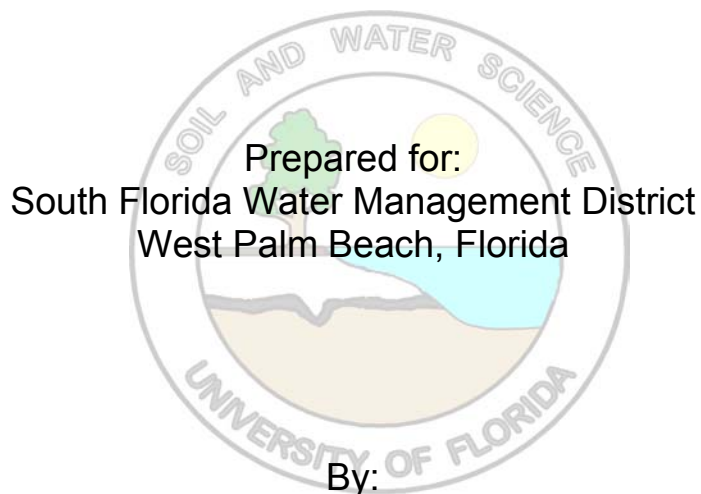


POTENTIAL IMPACTS OF SEDIMENT DREDGING ON INTERNAL PHOSPHORUS LOAD IN LAKE OKEECHOBEE

Summary Report
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1.0 Executive Summary

Lake Okeechobee, a large (1730 km²) shallow (average depth 2.7 m) lake in south Florida, has become more eutrophic due to excessive external phosphorus (P) loads (Fig. 1). The eutrophication of this lake has resulted in many changes over the past three decades that include: more than a doubling of total phosphorus (TP) in the water column, an increasing frequency of algal blooms, and an increasing abundance of nitrogen fixing cyanobacteria (Havens et al. 1996). Water quality monitoring by the South Florida Water Management District (SFWMD) revealed that mean annual total P concentrations in the lake water generally increased



Figure 1. Map showing Lake Okeechobee (source: South Florida Water Management District).

over the past three decades, more than doubling between 1973 (49 µg P/L) and 2001 (110 µg P/L) (Fig. 2). Increases in lake water P concentration were attributed to increased P loading to the system from various external sources such as the Kissimmee River, runoff from dairy farms at the north end of the lake, and agricultural activity on the southern shores of the basin. The excessive P loads have ultimately accumulated in mud sediments in the center of the lake. In 1988, a synoptic survey of Lake Okeechobee estimated that the surface 10 cm of mud sediment (44% of the lake) in Lake Okeechobee contained 28,600 metric tons of total P (TP). Flux measurements using sediment cores indicate that mud sediments may have contributed approximately 200 metric tons of dissolved P to the lake water column in 1988. There is a concern that the accumulation of mud sediments has continued over the past decade both in depth and extent, and that it may be impacting water quality in certain lake regions. Phosphorus budgets of Lake Okeechobee for the last ten years indicate that P has accumulated in the sediments at the rate 303 metric tons of P year⁻¹, representing a net increase of 10 percent (over a period of 10 years) in TP mass of

measurements using sediment cores indicate that mud sediments may have contributed approximately 200 metric tons of dissolved P to the lake water column in

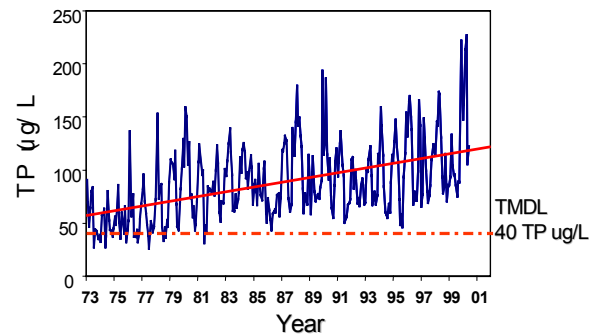


Figure 2. Total P concentration of the water column in Lake Okeechobee (Karl Havens, 2001, South Florida Water Management District).

surface 10-cm sediments. Since sediments tend to function as net long-term sinks for TP, external load reduction and other lake restoration efforts may be reflected in forms and storage of P in sediments. To decrease external P loading to the lake, Best Management Practices (BMPs) have been adopted in the Lake Okeechobee drainage basin. Some BMPs have been in place for a decade. Based on the field studies conducted in

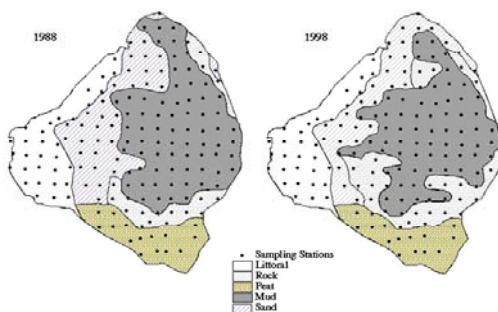


Figure 3. Maps of sediment zones and sampling locations in 1988 and 1998. Sediment zones are based on physical description of the surface 10-cm sediment.

