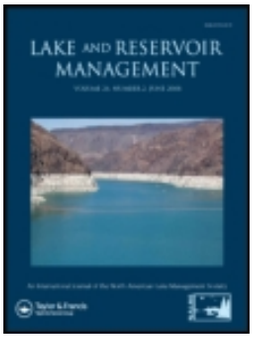

Lake and Reservoir Management

ISSN: 1040-2381 (Print) 2151-5530 (Online) Journal homepage: https://www.tandfonline.com/loi/ulrm20

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To cite this article: R. Thomas James , Victor J. Bierman Jr. , Michael J. Erickson & Scott C. Hinz (2005) The Lake Okeechobee Water Quality Model (LOWQM) Enhancements, Calibration, Validation and Analysis, Lake and Reservoir Management, 21:3, 231-260, DOI: 10.1080/07438140509354433

To link to this article: https://doi.org/10.1080/07438140509354433

Published online: 29 Jan 2009.

Lake and Reservoir Management 21(3):231-260, 2005

The Lake Okeechobee Water Quality Model (LOWQM) Enhancements, Calibration, Validation and Analysis

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Abstract

James, R.T., V.J. Bierman, Jr., M.J. Erickson and S.C. Hinz. 2005. The Lake Okeechobee Water Quality Model (LOWQM) enhancements, calibration, validation and analysis. Lake and Reserv. Manage. 21(3):231-260.

The Lake Okeechobee Water Quality Model (LOWQM) was enhanced to more accurately simulate sediment-water phosphorus (P) dynamics by separating the organic P (OP) into four classes (readily degradable, moderately degradable, non-degradable and dissolved), and to more accurately simulate algal dynamics by representing the phytoplankton community with the three distinct major algal groups (cyanobacteria, diatoms and green algae) observed in the lake. The model was calibrated and validated to observed water column nutrient data, sediment nutrient measurements and biovolume data for cyanobacteria, diatoms, and green algae. Model predictions were consistent with experimental observations and indicated that net sediment inorganic P (IP) loads were twice the external TP loads and net sediment inorganic nitrogen (IN) loads were 0.64 times the external total N loads. However, because of organic nutrient and algal settling the lake sediments are an overall nutrient sink. Sensitivity analysis indicated that total algal carbon, algal groups and chlorophyll *a* were very sensitive to changing algal parameters, parameters affecting light, temperature and supply of IP to the water column. Nutrients were less sensitive for two reasons: 1) algae represent a small fraction of the total nutrient mass, 2) the large pools of sediment nutrients, with long turnover times, buffer changes in the water column. Sensitivity analysis pointed to three potential management options to improve lake water quality: dredging, chemical treatment of sediments and external load reduction. These options were previously considered in a large sediment management feasibility study, which concluded that the last option–load reduction–was the most viable.

Key Words: Lake Okeechobee, water quality model, nitrogen, phosphorus, nutrient budget, nutrient flux

Site Description

Lake Okeechobee is one of the great shallow lakes of the world (Fig. 1). This large (1730 km^2), shallow (mean depth 2.7 m), subtropical lake in central south Florida has accumulated tons of P over the past few decades, leading to accelerated eutrophication (Havens *et al.* 1996). As the lake has become more eutrophic, increases in total phosphorus (TP), algal blooms (determined from events where chlorophyll *a* concentrations exceed 40 mg/m^3), and a shift from diatom to cyanobacteria dominance have occurred (Havens *et al.* 1996).

Numerous best management practices were put into place from the 1970s to the 1990s to reduce these excessive P loads. Despite some reduction of P loads to the lake, the average annual P concentration in the lake has remained above 0.090 mg P/l since 1994 and has increased in recent years (Havens and James 2005). The large store of P in sediments, estimated at 28,700 metric tons in the top 10 cm (Reddy *et al.* 1995), could explain the continued increase in P. Dissolved inorganic P (DIP) flux, estimated from sediment core measurements, is similar to external loads of TP (Moore *et al.* 1998). Annual P budgets show that P assimilation has declined in recent years (Havens and James 2004). Therefore, although sediments continue to be a net P sink, much of the P that accumulates in the surface sediments in the form of organic material is recycled to the water column as available

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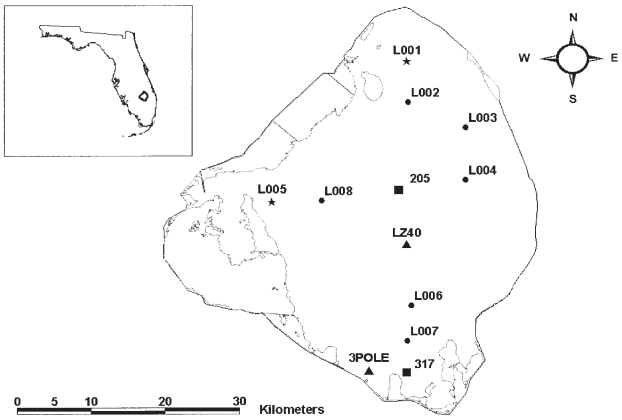

Figure 1.-Map of Lake Okeechobee showing the eight long term water sampling network and algal sampling networks (● Water Quality only ■ Algae only (1989-1992) ▲ Algae only (1993-2000) ★ All three).

inorganic P. This extra source of inorganic P contributes to algal growth, algal blooms and the further eutrophication of Lake Okeechobee.

Seasonal changes in water quality demonstrate the impact of sediment water interactions. During the winter dry season, cold fronts move through producing rather strong and consistent wind events. These wind events, along with the lake's long fetch produce significant waves that resuspend the bottom sediments (Jin and Ji 2001). Nutrients and solids are higher and light penetration is lower in the winter than in the summer (Maceina and Soballe 1990). During the summer calm season, strong wind events are much more isolated, sediment resuspension is less, the waters are warmer, and the light climate is significantly more favorable to algal growth (Havens *et al.* 1995a). Total and inorganic nutrient concentrations are lower as a result of the reduced sediment resuspension and increased uptake and settling of phytoplankton.

A very strong example of sediment water interactions was observed before and after Hurricane Irene passed close to the lake in late October 1999. Before the hurricane, Secchi Disk depth, total suspended solids (TSS) and TP were within the range of the historical data (a 20 year record). After the hurricane, measurements were either greater (for TSS or TP) or less (Secchi Disk) than 95% of the historical measurements (Havens *et al.* 2001). Sediment resuspension, due to the large waves created by the high winds, was considered the cause of this dramatic change.

Model Background

The concern over the cultural eutrophication of Lake Okeechobee precipitated efforts to improve understanding and management of this lake (Aumen and Wetzel 1995). Part of these efforts included development of a water quality model to estimate the P load reduction required to reduce algal blooms and water column P concentrations.

The Lake Okeechobee Water Quality Model (LOWQM) was developed to provide an understanding of internal nutrient processing within the lake, and to assess lake-wide responses to external nutrient load reductions. This model is an improvement over the previous modified Vollenweider Model used to assist management of P loads to the lake (Havens and James 1997), because in lake nutrient processes, including algal dynamics and sediment water interactions are simulated by the LOWQM.

The LOWQM is a spatially averaged, deterministic, mass balance model based on an enhanced version of EUTRO5, the eutrophication submodel of the Water Quality Analysis Simulation Program, Version 5 (WASP5; Ambrose *et al.* 1993a, Ambrose *et al.* 1993b). The LOWQM simulates the nitrogen (N), P, and oxygen cycles, as well as phytoplankton dynamics (Fig. 2, James and Bierman 1995, James *et al.* 1997). Water quality samples have been taken on this lake at eight locations every month since 1973. Because of the large size of Lake Okeechobee and the frequency of sampling, this model was calibrated and validated to whole-lake, monthly averaged observations.

The LOWQM was previously enhanced to incorporate a number of recommendations made by Bierman and James 1995), including the addition of: surface sediment layers and sediment processes that affect nutrient availability in the water column; the impact of sediment resuspension to the water column; and, dynamically changing depths and interfacial areas between water column model segments as the lake volume changes. These improvements are documented in James *et al.* (1997) and the equations that involve resuspended solids and their impact on light and dissolved inorganic nutrients are defined in Appendix 1 (Equations 1-9). This paper presents further enhancements that have been made to improve the model's ability to predict decadal-scale response to external nutrient load reductions, which is buffered by the large nutrient reservoir in the surface sediments. These enhancements included: four organic P (OP) classes based on degradability and solubility; three algal groups to represent the major algal classes in the lake including N-fixing cyanobacteria; and silica dynamics to simulate a diatom group (Fig. 2). These additions are more explicitly described below and in Appendix 1 (Equations 10 - 21).

The objectives of this study were to develop and implement these more accurate descriptions of algal dynamics and OP diagenesis reactions in the model and to calibrate and validate the model to an 18-year (1983-2000) period of observed data from Lake Okeechobee. Specifically, we wanted to determine the potential impact of sediment-water interactions, cyano-

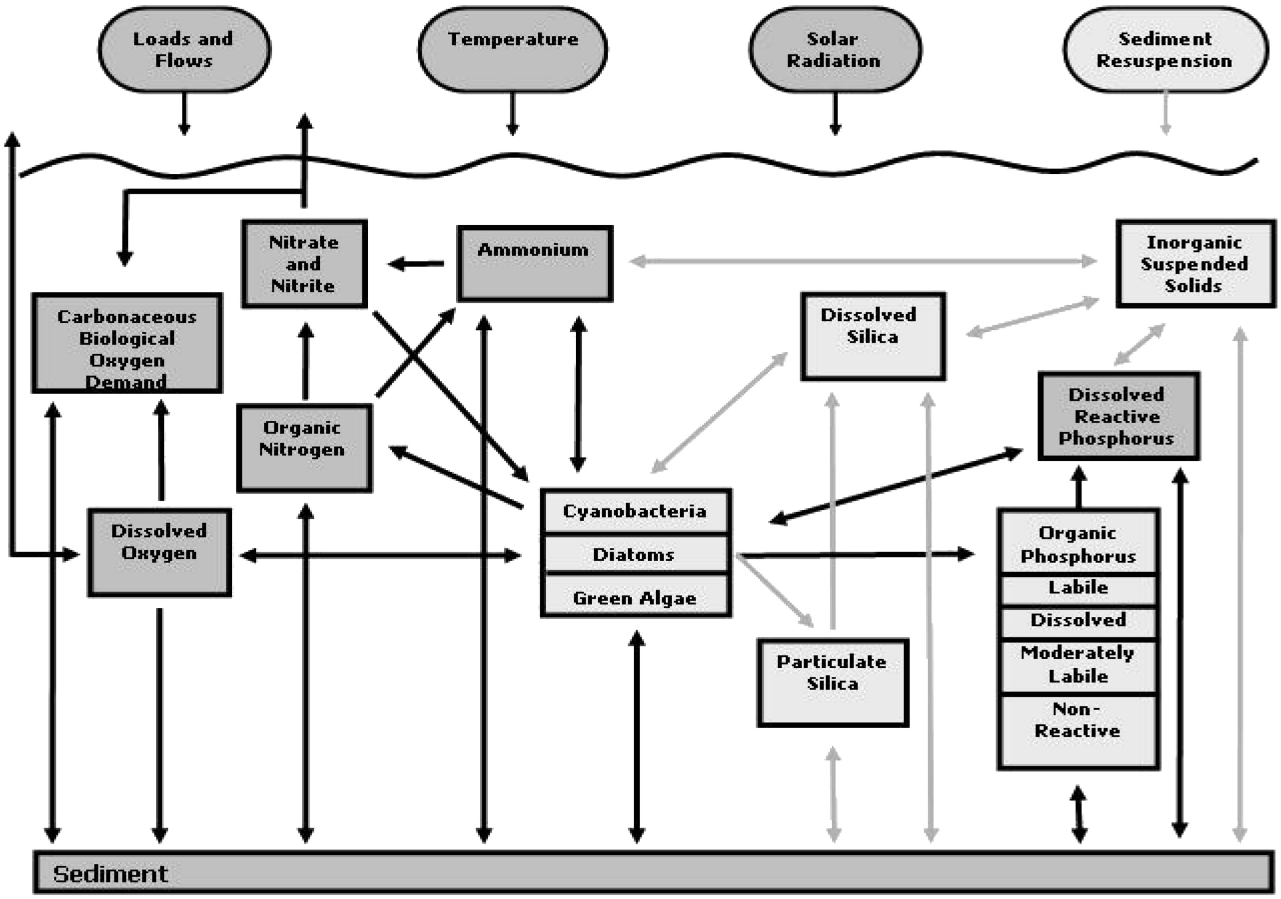

Figure 2.-Diagram of the enhanced Lake Okeechobee Water Quality Model (LOWQM). Lighter boxes and arrows indicate enhancements from previous version.

bacteria N-fixation, and the major algal groups' impact on P and N dynamics. Our attempt in this model development is to represent dominant long-term trends in the observed data. The model can then be used to determine if various management practices to reduce nutrient loads will provide long term improvements in lake-wide average water quality.

Model Enhancements

Phosphorus

The modified LOWQM has three particulate OP (POP) variables representing: readily- (OP1), moderately- (OP2), and non-degradable (OP3) POP (Fig. 2). These three types of non living OP use the same mathematical kinetic formulations as the original LOWQM, but with different process rates (Appendix 1: Equations 10, 11, 12). Mineralization rates – K_{OP1}, K_{OP2}, and K_{OP3} – distinguish the degree of resistance to degradation, from readily degradable to highly refractory (Appendix 2). The non living OP produced by algal respiration and death is apportioned among the three state variables through user specified fractions (a three-way split): X_{OP1}, X_{OP2} and X_{OP3}. Similar frameworks have been successfully applied in other studies (*e.g.,* DiToro and Fitzpatrick 1993, Smits and van der Molen 1993). The relative rates of conversion to inorganic P of the three POP fractions control the time scale over which changes in depositional fluxes will be reflected by changes in sediment-water fluxes of P (*c.f.* DiToro and Fitzpatrick 1993).

A fourth OP state variable was added to represent dissolved OP (DOP, Fig. 2). Formation of DOP is determined as a fraction of the POP mineralization product (Appendix 1, Equation 13). POP mineralization efficiency is specified for the moderately- and readily-degradable POP forms in the parameters Φ_{OP1} and Φ_{OP2}, respectively (Appendix 2). These fractions represent the degree of bacterially-mediated conversion of POP to inorganic P. This is consistent with the understanding that DOP is intermediate in the POP diagenesis process. A reaction rate for the conversion of DOP to DIP (K_{OP4}) is specified independent of the OP1 and OP2 diagenesis rates (Appendix 2).

