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# Lake Drawdown as a Method of Improving Water Quality

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Ecological Research Series

# LAKE DRAWDOWN AS A METHOD OF IMPROVING WATER QUALITY

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# LAKE DRAWDOWN AS A METHOD OF IMPROVING WATER QUALITY

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## FOREWORD

Effective regulatory and enforcement actions by the Environmental Protection Agency would be virtually impossible without sound scientific data on pollutants and their impact on environmental stability and human health. Responsibility for building this data bsse has been assigned to EPA's Office of Research and Development and its 15 major field installations, one of which is the Corvallis Environmental Research Laboratory (CERL).

The primary mission of the Corvallis Laboratory is research on the effects of environmental pollutants on terrestrial, freshwater, and marine ecosystems; the behavior, effects and control of pollutants in lake systems; and the development of predictive models on the movement of pollutants in the biosphere.

This report describes the results of pilot scale investigations to determine if the drawdown of Lake Apopka, Florida, would be an effective restoration technique in reducing algal blooms and increasing the population of desirable macroscopic aquatic plants.

A.F. Bartsch
Director, CERL

## ABSTRACT

Investigations were made to determine the feasibility of radical drawdown as a restoration technique for Lake Apopka, Florida, a 12,545 hectare lake in central Florida. Field studies showed the lake to be hypereutrophic with continual algal blooms, mats of floating water hyacinths, and a flocculent organic muck bottom rich in interstitial water nutrients. Sediments were dredged from the lake bottom and placed in aquaria, columns, tanks and pools. Following dewatering and varying drying periods, the containers of sediment were refilled. A large number of physical, chemical and biological parameters were monitored before, during and following sediment drying. Results indicate that drawdown improves subsequent refill water quality. In muck sediments, drying causes significant water loss and shrinkage. Loss of organic material is minimal. During and following refill, sediment is colonized by two macroscopic aquatic plants, Typha (cattail) and the alga, Chara. Drying results in only minor chemical changes in muck sediment. Refill water in the pool test simulations has the same or lower nutrient content, lower turbidity, higher dissolved oxygen, lower temperature, fewer algae and a more diverse benthic invertebrate population. Based on these laboratory scale investigations, drawdown appears to be an effective restoration technique for Lake Apopka.



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We also thank Donald Schults of the Corvallis EPA Laboratory, who served as Project Officer, for his advice and patience.

## SECTION I

## CONCLUSIONS

In laboratory simulations, we have found that drying muck sediments from Lake Apopka, Florida, results in significant water loss and shrinkage. Loss of organic material is minimal. Peat sediments, while they lose water as a result of drying, do not consolidate appreciably.

The only two aquatic macrophytes which sprout on drying sediments are Typha (cattail) and Eichhornia (water hyacinth). Following flooding of the dried sediments, Typha is the only vascular plant which survives. In several simulations, the macroscopic green alga Chara (stonewort) colonized the dried sediments after flooding. Chara is easily transplanted by sprigging. Dried sediments remain consolidated for long periods of time (at least a year) following refill.

The benthic invertebrates on dried sediments following four months of refill show greater diversity than populations on control sediments.

Few differences were found for nitrogen and phosphorus forms between dried and undried sediment. Drying results in only minor chemical changes in muck sediment. Lake Apopka sediment has a low capacity to absorb added phosphate, and mixing of unconsolidated sediment with overlying water acts as a source rather than a sink for aqueous phosphorus.

Nutrient levels in water above dried sediments (pool simulations) are the same or lower than levels in control simulations. Because of the higher turbidity levels in the controls, levels of particulate nitrogen and phosphorus are lower in the treated pools.

Refill water in the pool test simulations has lower turbidities, higher dissolved oxygen values, and lower temperatures (because of lower turbidities) than water in the control pools. Phytoplankton in refill water above dried sediments is less numerous and contains fewer cyanophytes than water above undried sediments. Chlorophyll levels are also lower in the test simulations. A laboratory algal assay showed that ground up dried sediment supports lower algal populations than does wet sediment. Adenosine triphosphate levels and total plate counts at the sediment water interface of dried versus undried sediments were not significantly different, lending support to the conclusion that the changes occurring during sediment drying are predominantly physical.

## SECTION II

## RECOMMENDATIONS

Drawdown/sediment consolidation appears to be a feasible lake restoration technique. Its use is recommended for shallow lakes in which extensive muck deposits cause problems because of their ready suspension by wind-driven waves and outboard motors and their consequent nutrient release. Drawdown is most attractive for lakes in which the littoral zone slopes gently so that a small vertical drawdown exposes large areas of bottom. Any drawdown project needs to consider the effects of increased flow and possible sediment transport on downstream water bodies. If pumping is required to achieve substantial drawdown, care must be taken to minimize the amount of loose sediment that is pumped from the lake.

A serious effort should also be made to evaluate the potential for physical removal of muck and peat from lake bottoms for purposes of soil amendment. It is not naive to consider muck and peat a misplaced resource; the highly fertile and profitable muck farms north of Lake Apopka attest to this fact. An integrated program of drawdown/sediment consolidation and sediment removal in selected areas to deepen the lake and provide sand bottom for fish spawning would appear to be the most desirable approach.

A number of questions about the drawdown/sediment consolidation technique remain unanswered by this project and deserve further study. Consolidation appears to be irreversible and primarily physical in nature. Further work on the mechanism of consolidation and on the physical-chemical changes in the sediment matrix causing the irreversibility would be in order. The general applicability of drawdown as a means of sediment consolidation needs further work. The sediment characteristics required for good consolidation should be explored, probably through demonstration project on various types of lakes.

With respect to Lake Apopka, drawdown appears to be an effective means of restoration, and if properly done, should result in enhanced water quality. Several engineering and socio-political problems, such as the extent to which the lake level can actually be lowered, the effect of drawdown on downstream water quality, and diminished freeze protection provided by the lake during drawdown, were not addressed by this study and will ultimately determine the feasibility of drawdown for restoring the lake. Drawdown should be undertaken during late winter and early spring because there is greater rainfall in summer and the colder temperatures will limit germination of terrestrial vegetation. Two months appears an adequate drying time; longer periods result in problems of vegetation growing on the dried soil. Hyacinths should be controlled during drawdown and refill and ideally should be physically removed from the lake.

Finally, an integrated approach should be taken to Lake Apopka restoration. Drawdown is not by itself a cure-all, but coupled with other measures drawdown can improve lake water quality. Serious consideration

should be given to removal of dried muck from shore areas, perhaps transporting it by truck to the nearby muck farms. The possibility of pumping the wet muck sediments onto the muck farms should also be explored. Fish spawning areas need to be added to the lake, either by removing the muck down to the sand bottom or by covering the dried muck with sand. The former method would seem preferable since it will increase the lake depth. Consideration should be given to planting or seeding the littoral areas with desirable forms of aquatic vegetation. The cessation of preventable nutrient loading is, of course, imperative.

## SECTION III

## INTRODUCTION

Florida's mild climate, extensive coastline, and more than 8,000 lakes effectively attract tourists, new residents, and industry. The rapid growth of Florida's population has exerted heavy demands on its water resources, especially for fishing, boating, and water contact sports. In an effort to meet the desire for water front property, Florida's coastlines are being developed using dredge, fill, and channelization techniques. The same procedures are being used on natural and manmade lake shorelines. Flood control structures often are installed and prevent natural fluctuations in water level. Development on floodplains sometimes necessitates river channelization for rapid removal of excess water. When these physical manipulations of natural systems are coupled with wastes generated by the resulting population increase, serious water quality degradation occurs.

Water quality degradation in lakes is characterized by a shortening of food chains and by respiratory dominance. Unique vascular plant problems occur in Florida as a result of the proliferation of introduced exotics, such as Florida elodea (Hydrilla verticillata) and water hyacinth (Eichhornia crassipes). In extreme cases, eutrophic lakes are characterized by odoriferous algal scums, fish kills, oxygen depletion and resultant hydrogen sulfide production, buildup of benthic organic deposits, and floating islands of hyacinths or impenetrable growths of submerged vegetation. Fish camps and resorts on such deteriorated lakes are unable to attract tourists and the resulting loss of income and decreased property values can be significant. Public discontent is proportional to economic loss.

### LAKE APOPKA BACKGROUND AND HISTORY

Lake Apopka, a 12,545 ha lake (Figure 1) in central Florida, exhibits most of the systems of degradation mentioned above. According to Brezonik et al. (1969), Apopka is the most eutrophic of the lakes in the Oklawaha chain (Figure 2). Gourd Neck Springs, in the southwestern portion of Lake Apopka is considered the headwaters of the chain. Water flows north through Lake Apopka into Lake Beauclair via the Apopka-Beauclair Canal. Lake Beauclair empties directly into Lake Dora, which drains into Lake Eustis through the Dora Canal. Lake Harris water also enters Lake Eustis via Dead River. Lake Eustis is connected to Lake Griffin by Haines Creek, and Lake Griffin discharges directly into the Oklawaha River. A manmade canal connects Lakes Griffin and Yale (State of Florida, 1970). Table 1

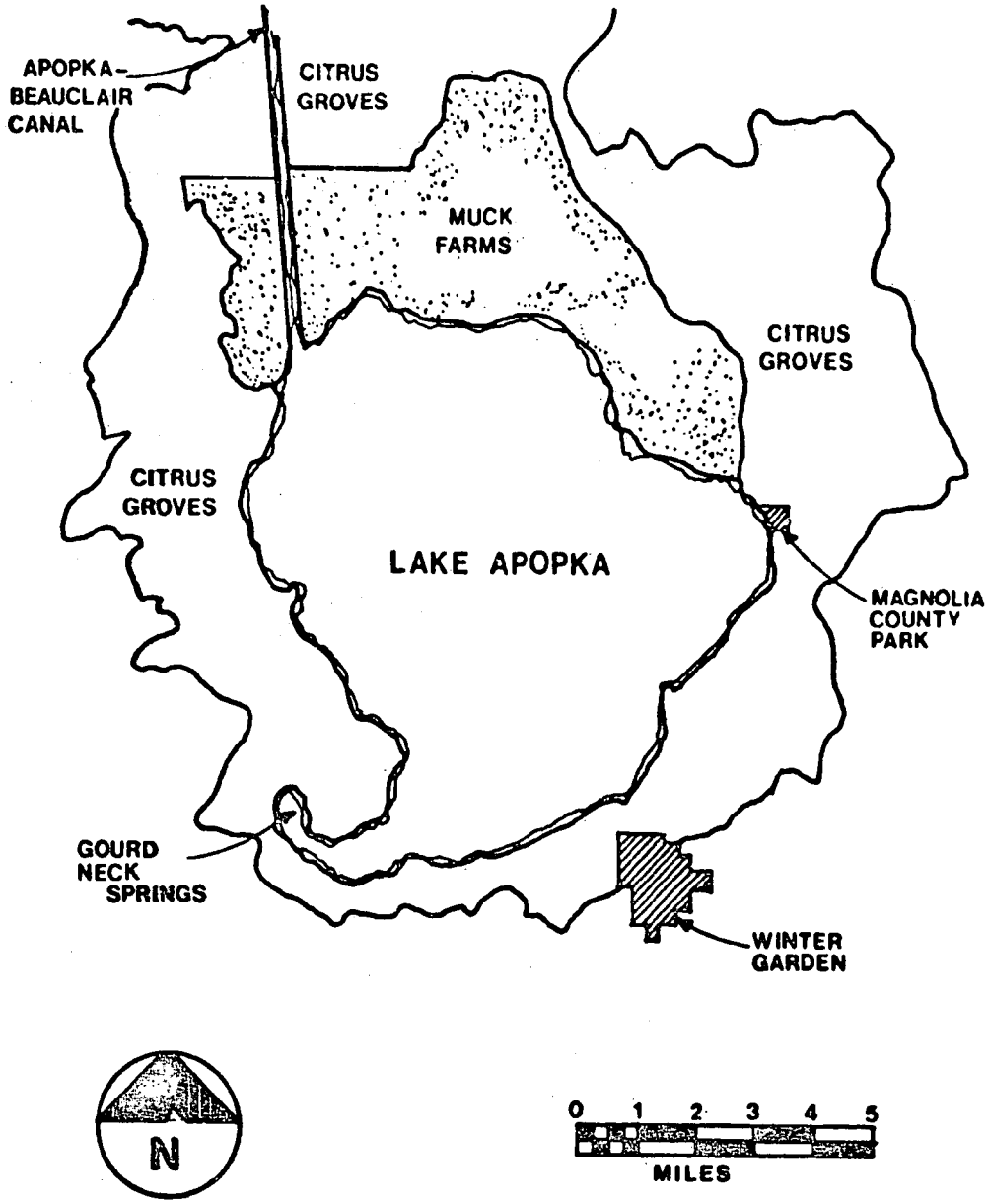


Figure 1. Lake Apopka and major land use types (modified from Perez 1972)

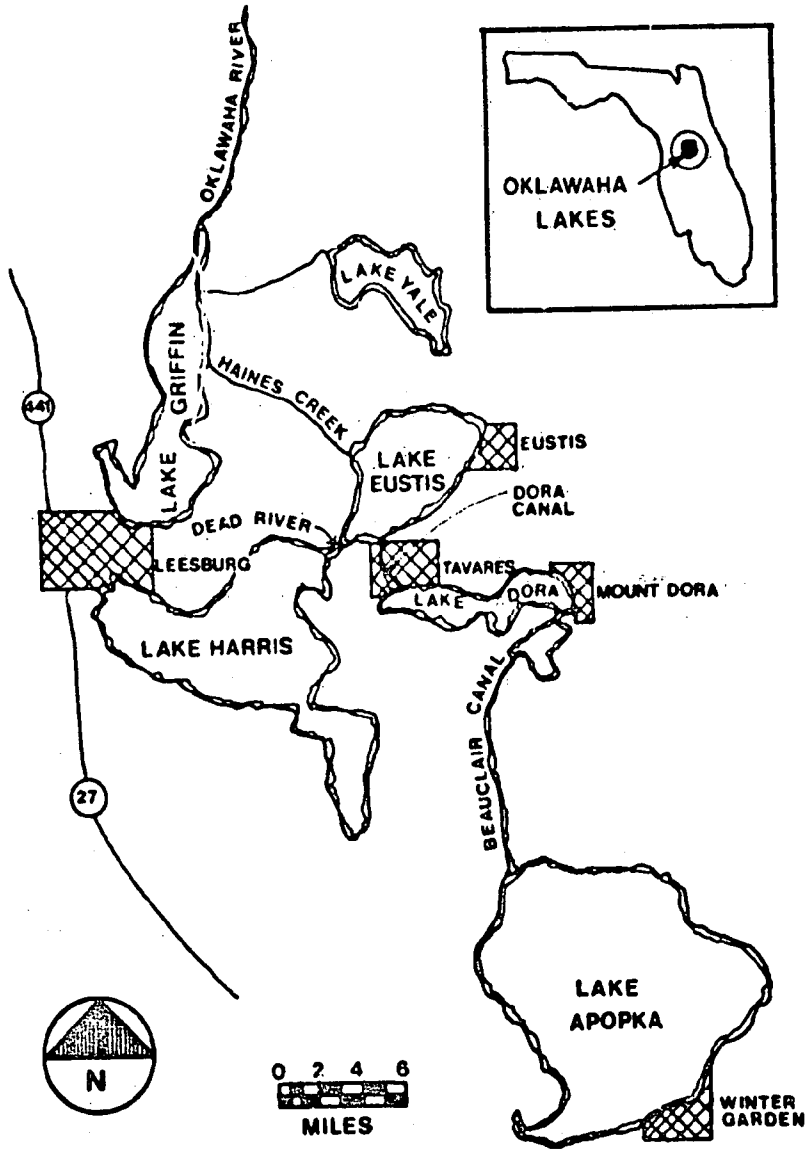


Figure 2. Oklawaha chain of lakes

provides a comparison of water quality parameters for the 5 lakes. The Oklawaha River has been and remains the subject of state and national concern, for it forms part of the Cross-Florida Barge Canal. A section of the Oklawaha already has been dammed as a part of canal construction. The resulting 5,260 ha impoundment is known as Rodman Reservoir. Further north and east, just above Lake George, the Oklawaha joins the St. Johns River, which discharges into the Atlantic Ocean east of Jacksonville.

A number of nutrient sources have been cited as causative factors in the eutrophication of Lake Apopka. These include agricultural runoff from vegetable farms (muck farms) and citrus groves, domestic waste effluent from the City of Winter Garden, and wastes from citrus processing plants. The relative importance of each of these nutrient sources has been the subject of considerable controversy. Perez (1972), Sherrield (1970), Heaney et al. (1971) and many others have all studied aspects of the problem. The lack of data has hindered the investigations, and an accurate hydrologic model and/or nutrient budget for Lake Apopka does not exist. Technical measures to control nutrient inputs, however, can be and are being instituted without precise information concerning the relative culpability of those responsible for nutrient contributions.

The history of Lake Apopka, presented in two Florida State Board of Health reports (1964 and undated) and summarized by a Federal Water Pollution Control report (Schneider and Little 1968), shows that a web of interrelated factors has led to the lake's present hypereutrophic condition. The lake first started receiving domestic waste effluent from Winter Garden in 1922 and citrus processing wastes in 1946. The vegetable farms, commonly referred to as muck farms, were completed by 1943. Excess runoff from the 7,290 ha (18,000 acres) of reclaimed lake bottom, until recently, has been pumped back into the lake or seeped under the dikes. By 1946, extensive growths of Vallisneria, Potomogeton, and Najas were present and bass and bream were abundant. The water was described as clear. These rooted aquatic plants were uprooted by a hurricane in 1947. Algal blooms and water hyacinths then began to appear. The submersed plants never returned, presumably because of the algal-caused turbidity. In 1948, the Beauclair Canal was constructed, connecting Lake Apopka with the downstream lakes in the Oklawaha Chain. Herbicides were also applied in 1948 to control the growth of water hyacinths. Algal blooms followed each treatment. In 1952, a water control structure was installed in the Beauclair Canal to regulate the water level of Lake Apopka. According to Schneider and Little (1968), since 1952 Lake Apopka has "changed from a lake to a mismanaged reservoir." A 1956-1957 study by the Florida Game and Fresh Water Fish Commission (1959, from Huffstutler et al. 1965) showed that game fish had dropped to 18 percent of the total population. The remaining rough fish were mostly shad, primarily gizzard and threadfin varieties. The bass population was found to consist mainly of adults, and reproduction levels were insufficient to maintain the population. In 1957, 1958, and 1959 rotenone was used to selectively kill the shad. According to Burgess (1964), "extremely dense phytoplankton blooms followed each treatment as a result of the released nutrients found in the bodies of the shad."

TABLE 1. CHEMICAL AND BIOLOGICAL CONDITIONS IN OKLAWAHA LAKES, OCTOBER, 1968[a] (FROM BREZONIK ET AL. 1969)

| Constituent[b] | Apopka | Dora | Harris | Eustis | Griffin |
|---|---|---|---|---|---|
| Diss. $O_2$ | 11.2 | 11.4 | 9.7 | 8.78 | 8.4 |
| Conductivity | 330 | 335 | 230 | 275 | 290 |
| pH | 9.5 | 9.5 | 8.8 | 8.9 | 8.9 |
| Alkalinity | 126 | 130 | 91 | 104 | 112 |
| COD | 113 | 157 | 48 | 112 | 78 |
| Color | 30 | 43 | 13 | 28 | 25 |
| Secchi Disc | 0.3 | 0.3 | 0.8 | 0.6 | 0.5 |
| Turbidity | 27 | 28 | 20 | 25 | 28 |
| Suspended Solids | 43 | 77 | 13 | 46 | 27 |
| TON | 3.8 | 5.3 | 1.7 | 4.1 | 3.0 |
| $NH_3$-N | 0.18 | 0.17 | 0.19 | 0.20 | 0.18 |
| Ortho $PO_4$ | 0.016 | 0.108 | 0.010 | 0.014 | 0.014 |
| $NO_2$-N | 0.004 | 0.005 | 0.004 | 0.006 | 0.007 |
| $NO_3$-N | 0.09 | 0.11 | 0.05 | 0.24 | 0.09 |
| Total P | 0.24 | 0.39 | 0.035 | 0.22 | 0.20 |
| $SiO_2$ | 6.4 | 0.05 | 3.4 | 1.4 | 0.40 |
| Fe (total) | T | 0.01 | 0.01 | 0.01 | 0.01 |
| Mn (total) | T | 0.003 | 0.006 | 0.005 | 0.003 |
| $Ca^{+2}$ | 25.1 | 30.5 | 23.1 | 26.3 | 28.9 |
| $Mg^{+2}$ | 14.7 | 14.2 | 6.1 | 10.7 | 9.6 |
| $Na^{+}$ | 11.7 | 18.1 | 9.7 | 14.5 | 14.0 |
| $K^{+}$ | 2.9 | 3.4 | 0.9 | 2.4 | 2.1 |
| $Cl^{-}$ | 21 | 22.3 | 12.8 | 18.7 | 17.5 |
| $SO_4^{=}$ | 10.2 | 12.3 | 4.9 | 7.1 | 6.7 |
| Chlorophyll a | 34.1 | 72.1 | 17.7 | 33.1 | 45.4 |
| Primary Prod.[c] | 0.386 | 1.02 | 0.037 | 0.274 | 0.183 |

[a]Results are average values for composite samples from three stations in each lake.

[b]Chemical species in mg/l except as follows: alkalinity in mg/l as $CaCO_3$, nitrogen species in mg N/l, phosphorus species in mg P/l, color in mg/l as Pt., specific conductance in $\mu mho\ cm^{-1}$, Secchi disc visibility in meters, chlorophyll a in $mg/m^3$.

[c]Primary production in mg C fixed/l-hr. Composite lake water samples were run in a laboratory incubator.

The persistence of phytoplankton blooms and adverse public opinion prompted the 1962-1964 Florida State Board of Health study (Huffstutler et al. 1965). In their report they concluded, among other things, that "the lake is in an undesirable biological condition and does not provide a satisfactory environment for fish food. This is caused by excessive nutrients contributed by many years, from the following sources: (a) domestic waste, presently treated, but untreated in years past, (b) untreated citrus packing, canning, and concentrating waste, (c) leaching from citrus growing, fertilizing and irrigating, (d) irrigation, drainage, and spraying in the truck farm areas, (e) allowing dead fish to remain and decay in the lake, and (f) allowing destroyed hyacinths to remain and decay in the lake." These findings, as well as newspaper articles such as the one entitled "Lake Apopka's Lingering Death" (Lane 1966, cited by Forbes 1968) further increased public discontent.

In 1967, the Governor of Florida requested the aid of the Federal Water Pollution Control Administration in evaluating the restoration of Lake Apopka. In March of 1968, the F.W.P.C.A. surveyed the lake to characterize the bottom materials and nutrient contributions by spring water, citrus grove runoff, and rainwater. They found 90 percent of the bottom to be covered by unconsolidated muck averaging 1.5 m (5 ft) in thickness. Only 5 percent of the bottom was characterized as being biologically productive. The distribution of nutrients in the bottom was relatively uniform over the lake. The muck contained from 1 to 4 percent (dry weight) total nitrogen, 0.02 to 0.21 percent total phosphorus, and 90 percent organic material. The F.W.P.C.A. considered citrus grove runoff and rainfall as significant sources of nitrogen and phosphorus. Gourd Neck spring water contributed low concentrations of these nutrients.

Additional data on the Apopka sediments were gathered by the Orange County Pollution Control Department (Sheffield 1967). This report concluded that the top portion of the sediment contained higher amounts of phosphorus and organic matter than underlying sediment. Calcium, magnesium, and sodium increased with sediment depth. The report also stated that "results of this study would tend to indicate that if the bottom material were allowed to dry and oxidize by the sun, a reduction in organic material would be realized."

The Corps of Engineers (Department of the Army 1967) also studied Lake Apopka sediments. In 1961, dry rod probings were made to determine sediment thickness along the northern shoreline. In 1965 and 1967, twenty-nine core borings were made in the same area, and lengths and physical descriptions of each core were presented. The surface of the study area was found to be covered by either peat or soft organic silt and clay deposits varying in depth from less than 0.3 m (1 ft) to over 15 m (50 ft). Below these organic sediments are interbedded deposits of clay, sand, and silt.

While there is a long history of disagreement concerning culpability for nutrient inputs to Lake Apopka, it is generally agreed that the lake is hypereutrophic and that something should be done about the situation. However, what should be done is also a subject of controversy, with suggestions ranging from commercial catfish farming to radical drawdown and sediment removal.

Some have suggested the entire lake bottom be converted to additional muck farm acreage.

## LAKE RESTORATION BY DRAWDOWN

The Florida Department of Pollution Control advocates drawdown and in 1972 submitted a proposal to the Environmental Protection Agency for funds to dewater the lake sediments (Florida Department of Pollution Control 1972). Drawdown as a method of lake restoration was suggested by reports of the Florida Game and Fresh Water Fish Commission on Lakes Trafford and Hancock. Both lakes showed improvement in water quality as a result of natural drawdown and subsequent exposure of bottom sediments. Lake Trafford, prior to its 1962-1963 drawdown, had a game to rough fish ratio of approximately 1 to 4, extensive algal blooms, and an unconsolidated muck bottom with few benthic organisms and little rooted aquatic vegetation. Drawdown caused by a drought exposed 15 percent of the bottom sediment for a period of 6 months. The original 30 cm (1 ft) of muck consolidated to 5-10 cm (2-4 in). After the lake refilled naturally, the consolidated sediments supported benthic organisms and rooted aquatic plants. With a restored habitat and food supply, the game fish returned. Less than a year after refilling, the game fish population had increased from its pre-drawdown 20 percent to 35 percent of the total. In 1968, 5 years after drawdown, game fish comprised 76 percent of the total population.

Information on Lake Hancock is less detailed, but the story is similar. In 1965, the lake was reported to be highly eutrophic, characterized by almost continual algal blooms, unconsolidated muck, and a paucity of both benthic organisms and rooted aquatic vegetation. In 1968, natural drawdown occurred and sediments were exposed for an unreported period. The effects of exposure included increases in water transparency, bottom firmness, benthic organisms in the littoral zone, and rooted aquatic vegetation.

More recent experiences with drawdown of Florida lakes suggest the method is effective in improving certain aspects of water quality. In 1971, Lake Tohopekaliga, a 9,200 ha lake in Osceola County, was drawn down in an effort to improve the fish habitat. According to a report by Florida Game and Fresh Water Fish Commission (1972-1973 Annual Report), the standing crop of fish doubled as a result of drawdown. The invertebrate population also increased. *Hydrilla verticillata*, however, became established following reflooding. *Hydrilla* is a particularly troublesome submersed vascular plant in Florida. Although the consolidated sediments remained firm following refill, the report concluded that at the present rate of organic deposition (of decomposing hyacinths and algae) the condition of the substrate in the littoral zone would be comparable to that which existed prior to drawdown in a relatively short period. In addition, algal diversity decreased and bloom frequency increased after refill, and chemical water quality did not improve. Total organic nitrogen and total phosphate were higher following drawdown. The Tohopekaliga experiment,

while successful in terms of increasing fish biomass, did not cause a long-term change in trophic status, undoubtedly in major part because significant amounts of sewage effluent (a total of 2.0 mgd) from 5 plants have continued to enter the lake. The authors concluded that "any long-term improvement in the lake requires complete removal of the large and ever-increasing volume of sewage being discharged to it."