1988 and 1999, sediments were classified as mud, littoral, peat, sand or rock (Fig. 3, Fisher et al. 2001). Although some minor redistribution has occurred over the previous decade, mud sediments of Lake Okeechobee essentially occupy the same total area. Total surface sediment P showed little overall change in the ten-year period (Fisher et al. 2001). However, lake-wide spatial patterns indicate some localized decreases of total P content in the littoral and northern regions of the lake.

The forms and amounts of P in Lake Okeechobee are a function of the P input from external sources such as agricultural and urban activities, P output from the lake through surface streams and groundwater seepage, and the interchange of P among the various sediment and water P components. Interchange between the dissolved and solid phase (particulate) P components occurs through chemical and biochemical reactions. The biological incorporation (uptake by algae) of dissolved inorganic P into insoluble organic P, and the subsequent settling of insoluble inorganic and organic P gradually remove P from the surface water. Thus, the direction of net transport of particulate P is from the overlying water to the sediments, while dissolved reactive (DRP) flux is from sediments to the overlying water column.

In 1998, the South Florida Ecosystem Restoration Working Group directed the Lake Okeechobee Issue Team to develop an Action Plan (Plan) for rehabilitation of the lake. This Plan also is included in the Lake Okeechobee Protection Act (Florida Statutes 373.4595 (2000)). One part of the Plan is to consider removal of the P-laden mud sediments within the lake by dredging. The mud in Lake Okeechobee covers an area greater than 80,000 hectares and has a volume of approximately 200 million cubic meters. Compared to other lake restoration projects (Cooke et al. 1993), removal of mud sediments from Lake Okeechobee will require an order of magnitude greater effort. However, dredging of surface sediments alone will not reverse eutrophication, unless external loads are also curtailed (Kleeberg and Kohl, 1999).

To support the Plan, the University of Florida conducted a study to determine the potential impacts that partial or complete dredging of mud sediments will have on mobility of P to and from the sediments. The study was designed to address the question: if mud sediments are partially or completely dredged, what are the potential impacts on water quality and sediment water interactions as older buried sediments are exposed to the water column? It is anticipated that dredging would remove recently accreted sediments enriched with labile pools of P and reduce potential P flux into the overlying water column, thus decreasing the internal P load and the time required for lake recovery.

Specifically, in this report we address the following specific questions by using the results obtained from a series of laboratory and field experiments conducted on Lake Okeechobee sediments. Specific questions to be addressed are: (i) What is the impact of dredging on the overall P assimilative capacity (and internal P load) of the lake? (ii) What is the impact of dredging on the P flux to the water column under oxic and anoxic conditions? (iii) What is the impact of dredging on resuspension of P? (iv) What is the present capacity of mud sediments in the lake to assimilate P, and how would this capacity change if the sediments were dredged to various depths (including complete removal of mud and exposure

of the underlying sand)? (v) What is the present rate of internal loading of P from the sediments to the water column (by diffusion and resuspension) and how would this rate change if sediments were dredged to various depths (including complete removal of mud and exposure of the underlying sand)? To address these questions, we conducted a series of experiments using sediment cores obtained from mud zone of Lake Okeechobee. Readers are cautioned that the summary results presented in this report reflect only the potential P retention and release capacity of sediments under controlled laboratory conditions and do not address the dynamic field conditions. However, results provide an indication of the effects of dredging mud sediments on P retention and release by mud sediments. In the context of above referenced questions, a summary of results are presented below. For experimental methods and detailed results, the reader is referred to the comprehensive report submitted to the South Florida Water Management District.

1.1 What is the impact of dredging on the overall P assimilative capacity (and internal P load) of the lake?

To determine the impact of dredging on the overall P assimilative capacity of mud sediments, first we need to understand relative proportion of various P forms of P present in mud sediments. Phosphorus is present in inorganic and organic forms. Inorganic forms include: porewater and exchangeable P (readily bioavailable); iron (Fe) and aluminum (Al) bound P (slowly available); and calcium bound P (very slowly available). Organic P forms include: dissolved organic P (DOP) (readily bioavailable); microbial biomass P (readily bioavailable); fulvic acid bound P (slowly available); humic acid bound P (very slowly available); and residual organic P (not available). We sampled sediments from nine locations in the mud zone of Lake Okeechobee. These sediments were analyzed for various P forms, using simple chemical extraction schemes as described by Olila et al (1994); Moore et al., (1998), and for mineralogical composition.

The mud sediments in Lake Okeechobee showed a distinct break in total P in the upper 10 cm when compared with the underlying layers. Total P was 33% higher in the top layer (0-10 cm) compared with the next sediment layer, immediately below (10-20 cm). The pools of total P, total inorganic and total organic P were all higher in the 0-10 cm. In terms of total P storage, there was a total of 192 g P m^{-2} in the top 56 cm of mud. If we dredge the top 30 cm sediment, we will remove 111 g P m^{-2} , 45 cm of dredging will account for an additional 50 g P m^{-2} , and 56 cm sediment dredging will remove all of the mud sediments above the native sediments. Approximately 58% of the total P storage is in the top 30 cm sediment. The relative proportion of each P fraction changes relatively little with depth. On an average, the Ca-Mg bound P comprises 83% of total P followed by residual P making up 12.4 % of total P. Fe-Al bound P made up 2.1%, just slightly higher than the 1.8 % of total P as alkali extractable organic P. The amount of P present in the porewater was small at 0.4% of the total P. These results suggest that approximately 95% of the total P is in relatively stable pool, with major portion of P present as Ca-Mg bound P. Changes in physico-chemical properties of sediments, such as decrease in pH, accumulation of organic matter, and increased biological activity can increase the dissolution of Ca-Mg bound P.

