Lake Okeechobee Phosphorus Dynamics Study

Volume I

Summary

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LAKE OKEECHOBEE PHOSPHORUS DYNAMICS STUDY:

SUMMARY

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Executive Summary

Agricultural runoff has resulted in higher phosphorus (P) inputs to Lake Okeechobee that threaten the lake with more frequent blooms of nuisance algae. To protect the lake, regulatory and management agencies, along with landowners, have acted to reduce P inputs through Best Management Practices, diversions, and other means. The goal of this effort is to reduce these inputs by 40 percent. However, at the time this goal was established, the lake's probable response to lower P loads was unknown. Projecting this response required additional information on nutrient processing and cycling in the lake. Phosphorus concentrations in this shallow lake depend on internal processes such as sediment-water column interactions, biogeochemical processes in the sediments, and littoral-open water exchanges, as well as external inputs. The Phosphorus Dynamics Study was undertaken to investigate these processes so that future conditions could be predicted.

The project goals were to: (i) understand and quantify key processes that affect P dynamics in Lake Okeechobee, and (ii) develop a framework or model to understand and predict changes in lake P concentrations as affected by basin management actions and natural variability. As part of the second goal, this project attempted to determine how long lake recovery might be delayed by internal P loading. (Lake recovery is defined here as an annual mean total phosphorus (TP) concentration of 50 μg/L or less.) To achieve the project goals, specific tasks were organized under six objectives:

- Identify, map, and characterize the major sediment groups in the lake.
- Compare recent rates of sediment and P deposition with accumulation rates in the geologic past.
- Quantify the flux of P between the water and sediments due to various biogeochemical processes. Measure the P retention capacity of different sediment types.
- Assess the influence of major inflows on P retention by lake sediments.
- Estimate annual P loading to the water column from the sediments.
- Determine if aquatic macrophytes are a source or a sink in exchanges of P between the littoral and pelagic zones.

The research was conducted by an interdisciplinary team of investigators with expertise in hydrodynamic processes, chemical and physical characterization of sediments, physical processes related to sediments, and biogeochemical processes in lakes. The project's objectives were accomplished through survey and collection of sediments, extensive laboratory and *in situ* experiments, statistical analysis of historical data, intensive field monitoring, and modeling of processes regulating P dynamics and water quality. Detailed results are presented in a 15-volume report entitled "Lake Okeechobee Phosphorus Dynamics Study" of which this is the first volume.

This summary lists the results obtained under each of the six objectives. It is followed by a brief discussion relating these results to the study's two main goals.

1. **Identification, distribution, and characterization of major sediment groups (Volume II--Reddy et al.)**

   - Based on surface sediment (upper 10 cm) characteristics, five major groups (mud, sand, marl, peat and littoral) exist in Lake Okeechobee. Mud sediments represent 44% of the lake bottom, and are found throughout the north and central regions of the lake.

   - In the surface sediments, the storage of carbon (as total organic carbon), total nitrogen, and TP is 4.53×10^9 kg, 3.81×10^8 kg, and 2.87×10^7 kg, respectively. This results in an overall C:N:P mass ratio of 158:12:1. Most of the P is stored in the muds (42%) and sands (41%). Carbon and nitrogen storage is more evenly distributed among mud, sand, peat, and littoral sediment types.

   - The storage of TP in the surface sediments is 55 times greater than the lake's mean annual TP input (5.18×10^5 kg) and is increasing by an average of 1.5% per year.

   - Total dissolved P in the sediment porewater accounts for 0.3 to 0.8% of TP in the surface sediment layers. In mud and littoral sediment, more than 50% of the total dissolved P is in the highly available inorganic form.

   - The percentage of TP as iron- and aluminum-bound P is highest in littoral sediments (18%) and less than 4% in other sediment groups.

   - The percentage of TP as calcium-bound P (non-labile) is highest in mud (44%) and sand (53%) sediments and less than 18% in other sediment groups.

   - In peat sediments, up to 75% of TP occurred as organic P forms. Organic P represents less than 44% of TP in other sediment groups.

2. **Comparison of recent rates of sediment and phosphorus deposition with accumulation rates in the geologic past (Volume V--Engstrom and Brezonik)**

   - Historical sediment and P accretion rates were measured using ^{210}Pb techniques. Although difficulties were encountered in interpreting ^{210}Pb data from some sites, reliable dating of sediments from the mud zone of Lake Okeechobee is possible. Results show that sediment accumulation rates have increased during this century at all mud zone sites by an average of twofold (from 300 $g/m^2 \cdot yr$ before the year 1910 to 700 $g/m^2 \cdot yr$ in the 1980s). Phosphorus accumulation rates have increased about fourfold over the same period (from about 250 mg $P/m^2 \cdot yr$ before 1910 to about 1000 mg $P/m^2 \cdot yr$ in the 1980s). Most of this increase has occurred during the last 40-50 years.

3. **Quantification of the P flux between water and sediments due to various biogeochemical processes, including measurement of P retention capacity of different sediment types (Volume III--Reddy et al.)**

- Phosphorus retention capacity of the sediments is in the order: mud > littoral > peat > sand. The buffer capacity (or P adsorption coefficient) is highest in mud and littoral sediments.

- Phosphorus solubility in mud and littoral sediments is governed by iron under oxidized (high Eh) conditions, and by calcium-phosphate mineral precipitation under reduced (low Eh) conditions.

- Average P flux from bottom sediments is 0.70, 0.91, 0.29 and 1.09 mg $P/m^2 \cdot day$ for mud, peat, sand and littoral sediments, respectively. The range in P flux is 0.14 to 2.22 mg $P/m^2 \cdot day$ for all sediment types.

- Internal P loads from bottom sediments are approximately equivalent to the external P loads ($\sim$1 mg $P/m^2 \cdot day$).

- At low total suspended solids (TSS) concentrations (<2 g/L) and low dissolved oxygen levels (<1 mg/L) in water column, the resuspension P flux is about 6 to 18 times the diffusive flux.

4. **Assessment of the influence of major inflows on P retention by lake sediments (Volume IV--Reddy et al.)**

- Phosphorus retention capacity of the sediments increases with loading of inorganic P to the water column.

- Phosphorus retention capacity of the sediment near the Taylor Creek/Nubbin Slough inflow is high, with 80% of the P load (loading rate = 5.4 mg $P/m^2 \cdot day$) assimilated.

- The predicted equilibrium P concentrations (EPC_w) in the water column are 38, 100, 104, and 230 μg P/L for South Bay, Fisheating Creek, Taylor Creek, and Kissimmee River inflow sediments, respectively. If the water column P concentration of the lake near the inflow area is lower than the EPC_w, then the sediments at that location will function as a "source" of P to the water column.

5. **Determine if aquatic macrophytes are a source or a sink in exchanges of P between the littoral and pelagic zones (Volume VI--Dierberg; Volume VII--Sheng et al.)**

- Detrital submerged aquatic vegetation decomposes rapidly with turnover rates of less than 30 days.

- Water column TP concentrations exhibited distinct horizontal gradients at the edge of the littoral zone on both calm and windy days, suggesting that hydraulic exchange between the two zones is minor except during episodic events. However, the P flux between vegetation and open water is expected to increase with higher lake stage. During low lake stage, there is little water movement to transport P between the two zones. During higher lake stage, circulation may intrude into the vegetation zone to cause resuspension of sediments and P from the shallow bottom and the subsequent transport of P into the open water zone nearby. However, further data collection and modeling are needed to confirm this supposition.

6. **Estimate the annual P loading to the water column from the sediments (Volume IX--Mehta et al.; Volume X--Dickinson et al.; Volume XI--Pollman; Volume XII--Sheng and Chen)**

- Significant correlation was observed in open-water zone between TP and TSS, and wind speed.

- A three-dimensional P transport model (includes 3-D wind-driven circulation model, sediment transport model) was used to determine the dynamic changes at the sediment-water interface, and impact on water quality.

- Using a simple box model (LOP0D), a nine-year simulation showed that over the long-term, sediments act as a sink and not as a source to the water column. The predicted diagenetic fluxes of P were in the same range as those measured.

- Internal loading processes are dominant on a daily or weekly time scale, whereas external loadings are dominant when the time scale is months to years. Erosion/deposition is a dominant factor for short-term response, while wind mixing is critical to understand short and long-term P dynamics of the lake.

- A simulation of a modest resuspension event resulting from 20 mph wind over a 3-hour period indicated that a total of 9.36×10^{10} g of fine sediments from the central mud zone will be suspended into the water column. If these sediments are uniformly distributed throughout the water column over the entire muddy bottom, the suspended sediment concentration will increase by about 100 mg/L. This should in turn lead to an increase in TP concentrations by about 180 μg P/L, according to the correlation between TP and TSS. The amount of TP transferred from the bottom sediments to the water column would be approximately 1.2×10^8 g or 120 metric tons. In comparison to nutrient budget data presented by James et al. (1995a), this amount is 33% of the mean content of TP in the water column (367 metric tons) and 23% of the mean annual external TP input (518 metric tons).

- The amount of soluble P released into the water column during a typical diurnal event is expected to be only a small fraction (e.g., 1-10%) of the estimated 120 tons of TP. Much of the TP settles back to the bottom within a few hours after the event, while there may be a small increase in soluble P concentration on the order of a few μg P/L. The amount of soluble P released into water column over the course of a year depends mostly on the number of major wind events in the lake.

- In the open water zone, lake stage does not appear to affect the sediment resuspension over the muddy bottom significantly. At a lower lake stage, waves are damped and weakened, but the waves can reach the shallower bottom more effectively, hence the amount of sediment resuspension remains comparable to the higher lake stage case. During low lake stage, wind-driven circulation gyres are generally confined to the open water zone with little flow into the vegetation zone.

7. **Influence of reduced tributary loadings on total P concentration of the water column in the lake (Volume X--Dickinson et al.; Volume XII--Sheng and Chen)**

- A nine-year simulation using the LOP0D box model indicates that reducing tributary loads of SRP and TP by 40%, 50%, and 70% will cause corresponding reductions in lake SRP concentrations of 26%, 33%, and 46% and reductions in lake TP concentrations by 19%, 23%, and 33%. The model also shows that the internal loadings of SRP via diffusion and resuspension are insensitive to changes in the external loadings. Using this model, the predicted lake response is not as great as that expected from the modified Vollenweider (1976) model. When tributary P loads are reduced by 70%, the LOP0D model predicts that lake TP will decline to 62.9 μg/L. However, this prediction is unlikely to be precise, and the important point is that significant in-lake TP reductions can be expected with large reductions in tributary loads. A return to lower lake TP concentrations of around 50 μg P/L is possible, but the lake may take up to ten years to respond to the reduced external loadings. In other words, after the target loading rate is achieved, there may be a delay of several years before the mean lake TP concentration begins to drop significantly below its current level of approximately 90 μg P/L. These predictions are similar to those given by Bierman and James (1995) and Federico et al. (1981).

- Results of the three month simulation of the 3-D model showed that a reduction in the loading only caused localized reduction in TP and SRP concentrations. This suggests that it is necessary to perform longer term simulations in order to see more significant changes. It should be noted that none of the model simulations have been validated or verified yet with historical water quality data.

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1.0 Introduction

1.1 Background and Purpose

Lake Okeechobee (26°58'N, 80°50'W) is a shallow, eutrophic lake (Figure 1) located in a predominantly agricultural watershed. The lake provides important flood control and water supply benefits for south Florida, and is a major recreational and ecological resource. In addition to supplying potable water to several municipalities around its shore and irrigation water for cropland to the south, the lake serves as the backup water supply for the urbanized southeast coast. It supports a recreational and commercial fishery that contributes $28 million per year to the local economy (Bell 1987), and its broad marshlands provide habitat for a rich array of wildlife species. Because of its economic and ecological importance to the region, the management of this large lake has received considerable attention.

The control of nutrient runoff to the lake has been the dominant management issue. Initial monitoring and evaluation by the U.S. Geological Survey, Central and Southern Florida Flood Control District, and others, led to the conclusion that the lake's nutrient loads should be substantially reduced (MacGill et al. 1976). These excessive loads mainly originated from dairy and beef cattle operations north of the lake, and the Everglades Agricultural Area (EAA), a region south of the lake of where sugar cane and other crops are grown. Steps were taken to reduce nutrient inputs through Best Management Practices (BMPs) and restriction of pumped discharges from the EAA, but these actions were not enough to reduce phosphorus (P) loads or stop the rising trend in lake P concentrations. Continued monitoring by the Flood Control District (which later became the South Florida Water Management District, or SFWMD) revealed that annual mean total phosphorus (TP) concentrations in the lake water rose from 49 μg/L in 1974 to 122 μg/L in 1988 (Janus et al. 1990).

Additional watershed management improvements were undertaken after the Florida legislature enacted the Surface Water Improvement and Management (SWIM) Act in 1987. This Act provided partial state funding for the SFWMD to carry out an approved SWIM program. Through this program, the SFWMD, along with landowners and other agencies and institutions, have made significant progress toward improving the quality of water discharged to the lake. The goal of this effort is to reduce P inputs by 40 percent. Watershed management activities have included the establishment of a regulatory program, buy-outs of cows from certain dairy farms unable to meet runoff criteria, and further implementation of BMPs. These BMPs include establishing buffer zones along natural tributaries and water supplies, fencing cattle out of waterways, providing shade structures for cattle, modifying feed, fertilization, and pumping practices, constructing wastewater treatment lagoons, and recycling wastewater, solids, and sludge through crop spray irrigation and land application. As a result, recent data indicate that P loads are declining and in-lake TP concentrations have stabilized at about 80 μg/L (Flaig and Havens 1995; James et al. 1995a, 1995b). Thus, the lake response has been observed during both rising and declining P inputs.

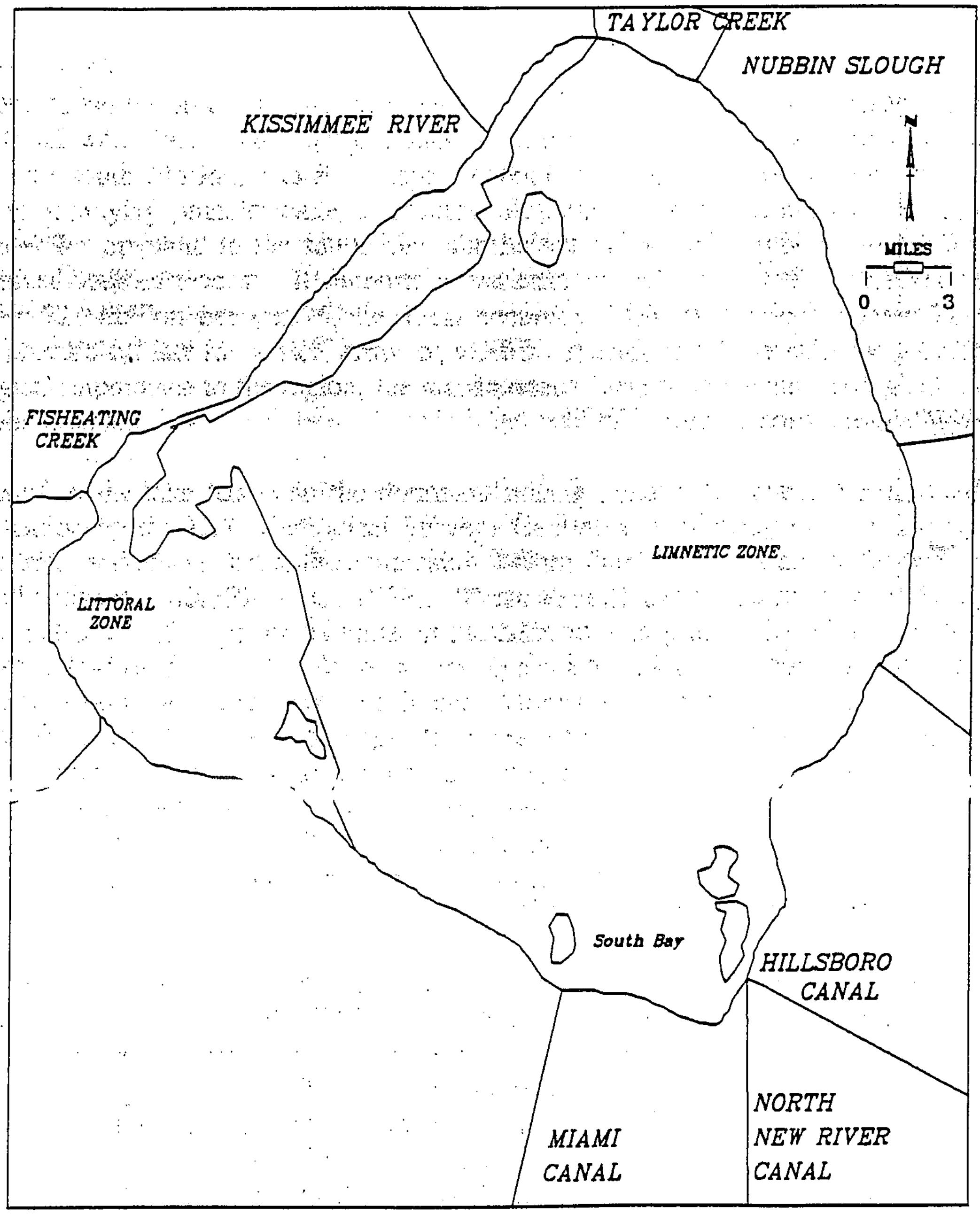

Figure 1. Map of Lake Okeechobee.

At the time phosphorus controls were initiated, little was known about the lake's probable response. The goal of 40 percent reduction in P loading was derived from the Vollenweider (1976) model modified for Florida lakes (Kratzer 1979). This model predicted that the targeted reduction would result in lower lake TP concentrations that would maintain the lake in a borderline meso-eutrophic condition (Federico et al. 1981).[1]

Although empirical models of this type need only a minimal amount of data and have been applied to many lakes throughout the world, they are limited to predictions of whole-lake, long-term conditions that may be inadequate for examining spatial and temporal scales important to management of Lake Okeechobee. For example, the modified Vollenweider (1976) model treats Lake Okeechobee as a completely mixed system, but this is an inaccurate assumption. Spatial variability in water quality has been proven through several studies of the lake (e.g., Aldridge et al. 1994, Havens 1994, Havens et al. 1994, Maceina 1993, Phlips et al. 1994b), and certain areas of the lake may differ in their responsiveness to nutrient inputs (Aldridge et al. 1994). The model also does not explicitly consider internal nutrient loading. As this report confirms, internal cycling greatly influences P concentrations in this shallow lake. Finally, the model assumes steady-state conditions with regard to nutrient loading, water input, residence time, and storage. Because of changes in watershed management and hydrologic variations, this assumption is violated over shorter time periods, as demonstrated by James and Bierman (1995) who found that the annual predictions from the Vollenweider model could not track changes in measured lake TP concentrations over a 20-year period. In short, steady-state models will estimate (with some uncertainty) the eventual condition of the lake after a change in the rate of nutrient loading, but they are inappropriate for predicting short-term shifts in nutrient concentrations under non-equilibrium conditions. For the latter purpose, a dynamic model is needed.

Dynamic models can estimate the response time of the lake to a change in P load, but this type of model requires information on hydrodynamic, sedimentary, and biogeochemical processes affecting P in the lake water column and sediments. The Lake Okeechobee Phosphorus Dynamics Study was initiated in 1988 to address this need. Its goals were to understand the factors that control P cycling and incorporate them into a framework for predicting the lake's response to the P reduction program. The products from this study will provide much of the groundwork for the SFWMD's ongoing modeling efforts. These models employ not only nutrient and hydrologic mass balance, but also nutrient uptake by algae, mineralization of organic material, and physical properties such as wind-driven resuspension of sediments. They structure information into a series of equations that can describe cause and effect relationships between nutrients, algae, and environmental conditions such as stage and wind. Through sensitivity analysis, the models can suggest research needs by determining which factors

[1] The Federico et al. (1981) report identified a value of 40 μg/L as the desired mean annual in-lake TP concentration. However, 50 μg/L also has been considered informally as being a more attainable goal. This slightly higher value is closer to the annual mean concentrations that were observed in the early 1970s.

are more important in determining algal productivity and bloom conditions. Ultimately, the SFWMD intends to use these models to determine the effects of nutrient load reductions and lake stage on the potential for algal blooms. The models also will simulate conditions in various "ecological zones" of the lake and describe localized impacts of nutrient inputs from certain tributaries.

1.2 Factors Affecting Phosphorus Dynamics

Phosphorus concentrations in Lake Okeechobee depend on interactions between the bottom sediments and water column, exchanges between the vegetated (littoral) and open water (limnetic) zones, internal biogeochemical processes, and inputs from tributaries, groundwater, and the atmosphere. Extensive studies of P cycling in lakes have identified the sediments as a major component (Syers et al. 1973; Bostrom et al. 1982). Previous work on Lake Okeechobee also indicates that internal nutrient sources are significant (Brezonik et al. 1979, 1983; Pollman 1983). Figure 2 shows the important components affecting the P dynamics in the limnetic zone (in boxes) and the interactions among these components (arrows).

In shallow lakes such as Lake Okeechobee, P flux across the sediment-water interface occurs in two different modes, depending on meteorological and hydrodynamic conditions. During calm days when vertical turbulent mixing and bottom shear stress are insufficient to resuspend sediment, dissolved P moves via passive diffusion and advection. The processes affecting P exchange in this mode include: (i) diffusion and advection due to wind-driven currents and wave-induced orbital motion, (ii) diffusion and advection due to flow and bioturbation within the interstitial water of bottom sediments, (iii) chemical processes within the water column (mineralization, adsorption/sorption, and biological uptake/release), and (iv) diagenetic reactions (mineralization, sorption, and precipitation/dissolution, etc.) within the bottom sediments. Although P exchange connected with these processes is continuous, the flux is small compared to the P movement due to resuspension and deposition of sediments during windy periods. Because resuspension events are transitory, the various hydrodynamic, sedimentary, and biochemical processes associated with them must be resolved on relatively short time scales (seasonal, monthly, and event-scale) to accurately estimate the annual P flux from sediment to the water column.

Lake littoral zones also can be a potential source of P to the pelagic water column. Aquatic vegetation may excrete a portion of the P taken up from the sediments (Carignan and Kalff 1982; Twilley et al. 1986). Macrophytes also release some of the nutrients stored in their biomass as they senesce and die (Landers 1981). The relative quantity of released P and its availability to phytoplankton varies depending on the system under study. Based on modeling work, Blancher (1979) determined that littoral macrophytes were the dominant suppliers of organic P to the pelagic zone of a central Florida lake, but most of it could not be utilized by the phytoplankton. Conversely, Carignan and Kalff (1982) reported only a small amount of P release by *Myriophyllum* into the surrounding water, but this P was in a highly available soluble form.

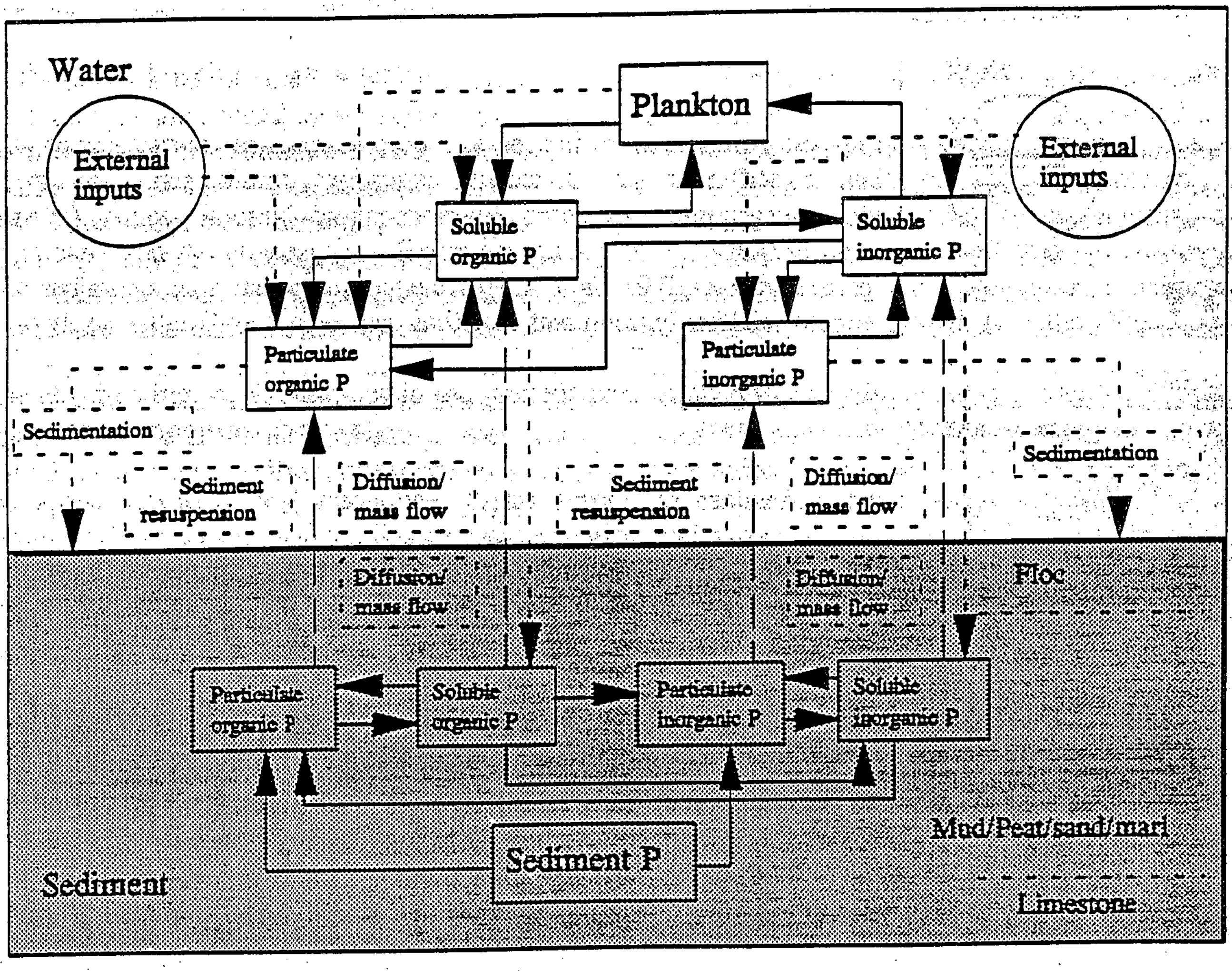

Figure 2. Phosphorus cycle in the sediment-water column of the Lake Okeechobee pelagic zone.