Under the direction of the Orange County Department of Pollution Control, Lake Eola, in Orlando, was drawn down in 1972. According to C. W. Sheffield, (1974), who directed the project, "at least partial success was attained." Although fishing has improved, there are still occasional algal blooms. The blooms are probably caused by the continued input of storm water drainage. The Eola experience again points to the need for curtailment of nutrient sources as an essential component of any lake restoration scheme.

## LAKE APOPKA RESTORATION PLAN

From a practical standpoint, drawdown appears to be the most feasible technique for restoring a lake the size of Apopka. Other restoration techniques, such as hypolimnetic aeration, dredging, bottom sealing, and chemical treatment are impractical for or not applicable to Lake Apopka for a number of obvious reasons. Because of past experience with drawdown and its apparent economic feasibility, the Florida Department of Pollution Control has chosen that method of restoration for Lake Apopka. The 1972 D.P.C. proposal to the Environmental Protection Agency basically calls for pumping Apopka water through the Beauclair Canal, through the Oklawaha lake chain and into the Oklawaha River. Extensive pumping would create a relatively narrow river of water from Gourd Neck Springs, in the southern end of the lake, to the pumps at the northern end. Following the drying period, refill would occur naturally, principally from Gourd Neck Springs inflow, subsurface seepage, and surface runoff. Table 2 shows a summary of the Florida Department of Pollution Control's restoration plan.

Anticipated results include lake deepening and an improvement in water quality due to retardation of nutrient flux to overlying waters. This reduction in nutrient transport will theoretically result from both oxidation of the sediment surface (Mortimer 1941) and the physical consolidation of the sediment. Consolidated sediments are less likely to be physically suspended by agitation than are unconsolidated sediments. This latter factor is particularly important in Lake Apopka, for turbulence produced by mild storms and even small outboard motor boats readily brings bottom sediments to the water surface. The hardened substratum should provide a growth platform for rooted aquatics, thereby increasing the available niches to support a diversified food web.

In view of the large size of the proposed project and the lack of information concerning the effect of drying on Apopka sediment, the Environmental Protection Agency understandably required further data before making a decision on the State's proposal. The present project was funded in order to obtain some basic data concerning the effects of dewatering Apopka sediments on subsequent refill water quality.

TABLE 2. MAJOR LAKE APOPKA RESTORATION FEATURES
(FROM FLORIDA DEPARTMENT OF POLLUTION CONTROL, 1972)

Abatement of man-made wastes into lake:

From farms by July - September 1973
From Winter Garden municipal waste treatment by mid-1973

Consolidation of lake bottom sediments:

Total bottom sediments exposed: 29,500 acres
Bottom sediments dried: 16,500 acres
% bottom sediments dried/total lake sediments: 50%

Water quantities:

Water drawdown from approx. 66.5 ft msl to 58 ft msl
Water in lake at 66.5 ft msl approx. 191,000 to 197,000 ac.-ft.
Water in lake at 58 ft msl approx. 1500 to 7500 ac.-ft.
Water into lake from springs: approx. 60 cfs (38.8 mgd)

Lake dredging:

To provide settling basin for silt removal prior to water being pumped downstream
To connect some deep water pockets (10 ft deep or more)

Treatment of drawdown water:

Silt settling using special barriers
Time phasing and flow regulation to reduce high nutrient loads
Contingency operations involving diversion and pump stoppage
In-canal aeration

Total project length: 700 to 850 days (3 concurrent fiscal yrs)

## SECTION IV

## STUDY APPROACH

In predictive studies of large lake phenomena, several approaches are possible. These include mathematical modeling, in situ lake studies, and laboratory simulations. Economic and scientific constraints usually dictate which technique is feasible. In this study, laboratory simulations were used for several reasons. Modeling was inappropriate because of the lack of quantitative information available concerning sediment-water exchange phenomena and Lake Apopka ecosystem dynamics. A more lengthy consideration was given to in situ drawdown studies. Early in the study, a trip by boat was made around the entire Apopka shoreline in a search for a small embayment which could be separated from the lake proper and drawn down. No such embayment could be found. We also made economic estimates of isolating a small section of the lake by using large diameter pipes or earthen dikes. These schemes were also found to be prohibitively expensive or technically unfeasible.

In all, five types of containers were used for the lake simulations. Preliminary experiments were conducted in small glass aquaria and plexiglas cylinders. For the full scale experiments, we used columns, tanks and pools. The columns (Figure 3) were constructed of 0.6 cm thick plexiglas and were 0.3 m by 1.2 m high. Drainage portals were made by drilling 2.5 cm diameter holes 2.5 cm from the bottom of each column. The holes were plugged from the inside with a rubber stopper, through which a glass tube extended to the outside. Attached to the glass tubes were plastic tubes controlled by pinch clamps.

The tanks (Figure 4) consisted of 1.5 m lengths of 1.2 m corrugated steel pipe embedded in concrete. Drainage ports consisting of 15.2 cm capped sections of 5.1 cm pipe were welded into holes about 7.6 cm from the bottom of each tank. Plankton netting (1 mm pore size) covered each portal to prevent loss of the bedding material. The concrete was washed with sodium tripolyphosphate and the metal surfaces of the tanks were cleaned using steel wool and steel scrub brushes. All surfaces (inside and out) were then covered with two coats of white epoxy concrete enamel. Brown pea gravel (12.7 cm) and builder's sand (2.5 cm) were placed in the bottom of each tank to facilitate draining.

A final series of experiments was conducted using plastic pools with a diameter of 2.4 m and a depth of about 0.5 m. Submersible electric water pumps were used to stir the water overlying the sediments.

The Lake Apopka sediment and water used in the simulations were retrieved as follows: A 3 hp (2237 J/sec) was used to pump muck from the Lake Apopka bottom into 55 gal (208 l) drums. The pump was operated from an 18 ft (5.5 m) boat powered by a 40 hp (29828 J/sec) outboard motor. The pump was equipped with 7.5 m rubber intake

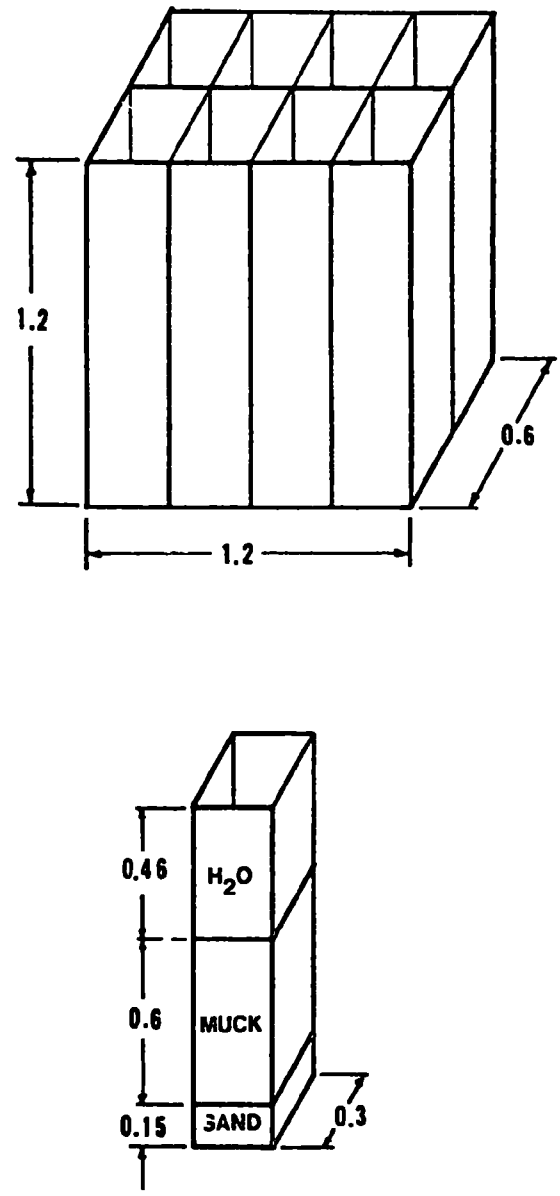


Figure 3. Plexiglas column design (dimensions in meters)

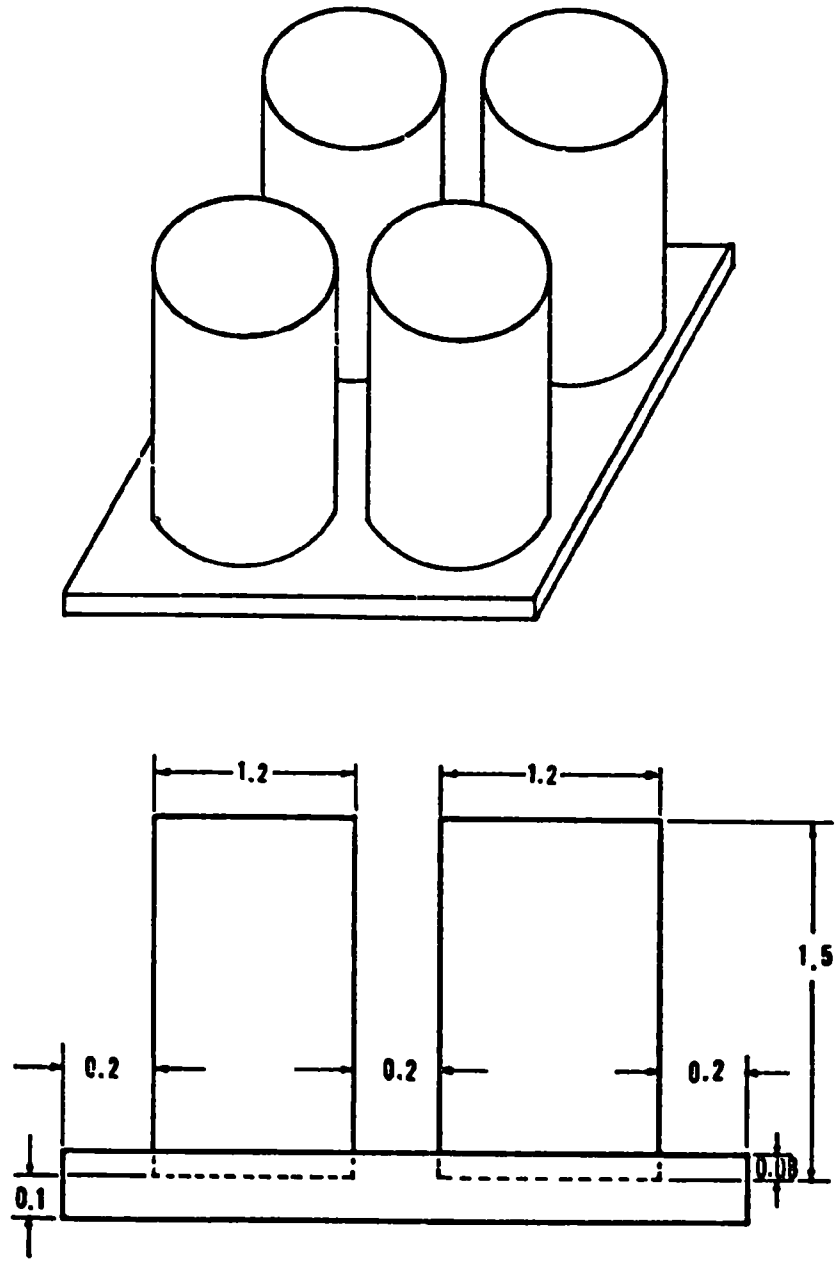


Figure 4. Steel tank design (dimensions in meters)

and discharge hoses. The intake hose had a weighted screen with a 0.64 cm mesh to prevent the access of larger pump damaging particles such as snail shells. During dredging, the boat was used to pull the hose through the sediment, providing continual intake of sediment from a depth of 15-30 cm below the sediment-water interface. When the intake hose was not dragged through the sediment, the end dug through the muck sediment to underlying sand and clay material, and brought up this material along with large amounts of water.

A wooden raft supported by eight 55 gal (208 l) drums was used to support five drums into which the dredged muck was discharged. The water depth we were working in was about 1.5 m. A total of 2,000 gal (7,570 l) of muck were retrieved in this manner and transported by truck to Gainesville, where the drums were emptied by pouring into the simulations. With the exception of special laboratory studies, the drawdown investigation was conducted out of doors.

The peat was recovered from about 20 m offshore in about 1 m of water in the same area of the lake. The material was dug up and shovelled into plastic garbage bags.

Overlying water was retrieved just offshore from the Winter Garden, Florida Municipal boat ramp. A gasoline powered centrifugal pump was used to pump the water directly from the lake into eighteen 55 gal (208 l) drums on the back of the rental truck. These were emptied by pouring over an inclined board over the sediment surface in the tanks and columns. Refill water was obtained by carrying the drums and pumps to Gourd Neck Springs in the research vessel and on a rented motorized pontoon boat, filling the drums, then transporting these back to the dock and pumping the water into drums inside the truck.

Before obtaining the large quantities of sediment for the simulations, studies were made of sediments from five locations in Lake Apopka. The results (see Section V) showed that, with the exception of sediment taken off the Winter Garden boat ramp, nutrient concentrations, volatile solids, and water content values for representative sediments were fairly consistent. Near Winter Garden, high phosphorus levels may be caused by sewage disposal. Because of the ease of access, we recovered sediment for the simulations from 200 m offshore from the Magnolia County Park on the northeast shoreline of Lake Apopka. Routine water quality data was also obtained from this area of the lake.

The basic study plan for the simulations included: (1) a period of equilibration and baseline data gathering, (2) sediment exposure, (3) refill, and (4) post-drawdown monitoring. The parameters measured and results for each are described in the following sections. A summary of methods is included in the Appendix. Special laboratory studies are included where appropriate.

## SECTION V

## TROPHIC STATE AND SEDIMENT CHARACTERISTICS OF LAKE APOPKA

This section summarizes the results of routine water quality monitoring conducted on Lake Apopka through the duration of the project. Major parameters considered were primary productivity, algal biomass (chlorophyll a) and levels of nitrogen and phosphorus species. These data are useful for comparison with the lake simulation results and, coupled with previous studies, provide a data base for the pre-drawdown trophic condition of Lake Apopka. This section also includes the results of a synpotic survey of Lake Apopka sediments to evaluate the potential for nutrient leachability.

Previous studies have clearly shown that Lake Apopka is greatly enriched. Brezonik and Shannon (1971) found that the lake had the highest trophic state index (TSI) of 55 lakes studied in northern and central Florida and suggested that agricultural runoff and domestic waste coupled with the lake's shallow depth ($z = 1.3$ m) were the primary factors responsible for its presently degraded conditions.

For the purposes of biological and chemical water quality monitoring, we selected three stations in the area from which sediments were collected for the simulations. The stations were located 40, 165, and 330 m offshore (stations 1-3, respectively) on a line extending east from the Magnolia County Park. The results of the chemical analyses are shown in Table 3 and represent grand means for all stations and all sampling times. No major seasonal trends could be discerned and differences in results between the individual locations were minor and inconsistent. Temporal variations in water quality are sporadic rather than seasonal and reflect algal bloom-crash phenomena which occur throughout the year.

The results in Table 3 compare well with two years of data (Table 4) obtained from composite samples of the lake (Brezonik and Shannon, 1971) and indicate that the lake contains extremely high levels of both organic and dissolved inorganic nutrients. High COD, Kjeldahl nitrogen and total phosphorus values reflect the abundance of algae. Secchi disk readings in Lake Apopka ranged from 15-30 cm and averaged about 22 cm. These are among the lowest transparencies we have measure in Florida lakes and indicate continuously high turbidity from profuse algal growth. By comparison, Secchi disk transparencies of 2-3 m are usually cited as the critical values separating oligotrophic and eutrophic lakes. High daytime pH and dissolved oxygen values also reflect intense algal activity. Among the inorganic nitrogen forms, ammonia is always higher than nitrate and in spite of the continually high algal concentrations, total inorganic N is never depleted. In fact, the minimum concentration found (0.4 mg N/l) during the two years of monitoring exceeds the often cited critical spring maximum figure for temperate lakes (0.3 mg N/l). Similarly, orthophosphate is never depleted, and the minimum value found (0.09 mg P/l) is far in excess of the critical concentrations cited by limnologists for algal bloom stimulation. Clearly inorganic nutrient

TABLE 3. SUMMARY OF WATER CHEMISTRY DATA FROM LAKE APOPKA MONITORING, 1972-1974

| Parameter | Mean[a] | Range |
|---|---|---|
| Dissolved $O_2$ (mg/l) | 10.2 | 9.5 - 12.9 |
| pH | 9.05 | 8.5 - 9.8 |
| Alkalinity (mg/l as $CaCO_3$) | 145 | 133 - 152 |
| Total Organic N (mg/l) | 3.00 | 1.95 - 3.98 |
| Particulate Organic N (mg/l) | 1.65 | 1.33 - 0.78 |
| Ammonia (mg N/l) | 0.55 | 0.33 - 0.78 |
| Nitrate (mg N/l) | 0.12 | 0.07 - 0.25 |
| Total Phosphate (mg P/l) | 0.26 | 0.09 - 0.32 |
| Total Soluble Phosphate (mg P/l) | 0.05 | 0.03 - 0.07 |
| Orthophosphate (mg P/l) | 0.024 | 0.016 - 0.035 |
| COD (mg/l) | 135 | 110 - 178 |

[a]Average values of all three stations for 12 sampling times over the period from August, 1972, through March, 1974

TABLE 4. SUMMARY OF WATER QUALITY DATA FROM LAKE APOPKA FOR 1969-1970[a]

| Parameter | Average | Range |
|---|---|---|
| Secchi Disc (m) | 0.22 | 0.15 - 0.30 |
| Primary Production (mg $C/m^3$-hr) | 337 | 231 - 521 |
| Chlorophyll a ($mg/m^3$) | 60.4 | 34 -73 |
| Dissolved Oxygen (mg/l) | 10.14 | 9.40 - 11.13 |
| pH | 8.85 | 8.42 - 9.57 |
| Alkalinity (as mg/l $CaCO_3$) | 140 | 126 - 148 |
| Conductivity (umho/cm) | 315 | 304 - 324 |
| Total Organic Nitrogen (mg/l) | 4.45 | 3.45 - 5.74 |
| Ammonia (mg/l of N) | 0.27 | 0.08 - 0.58 |
| Nitrate (mg/l of N) | 0.15 | 0 - 0.26 |
| Orthophosphate (mg/l of P) | 0.065 | 0.004 - 0.110 |
| Total Phosphate (mg/l of P) | 0.380 | 0.239 - 0.500 |
| $Na^+$ (mg/l) | 14.33 | 10.00 - 18.00 |
| $K^+$ (mg/l) | 3.73 | 1.20 - 5.00 |
| $Mg^+$ (mg/l) | 14.3 | 12.9 - 16.0 |
| $Ca^+$ (mg/l) | 55.3 | 25.0 - 100 |
| $Mn^+$ (mg/l) | 7.0 | 5.0 - 9.0 |
| $Cl^-$ (mg/l) | 23.5 | 21.3 - 27.5 |
| $SO_4^{-2}$ (mg/l) | 16.3 | 10.2 - 20.0 |
| $F^-$ (mg/l) | 0.41 | |
| COD (mg/l) | 159 | 113 - 191 |

[a]Data summarized from Brezonik and Shannon (1971)

concentrations do not limit algal production in Lake Apopka at the present time. More likely, physical factors, especially light, are more important in controlling *in situ* production. In summary, the chemical data indicate that Lake Apopka is hypereutrophic and has highly degraded water quality.

Primary production in Lake Apopka consistently occurred at extremely high levels. Carbon-14 uptake rates and chlorophyll levels for 1972-73 are plotted in Figure 5. Average chlorophyll a during 1972-73 was 84 $mg/m^3$, a value about 40 percent higher than the 1969-70 data reported by Brezonik and Shannon (1971). Chlorophyll a levels of 5-10 $mg/m^3$ are usually cited as the dividing point between oligotrophic and eutrophic lakes.

Although there are some temporal fluctuations in both chlorophyll *a* and productivity values, seasonal trends are not pronounced. The minimum chlorophyll *a* level occurred in October (56 $mg/m^3$) and the maximum productivity occurred in May and December (about 0.32 g $C/m^3$-hr). Maximum productivity occurred in June (0.53 g $C/m^3$-hr) and the mean over the period of measurement was 0.42g $C/m^3$-hr, a high rate also indicative of hypereutrophic conditions.

Measurement of total community metabolism using diurnal oxygen changes (Odum and Hoskin, 1958) also showed Lake Apopka to be extremely productive. Figure 6 shows typical diurnal changes in dissolved oxygen and temperature as measured in August, 1972. The computed productivity values for four dates in 1972-7 (Table 5) are within the range for some of the world's most productive systems during favorable periods (Odum 1971). The metabolic activity of Lake Apopka is largely due to the activity of blue-green algae (*Microcystis* sp., *Lyngbya* sp., *Oscillatoria* sp., and *Anabaena* sp.). Productivity levels in Lake Apopka compared closely to values found in tropical East African lakes (Melads and Kilham 1974). These lakes were also shallow (Z less than 2 m) and were dominated by cyanophytes (mainly *Spirulina* sp.).

An initial survey of Lake Apopka sediments was made for the purpose of determining the physical and chemical characteristics of muck from five lake locations. This survey was done to evaluate the variability of sediment conditions within the lake and determine the degree to which sediments collected for the large scale drawdown simulations reflected the average lake sediment conditions. The results shown in Table 6 indicate that muck sediment is rich in nutrients and organic matter. The high water content reflects the highly flocculent nature of the muck. No definite sediment-water interface exists in the lake and the muck freely mixes with overlying water.

In order to assess sediment nutrient release potential, 25 Eckman dredge samples were taken from points over the entire lake (Figure 7) and examined for interstitial water nutrient content. Table 7 shows that the interstitial water is quite rich in ammonia and orthophosphate, which is released to provide algal nutrients when mixing of the sediments and water occurs. Average orthophosphate in the interstitial water of all the samples was 3.23 mg P/l, while for muck sediment alone

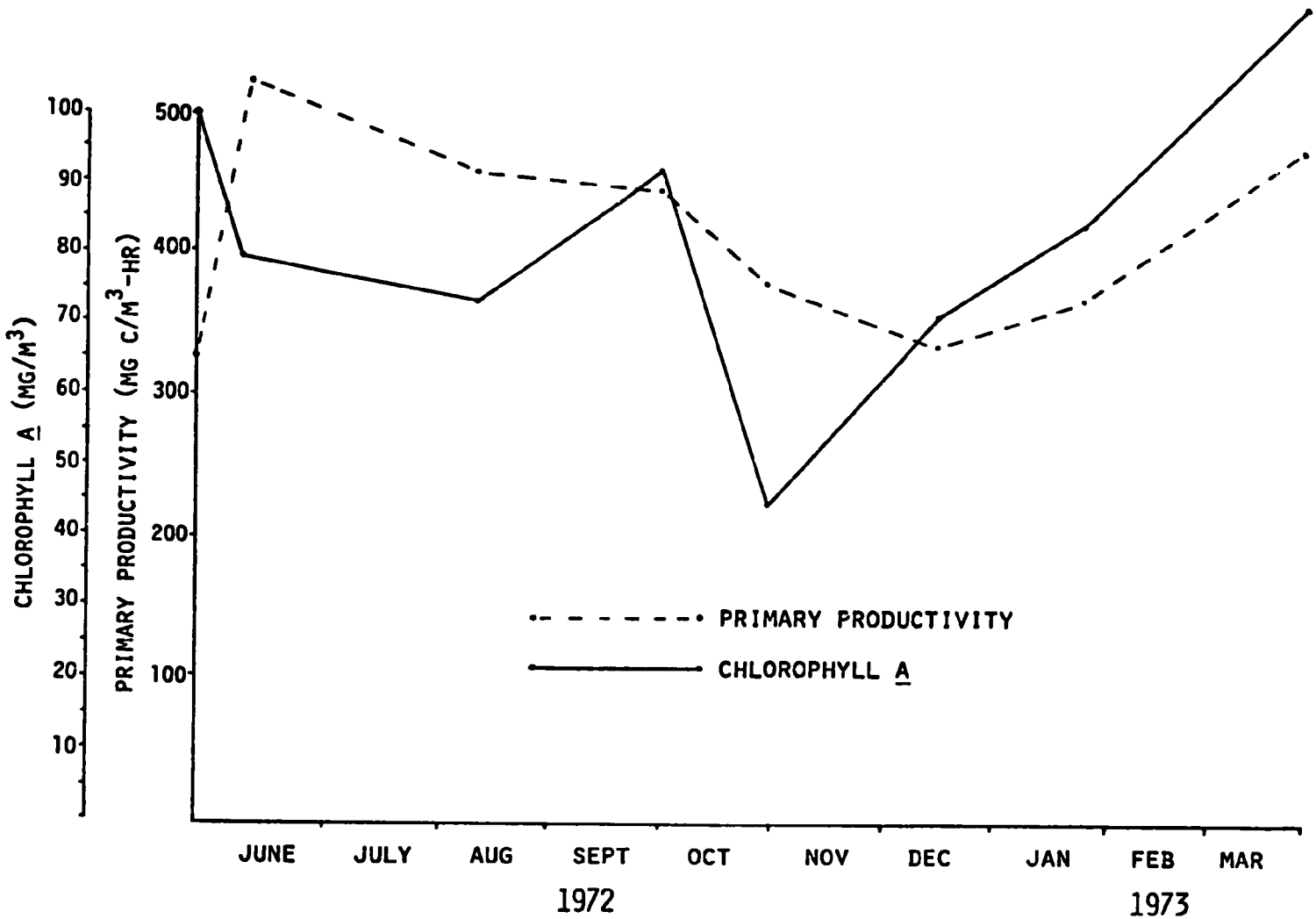


Figure 5. Chlorophyll a and carbon-14 productivity at Station 3, Lake Apopka

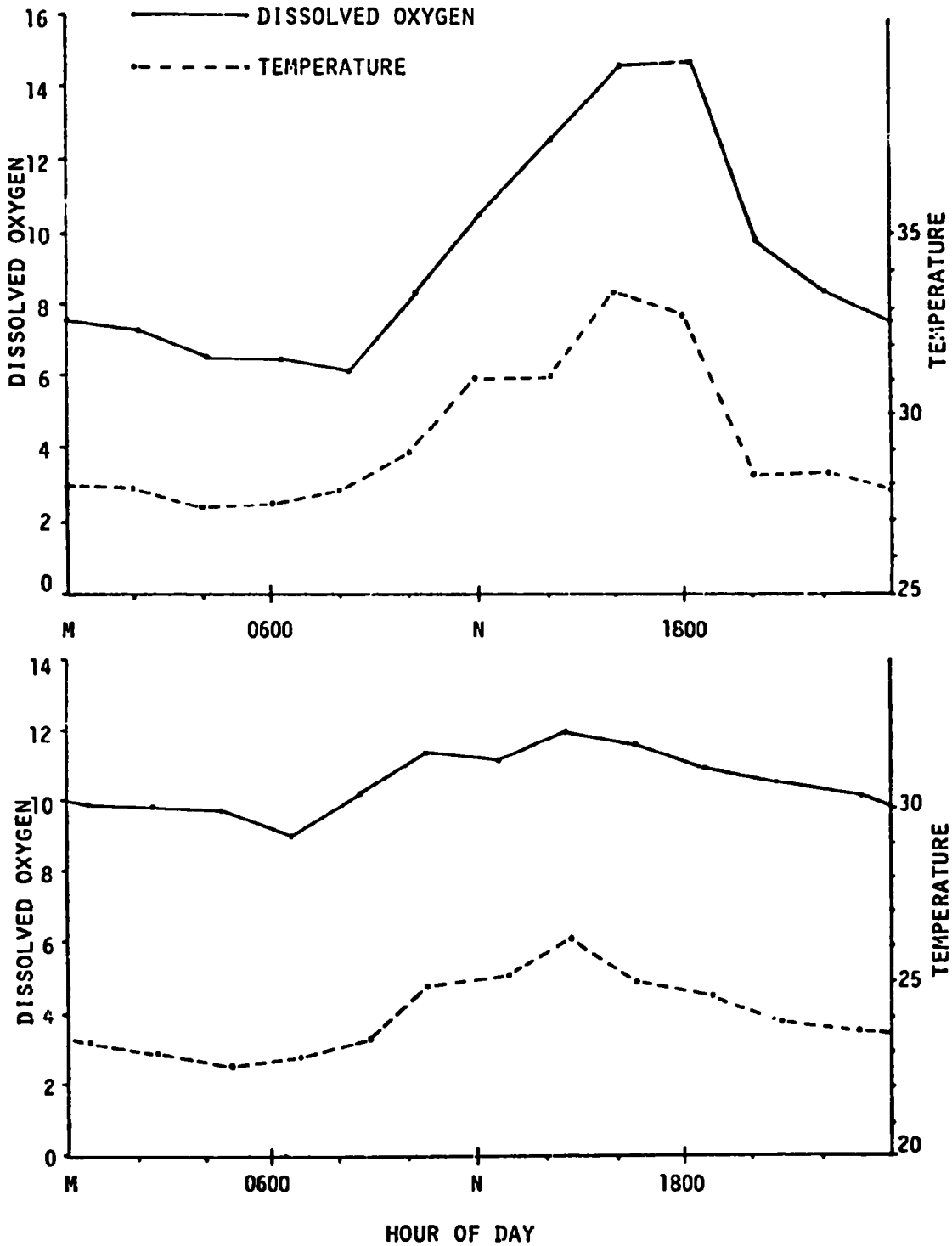


Figure 6. Diurnal changes of dissolved oxygen and temperature, Lake Apopka, August, 1972