The minerals in sediments were identified using several techniques, including x-ray diffraction (XRD) and scanning electron microscopy (SEM). Computer modeling of the

porewater chemical composition was also performed to determine which minerals should be chemically stable (i.e., supersaturated, and not subject to dissolution) in the sediment environment. Both solid-state and chemical modeling assessments of the Lake Okeechobee mud sediments and porewaters are consistent with an environment enriched in silica and bases (e.g., Ca and Mg). The minerals detected by XRD and corroborated by other techniques included: sepiolite, quartz, dolomite, smectite, palygorskite, and calcite. Modeling results indicate that apatite (the least soluble calcium phosphate mineral) and tricalcium phosphate would be stable (i.e., would not dissolve, and could precipitate given sufficient time) in the sediment environment. However, they are not controlling phosphate activity in solution. Some calcium phosphate was indicated in all 6 samples analyzed using SEM in conjunction with elemental microanalysis. The association of Ca and P was highly localized; i.e., there was no indication that P was diffusely associated with Ca in such a way as to suggest that it is significantly "co-precipitated" with the more abundant Ca phase - calcite (CaCO_3). High concentrations of Si and Mg, and the prevalence of the minerals sepiolite and palygorskite, may serve to inhibit phosphate sorption and precipitation, thus decrease overall P assimilative capacity of sediments.

To determine the influence of dredging on P release or retention, we obtained intact sediment cores from one location representing the mud zone. The following dredging treatments were implemented on intact sediment cores: control (no dredging-current conditions); 50% mud removal (approximately top 30 cm sediment); 75% mud removal (approximately top 45 cm sediment); 100% mud removal (dredged to sand sediments, top 55 cm sediment). At the end of dredging sediment cores were flooded to a depth 32 cm. Water column was spiked with different levels of DRP. Five water-column spiked DRP concentrations (0, 16, 32, 64, and 128 $\mu\text{g L}^{-1}$) were evaluated for each dredging treatment, using three replications per treatment. The water column in each treatment was aerated to maintain dissolved oxygen levels $> 5 \text{ mg L}^{-1}$. All sediment cores were incubated in the dark at approximately 22 °C. The water column in each sediment core was replaced once every 60 days with fresh lake water spiked with respective P loadings and this procedure was repeated for a period of 431 days.

Water column DRP concentrations (filtered lake water with no spike) were influenced by sediment dredging (Fig.4). Significant decreases in water column DRP were observed in sediment cores with 30 and 45 cm dredging. These effects were more pronounced during the first 7 months after dredging, followed by very little or no effect during the remaining 6 months of the study period. In no dredging treatment, initial increase in water column DRP concentrations was due to low DRP levels of the filtered lake water used and high sediment porewater concentrations.

These gradients decreased with time as sediment porewater DRP diffused into overlying water column. Under field

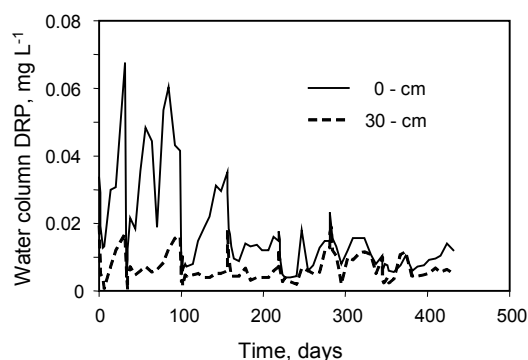


Figure 4. Influence of sediment dredging on DRP concentration of the water column under ambient conditions.

conditions, organic matter inputs and hydrodynamic effects at the sediment water interface potentially influence these DRP gradients.

Dissolved reactive P release from sediment was calculated based on accumulative P release at the each water column exchange cycle. At ambient water column P (less than 10 ug P/L) levels, sediment cores with no dredging released approximately 24 mg P m⁻² year⁻¹ (Fig. 5). Effect of dredging on percent increase or decrease in sediment P release was calculated as follows:

$$DE = [(P_o - P_d)/P_o] 100.$$

Where: DE is the percent change in sediment P release due to dredging, P_o is the sediment P release under no dredging conditions, and P_d is the sediment P release from sediment cores subjected to dredging. For the whole experimental period of 431 days, sediment dredging decreased approximately 90, 0.4, and 16% of sediment P release into water column for 30 cm, 45, and 55 cm sediment dredging, respectively, as compared to no dredging treatment. Although for the whole experimental period dredging the top 30 cm sediment seems to show significant positive effect on net P release to the water column, it should be recognized that the effect lasted only for the first 7 months only, followed by very little or no effect during the remaining 6 months of the study period.