Some researchers have identified Lake Okeechobee's littoral zone as a potential contributor to internal P loading (Canfield and Hoyer 1988; Phlips et al. 1994a, 1994b). They have suggested that higher lake stages resulting in littoral zone inundation may promote the release of nutrients from decomposing plants and oxidized organic sediments and facilitate nutrient transport to the open water. Evidence for this internal loading process is derived in part from the positive correlation of lake stage and lake TP concentrations (Federico and Jones 1982; Maceina 1993; Hanlon et al. 1994). Brezonik et al. (1979) concluded that P from macrophyte decomposition and "pumping" was only a minor contribution to Lake Okeechobee's P load, but they acknowledged a high degree of uncertainty in their estimate. Because macrophytes occupy about 25% of Lake Okeechobee's surface area, a more complete examination of P transport from the littoral zone formed an important element of the Phosphorus Dynamics Study.

1.3 Project Objectives

To resolve and synthesize these varying effects and processes into products useful for predicting Lake Okeechobee's response to management options, a comprehensive, interdisciplinary project was initiated. The team of investigators included researchers with expertise in hydrodynamic processes, chemical and physical characterization of sediments, physical processes related to sediments, and biogeochemical processes in lakes. The project's objectives were accomplished through intensive field monitoring and sampling, statistical analysis of historical data, extensive laboratory and *in situ* experiments, and modeling of processes regulating P dynamics and water quality. Detailed results are presented in a 15-volume report (Table 1) entitled "Lake Okeechobee Phosphorus Dynamics Study" of which this report is the first volume. This summary report reviews the results and relates them to the information needs of lake management.

The study's ultimate goals were to: (i) describe and quantify key processes that affect P dynamics in Lake Okeechobee, and (ii) develop a framework or model to understand and predict changes in lake P concentrations as affected by basin management and natural variability. As part of the second goal, this project attempted to determine how long lake recovery might be delayed by internal P loading. (Lake recovery is defined here as an annual mean TP concentration of 50 μg/L or less.) To achieve these goals, specific tasks were organized under six objectives:

- Identify, map, and characterize the major sediment groups in the lake.
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- Estimate the annual P loading to the water column from the sediments.

- Determine if aquatic macrophytes are a source or a sink in exchanges of P between the littoral and pelagic zones.

Table 1. List of project reports on Lake Okeechobee phosphorus dynamics study. Reports were submitted to the South Florida Water Management District. For additional information contact Mr. Brad Jones, South Florida Water Management District, West Palm Beach, FL. Specific technical questions should be addressed to the principal investigators of the project.

Volume I	Lake Okeechobee phosphorus dynamics study - Summary [K. R. Reddy, Y. P. Sheng, and B. L. Jones]
Volume II	Physico-chemical properties of sediments. [K. R. Reddy, M. Brenner, M. M. Fisher, and D. Ivanoff]
Volume III	Biogeochemical processes in the sediments (Part 1). [K. R. Reddy, O. Olila, P. A. Moore, M. M. Fisher, and D. Ivanoff]
Volume IV	Biogeochemical processes in the sediments (Part 2). [K. R. Reddy, P. A. Moore, M. M. Fisher, K. Van Rees, and P. S. Rao]
Volume V	Phosphorus accumulation rates. [D. R. Engstrom and P. L. Brezonik]
Volume VI	Littoral characterization - Associated biogeochemical processes. [F. E. Dierberg]
Volume VII	Hydrodynamic and sediment dynamics - Field and modeling study. [Y. P. Sheng]
Volume VIII	Assembly and time series analysis of historical flow and phosphorus data. [J. S. Carey and W. C. Huber]
Volume IX	Sediment characterization - Resuspension and deposition rates. [A. J. Mehta]
Volume X	Modeling of phosphorus dynamics in the water column. [R. E. Dickinson, W. C. Huber, and C. D. Pollman]
Volume XI	Development of a phosphorus diagenetic model for sediments. [C. D. Pollman]
Volume XII	Three dimensional numerical model of hydrodynamic transport and phosphorus dynamics in Lake Okeechobee: Theory and model development and documentation - User's manual. [Y. P. Sheng and Xin-Jian Chen]
Volume XIII	Phosphorus retention by sediments. [K. R. Reddy and O. G. Olila]
Volume XIV	Hydrodynamics and sediment and phosphorus dynamics during an episodic event. [Y. P. Sheng, X. J. Chen, S. Schofield, and E. Yassuda]
Volume XV	Spectoscopic characterization of carbon and phosphorus in the lake sediment and water column. [C. T. Johnston, K. R. Reddy, and O. G. Olila]

2.0 Hydrodynamics of Lake Okeechobee

2.1 Introduction

Phosphorus discharged into Lake Okeechobee is strongly influenced by the hydrodynamic processes which govern the circulation (or motion) and mixing of water parcels. These processes transport P from the point of discharge to other parts of the lake. Vertical mixing distributes constituents (e.g., phosphorus, oxygen, organic material) throughout the water column and facilitates exchange at the air and sediment interfaces. Thus, to predict P transport within the system, these circulation and mixing patterns need to be quantified.

Circulation and mixing in Lake Okeechobee are influenced by many factors, including heating and cooling at the water surface, surface inflows and outflows, evaporation and precipitation, vegetation, and the lake's complex geometry and bathymetry. However, the primary driving force is wind. The direct shearing action of wind at the air-water interface transfers much momentum and energy to the water and produces wind-driven circulation. Part of the momentum and energy in the air, however, is transferred into wind-generated waves. These waves produce orbital water motion which diminishes with depth below the water surface.

Wind-driven circulation and wave-induced motion have very different time scales. The wave-induced orbital motion generally varies quickly with time, with a dominant period on the order of a few seconds. The wind-driven circulation, on the other hand, varies more slowly, with a dominant period of a few hours to a few days. Both the slowly-varying circulation and the high-frequency, wind-induced waves must be examined to quantify the hydrodynamics. But due to the different time scales involved, different methodologies (field monitoring and numerical modeling) must be employed.

The objective of this portion of the study was to obtain a quantitative understanding of the wind-driven circulation and wind-induced waves in Lake Okeechobee. This information lays the groundwork for (i) quantifying the effects of hydrodynamic processes on sediment and P dynamics, with emphasis on benthic flux in the lake's limnetic zone, and (ii) quantifying the effect of vegetation on water movement and P flux between the littoral and limnetic zones.

2.2 Wind-Driven Circulation

To quantify the wind-driven circulation, parameters (wind speed and direction, water current at 1-3 vertical levels, and water temperature at 1-3 vertical levels) were measured at six locations (Figure 3) over two 1-2 month periods (one in 1988 and one in 1989). Two types of field platforms with mounted instruments were used (Sheng et al. 1992). A portable platform was used at most sites except at the Central Lake Station (UF Station C) and the Eastern Lake Station (UF Station D) where the bottom mud layer was too thick for placement of portable platforms. At each of these two stations, a fixed

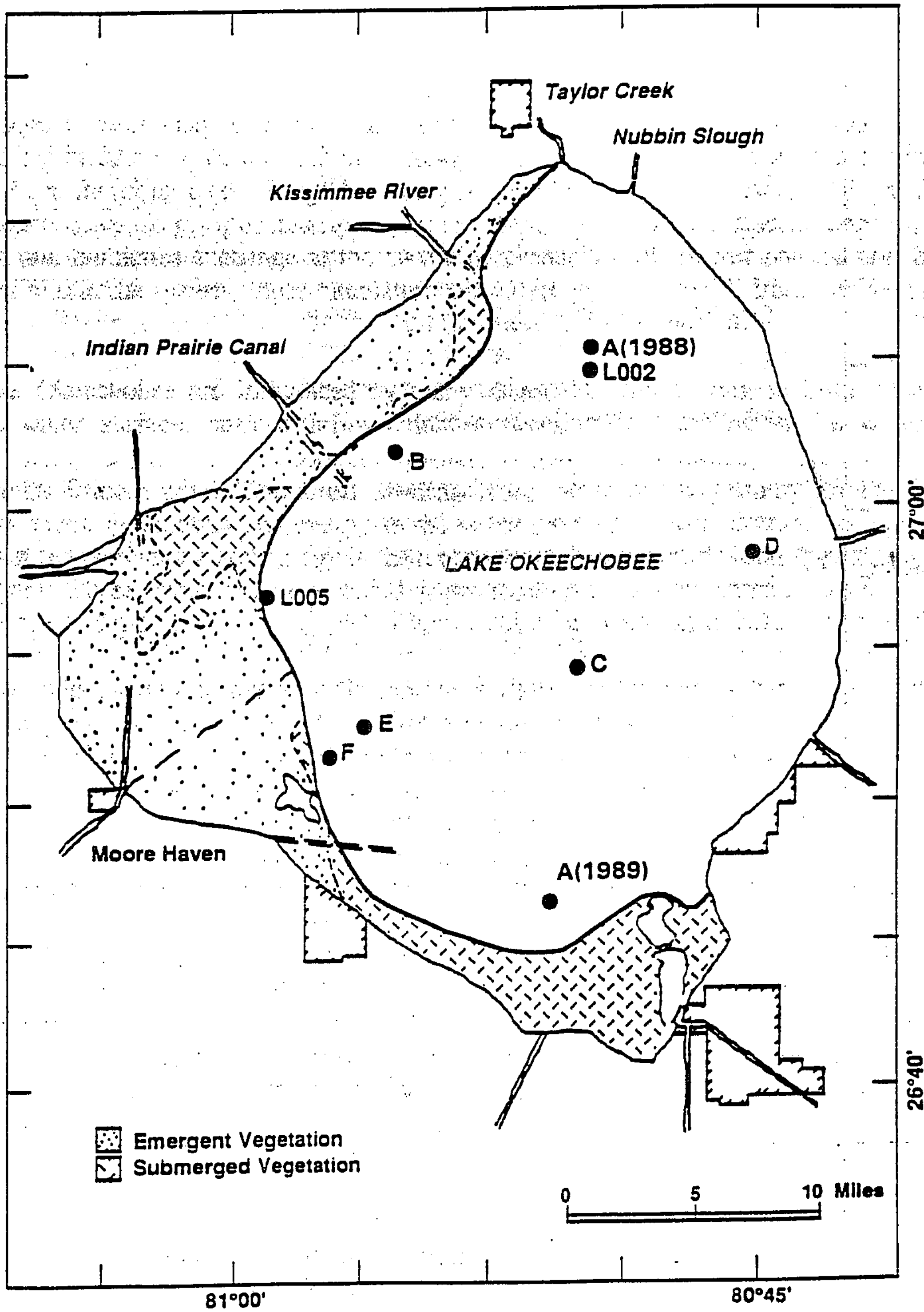


Figure 3. Lake Okeechobee with platform locations. Six platforms were placed in the lake for this study. Platform A was located in the north end in 1988 and in the south end in 1989. Stations L002 and L005 are routine monitoring stations operated by the SFWMD.

platform with piling was used instead. During the 1988 field study, three portable platforms were deployed in the vicinity of the littoral zone (Stations B, E, and F), where the bottom was primarily composed of fine sand, and one platform was deployed in the northern end of the lake (Station A). During 1989, Platform A was moved to the southern portion of the lake, where the bottom was composed primarily of peat.

2.2.1 Wind

Wind conditions near the water surface are influenced by three-dimensional atmospheric circulation (convective transport) over the lake. Except during thunderstorms, summer winds are generally weak with distinct lake breeze patterns. With the passage of cold fronts in the fall and winter, winds are generally stronger. Of course, tropical storms and hurricanes can drastically impact hydrodynamics, and it is important to realize that these extreme events may profoundly affect the sediment and P dynamics of Lake Okeechobee. However, these storms are rare and difficult to monitor, so sufficient data are not available to evaluate their significance. Therefore, the analysis of data collected for this study was necessarily restricted to the range of wind speeds commonly observed.

During the field study, 15-minute averaged wind data (speed and direction) were measured at most of the stations. Although wind speed and direction vary, even between different regions of the lake, there is a clear dominant diurnal cycle, particularly during the summer months. A representative 3-day wind record at one station is shown in Figure 4. The air is generally calm in the morning and early afternoon, then a breeze begins around late afternoon (approximately 6:00 pm) and remains strong until midnight. The dominant wind direction is from the east. A sudden change in wind conditions often leads to the development of seiche (standing wave) oscillations of water levels and currents.

Light winds are generally due to convectional air currents. These currents, along with localized thunderstorms, result in variations in wind speed and direction among different areas of the lake. Stronger winds usually accompany the passage of cold fronts (except in the case of tropical disturbances). Consequently, wind conditions over the lake are usually more uniform when the wind speed is greater. Because of the spatial variability in wind conditions, it is desirable to maintain more than one wind monitoring station on this large lake.

2.2.2 Currents

Significant diurnal variations in the speed of currents measured in Lake Okeechobee suggest that these currents are primarily driven by wind. Currents are usually weak (on the order of 10 cm/s or less), but can increase to 50-70 cm/s or higher during periods of seiche oscillation. A sample current meter record (Figure 5) at the Center Lake Station (Station C), corresponds to the wind data shown in Figure 4. Immediately after the wind speed increases in late afternoon, strong currents are

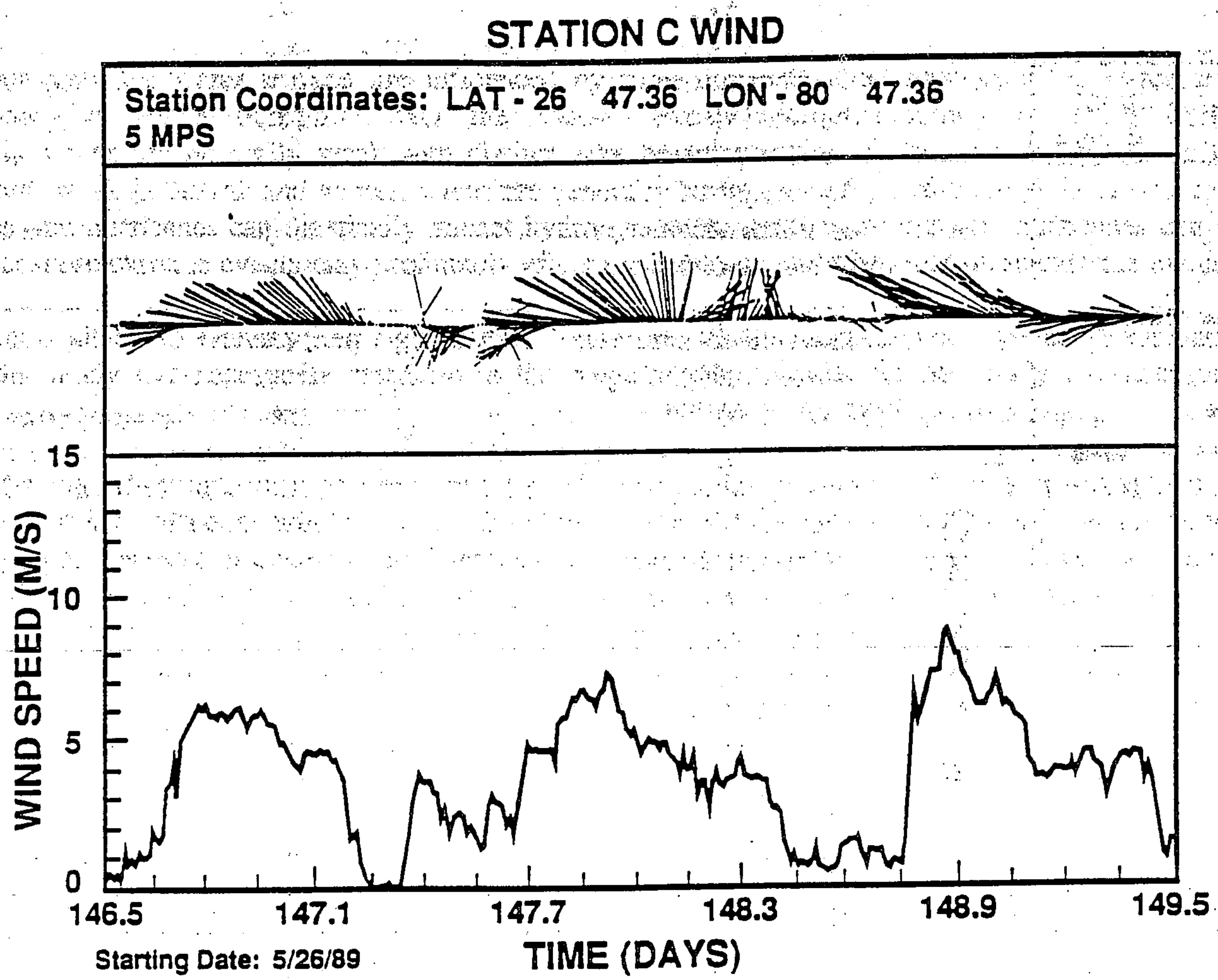


Figure 4. Wind velocity and wind speed at Station C during Julian days 146.5 (noon on May 26) to 149.5 (noon on May 29) in 1989. Sticks in the upper panel indicate the directions to which the wind is going (North is up). Length of the sticks indicates wind speed.

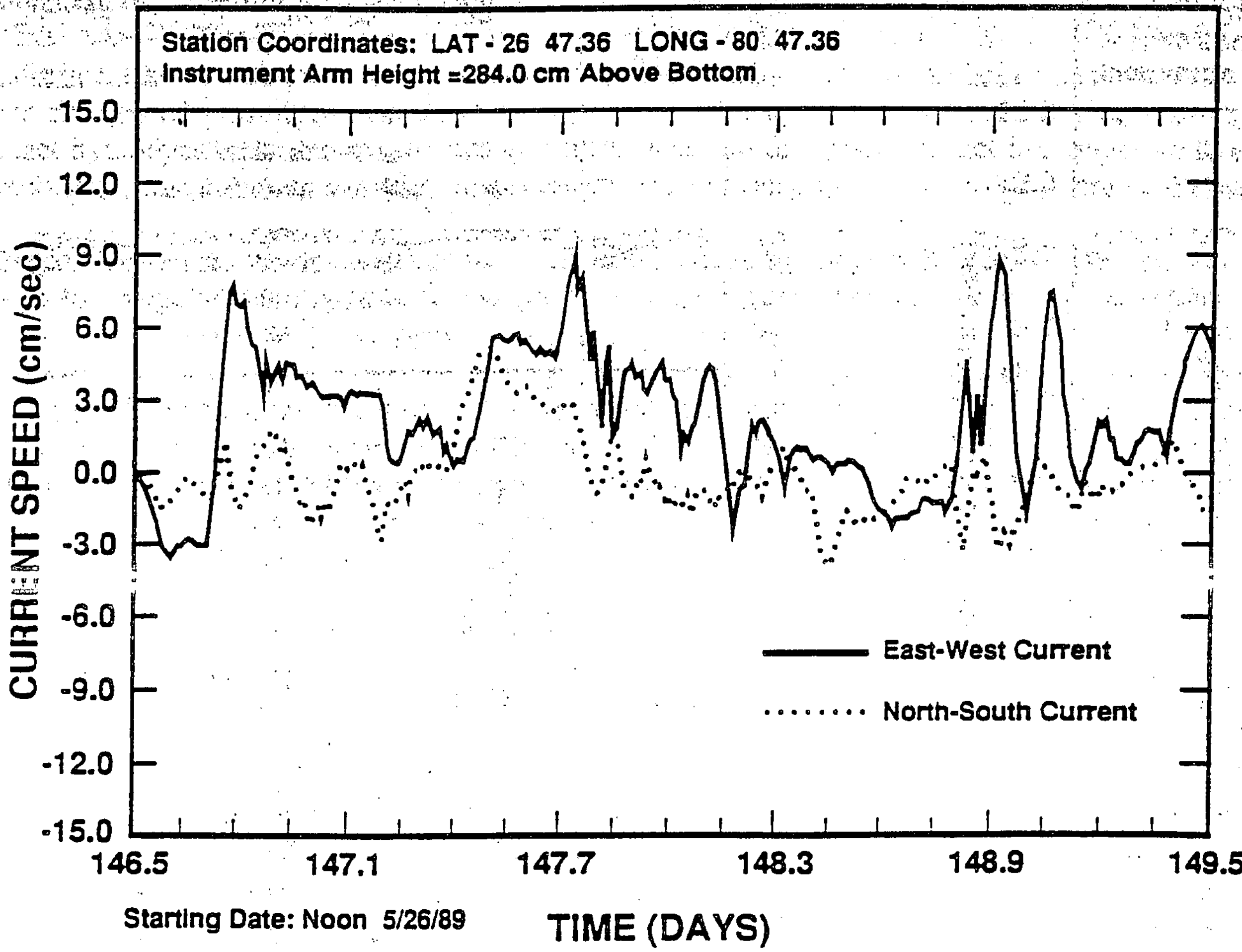


Figure 5. Wind-driven currents at Arm 3 of Station C (depth = 3.5 m) during May 26-29, 1989.

generated with an oscillation period of about 3-5 hours, which corresponds to the time that a surface disturbance takes to travel back and forth across the lake.

Spectrum analysis of the 15-minute averaged current data at the Central Lake Station shows that the east-west current has a seiche period of 3.5 to 4 hours, while the north-south current has a seiche period of 5 to 5.5 hours. The frictional resistance of the vegetation, together with the shallow depths in the western portion of the lake, apparently reduce the effective width and depth of the basin and lead to a shortening of the east-west seiche period. These seiche oscillations, however, generally do not produce enough bottom stress to resuspend bottom sediments.

The currents produced by these seiche oscillations are not deflected significantly by the Coriolis force. This effect, which results from the earth's rotation, directs atmospheric motions and sea currents, but most lakes are so small that shoreline influences preclude much of the Coriolis effect. In the shallow waters of Lake Okeechobee, bottom friction is quite important, and the Coriolis effect is not apparent as the data show no evidence of the Coriolis time scale.

Despite significant vertical mixing, currents do decrease somewhat from lake surface to lake bottom. Vertical turbulent mixing is primarily produced by the vertical shear associated with the wind-driven currents, instead of the wave orbital currents which do not vary significantly with depth except within the very thin wave boundary layer near the bottom.

During the relatively calm summer months, a diurnal thermal cycle is usually present in the water column due to diurnal variation in heat flux at the lake surface. During morning and early afternoon, the surface water is gradually heated by radiation and heat flux from the air. A thermally stratified water column usually develops between noon and 2 pm. A representative temperature record at the Center Lake Station is shown in Figure 6. The onset of lake breeze in late afternoon usually causes a sudden increase in current speed with differing current directions in the surface layer (downwind) and bottom layer (upwind). In the evening, the surface water is cooled, which generally leads to a turnover and de-stratification of the water column.

Currents vary spatially due to the lake's bathymetry, shoreline geometry, and vegetation. Under the forcing of a uniform and time-invariant wind field, steady-state lake circulation consists of vertically-integrated gyres which are generated by the spatially-varying topography. For example, with the saucer-shaped bathymetry shown in Figure 7, an easterly wind (wind from the east) generally leads to a steady-state lake circulation, consisting of a clockwise gyre in the southern portion of the lake and a counter-clockwise gyre in the northern portion. In the shallow southern and northern ends of the lake, surface and bottom currents are generally in the direction of the wind. In the deeper central portion of the lake, surface currents are in the direction of the wind, while bottom currents are opposite to the wind direction. A mild wind of 5-6 m/s is expected to cause a setup (i.e., pileup of water) on the order of 10-20 cm in the

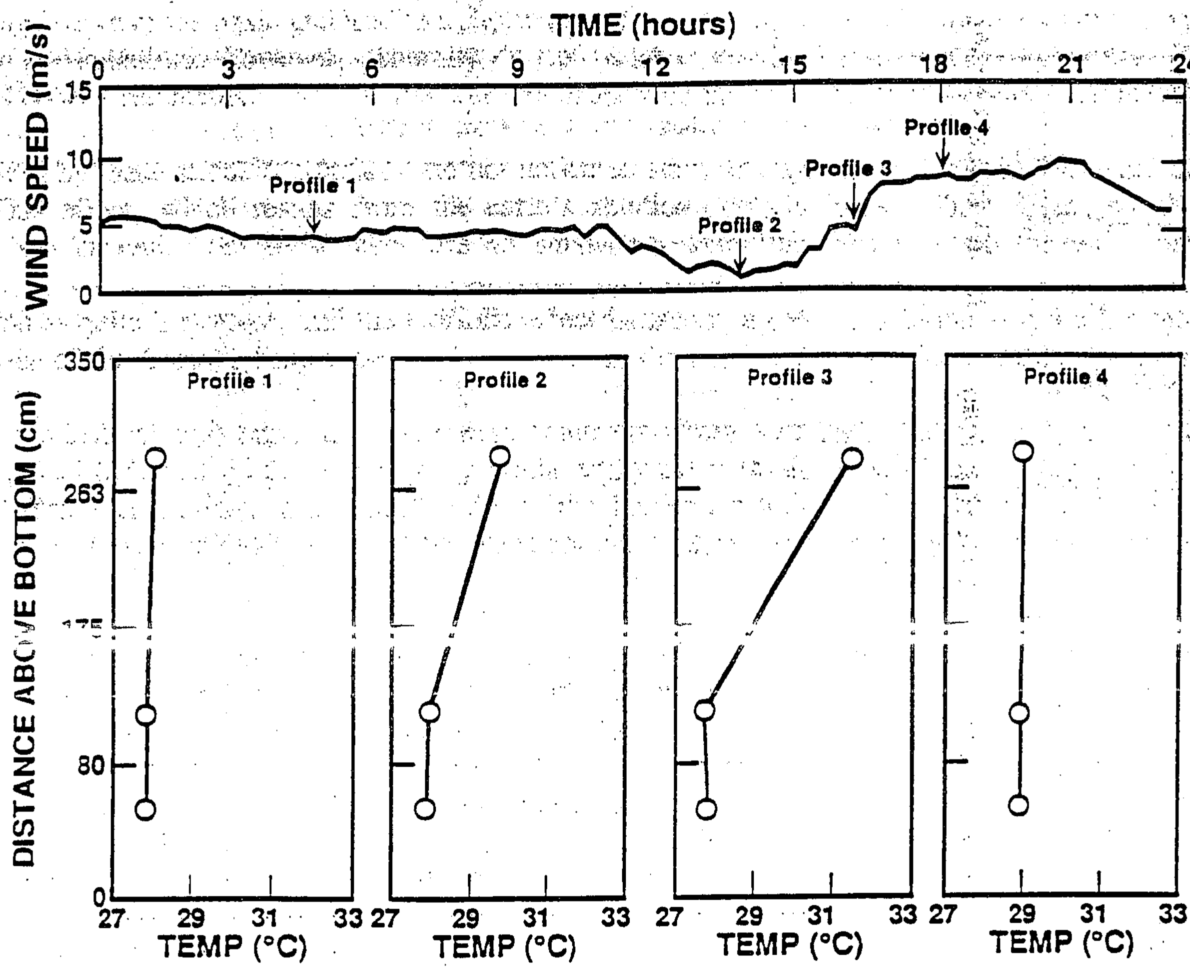


Figure 6. Wind speed and vertical profiles of temperature at Station C in Lake Okeechobee on June 4, 1989.