TABLE 5. COMMUNITY METABOLISM IN LAKE APOPKA

| Station and Date | Chlorophyll a (mg/m$^2$) | Gross Production (g $O_2$/m$^2$-day) | Community Respiration (g $O_2$/m$^2$-day) | Solar Energy (Kcal/m$^2$-day) | $P_g/R$ | Percent Efficiency ($P_g$ x 100/Solar Energy)[a] |
|---|---|---|---|---|---|---|
| 8/18/72 | | | | | | |
| Station 3 | 74 | 34 | 22 | 5870 | 1.5 | 2.89 |
| 10/30/72 | | | | | | |
| Station 3 | 46 | 6 | 8 | 4800 | 0.8 | 0.63 |
| 12/5/72 | | | | | | |
| Station 3 | 68 | 18 | 19 | 5840 | 0.95 | 2.05 |
| 4/11/73 | | | | | | |
| Station 3 | 121 | 38 | 29 | 7000 | 1.33 | 1.22 |
| Station 1 | 105 | 32 | 30 | 7000 | 1.07 | 1.02 |
| Mid-lake | 130 | 38 | 24 | 7000 | 1.57 | 1.21 |
| North Shore | 98 | 26 | 14 | 7000 | 1.77 | 0.80 |

[a] Calculated as Kcal/m$^2$-day, conversions (1 g $O_2$ = 1 g organic matter = 5 Kcal) from Odum, 1971

TABLE 6. COMPARISON OF MUCK FROM DIFFERENT LOCATIONS IN LAKE APOPKA

| Location | TON[a] | $NH_3$-N[a] | Total P[b] | % $H_2O$ | Volatile Solids[c] |
|---|---|---|---|---|---|
| Winter Garden Boat Ramp | 28.9 | 0.11 | 0.93 | 92.5 | 53.1 |
| Sandy Point | 23.2 | 0.10 | 0.41 | 95.6 | 70.2 |
| Gator Island | 25.4 | 0.10 | 0.32 | 97.3 | 67.4 |
| Winter Garden Pier | 28.1 | 0.15 | 0.62 | 96.8 | 67.2 |
| Station 3 | 29.6 | 0.15 | 0.62 | 96.8 | 67.2 |

a mg N/g (dry weight)

b mg P/g (dry weight)

c percent of dry weight

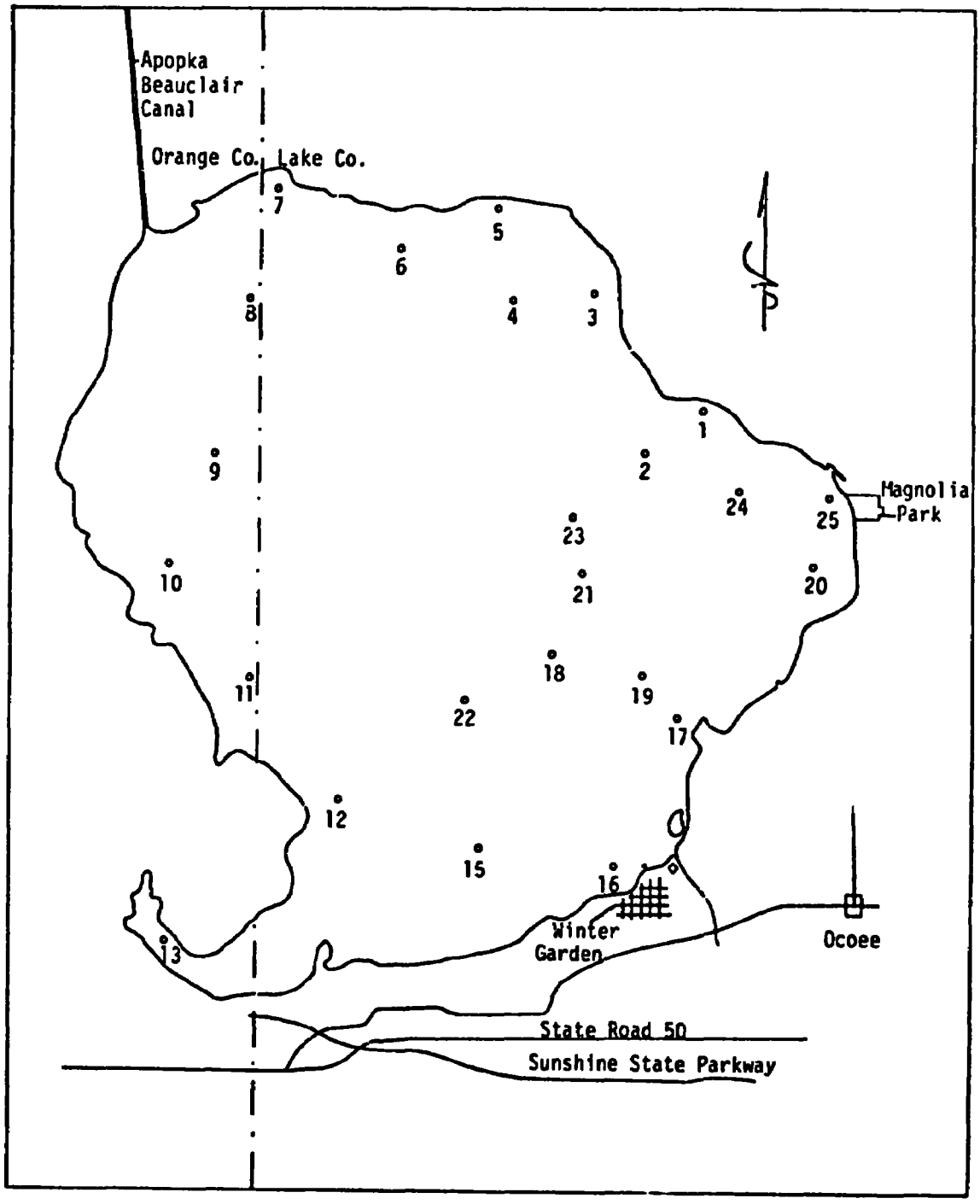


Figure 7. Lake Apopka sediment sampling locations

TABLE 7. NUTRIENTS IN INTERSTITIAL WATER OF LAKE APOPKA

| Station | Bottom Type | Concentration in Interstitial Water | | | |
|---|---|---|---|---|---|
| | | Ortho-Phosphate | | Ammonia | |
| | | mg P/l | mg P/g dry wt | mg N/l | mg N/g dry wt |
| 1 | Peat Bottom | 1.40 | 0.003 | 3.30 | 0.008 |
| 2 | Hard Sand and Rock (not analyzed) | - | - | - | - |
| 3 | Peat | 1.40 | 0.001 | 4.00 | 0.003 |
| 4 | Muck and Peat Mixed | 2.50 | 0.02 | 10.5 | 0.08 |
| 5 | Muck and Peat | 2.30 | 0.001 | 6.00 | 0.003 |
| 6 | Sand | 1.30 | 0.0002 | 6.00 | 0.001 |
| 7 | Muck and Peat | 2.70 | 0.02 | 10.5 | 0.08 |
| 8 | Muck | 3.40 | 0.03 | 11.0 | 0.11 |
| 9 | Soft Mud | 5.00 | 0.08 | >13.0 | >0.22 |
| 10 | Muck | 6.40 | 0.10 | >13.0 | >0.21 |
| 11 | Muck | 5.00 | 0.08 | >13.0 | >0.20 |
| 12 | Sand | 1.70 | 0.001 | 10.5 | 0.006 |
| 13 | Muck | 4.20 | 0.09 | >13.0 | >0.27 |
| 14 | Muck | 5.80 | 0.09 | >13.0 | >0.21 |
| 15 | Muck | 3.00 | 0.04 | 13.0 | 0.19 |
| 16 | Muck | 6.20 | 0.12 | >13.0 | >0.25 |
| 17 | Sand | 1.10 | 0.0001 | 5.30 | 0.0004 |
| 18 | Muck and Peat | 1.75 | 0.02 | 10.5 | 0.09 |
| 19 | Muck | 3.50 | 0.03 | 13.0 | 0.11 |
| 20 | Peat | 2.50 | 0.002 | 10.5 | 0.009 |
| 21 | Muck | 4.80 | 0.07 | >13.0 | >0.19 |
| 22 | Muck | 6.30 | 0.09 | >13.0 | >0.18 |
| 23 | Muck | 0.60 | 0.004 | 7.20 | 0.05 |
| 24 | Muck | 1.30 | 0.02 | 13.0 | 0.18 |
| 25 | Peat | Not enough interstitial water to analyze | | | |

the average was 3.74 mg P/l. Corresponding mean values for ammonia were about 10 and > 13 mg N/l. Muck sediments, therefore, have the greatest potential for nutrient release to overlying waters. They are more flocculent than other sediment types and hence more likely to be disturbed by physical agitation. Furthermore their interstitial waters contain higher concentrations of dissolved nutrient compared to the interstitial water found in peat, sand or mixed sediments.

## SECTION VI

## EFFECTS OF DRAWDOWN ON SEDIMENTS

### PHYSICAL EFFECTS

#### Preliminary Studies

For preliminary aquarium studies, muck sediment was obtained from the northeastern section of the lake by Ekman dredge. Wet sediment was added to glass aquariums to produce depths of 1 inch (2.5 cm), 3 inches (5 cm) and 6 inches (15 cm). All were allowed to dry in the sun for 14 days. During that time, moisture loss, consolidation and volatile solids were monitored. Following the two-week drying period, the aquaria were filled with equal volumes of Gourd Neck Spring Water (the principal groundwater source for the lake), and water quality in the aquaria was monitored for the next 28 days. The aquaria were maintained at 25°C and received approximately 0.45 langleys/min (1024 foot candles) of light during a 12-hour daily photoperiod.

Figures 8 and 9 show the moisture loss and extent of consolidation for the three depths of sediment. The original wet sediment was 96.4 percent water and the dry sediment contained 65.6 percent volatile solids. As might be expected, water loss and consolidation were dependent on the original sediment depth. Deeper sediment lost water more slowly and hence consolidated less in the same amount of time than did the more shallow sediments. After cracking occurred, consolidation measurements were discontinued since both horizontal and vertical shrinking occurred. No detectable decrease in volatile solids occurred as a result of drying indicating that consolidation is due primarily to water loss. Although consolidation measurements were discontinued following refill, it was apparent that only minimal swelling of the consolidated sediments occurred. The refill water quality parameters are discussed in following sections.

A second drying experiment was set up to test the affect of subsurface drainage on consolidation rate. Two 76 cm sediment columns 7.5 cm in diameter were allowed to consolidate in the sun for 24 days. One column was provided with free drainage through a bed of sand while the other was allowed to lose water only by evaporation. Figure 10 shows the consolidation rate in the drained and undrained columns. One day after the columns were filled with sediment, a zone of clear water appeared above the settling muck in both columns. On the second day this water was siphoned off and analyzed ($NH_3$-N = 45 mg/ℓ and o-$PO_4$-P = 10 mg/ℓ). The dashed lines in Figure 10 show the immediate effect of the removal of the supernatant on the rate of consolidation and indicate that drainage of the sediments from the bottom provides for much better consolidation than does surface evaporation alone. The results of this experiment are summarized in Table 8 . As in the aquaria experiment, no significant loss in volatile solids occurred. In the drained column,

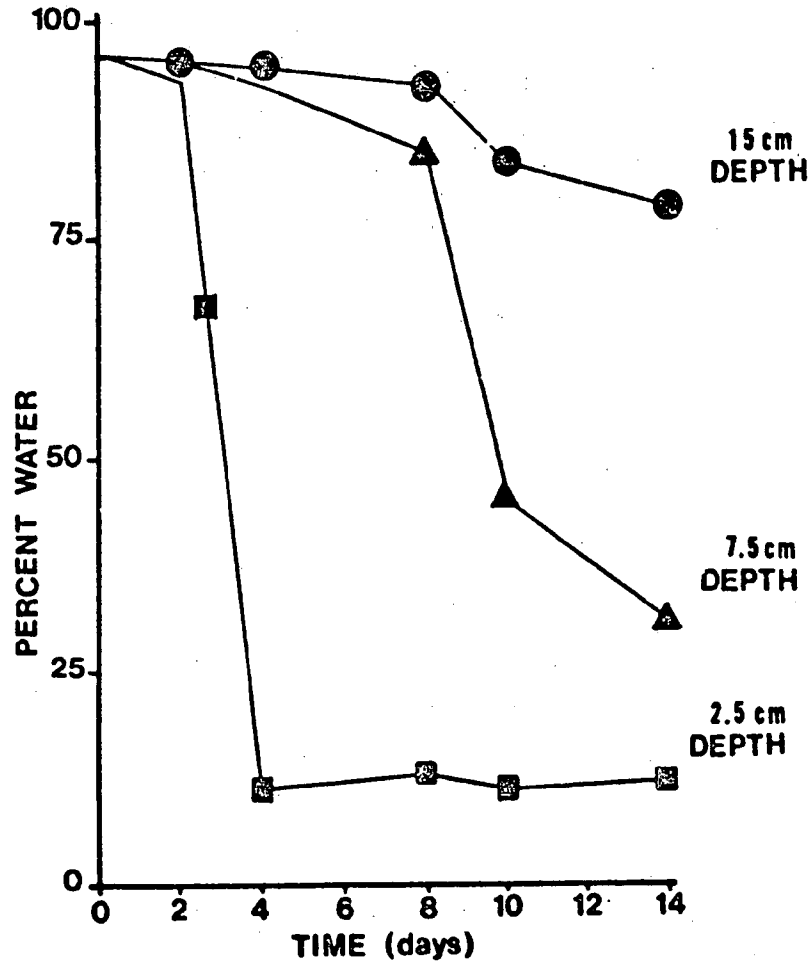


Figure 8. Effect of sun drying on moisture content of muck sediment (aquarium experiment)

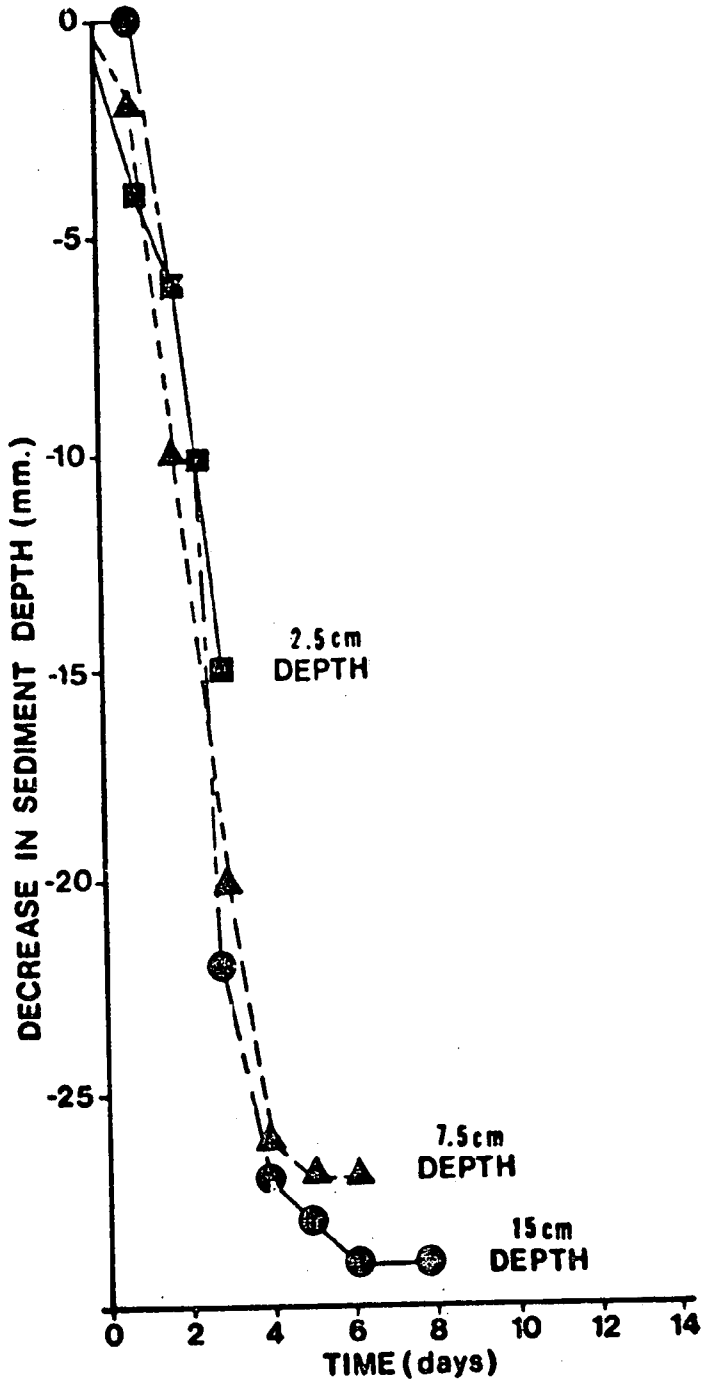


Figure 9. Consolidation of muck sediment upon sun drying (aquarium experiment)

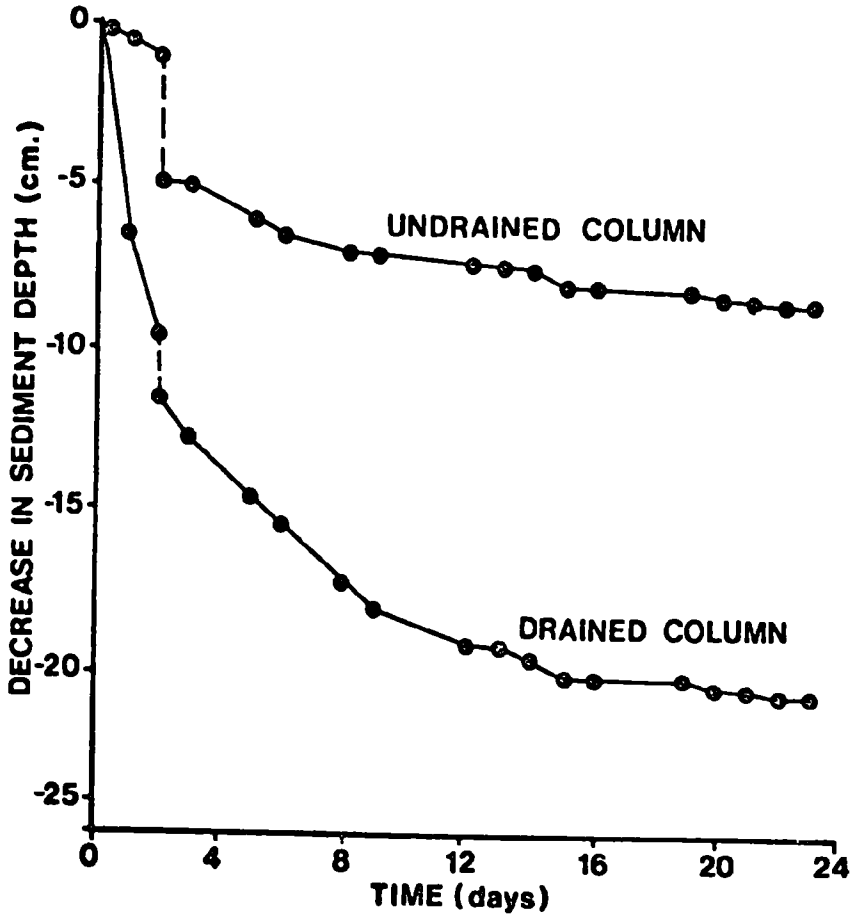


Figure 10. Consolidation of a drained and undrained column of muck sediment. Dashed lines indicate decrease in sediment depth caused by the removal of overlying water. Initial sediment depth was 76 cm.

TABLE 8. SUMMARY OF PHYSICAL CHARACTERISTICS OF MUCK IN COLUMN EXPERIMENT

| Date | | Drained Column % $H_2O$ | % Solids | % Volatile[a] |
|---|---|---|---|---|
| 5/31/72 | | 96.4 | 3.6 | 65.5 |
| 6/01/72 | | 88.5 | 11.5 | 66.8 |
| 6/02/72 | prior to $H_2O$ drawoff | 81.0 | 19.0 | 64.3 |
| 6/02/72 | after $H_2O$ drawoff | 71.3 | 28.7 | 65.8 |
| 6/06/72 | | 63.8 | 36.2 | 66.9 |
| 6/08/72 | | 60.4 | 39.6 | 67.1 |
| 6/12/72 | | 52.1 | 47.9 | 67.5 |
| 6/14/72 | | 45.8 | 54.1 | 66.3 |
| 6/22/72 | | 35.4 | 64.6 | 66.9 |

| Date | | Undrained Column % $H_2O$ | % Solids | % Volatile[a] |
|---|---|---|---|---|
| 5/31/72 | (initial | 96.4 | 3.6 | 65.5 |
| 6/01/72 | | 94.5 | 5.5 | 64.3 |
| 6/02/72 | prior to $H_2O$ drawoff | 93.6 | 6.4 | 62.5 |
| 6/02/72 | after $H_2O$ drawoff | 80.7 | 19.3 | 65.6 |
| 6/06/72 | | 80.6 | 19.4 | 66.4 |
| 6/08/72 | | 80.3 | 19.7 | 61.3 |
| 6/12/72 | | 79.5 | 20.5 | 66.5 |
| 6/14/72 | | 78.4 | 21.6 | 66.4 |
| 6/22/72 | | 76.5 | 23.5 | 66.8 |

a $\% \text{ volatile} = \frac{\text{dry wt} - \text{fired wt}}{\text{dry wt}} \times 100$

water content decreased from 96.4 percent to 35.4 percent after 24 days of drying. The minimum value for the undrained sediment was 76.5 percent water.

Lake Simulations (Columns and Tanks)

For initial lake simulations, 60 and 120 cm of muck sediment were placed in 19 plexiglas columns and 6 metal tanks. Three columns and one tank were not drawn down and hence served as controls. The sediment drying periods ranged from 69 days to 175 days for those which were subsequently refilled. Some sediment in tanks or columns developed leaks and remained dry for as long as 336 days.

Figure 11 shows the percent consolidation defined as loss in volume as a percentage of the original volume of the sediment. With a drying time of about 170 days, consolidation averaged 43 percent. The average consolidation for a 70 day drying time was 46 percent. Although there are higher and lower values, around 40 to 50 percent consolidation appears to be the point where an equilibrium is established between drying and rainfall. One tank was followed for 500 days following refill and maintained 30 percent consolidation throughout the period of observation. The significant point is that once sediment is consolidated, it remains in that state for a long period of time following refill. In Lake Apopka, such consolidation would increase the water depth in the areas with deep (e.g. > 2 m) layers of muck that dried well. The increased depth, as well as the more rigid structure of the consolidated sediments, would lessen the likelihood of sediment suspension caused by wave and outboard motor agitation.

Consolidation is due primarily to water loss. Figure 12 shows the average loss after 168 days of drying to be 33 percent. The dried sediments contained 25 percent less water 150 days after refill than did the control sediment. Apparently, the physical structure of the consolidated sediment inhibits reabsorption. Volatile solids (Figure 13) did not show significant changes, further indicating that the consolidation is due primarily to water loss.