DIP and particulate inorganic P (PIP) are influenced by redox-dependent partitioning to sediments. Moore and Reddy (1994) found that fluxes of DIP from the sediments of Lake Okeechobee were much higher under anaerobic conditions than aerobic ones. This is attributed to the high sediment iron content that sequesters IP under aerobic conditions and releases it under anaerobic conditions. To simulate this process indirectly, the sediment P partition coefficient, K_{PS} (Appendix 1, Equation 6), was adjusted to mimic iron in an oxic ferric (Fe^{3+}) state to sequester IP in a particulate form (K_{IPOV}, in the surface sediments, Appendix 3). In anoxic subsurface sediments iron is in a ferrous (Fe^{2+}) state and releases IP. However, much of this IP is bound to calcium (Ca) in the form of appetite (Brezonik and Engstrom, 1998). Thus K_{ipav}, in subsurface sediments was adjusted to reflect the binding affinity of Ca.

Nitrogen

The nitrogen cycle was investigated in the LOWQM to determine its impact on P and algal dynamics in the lake. This included the processes of N limitation, nitrification, denitrification, mineralization of organic N (ON), recycle of unavailable N forms, and N-fixation. All but N-fixation are available in the original EUTRO5 model (Ambrose *et al.* 1993ba). We added N-fixation (Appendix 1: Equation 18), as described below.

Silica

To adequately simulate diatom nutrient kinetics, a simple silica submodel was developed, consisting of a particulate and dissolved (available) silica pool (Fig. 2, Appendix 1: Equations 20 and 21). The particulate silica (PSI) is converted to the available silica (DSI) using a first-order dissolution constant K_{PSI} (Appendix 3). The dissolved silica is taken up by the diatoms as they grow. Diatom death completes the cycle by supplying PSI and DSI back to the water column and sediments.

Algae

The three groups of phytoplankton represented in this enhanced model include green algae (with no model modifications), N-fixing cyanobacteria, and diatoms (Appendix 1: Equation 14). To simulate N-fixing cyanobacteria a switch (NFIX) was added that specified the concentration of dissolved inorganic N (DIN) below which N-fixation is turned on (Appendix 1: Equation 18, Appendix 4). When N-fixation is turned on, cyanobacteria are no longer N-limited, and do not remove any DIN from the water column. The source for DIN uptake by cyanobacteria, rather, is implicitly assumed to be the atmosphere. The value for the switch was determined from Smith *et al.* (1995), who determined that N-fixation was most likely to occur in Lake Okeechobee when the DIN concentration in the water column was below 0.100 mg N/l. Diatoms are simulated by adding silica uptake kinetics (Fig. 2, Appendix 1: Equation 19, Appendix 4). DSI can influence diatom growth only if it is the most limiting nutrient. The three algal groups interact competitively through uptake of nutrients and impacting the light extinction coefficient as specified in Appendix 1 (Equations 5, 16, Appendix 4).

Model Development, Calibration and Validation

The modified LOWQM was applied to Lake Okeechobee for an 18-year period beginning in 1983 and extending through 2000. The model calculated the water budget of Lake Okeechobee from a set of time series that include monthly inflows, outflows and precipitation rates as developed by James *et al.* (1995a). An additional time series, monthly evaporation, was developed by adjusting model-predicted volumes to closely match observed volumes for the lake. The model used time series of monthly inflow-weighted nutrient concentrations to estimate loads as developed by James *et al.* (1995a), and monthly averaged water temperatures and solar radiation, taken from South Florida Water Management District's (District) data bases (www.sfwmd.gov/org/ema/dbhydro/index.html). The model computed time-varying volumes based on the flow rates and initial volumes. The model used a 0.08 day time step to calculate changes in the state variables. Model results were output daily and averaged by month for each year.

Resuspension and Settling

A constant gross settling rate, V_{ISS}, (59.3 cm/day) was specified for inorganic suspended solids (ISS) in the model, based on the work of Mehta (1991) (Appendix 1: Equations 1 and 2, Appendix 3). An additional time series of sediment resuspension, W_{ST} (Fig. 3a), was developed for the entire observed data set (1983-2000) by adjusting monthly values until simulated ISS values closely matched monthly observed values (Fig. 3b,c). W_{ST} averaged 0.0024 cm of sediment/day (95% confidence interval 0.0022 - 0.0027 cm of sediment/day), with a maximum of 0.02 and a minimum of 0 cm of sediment/day. To close the sediment/water mass balance loops, all particulate nutrients and algae in sediments were resuspended using this time series. Although the model could calculate these fluxes from wind speed and direction data, as in James *et al.* (1997), these data were incomplete for the 18 year period of the simulation.

Given W_{ST} for particulate N and P, the gross settling rate for organic species, V_O (6.7 cm/day) of P and N was calibrated to observed monthly averaged values (Appendix 1, Equations 10, 11, 12). This V_O value was constant throughout the

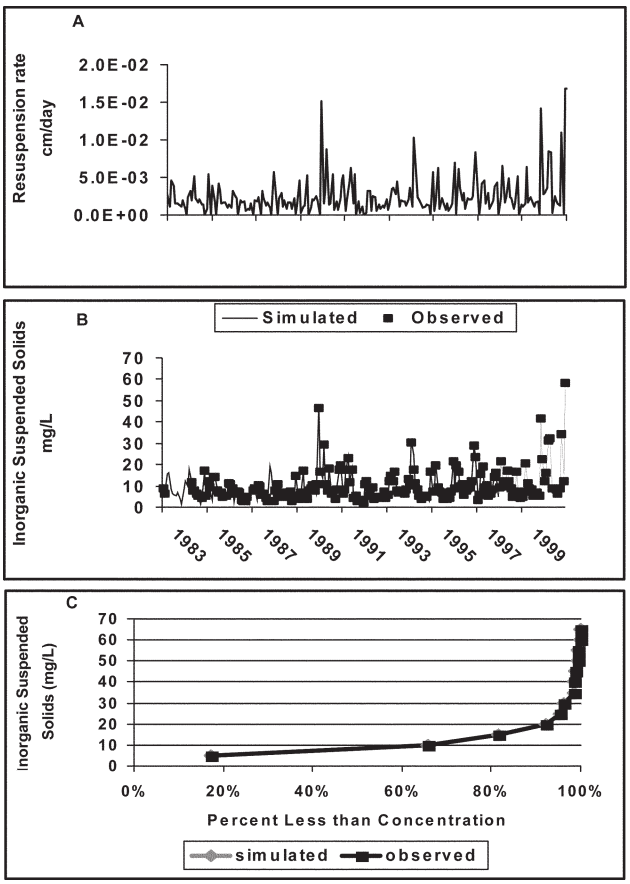

Figure 3.-A) Sediment resuspension rate (Wst) forcing function, B) monthly measured and simulated inorganic suspended solids, C) Cumulative frequencies of monthly averaged values.

simulation (Appendix 3). However, the fraction of dissolved ON (DON) in the water column and sediment (F_{DON}) was set at 0.002 mg DON/mg TN in the water column and sediments to produce simulated TN that closely matched observed monthly averaged TN values (Appendix 3). This constant fraction of ON was within the range of values of DON in Lake Okeechobee sediments as observed by Reddy (1991).

We specified a surficial aerobic sediment layer and a subsurface anaerobic layer, approximating the mixed layer depth. The surficial layer thickness was specified as 1 cm, consistent with measurements by Moore and Reddy (1994). The anaerobic layer thickness was specified as 5 cm. Sediments below 6 cm were considered permanently buried out of the system, there was no communication upward. Subsurface sediments and all contents were buried out of the system at a constant rate, V_b (0.095 cm/year)-determined from Pb^{210} studies of sediments in Lake Okeechobee (Brezonik and Engstrom 1998, Appendix 3).

Observed Data and Calibration and Validation Techniques

All initial values used for kinetic rates were based on those used by James and Bierman (1995), and James *et al.* (1997). We constrained all parameters within acceptable ranges as defined in Bowie *et al.* (1985), Thomann and Mueller (1987), and Ambrose *et al.* (1993b). Calibration was done in the order: ISS; TP; DIP; TN; DIN; DSI; chlorophyll *a* (CHLA), and phytoplankton carbon (cyanobacteria, diatom, and green algae) carbon. We fixed as many parameters as possible (Appendix 2, 3, 4), and systematically calibrated model output to monthly averaged data collected from January 1983 to December 1992 from the District's long-term, eight-station network (Fig. 1, Stations L001 - L008).

Phytoplankton biovolume estimates were taken from the raw data used by Cichra *et al.* (1995) at Stations 104 (L001), 204, 317 and 513 (L005) for the calibration time period 1988 to 1992 and from the District's studies that began in 1993 and continued to present at Stations L001, L005, LZ40, and 3-Pole for the validation period (Fig. 1). All biovolume data are available from the Lake Okeechobee Ecological Data Management System of the South Florida Water Management District.

The regression equations of Strathman (1967) were used to estimate the carbon (C pg) content of each algal group based on these biovolume (V µl) measurements:

$$\log(C\ cell^{-1}) = -0.460+0.866 \log(V\ cell^{-1}) \qquad (1)$$

for green algae and cyanobacteria and

$$\log(C\ cell^{-1}) = -0.422+0.758 \log(V\ cell^{-1}) \qquad (2)$$

for diatoms. The carbon per cell estimates were multiplied by the cell densities to determine mg C/l for each phytoplankton group and these values were used in comparison to model simulated values as described below.

Lake Okeechobee is one of the most studied lakes in the world (Havens *et al.* 1997, Aumen and Wetzel, 1995). It has been sampled on a monthly to twice monthly basis at eight locations within the pelagic region from 1973 until today (James *et al.* 1995b). Rigorous quality assurance and quality control procedures were not in place until 1982; therefore we used the data from 1983 to 2000 for calibration and validation. The model represents the lake as spatially and temporally averaged and it is compared to monthly averaged data using a number of numeric comparisons from the literature including percent bias (%Bias), R^2, and percent root mean square error (%RMSE) (see Janssen and Heuberger 1995).

Because there are 18 years of monthly to bi-monthly historical water quality data available for Lake Okeechobee, monthly averaged observations and associated standard

deviations can be calculated. As a result, two other numerics that demonstrate the ability of this lake-wide averaged model to simulate averaged conditions of the lake: Percent Correspondence (%Corr), which was used with the previous LOWQM model (James *et al.* 1997); and local model efficiency (LME), which we developed for use in this project. Percent Correspondence compares the number of monthly model predictions based on a Student's t-test that were not significantly ($p > 0.01$) different from the monthly averaged observed values–N_c–(sensu, Bierman and Dolan 1986a, b) to the total number of comparisons, N_t:

$$\%Corr = \frac{N_c}{N_t} \bullet 100\% \tag{3}$$

The other numeric, LME, was developed based on the following considerations:

1. Acceptable values should range from 0 to 1 with 1 equaling a perfect fit of observed to predicted measures; and,
2. It should account for variation in observed data for each individual comparison.

The equation is,

$$LME = 1 - \frac{\sum_{t=1}^{N_{comp}} \left[\frac{|\overline{S}_t - \overline{O}_t|}{2 \bullet O_{SDt}} \right]}{N_{comp}} \tag{4}$$

Where $\overline{O}_t$ is the averaged observed value for a given month (t), $\overline{S}_t$ is the monthly average of simulated values corresponding to the dates when the observed values were collected. This difference is divided by twice the standard deviation, O_{SDt}, of the monthly observed values:

$$O_{SDt} = \sqrt{\frac{\sum_{i=1}^{N_t} (O_{ti} - \overline{O}_t)^2}{N_t}} \tag{5}$$

O_{ti} is the mean monthly observed value from an individual station i in month t and N_t is the number of stations contributing to the samples in month t.

Monthly comparisons are summed and then divided by the total number of comparisons (N_{comp}) to obtain an overall average that could range from 0 to infinity. Subtracting this number from one gives the LME which can range from 1 to negative infinity. If, on average, the difference between $\overline{O}_t$ and $\overline{S}_t$ is less than twice the standard deviation (*i.e.*, within 95.46% of the probability curve of the observed data), then the LME will be between zero and one. If, on average, the difference $\overline{O}_t$ and $\overline{S}_t$ is greater than twice the standard deviation, then LME is less than zero and the comparisons were judged not acceptable. This measurement can be used with any model to observed water quality comparison provided that an observed mean and standard deviation can be developed for each comparison time period.

For the LOWQM model, the calibration period was the first ten years (January 1983 to December 1992) of the model simulation. The validation period was the next eight years (January 1993 to December 2000). The time periods were chosen because algal counts and biovolume measurement frequencies, labs, and locations were different in these two time periods. A calibrated and validated water column variable was one with a |%Bias| and %RMSE less than 25 percent, an LME greater than zero and a %Corr greater than 50 percent.

Bulk concentration data (*e.g.*, mg P/Kg Sediment) measured by Reddy (1991) and Fisher *et al.* (2001) in the top 10 cm of sediments from 171 locations were averaged for the lake as a whole and used to check the calibration and validation of the sediment model. These bulk data are available from the District's Lake Okeechobee Ecological Data Management System. The readily degradable fraction, OP1, was not measured directly in the sediments of Lake Okeechobee (Olila *et al.* 1995). This fraction was assumed to be a small fraction of the moderately degradable organic P measured by NaOH extraction. To verify the simulation of OP1 and OP2 fractions, they were added together and compared against the moderately degradable fraction measured by Olila *et al.* (1995) and Fisher *et al.* (2001). The highly-resistant OP fraction, represented by OP3 was compared to the residual OP measured by Olila *et al.* (1995).

We compared the average measured concentrations of these variables with the predicted annual average from the model for the same year. We used SAS (SAS Institute, Inc. 1989) to determine all statistical tests. Cumulative frequency diagrams were developed from monthly observed and simulated data using Microsoft Excel 2002 (Microsoft Corp, Redmond, WA).

Sensitivity Analysis

A sensitivity analysis was performed using the calibrated and validated model. Each individual parameter, all initial starting values, and all external forcing functions were increased and decreased individually by 10 percent. The impacts, averaged over the 18 year simulation, on chlorophyll *a*, TP, DIP, TN, DIN, cyanobacteria, diatoms, and green algae were determined using a relative sensitivity measure (Haan and Zhang 1996):

$$RS = \frac{\Delta S}{\Delta p} \bullet \frac{P}{\overline{S}} \tag{6}$$

Where the relative sensitivity (RS) is equal to the change in the simulated output (ΔS) divided by the change in the pa-

Table 1.-Calibration and validation numerics for water column observed and predicted variables.