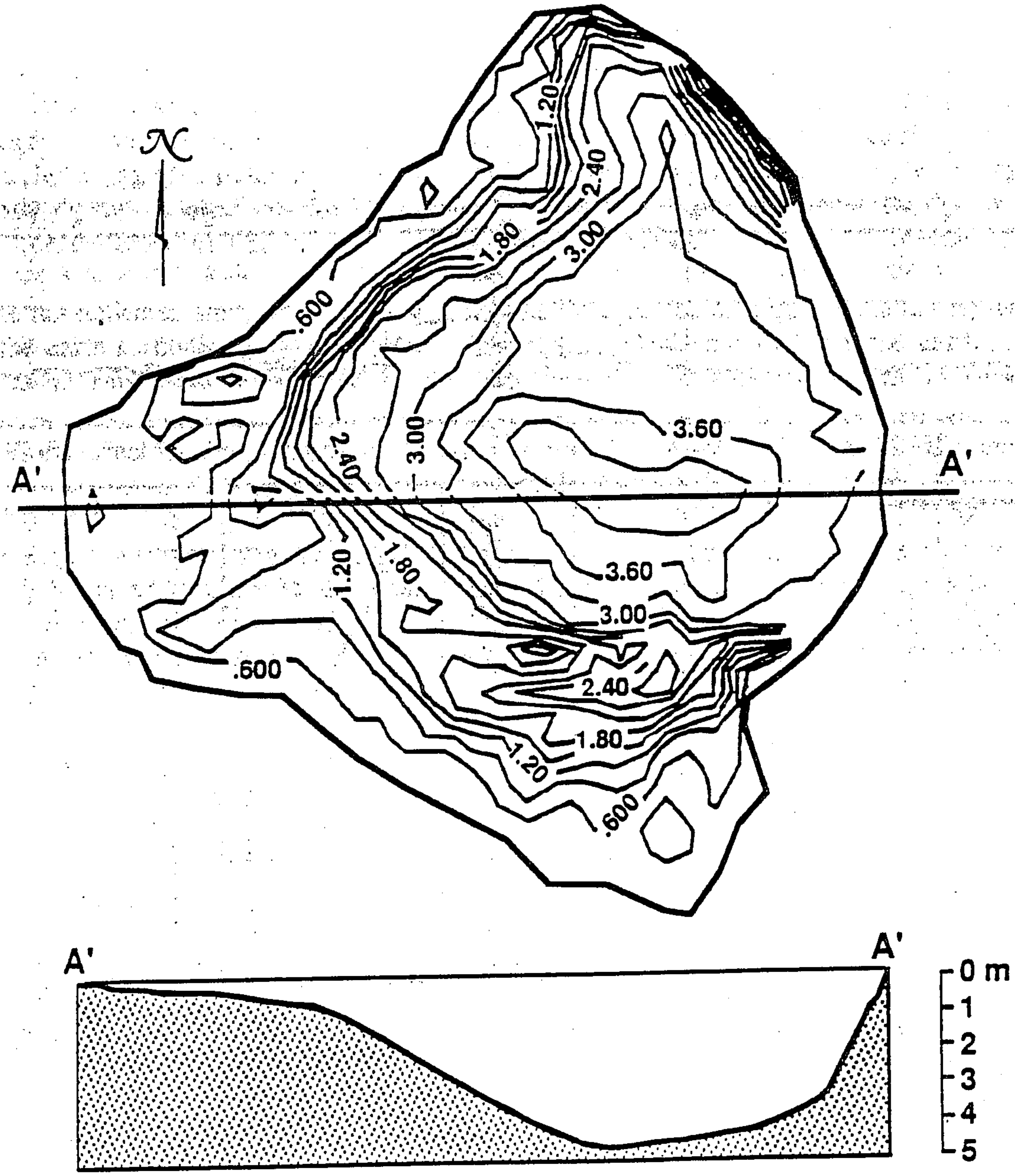


Figure 7. Geometry and bottom topography of Lake Okeechobee (Sheng et al. 1991a).

downwind direction (Sheng et al. 1988). However, a time-invariant wind field over an extended time period is rare, so steady-state circulation is rarely found in the lake. Instead, lake circulation is quite dynamic showing significant temporal variation. At any instant of time, there may be one, two, or more gyres in the lake, but these gyres change significantly with time.

Near the littoral zone, the dense vegetation causes the adjacent open water to flow parallel to the littoral-pelagic boundary. For example, at Station F near the southwestern littoral zone, flow direction is primarily northwest/southeast, as indicated by the strong north-south current and weak east-west current shown in Figure 8. In the vicinity of the northwestern littoral zone, the current at Station B flows primarily northeast/southwest.

Except during extreme storms, wind-driven currents are too weak to resuspend sediments or nutrients. Instead, sediment resuspension is caused by wave-induced orbital currents. Hence, the primary effect of wind-driven currents on sediment and nutrient dynamics is to provide transport (advection) and turbulent mixing. However, during hurricane wind conditions, wind-driven currents exceeding 1 m/s and a surface setup exceeding 1-2 m may be expected. Under these conditions, both wind-driven currents and wave-induced currents can cause significant resuspension of sediments and nutrients.

2.2.3 Circulation Modeling

A three-dimensional numerical model of wind-driven circulation in Lake Okeechobee (Sheng et al. 1991f, Sheng 1992) contains the following approximations: (i) vertical acceleration is small, (ii) density variation is small, (iii) vertical and horizontal turbulent mixing are of vastly different scales and can be parameterized separately, and (iv) a turbulent bottom boundary layer generally exists within the lowest 10% of the water depths. The model uses a robust model of turbulent transport to parameterize the vertical turbulent mixing, thus eliminating the need for *ad hoc* tuning of model coefficients for each model simulation. The model uses a uniform or non-uniform horizontal grid with a stretched vertical grid. To speed up simulations, the 3-D model uses a mode splitting technique to separate the fast external (surface gravity wave) mode from the slow internal (vertical flow structure) mode. Similar versions of the 3-D model have been developed for other lakes (e.g., Lake Erie, Green Bay, Lake Apopka) and estuaries (e.g., Tampa Bay, Charlotte Harbour, Mississippi Sound).

Application of the 3-D model to Lake Okeechobee (with a horizontal grid spacing of 2 km and five grid points in the vertical direction) showed that measured currents at most of the stations can be simulated by the model with reasonable accuracy (Sheng et al. 1991f, Sheng 1992). The model was used to produce flow fields in the lake over time periods ranging from a few days to three months. Model simulations showed that circulation gyres in the lake vary considerably with time, in response to the temporally and spatially varying wind. For example, circulation gyres for the surface and bottom layers of the water column at two instants of time are shown in Figures 9 and 10. Results of the model simulations were used to estimate sediment (Sheng et al. 1991c,

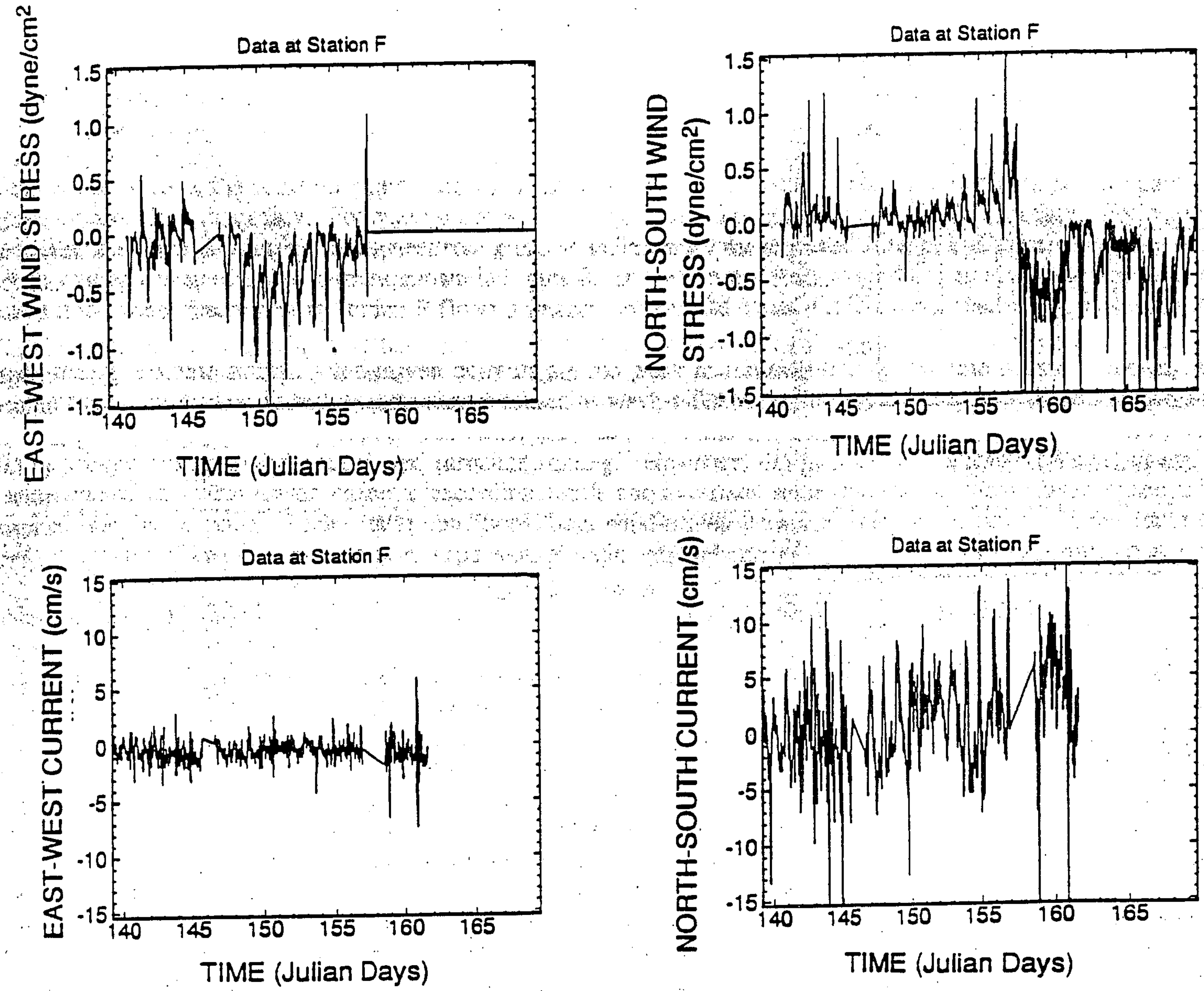


Figure 8. Wind stress and measured currents at Station F during Spring 1989.

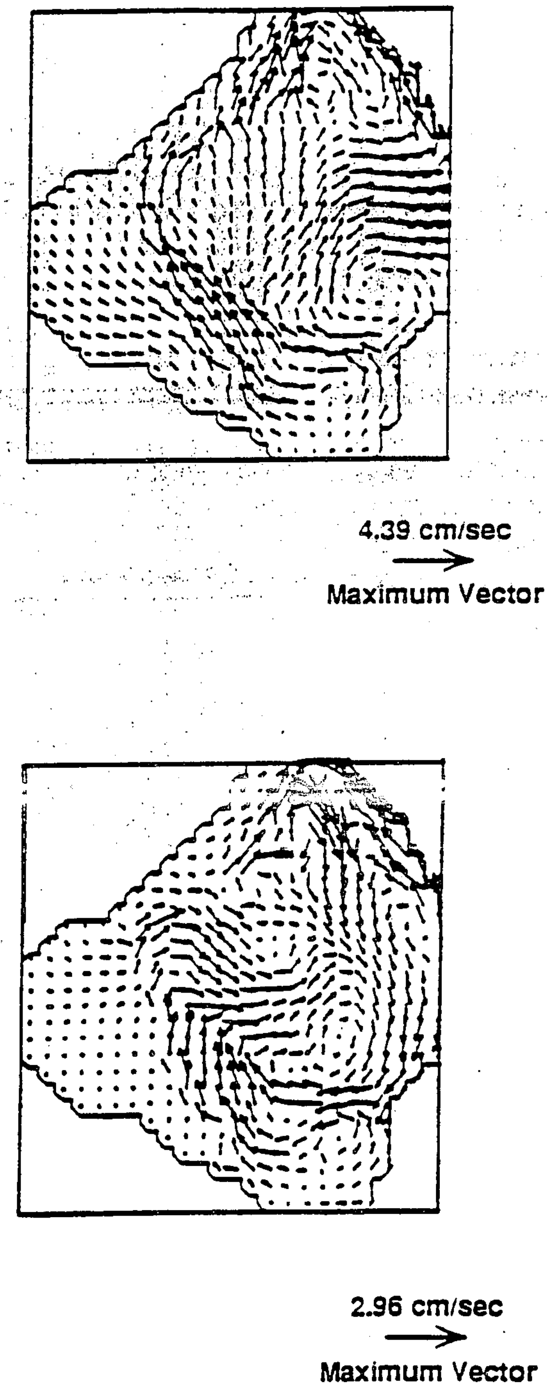
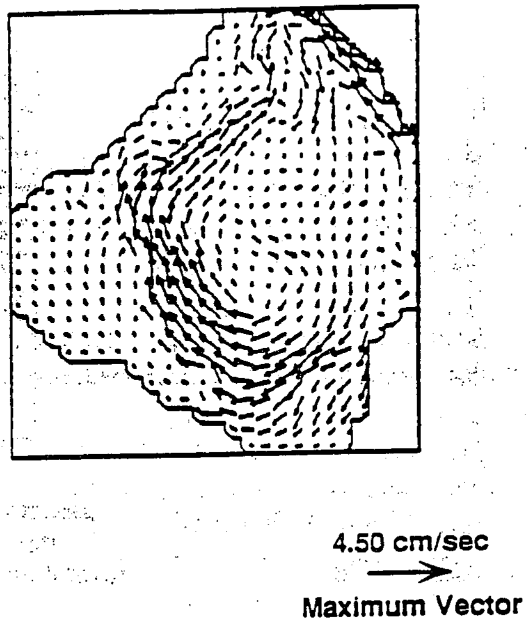


Figure 9. Near-surface (top panel) and near-bottom (lower panel) flow fields at the time of the third synoptic survey of Spring 1989. Notice the different gyres at the two vertical levels. The effect of vegetation is empirically parameterized in terms of added bottom friction.



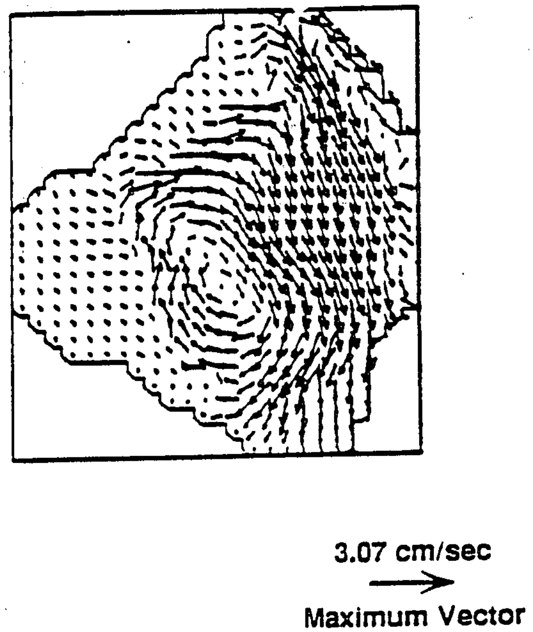


Figure 10. Near-surface (top panel) and near-bottom (lower panel) flow fields at the time of the fourth synoptic survey of Spring 1989. Notice the different gyres at the two vertical levels. The effect of vegetation is empirically parameterized in terms of added bottom friction.

Sheng et al. 1991d, Sheng et al. 1989) and P transport in the lake (Sheng et al. 1991d). The model was coupled to the sediment transport model and P dynamics model in a 3-D modeling system for Lake Okeechobee, LOHSP3D (Sheng and Chen 1992).

2.3 Wind-Induced Waves

2.3.1 Wave Dynamics

Short-period waves in Lake Okeechobee are generated by wind. The height and period of wind waves at any particular location in the lake depend on the wind speed and the fetch length, i.e., the distance over which the wind remains nearly uniform. For example, for an easterly wind, fetch length and wave height increase from Station D to Station C to Station E. In the littoral zone, profile drag due to vegetation and bottom friction due to shallow depth lead to damping of wave height and period. Wave data were collected at selected stations in open water with a 2 Hz sampling frequency. The energy spectrum of the measured wave data shows that wave energy is concentrated at waves with periods of 2 to 3 seconds (Figure 11).

Rather than exhibiting a single wave period at a given time, waves of different periods commonly appear together. Under typical modest wind speeds, wave height ranges from 10 to 60 cm. During storms, however, wave height could reach 1 m or higher. During the prevailing easterly wind, wave height grows from the eastern part of the lake towards the western marsh, where the waves are dampened when they reach the shallower depths and vegetation.

Based on measured wave height and period, the bottom orbital velocity and the bottom stress exerted on the sediments can be estimated. Under the forcing of a typical lake breeze in the late afternoon, significant bottom stress (1 to 5 dyne/cm^2) can be generated due to the sharp velocity gradient within the wave boundary layer, which is usually only a few centimeters thick. Due to difficulties in deploying instruments too close to the bottom, the currents within the bottom boundary layer could not be measured. Thus, bottom stress must be estimated by using numerical models.

2.3.2 Wave Modeling

Sediments are primarily resuspended by bottom stress induced by wave orbital currents near the bottom. Hence, before developing a sediment transport model (which includes advection, diffusion, settling, turbulent mixing, deposition, and resuspension), a wave model must be selected. Given a two-dimensional wind field over the lake, this wave model will estimate wave height and wave period throughout the entire lake.

In this study, three wave models were compared (Sheng et al. 1991c, Ahn and Sheng 1990): a spectral wave model, a parametric wave model, and a relatively simple empirical wave model. Results indicate that the parametric wave model used by the Great Lakes Environmental Research Laboratory (GLERL) of NOAA over-predicts the

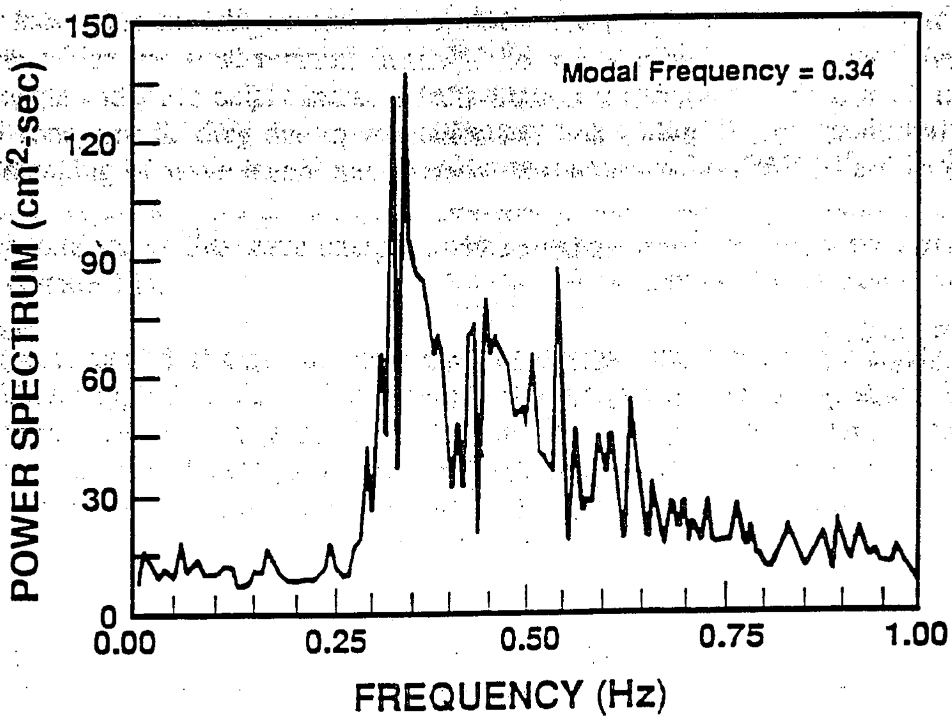


Figure 11. A typical wave spectrum during a windy period. The figure indicates that most energy is concentrated at 0.34 Hz (cycles/sec), which corresponds to a dominant wave period of 2.94 sec (Sheng et al. 1991f).

wave height and period in the lake, due to the lack of bottom dissipation in the model. The spectral model and the simple empirical model gave good results based on detailed comparisons with data. The simple empirical (SMB) model takes only 2.5% of the computational time required by the spectral model, so it was selected for practical application to Lake Okeechobee. Figure 12 shows an example of the SMB model's ability to simulate wave conditions.

Based on estimated wave height and wave period, the bottom stress may be computed by using a number of bottom boundary layers. To reduce computational time, the SMB wave model was combined with a simple bottom boundary layer model (Kajiura 1968) and included in the 3-D modeling system (LOHSP3D). Numerical modeling showed that bottom stress generated by waves was generally much larger than that due to currents. Thus, consideration of the current-wave interaction in the bottom boundary layer is not necessary except possibly during extreme storm events.

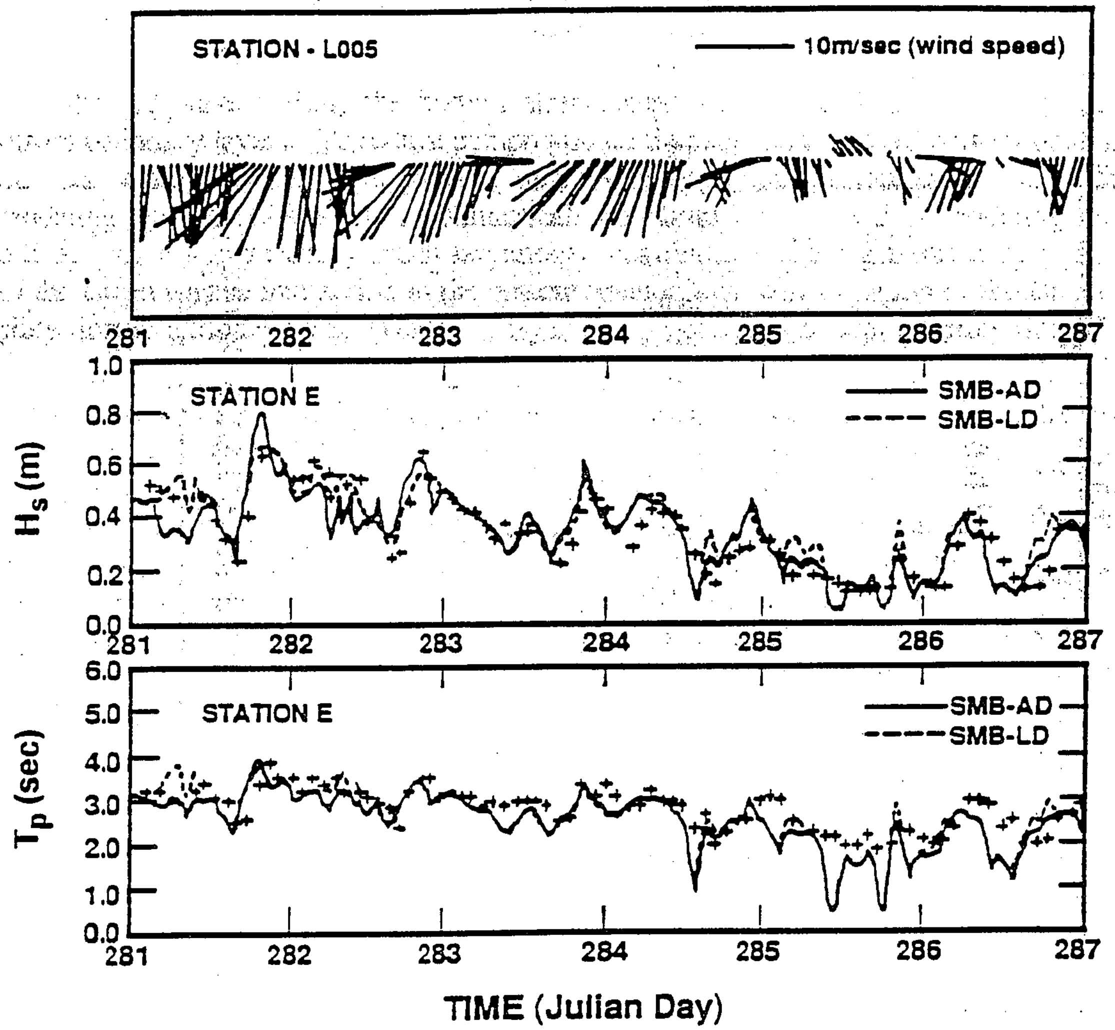


Figure 12. Comparison between simulated (by SMB-AD and SMB-LD wave models) and measured (+++) H_s (significant wave height) and T_p (peak wave period) at Station E in 1988. Stick diagram in the upper panel of measured wind at Station L005 indicates the directions to which the wind is going (North is up). Length of the sticks indicates wind speed.