During our sediment sampling off the northeastern shoreline of Lake Apopka, we found that an area of peat sediment extends from the shore some 100 meters out into the lake. The peat is apparently scoured by wave action and is not covered by muck sediment. Blocks 60 cm deep were placed in each of 3 plexiglas columns. The peat consists primarily of snarled fibrous plant roots, apparently remnants of previous rooted vegetation. This material does not consolidate upon drying. The 3 test columns were sun dried for an average of 180 days and the mean percentage consolidation was 7 percent. Two control columns of peat, each containing 60 cm, showed greater consolidation in spite of the fact that no drying occurred. The organic content (volatile solids) of individual peat samples varies between 80 and 90 percent of the dry weight. In this 3 column experiment, no change in the organic content of peat could be detected with drying times as long as 189 days. The water content of the peat also ranges between 80 and

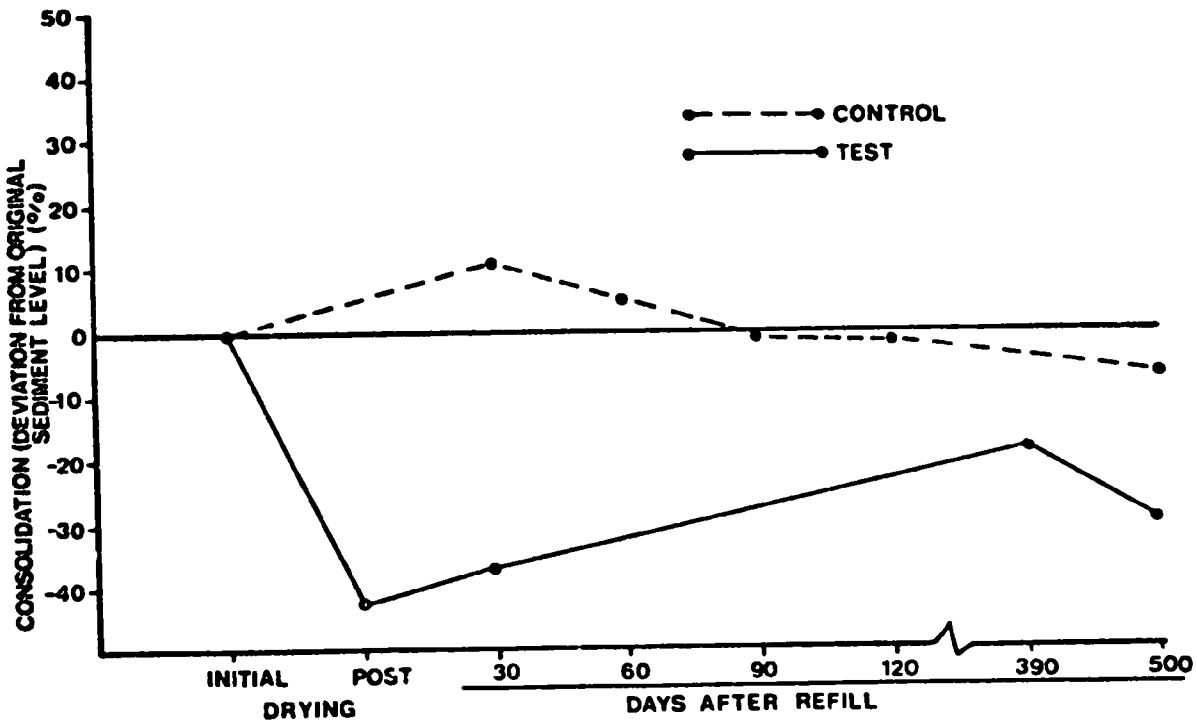


Figure 11. Consolidation of muck sediment (column-tank study)

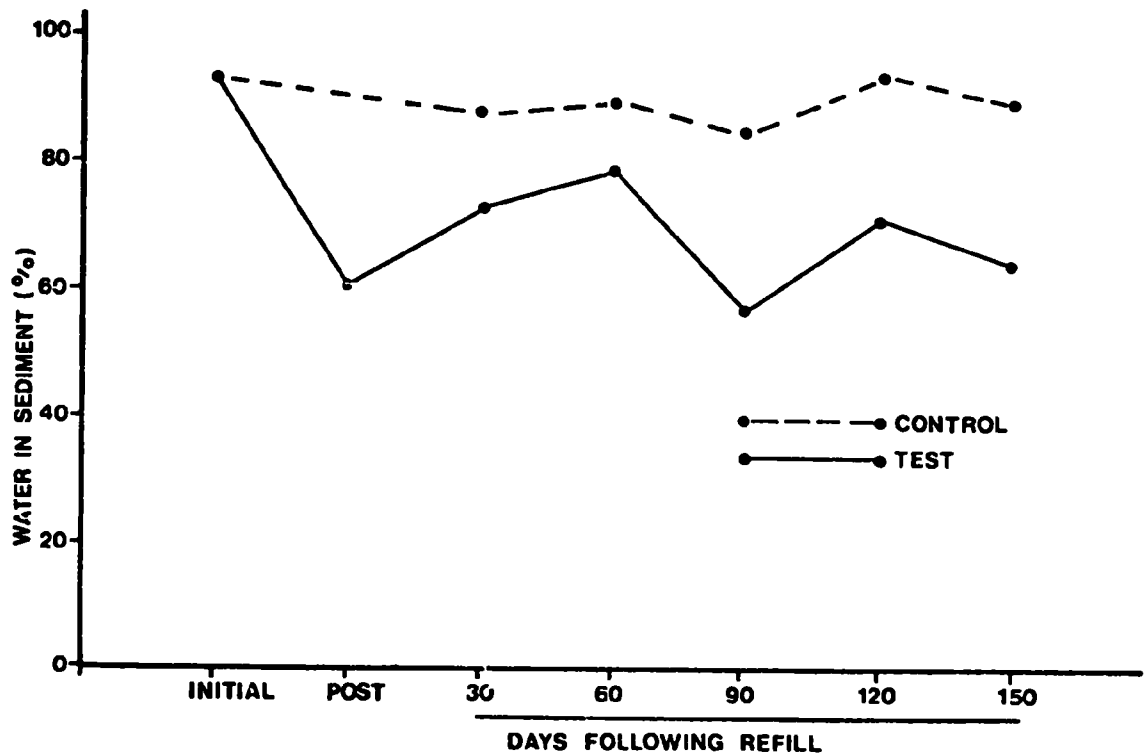


Figure 12. Percent water in muck sediment (column-tank study)

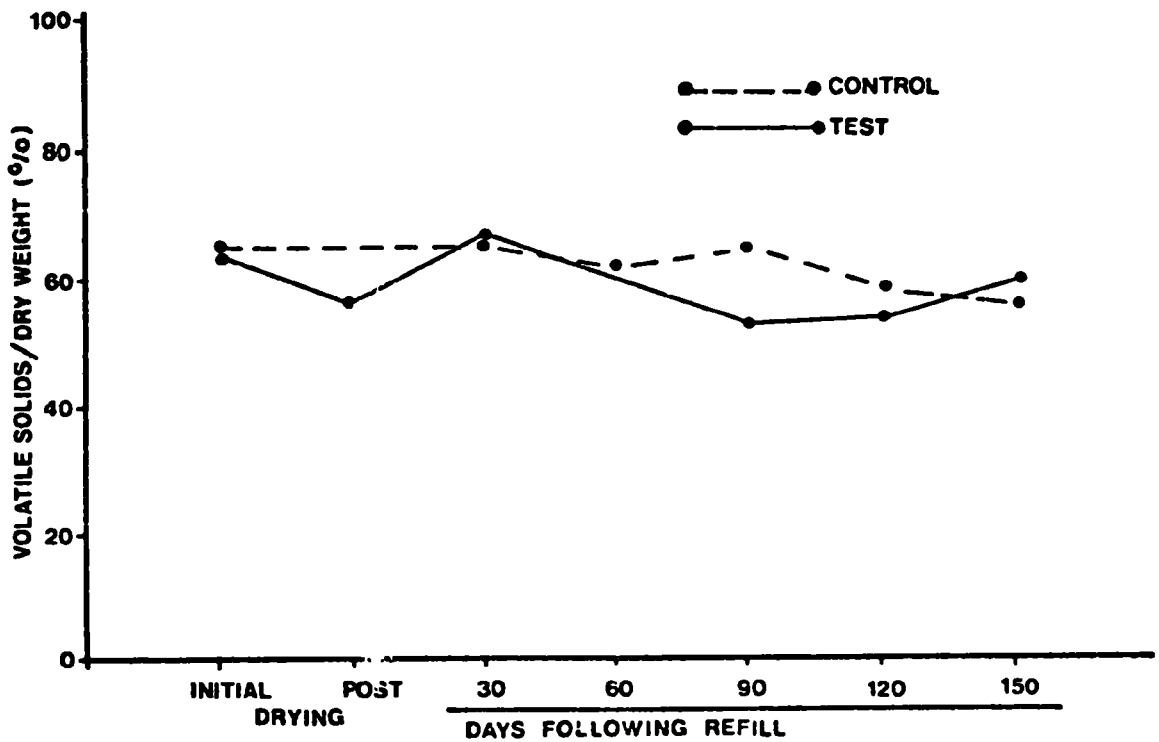


Figure 13. Volatile solids in muck sediment (column-tank study)

90 percent. The drying results showed great variation among samples. Apparently, peat dries better if it is loosely packed. It also reabsorbs moisture more rapidly if loosely packed. In areas of the lake where extensive peat deposits exist, there will be little increase in water depth resulting from drawdown and drying. Muck sediment on top of peat consolidates about the same (38 percent in 151 days) as muck alone. From our observations, this phenomenon does not occur commonly in the lake. The peat we observed was near shore and not covered by large amounts of muck sediment. During drawdown, muck which does cover the peat will probably move with the receding water into deeper areas of the lake. In order to deepen the lake, especially in the littoral zone, it may be necessary or at least desirable to remove some peat by physical means during drawdown.

Lake Simulations (Pools)

For the final lake simulation experiments, approximately 30 cm (1 ft) of muck sediment was placed in each of 5 plastic swimming pools 2.4 m (8 ft) in diameter and 45 cm (18 in) deep. Two pools ($\alpha$ and $\gamma$) were drawn down and dried for 4 months. The other 3 pools ($\beta$, $\Delta$, $\varepsilon$) served as controls. The major difference between the pool and the tank and column simulations was the fact that the water in all the pools was continually stirred by submersible water pumps. Additionally, the surface to volume ratio was greater in the pool experiments, thus affording greater wind mixing. Following refill of the test pools, some 17 parameters were followed for an additional four months. The physical parameters for the sediment included percentage water and volatile solids. Although sediment height (from which we calculated consolidation rates in the previous experiments) was followed, the data were unreliable because of sediment movement caused by forced circulation of the overlying water in the control pools. Changes occurring as a result of factors other than sediment shifting could not be detected.

Figures 14 and 15 show the water and volatile solids content of the sediment before, during and after drying. A two way analysis of variance (see appendix for details) was used to determine significant differences between the test and control pools. Although the lines on the figures represent means for all results obtained, the data were statistically analyzed for differences between test and control pools by grouping the data in three ways: (1) data from all 5 pools for the total time period: (2) data from 1 test pool ($\gamma$) and 2 control pools ($\beta$, $\Delta$) for the total time period and, (3) data from all 5 pools taken during the last two months of analysis. The second and third groups were used because leaks occurred in control $\varepsilon$ and in test $\alpha$. These sediments were immediately transferred to new pools but were disturbed in the process.

The water content of the sediment (Figure 14) shows a statistically significant difference ($P < 0.001$) due to drying when the data are grouped as in (2) and (3) above. The difference between test and control for all pools for the total time period was not significant ($P = 0.05$).

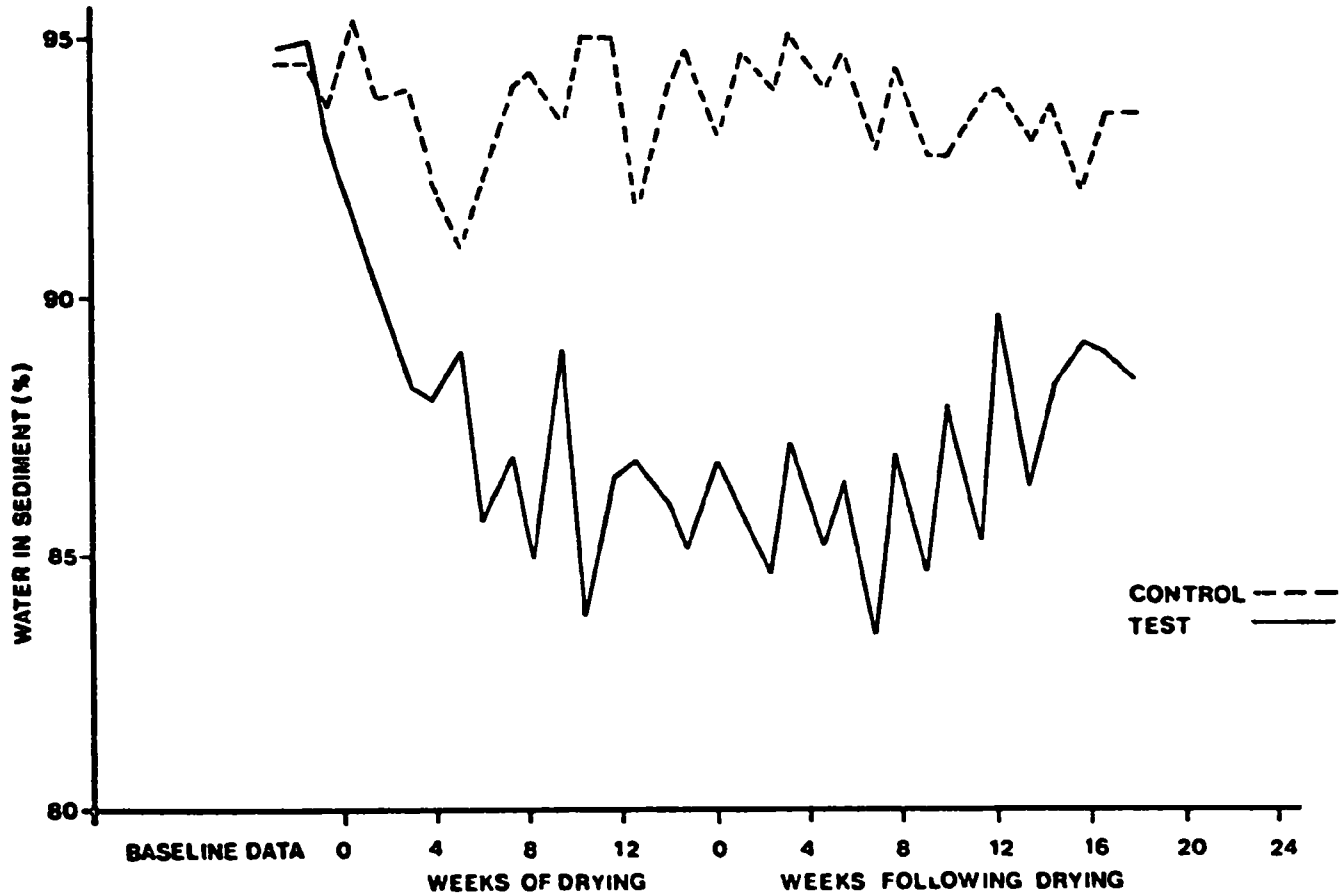


Figure 14. Percent water in muck sediment (pool study)

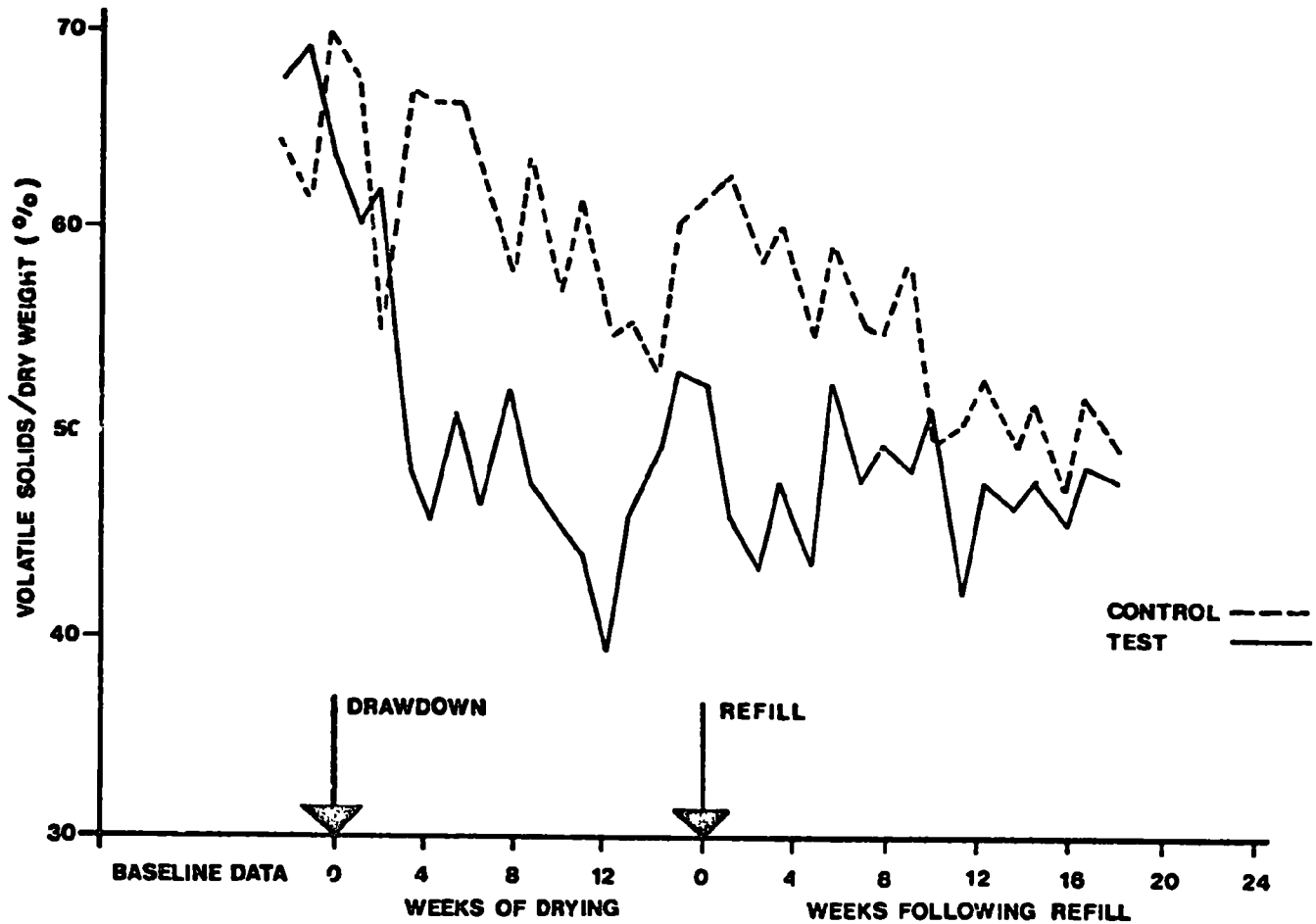


Figure 15. Volatile soldis in muck sediment (pool study)

Again, it appears that once drying takes place, the new structure of the sediment prevents reabsorption. In the pool experiments, drying reduced the water content from about 95 to 86 percent. In the tank and column experiments, the water content of the dried sediment was much lower, reaching a rough average of 65 percent. The primary reason for the difference is probably the method of drainage. The columns and tanks were underlain with sand and equipped with drainage valves, which were left open during the period of drawdown. The test pools, on the other hand, were pumped or siphoned following rainfalls. The latter method is obviously less effective and the drying sediments retained some rainfall. Furthermore, the columns and tanks were dried for over a month longer than were the pools. The experiment does show, however, that even under less than optimal drying conditions, significant water loss occurs and reabsorption is not important.

The effect of drying on the volatile solids content of the sediment was significant with data from 1 test pool and 2 controls are considered for the total time period. However, Figure 15 shows that volatile solids decreased in both test and control pools from around 65 percent to about 50 percent. The decline was more rapid in the test pools than the controls, but by the end of the study both sets were approaching the same volatile solids level. It should be noted that this is the only case we observed in which volatile solids showed a significant decrease as a result of drying. The rapid initial decrease upon drawdown (most of the decline occurred in the first two weeks of sediment exposure) suggests that part of the decline could be attributed to heterogeneous and non-representative samples. This suspicion is strenthened by the data in Table 2, which show there is a natural in-lake volatile solids range of 53.1 to 70.2 percent of muck sediment dry weight.

## BIOLOGICAL EFFECTS

### Aquatic Macrophytes

Nutrient additions and mismanagement of Lake Apopka have resulted in a phytoplankton-water hyacinth based ecosystem. Both types of primary producers die, decompose, and release nutrients which insure their survival. Both overwhelm their competitors, emergent and submersed macrophytes, by creating shade, turbidity, and a flocculent sediment. For drawdown to be effective in improving water quality, it will have to cause significant changes in the primary producer component of the Lake Apopka system. Submersed macrophytes would be the ideal replacement. They would provide a habitat for fish and periphyton, compete with the phytoplankton for nutrients, and reduce mixing velocities at the sediment-water interface. Their root structures would further aid in preventing sediment disruption.

According to a recent survey (Chesnut and Barman, 1974), the most common macrophyte on Lake Apopka is the hyacinth and no submersed plants

are present. The fourteen species observed by Chesnut and Barman are shown below in Table 9.

TABLE 9. AQUATIC VASCULAR PLANTS OF LAKE APOPKA (FROM CHESNUT AND BARMAN, 1974)

| | |
|---|---|
| Eichhornia crassipes | Panicum paludivagum |
| Alternanthera philoxeroides | Panicum hemitomon |
| Lemna sp. | Scripus validus |
| Nuphar advena | Typha domingensis |
| Pontederia lanceolata | Cladium jamaicensis |
| Sagittaria lancifolia | Hydrocotyle umbellata |
| Sagittaria latifolia | Jussiaea michauxiana |

We also observed a lack of submersed vegetation in Lake Apopka, and for that reason, investigated the sediments as a potential source for submersed macrophyte regrowth. Accordingly, in September, 1973, 17 sediment samples were taken by Ekman dredge at approximately equidistant intervals around the perimeter of the lake. Seeds present in each sample were separated by sieving and were identified. Table 10 summarizes that data. Several species of both emersed and submersed macrophytes were found, suggesting that the potential for regrowth does exist in the sediment as seeds.

In our own lake simulation studies, however, the only two aquatic macrophytes which sprouted on the drying sediment were cattail (Typha sp.) and water hyacinth (Eichornia crassipes). Table 11 shows a list of plant species which developed on dried sediments in our tanks and columns. For the tanks, all plants were harvested and weighed and the figures in Table 11 represent percent contribution of each species to total wet weight. The water hyacinth is not present in the list because of its short survival time on drying sediments. It appeared shortly after draining and lasted only a week or two. With the exception of Typha (cattail), all the species listed are terrestrial and apparently most germinated from seeds blown into the simulations. Eupatorium (dogfennel) and Typha (cattail) accounted for the majority of the biomass harvested from the tanks and columns.

The species list in Table 11 contains plants which grew on muck sediments exposed for about 7 months (non-refilled tanks and columns). Sediments dried for less time had fewer species growing. The first plant growth appears about 5 days after drying begins and consists of blue-green epipelic algae. After about 20 days, the algae is replaced

TABLE 10. SEEDS FOUND IN 17 SAMPLES OF SEDIMENT FROM LAKE APOPKA (SEPTEMBER 5, 1973)

| | | |
|---|---|---|
| 1. | Najas guadalupensis | Southern naiad |
| 2. | Sambucus canadensis | Elderberry |
| 3. | Potamogeton sp. | Pond weed |
| 4. | Brasenia schreberi | Watershield |
| 5. | Acnidia or Amaranthus sp. | Water hemp |
| 6. | Cladium jamaicensis | Sawgrass |
| 7. | Eichhornia crassipes | Water hyacinths |
| 8. | Cyperus sp. | Sedge |
| 9. | Scirpus sp. | Bulrush |
| 10. | Heteranthera sp. | Waterstar grass |
| 11. | Nymphaea sp. | Waterlily |
| 12. | Sagittaria lattifolia | Arrowhead |
| 13. | Myrica cerifera | Wax myrtle |
| 14. | Rumex cripus | Curly dock |
| 15. | Phytolacca americana | Pokeberry |
| 16. | Cicuta maculata or Conium maculatum | Water hemlock |
| 17. | Jussiaea leptocarpa | Water primrose |
| 18. | Anacharis sp. | Waterweed |
| 19. | Cephalanthus occidentalis | Button bush |
| 20. | Magnolia grandiflora | Magnolia |
| 21. | Proserpinaca palustris or Myriophyllum sp. | Mermaid weed Watermilfoil |
| 22. | Pontederia condata | Pickerelweed |
| 23. | Zostera marina | Eelgrass |
| 24. | Graminae | Unidentified grass |

TABLE 11. PLANTS COLONIZING DRIED SEDIMENT (EXPRESSED AS PERCENTAGE OF TOTAL DRY WEIGHT)

| Plant Species | Tanks | | Columns | | | | | |
|---|---|---|---|---|---|---|---|---|
| | 1 | 2 | 1 | 2 | 3 | 4 | 5 | 6 |
| *Amaranthus autralis* | 2% | | | | | | | |
| *Baccharus halimifolia* | .4% | .2% | | | | | | |
| *Celtis laevigata* | .01% | .6% | | | X[a] | | | |
| *Cocculus carolinus* | | .1% | | | | | | |
| *Conyza bonariensis* | | X[b] | | | | | | |
| *Conyza canadensis* | | X[b] | | | | | | |
| *Conyza parva* | | | | X | X | | | |
| *Cynodon dactylon* | | | | | | X | | |
| *Echinochloa walteri* | | | X | X | | | | |
| *Eupatorium capillifolium* | | | | | | | X | X |
| *Eupatorium leptophyllum* | 86% | 94%[b] | | | | | | |
| *Phytolacca rigida* | | 5% | | | | | | |
| *Sambuccus simpsonii* | | | | X | | | | |
| *Solidago* sp. | | | | X | | | | |
| *Typha latifolia* | 11% | | | | | | X | X |
| *Urtica urens* | <.01% | | | | | | | |
| Unidentified | <.01% | | | X | | X | | |

a X indicates presence only for columns
b *Eupatorium leptophyllum*, *Conyza canadensis* and *Conyza bonariensis* were weighed together

by moss and actinomycetes. These forms are slowly replaced by the vascular plants, which do not appear until about 60 days after drying starts. The first vascular plant to appear was Typha (cattail) on peat sediment. About 2 weeks later Typha and Eichhornia (hyacinths) sprouted in the muck sediment.

The only plant which survived refill in the tank and column experiment was Typha. Following refill, however, Chara braunii (stonewort), a macroscopic green benthic alga, grew on the sediments of 1 tank (69 day drying period) and 3 columns (175 day drying period). The Chara remained viable in the tank for over a year, at which time the tanks and columns were disassembled. Transplantation of the Chara into other columns or aquariums is easily accomplished by sprigging.

The same general pattern of plant succession occurred in the pool experiment. The biomass of the plants harvested from the test pools following 4 months of refill and the control pools are shown in Table 12. One test pool supported equal growth of Typha and Chara while the other contained primarily Typha. Over 2 kg of Chara was also harvested from one of the control pools. This was surprising, for Chara has not been reported as growing in Lake Apopka. Apparently, the laboratory simulations favor the growth of Chara, and we cannot state positively that drying is the sole factor responsible for its growth. However, the control pool in which Chara grew was the pool which sprung a leak and partially dried before the sediments could be transferred to a new pool. Perhaps the partial drying stimulated the growth of Chara.