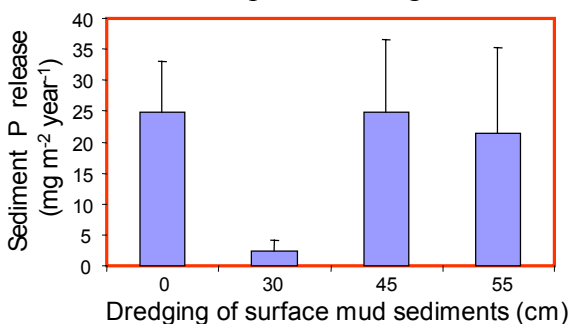


Figure 5. Influence of sediment dredging on sediment P release under ambient initial water column DRP levels.

for the first 7 months only, followed by very little or no effect during the remaining 6 months of the study period.

Considerable caution should be exercised in using the results presented in Fig. 5. The P flux represents a conservative estimate under very controlled experimental conditions. Lack of hydrodynamic events may potentially under estimate long-term P flux into water column. Apparent benefit of surface 30-cm sediment removal, as compared to 45-cm or 55-cm sediment removal, may be due to differences in physico-chemical properties at the sediment-water interface. The surface sediments (30-32 cm) exposed after 30 cm sediment dredging generally had higher P-binding strength than the sub-surface sediments (45-47 cm sediment). Further studies are needed to carefully examine the P-sorption characteristics of sediments at these boundary layers.

To determine the role of water column P concentrations on sediment P release or retention, we spiked water with different levels of DRP to simulate different loading conditions. Phosphorus release from sediments was highest at ambient water column DRP levels and decreased with an increase in P loading. However, it

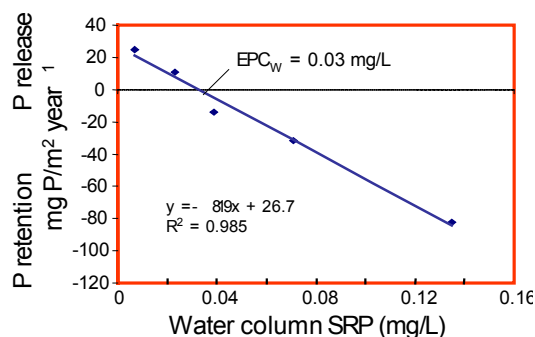


Figure 6. Influence of initial water column DRP levels on sediment P release or retention under no dredging conditions.

should be recognized that an increase in P loading also maintained high water column DRP concentrations.

Figs. 6 and 7 show the relationship between water column DRP concentration and P release or retention potential of sediments. These data represent total P release or retention at the end of 431 days study and are expressed on an annual basis. When sediment P release or

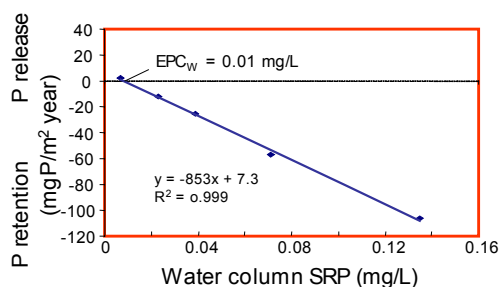


Figure 7. Influence of initial water column DRP levels on sediment P release or retention by sediment cores with 30 cm

retention equals “0”, the water column P concentration at this point is defined as equilibrium water column P concentration or EPC_w . If the water column P concentration is less than EPC_w , then the sediments will function as net source of P to the water column or release P from sediments to overlying water column. If the water column P concentration is greater than EPC_w , then the sediments will function as net sink for P or sediments retain P. Dredging the top 30 cm sediments decreased EPC_w from 0.03 mg L^{-1} to 0.01 mg L^{-1} , which means subsurface sediments have greater affinity to retain P. Dredging the top 45 and 55 cm sediments maintained EPC_w values of 0.02 and 0.04 mg L^{-1} , respectively. The EPC_w values reported here are based on an average for the whole study period.

In Fig. 8, we show the calculated EPC_w for each water exchange during the study period. Results indicate that the dredging had a positive effect in regulating EPC_w during the first 100 days. Dredging the surface 30 to 45 cm sediments had a pronounced effect on EPC_w , suggesting that the bottom sediments have high buffering capacity, which resulted in lowering water column

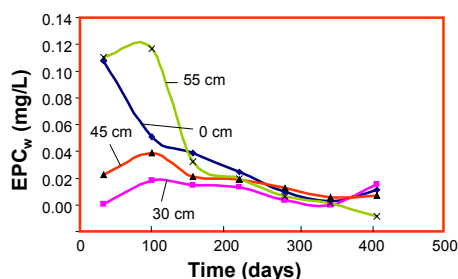


Figure 8. Influence of time after sediment dredging on EPC_w .

P concentrations. It should be recognized that these results are based on laboratory experiments conducted under static conditions with no external inputs of organic matter (derived from detrital planktonic tissue) or hydrodynamic events. However, the results show potential effects of dredging in decreasing water column DRP concentration.