3.0 Characterization of Bottom Sediments

Lake Okeechobee sediments were characterized for selected physical and chemical properties. Intact sediment cores were collected from 171 grid sites, 3.2 km apart, using a piston corer. Objectives included: (i) identify and map the major sediment groups in the lake, (ii) establish the spatial variability of C, N, and P fractions in the sediments, (iii) determine the soluble and insoluble forms of inorganic P in sediments, and (iv) estimate the historical sediment and P accretion rates in the lake using the ^{210}Pb technique. [For details see Reddy et al. 1991 (Vol. II), Engstrom and Brezonik 1991 (Vol. V), and Kirby et al. 1989 (Vol. IX).]

3.1 Sediment Types

The surficial sediments (10 cm depth) were classified into five major groups (mud, sand/shell/marl, peat, littoral, and rock), and their areal extent was mapped (Figure 13). Mud sediments were found throughout the lake's north and central region and occupy 44.3% of the lake bottom. A broad sandy area was observed between the mud zone and the western littoral area, and sandy deposits were found throughout the basin. Sands represent 24.5% of the lake bottom. The rocky reef in the southern part of Okeechobee represents only 3.7% of the lake area, and 8.9% of the bottom is peat in the most southern part of the lake. The western marsh has heterogeneous sediments, and these deposits occupy 18.6% of the lake. Qualitative differences among zonal sediment types were reflected in the physical and chemical measurements. Mean bulk density (g dry/cm^3) of surficial sands (1.213) was much higher than for mud (0.154), littoral (0.314), and peat (0.137) zones.

3.2 pH, Alkalinity, and Conductivity

Sediment porewater pH ranges were similar in the mud (6.7-8.3) and sand (6.9-8.2) zones. A narrower pH range was observed in peat deposits (7.0-7.5). The broadest pH range was found in littoral sediments (5.8-8.4), where low values reflect the accumulation of organic acids and CO_2 that accompany plant decomposition. Alkalinity ranged from 110 to 310 mg $CaCO_3$/L in the mud sediment and from 150 to 420 mg $CaCO_3$/L in the sand zone. Mean values for the mud and sand sediments were quite similar (194 and 204 mg $CaCO_3$/L respectively). Mean alkalinity of peat sediments was 152 mg $CaCO_3$/L. Littoral sediments generally had lower alkalinities (46-268 mg $CaCO_3$/L) reflecting, in part, the lower carbonate content of littoral sediments. Mean values for conductivity (μS/cm) among surface sediments of the mud (536), peat (617), sand (589) and littoral (516) zones were approximately the same. The minimum value (250 μS/cm) was recorded in the littoral area, reflecting low ionic strength at the site.

3.3 Dissolved Phosphorus and Nitrogen Forms in Sediment Porewater

Sediment porewater was separated from wet sediment by centrifugation, then filtered and analyzed for soluble reactive P (SRP) and total dissolved P (TDP). The

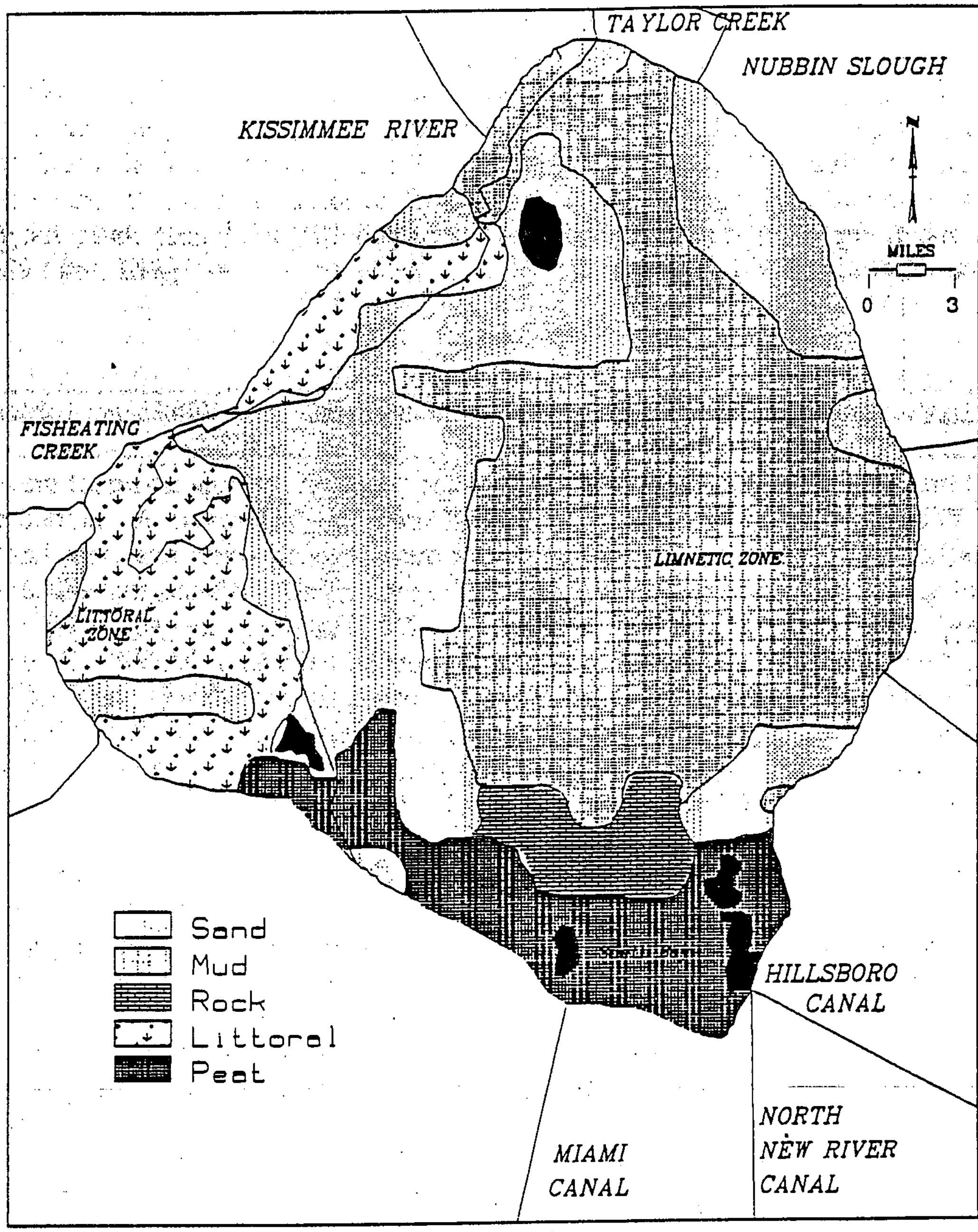


Figure 13. Major sediment groups of Lake Okeechobee.

difference between TDP and SRP yielded dissolved organic P (DORP). Soluble reactive P is the inorganic form of phosphorus that is most available for plant and algal uptake. [Note: In Figure 14, SRP is shown as "PW-Pi" (porewater inorganic P) and DORP is shown as "PW-Po" (porewater organic P)].

In the surface sediment layers (0-10 cm), TDP in the porewater represented 0.3 to 0.8% of the total P in the sediment. Total dissolved P concentrations were generally higher in surface sediments than in deposits from deeper in the profiles. Littoral deposits (0.17 mg/L) contained less TDP in the porewater than muds (0.63 mg/L), peats (0.52 mg/L), and sands (0.63 mg/L).

The mean concentration of porewater SRP was highest in the mud sediment. This concentration (0.392 mg/L) was significantly greater ($p < 0.05$) than averages measured in the peat (0.040 mg/L) and littoral (0.125 mg/L) areas, but not significantly different from the sandy deposits (0.241 mg/L). Soluble reactive P accounted for 62% of the TDP in the mud and 74% of the TDP in the littoral deposits. In the peat and sand, the TDP was dominated by DORP. Soluble reactive P constituted only 8% and 38% of the TDP in the peats and sands, respectively.

Three forms of dissolved N were measured in sediment porewater: nitrate (NO_3-N), ammonium (NH_4-N), and dissolved organic (DON). The only significant difference in mean NH_4-N concentrations was found between the sand (4.42 mg/L) and peat (1.70 mg/L) zones. Averages for the littoral area (2.71 mg/L) and mud zone (2.47 mg/L) were not statistically different ($p < 0.05$) from means calculated for other zones. Average DON concentrations in the peats (6.75 mg/L), sands (9.54 mg/L), and littoral area (7.17 mg/L) were statistically indistinguishable, but all three values were greater than the mean computed for the mud zone (2.89 mg/L).

3.4 Phosphorus and Carbon Content of Sediments

After removing the porewater, the following P fractions in the residual sediment were obtained through a series of sequential extractions:

- Potassium chloride extractable P (KCl-Pi): This is the loosely-bound inorganic P fraction that is potentially available to algae.

- Sodium hydroxide extractable P (NaOH-Pi and NaOH-Po): NaOH-Pi is the iron- and aluminum-bound inorganic P, and is also known as non-apatite inorganic P (NAIP). The NAIP fraction is considered to be potentially available, at least in part, to algae. NaOH-Po is composed of some organic P fractions associated with humic and fulvic acids.

- Hydrochloric acid extractable P (HCl-Pi): This is calcium-bound P, also known as apatite inorganic P (AIP). This form of phosphorus is tightly bound in the sediment.

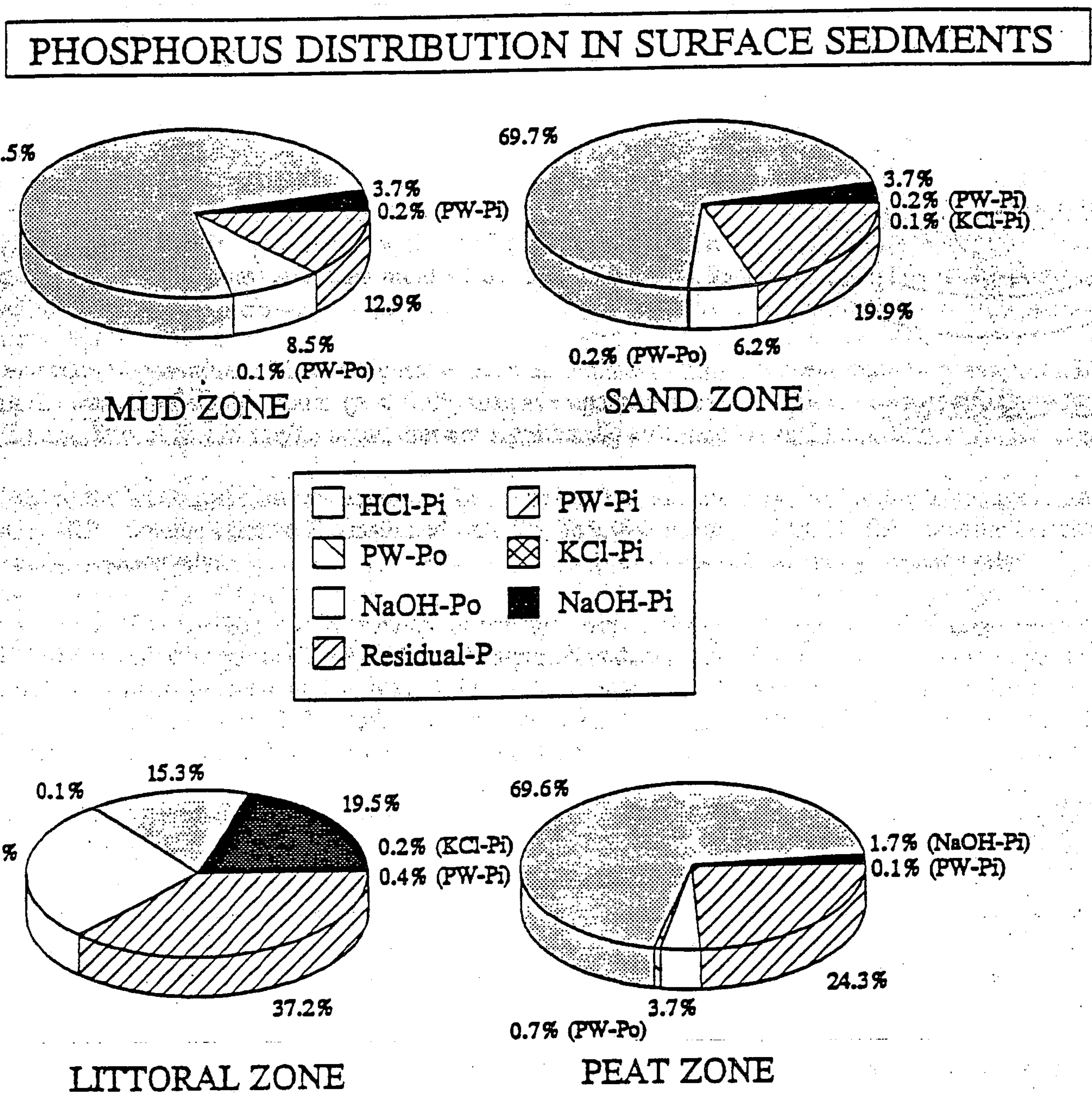


Figure 14. Relative distribution of P forms as determined by chemical fractionation in surface sediments of the mud and sand zones in Lake Okeechobee. PW-Pi = porewater inorganic P; PW-Po = porewater organic P; KCl-Pi = inorganic P extracted with potassium chloride; NaOH-Pi = inorganic P extracted with sodium hydroxide (Fe- and Al-bound inorganic P; also known as non-apatite inorganic P or NAIP); NaOH-Po = organic P extracted with sodium hydroxide; HCl-Pi = inorganic P extracted with hydrochloric acid (calcium-bound P; also known as apatite inorganic P or AIP); Residual-P = Total P minus other forms. See text for further explanation.

Total P in the porewater and sediment was also measured. Residual P (the portion not extracted with any of the above fractions) was calculated by subtracting the porewater and sediment fractions from the concentration of total P. The residual P fraction was assumed to represent organic P and any other mineral P fractions not extracted with NaOH or HCl reagents. Both residual organic P and calcium-bound P are relatively stable and not readily available for plant uptake.

Iron and Al-bound P (NaOH-Pi) constituted 18% of total P in the littoral zone sediments, but was less than 4% of total P in mud, peat and sand zone sediments (Figure 14). Littoral sediments also contained 25% of total P in readily hydrolyzable organic P form (NaOH-Po), whereas this fraction made up less than 8% of total P in the mud, peat and sand zones. Calcium-bound P (HCl-Pi) was proportionally higher in the mud and sand zones (44 and 53% of total P, respectively). Total P in the peat and littoral sediments contained only 18 and 15% as calcium-bound P. Peat sediments contained up to 75% of total P in organic form, while 36-44% of total P was present in organic form for other sediment types. Similar pool sizes were also observed in surface sediments of Lake Apopka, a hypereutrophic lake in central Florida (Olila et al. 1993).

Surficial peat deposits contained more total organic carbon (TOC; 40.2%) than sediments from other zones. Mean TOC concentrations for the mud (12%) and littoral deposits (15%) were not statistically different. Sands contained very little TOC (1.26%). Mud zone deposits were richest in total inorganic carbon (TIC), with a mean concentration of 2.9%. Peats (0.64%) and sand zone deposits (1.03%) contained less than muds, but were statistically indistinguishable from each other ($p > 0.05$). Littoral zone sediments contained very small amounts of TIC (0.05%). Surface mud zone deposits were rich in carbonates, and contained an average of about 24% carbonate by weight (as $CaCO_3$). Stratigraphic analysis of cores from the central mud zone indicates that carbonate-rich deposits underlie the muds in the center of the lake.

3.5 Nutrient Storage in Sediments

From the information on sediment chemistry and areal extent of various sediment types, the amount of TOC, total Kjeldahl nitrogen (TKN), and TP stored in the lake bottom can be estimated. About 4.53×10^9 kg of TOC is stored in the uppermost 10 cm of the sediments, and the TOC pool is divided among the various sediment zones as follows: mud (32%), littoral (34%), peat (19%), sand (15%). Total Kjeldahl nitrogen storage in surface sediments amounts to 3.81×10^8 kg, and is apportioned among the four zones in the following way: mud (29%), littoral (34%), sand (22%), and peat (15%). Total P storage in surficial sediments amounts to 2.87×10^7 kg. Mud and sand account for 42 and 41% of the TP storage, respectively, while littoral deposits (14%) and peat (3%) account for much smaller fractions of the total. On a mass basis, the overall TOC:TKN:TP ratio for the top 10 cm of the sediment bed is 158:12:1.

Over a recent 20-year period (1973-1992), the mean annual TP input to the lake was 5.18×10^5 kg. Over 80% of this input (4.18×10^5 kg) is retained in the lake (James et al. 1995a). Therefore, the TP storage in the surficial sediments is 55 times greater than the annual TP input and is currently increasing by an average of 1.5% per year.

Historical sediment and P accretion rates were measured using ^{210}Pb techniques. Although difficulties were encountered in interpreting ^{210}Pb data from some sites, reliable dating of sediments from the mud zone of Lake Okeechobee is possible. Details of this study are presented by in Volume V of this study (Engstrom and Brezonik 1991). Results show that sediment accumulation rates have increased during this century at all mud zone sites by an average of twofold. Sediment deposition increased from 300 $g/m^2 \cdot yr$ before the year 1910 to 700 $g/m^2 \cdot yr$ in the 1980s. Phosphorus accumulation rates have increased about fourfold since the 1900s (from about 250 mg $P/m^2 \cdot yr$ before 1910 to about 1000 mg $P/m^2 \cdot yr$ in the 1980s). Most of this increase has occurred during the last 40-50 years. Concentrations of all forms of sedimentary P are higher in the more recent sediments, especially NAI-P and organic P.

4.0 Physical Processes in Sediments

A large fraction of the nutrients entering the lake accumulate in the mud sediments. The dynamics of these fine sediments (sediment particles less than 30 microns in diameter) are complex and differ from those of coarse particles. Coarse sandy particles behave as individual grains. On the other hand, electrical charges on the surfaces of fine sediment particles can create cohesion of the primary particles to form flocs (or aggregates) which are larger in size but possess weaker bonding.

The objectives of this part of the study were to quantify: (i) the effect of hydrodynamics (currents and waves) on resuspension of fine sediments in the limnetic zone of Lake Okeechobee, (ii) the effect of wind-driven circulation on advection and mixing of fine sediments, and (iii) the effect of sediment dynamics on water quality. A combination of laboratory and field studies and numerical modeling was utilized to accomplish these objectives. For details, see reports by Mehta et al. (1991) and Sheng et al. (1991).

After bottom sediments are suspended into the bottom boundary layer by wave actions, they may be carried into the upper water column by turbulent mixing created by the vertical shear associated with the wind-driven currents. Depending on the strength of turbulent mixing, suspended sediments may reach the upper water column in a few hours or less. Once in the upper water column, suspended sediment particles can be transported by the advection and horizontal turbulent diffusion associated with the three-dimensional wind-driven circulation in the lake. They may also settle through the water column and be redeposited on the bottom. To quantify the three-dimensional sediment transport, it is necessary to first study the various processes involved which include advection, horizontal turbulent mixing, vertical turbulent mixing, settling, deposition, and erosion/resuspension.

4.1 Causes of Sediment Resuspension and Vertical Mixing

Analysis of field data of currents, waves, and sediments, plus modeling of the hydrodynamic and sediment processes, show that sediment resuspension is primarily caused by wave-induced bottom stress. As mentioned earlier, wind-driven currents are usually too weak to produce enough bottom stress for resuspension.

Measured suspended sediment concentration data show a dominant diurnal time scale related to the daily lake breeze activity. Waves created by the lake breeze generate significant bottom orbital currents and hence bottom stress. However, analysis of the high-frequency suspended sediment concentration data showed no dominant time period. This suggests that the effect of the waves is limited to a very thin boundary layer, which is underneath the location of the lowest OBS sensor (about 10 to 60 cm) above the bottom. Although seiche time scales are apparent in the energy spectrum of the mean currents, there is no seiche time scale in the mean suspended sediment concentration data. This suggests that seiche oscillation does not create sufficient bottom stress for sediment resuspension. Independent estimation of the bottom stress generated by seiche oscillation confirmed this observation. Proper instrumentation was unavailable for deployment within the thin wave boundary layer. Field measurements within the wave boundary layer are needed to quantitatively describe the sediment dynamics.

4.2 Vertical Turbulent Mixing

Once the sediments are resuspended from the bottom, they can be carried to the upper water column via turbulent entrainment and mixing. Turbulent mixing in the water column is generally created by the vertical shear in the wind-driven currents. Although the wind waves generate high-frequency orbital currents throughout the water column, there is usually little vertical shear in the water column, except within the thin wave boundary layer. Since the wind-driven currents are generally not very strong, suspended sediments may remain in the wave boundary layer for awhile before being carried to the upper water column by turbulence. Hence, there is usually a concentrated layer of suspended sediments over the bed within which a sharp density gradient may exist. This layer is similar to the thermocline in the water column, and is often called the "lutocline". Sediments in this layer behave more or less as a fluid, due to the continuous wave action on the bottom. During a lake breeze event, sediments are carried in the lutocline and then mixed up to the upper water column. Suspended sediment data show that the peak suspended sediment concentration near the lake surface usually lags the near-bottom peak concentration by 1 to 2 hours. During very strong wind events, turbulent mixing may increase to shorten the time lag between these peaks.

Vertical turbulent mixing is most vigorous in the surface mixed layer (usually between a few centimeters and one meter depending on wind) and the bottom boundary layer (usually a few centimeters for the wave boundary layer and up to one meter for the current boundary layer). Vertical mixing is difficult to measure, but can be estimated using the robust turbulence model contained in the circulation model (Sheng et al. 1992). Occasionally, during periods with little wind and significant stratification, flow in the lake may actually behave more like "laminar" (instead of "turbulent") flow. Compared to horizontal mixing, the effect of vertical mixing is much more pronounced due to the much smaller vertical length scale. The time scale of vertical turbulent mixing varies from a few minutes to more than two days, depending on the depth and wind condition.

4.3 Erosion/Resuspension Rate of Bottom Sediments

Suspended sediment concentrations measured in the field and numerical modeling were used to determine the erosion/resuspension rate of fine sediments. The erosion/ resuspension rate was found to be approximately a linear function of the excess bottom stress (the difference between the bottom stress due to waves and currents and the critical bottom stress which is 0.5 to 2 dyne/cm^2). The erosion constant, which is the proportionality constant between erosion rate and excess bottom stress, is on the order of 2 x 10^{-7} to 1 x 10^{-6} s/cm based on field data and numerical modeling (Sheng et al., 1992). As shown in Figure 15, the erosion rate determined in the field is different from

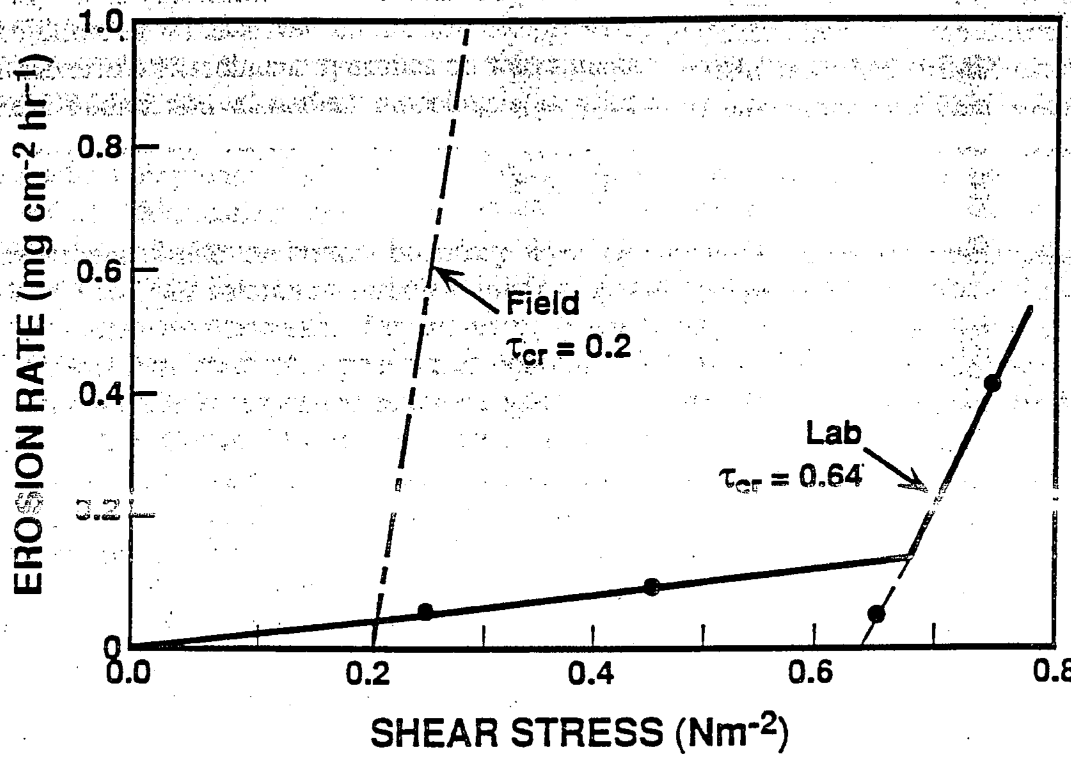


Figure 15. Erosion rates determined in the field (dashed line) and laboratory experiments (solid line). τ_{cr} is the critical shear stress below which no erosion occurs.

the erosion rate observed in the laboratory experiments. One possible explanation is that there was no wave activity in the laboratory experiments, resulting in lower erosion rates and less erosion/resuspension of sediments.