State Variable	Time Period	N	O	O_{sd}	S	S_{sd}	% Corr	% Bias	Local Efficiency	%RMSE	R^2
Inorganic Suspended Solids	Calibration	96	8.710	6.199	8.705	6.183	**94.8**	**-0.1**	**0.99**	**0.5**	**1.00**
mg/L	Validation	94	12.182	9.041	12.247	9.022	**95.7**	**0.5**	**0.98**	**0.2**	**1.00**
Total Phosphorus	Calibration	112	0.093	0.030	0.096	0.018	**73.2**	**3.2**	**0.53**	**2.5**	**0.34**
mg P/L	Validation	94	0.110	0.041	0.106	0.021	**70.2**	**-3.7**	**0.52**	**2.7**	**0.58**
Inorganic Phosphorus	Calibration	112	0.025	0.014	0.024	0.021	**76.8**	**-6.5**	**0.48**	**5.9**	**0.42**
mg P/L	Validation	92	0.031	0.015	0.033	0.025	**66.3**	**7.0**	**0.42**	**6.2**	**0.45**
Total Nitrogen	Calibration	112	1.545	0.301	1.450	0.277	**55.4**	**-6.2**	**0.30**	**2.4**	0.02
mg N/L	Validation	94	1.536	0.310	1.665	0.332	**63.8**	**8.4**	**0.34**	**2.0**	**0.44**
Dissolved Inorganic Nitrogen	Calibration	110	0.139	0.110	0.146	0.073	**59.1**	**4.6**	-0.53	**6.1**	**0.36**
mg N/L	Validation	94	0.191	0.142	0.184	0.098	**62.8**	**-3.7**	-0.24	**5.5**	**0.49**
Ammonia	Calibration	74	0.016	0.010	0.017	0.005	**66.2**	**2.4**	**0.16**	**8.1**	0.00
mg N/L	Validation	76	0.015	0.008	0.015	0.005	**50.0**	**-4.7**	-1.00	**7.8**	0.05
Nitrate+Nitrite	Calibration	110	0.124	0.110	0.129	0.076	**57.3**	**4.6**	-0.84	**6.8**	**0.37**
mg N/L	Validation	94	0.176	0.141	0.169	0.102	**64.9**	**-3.9**	-0.09	**5.8**	**0.51**
Dissolved Silica	Calibration	44	11.681	3.957	12.892	0.953	**65.9**	**10.4**	-0.34	**5.5**	0.01
mg Si/L	Validation	35	8.884	3.173	8.387	1.377	**60.0**	**-5.6**	-0.96	**5.3**	**0.23**
Chlorophyll a	Calibration	109	25.165	8.238	27.829	15.353	**66.1**	**10.6**	**0.35**	**5.4**	**0.18**
mg L-1	Validation	92	23.412	7.206	21.428	15.412	**59.8**	**-8.5**	**0.18**	**5.9**	**0.28**
Cyanobacteria Carbon	Calibration	40	0.529	0.257	0.643	0.355	**70.0**	**21.6**	**0.68**	**11.2**	**0.11**
mg C/L	Validation	41	0.473	0.461	0.365	0.275	**70.7**	**-22.9**	-0.28	**14.4**	**0.17**
Diatom Carbon	Calibration	40	0.041	0.023	0.041	0.027	47.5	**1.3**	**0.38**	**13.6**	0.00
mg C/L	Validation	41	0.095	0.142	0.088	0.086	**78.0**	**-6.7**	-1.83	26.6	0.00
Green Algae Carbon	Calibration	40	0.019	0.007	0.020	0.013	**52.5**	**2.5**	**0.66**	**10.6**	0.05
mg C/L	Validation	41	0.039	0.034	0.037	0.036	**90.2**	**-3.4**	-6.59	**18.9**	0.01
Total Algal Carbon	Calibration	40	0.590	0.254	0.704	0.372	**75.0**	**19.4**	**0.70**	**10.0**	**0.15**
mg C/L	Validation	41	0.606	0.565	0.490	0.383	**85.4**	**-19.1**	-0.02	**14.8**	**0.11**

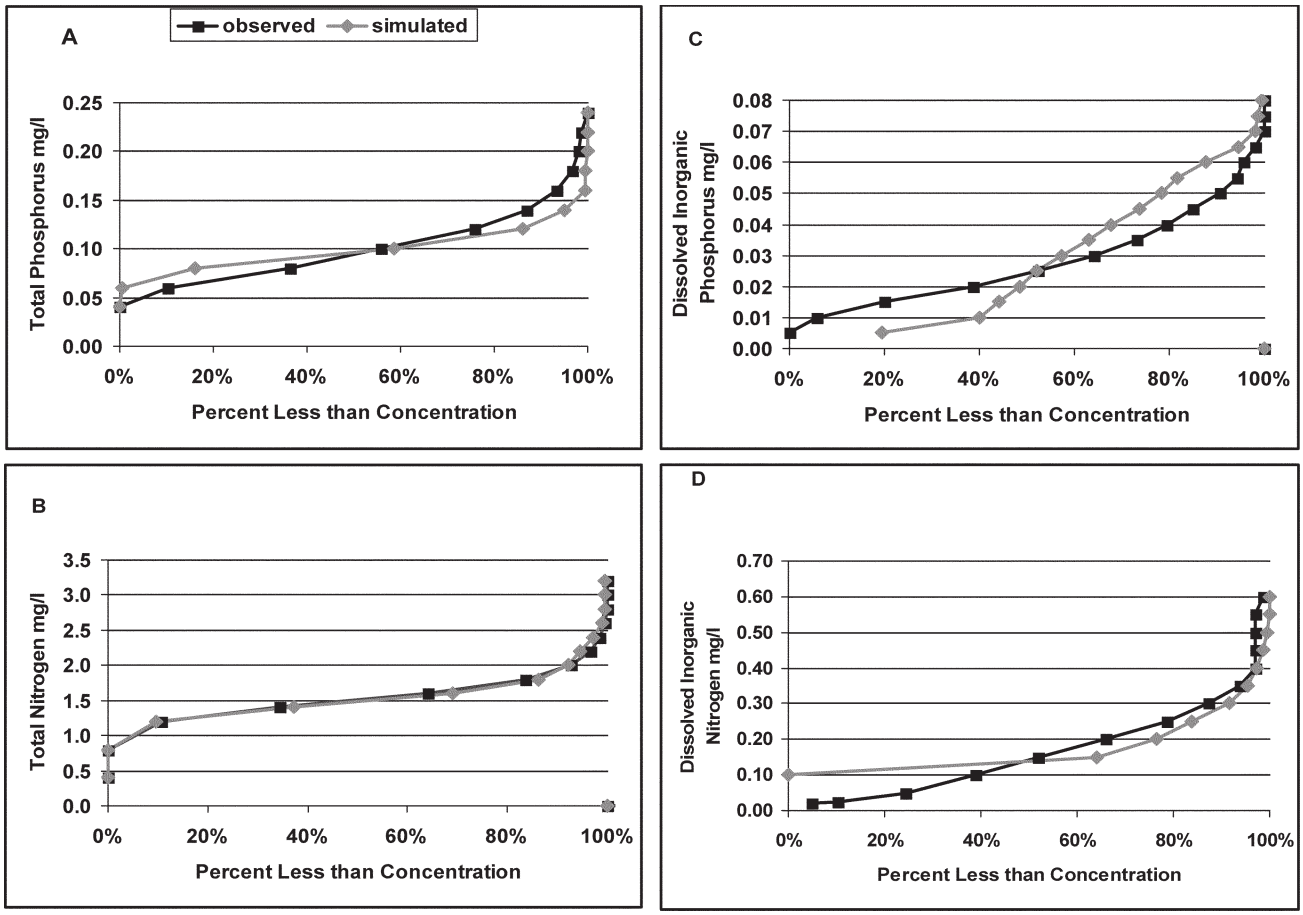

Figure 4.-Cumulative frequencies of model simulated and monthly average observed values for A) Total Phosphorus, B) Total Nitrogen, C) Dissolved Inorganic Phosphorus and D) Dissolved inorganic Nitrogen.

rameter (Δp). This is normalized to the calibrated parameter value P divided by the averaged calibrated simulated value ($\bar{S}$). Relative sensitivity can be used directly to compare all the parameters against one another.

Nutrient Fluxes

Using model predicted fluxes, annual averaged budgets were developed for N and P. These can be used to investigate the potential significance of various fluxes in relation to the external loads. These fluxes could also be used to determine potential areas of future research.

Results

Calibration and Validation

Because ISS was calibrated using resuspension velocity as a forcing function, the model result closely matched the observed mean monthly value (Fig. 3b). The cumulative frequencies of simulated and observed data are very close (Fig. 3c), which is also reflected in the numeric comparisons (Table 1).

Preliminary model simulations predicted dramatic declines of ISS in the sediments, primarily a result of the specified net burial rate, which was constrained by independent measurements from dated sediment cores. A review of sediment data for Lake Okeechobee (Brezonik and Engstrom 1998) and the model-computed fluxes indicate that the model was apparently missing an inorganic solids source. That source was assumed to be from settling of calcium carbonate, which is not simulated in the model but makes up 12 to 40 percent of dry sediments in the lake (Brezonik and Engstrom 1998). Given that the range of solids accumulation in sediments was estimated between 300 to 1150 g/m^2/year (Brezonik and Engstrom 1998), we then estimate that the accumulation of $CaCO_3$ in the sediments is between 36 and 460 g/m^2/year. For the present model, the missing mass was calibrated as a load of 327 g/m^2/year to the surface sediments.

Nutrient predictions met most calibration and validation numeric standards (Table 1). Cumulative frequency diagrams

show that TP is slightly over predicted at values less than 0.1 mg P/l and is slightly under predicted above this value (Fig. 4a). For TN the cumulative frequencies for predicted and observed are very similar (Fig. 4b). These calibration and validation results were partly the result of adjusting the settling rate (V_O) of the organic nutrients and adjusting the percent dissolved concentration for ON.

This settling rate, V_O, also influenced the calibration measurements for DIP and DIN, which also were acceptable with the exception of the negative LME for DIN (Table 1). The cumulative frequency diagram for DIP indicates the model under predicts below 0.025 mg/l and slightly over predicts at values above this (Fig 4c). For DIN the cumulative frequency diagram shows that the model is unable to predict the observed occurrence of DIN concentrations below 0.1 mg/l, and slightly under-predicts above this value (Fig. 4d). This is attributable to the NFIX option, which turns on N-fixation when the DIN concentration falls below 0.1 mg N/L. In reality, the rate of N-fixation is probably a variable continuous function that is dependent on a variety of factors, rather than the step function switch represented in the model. However, research on Lake Okeechobee has not provided any method to formalize a continuous function describing how this process operates under DIN-limiting conditions. An accurate representation of the contribution of N fixation to the overall total N budget of the lake was a more important calibration constraint than matching computed and observed DIN concentrations in the range of low, rate-limiting values.

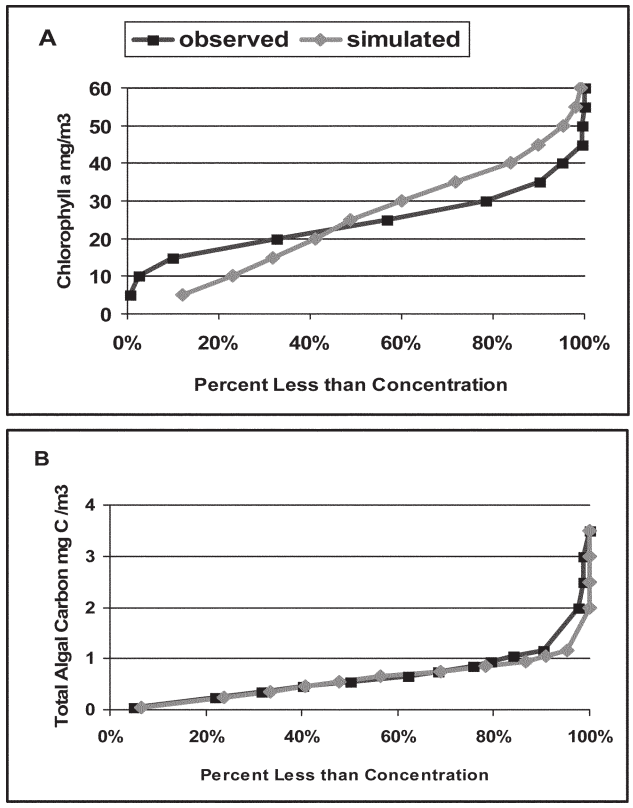

Figure 5.-Cumulative frequencies of model simulated and monthly averaged observed values for A) chlorophyll *a*, B) Total Algal Carbon.

For DSI, the predictions failed to meet the LME in the calibration period, while all other numeric conditions were met (Table 1). The sensitivity analysis indicated that changing any parameter, initial condition or load related to DSI had no impact on other nutrients or algae (see Discussion). Therefore, silica has little impact on model calibration results.

The three separate algal species, cyanobacteria, diatoms and green algae, produce chlorophyll *a* values that meet the numeric criteria for %Bias, %RMSE, %Corr and LME (Table 1). From 1997 to 2000 the cyanobacteria represented between 50 and 80 percent of the total biovolume of algae in the lake (Havens *et al.* 2003). Model results are consistent with this finding; cyanobacteria make up a majority of the algal biomass followed by diatoms and then green algae. For the calibration and validation periods all calibration numerics are met by the algal groups with the exceptions of %Corr and %RMSE for Diatoms and LME for all algal groups in the validation period. Cumulative frequency diagrams indicate that chlorophyll *a* is under predicted below 25 mg/m^3 and over predicted above that (Fig 5a). Total algal carbon is slightly under predicted between 1 and 2 mg C/l (Fig 5 b). Cyanobacteria is slightly under predicted between 0.5 and 2 mg/l (Fig. 6a), Diatoms are slightly under predicted above 0.3 mg/l (Fig. 6b) and green algae predictions are very similar to observed values (Fig. 6c).

Sediment N and P content varies widely within and among sediment types and locations in the lake (Fisher *et al.* 2001), which is indicated by the large standard deviation of the observed data (Table 2). With a few exceptions, the average model-predicted values for the different forms of these nutrients were all within one standard deviation of the observed data for the given years. Only the inorganic solids, non-degradable OP, DIP, TP and Particulate IN were within 20 percent of the mean observed values. The larger differences between modeled and observed for the other nutrients could partially be because the observed data were estimated from the top 10 cm of sediment while the predicted values were from the top 6 cm of sediment.