TABLE 12. PLANT BIOMASS HARVESTED FROM POOL SIMULATIONS. TEST POOLS HARVESTED FOLLOWING DRAWDOWN AND FOUR MONTHS OF REFILL

| Pool | Plant Harvest (kg) | | |
|---|---|---|---|
| | Chara | Typha | Other |
| Test (α) | 6.97 | 6.52 | 0.20 |
| Test (γ) | 0.61 | 16.9 | 0.42 |
| Control (β) | 0 | 0 | 0 |
| Control (Δ) | 0 | 0 | 0 |
| Control (ε) | 2.59 | 0 | 0 |

One impediment to the establishment of Chara and Typha in Lake Apopka is the water hyacinth. According to Chesnut and Barman (1974), as well as our field observations, mats of hyacinths can flatten and destroy stands of emergent littoral vegetation. The mats also shade submersed vegetation. A continuing hyacinth removal program (preferrably mechanical) may be necessary to prevent the destruction of cattail stands which will become established during and following drawdown. Protective fencing measures will probably be ineffective due to the large hyacinth biomass and wind forces involved. The cattail would occur in the shallow water areas of the lake (<1.3 m). Chara, should it become established as a result of drawdown, might grow in deeper water. There, it would less likely be adversely affected by hyacinths, which tend to blow into shore. When hyacinths are in deeper water, we have noticed they occur as small clumps, which would not significantly shade submersed plants. The establishment of Chara, however, is not certain. Two significant factors affecting its growth are water turbidity and calcium content. If turbidity can be kept low during refill its growth seems likely. The calcium content of Gourd Neck Springs water (the primary means by which the lake will be refilled) is moderately hard (Ca> 25 ppm), which should be adequate for Chara growth.

Benthic Invertebrates

The Lake Apopka benthic invertebrates sampled by the Florida State Board of Health during their 1962 to 1964 survey (undated report) indicated oxygen stress. The predominant forms were sludgeworms (Oligochaeta), bloodworms (Chironomidae) and phantom midges (order Diptera, family Culicidae). These three forms were found consistently in the muck sediment. At several muck bottom stations, no invertebrates were found. Both sludgeworms and bloodworms contain erthyrocruorin, a hemoglobin-like pigment which allows them to live at low levels of dissolved oxygen for extended periods of time. When they are the only forms found (low diversity), oxygen stress is usually the reason. The phantom midges are capable of vertical migration and hence are considered poor indicators of water quality.

Other benthic invertebrates found in the early 1960's included mayflies (class Insecta, order Ephemeroptera), scuds (class Crustacea, order Amphipoda), and clams and snails (phylum Mollusca, Classes Pelycypoda and Gastropoda). None were found frequently and they usually were not present in muck sediment. These organisms, taken as a group, indicate adequate dissolved oxygen.

In 1970 and 1971, Florida Technological University conducted a survey of the benthic invertebrates of the Oklawaha chain. Their raw data is included as Appendices B-1 to B-4 in the Florida Department of Pollution Control's Restoration Plan (1972). The Apopka survey consisted of a one year seasonal sampling of about 30 stations. Again, the predominant forms were bloodworms, sludgeworms and phantom midges. A few leeches (phylum Annelida, class Hirudinea), which also can tolerate low dissolved oxygen, were found. Scuds and snails were occasionally

observed. A total of 2 mayflies, the least pollution tolerant form found, were recovered from the 30 stations for the 4 sampling periods.

Samples of flocculent muck and peat from the northeastern section of Lake Apopka taken for simulations in this study contained no benthic invertebrates. The only signs of life were shells of long dead snails (mostly Viviparus sp.), which were abundant in muck and peat and probably predate the 1947 hurricane. In order to minimize sediment perturbation, we did not monitor benthic invertebrates during the course of the experiment. We did observe, however, that prior to drawdown, the plexiglas columns and tanks were rapidly colonized by bloodworms. Possibly in our initial examination of the sediments for invertebrates, eggs or very early instar larvae passed through the U.S. Standard No. 30 sieve (595 micron mesh) and were therefore missed. It is more likely that adult female chironomids laid eggs on the surface of the tank and column water covering the sediments. The simulations were near a sewage treatment plant oxidation pond, where large numbers of chironomids thrive.

About 3 months after the control simulations were constructed, the trophic structure shifted from a phytoplankton dominated system to one in which comparatively large biomasses of zooplankton were supported by epipelic algae. This shift was probably due to the cessation of mixing, which, in the lake, keeps the phytoplankton supplied with nutrients by accelerating sediment-water exchange. The same phenomenon occurred in the test simulation following refill. The green filamentous alga, Oedogonium, was the predominant epipelic growth form. In one control tank, this alga covered the bottom to a depth of several centimeters and supported a community of diving beetles (order Coleoptera, family Dytiscidae), dragonfly nymphs (order Odonata), and an occasional water scorpion (order Hemiptera, family Nepidae, genus Ranatra). These unusual biotic conditions, which we felt were the result of the lack of mixing, were responsible for the disassembly of the tank and column experiments and the assembly of the mixed pool simulation.

The lake sediment placed in the pool simulations also contained no living benthic invertebrates at the start of the experiment. Following drying and the 4 month refill period, the benthic invertebrates were recovered from the test and control pools. Recovery was by dip net, sieve and siphon. The results are shown in Table 13. Although the figures in Table 13 are not on a quantitative areal basis, they do represent the results of equal sampling effort per pool and hence can validly be used to compare test versus control benthic invertebrate populations. The diversity indices presented in Table 13 are Shannon-Weaver (base 2) family diversity indices.

In interpreting Table 13, probably the most significant point is that diversity in the test pools was higher than in the controls. Since the number of replicates was so small, no statistical analysis was possible. The diversity of control $\varepsilon$ (1.94), however, is probably not significantly different from that of test pool $\gamma$ (2.14). Control

TABLE 13. BENTHIC INVERTEBRATES POPULATING POOL SIMULATIONS FOLLOWING DRAWDOWN AND REFILL. NUMBER EXPRESSED ON THE BASIS OF EQUAL SAMPLING EFFORT PER POOL

| | Test Pools | | Control Pools | | |
|---|---|---|---|---|---|
| | α | γ | β | Δ | ε |
| Phylum Annelida | | | | | |
| Family Tubificidae | | | | | 1 |
| Phylum Arthropoda | | | | | |
| Class Insecta | | | | | |
| Order Diptera | | | | | |
| Family Chironomidae | 11 | | 15 | 41 | 8 |
| Order Ephemeroptera | | | | | |
| Family Caenidae | 1 | 1 | | 2 | |
| Family Leptophlebiidae | 1 | 2 | 12 | 3 | 7 |
| Order Hemiptera | | | | | |
| Family Notonectidae | 15 | 1 | 3 | 6 | 2 |
| Family Nepidae | 3 | 1 | | | |
| Order Odonata | | | | | |
| Family Coenagrionidae | 22 | 8 | | | |
| Family Libellulidae | 16 | 13 | 4 | | 2 |
| Phylum Chordata | | | | | |
| Class Amphibia (unidentified tadpoles) | 11 | 3 | | | |
| Total | 80 | 29 | 34 | 52 | 20 |
| Number of Families | 8 | 7 | 4 | 4 | 4 |
| Diversity Index | 2.55 | 2.14 | 1.72 | 1.05 | 1.94 |

pool ε, as explained earlier, developed a leak and the sediments had to be transferred to a new pool. As a result of the leak and water loss during transfer, the sediment in ε underwent some drying. In reality, therefore, control pool ε might be considered intermediate between a test and a control pool. This hypothesis is strengthened by the fact that Chara grew only in control pool ε and the two test pools, α and γ. No macroscopic plants grew in either of the other two control pools (β and Δ).

The presence of the emersed and submersed plants provided a habitat suitable for a greater diversity of benthic invertebrates in the test pools. The predominant organisms in all control pools were bloodworms (Chironomidae). In the test pools, dragonfly nymphs were the most numerous forms, with air breathing hemipterans a close second in test α. Since the pool simulations were located in Gainesville, the benthic fauna which develops following drawdown and refill in Lake Apopka might be slightly different. If submersed aquatic plants, either an alga such as Chara or vascular plants, develop, however, an increase in benthic invertebrate diversity and biomass can be expected. Although we did not measure biomass, the combined weight of the tadpoles in the test pools easily surpassed the combined biomass of the invertebrates recovered from the control pools. The shift in primary producers from phytoplankton to macroscopic plants should be evident all the way up the food chain. Fish should respond positively to the increased bottom stability, the increased abundance of their invertebrate food source, and the fingerling habitat provided by the plants.

## CHEMICAL EFFECTS

Effect of sediment drying on major nutrient forms and general chemical characteristics of the sediments appear to be minimal. Although variations occurred both temporally in a given simulation and between dried and undried sediments, they were generally no greater than those attributable to sampling variations resulting from sediment heterogeneity. Figures 16 shows measurement of total phosphorus and total nitrogen in the column and tank simulations. The apparent reduction in phosphorus in the dried sediments is not so significant as it first appears in Figure 16. Phosphorus levels within a pool were found to vary by about ± 20 percent areally. The areal variation in total nitrogen was about ± 21 percent, making the reduction below control levels 150 days after refill barely significant. Free ammonia was highly variable in the column and tank sediments (Figure 17) and represented only about one percent of the total nitrogen. Free ammonia levels were always less than 0.4 mg/g dry wt. whereas TKN averaged 30 mg/g dry wt. of sediment. Ammonia in the undried sediment (control) appeared to rise slowly during the period of study. The drying (test) sediments showed higher ammonia levels during most of the drying period. At the end of the drawdown period, however, the ammonia concentration of the dried sediments was below that of the control sediments.

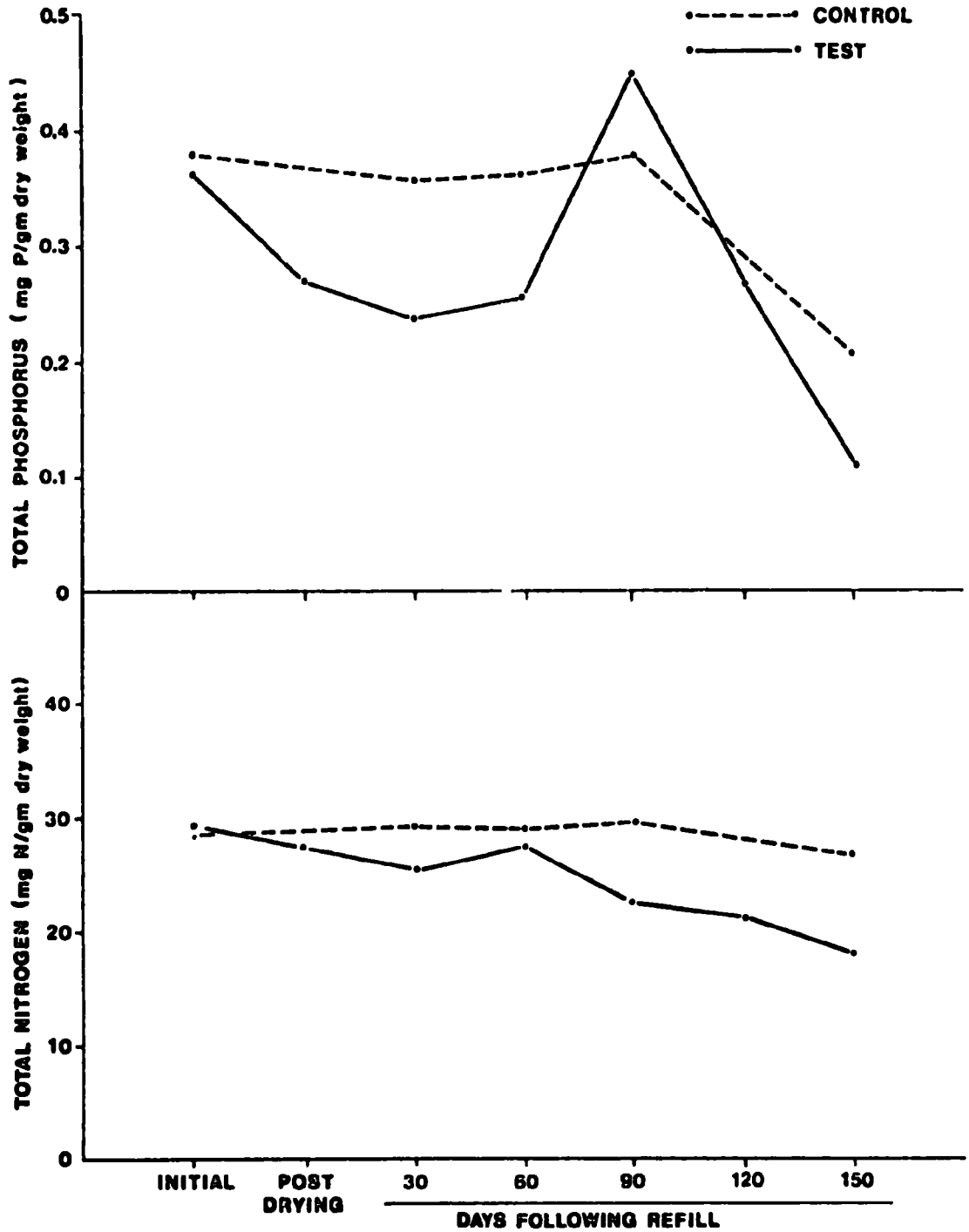


Figure 16. Total phosphorus and nitrogen changes in column and tank sediments

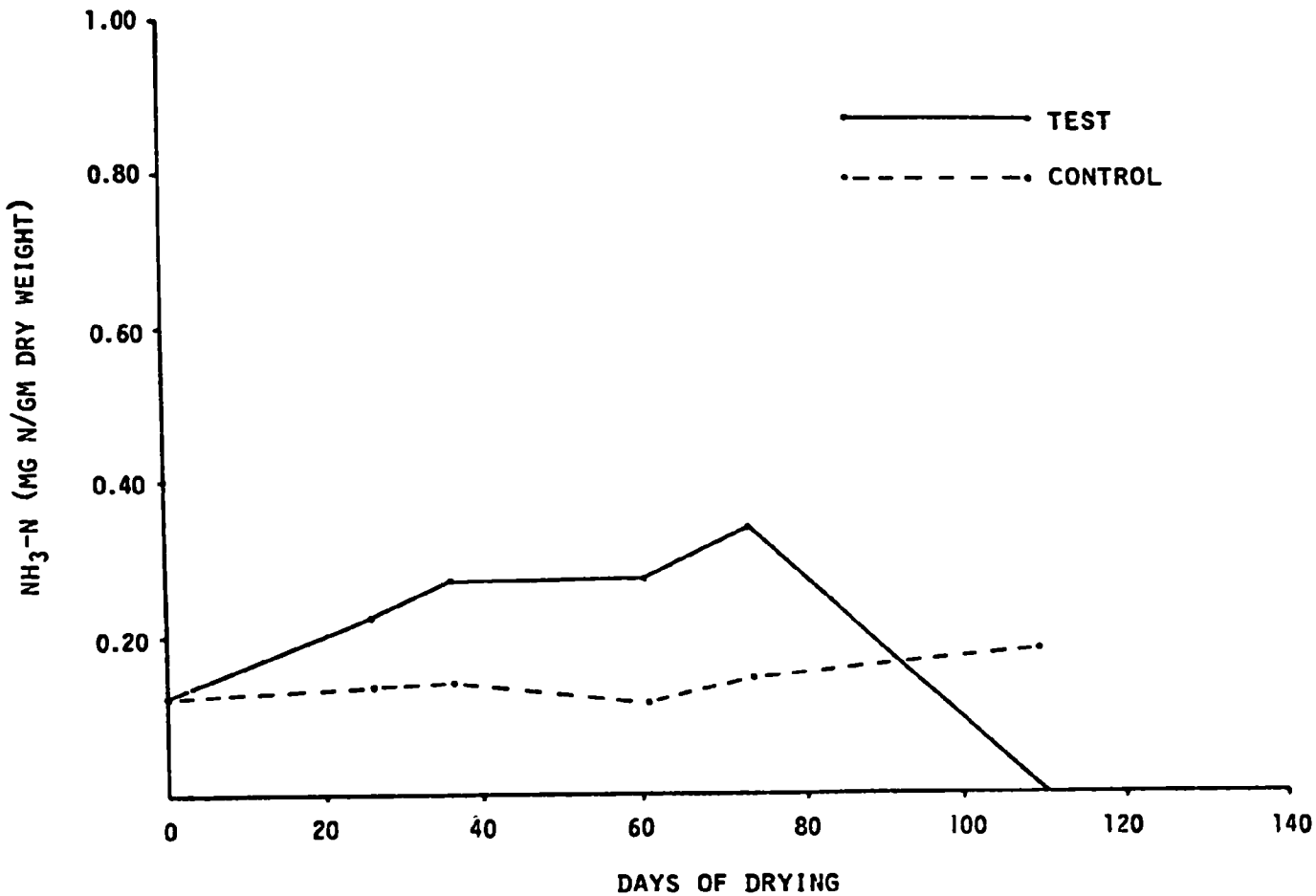


Figure 17. Changes in ammonia nitrogen in column and tank sediments

Table 14 shows results of sediment cores taken from a control and dried sediment tank. No significant pattern of stratification in TON and total P can be seen for either series of cores. Data for the dried cores gives further weight to the conclusion that drying results primarily in physical changes. The average TON increased about 15 percent in the dried sediment core, but total P stayed about the same. If substantial oxidation had occurred, the phosphorus and perhaps the nitrogen would act as conservative substances and their concentrations on a dry weight basis would have increased.

Sediments from control and test tanks have also been examined to determine the forms (speciation) of nitrogen and phosphorus. Fractionation of sediment phosphorus was done according to a modified Chang and Jackson procedure for soil analysis (Peterson and Corey, 1966). To avoid changes in sediment characteristics the studies were done on fresh wet and dried sediment rather than oven-dried sediment. Interstitial water was removed by centrifugation. Nutrient analyses were performed on the supernatant, and nutrient fractionation tests were done on the centrifuged sediment.

In general the fractionation results show little qualitative difference between dried and undried sediments with respect to nitrogen and phosphorus forms. Preliminary experiments with fresh Lake Apopka sediment indicated the presence of phosphorus in only one fraction; analyses on surface samples from three tanks (B, F and J) likewise indicated inorganic phosphorus occurred in the reductant soluble fraction. No phosphorus was found in the "easily soluble" phosphate, aluminum phosphate, iron phosphate, calcium phosphate, or occluded phosphate fractions. Thus inorganic phosphorus in Lake Apopka sediment is apparently bound entirely in the reductant soluble form (mainly ferric phosphate coated with ferric hydroxide). Organic phosphorus ranged from 0.12 to 0.37 mg/g in the three sediments, or from 32 to 56 percent of the total P. Results for the pool sediments indicated the same simple P speciation both before and after drying. Table 15 presents results of metal analyses on sediments used in the pool simulations. Results verify the low marl character of the sediment; calcium content was 5 percent or less and some of this was undoubtedly tied up in snail shells. The iron levels are moderately high and most of the inorganic P was found to be sorbed to the iron hydroxides.

Sediment interstitial water levels of ammonia and orthophosphate were measured on three occasions in the pool simulations; shortly after they were constructed and filled, at the end of the drying period and three months after refill. Interstitial water was separated from the sediment by centrifugation and nutrient analyses were performed on the supernatant using the Auto Analyzer techniques described in the appendix. Results are summarized in Table 16. Pore water of the Lake Apopka sediments in the pool study initially contained 0.1 to 2.6 mg/l of orthophosphate-P and from 0.3 to 1.5 mg/l of ammonia-N. This material would be readily transported to the overlying water when the sediments are suspended by wind driven currents and wave action. The high initial variability among the pools probably reflects distur-

TABLE 14. NITROGEN AND PHOSPHORUS DISTRIBUTION IN SEDIMENT CORES FROM TANK EXPERIMENT

| Depth Cm | Control (D) Water % | TON mg/g-dry wt | Total P mg/g-dry wt | Dried (B) Water % | TON mg/g-dry wt | Total P mg/g-dry wt |
|---|---|---|---|---|---|---|
| | | | August 11, 1972 (Predrawdown) | | | |
| Surface | 97 | 32 | 0.46 | 697 | 29 | 0.31 |
| 5 | 98 | 32 | 0.55 | 97 | 26 | 0.35 |
| 10 | 95 | 27 | 0.56 | 93 | 25 | 0.37 |
| 15 | 91 | 27 | 0.43 | 92 | 29 | 0.26 |
| 20 | 90 | 36 | 0.46 | 92 | 23 | 0.28 |
| 50 | 88 | 35 | 0.50 | 87 | 26 | 0.30 |
| Ave. | 93 | 36 | 0.49 | 93 | 26 | 0 3· |
| | | | September 18, 1972 | | | |
| Surface | | | | 75 | 30 | 0.38 |
| 5 | | | | 74 | 29 | 0.35 |
| 10 | | | | 76 | 24 | 0.31 |
| 15 | | | | 79 | 29 | 0.30 |
| 20 | | | | 80 | 29 | 0.32 |
| 25 | | | | 80 | 30 | 0.36 |
| Ave. | | | | 77 | 28 | 0.33 |
| | | | October 20, 1972 | | | |
| Surface | 97 | 29 | 0.57 | 59 | 26 | 0.35 |
| 5 | 95 | 35 | 0.43 | 65 | 32 | 0.25 |
| 10 | 92 | 32 | 0.51 | 74 | 32 | 0.30 |
| 15 | 90 | 33 | 0.46 | 73 | 30 | 0.36 |
| 20 | 85 | 31 | 0.50 | | | |
| 50 | 83 | 30 | 0.49 | — | — | — |
| Ave | 90 | 32 | 0.49 | 68 | 30 | 0.32 |

TABLE 15. METAL ANALYSES ON SEDIMENTS FROM LAKE APOPKA POOL SIMULATIONS

| Pool | Mg mg/g | Mn μg/g | Fe mg/g | Ca mg/g |
|---|---|---|---|---|
| Test α | 4.18 | 3 | .13 | 19.5 |
| Control β | 6.63 | 4 | .25 | 33.2 |
| Test γ | 5.10 | 5 | 1.08 | 51.0 |
| Control Δ | 3.92 | 4 | .30 | 27.4 |
| Control ε | 4.77 | 4 | .25 | 45.3 |

TABLE 16. INTERSTITIAL WATER NUTRIENT CONTENT OF APOPKA POOL SIMULATIONS[a]

| Pool | Beginning | | Middle | | End of Study | |
|---|---|---|---|---|---|---|
| | $NH_3$ | OP | $NH_3$ | OP | $NH_3$ | OP |
| Test α | 0.28 | 0.14 | .21 | .13 | 6.4 | .15 |
| Control β | 1.54 | 2.6 | .20 | .60 | 7.3 | .63 |
| Test γ | 0.95 | 2.2 | .20 | .41 | 5.1 | .20 |
| Control Δ | 1.33 | 0.64 | .20 | .20 | 7.8 | .20 |
| Control ε | 0.84 | 0.94 | .30 | .16 | 9.0 | .97 |

[a] Results in mg/l of pore water

barces resulting from the collection, transport and addition of the sediments to the pools and implies that sediment/interstitial water equilibrium had not been achieved.[a] Both ammonia and orthophosphate were consistently lower in all pools by mid-January (just prior to refill of pools α and γ). By the end of the experiment ammonia levels had increased somewhat in all the waters while orthophosphate changed deviated little from predrawdown levels. The final orthophosphate and ammonia levels for the dried sediment pore water were somewhat lower than the controls, but considering the variability among the pools the significance of these differences is questionable. Although the results in Table 16 indicate slight if any differences between dried and undried sediments with respect to nutrient levels in pore water, it should be noted that dried sediments have a lower water content and are less likely to be suspended by turbulence than the undried sediments. The potential contribution of nutrients from pore water of dried sediments to the overlying lake water thus should be lower than that from undried sediments.

A further question that arises concerning the effectiveness of sediment drying is the extent of nutrient leachability from dried and undried sediments. A priori arguments can be made in either of two directions. If substantial oxidation occurs as a result of sediment dessication, then organic N and P forms should be converted to soluble inorganic forms, and reduced N should be oxidized to nitrate, which is readily leachable. On the other hand, oxygenated sediments are generally considered less susceptible to nutrient release than are reduced sediments. Formation of an oxidized crust at the sediment-water interface is generally thought to prevent or inhibit nutrient release (e.g. Mortimer 1941-42). Such crusts were observed on the dried sediments of the experimental pools. The tank and pool experiments can be considered as large scale experiments and differences in trophic conditions between control and dried systems provide an indication of differences in nutrient leaching from dried and undried sediment. However, interpretation of the tank and pool data directly in terms of N and P leaching rates is confounded by many factors.

In order to obtain direct data on the sorptive-leaching properties of dried and undried sediment, several short term laboratory studies were undertaken. These included measurements of leachable and exchangeable orthophosphate and ammonia. The former is defined as the amount of orthophosphate or ammonia desorbed from sediment by mixing with demineralized water, and the latter represents the amounts of the nutrients desorbed (or exchanged) using 1 N $NH_4Cl$ (for ortho P) or 2 N KCl (for $NH_3$) as the leaching solution. Table 17 presents a summary of an extensive synoptic survey undertaken on Lake Apopka sediments. Twenty-five sediment samples were collected from stations scattered throughout the lake. A portion of each was centrifuged to measure interstial

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a Some of the variation also results from differing water content (i.e. ml interstitial water/g sediment) present initially.

water content of $NH_3$ and ortho P, and another portion was dried under a heat lamp to a degree simulating natural drying for several months. Leaching and exchange measurements were made on dried and undried aliquots of the sediment. The results (Table 17) indicate an increase in leachable and exchangeable ammonia. Thus drying would seem to create the potential from greater phosphate release. However, this result must be interpreted cautiously with respect to the effect of drawdown on the lake proper. The leaching experiments were conducted under drastic mixing conditions, whereas the net effect of drawdown drying will be to decrease sediment suspension by creating a stable sediment structure. The lack of an increase in leachable $NH_3$ as a result of drying perhaps resulted from some ammonia volatilization during drying.

Table 18 presents results of several sets of leaching experiments done on sediments from the pool simulations before and after drawdown and refill. Before drying all pools had a rather uniform ammonia leaching rate (0.35-0.40 mg/g). The somewhat increased $NH_3$ leachability in all sediments after the drying period probably reflects disturbance of the sediments in setting up the pools (hence depressed initial values). The post drying results indicate lower $NH_3$ leaching from the dried sediments (means = 0.47 and 0.57 mg/g) for the two dates compared to the controls (means = 0.66 and 0.94 mg/g, respectively). However, the range of control values indicates that the differences may be less than those suggested by the means. For phosphate the initial leaching rates were more variable (0.02 to 0.06 mg/g) than for $NH_3$. After drying leachable ortho P was lower in all pools, again probably an artifact of initial sediment disturbance. Just after refill the leachable ortho P in the dried pools was slightly less than in the controls. A further decline in leachable ortho P was noted in four of the pools at the end of the study (about 3 months later). However, pool $\gamma$ (a drawdown pool) showed a small increase, and as a result, had a considerably higher rate of ortho P leaching than did the other dried sediment ($\alpha$), which had a rate comparable to the controls. Reasons for the discrepancy in pool $\gamma$ are not apparent; perhaps they reflect differences in the extent of drying.

In general, results of these leaching test and other studies not described here indicate that lake drawdown and moderate sediment desiccation leads to only slight changes in nutrient leachability. In some cases dried sediments leached less material and in other cases more, but overall, drying does not seem to influence the chemical factors affecting nutrient leachability. The principal effects of drying then relate to physical consolidation.