4.4 Movement of the Sediment Bed

Due to erosion/resuspension and deposition events, the bed can move up or down. Vertical movement of the bed was estimated by using the numerical model of sediment transport. Model simulation showed that during a typical diurnal cycle, vertical movement of the bed is generally less than 1 cm. However, this estimate needs to be validated with field data.

Under the influence of wind waves, the fluid mud at the bottom can experience oscillating porewater pressure and even some slight horizontal movement (Mei and Foda 1981). Mehta and Jiang (1990) measured the wave-induced acceleration of bottom sediments at a station near the southeastern shore of Lake Okeechobee. However, there was no evidence of significant bed movement in the horizontal direction, so the horizontal movement of the bed appears to be extremely small.

4.5 Settling and Deposition

The fine muddy sediments of Lake Okeechobee consist of particles with diameters of less than 30 microns (Hwang and Mehta 1989). The sandy sediments consist primarily of coarser particles of 100 to 1000 microns, (i.e., 0.1 to 1 mm). The muddy sediment particles have densities ranging from 1.05 to 1.3 g/cm^3, while the sandy sediment particles have a density of 2.56 g/cm^3. Thus, muddy particles have a much lower settling velocity (0.01 to 1 mm/s) than the sandy particles (1 to 150 mm/s), and the time scale for settling varies from several days to only a few seconds depending on wind conditions and the type of particle. In the central mud zone, fine suspended sediments may remain in the water column for many hours before settling out to the bottom. Because of the strong wave-induced bottom stress, open-water areas near the vegetation do not have much fine sediments on the bottom. The critical stress for sediment erosion in this zone is several orders of magnitude larger than in the mud zone. Settling velocity and critical stress for erosion are difficult to determine in the field, so they are usually determined in the laboratory. In this study, however, field data and modeling were used to quantify the critical stress for erosion.

Suspended sediments usually exist in various sizes. Particles in different size groups can actually migrate from one group to the other by flocculation and breakup. At low concentration (depending on the environment) and low turbulence, particles settle individually. At sufficiently high concentration and turbulence, particles can collide more frequently and form larger particles (flocs) due electrical forces among the particles. At the same time, larger particles can be broken up by the shear due to decreased floc strength. The laboratory experiment of Hwang and Mehta (1989) produced an empirical relationship between settling velocity and concentration, which partially contains the effect of flocculation on settling. Due to difficulty in measuring turbulence, the effect of turbulence on flocculation was not considered. To quantify the influence of turbulence, some direct measure of turbulence must be obtained in the field using recently available turbulence sensors.

Because particle size distribution plays such an important role in sediment transport and P adsorption/desorption, further study of flocculation dynamics would be useful. Hydrodynamic conditions affect the particle size distribution, so the distributions under laboratory and field conditions may be different depending on the mixing regime. To explain the flocculation dynamics in the lake, more field data and numerical modeling are needed to quantify the variability of particle size distribution *in situ* and provide guidance for laboratory experiments that attempt to simulate actual conditions.

Sediment deposition is the process by which the sediment particles travel through the various layers near the bottom (current boundary layer, wave boundary layer, viscous sublayer, and vegetation layer, etc.) to reach the bottom (Figure 16). Thus, deposition is different from gravitational settling in the water column and depends on the dynamics of the various near-bottom layers. This study used a deposition model developed by Sheng (1984) based on fundamental principles of fluid dynamics. Over the muddy bottom, during calm days, there is little turbulent mixing near the bottom and deposition could be very slow (on the order of hours to days) for fine particles within a certain range, thus favoring the formation of a sharply defined flocculent layer with high concentration, i.e., the lutocline. During slightly windy days, the flocculent layer or lutocline may thicken due to more sediment erosion. During very windy days, erosion is very vigorous, and the lutocline may become very thick and extend almost to the lake surface. To further understand the lutocline dynamics, measurements must be performed within a few centimeters from the bottom.

4.6 Three-Dimensional Transport of Suspended Sediments

Both horizontal advection and horizontal turbulent mixing can affect the long-term and large-scale transport of materials in the lake. Due to the relatively weak residual currents in the lake (currents averaged over long time periods, e.g., one day, usually on the order of a few cm/s), the time scale of horizontal advection is very long. Dissolved materials that enter into the lake may take 1-2 weeks to reach the Center Lake Station (about 20 km). For sediments, this time scale is expected to be longer, because sediments may go through a sequence of transport-deposition-resuspension-transport events before reaching the Center Lake Station. Long-term model simulations confirmed this expectation. Another important finding is that horizontal advection between the open water and the vegetation zone is usually weak, since the rather dense vegetation causes the adjacent open water to flow parallel to it. Thus, there is little exchange of sediment between the two zones.

Treatment of horizontal turbulent mixing in the numerical transport model is essentially a parameterization of the effect of small scale motion (scales less than the size

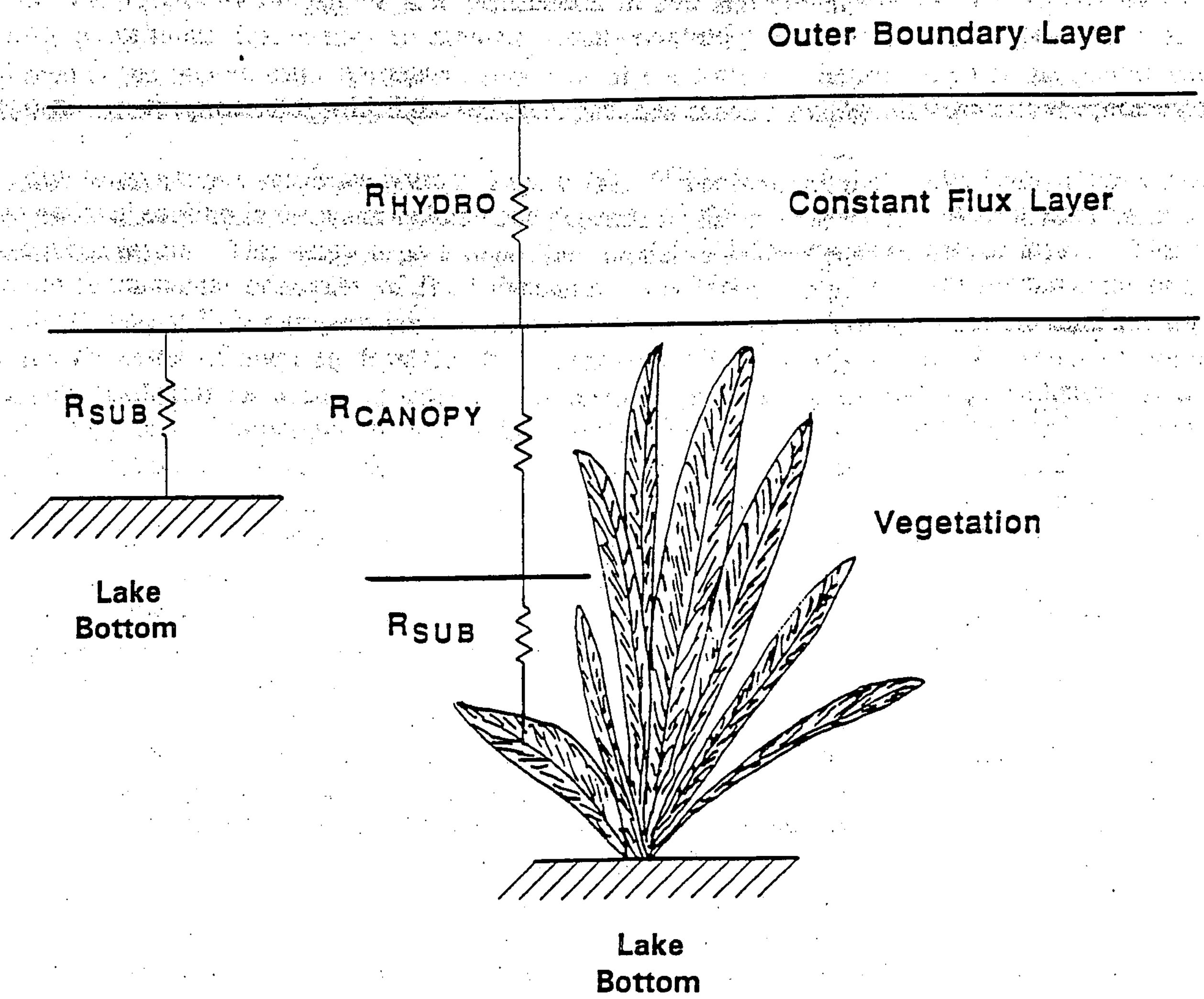


Figure 16. Various layers to be considered for deposition analysis of sediment particles.

of the numerical grid) on large scale transport, and also may include the effect of Langmuir circulation and breaking waves on horizontal mixing. The horizontal turbulent mixing coefficient scales with a length scale and a velocity scale, and is on the order of 1 to 100 m^2/s. Horizontal turbulent mixing is reduced in the vicinity of the vegetation zone, since the current and the horizontal scale of motion are usually significantly reduced. The time scale of horizontal turbulent mixing is usually longer than that of horizontal advection. Spatially uniform horizontal mixing coefficients were generally used in the modeling study.

Model simulations were performed over various periods ranging from one week to three months (Sheng et al. 1991c, Sheng et al. 1991f), using the continuous data collected at a few fixed stations and the synoptic data collected at many stations but at weekly intervals. Since there are many processes and model constants for parameterizing the processes, a sensitivity analysis was performed (Sheng et al. 1991b) on sediment transport modeling. Results showed that the 3-D model could simulate the sediment transport during the Spring 1989 survey quite well, provided proper bottom sediment conditions were specified for the entire lake. Examples of model simulations are shown in Figure 17 for the suspended sediment concentration at the Center Lake Station (Station C), and in Figure 18 for near-bottom suspended sediment concentration over the entire lake at an instant of time. The simulations appear to be reasonable. Sensitivity runs revealed that tributary sediment loading contributes little to the suspended sediment concentration in the central part of the lake. The horizontal advection and vertical turbulent mixing by wind-driven currents are primarily responsible for distributing the suspended sediments around the lake. Neglecting the advection by wind-driven currents produced too much vertical mixing of suspended sediments in the water column. These results also indicated the importance of getting accurate wind measurement for model simulations. Details of the sediment transport model are given in Sheng and Chen (1992).

4.7 Laboratory Experiments and Other Field Experiments on Sediment Dynamics

Other laboratory and field research was conducted to determine the physical processes related to sediment dynamics. Results of these studies are presented by Hwang and Mehta (1989) and Mehta and Jiang (1990) in Volume IX.

The total amount of mud in the lake bottom was estimated to be 193 x 10^6 m^3, with the thickness ranging from a few centimeters at the periphery to over 75 cm in the deep center of the lake. The distribution of bottom sediments suggests that the northern tributaries have supplied most of the mud and sand. Based on density profiles obtained in the field, the presence of a fluid mud layer (0-10 cm) in the upper part of the core was established. This is consistent with the numerical modeling study of Sheng et al. (1992) which indicates the existence of a fluid mud layer (0-20 cm) at the bottom of the water column.

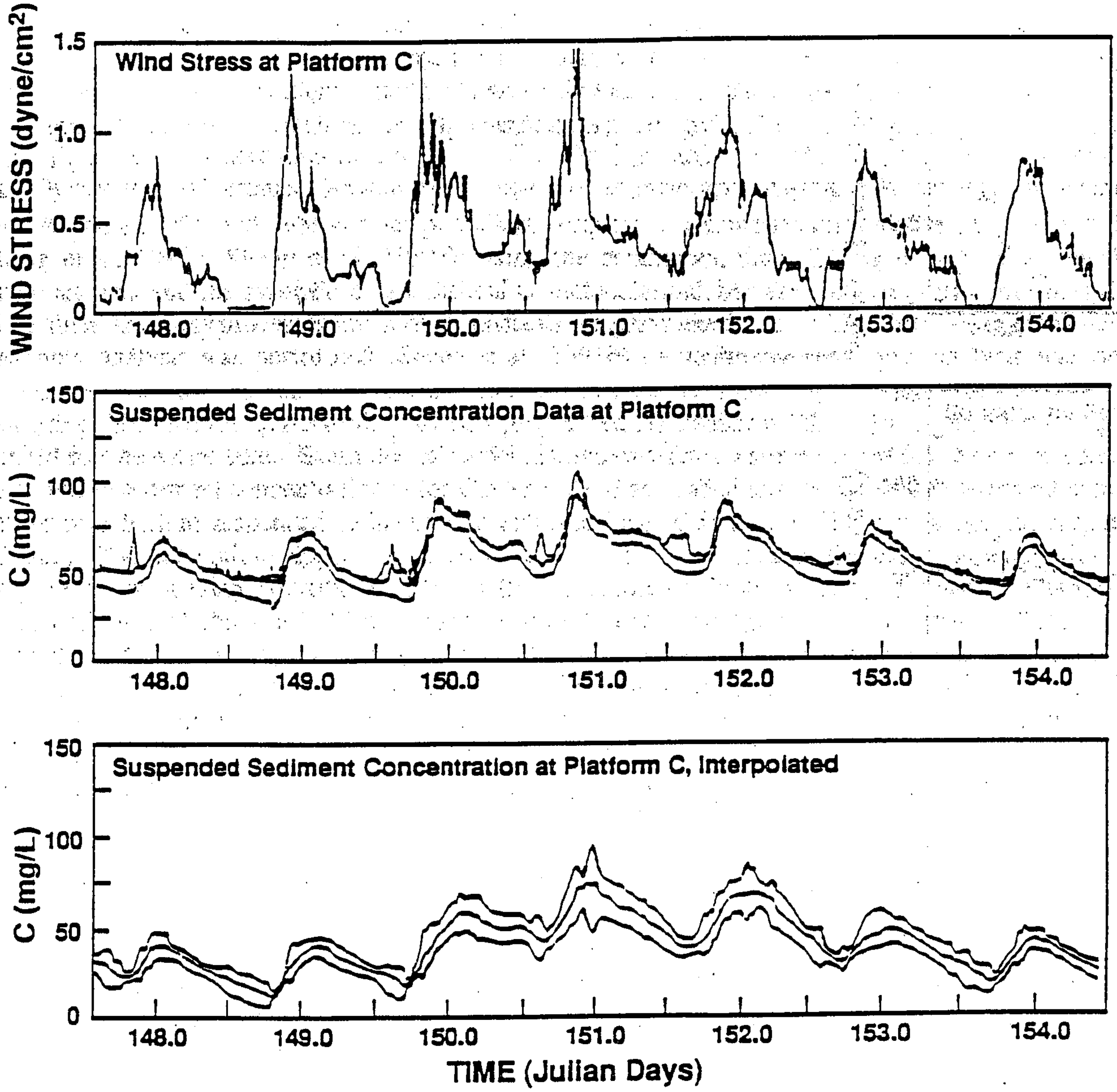


Figure 17. Wind stress (top panel), measured suspended sediment concentration at three depths (middle panel), and simulated suspended sediment concentration at three depths (bottom panel) at Station C (Center Lake Station) in Lake Okeechobee during a one-week period.

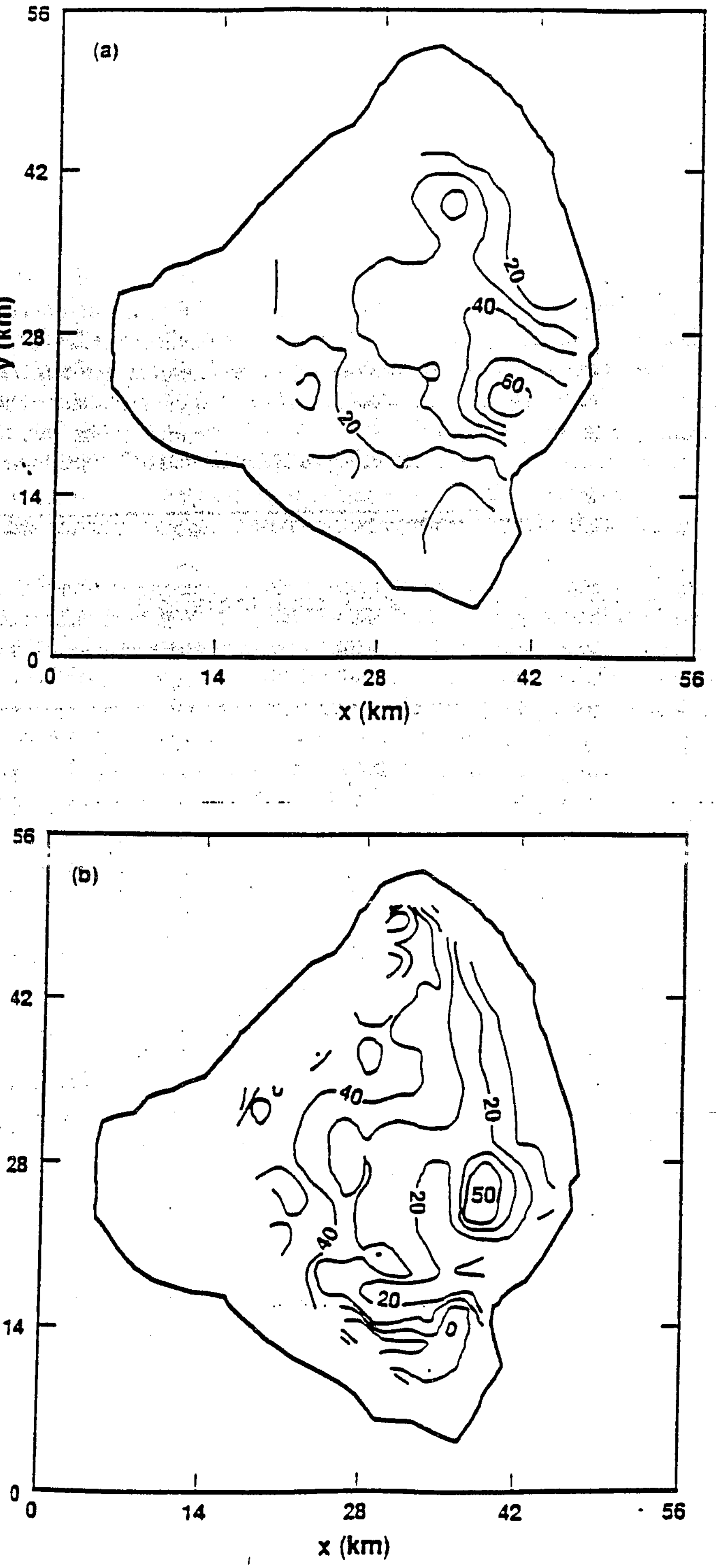


Figure 18. Near-bottom suspended sediment concentrations for the second week of Spring 1989 field experiments: (a) measured data, (b) model results.

Additional field experiments confirmed that the top 10-20 cm of the mud sediments are kept in fluidized state due to wave action that may be too weak to cause resuspension. The thickness of the fluidized mud layer varies with the wave condition and water depth. The thickness of the mud layer is comparable to that in Lake Apopka (Sheng: ongoing study of Lake Apopka). Particles from the mud zone generally range from 1 to 30 microns in diameter. The particle size distribution varies significantly with location and time, depending on sediment type and water turbulence.

Settling velocity appears to be independent of the suspended sediment concentration at low concentration (about 100 mg/L), but increases with higher concentrations up to about 2 g/L, apparently due to the influence of flocculation. Above 2 g/L, the high concentration of suspended sediments hinders the settling velocity. The 3-D sediment transport model of Sheng et al. (1991a) uses an empirical method of parameterizing the effect of flocculation on settling. The flocculation process in the water column is complex, but very important in determining the surface area of suspended sediment particles available for desorption/adsorption of phosphorus. Thus, it should be studied with a combination of numerical modeling and laboratory and field experiments.

5.0 Biogeochemical Processes in Sediment and Water Column

The specific objectives of the studies on biogeochemical processes (Volumes III and IV) were to: (i) determine the P retention/release capacity of bottom sediments, (ii) establish the role of redox potential and pH on phosphate solubility, (iii) determine the rate of organic P mineralization on P release, (iv) evaluate the rates of P exchange between the sediment and the overlying water column, and (v) establish the role of sediment resuspension on P release into the water column.

5.1 Phosphate Sorption by Sediments

The capacity of bottom sediments to retain or release P to the overlying water column is controlled by the sediments' physico-chemical properties. In this study, the ability of sediments to regulate dissolved inorganic P in the overlying water was evaluated by a series of batch incubation experiments to determine P adsorption coefficients and equilibrium P concentrations (EPC) under a range of conditions likely to be found in the lake (see Reddy and Olila 1991 for details).

The EPC is the concentration at which P sorption by sediments is equal to desorption. The EPC values determined from sorption isotherms are used to determine the direction of P release between sediment and the overlying water column. The EPC values of surface (0-5 cm) mud sediments were in the range of 1-56 μg P/L. Ranges of EPC for other sediments were: peat (1-70 μg P/L), sand (1-59 μg P/L), and littoral (1-669 μg P/L). The soluble P concentration of the open lake water is generally <50 μg/L, and when the EPC of sediment is greater than the SRP of the water column, the sediments may function as source of P. A wide range in EPC values suggests that the capacity of sediments to retain P is spatially highly variable.

The phosphate retention capacity of the sediments was in the order of mud > littoral > peat > sand. The P adsorption coefficient or buffer capacity (ratio of P sorbed on solid phase to the P in solution) for surface sediments was in the order of littoral > mud > peat > sand. High adsorption coefficients indicate strong P retention by sediments and low solution P concentration.

In mud sediments, significant correlations were observed between the P adsorption maximum of sediments and related sediment properties such as oxalate and citrate dithionate bicarbonate (CDB) extractable Fe and Al, and exchangeable Ca and Mg. For peat and sand sediments, these relationships were not significant. However, in sediments at major inflows to the lake and littoral zone, the CDB and oxalate extractable Fe and Al also showed positive relationships with P adsorption maximum. The significant positive relationships suggests that Fe and Al were controlling the P sorption in mud sediments. The ratio of CDB to oxalate extractable Fe was about 1:1 suggesting that most of the Fe is in amorphous form. These relationships suggest that under anaerobic conditions, presence of ferrous hydroxide ($Fe(OH)_2$) results in more sorption sites, and may regulate P retention. Other possibilities include the formation of vivianite

$(Fe_3(PO_4)_2 \cdot 8H_2O)$ in the sediment. A significant relationship between exchangeable Ca and P sorption capacity also suggests that calcium compounds (such as $CaCO_3$) are actively involved in P retention in these sediments.

On an areal basis, surface sediments (0-5 cm) of the mud zone have a maximum P retention capacity of about 105 kg P/ha. The P retention capacity of the sediments can be significantly altered by water column depth, biogeochemical activity in the water column, and diffusion of P from the water column to the sediment surface.

Batch incubation experiments have shown that Lake Okeechobee sediments can assimilate large quantities of P under long-term loading conditions. Sediment porewater P decreased dramatically with time in P loaded sediments under experimental conditions. At concentrations of 10 mg P/L of sediment, the porewater P decreased to below 0.1 mg P/L in the mud, littoral, and sand sediments after only six months. Even at concentrations of 100 mg P/L sediment, the dissolved P decreased to around 1 mg P/L in the mud, littoral and sand sediments. The rate of P assimilation in peat sediments was much slower, with porewater concentrations of 1.5 and 29 mg P/L after one year in the 10 and 100 mg P/L sediment treatments, respectively.

The P concentration in the sediment porewater is regulated by sorption, desorption, or dissolution reactions. The removal or sorption of P from solution by the solid phase of the sediments is largely regulated by the presence of amorphous oxides of iron and aluminum. The mud zone sediments contain high levels of amorphous Fe and Al that play a significant role in P retention. In addition, sorption and precipitation of inorganic P by $CaCO_3$ can be very significant in sediments. In Lake Okeechobee, the mud sediments are highly reactive and have a large capacity to retain P. Because of their reactive nature, these sediments can potentially function as a sink for water column P.

5.2 Effect of Redox Potential and pH of the Sediments on Phosphorus Solubility

In shallow lakes, surface layers of the bottom sediments can remain oxidized (aerobic) as a result of oxygen diffusion through the water column and mixing during sediment resuspension. The thickness of this oxidized layer is dependent on the oxygen consumption rate in the sediment. The oxidized (aerobic) and reduced (anaerobic) conditions of the sediments can be characterized by redox potential (Eh). Sediment Eh values of >300 mV represent aerobic conditions, while the values in the range of 300 to -250 mV represent anaerobic conditions. In Lake Okeechobee, sediment Eh is typically high at the sediment-water interface and decreases with depth (Moore and Reddy 1991).