Lake Okeechobee is a very complex system and this water quality model contains many deliberate simplifying assumptions with respect to spatial and temporal scales, and the level of resolution of chemical-biological processes. Based on the numeric comparisons in the calibration and validation periods for water quality parameters and algal groups (Table 1), the predicted and observed sediment parameter values (Table

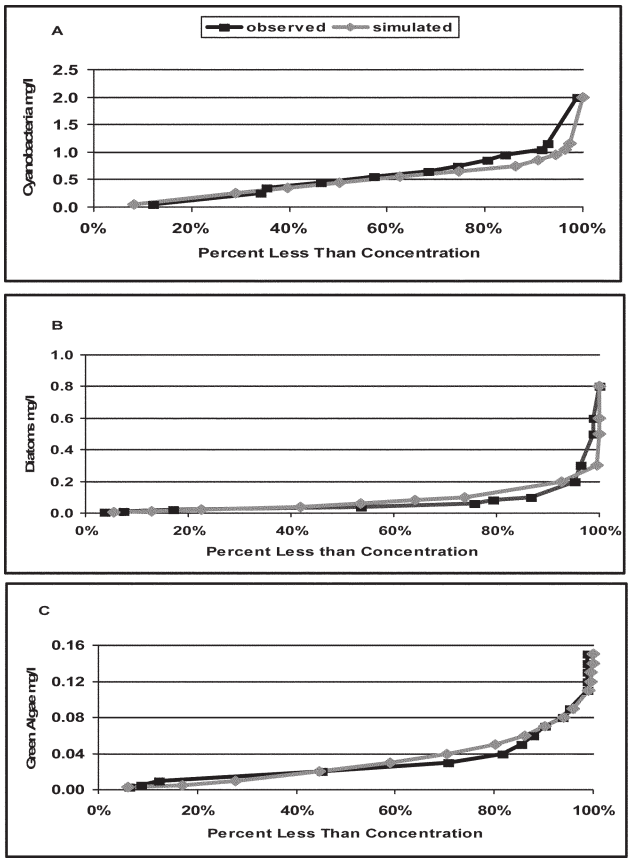

Figure 6.-Cumulative frequencies of model simulated and monthly averaged observed values for A) cyanobacteria, B) diatoms, C) green algae.

2), and the results from the cumulative frequency diagrams (Figs. 4, 5 and 6), this model is considered reasonably and acceptably calibrated and validated. In the authors' view, this model provides an acceptable and credible representation of the available data, such that it is a useful tool to further our understanding of long-term P dynamics in Lake Okeechobee.

Sensitivity Analysis

The results of the model sensitivity analysis are presented on a relative basis. As such, the influence of a tested parameter on a particular state variable indicates that a percent change in the parameter value will lead to a proportional change in the state variable (Equation 6). Thus if the sensitivity is 1.5 then a 10% change in the parameter value will lead to a 15% change in the state variable. If the sensitivity is 0.5, then a 10% change in the parameter variable will lead to a 5% change in the state variable. The sensitivity analysis investigated 159 parameters/forcing functions. Parameters that produced a relative sensitivity of 0.30 or greater for at least one nutrient (DIP, DIN TP TN) were tabulated (Table 3).

Chlorophyll *a* is most sensitive to factors influencing growth and death rates of cyanobacteria and diatoms (Table 3). If cyanobacteria out-compete the other algal groups, chlorophyll *a* concentrations increase, and vice versa. Cyanobacteria are responsible for approximately 80 percent of the total algal biomass. Consequently, changes in cyanobacteria will have a stronger influence on other algal-related state variables than changes in diatoms or green algae.

Results for green algae, diatoms and cyanobacteria are extremely sensitive to model parameters that affect their net growth rates and thus their competitive abilities (Table 3). These include growth, respiration, algal settling, temperature, water volume (affecting water depth), light extinction, uptake kinetics, supply and removal of available P. There are other parameters that affect these algal groups but have very little impact on nutrients. These include external inflows of algal, ratios of chlorophyll *a*, N and phosphorus to carbon, algal settling rates, and supply of inorganic N to the water column (data not shown). While there can be large changes in magnitudes and relative proportions of individual algal groups, the corresponding changes in total algal concentration are much smaller. The reason is that total algal concentrations are constrained by ambient light, temperature and nutrient concentrations. The proportional composition of the algal groups is much less constrained because it is a function of differences among the algal groups with respect to process rates, competitive advantages and temperature dependencies.

DIP results are influenced by model parameters affecting algal growth, light, resuspension, particulate settling, sediment water interactions and external P inflows (Table 3). As algae growth, solids settling, or aerobic sediment partition coefficient for P is increased, DIP declines. As algal respiration, settling, light extinction, water volume (stage) or supply of P from external loads or the sediment (IP in surface sediments) is increased, the reverse occurs. DIP is influenced positively by light extinction coefficients and water volume (stage) because increases in these factors reduce the light availability in the water column and reduces the growth rate of the algal groups, allowing an increase in the stock of DIP. Resuspension positively and settling negatively influences DIP through two processes, increasing and decreasing the supply of DIP to the water column from the sediments and decreasing and increasing the light availability, which as stated above influences algal growth which influences DIP. Increasing the aerobic sediment partition coefficient for IP reduces the available IP and increases the amount settled to the sediment.

DIN is strongly influenced by growth and respiration. As the growth of green algae or diatoms increases, more DIN is removed, as this growth is reduced, less DIN is removed.

Table 2.-Observed (from raw data of Fisher *et al.* 2001, Olila *et al.* 1995, and Reddy 1991) and predicted mean and standard deviations for sediments of Lake Okeechobee mg/m^2 for the top 6 cm of sediment.

	1988				1998			
	Observed		Predicted		Observed		Predicted	
State Variable	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation
Inorganic Solids	13,489,740	15,904,403	14,894,839	14,422	15,896,100	15,896,100	14,891,094	30,701
Total Phosphorus	1,246	5,818	8,439	80	10,668	13,346	8,577	79
Degradable Particulate Phosphorus	879	1,535	223	20	150	191	189	42
Non Reactive Organic Phsophorus	2,218	3,171	2,365	45	2,573	2,316	2,751	33
Dissolved Organic Phosphorus	89	173	6	3	2	7	6	3
Total Inorganic Phosphorus	5,264	4,464	5,821	56	9,636	15,119	5,612	93
Dissolved Inorganic Phosphorus	16	21	18	12	22	18	17	11
Total Nitrogen	90,101	89,290	99,646	690	82,050	56,861	103,315	1,107
Dissolved Ammonium	130	118	5	2	50	38	9	8
Particulate Ammonium	422	336	408	19	414	329	405	46

Table 3.-Relative sensitivity values for selected variables (bolded values are greater than 0.30 or less than -0.30).

Description	Chlorophyll *a*	Phytoplankton Carbon	Cyano-bacteria	Diatom	Green Algae	DIP	DIN	TP	TN
Blue Green Growth Rate	**2.65**	**2.12**	**6.41**	**-13.88**	**-23.15**	**-0.76**	**0.43**	-0.12	0.21
Blue Green Growth Temperature Coefficient	**1.84**	**1.42**	**4.90**	**-11.41**	**-19.47**	0.10	**0.45**	-0.20	0.04
Blue Green Half Saturation Coefficient P	**-1.05**	**-0.85**	**-2.86**	**6.98**	**10.28**	-0.17	**-0.34**	0.18	0.00
Blue Green Respiration Rate	**-2.85**	**-2.34**	**-6.49**	**13.17**	**22.16**	**0.37**	**-0.42**	0.17	-0.09
Blue Green Respiration Temperature Coefficient	**-1.58**	**-1.27**	**-3.73**	**8.26**	**12.49**	-0.15	**-0.36**	0.24	0.01
Diatom Death Rate	**1.07**	**0.55**	**2.85**	**-15.33**	**4.43**	**0.50**	**0.45**	0.00	0.01
Diatom Growth Rate	**-1.63**	**-0.66**	**-5.01**	**28.78**	**-6.55**	**-1.05**	**-1.31**	**-0.33**	-0.15
Diatom Growth Temperature Coefficient	**-1.36**	**-0.73**	**-3.55**	**18.36**	**-4.61**	**-0.51**	**-0.40**	-0.05	-0.02
Diatom Half Saturation Coefficient P	**0.97**	**0.62**	**2.45**	**-11.73**	**2.99**	**0.31**	0.30	-0.06	0.00
Diatom Respiration Rate	**1.38**	**0.53**	**4.36**	**-25.44**	**5.71**	**0.81**	**0.66**	0.24	0.08
Diatom Settling Rate	**0.75**	**0.41**	**1.90**	**-9.99**	**3.15**	**0.32**	**0.32**	-0.11	-0.03
Diffusion between water and sediment	0.17	0.18	0.19	0.14	0.14	0.00	0.02	0.08	**0.31**
Green Death Rate	**0.44**	**0.53**	**2.70**	**3.17**	**-38.09**	**0.34**	**0.38**	-0.06	0.00
Green Growth Rate	**-0.39**	**-0.54**	**-4.45**	**-4.98**	**68.11**	**-0.74**	**-1.16**	-0.24	-0.12
Green Growth Temperature Coefficient	**-0.61**	**-0.69**	**-3.19**	**-3.21**	**42.43**	**-0.31**	**-0.32**	0.05	0.00
Green Respiration Rate	**0.32**	**0.44**	**3.93**	**4.23**	**-60.45**	**0.54**	**0.53**	0.16	0.07
Light Extinction Coefficient 1	**-0.60**	**-0.62**	**-0.67**	**-0.41**	-0.26	**0.99**	**0.66**	0.06	-0.07
light Extinction Coefficient 2	**-0.47**	**-0.48**	**-0.57**	-0.13	0.00	**0.68**	**0.54**	0.04	-0.04
Nitrogen Switch	0.08	0.08	0.07	0.15	0.18	-0.14	**0.42**	0.00	0.06
Organic Nitrogen Concentration in Bottom Sediments	0.01	0.02	-0.05	**0.31**	**0.38**	-0.09	0.19	0.00	**0.32**
Organic Nutrient Settling	-0.13	-0.13	-0.12	-0.20	-0.23	0.04	-0.08	**-0.36**	**-0.67**
Sediment Resuspension	-0.14	-0.13	-0.21	0.18	0.27	**0.90**	**0.61**	**0.39**	**0.36**
Solids bottom sediment concentration	**-0.62**	**-0.60**	**-0.73**	-0.10	0.05	0.10	**0.36**	-0.18	-0.11
Solids Settling	**0.47**	**0.46**	**0.54**	0.14	0.03	**-0.74**	**-0.54**	-0.06	0.04
SRP in Bottom Sediments	**0.45**	**0.46**	**0.53**	0.13	0.08	**0.53**	0.05	**0.33**	0.14
SRP solids partition coefficient (oxic)	**-0.35**	**-0.36**	**-0.43**	-0.06	-0.01	**-0.54**	-0.04	-0.29	-0.11
Temperature forcing function	**1.28**	**1.15**	**2.55**	**-4.64**	**-5.76**	**-1.47**	**-0.61**	-0.15	**0.47**
TP concentration in inflowing water	**0.41**	**0.40**	**0.45**	0.22	0.20	**0.39**	0.06	**0.31**	0.12
Volume	**-1.83**	**-1.77**	**-2.74**	**2.65**	**1.98**	**1.02**	**0.37**	0.03	-0.10

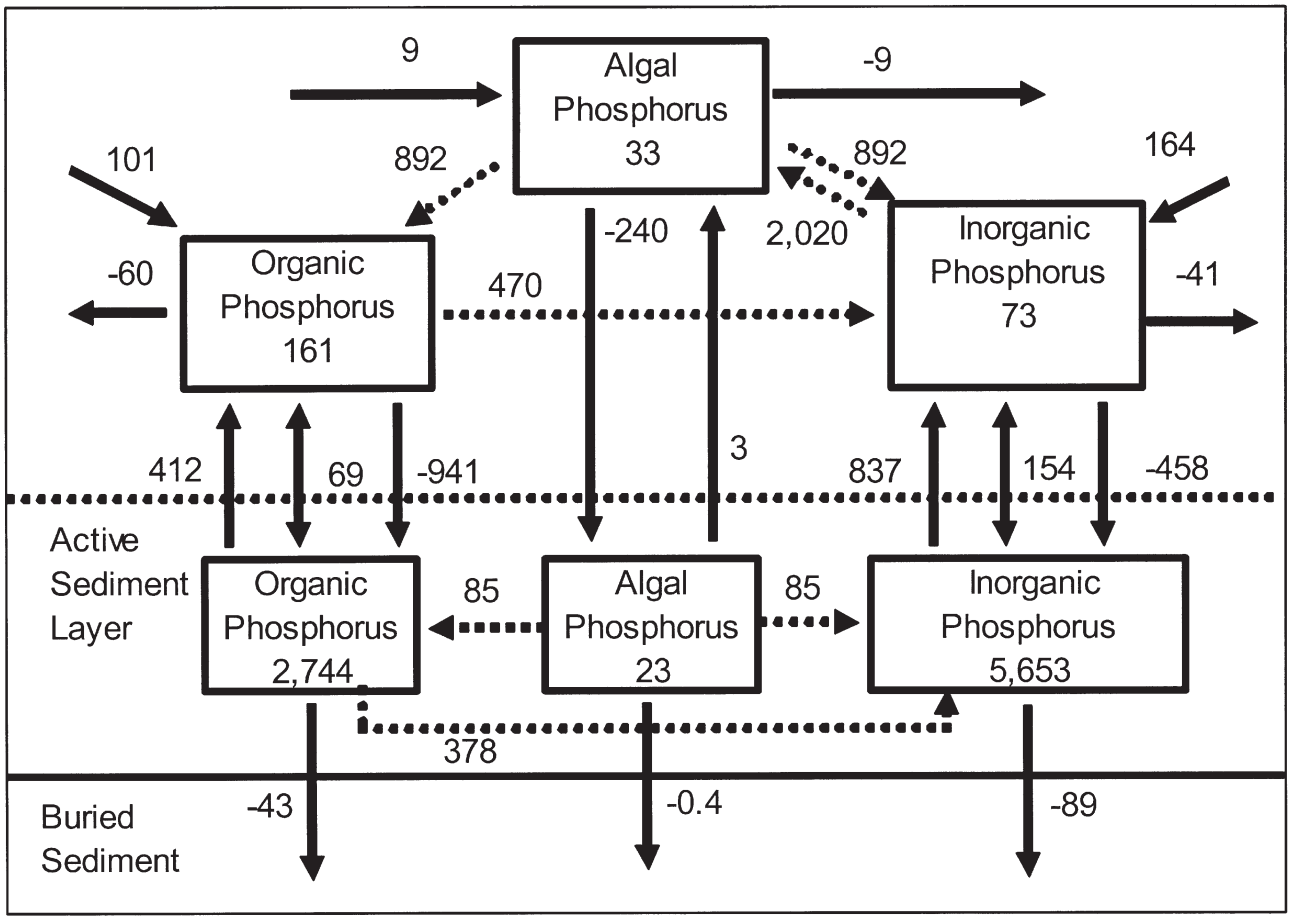

Figure 7.-Average flux (mg/m²/ year) and standing stock values (mg /m²) of phosphorus predicted from the calibration/validation of the Lake Okeechobee Water Quality Model. Negative values are fluxes out of the water column; positive values are fluxes into the water column. Double sided arrows indicate diffusive fluxes. Solid arrows represent transport. Dashed arrows represent transformations.