1.2 What is the impact of dredging on the P flux to the water column under oxic and anoxic conditions?

Laboratory studies using intact cores were conducted to determine: (i) the rate of oxygen consumption by sediments under four dredging treatments, and (ii) the rate of P release from sediment to overlying water column during and after oxygen depletion. No dredging treatment showed high oxygen consumption rates as a result of high organic matter content and presence of other oxygen demanding species. Dredging decreased overall oxygen consumption rates, with low rates observed by sediments below 45 cm depth (Fig. 9). These subsurface sediments

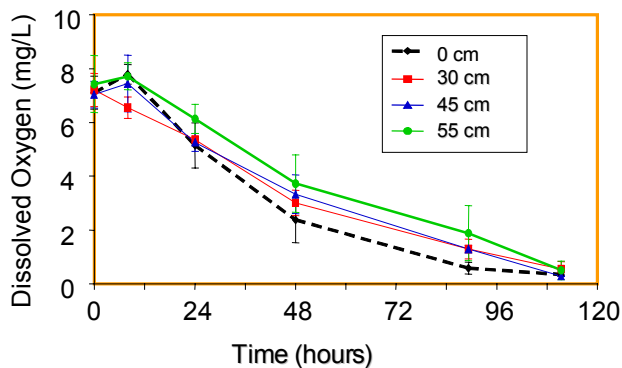


Figure 9. Influence of dredging on oxygen consumption by sediments.

treatments, with flux into the water column of approximately $5.4 \text{ mg P m}^{-2} \text{ day}^{-1}$ (Fig. 10). Dredging significantly reduced P flux under oxygen depleted water column conditions, with P flux in the range of 1 to $1.4 \text{ mg P m}^{-2} \text{ day}^{-1}$ for all dredging treatments. Phosphorus release (measured during a 32 day incubation) from sediments into **aerobic** (oxygenated) water column was significantly lower in no dredging treatment, with flux of approximately $0.4 \text{ mg P m}^{-2} \text{ day}^{-1}$ (Fig. 11). Dredging significantly reduced P flux under oxygenated water column conditions, with P flux in the range of 0.1 to $0.35 \text{ mg P m}^{-2} \text{ day}^{-1}$ for all dredging treatments. These results showed the impact of oxygenated water column on sediment P release. Under aerobic water column conditions, the sediment-water interface is aerobic and below this layer the sediment is anaerobic. This aerobic layer is typically 1 cm thick. Mud zone sediments are rich in iron. In anaerobic sediments, iron is present in reduced form (ferrous iron). This form of iron is less effective in adsorbing soluble P in sediments. This is the reason, why anaerobic sediments typically contain high levels of soluble P and ferrous iron in porewaters. However, under aerobic conditions, ferrous iron is oxidized to ferric iron and this form of iron is highly effective in adsorbing soluble P. As both soluble P and ferrous iron diffuse from anaerobic sediments into aerobic sediments or water column, ferrous iron is rapidly oxidized to ferric iron which scavenges soluble P from the water column, resulting in lower P flux. Under field conditions, this aerobic sediment-water interface can be disrupted by the addition of organic inputs which create high oxygen

probably contained more recalcitrant organic matter content, as compared to surface sediments. Oxygen levels reached negligible levels within 5 days. Calculated oxygen consumption rates were: $40 \text{ mg O}_2 \text{ m}^{-2} \text{ hour}^{-1}$ for no dredging treatment, $25 \text{ mg O}_2 \text{ m}^{-2} \text{ hour}^{-1}$ for 30-cm dredging, and $30 \text{ mg O}_2 \text{ m}^{-2} \text{ hour}^{-1}$ for both the 45 and 55-cm dredging, respectively.

Phosphorus release (measured during a 30 day incubation) from sediments was rapid under **anaerobic** (no oxygen) water column conditions, especially in no dredging

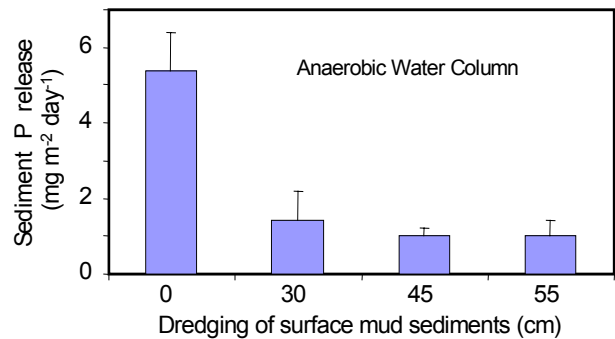


Figure 10. Influence of dredging on sediment P flux into anaerobic water column.

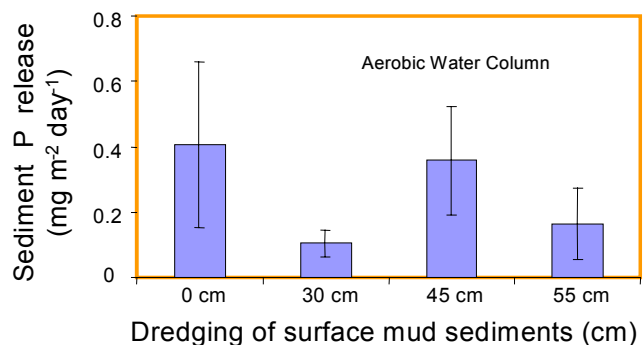


Figure 11. Influence of dredging on sediment P flux into aerobic water column.

demand, resulting in anaerobic conditions. In addition, wind-driven sediment resuspension can also disrupt this boundary layer, creating high oxygen demand at the sediment-water interface.