Water-soluble P levels in mud zone sediment were about an order of magnitude lower under oxidized conditions than under reduced conditions (Figure 19). Decreases in water-soluble P concentrations under oxidized conditions were coupled with decreases in dissolved Fe, indicating that precipitation of ferric phosphate or phosphate adsorption by Fe oxides was the removal mechanism. Mineral equilibria calculations indicated that

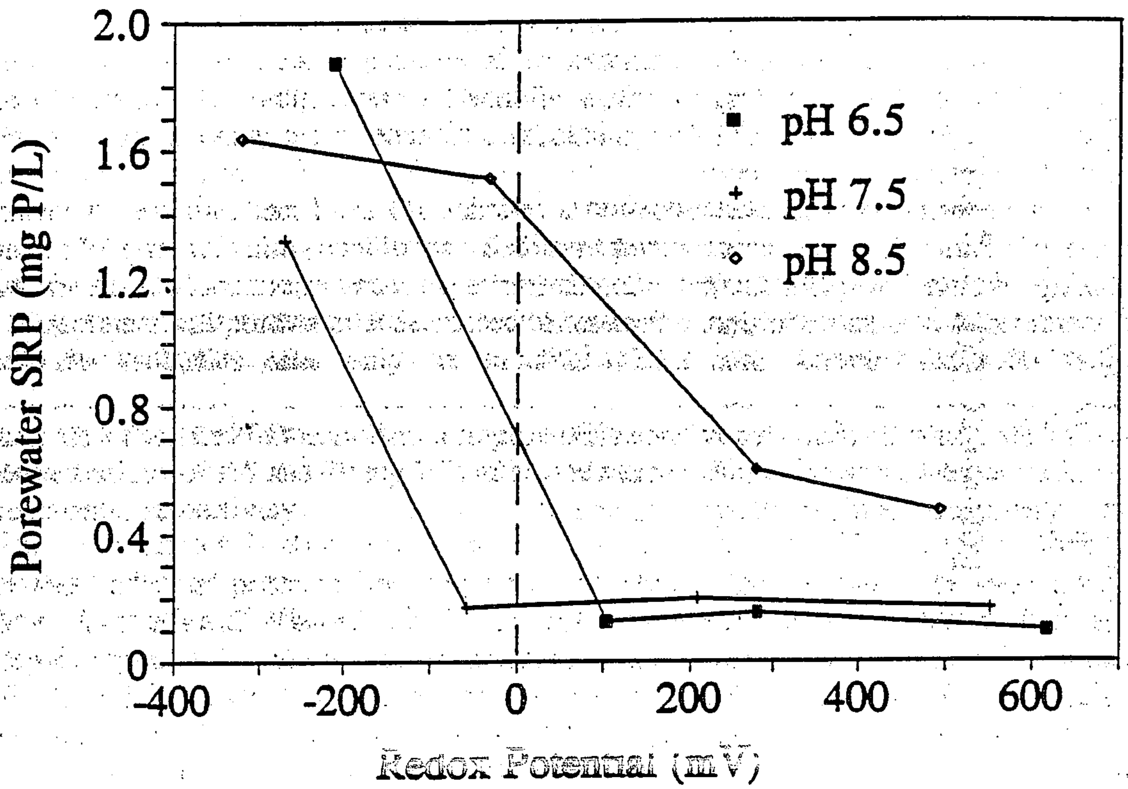


Figure 19. Phosphorus solubility as a function of pH and redox potential in the mud zone sediment.

B-$Ca_3(PO_4)_2$ or whitlockite governs water-soluble P under reduced conditions. This hypothesis was substantiated by P fractionation data that showed Fe-bound P increased under oxidized conditions at the expense of Ca-bound P. Based on these results, iron controls the behavior of P in Okeechobee sediments under oxidizing conditions, whereas calcium phosphate mineral precipitation governs P solubility under reducing conditions. These studies indicate that the surface sediments in the mud zone when exposed to oxygenated water column conditions are highly reactive and adsorb P to Fe components of the sediments.

5.3 Mineralization of Sediment Organic Phosphorus

Although organic P frequently comprises a major fraction of the P in both fresh and marine waters, little is known of the nature and behavior of these compounds. In Lake Okeechobee, organic P comprises 15-75% of the total P in the sediments (Figure 14). While most of these naturally occurring organic P compounds are associated with dead particulate biological components, a significant fraction exists in the "dissolved" phase (Messer and Brezonik 1978). Therefore, it seems that mineralization of organic P could contribute significantly to the overall P flux from the Lake Okeechobee sediments. Consequently, this study attempted to identify important dissolved organic P compounds and determine their biological availability.

Using chemical extraction procedures, four pools of organic P were identified: (i) labile organic P associated with microbial biomass, (ii) moderately labile organic P, (iii) moderately resistant organic P (fulvic acid P), and (iv) highly resistant organic P (humic acid P and residual organic P not extracted with HCl and NaOH). [For details, see Reddy and Ivanoff 1991 (Vol. III).]

Aerobic decomposition of organic matter resulted in more P accumulation in microbial biomass than did anaerobic decomposition. During a 100-day decomposition study, about 16 and 36% of organic P was released as inorganic P from mud and peat sediments, respectively. Significant correlations of various P pools with total organic P suggest that these pools are interrelated and are an integral part of organic matter decomposition in sediments. A large proportion of organic P is present in humic and fulvic acid P fractions, suggesting their slow bioavailability. The amount of inorganic P released during anaerobic decomposition may be very small and is limited by the supply of electron acceptors and the nature of organic matter present. Solubility and release of low molecular weight organic P compounds was shown to increase with decrease in Eh. Decomposition of organic matter may have several indirect effects on P release. For example, accumulation of organic acids and chelation of metals may solubilize insoluble Ca, Fe, and Al phosphates. Anaerobiosis increases humic and fulvic acid fractions which may form a protective surface over colloidal Fe/Al-oxides and may result in reduction in phosphate adsorption.

Among the various organic P pools, bicarbonate-extractable P, which represents microbial biomass P, is the most dynamic and may be the most significant pool. In sediments, the biomass pool of P can also be comprised of benthic organisms other than bacteria. Because of rapid turnover of the biomass, this pool of P may constitute a significant portion of P flux from sediment to the water column. In Lake Okeechobee, organic P mineralization can contribute significantly to the overall P flux from sediments, especially in the mud and peat zones. Organic P from sand zones may not significantly contribute to the overall inorganic P pool. Organic P contribution to P flux from littoral sediments is highly variable both spatially and seasonally.

5.4 Phosphorus Exchange Between Sediment and the Overlying Water Column

Phosphorus flux from Lake Okeechobee sediments is very sensitive to changes in the O_2 status of the overlying water, with anaerobic conditions resulting in extremely high rates of P release (Moore et al. 1991). Average P flux from mud zone sediments was 0.70 mg P/m^2•day (0.14 to 1.89 mg P/m^2•day), compared to peat (average = 0.91; range = 0.16 to 2.22), sand (average = 0.29; range = 0.11 to 0.52), and littoral (average = 1.09; range = 0.64 to 1.54) sediment zones. The P flux from the sediments at major inflows to the lake also varied; Taylor Creek (average = 3.18 mg P/m^2•day; range = 0.40 to 5.79 mg P/m^2•day), Kissimmee River (average = 0.74; range = -0.46 to 3.35), Fisheating Creek (average = -0.44; range = -0.22 to -0.86).

Phosphorus flux from intact sediment cores was several times higher when the overlying water was anaerobic rather than aerobic (Figure 20). Although there were steep porewater SRP gradients in Okeechobee sediments (varying from 0.1 mg P/L at the sediment/water interface to over 1 mg P/L at the lower depths; Figure 21), P flux was not regulated by such gradients. The lack of dependence of P flux on SRP gradients found in this study is indicative of the role redox reactions (involving Fe) play with respect to P chemistry in the top few cm of the sediment. The most probable mechanism of SRP control by Fe in this layer is the formation of an amorphous ferric phosphate mineral phase. The mechanism controlling SRP concentrations under anaerobic conditions in Lake Okeechobee sediments appears to be Ca-P precipitation, although it is possible that P precipitation by Fe(II) also could be occurring. The experiments on the redox chemistry of Lake Okeechobee sediments, as well as the P flux studies under anaerobic conditions, would indicate that the total P maxima observed near the sediment-water interface in Okeechobee may be the result of diagenetic processes (P precipitation). Redox potential and water soluble P profiles obtained from intact cores and porewater equilibrators indicate that surface sediments are oxidized and low in soluble P, whereas subsurface sediments are reduced and higher in water soluble P. Therefore, this study concludes that short-term flux is mediated by Fe, whereas on a longer time scale (i.e. weeks), Ca phosphate burial is the dominant P removal mechanism (Figure 22).

The possible significance of co-precipitation of inorganic P by $CaCO_3$ is important because James et al. (1995b) observed that alkalinity and Ca in the water column declined over a recent 20-year period, while TP concentrations rose and net P sedimentation rates decreased. They found a strong negative relationship between Ca and

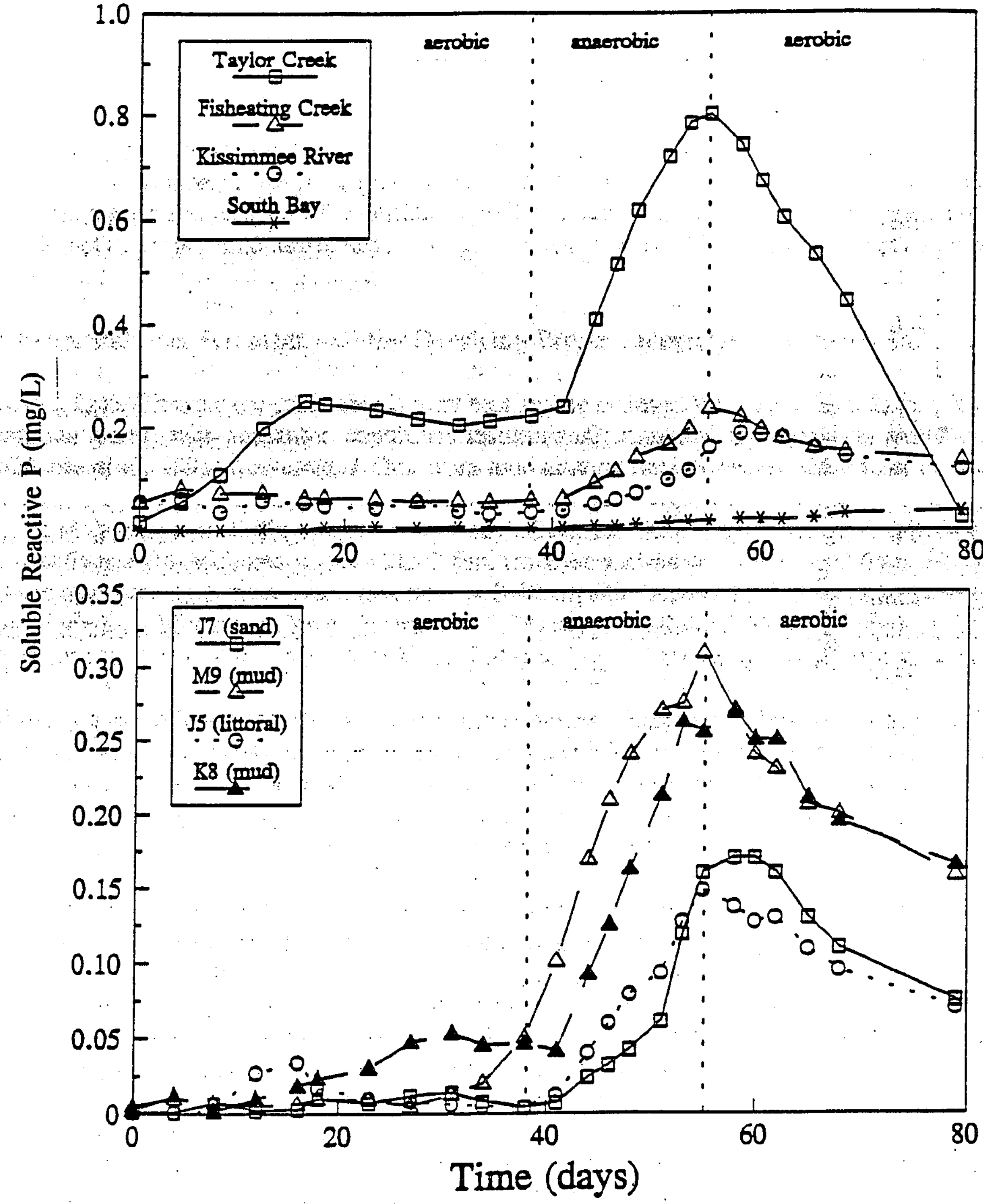


Figure 20. Phosphorus flux under aerobic and anaerobic water column conditions.

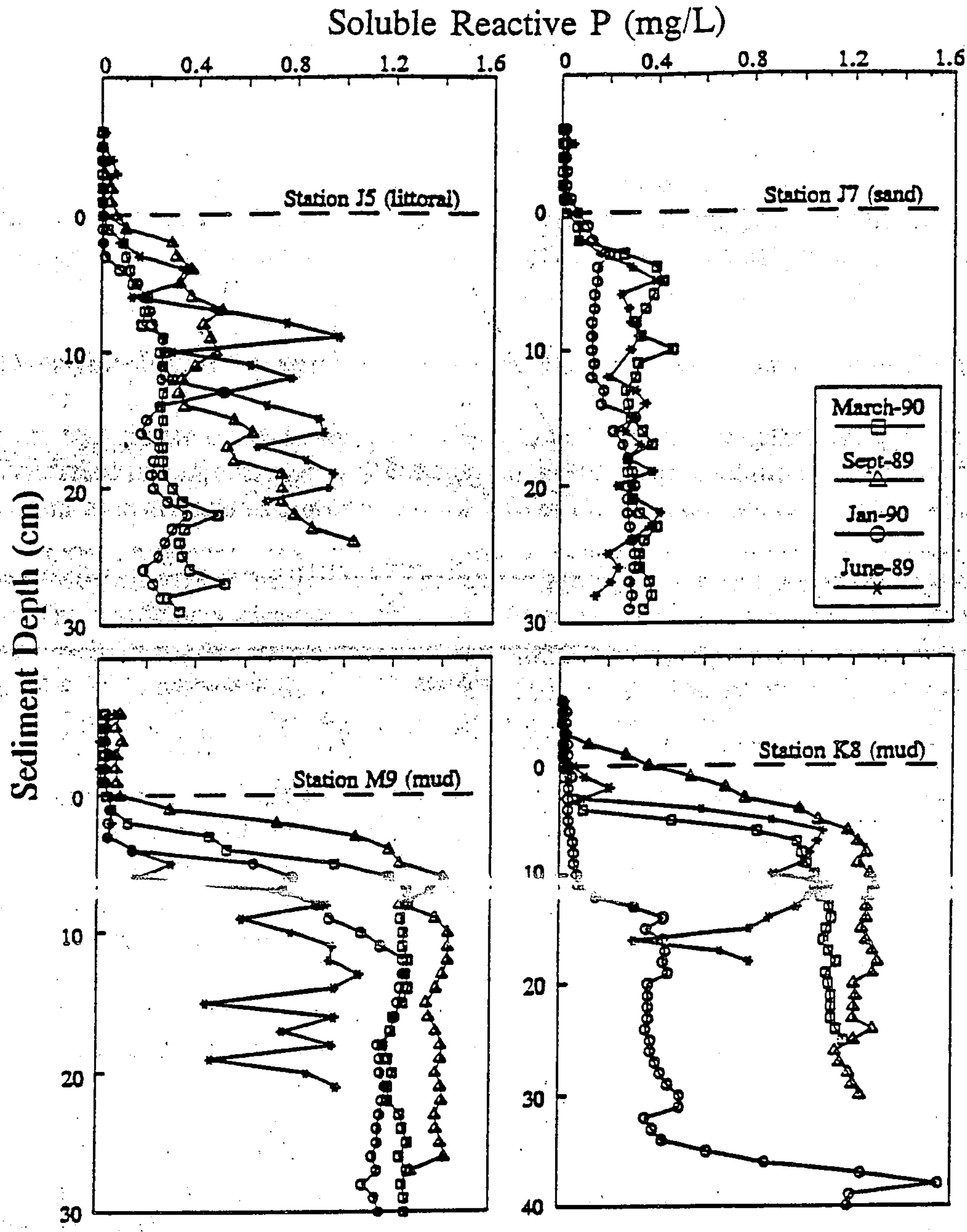


Figure 21. Distribution of dissolved phosphorus in the sediment and water column at selected stations in Lake Okeechobee.

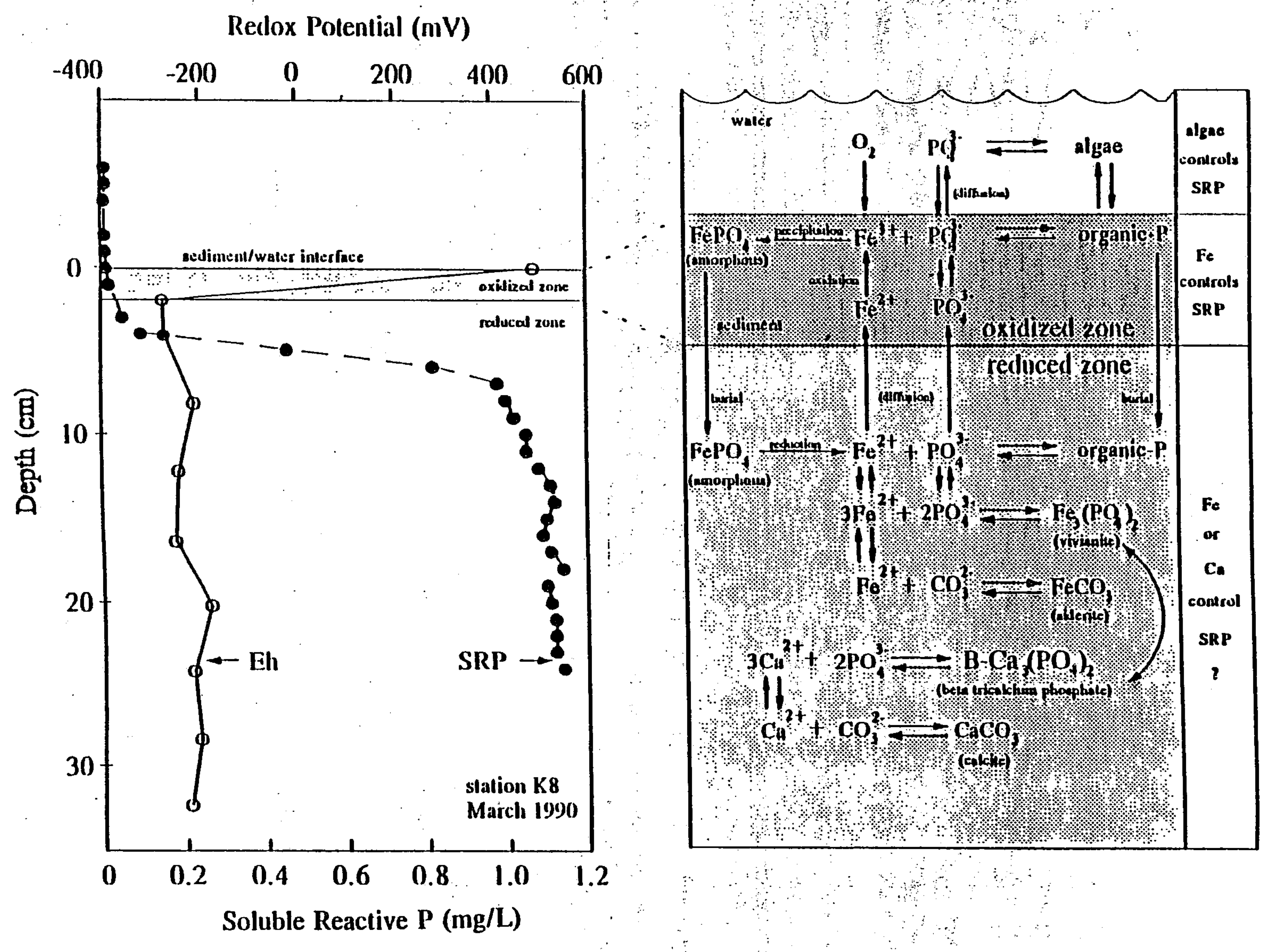


Figure 22. Redox profile, porewater phosphorus distribution, and associated biogeochemical processes in the sediment and water column (mud zone) of Lake Okeechobee.

TP concentrations and a strong positive relationship between Ca concentration and net P sedimentation rate. This suggests that the decline in Ca concentrations may have directly affected the net P sedimentation rate, and thus maintained TP concentrations at a higher level. The cause of the declining trend in Ca concentrations is currently unknown.

Phosphorus flux from sediments also can be influenced by bioturbation in the surface sediments. The effect of benthic organisms (bioturbation) on solute diffusion coefficients (D_e) was quantified using laboratory batch studies (Van Rees et al. 1991a). Estimates of D_e using mud and littoral zone sediments with benthic populations (predominately oligochaete species) were 1.6 to 15 times higher than those in sediments without any benthic organisms. Mud sediment cores of varying surface area had similar estimates of D_e for tritium; however, increasing surface area in littoral sediments resulted in increasing estimates of D_e, reflecting the heterogeneity of benthic population densities and activity. *In situ* estimates of D_e, however, are still needed in order to validate the laboratory estimates. These results suggest that the diffusive P flux measured using intact sediment columns under laboratory conditions may underestimate overall P flux by as much as 15 times. Future research should focus on the role of benthic activity on P release.

Laboratory studies using radioactive P (^{32}P) showed that the uptake of ^{32}P by sediments from the major inflows and areas representative of major sediment types was very rapid in the first two days for all cores after 14 days (Van Rees et al., 1991b). Uptake was still occurring after 14 days in sediments from Fisheating Creek and M17 (peat) whereas the other sediments had approached some "steady state" condition. These results suggest the potential for aerobic sediments to act as a sink, but do not elucidate the role of the sediments as a potential source of P to the overlying lake water.

Radioactive P applied to mud sediments showed that after 56 days the majority of the ^{32}P was Ca-bound, further supporting the importance of Ca in mud sediments in controlling P solubility over a long period of time. However, on a short-term basis, larger recovery of ^{32}P in NaOH extractions suggests the importance of Fe in regulating P retention. Results also show that up to 76% of the inorganic P present in the sorbed phase is rapidly exchanged with P in the liquid phase.

There are important management implications in the fact that internal P loads (i.e. sediment P flux) are roughly equivalent to external P loads (~ 1 mg P/m^2•day). These internal loads could increase dramatically if the surface oxidized zone in the sediments becomes reduced (anoxic). This could occur if external loading of organic matter increased dramatically. Flux studies also showed high P release from submerged aquatics occurred in the dark. Therefore, if the littoral zone were flooded quickly to a depth which resulted in plant mortality, large amounts of P release could occur.

5.5 Effect of Sediment Resuspension on Phosphorus Release

Dissolved P in the interstitial water of bottom sediments can be diffused/advected into the water column by molecular diffusion and bioturbation. On the other hand, particulate P can enter the water column by bottom sediment resuspension and bioturbation. The aerobic conditions that exist in the water column can have a significant effect on soluble P release during sediment resuspension.

Potential soluble P release rates under aerobic conditions were higher for sand and peat sediments than for mud and littoral sediments. Although the sand and peat sediments may show high rates of P release, their total P release capacity is generally low due to their inability to resuspend into the overlying water column. Laboratory studies showed the resuspension of the top 10 cm of bottom sediments from the mud zone into the overlying 30 cm of the water column resulted in an appreciable increase in soluble P. This release was essentially due to high total suspended solids (TSS) concentrations in the water column. Recently, this phenomenon has also been observed in the field at station L002. The P release due to resuspension was about 90 times the diffusive flux of P. However, these high TSS concentrations are not usually found in the lake. At low TSS concentrations (<2 g/L) and low dissolved oxygen (DO) levels (<1 mg/L) in the water column, the resuspension flux was about 6-18 times the diffusive flux measured for the same sediments. Under oxygenated water column conditions and at low TSS concentrations (<2 g/L) the suspended sediment particles decreased soluble P concentration in the water column as a result of adsorption and precipitation.

The results presented in this study raise questions as to the significance of wind-induced sediment resuspension in soluble P release into the water column. Results suggest that P release during sediment resuspension is short-term and occurs under low DO water column conditions. In Lake Okeechobee, it is likely that low DO and high TSS concentrations can occur in the water column during the night time, thus suggesting soluble P release. Lack of correlation between TSS and soluble P, based on the field data, suggest that the P reactions are governed by complex abiotic and biotic processes, and the kinetics of these processes are based on much shorter time scales than those used in this study.

Suspended solids and surface sediments show a high degree of variability with respect to P release capacity. The net increase in soluble P concentration of the water column was calculated for two conditions: (i) ambient water column conditions with TSS concentrations in the range of 21 to 74 mg/L, and (ii) maximum TSS concentration of 200 mg/L which can potentially occur during storm events (Reddy and Olila 1993). Water column depth was assumed to be 3 m and well mixed with no stratification. Among the sites examined, the suspended solids obtained from station L002 had a greater proportion of total P in desorbable form (up to 9% of total P). At this station, if all desorbable P is released, the SRP of the water column will be increased by about 15 μg/L. The net increase at other stations was in the range of 1.5 to 3.5 μg P/L. The SRP increase was estimated to be higher if the TSS concentration reaches 200 mg/L.

The TSS concentration at L002 and Taylor Creek inflow indicated an increase of 47 and 33 μg P/L, respectively. Even at high TSS concentration of 200 mg/L, the increase in SRP levels at some stations was appreciably lower (5.8 to 7.6 μg/L) than at other stations. It should be noted that the SRP released represents maximum values and under ambient conditions suspended solids seldom exceed 100 mg/L, thus the SRP release may be significantly lower. The suspended solids and surface sediments showed high affinity to P retention, thus they may actually remove P from the water column during resuspension events (for details see Reddy and Olila 1993). The kinetics of P release appear to be very rapid and occur at very low P concentrations, often below analytical detection limits. To further understand the mechanism of P retention and release, tracers such as ^{32}P must be used to follow the kinetics under in-situ conditions.

Abiotic processes include the complexing of P with ferric and calcium compounds, and biotic processes include algal and microbial assimilation (Figure 22). In order to establish the role of suspended particles on P release or retention, it is critical that these suspended sediments are characterized for various P forms and for associated physico-chemical properties. The kinetics of short-term P release between suspended particles and the water column needs further investigation. Field data are needed for diel variations in soluble P, DO, pH, and total suspended solids at the sediment-water interface to establish and confirm the significance of resuspension effects on soluble P release or retention.

Resuspension of P occurs as a result of resuspension of sediments and the subsequent release of P from the sediments (Figure 23). Based on field data of TSS and P in the water column and numerical modeling, Sheng (1993) estimated that the resuspension of fine sediments is a linear function of the excess bottom stress (which is the difference between bottom stress and the critical stress below which no resuspension takes place), with the proportionality constant being the erosion constant. Using the correlation between SRP and TSS presented in the previous sub-section, Sheng (1993) found that the resuspension flux of SRP is on the order of 92.7 mg/m$^2\cdot$day. He also used the "equilibrium partitioning" assumption and laboratory-determined adsorption constant to determine the resuspension rate to be between 6.74 and 17.2 mg/m$^2\cdot$day. In both estimations, Sheng assumed the bottom stress to be on the order of 1 dyne/cm^2 and a critical stress of 0.3 dyne/cm^2.