Parameters that influence the competitive interactions between cyanobacteria and diatoms and green algae (*e.g.*, P uptake half saturation coefficient, Respiration temperature coefficients, and settling rates) also have an impact on DIN, positive when cyanobacteria increase, and negative when diatoms or green algae increase. As with DIP, factors that affect light availability (water volume-stage, light extinction coefficients, resuspension) influence DIN positively. Increasing or reducing the N-fixation switch also influences DIN by reducing or increasing the lower limit of the predicted concentration values. DIN is also positively sensitive to sediment resuspension and negatively sensitive to temperature.

Very few parameters influence TN or TP significantly and of these only two–organic nutrient settling and temperature–exceed an absolute value of 0.4 for the relative sensitivity of TN. For TP those parameters with a significant influence include settling, sediment resuspension, supply of P to water column (TP in inflowing water and IP in sediments) and diatom growth, for TN they also include supply of N to the water column (ON in sediments, diffusion rate, resuspension).

Nutrient Fluxes

A mass balance diagram of average annual P fluxes, in mg/m²/year of P, predicted by the LOWQM was developed to show the importance of sediment-water interactions compared to both external loads and water column nutrient recycling via mineralization and algal dynamics (Fig. 7). This model simulation estimates that sediments accumulate 163 mg/m²/year of P. Thus 59 percent of the total external load (274 mg/m²/year) is assimilated into the sediments on an annual basis. This net settling of P to the sediments (the sum of all fluxes to and from the sediments) settles to the sediment primarily in the form of algal P (AP, 237 mg/m²/year) and non-living OP (460 mg/m²/year). Sediment OP (2,744 mg/m²) is recycled to IP (378 mg/m²/year) at the rate of 17 percent by mass per year, while sediment AP (23 mg/m²) is recycled to IP (85 mg/m²/year) at the rate of 470 percent per

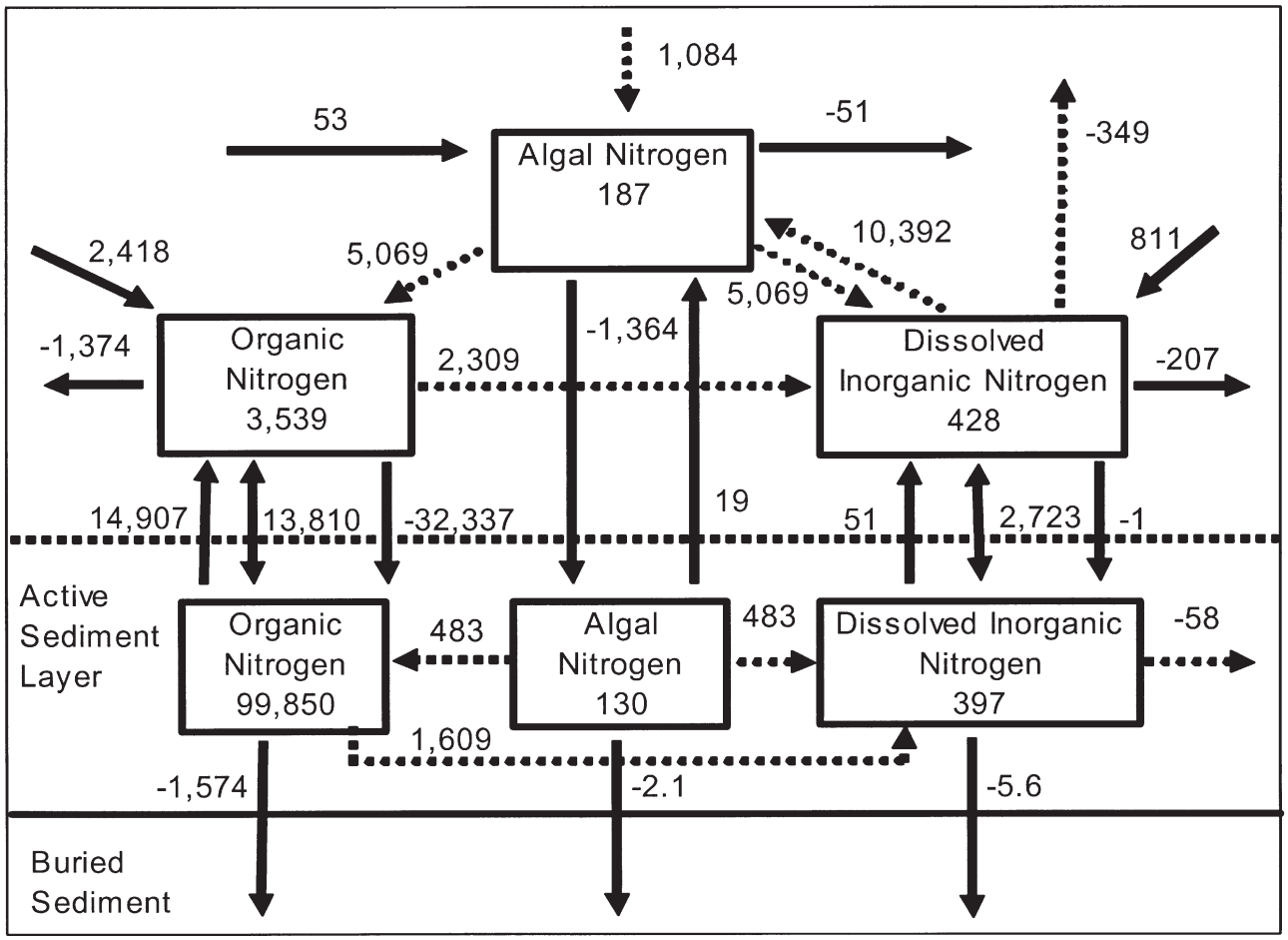

Figure 8.-Average flux (mg/m2/year) and standing stock values (mg/m2) of nitrogen predicted from the calibration/validation of the Lake Okeechobee Water Quality Model. Negative values are fluxes out of the water column; positive values are fluxes into the water column. Double-sided arrows indicate diffusive fluxes Solid arrows represent transport. Dashed arrows represent transformations.

year. This sediment recycling supports the supply of IP back to the water column (see below). To completely bury the top 6 cm of sediment P (consisting of OP, AP and IP of masses 2,744, 23, and 5,653 mg P/m^2 respectively) based on the burial terms, (consisting of OP, AP and IP fluxes of 43, 0.4, and 89 mg/m^2/year respectively) would take approximately 63 years. This is consistent with the burial rate of 6 cm of sediments which is also approximately 63 years. This is a significant result that indicates P buried in the sediments can influence water column P for decades.

The total DIP flux to the water is 1,167 mg/m^2/year. This includes 533 mg/m^2/year of IP from net sediment fluxes, 164 mg/m^2/year from external loads and 470 mg/m^2/year from OP mineralization. The unidirectional decay of OP to IP is considered a load to the IP compartment, while recycling between AP and IP is not considered a load to the water column. The net flux of IP from the sediment (533 mg/m^2/year) is 46 percent of the total IP flux to the water column (1,167 mg/m^2/year) and is almost two times greater than the external load of TP to the lake (274 mg/m^2/year). This sediment flux is important because it continually supports algal growth.

A similar diagram for N was also developed (Fig. 8). This indicates that the net removal of N into the sediments is approximately 2,192 mg/m^2/year. Thus 50 percent of the total external loads (4,366 mg/m^2/year) are assimilated by the sediments on an annual basis. The external loads include 1,084 mg/m^2/year of N fixation, which is 25 percent of the external load and is close to the estimate of 33 percent by Phlips and Ihnat (1995). This net assimilation of N by the sediments (the sum of all fluxes to and from the sediments) is primarily in the form of algal N (AN, 1,345 mg/m^2/year) and non-living ON (3,620 mg/m^2/year). Sediment ON (99,850 mg/m^2) is recycled to IN (1,609 mg/m^2/year) at the rate of 1.6 percent by mass per year, while AN (130 mg/m^2) is recycled to IN (483 mg/m^2/year) at the rate of 371 percent per year. This recycling supports the supply of IN back to the water

column (see below). As with P the turnover rate of sediment N (99,850; 130; 397 mg/m^2 of ON, AN and IN respectively) due to burial (1,574; 2.1, 5.6 mg/m^2/year of ON, AN and IN respectively) is 63 years.

The total DIN flux to the water column is 5,893 mg/m^2/year. This includes 2,773 mg/m^2/year of IN from net sediment fluxes, 811 mg/m^2/year from external loads and 2,309 mg/m^2/year from ON mineralization. The flux of IN from the sediment (2,773 mg/m^2/year) is 47 percent of the total IN flux to the water column (5,893 mg/m^2/year) and is equivalent to 63 percent of external load of TN to the lake (4,366 mg/m^2/year).

Discussion

Sediment water interactions in this shallow lake are important because these generate a significant source of available P (Fisher *et al.* 2001, Moore *et al.* 1998) to the water column and the resuspension of sediment results in reduced light penetration, inhibiting algal growth (Phlips *et al.* 1995a, Phlips *et al.* 1993, Phlips *et al.* 1995c). The model indicates that the combination of resuspension and diffusion contributes approximately twice the amount of P from sediments to the water column as external loads (Fig. 6). Similarly, the model indicates there is a large sediment flux of IN equivalent to 63 percent of the external load. These fluxes of nutrients reflect the large storage of P and N in Lake Okeechobee sediments of 28,700 and 381,000 tons respectively (Reddy *et al.* 1995) and demonstrate the importance of sediment-water interactions on the nutrient dynamics of Lake Okeechobee. This large nutrient pool can also influence lake response to external load reductions (Bierman and James 1995).

Phosphorus

Evidence for the role of sediments in controlling lake response to external P load reductions is well documented, especially for highly eutrophic lakes (*e.g.*, Shagawa Lake, USA, [Larsen *et al.* 1981], Lake Sammamish, USA, [Welch *et al.* 1986], and Lake Varese, Italy, [Rossi and Premazzi 1991]). Thus to predict lake response to external load reductions, sediment nutrient reservoirs and their interaction with the water column must be considered.

A fundamental concept underlying studies of sediment-water P interactions is that seston and algae deposited to the sediments consists of OP material exhibiting a wide range of resistance to decomposition. Readily degradable particulate OP fractions undergo transformation to IP and become available for potential release to the water column, whereas more resistant particulate OP fractions are largely retained and eventually buried to the deep sediments. Determination of the nature and abundance of the readily degradable fraction is important in predicting the magnitude and time-course of sediment P release rates in lakes (*c.f.* Penn *et al.* 1995).

In sediment P models that include only one OP state variable, such as the previous versions of the LOWQM, all OP is potentially degradable. These models may have reduced predictive ability on decadal time scales due to this limitation. In long-term dynamic simulations, such models cannot accurately describe both the short-term behavior due to decomposition of the readily degradable OP fraction and the longer-term behavior due to decomposition of the moderately- to non-degradable fractions.

While natural marine organic matter is made of hundreds if not thousands of different compounds, each having its own reactivity toward decay (Boudreau 1992), sediment model frameworks representing three degradability classes successfully describe the diagenesis process in marine and freshwater sediments (*e.g.*, Westrich and Berner 1984). Representation of three reactivity classes of OP is practical because they can be operationally defined and their respective decay rates can be measured.

The enhancement to represent these three reactivity classes of OP and to also include dissolved OP was reasonably achieved, with water column TP and DIP being well calibrated and validated (Table 1), and the various classes of sediment P being within a standard deviation of the observed values (Table 2). The large variation of the observed sediment P classes is attributed to spatial variation in the lake, which is discussed below.

A major assumption of the LOWQM is that sediments below 6 cm act only as a sink, not a source of material to the lake. No materials were allowed to communicate upward from these deep sediments. Support for this assumption comes from three independent sources, 1) Brezonik and Engstrom (1998) found highest TP concentrations in the top 5 cm of sediment, 2) Kirby *et al.* (1994) found sub-millimeter laminations in mud sediments that disappeared toward the top of the sediments suggesting that resuspension only affects the upper few cm of sediments, and 3) sediment cores taken after Hurricane Irene passed over the lake in October 1999 did not show signs of major disturbance below the first 5 cm of sediments. (John White, Research Scientist, University of Florida, personal communication).

Another enhancement of the LOWQM was to implicitly simulate aerobic surface sediments and anaerobic subsurface sediments. DIP levels in Lake Okeechobee sediments were found to be lowest in the oxidized surface sediments, increasing from less than 0.1 mg/l to approximately 1.1 mg/l at 10 cm in anaerobic sediments (Moore and Reddy 1994). The model results also replicate this trend, with dissolved P in the surface of 0.06 mg/l and 0.36 mg/l in the subsurface sediments (results not shown).

Nitrogen

N-fixation by cyanobacteria was implemented in this model because independent measurements indicated that this process was responsible for approximately a third of the N load to the lake (Phlips and Ihnat 1995). The model predictions were similar, with about 25 percent of the N load coming from N fixation to the water column. TN was well calibrated and validated and DIN was well calibrated and validated with the exception of negative LMEs. This failure can be attributed to the N-fixation switch used for cyanobacteria. The lower limit of the predicted DIN is at the concentration where the N-fixation switch is turned on (Fig. 5). A lower NFIX would improve the LME but would also produce a more negative %BIAS (Table 1). The NFIX value used (0.10 mg/l) was chosen to reflect the overall trends more so than replicating the full range of observed seasonal variation in DIN.

Silica

As noted above, parameters that are involved in the silica cycle did not have any influence on nutrients and algae (data not shown). Both observed and computed values for DSI, for both the calibration and validation periods (Table 1), are far in excess of values that would limit diatom growth (*e.g.*, Ksi = 0.050 mg Si/l, Appendix 4). Clearly there is not a full understanding of silicon dynamics in the lake and/or there are errors in external silicon loading estimates; however, these issues have little impact on the model calibration results.

Algae

The enhancement to the model of three separate groups of algae was implemented to improve the description of algal dynamics. The rationale in the attempt is predicated on three observations: 1) Diatoms were the dominant algal species in Lake Okeechobee in the 1970s (Havens *et al.* 1996), 2) Cyanobacteria have become dominant since then resulting in an algal bloom of *Anabaena circinalis* that covered over 310 square km of Lake Okeechobee in August of 1986, (Jones 1987), and 3) Studies have also shown that N-fixation by cyanobacteria is a major pathway of N loads to the lake (Phlips and Ihnat 1995). This enhancement has provided some further understanding of the algal nutrient dynamics in Lake Okeechobee.