TABLE 17. SUMMARY OF LAKE APOPKA SEDIMENT SYNOPTIC STUDY[a]

| | OP | | $NH_3$ | |
|---|---|---|---|---|
| | Fresh | Dried | Fresh | Dried |
| Interstitial Water Content | 3.09 | --- | 9.6 | --- |
| Leached with Demineralized Water | 0.106 | 0.315 | 0.88 | 0.82 |
| Exchangeable N or P[b] | 0.24 | 0.42 | 1.58 | 1.47 |

a Results are mean values (in mg N or P/l) for 25 samples collected around Lake Apopka

b Amount of nutrient leached into 1N $NH_4Cl$ (for phosphate or 2N KCl (for ammonia)

TABLE 13. NUTRIENT LEACHABILITY OF APOPKA POOL SIMULATION SEDIMENTS[a]

| Pool | Before Drying | | After Refill | | End of Study | |
|---|---|---|---|---|---|---|
| | $NH_3$ | OP | $NH_3$ | OP | $NH_3$ | OP |
| Test α | 0.40 | 0.02 | .45 | 0.007 | 0.49 | .003 |
| Control β | 0.39 | 0.06 | .65 | 0.015 | 1.20 | .002 |
| Test γ | 0.37 | 0.06 | .50 | 0.010 | 0.66 | .015 |
| Control Δ | 0.38 | 0.05 | .95 | 0.015 | 0.96 | .004 |
| Control ε | 0.35 | 0.05 | .45 | 0.010 | 0.66 | .003 |

a Results in mg N or P/g dry weight sediment

Experiments conducted by placing ∿0.5g (wet wt) of centrifuged sediment (to remove interstitial water) into 10 ml deionized water and mixing for one hour.

## SECTION VII

## EFFECTS OF DRAWDOWN ON OVERLYING WATER QUALITY

### PHYSICAL - CHEMICAL EFFECTS

For the initial lake simulations, muck sediment (60 and 120 cm deep) was placed in 19 plexiglas columns and 6 metal tanks. Three columns and one tank served as controls. Following a drying time ranging from 69 to 175 days (dried a mean of 117 days), 3 tanks and 10 columns were refilled with Gainesville ground water, which is chemically similar to Gourd Neck Spring water. A variety of routine water quality measurements were made on the tanks and columns at weekly to monthly intervals. Measured parameters included physical conditions, such as temperature, turbidity and conductance, and chemical conditions, such as dissolved oxygen, pH, major ions and all N and P forms. Of primary interest with respect to the assessment of drawdown feasibility as a lake renewal technique are the levels of nutrient forms and related parameters in the water following refill of the test simulations.

Orthophosphate levels in the waters of the experimental columns and tanks (Figure 18) exceeded control levels by relatively small amounts (0.01 - 0.03 mg P/l) for the first 90 days after refill, but after 150 days of refill, water above the dried sediment became highly enriched (average 0.131 mg P/l), while the controls remained low (average 0.03 mg/l). Reasons for this increase are not entirely clear, but as discussed later, the lack of adequate mixing in the tanks led to dense growths of filamentous algae on the sediment of some of the test tanks, and the conditions in the tanks after the onset of these growths are not considered representative of <u>in situ</u> lake conditions.

Nitrate (Figure 19) in the refill water, on the other hand, remained lower in the test than in the control simulations, even after 180 days of refill. Total nitrogen (Figure 20) was also slightly lower in the test tanks just after refill, and later (after 90 days of refill) the control tank water increased markedly in total N compared to the test tanks. Total N levels in both test and control tanks remained above 1.5 mg/l, indicative of eutrophic conditions. A pulse of ammonia nitrogen to almost 0.7 mg N/l (Figure 21) occurred following refill of the of the test tanks, but the level dropped rapidly and ammonia remained depleted after 60 days of refill. The ammonia levels in the controls declined almost linearly during the study period, eventually becoming depleted by day 180 of the refill period.

Dissolved oxygen and pH changes in the tanks are shown in Figures 22 and 23. There were also wide diurnal variations in oxygen levels not reflected in Figure 22. In general, test simulation oxygen levels were at or above saturation and exceeded control levels after 30 days of refill. This condition remained until 120 days of refill. The pH levels varied between about 7.5 and 9.2, and generally followed the trend in dissolved oxygen level, suggesting photosynthetic activity as the main cause of the changes.

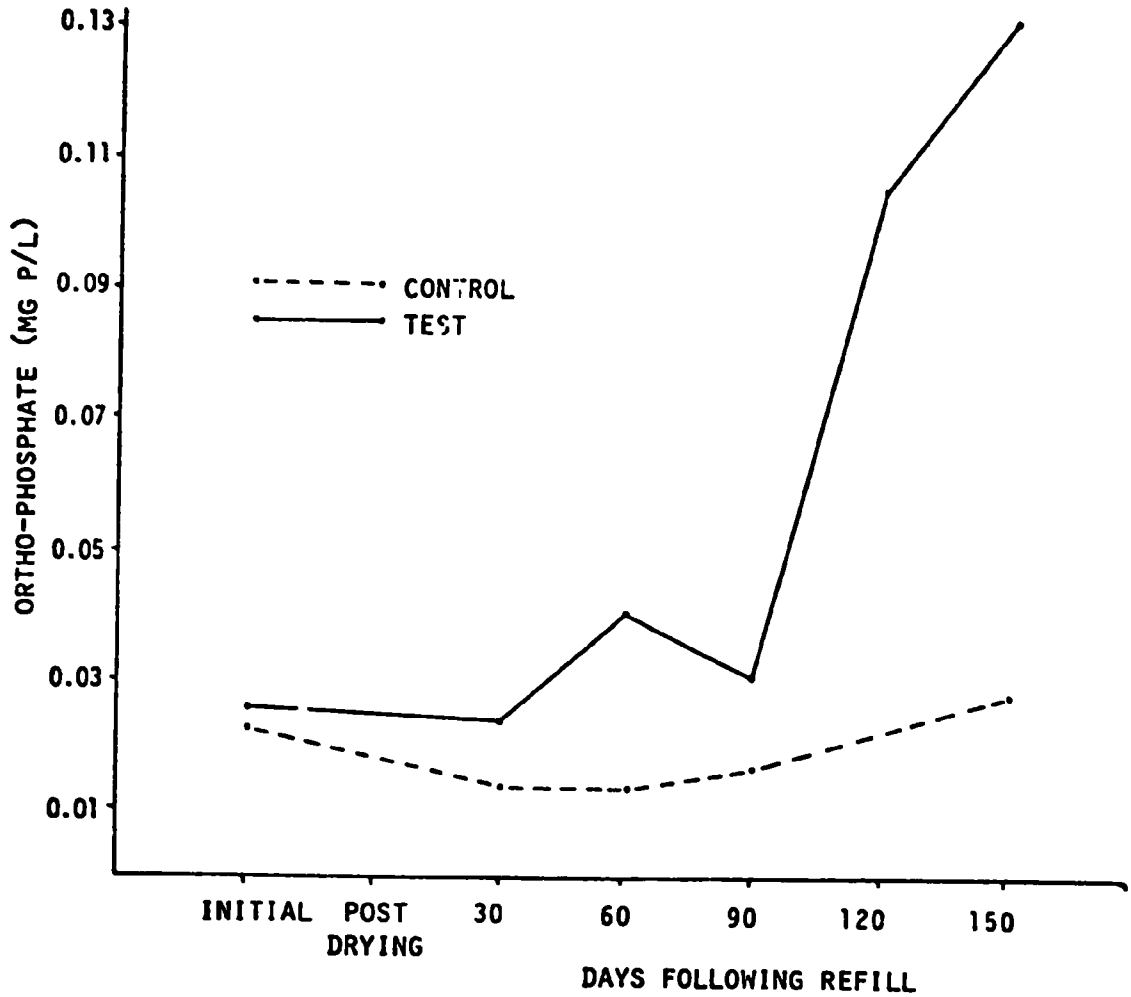


Figure 18. Changes in orthophosphate in column and tank simulation water

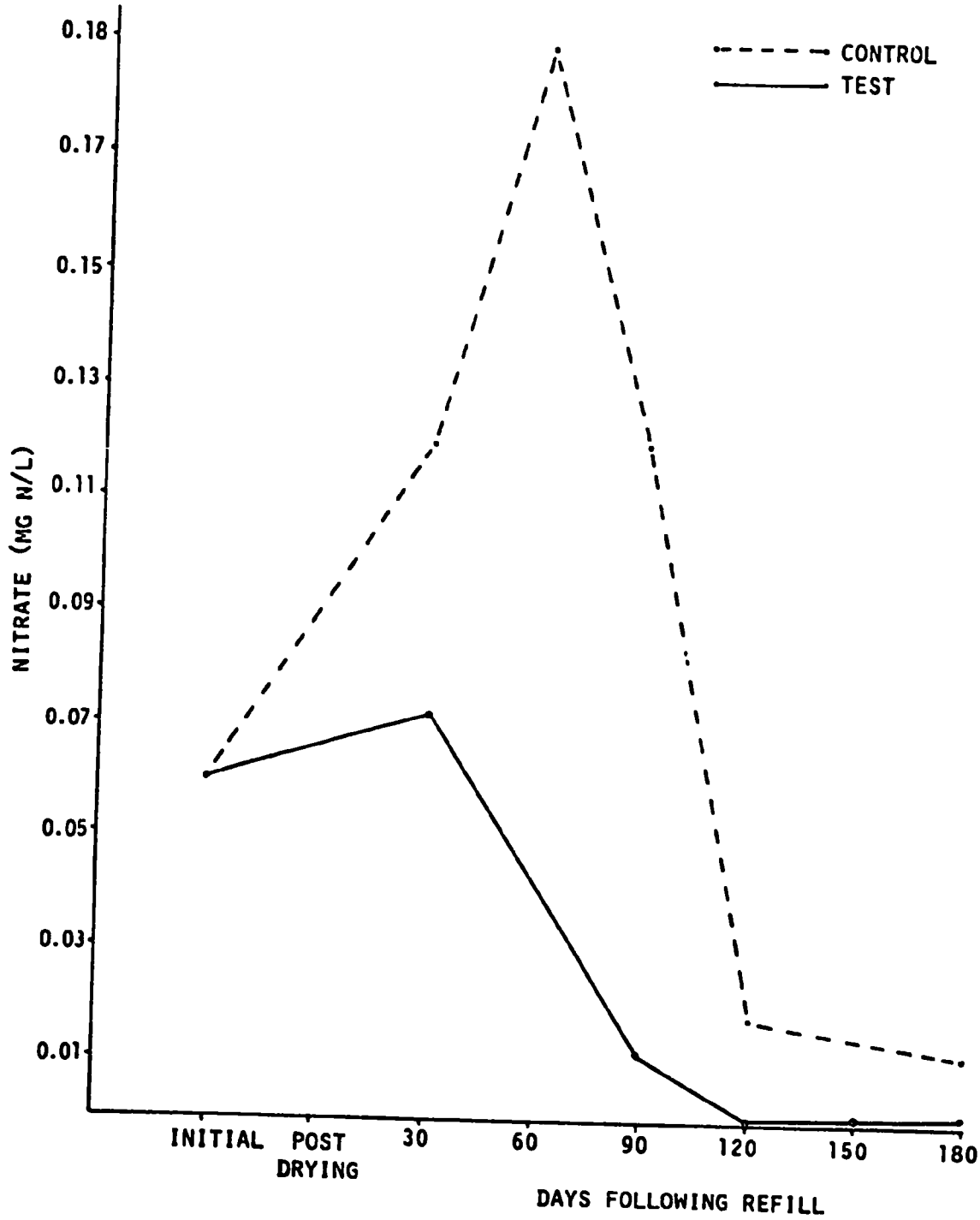


Figure 19. Changes in nitrate in column and tank simulation water

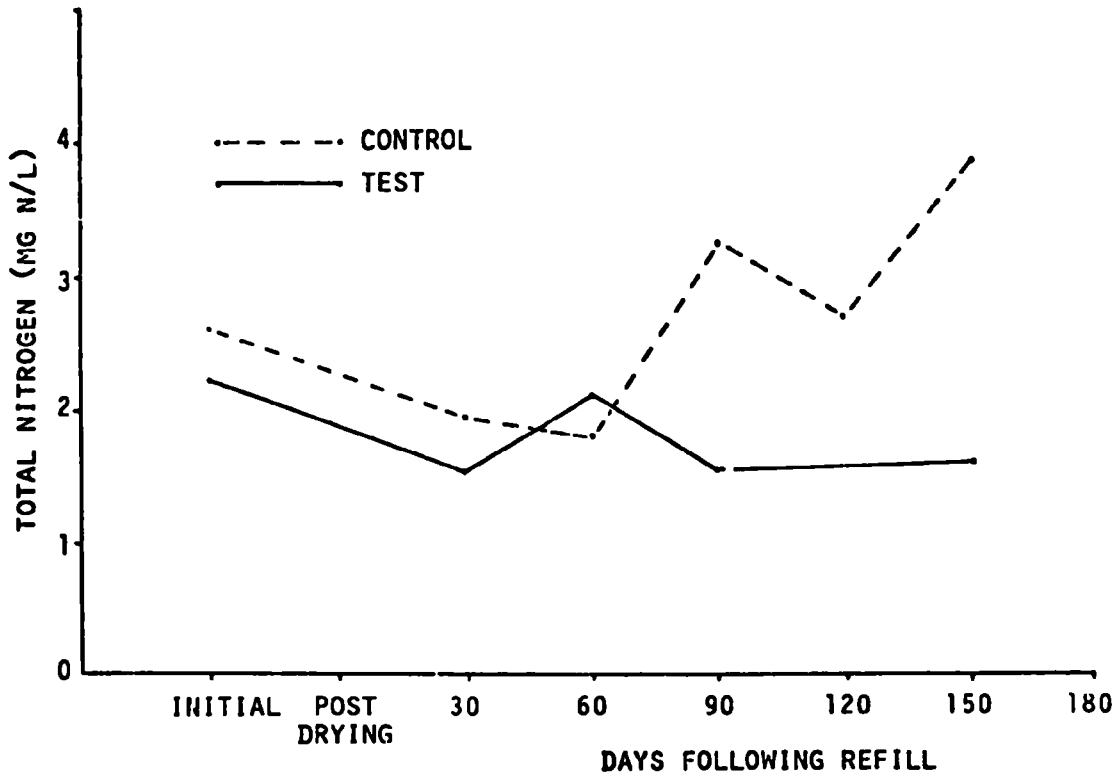


Figure 20. Changes in total nitrogen in column and tank simulation water

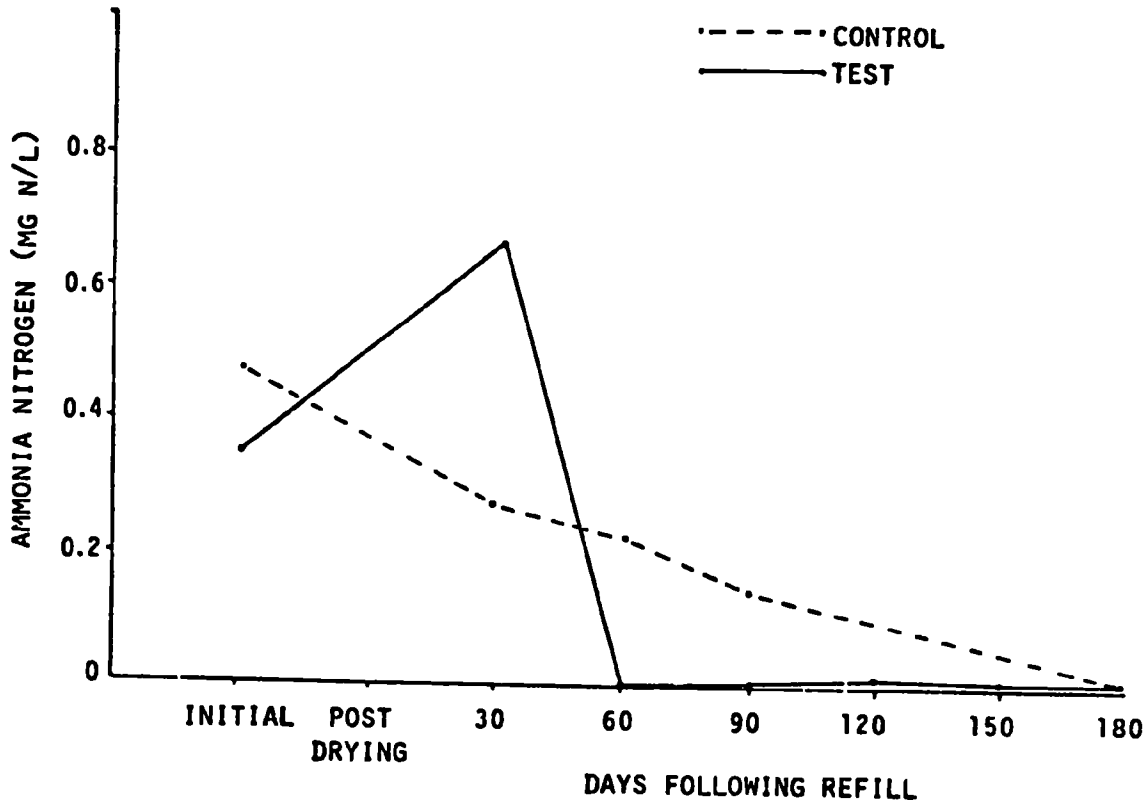


Figure 21. Changes in ammonia in column and tank simulation water

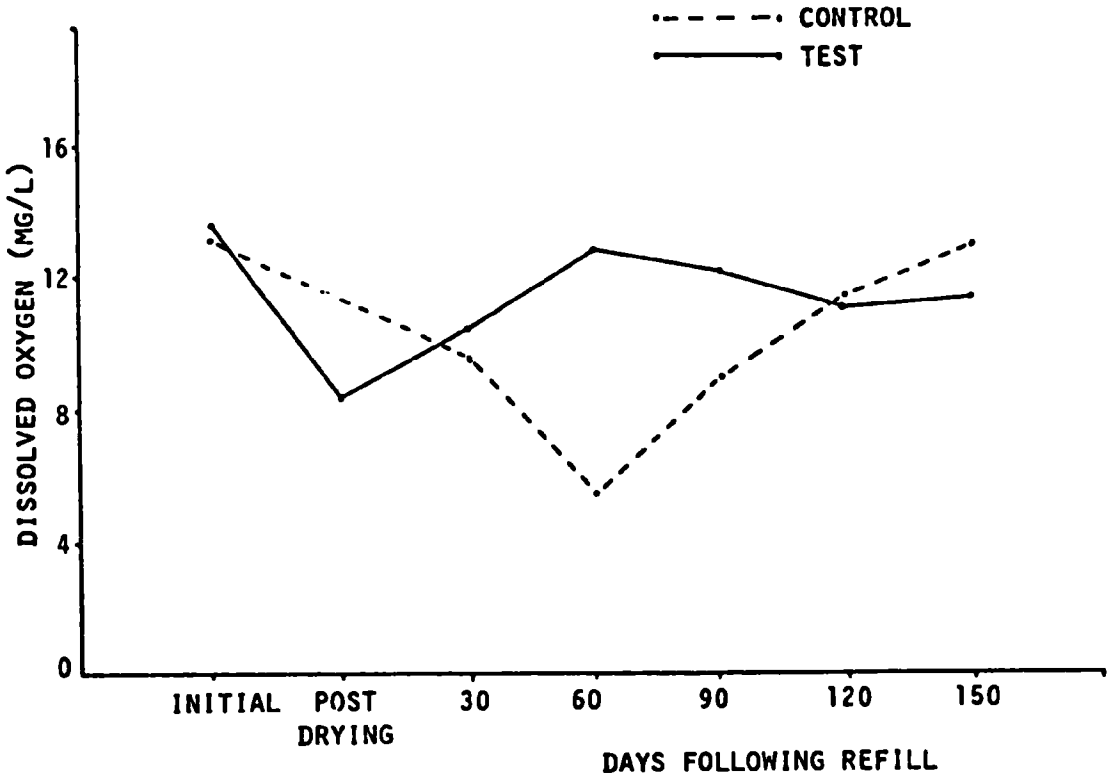


Figure 22. Changes in dissolved oxygen in column and tank simulation water

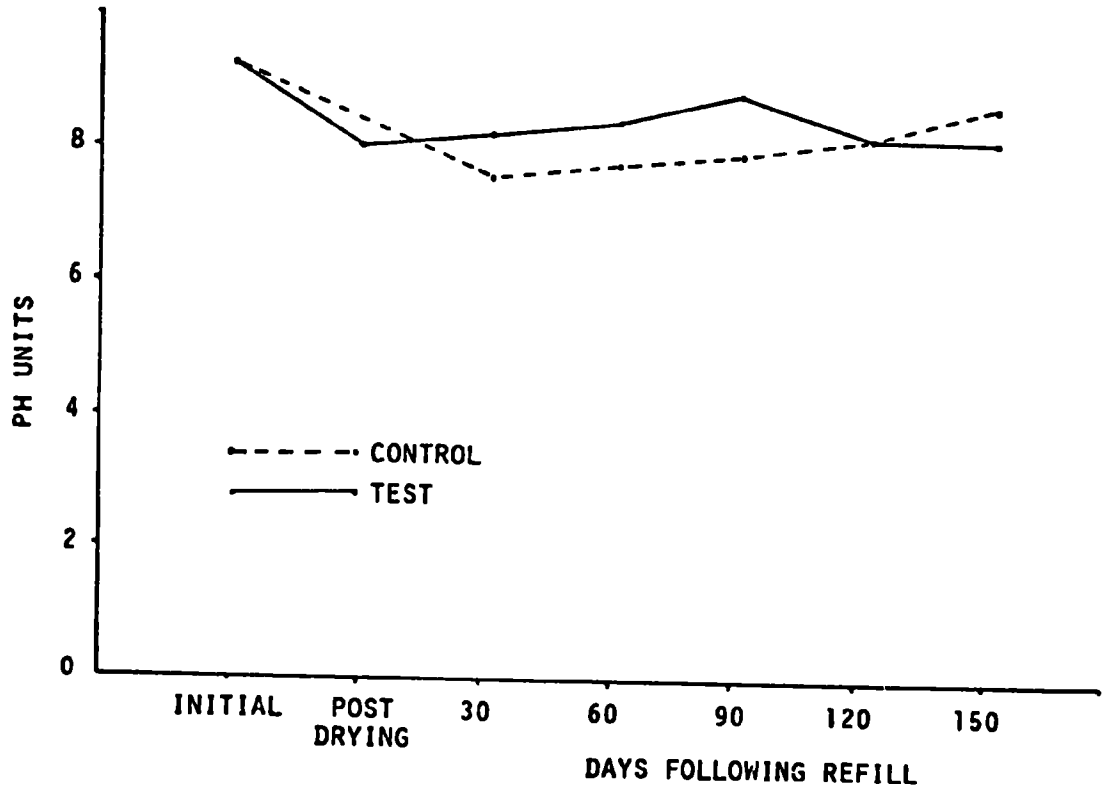


Figure 23. Changes in pH in column and tank simulation water

Similar measurements were made on water quality in the test and control pool simulations and Figures 24 to 32 summarize the results. For purposes of this report the data for the test pools and the control pools were averaged and the graphs show the mean values vs time for the two treatments (test--dried sediment--and control--undried sediment).

Water temperature in the pools reflected the changing season and short term weather changes, with a general trend of increasing temperature in the weeks following refill (which occurred on Jan. 15, 1974). Control and test pool temperatures closely paralleled each other but there was a distinct trend for slightly warmer water (by about 1°C) in the control pools. This evidently resulted from increased turbidity levels in the control pools (Figure 24), which caused greater absorbance of solar radiation. Comparison of Figures 24 and 25 indicates that maximum temperature differences between the test and control pools coincided with maximum differences in turbidity. Drawdown had a marked effect on turbidity levels in the pools; values in the test pools ranged from less than 1 JTU to about 4 JTU in the weeks after refill. Turbidity in the control pools was consistently higher than in the test pools, and large fluctuations in the control pools reflected periodic sediment resuspension. The mean turbidity for the control pools was nearly 8 JTU and the range was 2 to 17 JTU.

Alkalinity (Figure 26) and pH (Figure 27) levels in the test and control pools showed relatively large temporal variations during the weeks following refill, obscuring any differences which may have existed between the two sets. Variations in alkalinity resulted primarily from evaporation during dry weather plus the dilution effect of periodic rainfall. Variations in pH also resulted from these factors plus biological reactions such as photosynthesis and respiration. In general there was a trend toward higher pH in both sets of pools in the weeks following refill with initial values less than 8.0 in both sets. The pH of the test pools approached 9.0 by the end of the study, while the control pools fluctuated between 8.2 and 8.6.

Ortho and total phosphate levels (Figures 28 and 29) exhibited similar trends in both control and test pools, with orthophosphate variations being especially close. No initial spike of either ortho or total phosphorus was noted in the test tanks upon refill. Concentrations of orthophosphate in both sets of pools ranged from moderate to high during the monitoring period. Figure 28 shows a slight tendency for lower orthophosphate in the test pools, but this trend was not statistically significant. Except for two brief peaks of 0.11 and 0.060 mg P/l, orthophosphate remained less than 0.050 mg P/l in the test pools and averaged 0.028 mg P/l (excluding the two peaks) and 0.038 mg P/l including all the data. The mean for the control pools for all dates was 0.044 mg P/l.