5.6 Phosphorus Retention by Sediments Near Major Inflows to Lake Okeechobee

Over 50% of the P inputs to Lake Okeechobee are contributed by five inflows: the Kissimmee River, Taylor Creek/Nubbin Slough, Fisheating Creek, and pump stations S-2 and S-3 in the Everglades Agricultural Area. The phosphorus retention capacity of sediments obtained near these inflows was highly variable. Phosphorus retention increased with P loading to the water column. The sediment cores near the Taylor Creek/Nubbin Slough inflow had high P retention capacity, with 80% of the P load (loading rate = 5.4 mg/m$^2\cdot$day) assimilated (for details see Reddy and Fisher 1991).

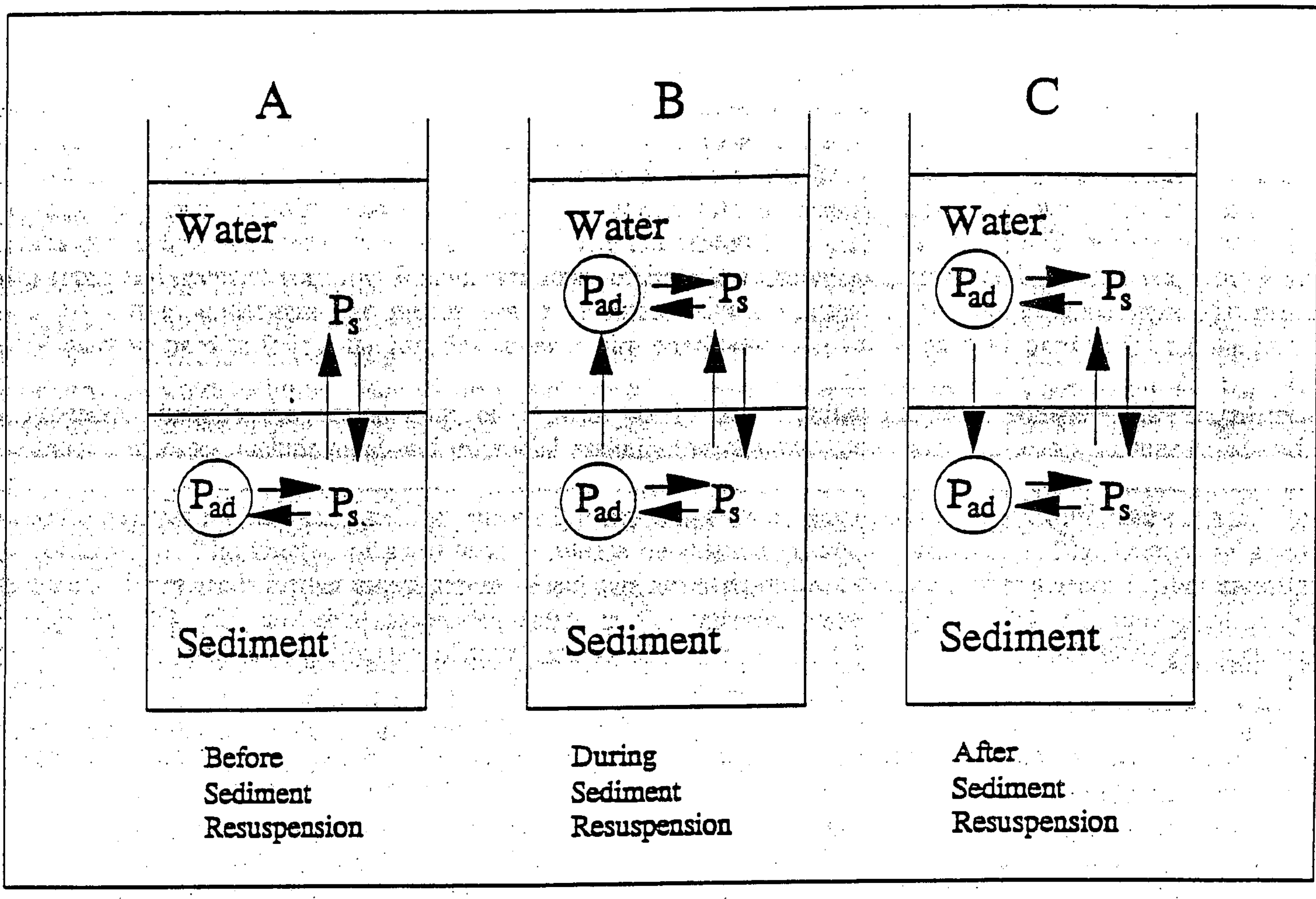


Figure 23. Schematic showing the phosphorus exchange processes during sediment resuspension events.

In laboratory experiments, Fisheating Creek, Taylor Creek, and South Bay sediments functioned as net sources of P to the overlying water column at low P loading (loading = <0.4 mg P/m$^2 \cdot$day). Kissimmee River sediments functioned as a P source at loadings of <0.7 mg P/m$^2 \cdot$day. The predicted equilibrium P concentrations (EPC_w) (EPC_w is defined as the concentration of P in the water column for which neither release nor retention of P occurs by the sediment) in the water column were 38, 100, 104, and 230 μg/L for South Bay, Fisheating Creek, Taylor Creek, and Kissimmee River inflow sediments, respectively. For example, if the water column P concentration is higher than EPC_w, the sediments will function as net sink. These results also suggest that if the external loads of P are reduced, the sediments will function as a source of P.

Phosphorus release from the sediments can be seasonally variable. During periods of high P loading from Taylor Creek/Nubbin Slough, the water column P concentration would increase, probably higher than the estimated EPC_w. During this period, sediments probably function as a net sink for P. However, during the periods of heavy rainfall, the dissolved P concentration of the lake inflows is diluted, and thus may not significantly increase water column P above the EPC_w. In some cases, the dilute inflow may actually decrease the dissolved P concentration of the water column. Under these conditions, sediments may function as a source of P to the water column.

5.7 Summary of Biogeochemical Processes

These results show that biogeochemical processes related to P cycling in Lake Okeechobee are dynamic and vary with both time and location. Sediments can function as sink or source of P to the water column, depending on the physical, chemical and biological conditions in the lake. Of the major four sediment groups identified in the lake, mud and littoral sediments are probably more important than sand and peat sediments with regard to P exchange with the overlying water column.

Although these results help to understand the biogeochemical processes in the bottom sediments and their relative importance to the overlying water quality, many of the process rates were measured under laboratory conditions. The absolute rates may be different under field conditions where they are influenced by various environmental factors. Many of the important processes occurring in the lake may be better understood by their evaluation at the sediment-water interface (possibly *in situ*) at a much-reduced time scale.

6.0 Biogeochemical Processes in the Littoral Zone of the Lake

The effects of macrophytes on nutrient cycling in lakes are not understood as well as nutrient cycling through pelagic organisms and profundal sediments (Granéli and Solander 1988). Interactions among aquatic plants, sediment, and water may result in an increase or a decrease of the P concentration in lake water. Because aquatic macrophytes occupy about a quarter of the lake's surface, it is important to understand the impact they might have on nutrient cycling in the littoral zone and pelagic waters. In the nearshore areas with macrophytes, two broad plant communities can be identified: a littoral region mostly consisting of submerged aquatic vegetation (SAV) and a marsh zone dominated by emergent vegetation. The SAV community was examined closely for its potential to contribute P to the pelagic waters of Lake Okeechobee. A common scenario is that of lake drawdown, which exposes the SAV community to desiccation, decomposition, and subsequent release of P during re-flooding.

The littoral region has a distinct vegetation distribution along most of the western shoreline. Bulrush (*Scirpus californicus*) is the emergent species which is exposed to the high energy waves of the open water. There are occasional patches of Illinois pondweed (*Potamogeton illinoensis*) lakeward from the bulrush. Inland (westerly) from the bulrush zone there is a narrow open water area with patches of cattail (*Typha augustifolia*) and maidencane (*Panicum hemitomon*). The open water areas contain dense stands of SAV, predominated by eelgrass (*Vallisneria americana*); hydrilla (*Hydrilla verticillata*) and pondweed are sub-dominants. Inland from the open water are dense stands of impenetrable cattail or, in the southwest, coastal plain willow (*Salix caroliniana*). The distances between the emerging bulrush and the beginning of the cattail and willow zones range from 120 to 340 m. This part of the littoral zone is termed the "fringing" littoral zone in this report. Two portions of this fringing littoral zone were studied: a northwestern section from Buckhead Ridge to Indian Prairie Canal, and a western section along Observation Shoal. For details, see Volume VI of the final report (Dierberg 1991).

Beginning with the cattail zone and proceeding (westerly) inward to the shore are large areas of marsh consisting of cattail, torpedo grass (*Panicum repens*), water lily (*Nymphaea odorata*), spike rush (*Eleocharis cellulosa*), and beak rush (*Rhynchospora tracyi*). The marsh is not included as part of the littoral zone in this report. It is treated as being hydrologically and materially isolated from the adjoining littoral zone.

There are other littoral areas in the lake besides the "fringing" littoral zone. These larger areas, which are in the vicinity of Fisheating Bay, Ritta Island, Kreamer Island, Torry Island, and King's Bar, also have dense stands of SAV.

Emergent macrophytes were not studied because: (i) they do not comprise the dominant plant community in the littoral areas adjacent to the open water; (ii) they have low nutrient content and have supporting tissue which is resistant to microbial attack, while submerged plants do not contain much cellulose and are more easily mineralized upon death (Twilley et al. 1986); (iii) they do not "turn over" as rapidly as the SAV and can withstand drawdown effects more than SAV; and (iv) from 25 to 50% of the above ground peak standing stock of P in perennial emergent macrophytes is returned to the rhizomes in late summer (Prentki et al. 1978; Davis and van der Valk 1983; Gopal and Sharma 1984; Morris and Lojtha 1986). In lieu of emergents, the SAV community was examined closely for its potential to contribute P to the pelagic waters of Lake Okeechobee through decomposition and seasonal P uptake and release. Bioassays also were conducted to test the availability of released P for algal growth.

6.1 Areal and Temporal Variations in Water Column Phosphorus

Intensive water sampling was conducted to determine spatial gradients and temporal variations along the littoral zone boundary. Water quality samples were taken along 24 transects in the two sections of littoral zone identified above during September 29-October 29, 1988 and May 20-June 25, 1989. These transects, each with three sampling sites, ran perpendicular to the edge of the littoral zone, with two stations in the littoral zone and one in the open water.

Significant aqueous P concentration gradients existed between the open water and the two littoral zone stations. The magnitude and direction of these gradients changed with the area observed and the period of sampling. Higher concentrations of SRP and TP were found at the littoral zone stations than at the open water stations, except for the northwest littoral zone during 1989, when the reverse was true. Differences in P concentrations between the two littoral zone stations were insignificant, indicating that sampling more than one littoral zone site along the transects was redundant. Also, high-frequency sampling showed that these gradients changed daily, indicating that P concentrations in the fringing littoral zone depend on short-term changes in wind and seiche conditions. However, because steep P concentration gradients between the open water and littoral zone existed even on windy days, hydraulic exchange between the two zones is probably minor except during episodic events. Most occurrences of higher total P concentrations in the littoral zone were probably due to resuspended particulate matter, which settled back to the bottom when wind velocities diminished and was not transported out of the littoral zone.

6.2 Phosphorus Storage/Release by Submersed Aquatic Vegetation

Eelgrass dominated the SAV community in the fringing littoral zone, comprising 81% of the total biomass (see Volume VI). It was less dominant in the SAV of Fisheating Bay. Hydrilla, pondweed, and *Nitella* also made up significant portions of the SAV community. Hydrilla exhibited the greatest change in tissue P concentrations (5-fold range over the annual cycle) and had the highest seasonal uptake rates (667 mg P/m^2 between January 21 and May 6, 1989). It was also the species that was most sensitive to changes in lake stage.

In laboratory tests, submersed aquatic vegetation in the littoral zone of Lake Okeechobee decomposed and released soluble P at rapid rates when the plant material was dried and then reflooded with lakewater. A simple exponential decay model accurately described the decomposition and P loss with about 50% initial P released in 7 to 28 days.

From 23-50% of the tissue P in dried eelgrass, hydrilla, and pondweed was released within five hours of reflooding. The released P was biologically available for the growth of an algal assay organism (*Selenastrum capricornutum*), indicating that a potential exists for algal growth stimulation. Yield coefficients, defined as the mass of algae (dry wt) produced per mass of P taken up, averaged 599.

Roots were the major site of P uptake in living SAV according to an empirical model. Contact of the decomposing SAV with sediment, such as what occurs under field conditions, resulted in one-third to one-half less P release than in the absence of sediment due to P immobilization by the sediment. In addition to attenuation by sediment, P released by decomposing SAV would be further subject to recycling by the periphyton community within the littoral zone and to co-precipitation with calcite or precipitation as apatite on the surfaces of the SAV. The exchange of water between the littoral and pelagic zones is probably low (except for episodic events), which would favor P recycling within the littoral zone while making the transport of significant quantities of P to the open water unlikely.

6.3 Phosphorus Export from the Littoral Zone to the Open Water

The probable low hydrologic exchange processes notwithstanding, the potential contribution of P from the littoral zone to the pelagic zone was calculated based on field- and lab-desiccated release experiments, *in situ* decomposition studies, and changes in SAV standing crops. Given a completely-mixed pelagic zone, a lake stage of 12.2 ft NGVD, and SAV coverage of 7,228 hectares, release and transport of P from SAV in the littoral zone could increase the pelagic TP concentration by a maximum of 11 μg P/L. This assumes no immobilization or recycling of the P within the littoral zone and complete transport from the littoral zone to the pelagic zone. When sediment immobilization is taken into account, then as little as a 1 μg P/L increase in pelagic TP concentration could be expected. Thus, the contribution of P to the limnetic zone from SAV senescence and decay would be negligible (Dierberg 1991). Numeric modeling of the circulation and transport of P in the vicinity littoral zone/open water boundary (Sheng et al. 1992) also showed that there was negligible transport of P from littoral zone to the open water, at least during low stage. As explained in Section 8.2, there is greater opportunity for littoral zone transport at higher lake stages, although no data exist to show that this actually occurs.

In conclusion, there is not enough SAV biomass within the littoral zone to contribute significant quantities of P to the pelagic zone either from senescence and decay or from massive herbicide application, freezing, or lake drawdown followed by reflooding, even under the most liberal assumptions (Dierberg 1992). Therefore, the mobilization and transport of P from this source is an unlikely cause of the lake-wide rise in TP concentrations. However, this apparently insignificant contribution to the lake-wide phosphorus pool does not preclude the importance of localized effects. It is known that chlorophyll *a* concentrations are typically higher in areas closer to shore than at mid-lake sites. These nearshore chlorophyll concentrations are correlated with lake stage, but the mechanism for this relationship is still unresolved and may be due to offshore influences as well as littoral zone processes (see Sections 8.2 and 9.2).

7.0 Modeling of Phosphorus Dynamics

As shown in the schematic representation of Lake Okeechobee's P cycle (Figure 2), most of the P processes (mixing, resuspension, deposition, settling, and diffusion) are affected by the hydrodynamic and sediment dynamics. Even the transformation process is affected by the hydrodynamics and sediment dynamics to some extent. Thus, field data were collected to quantify the effects of hydrodynamics and sediment dynamics on P dynamics. These data also facilitated the development and calibration of a P model for Lake Okeechobee. The following sections present a summary of (i) the water column P, and (ii) the P modeling.

7.1 Water-Column Phosphorus

During each of the 1988 and 1989 synoptic surveys conducted, water samples collected from 2 depths at 25 stations in the lake were analyzed for SRP, total P, and total suspended solids (TSS) (Sheng et al. 1989; Sheng 1993; Sheng et al. 1991b). Significant correlation between TP and TSS, with $r=0.77$, is shown in Figure 24. An equally significant correlation was found between PP (particulate P) and TSS, while weaker correlation (with $r=0.40$) existed between SRP and TSS. In addition, both TP and TSS responded to wind speed (Figure 25), suggesting that P concentration in Lake Okeechobee is significantly influenced by the hydrodynamics and sediment dynamics which are forced by the wind. Since more than 90% of the total P is in particulate form, both PP and TP correlated very well with the TSS concentration. High concentrations of TP and PP are generally found in areas with high TSS, where the muddy bottom is very thick. The correlation between SRP and TSS is weaker because of the dynamics of P exchange between particulate matter and water. Although TP and PP increases during a resuspension event, SRP may not increase immediately because the redox potential and ion concentration may not favor the release of P from sediments/aggregates into the water (see section 5.5 for more detailed explanation).

It is apparent that diurnal and weekly variations in P concentrations are primarily due to wind variations. The influences of the rivers and canals are not apparent. However, one should not conclude that rivers/canals do not contribute to the increased P concentration in the lake. Nor should one conclude that the lake "pollutes itself" regardless of how much P enters into the lake. As mentioned before, the transport of P from rivers/canals is a very slow process, involving sequences of transport-deposition-transformation-resuspension-transport events that may last for several weeks before the unit of P is deposited in the center of the lake.

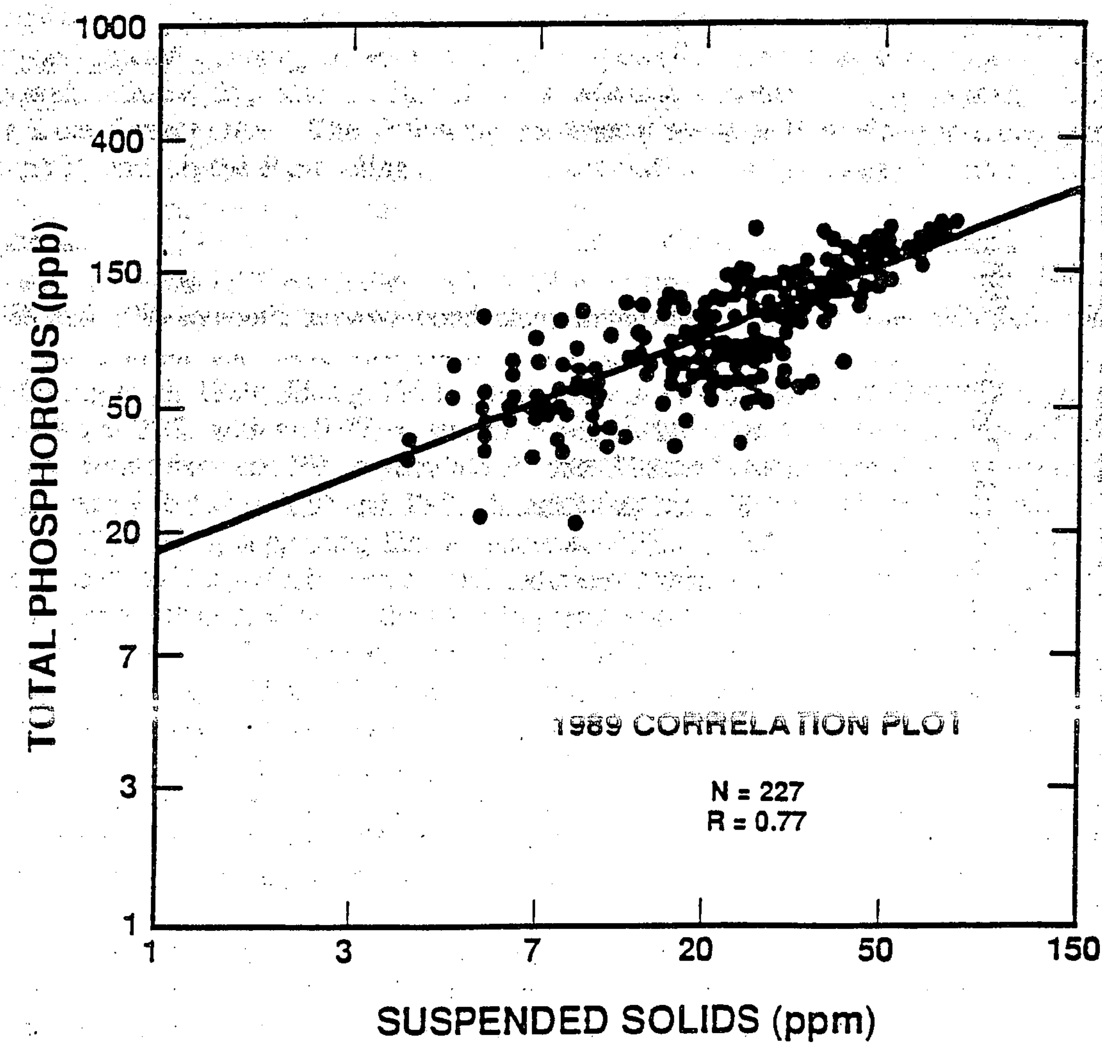


Figure 24. Correlation between TP and TSS measured in the 1989 synoptic survey of Lake Okeechobee.

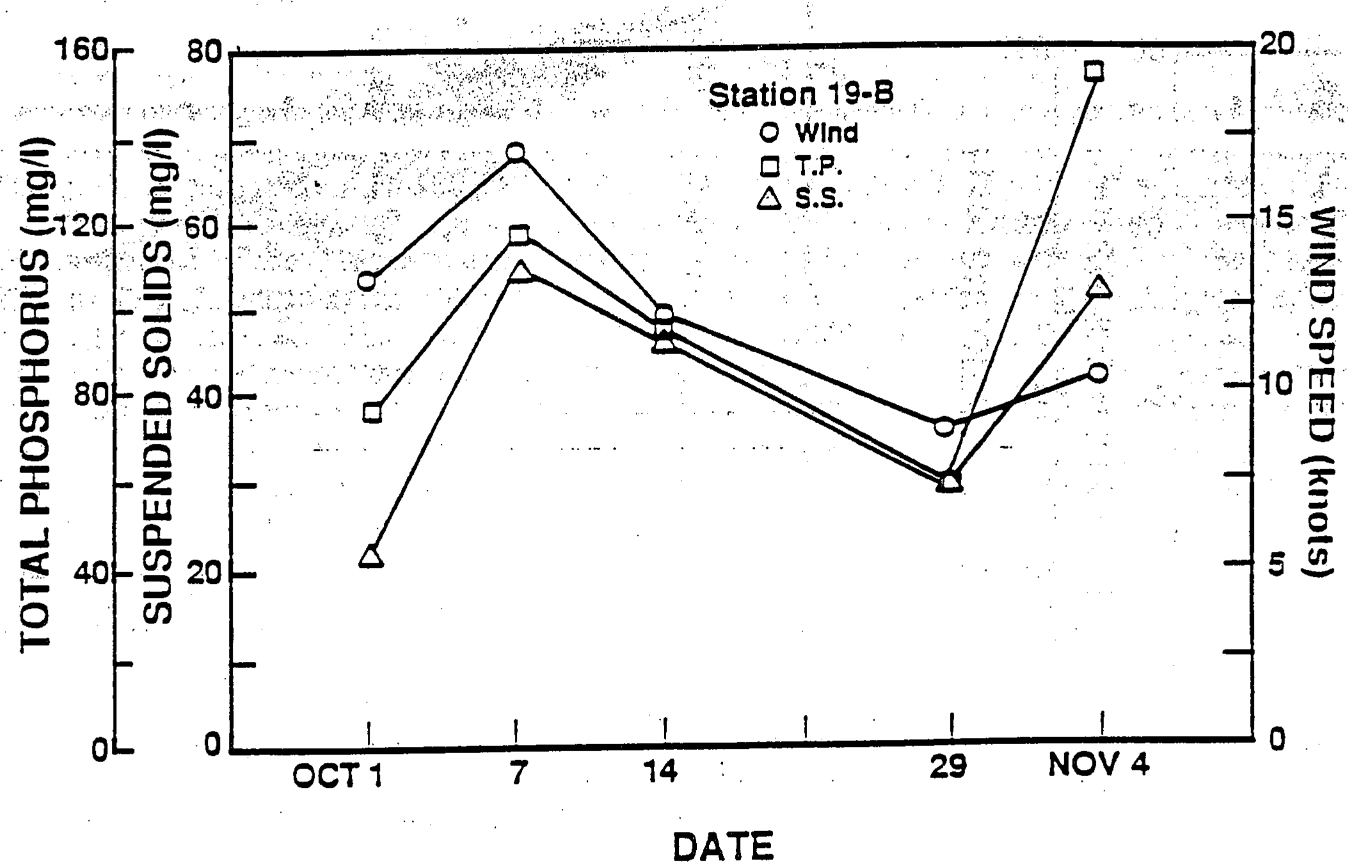


Figure 25. Wind speed, TP, and TSS measured at a mid-lake site (Station 19-B) during the 1988 synoptic survey (from Sheng et al. 1991e).

7.2 Phosphorus Modeling

7.2.1 Three-Dimensional Model

A three-dimensional model of P transport has been developed by Dickinson et al. (1992) by incorporating the P transformation into the sediment transport model developed by Sheng et al. (1991a). The model, LOP3D, uses the same grid structure and finite-difference formulation as the hydrodynamic and sediment transport models. The model includes the following water column P components: SRP, dissolved organic P (DOP), green algae (GRN), blue-green algae (BLU), zooplankton (ZOO), particulate inorganic P (PIP), and particulate organic P (ORG). Water column P has time scales comparable to the hydrodynamic and sediment scales, due to the significant influence of hydrodynamics and sediment dynamics on P dynamics. In addition, P dynamics of the bottom sediments are modeled. The diagenesis in the sediments, however, has much longer time scales compared to time scales in the water column P, due to the relatively slow process of diffusion and mixing in the sediment.

The 3-D model of P transport was run for three consecutive months to simulate the Spring 1989 scenario. Results from the 3-D wind-driven circulation model and the sediment transport model were used to drive the P model. Numerous model constants were determined by using best available information and the experimental data from the studies conducted by Reddy and associates (Section 5). Results of the three-month simulation of P transport in Lake Okeechobee are summarized in Sheng et al. (1991c). Despite the complexity, the model has been found to give useful results with relatively little computer time. The model has been used to evaluate the impact of reduced tributary loading on the P distribution in the lake. Unfortunately, due to the short duration of model simulation, no significant difference in water column P was detected in the majority of the lake, except in the vicinity of the Kissimmee River. With some additional effort to streamline the computer code, it is feasible to use the model for long-term (e.g., 1-3 years) simulations. As mentioned before, running the P model with the same time step and spatial grid as the hydrodynamic and sediment model eliminates the need for *ad hoc* assumptions, which are often needed in box models. Thus, the 3-D model is well-suited for investigating the detailed P dynamics in various zones of the lake.