The three groups of algae represent approximately 93 % of the algal biomass in Lake Okeechobee (Havens *et al.* 1996). The poor LME result in the validation period could be attributed to two causes. 1) The observed error measurement is probably greater than calculated because it is based on the variability among carbon estimates only; it does not include the error from equations used to convert biovolume to carbon. If the observed error is greater than measured, it is possible that the LME would be positive. 2) The validation data are sparser over time (on average less than 6 samples/year) than the calibration data (on average more than 9 samples/year). The validation data set may not capture the seasonal variability that was portrayed in the model calibration results.

Sensitivity Analysis

Model results show that the three algal groups are extremely sensitive to factors that affect their own growth rate and factors that affect the other algal groups (Table 3). This is most true for green algae and diatoms, in part because they represent less than half the biomass of the cyanobacteria. Due to competition among the three species, if cyanobacteria biomass is reduced by half, diatoms could more than double and green algae could increase an order of magnitude. Algal groups were also more sensitive to light than supply of inorganic nutrients. Algal carbon as a whole was less sensitive than the individual groups because it is constrained by light, temperature and nutrient supply rates. The relationships among individual algal groups are less constrained because they are a function of differences among the algal groups with respect to process rates, temperature dependencies, and competitive advantages. This is consistent with the observed strong light limitation observed in Lake Okeechobee due to resuspended sediments (Aldridge *et al.* 1995, Phlips *et al.* 1995a).

Despite the strong sensitivity of algal groups, the inorganic and total N and P were less sensitive than algae to most changes in algal dynamics and parameters affecting nutrient supply. This can be attributed to the non-linear nature of this model that reflects the nutrient cycle, algal dynamics, and sediment water interactions. For example, as supply of IP is increased, algae respond by increased growth, absorbing a majority of that supply. Thus algae standing stock increases faster than IP standing stock as IP supply increases. The results from the mass budgets indicate that algal P and N constitute only 12 and 5 percent, respectively, of the TP and TN in the water column. Consequently, changes in algal concentrations and/or dynamics should not be expected to cause large changes in water column TP or TN concentrations. This observation is consistent with the sensitivity analysis.

The sensitivity analysis of DIP and TP indicates potential management opportunities that could benefit the nutrient conditions of Lake Okeechobee while reducing cyanobacteria biomass (Table 3). These include increased sediment partition coefficient; reduced resuspension; reduced DIP in the sediments; and reduced P load. These were explored in the Lake Okeechobee Sediment Management Feasibility Study by BBL (2003), which looked at chemical treatments, such as alum, that could increase the partition coefficient in the sediments and reduce resuspension of solids, and dredging that could reduce the DIP in sediments. Chemical treatment was estimated to reduce the time to reach a goal of achieving a

lake-wide water column TP concentration of 0.04 mg P/l to 15 years earlier than by external load reduction alone. Dredging was estimated by engineers to leave a veneer of mud sediments that, based on the time frame required to implement dredging, will not accelerate any improvement in water quality conditions beyond the rate of recovery predicted without dredging, and may actually exacerbate the recovery. Based on these model results and other comprehensive technical and economic evaluations of the scenarios, neither scenario was recommended as the preferred plan, and the management focus will remain on reduction of external P loads.

Surprisingly, neither TP nor TN was sensitive to a potentially important parameter, burial out of the system. A simple explanation is that the sediments contain so much P and N that the burial rate does not remove a significant amount of P, even in this 18-year simulation. This can be attributed to the long turnover time in the sediments of approximately 63 years. If the simulation were extended to 100 years, both N and P could potentially be impacted by the sediment burial rate.

Nutrient Fluxes

The current LOWQM provides predictions of internal loadings of sediments to the water column (Fig. 6). The net flux of TP is from the water column to the sediments (163 mg/m^2/year), with a majority of that being in the form of organic material. The model also predicts a significant flux of available IP from the sediment to the water column that is twice the external load of P to the lake. The model is consistent with previous research of the lake that indicates that sediments are a net sink for total P (Havens and James 1997), yet they are a net source of inorganic P to the water column (Moore *et al.* 1998).

Predicted fluxes show that sediments are a net sink for TN while they are a net source of IN to the water column. Loading of IN to the water column from the sediment is 47 percent of total IN loading to the water column. This is approximately the same percentage as the loading of inorganic P to the water column from the sediment as compared to the total IP loading to the water column (46 percent). One difference is that the inorganic P consists of both particulate (sorbed to solids) and dissolved forms, while most of the IN is immediately available; however, particulate P can become available via desorption in the water column. Loading of IN to the water column from the sediment is 64 percent of the total external N loading (including N fixation). This is much lower than the corresponding ratio (195 percent) for P. Relative to external loadings to the lake, the sediments contribute a much higher internal loading of P than of N.

The addition of N-fixing cyanobacteria provides a more accurate representation of the N cycle in Lake Okeechobee. The model predicted N fixation is a significant net load (25 percent) of new N to the lake. It is consistent with estimates based on independent experimental measurements (Phlips and Ihnat 1995). It is also consistent with observations from lakes such as Okeechobee that are highly enriched in P, and in which N is relatively more limiting to algal growth than P (Aldridge *et al.* 1995). From a management standpoint, however, it is not possible to directly control N-fixation. The most effective strategy is to reduce P loadings to reduce total algal concentrations and to restore a more balanced condition between P and N limitation, and a more balanced ecosystem, with lower concentrations of N-fixing cyanobacteria.

Conclusions

The predictions from this calibrated and validated model are consistent with a number of observations of Lake Okeechobee. These include cyanobacteria dominance, significant DIP flux from sediments (Fig. 6), significant N-fixation (Fig. 7) and the importance of light limitation on the lake (Table 3). An interesting result is the low sensitivity of TP and TN to changes in external loads of these nutrients. This is a result of the balance in the model between external and internal (sediment) sources and recycling of the nutrients through the algal groups. All sensitivity analyses were conducted only for the 18 year application period. As discussed above, the turnover time for TP and TN reservoirs in the top 6 cm of sediments is estimated at 63 years, which is almost 4 times the simulation length. These large sediment pools along with the strong sediment water interactions reduce the sensitivity of water column TP and TN concentrations to changes in external loading rates over the 18 year application period. This insensitivity will be the topic of a separate paper on the LOWQM which will involve use of the model to conduct load-response analyses.

Given that it would be difficult to manage the internal cycle of P through chemical treatment and dredging, the only other viable alternative is external P load reduction. The sensitivity analysis indicates a similar response of algal groups, in-lake TP or DIP concentrations to P load reduction than. Since the ultimate goal is to reduce algal blooms, this alternative appears quite viable.

Future Development

Three concerns for future development of modeling for Lake Okeechobee are spatial variability and emergent and submergent aquatic vegetation. This spatially-averaged model cannot replicate spatial variability observed in the water column and sediments of the lake. Water quality dynamics have been shown to vary among the different regions of the lakes (Havens *et al.* 1995b, Phlips *et al.* 1995b, Phlips *et al.* 1993, Phlips *et al.* 1995c, Schelske 1989), and lake response to external forcings should be viewed within a

regional context (Phlips *et al.* 1993). Also, the model does not explicitly represent two important biological pathways within the lake, submergent and emergent vegetation, which comprise approximately 20 percent of the lake area (Phlips *et al.* 1995b). The lowering of the lake levels and the subsequent increase in submerged aquatic vegetation is likely the cause for the improvement in water quality observed in the summer and fall of 2000 due to increased light penetration because of lower lake levels, and reduced nutrient levels because of increased submerged aquatic vegetation (SAV) growth and subsequent competition for nutrients between SAV and algae (Havens *et al.* 2001).

To address these deficiencies, the information and parameters developed in the LOWQM are being used to assist the development of a spatially explicit 3-D hydrodynamic, sediment and water quality model. This spatially explicit model, the Lake Okeechobee Ecosystem Model (LOEM), is based on the current three-dimensional Lake Okeechobee Hydrodynamic and Sediment Transport Model (Jin *et al.* 2000). The LOEM is currently being developed to define impacts of water levels and major storm events on algal blooms, P, and the submerged aquatic vegetation on the lake. The LOEM will run at shorter time scales (less than 10 years) and is being calibrated and validated to more extensive (spatially and temporally), short term (5 years or less) data sets.

Acknowledgments

This work was funded partially through two contracts: the first between the South Florida Water Management District and AScI corporation (C-5236), and the second between the South Florida Water Management District and Limno-Tech Inc (C-7613). The authors thank Curt Pollman, Karl Havens, Susan Gray, and Todd Tisdale and three anonymous reviewers for their review of a previous version of this manuscript.

References

Aldridge, F.J., E.J. Phlips and C.L. Schelske. 1995. The use of nutrient enrichment bioassays to test for spatial and temporal distributions of limiting factors affecting phytoplankton dynamics in Lake Okeechobee, Florida. Pages 177-190 *In* N.G. Aumen and R.G. Wetzel, eds. Advances in Limnology. vol. 45. Schweizerbart, Stuttgart, Germany.

Ambrose, R.B., Jr., T.A. Wool and J.L. Martin. 1993a. The Water Quality Analysis Simulation Program, WASP5. Part B: The WASP5 Input Dataset. Version 5.00. U.S. Environmental Protection Agency, Center for Exposure Assessment Modeling,, Athens, Georgia.

Ambrose, R.B., Jr., T.A. Wool and J.L. Martin. 1993b. The Water Quality Analysis Simulation Program, WASP5; Part A: Model Documentation. U.S. Environmental Protection Agency, Environmental Research Laboratory, Athens, GA.

Aumen, N.G. and R.G. Wetzel, eds. 1995. Advances in Limnology Ecological studies of the littoral and pelagic systems of Lake Okeechobee, Florida (USA). vol. 45. Schweizerbart, Stuttgart, Germany.

Bierman, V.J., Jr. and R.T. James. 1995. A preliminary modeling analysis of water quality in Lake Okeechobee, Florida: diagnostic and sensitivity analyses. Water Research 29:2767-2775.

Bierman, V.J., Jr. and D.M. Dolan. 1986a. Modeling of phytoplankton in Saginaw Bay: I. Calibration Phase. Journal of Environmental Engineering 112:400-414.

Bierman, V.J., Jr. and D.M. Dolan. 1986b. Modeling of phytoplankton in Saginaw Bay: II. Post Audit Phase. Journal of Environmental Engineering 112:415-429.

BBL. 2003. Lake Okeechobee Sediment Management Feasibility Study. Report Submitted to the South Florida Water Management District, West Palm Beach FL. Contract number C-11650. Prepared by Blasland, Bouck and Lee, Inc. (BBL).

Boudreau, B.P. 1992. A kinetic model for microbic organic-matter decomposition in marine sediments. FEMS Microbiology Ecology 102:1.

Bowie, G.L., W.B. Mills, D.B. Porcella, C.L. Campbell, J.R. Pagenkopf, G.L. Rupp, K.M. Johnson, P.W.H. Chan, S.A. Gherini and C.E. Chamberlain. 1985. Rates, Constants, and Kinetics Formulations in Surface Water Quality Modeling (Second Edition). U.S. Environmental Protection Agency, Environmental Research Laboratory, Athens, Georgia.

Brezonik, P.L. and D.R. Engstrom. 1998. Modern and historic accumulation rates of phosphorus in Lake Okeechobee, Florida. Journal of Paleolimnology. 20:31-46.

Cichra, M.F., S. Badylak, N. Henderson, B.H. Rueter and E.J. Phlips. 1995. Phytoplankton community structure in the open water zone of a shallow subtropical lake (Lake Okeechobee, Florida USA). Pages 157-175 *In* R.G. Wetzel, ed. Ecological studies of the littoral and pelagic systems of Lake Okeechobee, Florida (USA). vol. 45. Schweizerbart, Stuttgart, Germany.

DiToro, D.M. and J.J. Fitzpatrick. 1993. Chesapeake Bay Sediment Flux Model. US.E.P.A., Annapolis, MD.

Fisher, M.M., K.R. Reddy and R.T. James. 2001. Long-term changes in the sediment chemistry of a large shallow subtropical lake. Lake and Reservoir Management 17:217-232.

Haan, C.T. and J. Zhang. 1996. Impact of uncertain knowledge of model parameters on estimated runoff and phosphorus loads in the Lake Okeechobee Basin. Transactions of the American Society of Agricultural Engineers 39:511-516.

Havens, K. and R.T. James. 2005. The phosphorus mass balance of Lake Okeechobee, Florida: implications for eutrophication management. Lake and Reserv. Manage. 21(2):139-148.

Havens, K.E., N.G. Aumen, R.T. James and V.H. Smith. 1996. Rapid ecological changes in a large subtropical lake undergoing cultural eutrophication. Ambio 25:150-155.

Havens, K.E., C. Hanlon and R.T. James. 1995a. Historical trends in the Lake Okeechobee ecosystem V. algal blooms. Archiv für Hydrobiologie Suppl 107:89-100.

Havens, K.E., C. Hanlon and R.T. James. 1995b. Seasonal and spatial variation in algal bloom frequencies in Lake Okeechobee, Florida, U.S.A. Lake and Reservoir Management 10:139-148.

Havens, K.E. and R.T. James. 1997. A critical evaluation of phosphorus management goals for Lake Okeechobee, Florida, USA. Lake and Reservoir Management 13:292-301.

Havens, K.E., K.-R. Jin, A.J. Rodusky, B. Sharfstein, M.A. Brady, T.L. East, N. Iricanin, R.T. James, M.C. Harwell and A.D. Steinman. 2001. Hurricane effects on a shallow lake ecosystem and its response to a controlled manipulation of water level. The Scientific World 1:44-70.

James, R.T. and V.J. Bierman, Jr. 1995. A preliminary modeling analysis of water quality in Lake Okeechobee, Florida: calibration results. Water Research 29:2755-2766.

James, R.T., B.L. Jones and V.H. Smith. 1995a. Historical trends in the Lake Okeechobee ecosystem II. nutrient budgets. Archiv für Hydrobiologie Suppl. 107:25-47.

James, R.T., J. Martin, T. Wool and P.F. Wang. 1997. A sediment resuspension and water quality model of Lake Okeechobee. Journal of the American Water Resources Association 33:661-680.

James, R.T., V.H. Smith and B.L. Jones. 1995b. Historical trends in the Lake Okeechobee Ecosystem III. water quality. Archiv für Hydrobiologie Suppl. 107:49-69.

Janssen, P.H.M. and P.S.C. Heuberger. 1995. Calibration of process-oriented models. Ecological Modelling 83:55-66.

Jin, K.-R., J.H. Hamrick and T. Tisdale. 2000. Application of three-dimensional hydrodynamic model for Lake Okeechobee. Journal of Hydraulic Engineering 126:758-771.