Total phosphate levels in both sets of pools started at about 0.080 mg P/l just after refill and showed a trend toward gradually increasing values in both sets throughout the monitoring period. Control levels were higher than test pool levels on all but one date and the differences appeared to increase as the post drawdown monitoring period continued. A maximum of 0.21 mg P/l was reached in the control pools

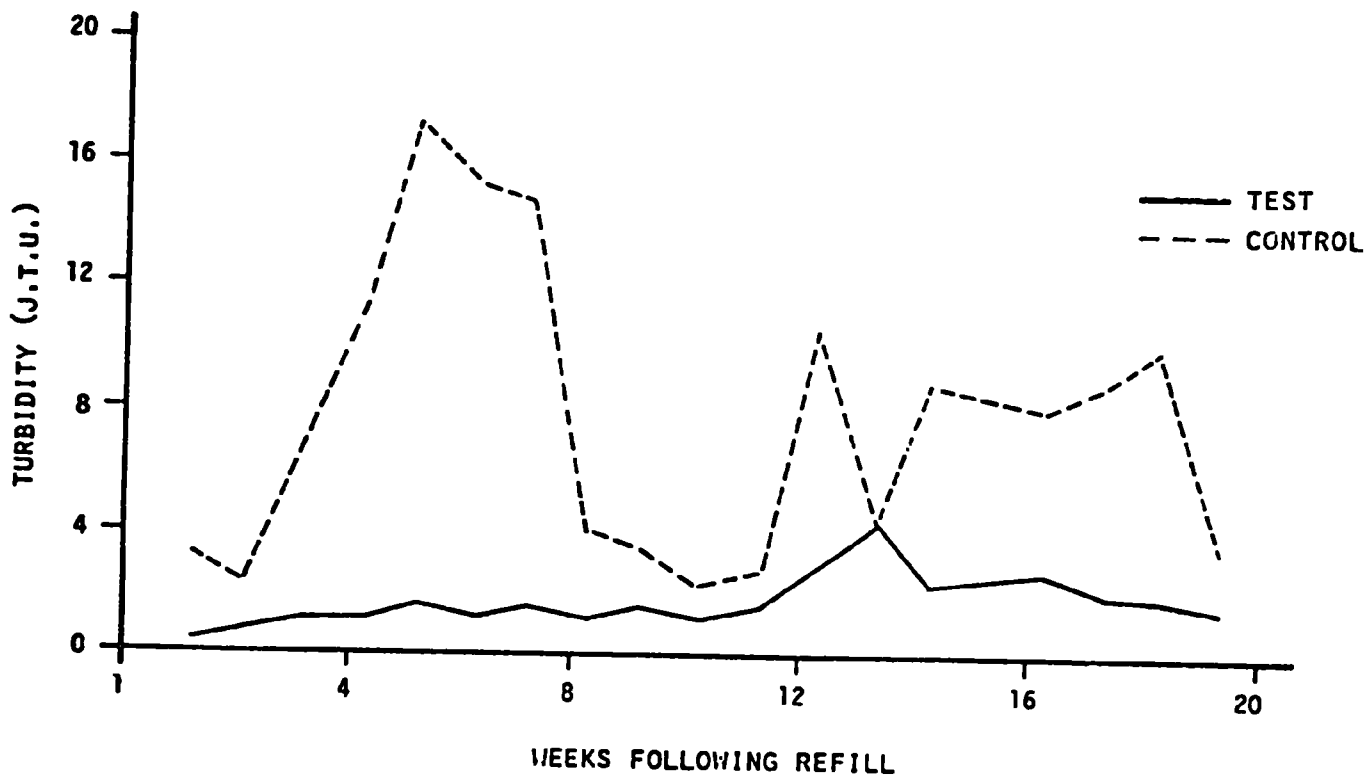


Figure 24. Weekly changes in turbidity in pool simulations

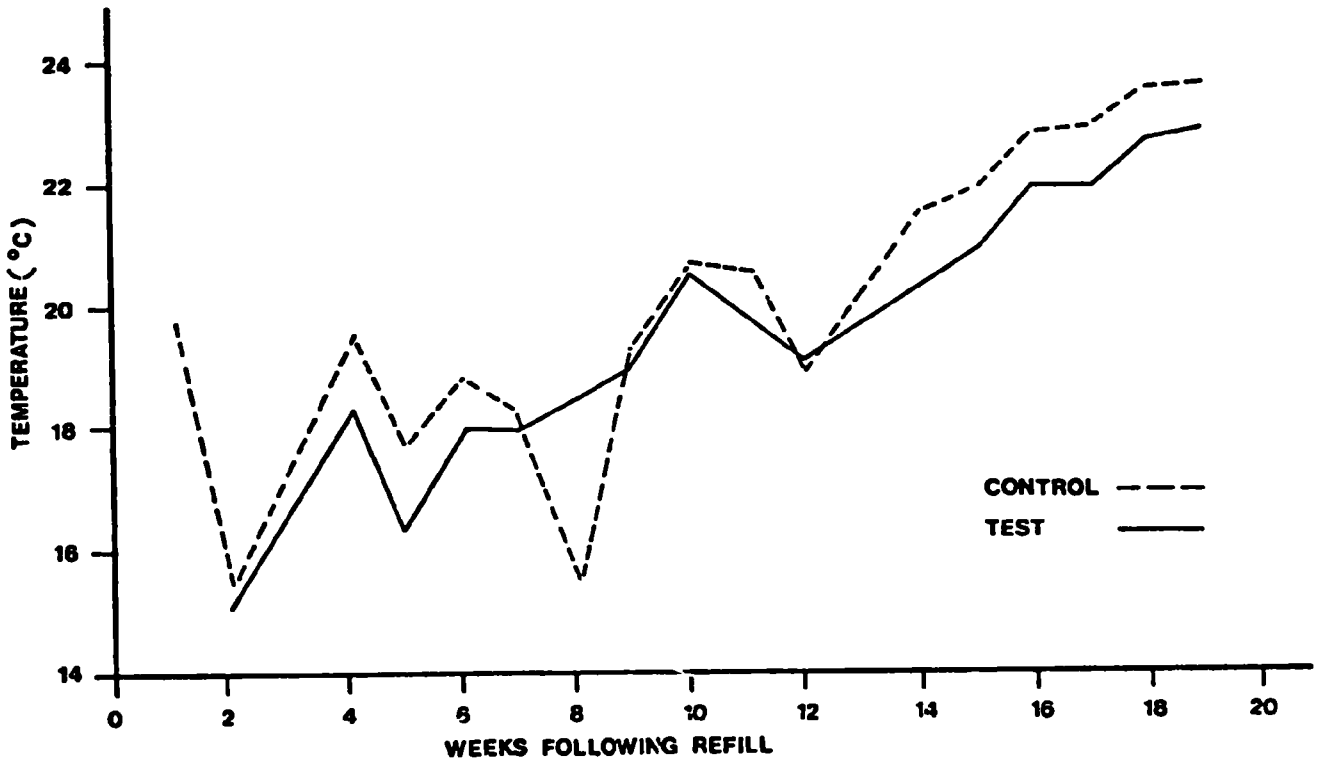


Figure 25. Weekly changes in temperature in pool simulations

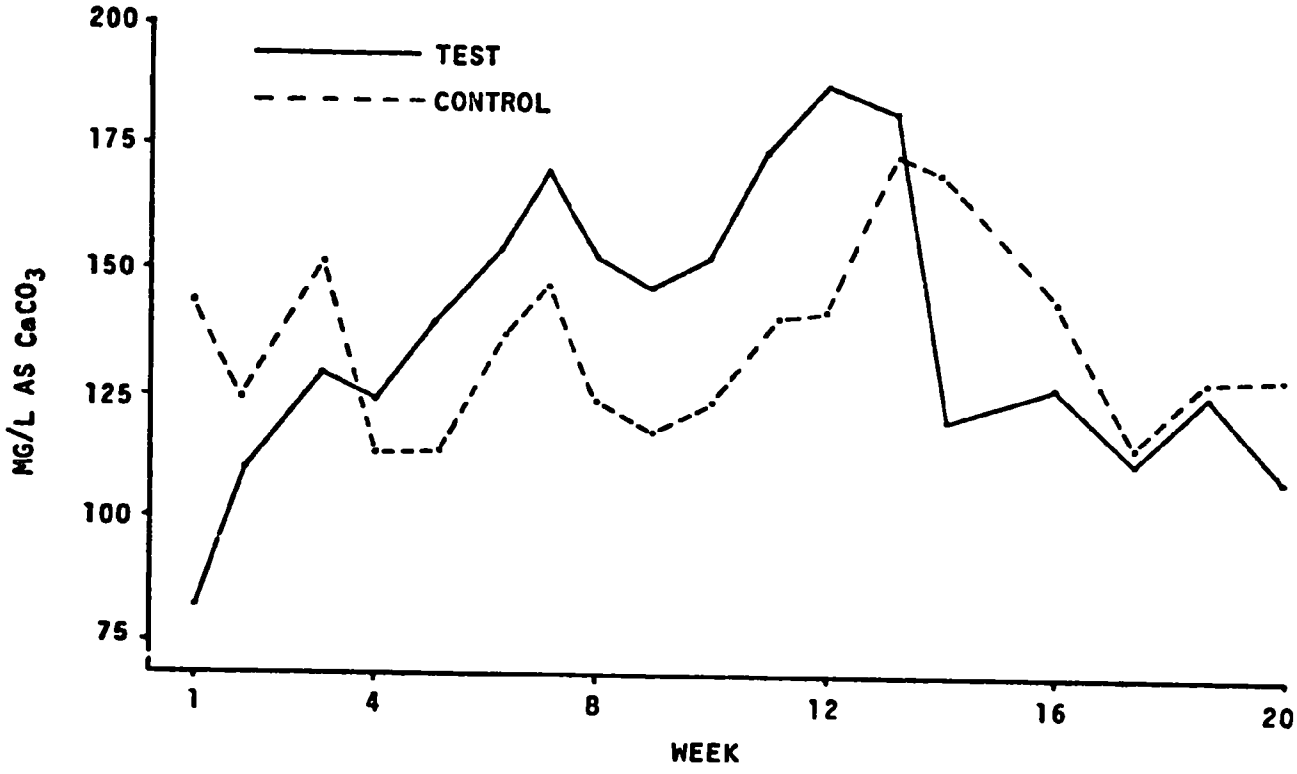


Figure 26. Weekly changes in alkalinity in pool simulations

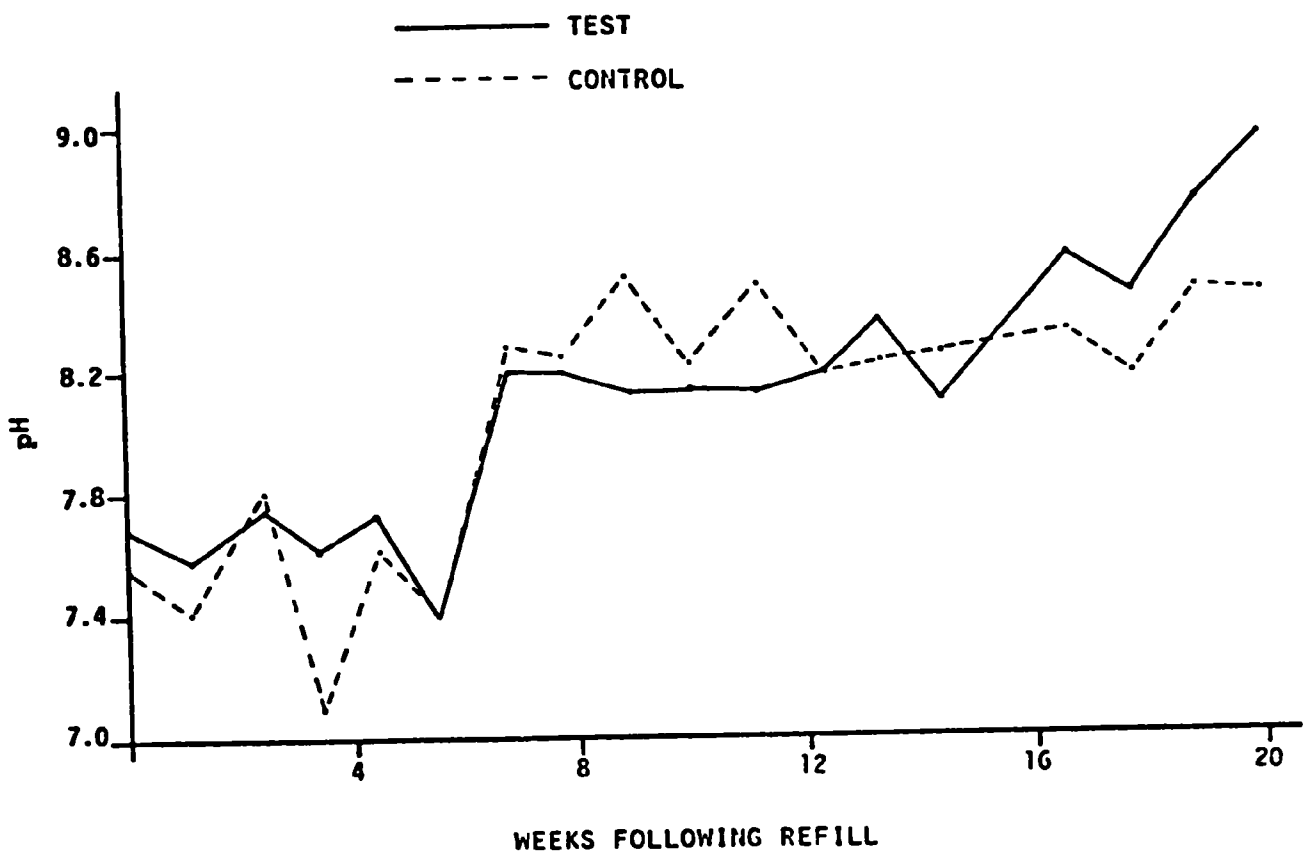


Figure 27. Weekly changes in pH in pool simulations

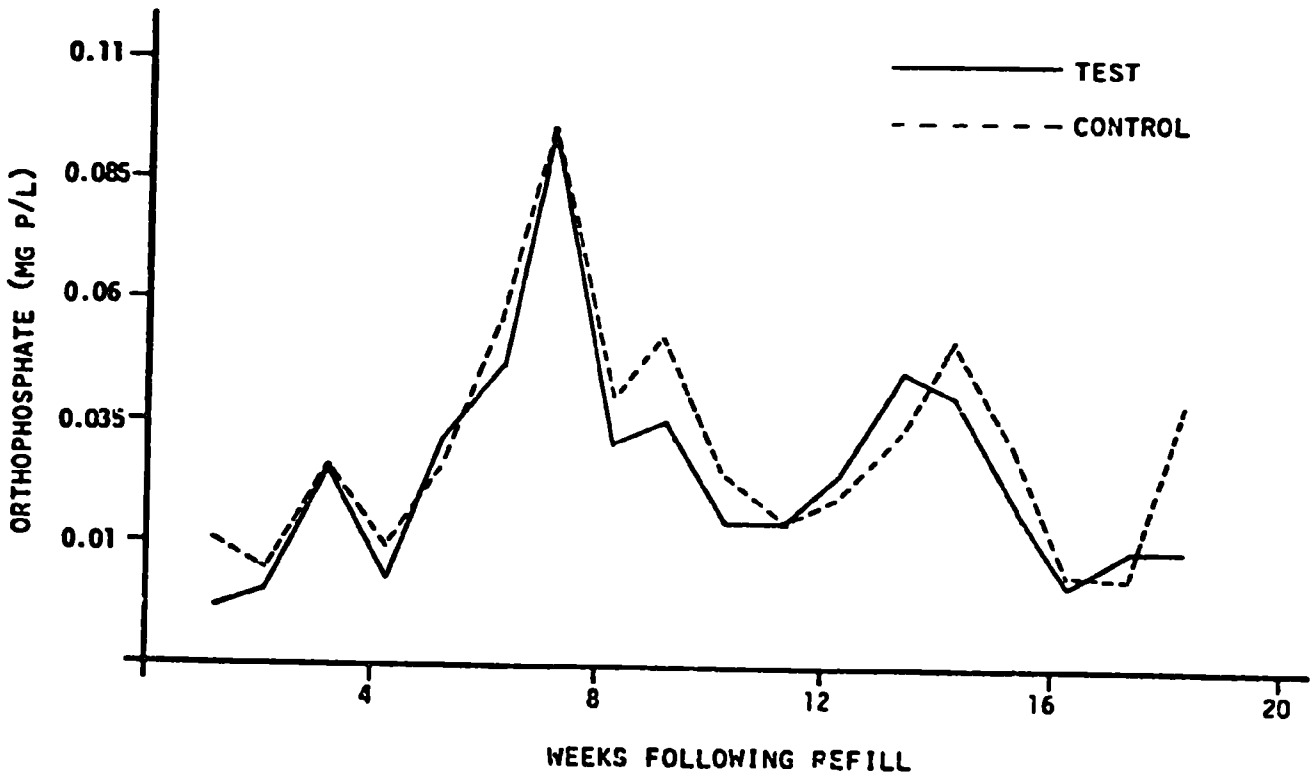


Figure 28. Weekly changes in orthophosphate in pool simulations

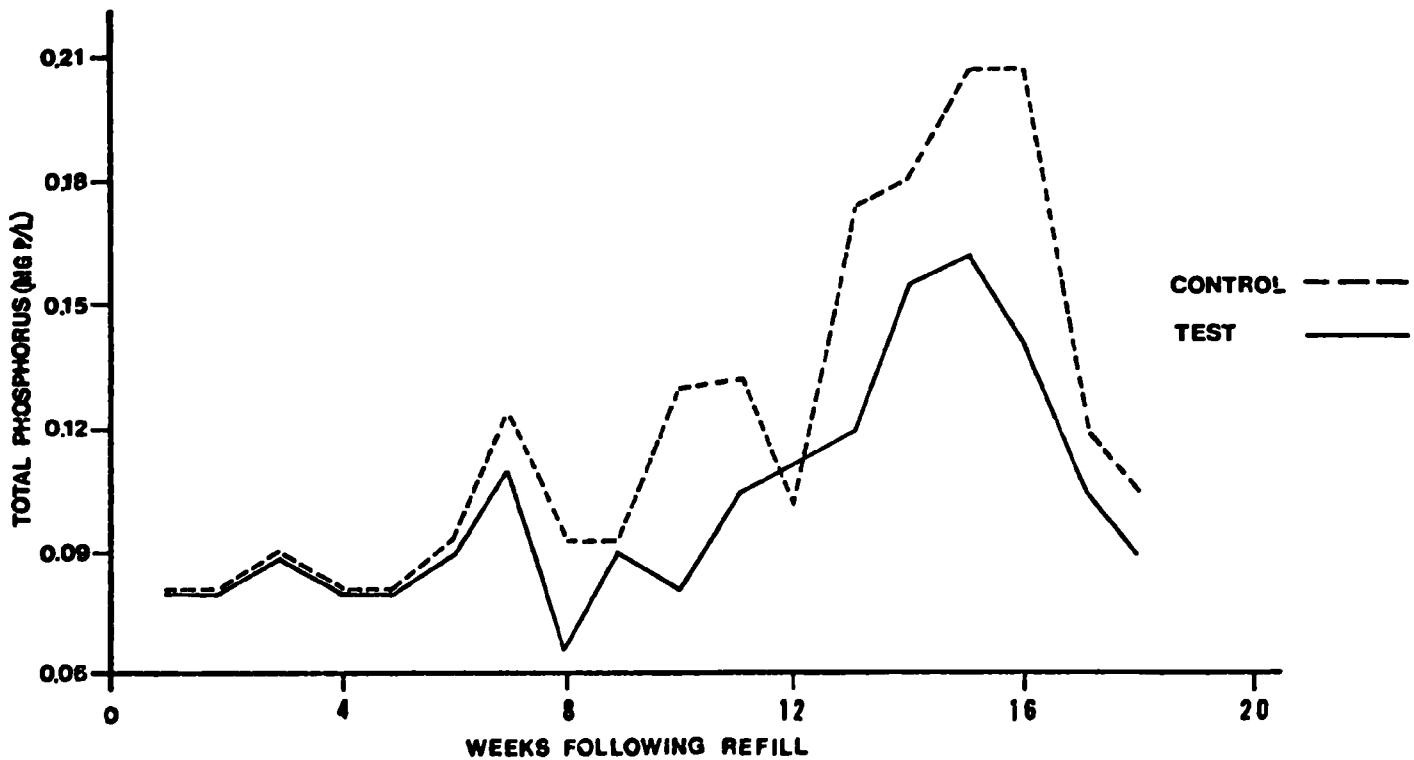


Figure 29. Weekly changes in total phosphorus in pool simulations

by the 15th week. The test pool maximum occurred simultaneously but reached only 0.16 mg P/l.

An initial spike in the aqueous ammonia content occurred in the test pools when they were refilled, but this spike rapidly dissipated (Figure 30). Consequently, the initial ammonia level was higher in the test pools (0.45 mg N/l) than in the controls (0.33 mg N/l), but the control pools maintained high levels (0.3-0.4 mg N/l) until the seventh week after refill, whereas the ammonia in the test pools dropped to less than 0.1 mg N/l by the second week. Two mechanisms appear responsible for this rapid decline: readsorption of ammonia by the sediment--probably by a layer of benthic algae and bacteria--and volatilization of ammonia to the atmosphere under the slightly alkaline pH conditions (Bouldin et al. 1974). There was not a sufficient increase in planktonic algal biomass during this period to account for the rapid ammonia loss (see later discussion of chlorophyll data). Ammonia in the test pools subsequently fluctuated between 0.05 and 0.2 mg N/l, showing at first a slowly increasing trend to a maximum of 0.2 mg/l by the ninth week and then rapidly declining and remaining below 0.1 mg/l until the end of the monitoring period. Ammonia in the control pools declined rapidly after the seventh week, and test and control levels were equal by the ninth week. However, the decline in control pool ammonia was slower thereafter, and the control pools exhibited significantly higher ammonia levels compared to the test pools for most of the remainder of the monitoring period.

Nitrate levels in the pools were temporally highly variable (Figure 31), but no initial spike was noted in the test pools following refill. Evidently some organic oxidation occurred in the drying sediments, releasing ammonia from organic N forms, but under the drying conditions used for these simulations nitrification of the released ammonia did not occur. This is probably an advantage since nitrate is more easily leached from sediment upon refill than is ammonia, and conversely ammonia is more easily lost from solution by volatilization and sediment resorption than is nitrate. The relatively large and rapid oscillations in nitrate levels in the pool waters after refill preclude detailed discussion of the data, but it is clear from Figure 31 that nitrate was generally higher in the control pools than in the test pools during the first nine weeks following refill. Nitrate levels in both sets of pools showed similar fluctuations thereafter, but the maximum values associated with two short-lived peaks in nitrate were higher in the control pools.

Interpretation of the temporal variations in inorganic nutrient forms is a complicated matter. Obviously the variations are the result not only of sediment water interactions but of seasonal and short-term changes in benthic and planktonic algal growths, as well as periodic nutrient inputs to the open pools from rainfall and dry fallout. It can be concluded that sediment drying did not cause any spikes of orthophosphate and nitrate in the refill water and that an initial ammonia spike was short-lived with values rapidly falling below the control pool levels. Furthermore, inorganic nitrogen levels were

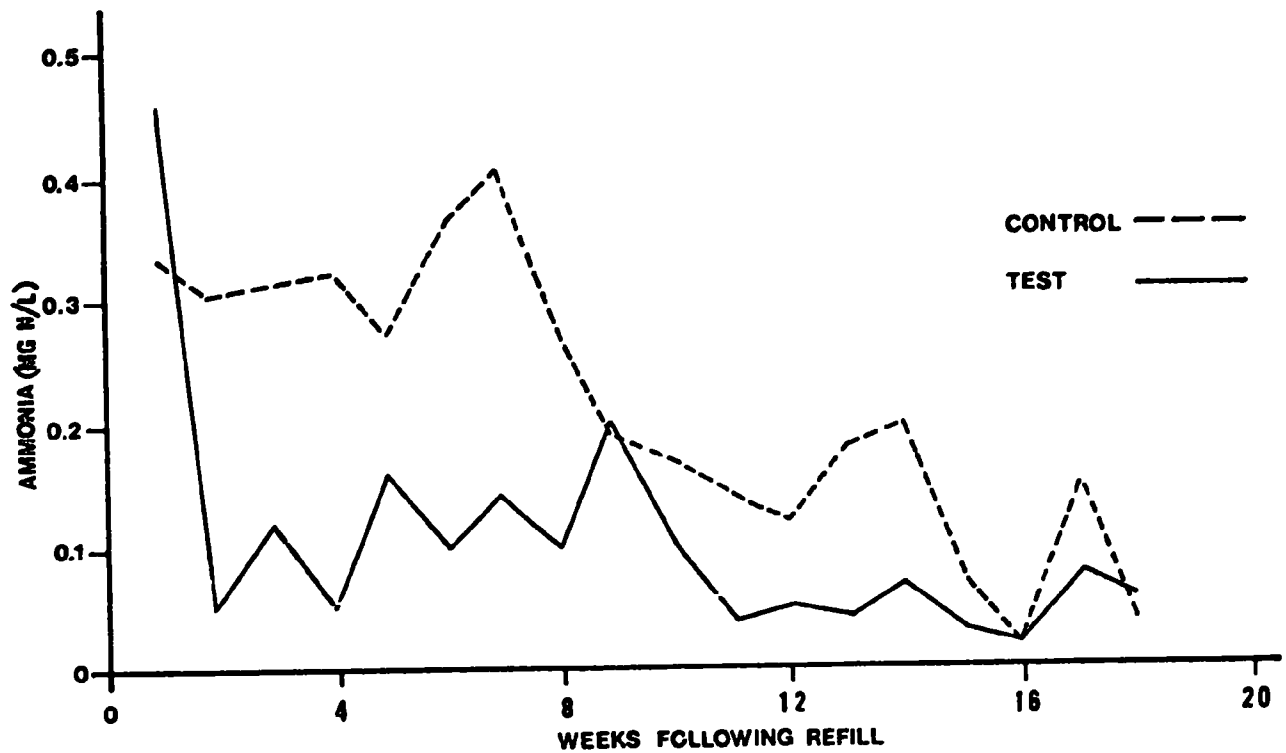


Figure 30. Weekly changes in ammonia in pool simulations

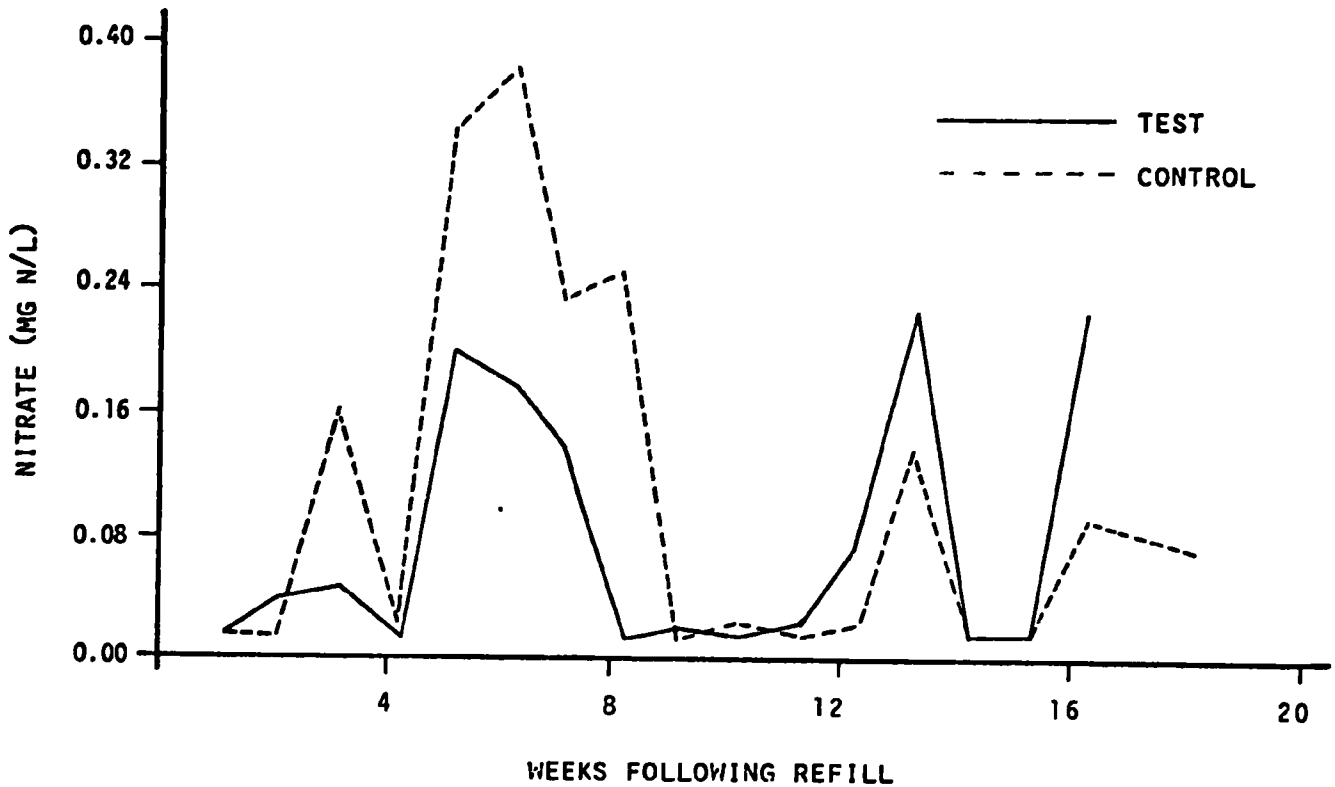


Figure 31. Weekly changes in nitrate in pool simulations

generally lower in the test pools compared to the controls. Inorganic phosphate was nearly the same in both sets of pools, but total phosphate was definitely lower in the test pools. Total organic nitrogen (not graphed) was also lower in the test pools, but this conclusion is based on limited data.