7.2.2 Box Model

In addition to the 3-D model, Dickinson et al. (1992) also developed a box model (LOP0D) to describe P dynamics in the lake. The model contains five large boxes: north lake, mud zone, sand zone, South Bay, and littoral (macrophyte) zone. This model is very efficient and has been used for a nine year simulation. However, because of the crudeness in spatial and temporal resolution, this model is not suitable for quantitative and detailed simulations. Thus, results of this box model should be interpreted with caution and used as a qualitative guide for planning the more detailed 3-D simulations.

Nevertheless, based on a nine-year simulation using the box model, Dickinson et al. (1992) obtained the following results:

Particulate inorganic P (PIP) erosion	-220 $mg/m^2 \cdot yr$ (from sediment to water)
SRP diffusion	-59 $mg/m^2 \cdot yr$ (from sediment to water)
Algal settling	206 $mg/m^2 \cdot yr$ (from water to sediment)
PIP deposition	490 $mg/m^2 \cdot yr$ (from water to sediment)
Net TP exchange	418 $mg/m^2 \cdot yr$ (from water to sediment)

Thus, over the long-term the sediment acts as a sink, not a source of P to the water column in the lake. The box model (LOP0D) results gave diagenetic fluxes in the same range as those reported by Moore et al. (1991). Internal loading processes are dominant on a daily or weekly time scale, whereas external loadings are dominant when the time scale is months to years. Erosion/deposition is the dominant factor for short-term response, while wind mixing is critical to understanding the short and long-term P dynamics of the lake.

The water column P data (section 7.1) also showed that there is significant spatial variation in the P concentrations. The spatial scale of variation is rather fine compared to the size of "boxes" used in a typical "box model" for describing the water quality and ecology of Lake Okeechobee. If a box model is used to model the P dynamics of Lake Okeechobee, it would require measured P concentrations averaged over the large "boxes" in the lake. In order to obtain meaningful "averaged" P concentrations for model calibration, an exceedingly large number of samples may be required. Thus, although a box model may be easier to develop and much faster to run, it actually requires more data to calibrate than does a three-dimensional model. A three-dimensional model can actually use all the data collected at discrete stations without having to perform spatial averaging, so long as the grid size is not too large.

8.0 Summary of Modeling Results

In this section, the major findings are summarized in terms of the following questions: (i) What is the annual loading of P from bottom sediments to water column? (ii) Is there a significant P flux between the vegetation zone and the open water? (iii) How does lake stage affect P dynamics? and (iv) What is the influence of reduced tributary P loading on the P distribution in the lake?

8.1 Annual Loading of Phosphorus from Bottom Sediments to Water Column

Although the benthic P flux has been estimated in Section 5, providing an accurate determination of the total annual loading of P from the bottom sediments to the water column is more challenging and requires a comprehensive understanding of the phosphorus dynamics and the factors that influence it (e.g., redox potential, hydroxide concentration, pH, microbial biomass, chlorophyll-a concentration, etc.). This study has used state-of-the-art techniques to attempt to understand and quantify these factors and processes as completely as possible. It has shown that total and particulate phosphorus concentrations do respond to the hydrodynamics and sediment dynamics, but has yet to establish a cause-effect relationship that would allow a precise estimate of internal loading. Exact estimates would require more detailed study involving monitoring at extremely high frequency and resolution. In fact, a complete explanation of some processes would demand field instrumentation that does not currently exist. Nevertheless, based on the best data obtained from the field and laboratory experiments described above, the following discussion presents an approximate estimate of internal P loading that should be useful for management application.

As an example, a modest wind event might consist of a 3-hour resuspension period during which an average bottom stress of 5 dyne/cm^2 (requiring a wind speed of 10 m/s, i.e. 20 mph) is exerted on the muddy bottom area (approximately 20 km x 15 km). During this time, a total of 9.36 x 10^{10} g of fine sediments will be suspended into the water column. If these sediments are uniformly distributed throughout the water column over the entire muddy bottom, the suspended sediment concentration will increase by about 100 mg/L. This should lead in turn to an increase in TP concentrations by about 180 μg P/L, according to the correlation between TP and TSS. The total amount of TP transferred from the bottom sediments to the water column would be about 1.2 x 10^8 g (or 120 metric tons). In comparison to nutrient budget data presented by James et al. (1995a), this amount is 33% of the mean content of TP in the water column (367 metric tons) and 23% of the mean annual external TP input (518 metric tons).

Most of the resuspended particulate P will settle back to the bottom and will not be available for phytoplankton uptake. The amount of time that the particulate P stays in the water column depends largely on the size of the sediment particles. The coarse sandy particles may stay in the water column for less than half an hour, while the fine micron-size particles can stay in the water column for several days. Moreover, during the resuspension event, the DO concentration may be sufficiently high to prevent the desorption of P from the sediment particles. With the increased suspended sediment concentration, oxygen consumption may cause the DO level to decrease and allow the release of dissolved P. Part of the soluble P in the water column may diffuse back to the bottom sediments. Thus, the amount of SRP released into the water column during a typical wind event is expected to be only a small fraction (e.g. 1-10%) of the estimated 120 tons of TP. Much of the TP settles back to the bottom within a few hours after the event, and there may be a small increase in SRP concentration on the order of a few μg P/L (Reddy and Olila 1993). This scenario agrees with long-term water quality monitoring data, which show seasonal peaks in TP and SRP during the more windy months (Federico et al. 1981).

The annual loading of P from sediments to water column depends on the magnitude of the diurnal events. During a small event with rather low wind speed of less than 4-5 m/s (about 10 mph), resuspension of sediments is limited to the very top millimeter of the sediment column, i.e., the aerobic layer where desorption is unlikely to occur. Thus, resuspension of these sediments into the water column will not lead to release of SRP into the water column. During a major event with wind speed exceeding 15-20 m/s (34-45 mph), sediments will be brought up from the deeper anaerobic layer, where desorption is favored. Thus, the amount of SRP brought into the water column during one year are determined by the number of major wind events. Assuming there are five such major events in one year, the resuspension flux of SRP should be about 5 to 60 metric tons. The mean SRP content in the water column is 100 metric tons (James et al. 1995a, 1995b), so this flux represents 5 to 60 percent of the mean SRP content. The P loading from the sediments may vary significantly from year to year. During a calmer year, there may be very little resuspension flux. During a very energetic year, TP and SRP concentrations may become significantly elevated and remain high for months.

8.2 Phosphorus Flux Between Vegetation Zone and Open Water, and the Effect of Lake Stage

Model simulations (Sheng and Lee 1991a) for the one month period in 1989 when extensive field data were taken show that the net P flux between the littoral and pelagic zones was about one metric ton, which is small compared to the 14 tons contributed from external sources during the same period. These results should be valid for most of 1989, because there was usually little cross-boundary current between the vegetation zone and the open water. Moreover, the horizontal turbulent mixing was usually small in the vicinity of the vegetation (Sheng and Lee 1991b). Nevertheless, the lake stage was low (below 13 ft NGVD) for most of 1989, and extrapolation of the results to other years may not be without error.

The P flux between vegetation and open water is expected to depend on the lake stage. At low lake stage (e.g., 1989), much of the marsh is dry and the littoral zone may have less than 50 cm of water depth. During these periods, water generally circulates only in the open water zone, and there is no significant P transport from the littoral zone to the open water. When lake stage rises, emergent vegetation becomes submerged and there is more circulation in the region of the fringing littoral zone. Unfortunately, no field data were collected at high lake stages, but it is expected that circulation can intrude into the emergent vegetation at high lake levels to cause some resuspension and transport of P to the open water nearby.

The extent of circulation and the magnitude of P transport at higher stages is still open to conjecture. From their own model simulations, Richardson and Hamouda (1994) contended that water movement due to seiches dampens out exponentially with distance into the vegetation. Therefore, they believe that nutrient transport due to seiches is probably negligible because such a narrow band of the littoral zone is affected. On the other hand, Phlips et al. (1994a, 1994b) proposed that seasonal patterns in phytoplankton chlorophyll a concentrations near the littoral zone are at least partly based on a seasonal shift in N and P distributions between the SAV and phytoplankton communities. They went on to suggest that areas offshore from the littoral fringe exhibit high chlorophyll levels in the fall and winter because of internal loading from senescing vegetation. In support of this idea, chlorophyll a concentrations near the littoral zone were observed to increase following a die-off of submerged vegetation. This die-off, which resulted in regional anoxia, occurred in the fall of 1991 (after the field work for the Phosphorus Dynamics Study had concluded) following an increase in lake stage during the summer. They postulated that the nutrients released from decomposing plants could have provided the large nutrient pool needed for the consequent fall and winter bloom of phytoplankton.

Alternatively, P from the limnetic zone may be more easily circulated to the nearshore areas at higher stages and stimulate bloom formation. Maceina (1993) hypothesizes that high lake stages may allow more circulation of nutrient rich water from the mud zone to areas closer to shore. At this time, it appears that the Phosphorus Dynamics Study has shed much light on this difficult problem, but the work was not conducted over a long enough period to provide a definitive answer. Because algal bloom frequencies are highest on the west side of the lake (see Section 9.1), which is the region farthest from the most nutrient-rich inflows, continued research is needed to quantify the magnitude of littoral-pelagic transport of P under different conditions and determine the reason for algal blooms in the nearshore regions.

In the pelagic zone, normal variations in lake level do not appear to significantly affect the magnitude of phosphorus resuspension from the flocculent mud sediment (Sheng et al. 1991c). Wave-induced motion can reach the bottom at even the highest lake stages. At lower stage, waves are damped and weakened, but they can reach the shallower bottom more effectively, so the amount of sediment resuspension remains comparable to the higher lake stage case.

8.3 The Effect of Reduced Tributary Loadings on TP and SRP in the Lake

A nine-year simulation using the LOP0D box model indicates that reducing tributary loads of SRP and TP by 40%, 50%, and 70% will cause corresponding reductions in lake SRP concentrations of 26%, 33%, and 46% and reductions in lake TP concentrations by 19%, 23%, and 33%. The model also shows that the internal loadings of SRP via diffusion and resuspension are insensitive to changes in the external loadings. Using this model, the predicted lake response is not as great as that expected from the Vollenweider model discussed in the introduction of this report. When tributary P loads are reduced by 70%, the LOP0D model predicts that lake TP will decline to 62.9 μg/L. However, this prediction is unlikely to be precise, and the important point is that significant in-lake TP reductions can be expected with large reductions in tributary loads. A return to lower lake TP concentrations of around 50 μg P/L is possible, but the lake may take up to ten years to respond to the reduced external loadings. In other words, after the target loading rate is achieved, there may be a delay of several years before the mean lake TP concentration begins to drop significantly below its current level of approximately 90 μg P/L. These predictions are similar to those given by Bierman and James (1995) and Federico et al. (1981). Using a preliminary version of their WASP model application, Bierman and James predicted that a 40% step reduction in external P loading could result in a 25% reduction in lake TP concentration after five years.

Results of a three-month simulation using the 3-D model shows that a reduction in external loading only causes localized reductions in lake TP and SRP concentrations. This suggests that longer simulations are needed to see more significant changes. Long-term simulations require further refinement of the numerical models (Sheng et al. 1991b). Since the completion of this study, work has continued on these models, and the computer codes have been streamlined to take advantage of newly available computer architecture. Faster calculations will permit long-term model simulations. With some additional effort, the present models could be improved so that simulations of one to three years are possible.

More extensive data collection could improve the models' accuracy and precision. The estimates from both the LOP0D and 3-D model simulations contain significant uncertainties. With the existing field and laboratory data, sufficient calibration and validation of these models were not possible. Although the uncertainty of the sediment transport model was studied, the uncertainty analysis was too complicated to apply to the P model. The uncertainty of the P model is expected to be much higher than that for the sediment model. Sensitivity analysis suggests that more field and laboratory data are needed before the models can be considered as precise tools for predicting the impact of various management options (Sheng et al. 1991a). Nevertheless, the existing estimates can be used with caution as guidance for management action.

In developing robust numerical models, initial emphasis was placed on capturing the essence of the physical processes. During a second phase of the study, investigators collected nutrient data in addition to the other field data at high frequency over a 3-day period immediately following a storm in 1993. Nutrient concentrations varied significantly with time even though the sediment concentration did not change much. The models simulated the data accurately when the effects of DO and pH on sorption kinetics were considered (Sheng et al. 1993).

9.0 Summary and Application to Lake Management

Decisions regarding watershed management and lake protection are often difficult and controversial, and require a strong basis of technical support. In cases such as Lake Okeechobee, where these decisions may have significant costs and economic impacts, a scientific foundation for decision-making is critical. Even so, the best available information may not satisfactorily answer all questions about the potential effectiveness of management actions. Further research can reduce the uncertainty involved in the decision-making process by increasing knowledge of how the system functions and improving forecasting ability.

The recommendation to reduce nutrient loading to Lake Okeechobee was based on routine monitoring of the lake and empirical relationships derived from other lakes. However, the challenging technical issues posed by this complex ecosystem required more intensive data collection and research. The Phosphorus Dynamics Study and the Lake Okeechobee Ecosystem Study were two major projects that were undertaken to fulfill this need. These studies have resulted in major progress toward understanding environmental trends and processes in the lake. The foundation now exists for building detailed models that describe eutrophication processes and improve predictive capabilities. The following sections discuss ways in which the Phosphorus Dynamics Study can be applied to lake management and future research.

9.1 Current Lake Status and Historic Trends

In evaluating the detrimental effects of pollution and the possible benefits of improved management, anthropogenic impacts must be distinguished from the natural variability of the system. This requires a thorough awareness of the lake's current status and historic trends. This study contributes a great amount of information on present conditions and trends related to P dynamics. Because various factors exert their influences over very different time scales, the research encompassed both short-term intensive monitoring and examination of long-term trends.

With regard to current status, an abundance of hydrologic and benthic data have been gathered at a time when nutrient runoff appears to have peaked. These data will serve as a valuable reference for any future assessments of the benefits resulting from nutrient load reduction efforts.

Of more immediate interest is the description of processes affecting P over short periods of time. Until recently, only longer-term water quality trends could be examined, as the SFWMD's data (collected at 2-4 week intervals) did not permit investigation of short-term phenomena. High-frequency data gathered by automated methods showed that water quality is significantly influenced by short-term events such as wind-induced sediment resuspension and the diurnal cycle of thermal stratification and convective mixing. These results indicate that intensive data collections are necessary to understand how algal blooms form in the lake and why they occur in certain areas. Further automated monitoring has continued as a consequence of this research.

The lake's trophic history, including the period prior to basin development, was investigated by examining the stratigraphic record in the sediments. Paleolimnological investigations have established that the lake has been responsive to changes in its watershed during the last century. Radioisotopic analysis of sediment cores shows that sediment and phosphorus accumulation rates have increased during this century. These accelerated rates of deposition, coincident with the development of agriculture in the basin, suggest that agricultural runoff has had a lasting effect on these sediments. This sediment enrichment coincides with a shift in diatoms toward species that are tolerant of eutrophication (Stoermer et al. 1992).

Another important outcome of this study is the evidence that the lake's sediments have a dominant influence on the spatial variability of various ecosystem components. In the Lake Okeechobee Ecosystem Study, Phlips et al. (1993, 1994b) used discriminant function analysis to partition the open waters of the lake into four "ecological zones" based on light availability and water column concentrations of TN, TP, and chlorophyll a. They found that these zones coincide roughly with the lake's morphology and the distribution of sediment types. Spatial differences in phytoplankton growth limitation also correspond to these zones. Nutrient enrichment bioassays show that algal growth is limited by nitrogen, except in the mud zone where light availability is probably the limiting factor (Aldridge et al. 1994). Nitrogen fixation rates (Phlips and Ihnat 1994), phytoplankton biovolume (Cichra et al. 1994), algal bloom frequencies (Havens et al. 1994), benthic invertebrate populations (Warren et al. 1994), and fish populations (Bull et al. 1994) also exhibit zonal patterns coincident with the sediment type distribution. Especially significant is the fact that algal blooms are most prevalent in the regions that are geographically distant from the phosphorus-rich inflow to the north. These areas to the west and south are less affected by sediment resuspension, and biologically available forms of P are apparently in excess of phytoplankton demands, resulting in the predominance of N-limitation (Aldridge et al. 1994, Havens 1994) and the growth of nitrogen-fixing cyanobacteria (Cichra et al. 1994, Phlips and Ihnat 1994). In sum, these findings suggest that spatial variability in the lake is affected more by sediment type than by proximity to surface inflows.

The study of P accumulation in the sediments indicates that sediment enrichment continued to increase, particularly since about 1940, along with the intensification of agricultural activity. Given the strong interaction of these sediments with the water column, continued enrichment could explain several signs of accelerating eutrophication over the last two decades: increasing TP concentration in the water column since the 1970s (James et al. 1995b), declining net sedimentation rate for P (James et al. 1995a), declining TN:TP ratio (Smith et al. 1995), more frequent algal blooms (Havens et al. 1995), increases in rotifers and copepods among the zooplankton (Crisman et al. 1994), and increase in oligochaete dominance among benthic invertebrates (Warren et al. 1994). These eutrophication trends are probably controlled more directly by the increasingly rich reservoir of P in the sediments rather than by the P inputs themselves. Thus, although the short-term impacts of internal cycling processes are most apparent, the long-term effects of sediment enrichment on water quality and productivity also can be inferred from the evidence gathered from this research.

9.2 Hydrodynamic and Biogeochemical Processes

The apparent relationship between sediment enrichment and the eutrophication of Lake Okeechobee implies that the lake's potential recovery depends greatly on the present condition of its sediments. If Lake Okeechobee's sediments were enriched with phosphorus to the degree observed in Lake Apopka and other hypereutrophic lakes, then recovery of Lake Okeechobee would be extremely long. However, the laboratory tests discussed in Section 5 indicate that the mud sediment is still effective in assimilating dissolved inorganic P. This conclusion is supported by mass balance analysis showing that over 80% of the P input is trapped in the lake (James et al. 1995a).

Even though the sediments are still a strong net sink for phosphorus, they also can contribute dissolved P to the water column under certain conditions. Flux of SRP from the sediments can increase greatly when the overlying water becomes anaerobic or the anoxic subsurface sediments are exposed by wave erosion. Wind-driven resuspension can cause large increases in the water column total P concentrations, and major events also can elevate concentrations of SRP. In a typical year, the annual flux of SRP from the sediments is about equal to the external SRP input.

In comparison to loading from the sediments, phosphorus transport from the littoral zone appears to be negligible, at least at the low lake levels encountered in this study. However, this issue is still controversial and deserves priority for further investigation. Future research should focus on the nearshore areas between the vegetated zones and the central mud zone, and should attempt to quantify the relative importance of lateral P transport from the littoral and mud zones and competition between benthic and planktonic primary producers. This research is important because the nearshore areas receive the most recreational use and have the highest frequency of algal blooms (Havens et al. 1994, 1995). Also, they are more sensitive to nutrient limitation than the light-limited mud zone, and might exhibit greater sensitivity to changes in nutrient loading (Aldridge et al. 1994).

At higher lake stages, it is possible that lateral transport of P could increase. The results of this study provide support for Maceina's (1993) hypothesis that phosphorus-rich muddy sediments from the center of Lake Okeechobee are more evenly distributed through the lake at high lake stage, thereby providing the nutrients needed for the algal blooms along the western edge of the lake. Therefore, a higher average lake stage may predispose this area toward more frequent algal blooms. However, more work must be done to verify Maceina's proposal that lake stage is the principal controlling factor for phytoplankton standing crop.

9.3 Predicted Success of the Lake Okeechobee Management Program

In conclusion, the following questions put forth by lake managers must be addressed: Will the lake respond to phosphorus load reductions? If so, how long will it take? Should more be done now to speed up the lake's recovery?

Before responding to these questions, it should be stated that the work cited in this report shows that the lake exhibits the cumulative effect of higher nutrient inputs associated with basin development. These studies substantiate previous investigations (Federico et al. 1981; Janus et al. 1990) that recommended P load reductions and affirm the current management strategy of controlling P runoff at its source. If these inputs had been allowed to continue unchecked, the lake's trophic state would have proceeded to a hypereutrophic condition.

However, even though the lake has moved toward the threshold of a hypereutrophic condition, this study does not support the view that the lake is in immediate danger of ecological collapse. Unlike certain hypereutrophic lakes, Lake Okeechobee's sediments still have a strong capacity to assimilate incoming phosphorus. Furthermore, the mass balance model developed for this study indicates that Lake Okeechobee will eventually recover if P loads are successfully reduced, although internal loading will delay the lake's recovery by as much as 10 years. Recent data show that many of the P reduction efforts have been successful (Flaig and Havens 1995) and P inputs are moving close to the target (Jones and James 1993). Whether or not the loading target is achieved soon (possibly through the enforcement of additional P control measures) or is not reached for several years, it can be assumed that the lake is now responding to its current level of nutrient input. Therefore, considering the present health of the lake ecosystem, the current condition of its sediments, and the changing trends in P loads and lake TP concentrations over the last few years, expensive in-lake restoration techniques are not justified as a general eutrophication control strategy at this time.

10 Research Recommendations and Ongoing Work

As a scientific study of the internal processes affecting P dynamics in the lake, this project has been highly successful. The quantification of phosphorus fluxes is a major step toward understanding the dynamics of lake productivity, and the models presented here are important tools for ongoing research. However, the ability to make exact predictions based on this new information has been difficult due to the study's short time frame. Further data collection and model enhancement are necessary to improve forecasting precision.

Although knowledge of the P biogeochemistry in the lake has increased considerably, more information on several biogeochemical processes could complete our understanding of the role of bottom sediments. Additional studies should focus on short-term processes controlling long-term P retention and release in the sediment and water column. Some suggested topics for further research are: (i) relative contributions of sediment resuspension and diffusion to overall P flux from sediments, (ii) phosphorus exchange reactions between suspended sediments and the water column, and effects of water column conditions such as pH and DO on P release or retention, (iii) quantification of the physico-chemical characteristics of suspended sediments, in relation to their P retention/release capacity, (iv) identification of the labile and non-labile pools of P in the suspended sediments and determination of their bioavailability, (v) characterization of the oxygen and redox status at the sediment-water interface in relation to P flux, (vi) determination of the role of sediment reduction processes (e.g., nitrate reduction, sulfate reduction and methanogenesis) on organic matter decomposition and P flux, (vii) determination of the effects of fluctuating water levels and sediment drying in the littoral zone on P release or retention, (viii) relationship of P dynamics to carbon and nitrogen dynamics in the sediment and water column, and (ix) development of an internal nutrient budget for the lake.

Future studies should also strive to link the physical/chemical data obtained from this study with biological processes. More ecologically-oriented research might investigate: (i) fluxes of dissolved organic and inorganic P between sediments, benthic microflora, littoral zone vegetation, phytoplankton, and bacterioplankton, and (ii) the role of benthic flora in stabilizing sediments, reducing resuspension of sediments, and thereby reducing flux of P from sediments to water column, (iii) the role of benthic activity (bioturbation) in P exchanges between the sediments and water column, and (iv) relating findings from Lake Okeechobee to conditions in other eutrophic Florida lakes.

With regard to data collection, this study placed heavy emphasis on laboratory data in measuring parameters related to P dynamics. Consequently, a few links in the P cycle still have not been elucidated. If further research is undertaken to fill the information gaps, the relevant data should be collected within the lake itself. This work would require frequent sampling of all important parameters (wind, hydrodynamics, sediment, oxygen, pH, ion concentration, chlorophyll *a*, phosphorus, nitrogen, microbial biomass, etc.) during episodic events. State-of-the-art high frequency sampling devices (such as sensors capable of measuring the detailed vertical structure of currents, temperature, and dissolved oxygen) should be utilized or developed for measuring as many parameters as possible. Of course, laboratory experiments still will be needed to help quantify the kinetic constants of the transformation processes. One particular study that would benefit from field data collection is an investigation of flocculation dynamics, as discussed in Section 4.5.

As modeling of the lake progresses, more precise forecasts of trends in specific areas of the lake can be made. Special attention should be given to the nearshore areas that are most sensitive to nutrient limitation. Because this study was conducted when lake levels were low, further research is warranted to quantify the littoral-pelagic flux of P. These investigations should seek to quantify the circulation and sediment and P dynamics within the littoral zone, and then quantify the P flux between the two zones during different lake stages. This work would require extensive collection of field data (water level, wind, wave, currents, sediments, P, etc.) within the littoral and nearby pelagic zones, and the enhancement of numerical models.

Some of this follow-up research has been done already. Rhodamine dye has been used to trace water movement inside the edge of the littoral zone. Dye movement was observed up to 100 m inside the emergent marsh vegetation; this distance varied widely among the areas examined. In another ongoing project, five stations have been established along a 2000 m transect from the littoral fringe toward the northwest levee. Stage, temperature, and conductivity have been measured at these sites every 15 minutes with automated equipment to detect water movements over time.

The 3-D numerical models should be refined and linked with the WASP water quality model which is being applied to the lake by SFWMD research staff. The WASP model is intended to be the main model for predicting future trophic conditions under various management scenarios. The SFWMD has chosen to use a modified version of the WASP model that explicitly includes the impact of hydrodynamics and sediment resuspension on light attenuation and phosphorus concentrations. With some additional calibration and validation, enhanced versions of the numerical models could be used in conjunction with this WASP model to perform high-resolution, quantitative simulations of phosphorus and plankton dynamics. To be fully applicable to this modeling effort, the hydrodynamic model would need to demonstrate the ability to maintain a mass balance, to be calibrated to sufficient site-specific data, and to have its high resolution output translated to input appropriate to the needs of WASP. These simulations could provide much more insight and predictive capability than those from simple box models.

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