1.3 What is the impact of dredging on resuspension of P release into water column?

Laboratory studies were conducted to determine: (i) the rate of P desorption from solid phase to liquid phase, and (ii) the rate of P release during sediment resuspension under low sediment shear stress where 2 cm of sediment was resuspended ($\sim 0.2 \text{ N m}^{-2}$ representing low wind velocity) and high shear stress where 10 cm of sediment was resuspended ($\sim 0.75 \text{ N m}^{-2}$ representing high wind velocity).

In the first experiment, bulk sediments were sampled at 0-2, 30-32, 45-47 and 60-62 cm depth from sediment cores collected from Station M-9 of the mud zone. Each of these sediments represents surface sediments after dredging treatments have been implemented. Wet sediment (approximately 4 grams) samples were placed in 500-mL flasks containing 400 mL of filtered Lake Okeechobee water to obtain total suspended solids concentration of 10 g L^{-1} . The suspensions were sampled at 15 and 30 min, and 1, 2, 4, 8, 24, 48 and 96 h, and filtered through $0.45 \mu\text{m}$ filter and analyzed for DRP.

The accumulative P release by sediments suspended for 96 hours in an aerobic water column was significantly greater for sediments from 0-2 cm depth (8.3 mg P kg^{-1} per 96 hours) than that of the other depths studied (30-32, 45-47 and 60-62 cm). The amount of P released was lowest by sediments from 45-47 cm depth ($-1.6 \text{ mg P kg}^{-1}$ per 96 hours; negative sign denotes net adsorption) followed by sediments from 60-62 cm (0.7 mg P kg^{-1} per 96 hours) and 30-32 cm depth (4.0 mg P kg^{-1} per 96 hours). The results also demonstrated that the maximum P release from suspended sediments could be sustained up to 48 h after resuspension (Fig. 12). A longer time period (96 hours) resulted in re-adsorption of released P. Both the maximum rate ($0.23 \text{ mg P kg}^{-1} \text{ h}^{-1}$) and cumulative P released (8.3 mg P kg^{-1} per 96 hours) at the end of 96 hours were high for sediments from 0-2 cm

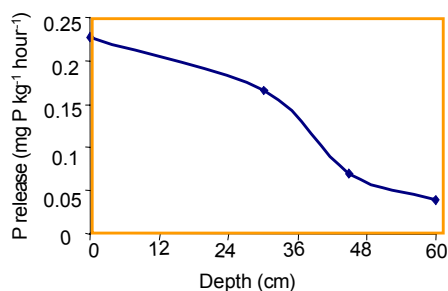


Figure 12. Maximum P release potential of sediments suspended into aerobic water column.

depth. Dredging P rich sediments decreased P release potential of sediments into an aerobic water column.

A second experiment was conducted to determine the P release from intact sediment-cores (60-cm sediment depth) during and after resuspension of surface sediments after simulated dredging of 0, 30, 45, and 55 cm removal. Simulations of low wind conditions (resuspension of top 2 cm of sediment; $\sim 0.25 \text{ N m}^{-2}$), and high wind conditions (resuspension of 10 cm of sediment; $\sim 0.75 \text{ N m}^{-2}$), were followed by water sampling for 95 h. Sediments were resuspended for one hour, followed by 95 hours of settling of suspended particles. The simulation of low wind-induced sediment re-suspension showed that during an hour long induced re-suspension of sediments, P in the water column increased substantially (Fig. 13). The P release increased gradually up to 6 h after sediment resuspension was stopped. The 0,

30 and 45-cm sediment removal of mud sediments yielded less than $0.005 \text{ mg P L}^{-1}$ in the water column at the end of 95 h after resuspension was discontinued. However, in case of 55 cm mud sediment removal, the water column P remained about $0.025 \text{ mg P L}^{-1}$. These results

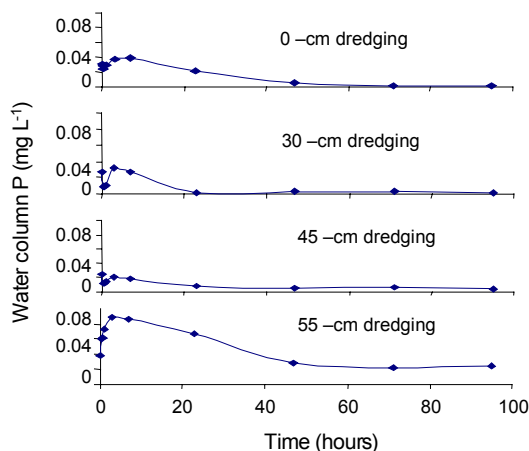


Figure 13. Influence of surface 2 cm sediment resuspension on P release into aerobic water column.

support findings from the first experiment that the dredging to 30 or 45 cm would be better options to reduce short-term release of P to the water column than no dredging or complete removal of the mud.

