on primary productivity and nutrient limitation in the lower St. Johns River. Contract #20569, St. Johns River Water Management District, Palatka, FL.
- Schindler, D. W. 1977. Evolution of phosphorus limitation in lakes: Natural mechanisms compensate for deficiencies of nitrogen and carbon in eutrophied lakes. *Science* 195:260-262.
- Shen, J. and H. V. Wang. 2007. Determining the age of water and long-term transport timescale of the Chesapeake Bay. *Estuarine, Coastal and Shelf Science* 74:585-598.
- Smayda, T. J. 2002. Adaptive ecology, growth strategies and the global expansion of dinoflagellates. *J. Oceanography* 58: 281-294.
- Sucsy, P. and J. Hendrickson. 2004. Calculation of Nutrient Reduction Goals for the Lower St. Johns River by Application of CE-QUAL-ICM, a Mechanistic Water Quality Model. Draft Water Resources Department Technical Memorandum. St. Johns River Water Management District, Palatka, FL. 91 pp.
- Tillman, D.H., C.F. Cerco, M.R. Noel, J.L. Martin and J. Hamrick. 2004. Three-Dimensional Eutrophication Model of the Lower St. Johns River, Florida. U.S. Army Corps of Engineers Engineer research and Development Center Environmental Laboratory Report #ERDC/EL TR-04-13.
- USEPA. 2008. Total Maximum Daily Load (TMDL) for Nutrients: Lower St. Johns River. United States Environmental Protection Agency, Region 4, Atlanta, GA. 38 pp.
- Utermöhl, H. 1958. Zur vervollkommnung der quantitativen phytoplankton-methodik. *Mitteilungen-Internationale Vereinigung für Theoretische und Angewandte Limnologie* 9:1-38.
- World Health Organization. 2003. Guidelines for Safe Recreational Water Environments. Volume 1: Coastal and Fresh Waters. Chapter 8: Algae and Cyanobacteria in Fresh Waters. Table 8.3. Geneva, Switzerland.

## **7 APPENDICES**

### **APPENDIX 8.A. RELATIONSHIPS OF WATER FLOWS WITH PLANKTON AND WATER QUALITY**

### **APPENDIX 8.B. EFFECT OF WATER WITHDRAWAL ON PHYTOPLANKTON BIOMASS IN LAKE POINSETT**

### **APPENDIX 8.C. EMPIRICAL REGRESSION MODELS AND GOODNESS-OF-FIT DIAGNOSTICS**

### **APPENDIX 8.D. CE-QUAL-ICM SETUP, CALIBRATION AND WITHDRAWAL SCENARIO RESULTS**

### **APPENDIX 8.E. NITROGEN AND PHOSPHORUS CONCENTRATIONS, FLUXES, AND LOADS IN THE ST. JOHNS RIVER**

### **APPENDIX 8.F. DOMINANT PHYTOPLANKTON TAXA AT LAKE GEORGE AND RACY POINT AND DINOFLAGELLATE TAXA IN THE LOWER ST. JOHNS RIVER**