Jin, K.-R. and Z.G. Ji. 2001. Calibration and verification of a spectral wind-wave model for Lake Okeechobee. Ocean Engineering 28:571-584.

Jones, B. 1987. Lake Okeechobee eutrophication research and management. Aquatics 9:21-26.

Kajiura, K. 1968. A model of the bottom boundary layer in water waves. Bulletin of the Earthquake Research Institute 46: 75 123.

Kirby, R., C.H. Hobbs and A.J. Mehta. 1994. Shallow stratigraphy of Lake Okeechobee, Florida: a preliminary reconnaissance. Journal of Coastal Research 10:339-350.

Larsen, D.P., D.W. Schults and K.W. Malereg. 1981. Summer internal phosphorus supplies in Shagawa Lake, Minnesota. Limnology and Oceanography 26:740-753.

Maceina, M.J. and D.M. Søballe. 1990. Wind-related limnological variation in Lake Okeechobee, Florida. Lake and Reservoir Management 6:93-100.

Mehta, A.J. 1991. Lake Okeechobee Phosphorus Dynamics Study. Vol. IX. West Palm Beach, Florida. Final report submitted to the South Florida Water Management District, Contract Number C91-2393.

Moore, P.A.J. and K.R. Reddy. 1994. Role of Eh and pH on phosphorus geochemistry in sediments of Lake Okeechobee, Florida. Journal of Environmental Quality 23:955-964.

Moore, P.A.J., K.R. Reddy and M.M. Fisher. 1998. Phosphorus flux between sediment and overlying water in Lake Okeechobee, Florida: spatial and temporal variations. Journal of Environmental Quality 27: 1428-1439.

Olila, O.G., K.R. Reddy and W.G. Harris. 1995. Forms and distribution of inorganic phosphorus in sediments of two shallow eutrophic lakes in Florida. Hydrobiologia 302:147-161.

Penn, M.R., M.T. Auer, E.L.V. Orman and J.J. Korienek. 1995. Phosphorus diagenesis in lake sediments: investigations using fractionation techniques. Marine and Freshwater Research 46:89-99.

Phlips, E.J., F.J. Aldridge and C. Hanlon. 1995a. Potential limiting factors for phytoplankton biomass in a shallow subtropical lake (Lake Okeechobee, Florida, USA). Pages 137-155 *In* N.G. Aumen and R.G. Wetzel, eds. Advances in Limnology. vol. 45. Schweizerbart, Stuttgart, Germany.

Phlips, E.J., F.J. Aldridge and P. Hansen. 1995b. Patterns of water chemistry, physical and biological parameters in a shallow subtropical lake (Lake Okeechobee, Florida, USA). Pages 117-135 *In* N.G. Aumen and R.G. Wetzel, eds. Advances in Limnology. vol. 45. Schweizerbart, Stuttgart, Germany.

Phlips, E.J., F.J. Aldridge, P. Hansen, P.V. Zimba, J. Ihnat, M. Conroy and P. Ritter. 1993. Spatial and temporal variability of trophic state parameters in a shallow subtropical lake (Lake Okeechobee, Florida, USA). Archiv für Hydrobiologie 128:437-458.

Phlips, E.J., F.J. Aldridge, C.L. Schelske and T.L. Crisman. 1995c. Relationships between light availability, chlorophyll *a*, and tripton in a large, shallow subtropical lake. Limnology and Oceanography 40: 416-421.

Phlips, E.J. and J. Ihnat. 1995. Planktonic nitrogen fixation in a shallow subtropical lake (Lake Okeechobee, Florida, USA). Pages 191-201 *In* N.G. Aumen and R.G. Wetzel, eds. Advances in Limnology. vol. 45. Schweizerbart, Stuttgart, Germany.

Reddy, K.R. 1991. Lake Okeechobee phosphorus dynamics study: Physico-chemical properties in the sediments. Vol. II. South Florida Water Management District, West Palm Beach, FL. Final report submitted to the South Florida Water Management District, West Palm Beach, Florida, Contract Number C91-2393.

Reddy, K.R., Y.P. Sheng and B.L. Jones. 1995. Lake Okeechobee Phosphorus Dynamics Study. Vol. I. West Palm Beach, FL. Final report submitted to the South Florida Water Management District. West Palm Beach, Florida, Contract Number C91-2393.

Rossi, G. and G. Premazzi. 1991. Delay in lake recovery caused by internal loading. Water Research 25:567-575.

SAS Institute, Inc. 1989. SAS/STAT® Users Guide: Version 6, Fourth edition. SAS Institute, Cary, NC.

Schelske, C.L. 1989. Assessment of nutrient effects and nutrient limitation in Lake Okeechobee. Water Resources Bulletin 25:1119-1130.

Smith, V.H., V.J.B. Jr., B.L. Jones and K.E. Havens. 1995. Historical trends in the Lake Okeechobee ecosystem IV. nitrogen: phosphorus ratios, cyanobacterial dominance, and nitrogen fixation potential. Archiv für Hydrobiologie Suppl. 107:71-88.

Smits, J.G.C. and D.T. van der Molen. 1993. Application of SWITCH, a model for sediment-water exchange of nutrients, to Lake Veluwe in The Netherlands. Hydrobiologia 253:281-300.

Strathman, R.R. 1967. Estimating the organic carbon content of phytoplankton from cell volume or plasma volume. Limnology and Oceanography 12:411-418.

Thomann, R.V. and J.A. Mueller. 1987. Principles of Surface Water Quality Modeling and Control. Harper and Row Publishers, New York.

U.S. Army Coastal Engineering Research Center, 1984. Shore Protection Manual 1. U.S. Army Corps of Engineers, Fort Belvoir, Virginia.

Welch, E.B., D.E. Spyridakis, J.I. Shuster and R.R. Horner. 1986. Declining lake sediment phosphorus release and oxygen deficit following wastewater diversion. Journal of the Water Pollution Control Federation 58:92-96.

Westrich, J.T. and R.A. Berner. 1984. The role of sedimentary organic matter in bacterial sulfate reduction: The G model tested. Limnology and Oceanography 29:236 - 237.

Appendix I – Equations

Modifications from the original EUTRO5 to the current LOWQM included: 1) the addition of an inorganic suspended solids (ISS) state variable and related interactions, 2) the partitioning of organic phosphorus into 4 components, 3) the partitioning of algae into three groups to provide the ability to simulate nitrogen fixing cyanobacteria, diatoms, and green algae, and 4) silica and the related interactions. A thorough description of 1 is given in James *et al.* (1997). All modifications are summarized below:

Solids:

In this LOWQM, resuspension of solids from sediments to the water column could be represented by a simple wind-wave model that uses algorithms based on the shallow water Sverdrup-Munk-Bretschnieder (SMB) model for Cartesian grids (U.S. Army Coastal Engineering Research Center 1984) that computes the bottom shear stress at every grid point using the Bottom Boundary Layer (BBL) model (Kajiura 1968, James *et al.* 1997). Because wind data for the lake were incomplete for the 18 year period of the model, we modified the change in solids equation to use a forcing function to resuspend solids:

$$\frac{\partial M_{ISS}}{\partial t} = Q_{it} C_{ISS\,it} - Q_{et} C_{ISS} + \frac{W_{st}}{H_s} M_{ISS\,ss} - \frac{V_{ISS}}{H_w} M_{ISS} \tag{1}$$

$$\frac{\partial M_{ISS\,s}}{\partial t} = -\frac{W_{st}}{H_s} M_{ISS\,ss} + \frac{V_{ISS}}{H_w} M_{ISS} - [(C_{ISS_s} - C_{ISS_b}) SMIX_{ISS} \Theta_{SMIX}^{t-20} \frac{A_{sb}}{H_{sb}}] + L_{ISS} \tag{2}$$

$$\frac{\partial M_{ISSb}}{\partial t} = (C_{ISS_s} - C_{ISS_b}) SMIX_s \Theta_{SMIX}^{t-20} \frac{A_{sb}}{H_{sb}} - \frac{V_b}{H_b} M_{ISSb} \tag{3}$$

where

t = time (day)

M_{ISS}, M_{ISS_S}, M_{iss_B}, M_{iss_i} = inorganic solids mass in the water column, surface sediments, bottom sediments and inflowing water, respectively (g) at time t

C_{ISS}, C_{ISS_S}, C_{iss_B}, C_{iss_i} = inorganic solids concentrations in the water column, surface sediments, bottom sediments and inflowing water, respectively (mg/l, g/m^3)

Q_{it} = inflow at time t (m^3/day)

Q_{et} = outflow at time t (m^3/day)

W_{st} = the resuspension velocity at time t (m/day)

H_w, H_s, H_B = depth of the water column, surface sediment and bottom sediment (m) respectively

V_{ISS}, V_b = settling velocity of inorganic solids from the water column and burial rate from bottom sediments, respectively (m/day)

$SMIX$ = sediment mixing rate (m/day)

Θ_{SMIX}^{t-20} = temperature coefficient for sediment mixing rate (unitless)

A_{sb} = interfacial area between surface and bottom sediments (m^2)

L_{ISS} = make up load of inorganic solids that implicitly is not being simulated (*i.e.*, $CaCO_3$) and is added to the surface sediment of the model (g)

The impact of solids and algae on light penetration through the water column is computed from the Beer-Lambert law:

$$I = I_o e^{-K_H H} \tag{4}$$

where

I_o = the surface light intensity (langleys)

H = average water column depth (m)

K_H = light extinction coefficient (/m) related to solids and phytoplankton by the equation:

$$K_H = LCF_1 + LCF_2\, C_s + \sum_{i=1}^{3} C_{A_i} \frac{XKC_{A_i}}{CCHL_{A_i}} \tag{5}$$

where

LCF_1 = portion of extinction attributed to lake color and reflection (/m)

LCF_2 = proportional impact of sediment solids on the extinction coefficient (/(m•mg))

C_{A_i} = carbon concentration of phytoplankton group i (mg/l)

$CCHL_{A_i}$ = carbon to chlorophyll *a* ratio for phytoplankton group i (unitless)

XKC_{A_i} = light extinction impact of the chlorophyll *a* in phytoplankton group i (/(m•mg chlorophyll *a*).

The impact of solids on ammonium-nitrogen, dissolved inorganic phosphorus and dissolved silica is based upon user-specified partition coefficients. The equation is a linear relationship between dissolved (C_w, mg/l) and particulate (C_s, mg-chemical kg-solids^{-1}) forms that is given by (Karickhoff 1985):

$$C_s = K_{ps}\, C_w \tag{6}$$

where

K_{ps} = a partition coefficient (l/kg).

At equilibrium, the distribution among the dissolved and particulate phases is controlled by the partition coefficient Kpsx, where X represents the inorganic nutrient. The total mass of chemical in each phase is controlled by Kpsx and the amount of the solids present,

$$C = C_w n + C_s S \tag{7}$$

where

C = the total chemical concentration and n the porosity. Substituting equation 4 into 5 and rearranging terms gives the dissolved fraction f_D

$$f_D = \frac{C_w n}{C} = \frac{n}{n + K_{psx}\, S} \tag{8}$$

and the sediment sorbed fraction is

$$f_s = \frac{C_s S}{C} = \frac{K_{psx}\, S}{n + K_{psx}\, S} \tag{9}$$

In the LOWQM, these fractions are determined in time and space throughout a simulation from the partition coefficients, internally calculated porosities, and simulated solids concentrations.

Organic Phosphorus:

The equations that describe the kinetic modifications for the 4 classes of organic phosphorus state variables are listed below:

For OP_1 (labile organic P):

$$\frac{\partial M_{OP1}}{\partial t} = \sum_{i=1}^{3} M_{A_i}\lambda_{A_i}a_{pc_i}f_{OP}X_{OP1} - (1-\phi_{OP1})K_{OP1}\Theta_{OP1}^{T-20}\kappa_{prc}M_{OP1} - \frac{v_O}{H_w}M_{OP1} + \frac{W_{st}}{H_s}M_{OP1s} - M_{OP1}\phi_{OP1}K_{OP1}\Theta_{OP1}^{T-20}\kappa_{prc} + (Q_{it}C_{OP1i} - Q_{et}C_{OP1}) \tag{10}$$

For OP_2 (moderately labile organic P):

$$\frac{\partial M_{OP2}}{\partial t} = \sum_{i=1}^{3} M_{A_i}\lambda_{A_i}a_{pc_i}f_{OP}X_{OP2} - M_{OP2}(1-\phi_{OP2})K_{op2}\Theta_{OP2}^{T-20}\kappa_{prc} - M_{OP2}\frac{v_O}{H_w} + M_{OP2}\frac{W_{st}}{H_s} - M_{OP2}\phi_{OP2}K_{OP2}\Theta_{OP2}^{T-20}\kappa_{prc} + (Q_{it}C_{OP2i} - Q_{et}C_{OP2}) \tag{11}$$

For OP_3 (recalcitrant organic P):

$$\frac{\partial M_{OP3}}{\partial t} = \sum_{i=1}^{3} M_{A_i}\lambda_{A_i}a_{pc_i}f_{op}X_{OP3} - M_{OP3}\frac{v_O}{H_w} + M_{OP3}\frac{W_{st}}{H_s} + (Q_{it}C_{OP3i} - Q_{et}C_{OP3}) \tag{12}$$

For OP_4 (dissolved organic P):

$$\frac{\partial M_{OP4}}{\partial t} = \sum_{i=1}^{3} M_{A_i}\lambda_{A_i}a_{pc_i}f_{op}X_{OP4} - M_{OP4}K_{op4}\Theta_{OP4}^{T-20}\kappa_{prc} + (C_{OP4i} - C_{OP4})\frac{D_{ws}A_{ws}\Theta_D^{T-20}}{H_s} + \sum_{j=1}^{2} M_{OPj}\phi_{OPj}K_{OPj}\Theta_{OPj}^{T-20}\kappa_{prc} + (Q_{it}C_{OP41} - Q_{et}C_{OP4}) \tag{13}$$

Where

M_{OP1}, M_{OP2}, M_{OP3}, M_{OP4} = OP_1, OP_2, OP_3 and OP_4 mass, respectively, in water column (g P)

C_{OP1}, C_{OP2}, C_{OP3}, C_{OP4} = OP_1, OP_2, OP_3 and OP_4 concentrations, respectively in water column (g P/m)

M_{OP1s}, M_{OP2s}, M_{OP3s}, M_{OP4s} = OP_1, OP_2, OP_3 and OP_4 mass, respectively in sediment column (g P/m^3 of sediment)

C_{OP4s} = OP_4 concentration in the sediment column (g P/m^3 of pore water)

C_{OP1i}, C_{OP2i}, C_{OP3i}, C_{OP4i} = OP_1, OP_2, OP_3 and OP_4 concentrations, respectively in inflowing water column (g P/m^3)

M_{Ai} = algal groups i=1 to 3 Mass (g C)

λ_{Ai} = temperature corrected algal respiration plus death rate for algal group "i" (/d)

a_{pci} = algal carbon to phosphorus ratio for species "i" (mg P/mg C)

f_{op} = fraction of algal-derived phosphorus recycled to OP (unitless)

X_{opk} = fraction of algal-derived OP recycled to specific forms OP_k, k = 1 to 4 (unitless)

k_{opk} = mineralization rates for OP_k, k = 1 to 4 (/d)

ϕ_{opk} = mineralization efficiency factor for labile OP_k forms (k = 1 to 2) that represents dissolution to OP_4 or "DOP" (unitless)

θ_{opk} = mineralization rate temperature correction factor for OP_k, k = 1 to 4 (unitless)

C_{ai} = algal carbon concentration for group i = 1 to 3 (g C/m^3)

κ_{prcs} = half-saturation limitation on phosphorus mineralization (g C/m^3)

v_O = particulate settling rate for organic phosphorus (m/d)

H_w, H_s = water and sediment segment depth (m)

D_{ws} = diffusion coefficient (m^2/d)

A_{ws} = interfacial area between surface sediment and water column (m^2)

Θ_D^{T-20} = temperature coefficient for diffusion (unitless)

For the top sediments the signs for the resuspension, settling and diffusion components are reversed. Additional terms representing mixing, burial and diffusion (S_{mix}, V_b, D_{ss}, respectively) between the bottom and surface sediments is also added. Finally a burial term (V_b) is added to the bottom sediment that represents material that is permanently removed from the system.