Resuspension of top 10 cm sediment for one hour into water column showed that water column P (0.02 mg P L^{-1}) for sediment-core dredged to 30 cm and the highest (0.6 mg P L^{-1}) for the cores dredged to 45 cm 95 hr after resuspension was discontinued. Both the 0 and 55 cm dredged cores yielded 0.08 and 0.16 mg P L^{-1} , respectively. These results also indicate that dredging to 30 cm may provide better options for

the control of water column P even in high wind-induced events. However, it should be recognized that resuspension of top 10-cm sediment may be unrealistic even under heavy winds and represents worst condition.

Phosphorus release during sediment resuspension represents only one event. However, under field conditions repeated sediment resuspension events can occur periodically in response to wind events and maintain P flux into water column, through desorption P from suspended particles, especially during first few hours. At the same time oxidation of ferrous iron (Fe^{2+}) to ferric iron (Fe^{3+}) can remove DRP from the water column by forming ferric phosphate. This precipitate attached to suspended particles will settle on sediment surface. Diffusion of P from layers below the sediment resuspension zone (0-2 cm depth) will provide constant supply of P to surface layers. Thus on a long-term, diffusion of P to surface sediment layers regulates overall P flux into water column.

1.4 What is the present capacity of mud sediments in the lake to assimilate P, and how would this capacity change if the sediments were dredged to various depths.

Phosphorus adsorption (retention) and desorption (release) experiments were conducted on sediments, using batch incubation technique under aerobic anaerobic conditions (methods described by Pant and Reddy, 2001; Pant et al., 2001).

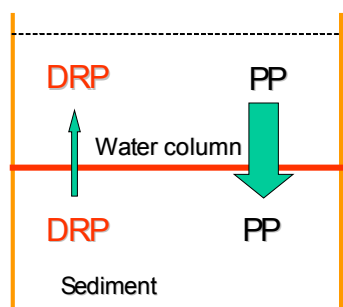
Under aerobic conditions, the EPC_0 (equilibrium P concentration, where adsorption equals desorption) and S_{max} (maximum amount of P adsorbed) were highest for sediments from 0-2 cm. High EPC_0 values of surface sediments indicate high P enrichment in surface sediments. However, the P desorption study showed that 0-2 cm sediments had lower capacity to retain sorbed P than that of 30-32 and 45-47 cm. The sediments from 60-62 cm, however, had the highest EPC_0 and the lowest S_{max} compared to sediments from above 60 cm,

suggesting poor sorptive capacity of native sandy sediments. Though most of the P sorption parameters were not significantly different for sediments from 30-32 and 45-47 cm, EPC_0 for sediments from 30-32 cm (0.011 mg L^{-1}) was about 2 times that of sediments from 45-47 cm (0.004 mg L^{-1}), suggesting that bottom sediments have greater binding capacity to retain P.

The EPC_0 under anaerobic conditions showed no significant difference among sediment depths, but was generally higher than sediments exposed to aerobic conditions. The S_{\max} was greatest for sediments from 0-2 cm, and decreased in the order of 30-32 cm, 45-47, and 60-62 cm depth. The S_{\max} had significant positive correlation (at $p < 0.05$) with ox-Fe ($r = 0.87$) and ox-Al ($r = 0.60$), indicating that the greater amount of amorphous and poorly crystalline Fe and Al, the greater the S_{\max} .

1.5 What is the present rate of internal loading of P from the sediments to the water column (by diffusion and resuspension) and how would this rate change if sediments were dredged to various depths (including complete removal of mud and exposure of the underlying sand)?

Both biotic and abiotic reactions regulate DRP concentration of the water column. In eutrophic lakes such as Lake Okeechobee, much of the DRP added is rapidly assimilated by algae. Dead algal cells, along with the particulate inorganic and organic solids accrete on bottom sediments. Sediment bound P (particulate P or PP) accretion rates increase with P loading.



For example, in Lake Okeechobee, P accretion rates have increased about four-fold since the 1900s (from about $0.25 \text{ g P m}^{-2} \text{ year}^{-1}$ before 1910 to about $1 \text{ g P m}^{-2} \text{ year}^{-1}$ in the 1980s) (Brezonik and Engstrom, 1998). Concentrations of total P and several P fractions are generally higher in recent sediments, and decreased with depth, suggesting the influence of recent P

loading. Although accretion of sediment bound P suggests that PP flux is downward (i.e. from water column to sediments), the DRP flux is upwards (i.e. from sediments to water column) in response to concentration gradients established at the sediment-water interface. Present internal loading was estimated two ways: (1) sediment cores were incubated for 431 days under laboratory conditions, with water exchange period of approximately 60 days, and (2) short-term incubation of sediment cores for 30 days under both aerobic and anaerobic water column conditions. These were measured in sediment cores dredged for 0, 30, 45, and 55 cm. This flux represents diffusive flux only and does not consider any potential bioturbation or sediment resuspension.

Estimated annual P flux from sediments to the water column is 25, 2.4, 25, and $21 \text{ mg P m}^{-2} \text{ year}^{-1}$ for no dredging, 30, 45, and 55 cm dredging treatments, respectively. These flux measurements are very low compared those measured during short term incubations. In our opinion, these annual fluxes are not realistic, since the system did not receive any external inputs or subjected to any hydrodynamic events. Sediment P fluxes measured during short-term incubations may provide more realistic diffusive flux P into water column. Under aerobic water column conditions, P flux measured was: 0.41, 0.11, 0.36, and $0.16 \text{ mg P m}^{-2} \text{ day}^{-1}$ for no dredging, 30, 45, and 55 cm dredging treatments, respectively. Sediment P flux

into anaerobic water column was: 4.5, 1.1, 0.7, and 0.9 mg P m⁻² day⁻¹ for no dredging, 30, 45, and 55 cm dredging treatments, respectively. High anaerobic flux reflects dissolution of iron bound P from the surface sediments. Under both conditions, dredging decreased sediment P flux into water column. In earlier studies, we measured sediment fluxes from sediment cores obtained from the mud zone station used in the current study. Sediment P fluxes measured were: 1.7 and 1.0 mg P m⁻² day⁻¹ for 1988 and 1999, respectively (Fisher et al. 2002).

During windy periods, resuspension and deposition of sediments may be an important mode of P transfer to the water column. Because sediment resuspension events are transitory, P flux due to this process may occur at short-time scales at a more rapid rate, compared to diffusive flux. Resuspension of 2 cm surface sediments into aerobic water column increased the DRP concentration to 0.4 mg L⁻¹ in no dredging treatment, as compared to 0.1 to 0.8 mg L⁻¹ in dredged treatment. Within 96 hours after sediment resuspension, DRP reached background levels in 0, 30, and 45 cm dredging treatments, suggesting high reactivity of surface sediments. When native subsurface sediments were resuspended into water column, background levels increased from 0.005 mg L⁻¹ to 0.024 mg L⁻¹, suggesting poor adsorptive capacity of these sediments.

The total P storage in the top 30 cm sediment is estimated at 110 g P m⁻². This is main concern of lake managers. If these recently accreted sediments are not dredged and assuming 5% or 5.5 g P m⁻² of this stored P is bioavailable, then at current rate of P flux of 0.4 mg m⁻² day⁻¹ into aerobic water column, it will take approximately 38 years to deplete this pool. If top 30 cm of sediment is dredged, the estimated P in remaining sediment is about 82 g P m⁻². If we assume only 5% of this stored P is bioavailable, then at current rate of P flux it will take approximately 28 years to deplete this pool. Internal biogeochemical reactions can potentially increase this bioavailable pool as a result of desorption and dissolution reactions and mineralization of organic P.

1.6 Conclusions

All laboratory experiments suggest that dredging the surface 30 to 45 cm sediments can have a positive effect in improving water quality. It should be recognized that laboratory studies consider only diffusive flux from sediment to overlying water column and does not include repeated sediment resuspension events and inputs of organic material at the sediment water interface. Dredging removes substantial amount of P stored in sediments. Removal of top 30 cm sediment accounts approximately 65% of TP storage. Implementation of P reduction goals such as dredging may have significant costs and economic impacts. Thus, management focus should be on reduction of external loads, which will ultimately have a positive effect in reducing the internal load. To determine the regulators of internal load, it is critical that we have a thorough understanding of the dynamics of physical, chemical, and biological processes at sediment water interface regulating the internal load within the lake. The key questions often asked are: (i) will Lake Okeechobee respond to P load reduction? (ii) if so, how long will it take for the lake to recover and reach its background condition? and (iii) are there any economically feasible management options to hasten the recovery process?

The internal load (once the external load) can extend the time required for lake to reach its original condition. This lag time for recovery should be considered in developing management strategies for the lake. Decisions regarding management and restoration of lakes are often difficult and controversial as they involve regulating P loads from both internal and external sources.

1.7 Recommendations

All laboratory studies indicate that removal of surface sediments reduce potential P flux into water column. Experiments conducted primarily used a black box approach in addressing the effects of dredging on water quality. The next step is to identify biotic and abiotic controls regulating P flux from sediments to overlying water column. Future studies should focus on determining the interactions between biotic (microbially mediated processes, electron acceptors, and organic matter turnover) with abiotic (adsorption-desorption, redox, and pH) solubility and stability of P in subsurface sediments. These processes should be carefully examined to obtain insight on why deeper sediments have poor P retention capacity.

Phosphorus transport from sediments to overlying water column is primarily due to diffusion and resuspension. Phosphorus release due to resuspension is rapid, but is restricted to few events. Diffusion is a steady process and constantly supplies P to surface sediments, and resuspension of surface sediments transports P into water column. Future studies should repeat resuspension events for several days to evaluate sustained P release from sediments to water column. Sediment particle suspended into water column play a significant role in adsorbing P. For example, sediments suspended into high pH water during day time may behave differently than those suspended during night time into near neutral pH waters. Under alkaline conditions, OH^- ions can displace PO_4^{3-} ions from iron oxide minerals, thus making more P available to phytoplankton. Thus detailed characterization of suspended sediments is essential in evaluating internal load of P. Diffusive flux mediated by benthic organisms (bioturbation) in P exchanges between sediment and overlying water column needs to be determined.

Alternative approaches to dredging to reduce internal P load should be examined. These include the application of chemical amendments such as alum and lime should be considered. Interactive effects of these chemicals with sediment P forms should be evaluated in the context of stability and reactivity of the stored P.

1.7 References

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