Algal Groups:

The changes in mass of the algal groups are defined by the following equation:

$$\frac{\partial M_{A_i}}{\partial t} = M_{A_i}(G_{1\max}\Theta_1^{t-20}G_I G_N) - M_{A_i}(k_{id} + k_{ir}\Theta_{ir}^{t-20}) - M_{A_i}\frac{V_{sA_i}}{H_w} + M_{A_{is}}\frac{W_{st}}{H_s} + (Q_{it}C_{A_{ii}} - Q_{et}C_{A_i}) \tag{14}$$

Where:

C_{Ai}, C_{Aii} = concentration of algal species i at time t in the water column (g/m^3) and inflowing water (g/m^3), respectively.

M_{Ai} = mass of algal species i at time t in the water column (g C).

k_{id} = non predatory mortality (/day) for phytoplankton group i.

k_{ir} = respiration rate of phytoplankton group i at 20°C (/day)

Θ_{ir}^{t-20} = temperature correction function for respiration of algal group i (unitless).

V_{sA_i} = average settling velocity of algal group I (m/day).

$G_{1\max}$ = maximum growth rate at 20°C (/day)

Θ_1^{t-20} = temperature correction for maximum growth rate (unitless)

G_I = growth rate reduction integrated over depth (unitless, Smith 1980):

$$G_I = \frac{e^1}{K_d H}\left[e^{-\frac{I_o}{I_{si}}e^{-K_d H}} - e^{-\frac{I_o}{I_{si}}}\right] \tag{15}$$

where

I_{si} = saturating light intensity for algal group i

G_N = growth rate reduction due to nutrients and is based on the minimum of three Michaelis Menten uptake kinetic parameters:

$$G_N = MIN[MM_{pi}, MM_{Ni}, MM_{SIi}] \tag{16}$$

where

$$MM_{Pi} = \frac{C_{DIP}}{C_{DIP} + K_{MPi}} \tag{17}$$

where

C_{DIP} = the concentration of DIP (g/m^3)

K_{MPi} = half saturation constant of P for algal group i (g/m^3)

$$MM_{Ni} = \frac{C_{DIN}}{C_{DIN} + K_{Mi}}, or\ 1\ \ if\ \ (C_{DIN} < NFIX) \tag{18}$$

where

C_{DIN} = the concentration of DIN (g/m^3)

K_{MNi} = half saturation constant of N for algal group i (g/m^3)

NFIX = switch that turns on nitrogen fixation (*i.e.*, nitrogen no longer limiting) (g/m^3)

$$MM_{SIi} = \frac{C_{DSI}}{C_{DSI} + K_{MSIi}} \tag{19}$$

where

C_{DSI} = the concentration of DSI (g/m^3)

K_{MSIi} = half saturation constant of SI for algal group i (g/m^3)

Silica Cycle:

$$\frac{\partial M_{DSI}}{\partial t} = \sum_{i=1}^{3} M_{A_i} a_{SIC_i} (\lambda_{A_i} f_{DSI} - G_{1\max} \Theta_1^{t-20} G_I G_N) + M_{PSI} K_{PSI} \Theta_{PSI_4}^{T-20} + (C_{DSI_s} - C_{DSI}) \frac{D_{ws} A_{ws} \Theta_D^{T-20}}{H_s} + (Q_{it} C_{DSI\,i} - Q_{et} C_{DSI}) \tag{20}$$

$$\frac{\partial M_{PSI}}{\partial t} = \sum_{i=1}^{3} M_{A_i} a_{SIC_i} (\lambda_{A_i} (1 - f_{DSI})) - M_{PSI} K_{PSI} \Theta_{PSI_4}^{T-20} - M_{PSI} \frac{V_{PSI}}{H_w} + M_{PSI_s} \frac{W_{st}}{H_s} + (Q_{it} C_{PSI\,i} - Q_{et} C_{PSI}) \tag{21}$$

Where

C_{DSI}, C_{PSI}, C_{DSI_i}, C_{PSI_i}, C_{DSI_s}, C_{PSI_s} = concentration of dissolved and particulate silica in the water column, inflowing water and sediments, respectively (g/m^3).

M_{DSI}, M_{PSI}, M_{DSI_s}, M_{PSI_s} = mass of dissolved and particulate silica in the water column and sediments, respectively (g).

a_{CSIi} = carbon to silica ratio in algal species i. (unitless)

f_{DSI_i} = fraction of algal silica that recycles to the dissolved pool (unitless)

K_{PSI} = particulate silica dissolution rate (day^{-1})

Θ_{KPSI}^{T-20} = temperature correction for particulate silica dissolution rate (unitless)

Appendix 2 – Organic Phosphorus Mineralization Parameters

Parameter	Organic Phosphorus Class					
	Readily Degradable	Moderately Degradable	Non-Degradable	Dissolved	Description	Reference
i	1	2	3	4		
K_{OPi}	0.031	0.0034	0.000	0.031	Mineralization rate of organic P (/day)	Calibration, Bowie (1985)
K_{SOPi}	0.014	0.0014	0.000	0.014	Mineralization rate of organic P in sediments (/day)	Calibration, Reddy (1991)
V_O	6.7	6.7	6.7		Settling of organic nutrients (cm/day)	Calibration
X_{OPi}	0.60	0.10	0.10	0.20	Fraction of algal P that is recycled to this form of organic P (mg P/mg Algal P)	Calibration
Θ_{OPi}	1.06	1.06	N/A	1.06	Temperature coefficient for mineralization (unitless)	Bowie (1985)
Θ_{SOPi}	1.06	1.06	N/A	1.06	Temperature coefficient for mineralization in sediments (unitless)	Bowie (1985)
Φ_{OPi}	0.15	0.15	N/A	N/A	Fraction mineralized to dissolved organic P in water column (mg dissolved OP/mg organic P)	Calibration
Φ_{SOPi}	0.15	0.15	N/A	N/A	Fraction mineralized to dissolved organic P in sediments (mg dissolved OP/mg organic P)	Calibration

N/A – Not applicable

Appendix 3 – Additional Parameters

Parameter	value	Parameter Discription	Reference
D_{SS}	8.00E-05	Diffusion rate between surface and bottom sediments (m^2/sec)	Calibration
D_{WS}	8.00E-05	Diffusion rate between water column and sediment segment (m^2/sec)	Reddy (1991b)
F_{DON}	0.0020	Fraction of organic nitrogen that is dissolved (mg DON/mg ON)	Reddy (1991)
K_{BOD}	0.5	Half saturation coefficient of oxygen limitation (mg O_2/L) for the decay rate of organic carbon	Ambrose *et al.* (1988)
K_{DC}	0.1	Carbonaceous deoxygenation rate constant (/day)	Ambrose *et al.* (1988)
K_{DENIT}	0.14	Maximum denitrification rate at 20°C (/day)	Bowie *et al.* (1985)
K_{PSPAV}	1000	Partition coefficient between soluble reactive phosphorus and anaerobic sediments Kg/ L	Comparison of dissolved and particulate inorganic phosphorus in sediments (Reddy 1991)
K_{PSPOV}	6000	Partition coefficient between soluble reactive phosphorus and aerobic sediments Kg/ L	Comparison of dissolved and particulate inorganic phosphorus in sediments (Reddy 1991)
K_{NIT}	0.40	Maximum nitrification rate at 20°C (/day)	Calibration
K_{NO3}	0.1	Denitrification half saturation factor (mg O_2/L)	Ambrose *et al.* (1988)
K_{ON}	0.0045	Maximum Organic N Mineralization at 20°C (/day)	Calibration
K_{ONs}	0.000045	Maximum organic N mineralization at 20°C in the sediment (/day)	Calibration
K_{OXNIT}	1	Nitrification half saturation (mg/ L)	Ambrose *et al.* (1988)
K_{PSi}	0.1	Solubilization of particulate silica (/day)	Calibration
K_{PSN}	30	Partition coefficient between ammonium and sediment (Kg/L)	Comparison of dissolved and particulate ammonia in sediments (Reddy 1991)
K_{PSSI}	110	Partition coefficient between available silica and sediment (Kg/L)	Calibration
L_{DIP}	72.5	Atmospheric load of inorganic P (Kg/day)	Ratio of SRP to TP averaged concentration in wet rainfall * 35 metric tons (FDEP 2000)
L_{ISS}	1.13E+06	Additional load of solids (as $CaCO_3$) to surface sediment (Kg/day)	Brezonik and Engstrom. (1998)
L_{NH4}	661.7	Atmospheric load of ammonia (Kg/day)	Ratio of NH4 to TP averaged concentration in wet rainfall * 35 metric tons (FDEP 2001)
L_{NOX}	714.6	Atmospheric load of nitrate+nitrite (Kg/day)	Ratio of NOX to TP in averaged concentration in wet rainfall * 35 metric tons (FDEP 2001)

Appendix 3 – Additional Parameters (continued)

Parameter	value	Parameter Discription	Reference
L_{ON}	2003	Atmospheric load of organic nitrogen (Kg/day)	Ratio of Organic Nitrogen to TP averaged concentration in wet rainfall * 35 metric tons (FDEP 2001)
L_{OP}	23.3	Atmospheric load of organic phosphorus (Kg/day)	Ratio of Organic Phosphorus to TP averaged concentration in wet rainfall * 35 metric tons (FDEP 2001)
OCRB	2.67	Oxygen to carbon ratio (mg O_2/mg C)	Ambrose *et al.* (1988)
SMIX	0.03	Solids mixing between sediment segments (cm/day)	Calibration
Vb	0.095	Burial rate of sediments (cm/year)	Brezonik and Engstrom (1998)
Viss	59.3	Settling of suspended solids (cm/d)	Mehta (1991)
Hss	0.215	Light extinction coefficient due to suspended solids (/m•mg ISS/L)	Relation of Secchi depth and suspended solids
Hw	2.83	Light extinction coefficient due to color in water (/m)	Relation of Secchi depth and suspended solids
Θ_D	1.06	Temperature coefficient for diffusion rates (unitless)	Calibration
Θ_{DC}	1.06	Temperature coefficient for carbonaceous deoxygenation (unitless)	Ambrose *et al.* (1988)
Θ_{DENIT}	1.06	Temperature coefficient for denitrification (unitless)	Bowie *et al.* (1985)
Θ_{NIT}	1.06	Temperature coefficient for nitrification (unitless)	Bowie *et al.* (1985)
Θ_{on}	1.06	Temperature coefficient for organic N mineralization (unitless)	Bowie *et al.* (1985)
Θ_{ons}	1.06	Temperature coefficient for N mineralization in sediment (unitless)	Bowie *et al.* (1985)
Θ_{psi}	1.06	Temperature coefficient for silica solubilization (unitless)	Calibration
Θ_{SMIX}	1.06	Temperature coefficient for solids mixing (unitless)	Calibration

Appendix 4 – Algal Group Parameters (based on ranges from Bowie *et al.* (1985))

	Algal Group			
Parameter	**Cyanobacteria**	**Diatoms**	**Green Algae**	**Description**
CCHL	22	29	22	Carbon to chlorophyll *a* ratio (mg C /mg CHLA)
F_{ON}	0.5	0.5	0.5	Fraction of nitrogen that recycles to organic pool (mg organic N/mg total N)
F_{OP}	0.5	0.5	0.5	Fraction of phosphorus that recycles to organic pool (mg organic P/ mg total P)
FSAP	N/A	0	N/A	Fraction of silica that is dissolved (mg dissolved Si/mg total Si)
G_{1max}	1.5	2	2	Maximum growth rate/day
IS1	45	100	105	Maximum light saturation Langleys/day
K_{MPHYT}	1	1	1	Half saturation constant for phytoplankton mineralization in sediments mg C/L
κ_N	0.032	0.032	0.032	Half saturation for DIN
κ_P	0.0032	0.0032	0.0032	Half saturation for DIP
K_{PZDC}	0.02	0.02	0.02	Decomposition rate of phytoplankton in sediment /day
K_{PZDT}	1.06	1.06	1.06	Temperature coefficient of phytoplankton decomposition in sediments (unitless)
κ_{SI}	N/A	0.05	N/A	Half saturation for dissolved silica (mg Si/L)
N2FIX	0.1	N/A	N/A	Nitrogen fixation switch (mg DIN/L)
NCRB	0.125	0.125	0.125	Nitrogen to carbon ratio (mg N/mg C)
PCRB	0.022	0.022	0.022	Phosphorus to carbon ratio (mg P/mg C)
SICARB	N/A	1	N/A	Silica to carbon ratio (mg Si/mg C)
V_a	4.32	9.43	8.49	Algal settling rate (cm/d)
η_{phyt}	0.029	0.021	0.023	Algal light extinction coefficient (/m*mg Chl*a*)
Θ_1	1.08	1.06	1.06	Temperature coefficient for algal growth (unitless)
Θ_r	1.06	1.06	1.06	Temperature coefficient for respiration (unitless)
λ_d	0.020	0.053	0.056	Non-predatory mortality (/day)
λ_r	0.087	0.072	0.072	Respiration rate (/day)

N/A – Not Applicable