Dissolved oxygen values (Figure 32) in the test pools were, with few exceptions, higher than those in the control pools, and both sets were generally above saturation values. By the end of the monitoring period, both sets had dissolved oxygen values considerably higher than saturation, implying high rates of photosynthesis. The generally higher oxygen values in the test tanks may be explained by their lower turbidity values. The high turbidities in the control pools reflect suspension of the flocculent sediment; consequently sediment oxygen demand is likely to have been a more important factor in the control pools than in the test pools with their consolidated sediments.

The data shown in Figures 24 to 32 were subjected to statistical analysis in order determine the significance of apparent differences between the two sets of pools. Analysis of variance was performed (Table 19) for 14 parameters on three combinations of the pool water quality data: 1) total time following refill using all pools, 2) total time following refill excluding two pools ($\alpha$ and $\varepsilon$) that devoloped leaks and were inadvertently partially dried (see Section VI), and 3) last two months of monitoring period using all pools. The effects of the partial drying and transfer operations on conditions in pools $\alpha$ and $\varepsilon$ were felt to have been eliminated by that time. Hence this treatment probably represents the best comparison of refill water quality in the dried and undried pools. The results in Table 19 corroborate the qualitative judgements made earlier in describing Figures 24 to 32, and indicate that among the nutrient forms only total phosphorus and ammonia were significantly different in the two sets of pools. Temperature was significantly different during the last two months for all pools, and turbidity just missed being statistically different (at the 95% confidence level) using the same sampling period. However, turbidity differences were highly significant for the total refill time when the two leaky pools were excluded. Dissolved oxygen differences were significant for both of the latter two data treatments but were not different considering all pools for the total time. Only one parameter (total phosphate) was significantly different for all pools and sample times, but seven parameters were significantly different when the two leaky pools were excluded, and six parameters showed significant differences using all pools for the last two months of monitoring.

A laboratory experiment was conducted to determine the effect of drying extent on refill water quality. Six small (15 cm x 25 cm x 20 cm) aquaria were filled with fresh Lake Apopka sediment to a depth of 6.5 cm. One aquarium was then filled with Gourd Neck Springs water and served as a control, while the other five were subjected to accelerated drying conditions under 150 watt spotlights. Aquaria remained under the lights for varying periods in order to achieve a gradient of dryness; as each aquarium was removed from the lights, it was covered with

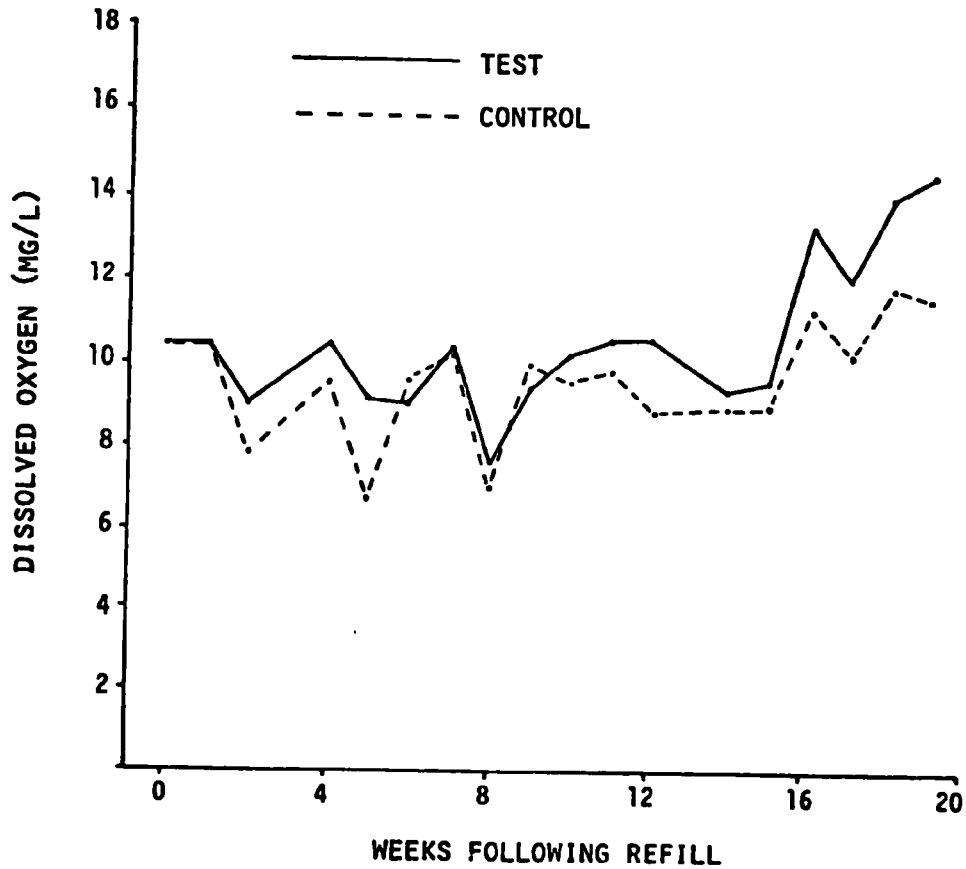


Figure 32. Weekly changes in dissolved oxygen in pool simulations

TABLE 19. ANALYSIS OF VARIANCE RESULTS FOR WATER QUALITY PARAMETERS IN POOL SIMULATIONS

| Parameter | Sampling Periods | | |
|---|---|---|---|
| | Total Time (All Pools) | Total Time (Excluding Two) | Last Two Months (All Pools) |
| pH | 0.32 | 0.75 | 0.16 |
| Conductivity | 0.29 | 0.001[a] | 0.14 |
| Turbidity | 0.44 | 0.001[a] | 0.07 |
| Dissolved Oxygen | 0.19 | 0.001[a] | 0.04[a] |
| Temperature | 0.08 | 0.40 | 0.001[a] |
| Alkalinity | 0.59 | 0.001[a] | 0.71 |
| Ammonia | 0.98 | 0.001[a] | 0.03[a] |
| Nitrate | 0.27 | 0.48 | 0.10 |
| Orthophosphate | 0.79 | 0.76 | 0.37 |
| Total Phosphorus | 0.001[a] | 0.30 | 0.02[a] |
| Percent Water | 0.15 | 0.001[a] | 0.001[a] |
| Volatile Solids | 0.83 | 0.001[a] | 0.67 |
| Oxygen Saturation | 0.26 | 0.04 | 0.09 |
| Chlorophyll a | 0.17 | 0.87 | 0.03[a] |
| Adenosine Triphosphate | 0.95 | 0.10 | -- |
| Bacterial Counts | 0.47 | 0.81 | -- |

[a] Denotes significant difference (at or greater than 95% confidence level) between test and control pools.

aluminum foil to prevent further evaporation. The least dried sediment had only slight cracks in its surface while the most dried had developed cracks down to the bottom of the aquarium and had separated into individual hard cakes. After all aquaria had been removed from the lights, they were filled with fresh Gourd Neck Springs water by carefully siphoning water down the sides of the containers to minimize sediment disruption. Air spargers were placed in the aquaria to keep the water mixed and aerated and samples were taken from each aquarium twice a week for a one-month period for turbidity and inorganic nutrient analyses. Results of the monitoring are summarized in Figure 33.

The extent of drying is shown by the percent water in sediment samples taken prior to refill. The control sediment (#1) was 96.6 percent water, the least dried sediment had a water content of 91.6 percent, and the most dried had only 16.1 percent water. It is obvious that drying had little effect on organic content. Even the driest sediment had a percent volatile solids only 1.2 percent lower than the control, and the range for all six aquaria was 67.2 to 69.0 percent. Of course, under slower, natural drying conditions microbial oxidation may result in somewhat larger volatile solids decreases, but the results of our larger simulations also indicate minor losses.

The effect of drying period on inorganic nutrient levels in the refill water was different for each of the three nutrient forms. Drying enhanced nitrate release from the sediments in that all dried aquaria had higher nitrate levels than the control. However, initial nitrate release was inversely related to the extent of drying, with the least dry sediment yielding nitrate spikes greater than 2 mg N/l and concentrations then declining with time. The more highly dried sediments showed lower initial spikes but increasing nitrate levels with time following refill. Apparently a small amount of drying enhances nitrification but extended drying inhibits further nitrifying activity. When the highly dried sediments are reflooded they provide a better substrate for nitrifiers than the undried muck, and consequently nitrate levels eventually increase.

The opposite results are seen for ammonia in the refill water. Slight to moderate drying depressed ammonia release, but extended drying caused sufficient organic matter decomposition to produce elevated ammonia levels in the refill water. However, only the most highly dried sediment (which is unlikely to be achieved in the environment) produced excessive ammonia levels. It should be noted that the control ammonia levels were much lower than values found in Lake Apopka. The refill water (from Gourd Neck Springs) is naturally low in ammonia. Under the relatively quiescent laboratory conditions, a defined sediment/water interface was maintained, thus minimizing ammonia release.

Orthophosphate levels in all aquaria showed increasing concentrations with time following refill. Extensive drying led to elevated phosphate releases compared to the control sediment, whereas the slightly to moderately dried sediments had orthophosphate levels somewhat lower to slightly more than the control. Turbidity levels in the refill

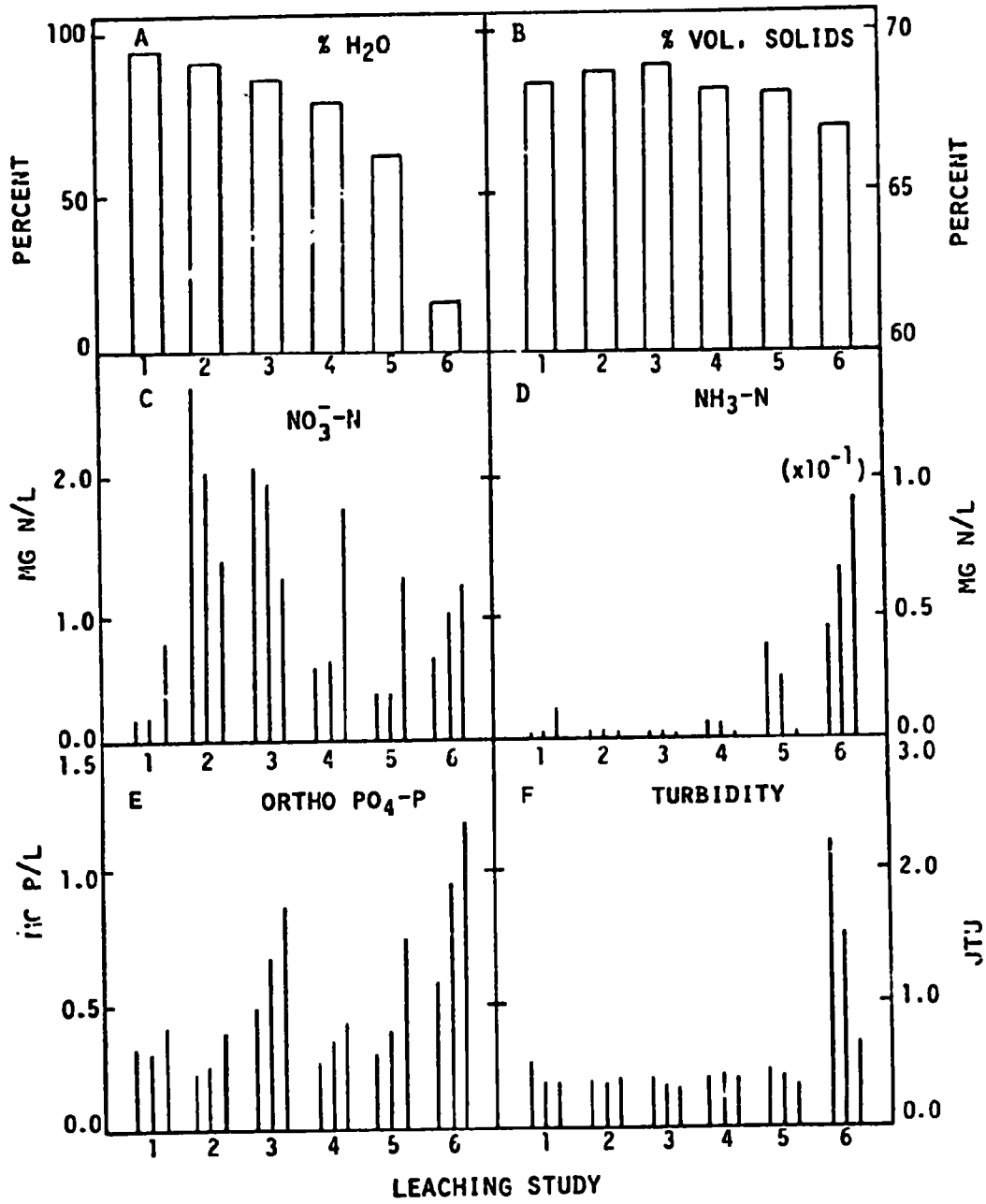


Figure 33. Effect of drying period on sediment and refill water quality in laboratory (aquarium) experiment. Numbers on abscissa represent different aquaria (dried to varying extents). The three vertical lines in C-F for each aquarium represent analysis made one, two, and three weeks after refill, respectively

waters of all aquaria (including the control) were lower than typical values for Lake Apopka. Slight to moderate drying decreased the already low control values, but extensive dr ing led to elevated turbidity.

The results in Figure 33 indicate there is an optimum level of drying in terms of effects on water quality. Too little drying stimulates nitrate release and more seriously will not consolidate the sediments and stabilize the interface. On the other hand extensive drying tends to release more ammonia and phosphate, at least in short term tests, than does undried sediment.

## BIOLOGICAL EFFECTS

Biological parameters measured in water overlying the sediments in the test and control columns and tanks included primary production and chlorophyll a. Production was measured by the C-14 uptake technique (Parsons and Strickland, 1968). As mentioned earlier, the primary producer component of the control columns and tanks shifted from phytoplankton to periphyton. Shortly after refill, the same phenomenon occurred in the test columns and tanks. This shift from plankton to periphyton in the controls, which was clearly unrepresentative of Lake Apopka, necessitated abandonment of the tanks and columns and the construction of the continually mixed pool simulations. Figure 34 contains typical control tank data showing the rapid decline in C-14 productivity and chlorophyll a levels, both indicating crashes in the phytoplankton populations. Sixty centimeters of muck sediment was placed in the control tank on July 29, 1972. A dense mat of Oedogonium sp. dominated the tank by late November, 1972 and remained until the end of the experiment. The test tank had the same amount of sediment added to it on July 29 and was drawn down on August 11 for approximately 2 months. Refill occurred on October 19, 1972. After the test tank was refilled, a growth of Chara rapidly covered the dried sediments and apparently outcompeted the phytoplankton for nutrients, as did the Oedogonium in the control tank. Chlorophyll a levels and C-14 primary production remained well below values measured during the same months in Lake Apopka. In December, for instance, the chlorophyll level in the control tank was about 10 $mg/m^3$ and only slightly over 20 $mg/m^3$ in the test tank. In the lake, the chlorophyll level was about 70 $mg/m^3$. The primary production value for the lake at that time was about 350 mg $C/m^3/hr$, as compared to only 20 mg $C/m^3/hr$ for the test and control tanks. With our controls behaving so differently from Lake Apopka, it was impossible to determine whether or not the low chlorophyll and productivity values in the test tank refill water were a result of drawdown and sediment drying or simply a simulation anomaly. This was one of the major reasons for changing to the mixed pool simulations.

In an effort to determine whether or not dried sediments mixed in water stimulated the growth of algae, a series of algal bioassays were performed. In discussions of drawdown as a lake restoration technique, it is often hypothesized that sun drying will, through oxidation, convert

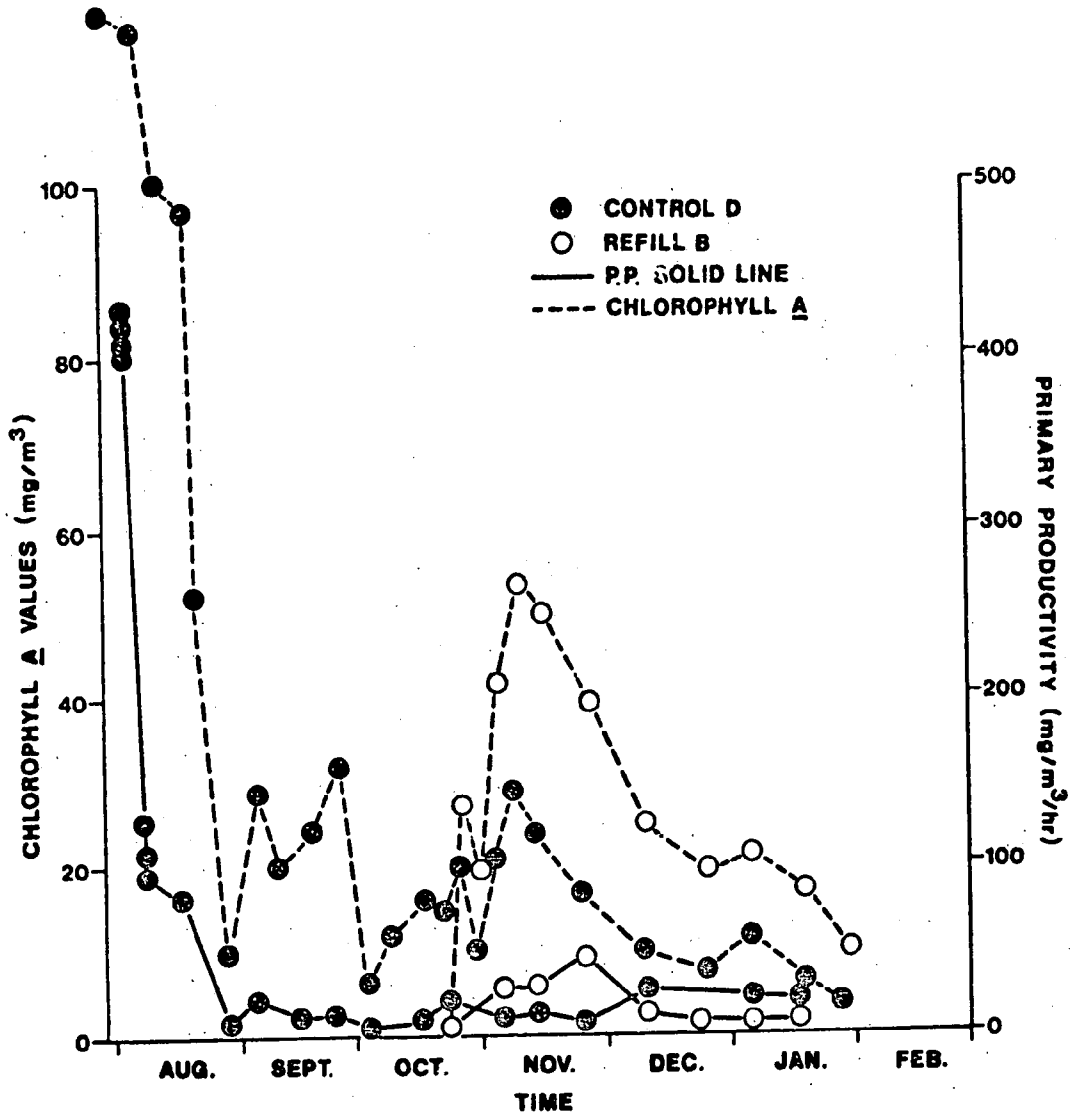


Figure 34. Primary productivity and chlorophyll a in control and test tanks

organic nitrogen and phosphorus compounds in the sediment to inorganic forms, which upon refill might be released and stimulate algal growth. In our pool experiments, we monitored algal growth following refill. Those results (described later) can be considered more representative of what might happen in Lake Apopka itself. The algal bioassay data reported here represent "worst case" results. Because the dried sediments are almost rock hard, very little mixing with the overlying water occurred in the pool simulations or would occur in the lake. In our bioassay, however, we ground the hardened sediment to a sand-like consistency and then used a magnetic stirrer to mix the sediment with deionized water for 24 hours. Obviously, no such mixing forces are present in Lake Apopka. Prior to grinding, the sediment was exposed to sunlight in a drained plexiglas cylinder for one month.

Two mixtures of dried sediment were used: 0.5 g/100 ml deionized water and 1.0 g/100 ml. Fresh wet sediment was also mixed with deionized water to yield 0.5 and 1.0 g/100 ml deionized water on a dry weight basis. A fresh sediment water content of 95 percent was used in making those calculations. The mixtures were centrifuged to remove suspended solids and the supernatants were Millipore (0.45 μm) filtered prior to testing. Three 100 ml filtered replicates of each dried sediment solution and duplicates of the wet sediment solution were inoculated with Selenastrum capricornutum to produce an initial cell concentration in the test flasks of 1000 cells/ml. Media and test procedures followed those recommended by the Environmental Protection Agency (1971a). Cell counts were made daily until the increase in cell number was less than 5 percent per day. The results are plotted in Figures 35 and 36. At 0.5 g/100 ml of deionized water, dry sediment supports less growth than does wet sediment. At 1.0 g/100 ml, however, slightly higher populations of Selenastrum were reached with the dried sediment mixture. The population of Selenastrum grown in algal assay procedure medium exceeded all test replications, reaching almost 5 x $10^6$ cells/ml after 12 days. Possibly our initial sediment inoculations were too small. This experiment did indicate, however, that dried sediment, even when thoroughly ground up and vigorously stirred with water, does not support a significantly higher algal biomass than does a comparable amount of wet sediment. Rather than continue this rather unrealistic line of experimentation, we set up the large mixed pool simulations and followed algal populations in them.

Phytoplankton sampling was begun in early April, 1974, some 5 weeks following refill. All pools were sampled on an approximate weekly basis until June 7, 1974. Using the methods recommended by the Environmental Protection Agency (1973), Shannon-Weaver generic diversity indices were calculated and are shown in Figure 37. Although an analysis of variance shows that there is a significant difference ($P = 0.07$) between test and control tank diversity (test being higher), the interpretation of the phytoplankton data is complicated by several factors. The refill water added to the test pools was originally free of algae and therefore required some time to develop a flora, which has as its origin resistant algal forms in the dried sediment and airborne algae. Although Gourd Neck Springs water is also algal

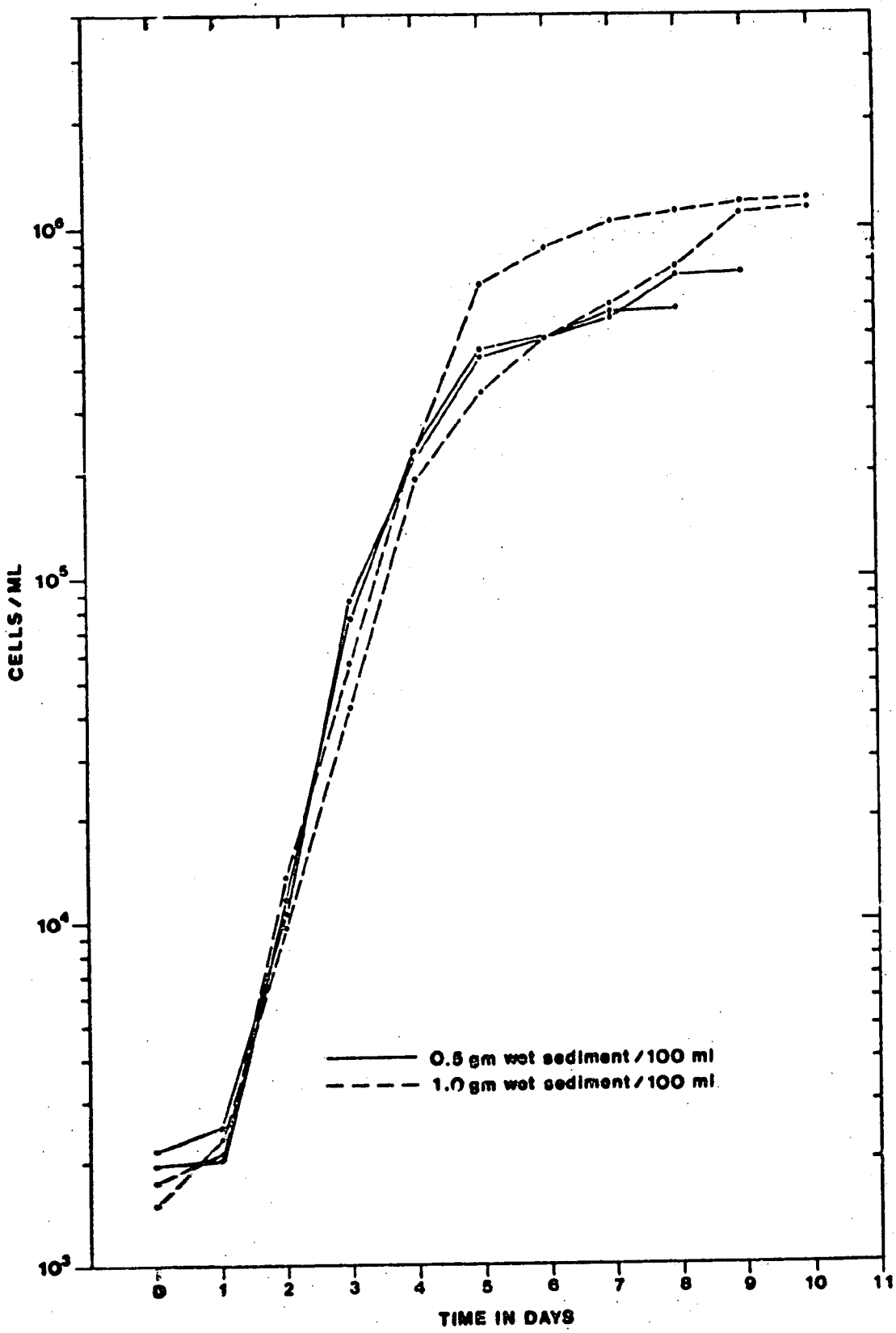


Figure 13. Selenastrum growth in wet sediment suspensions

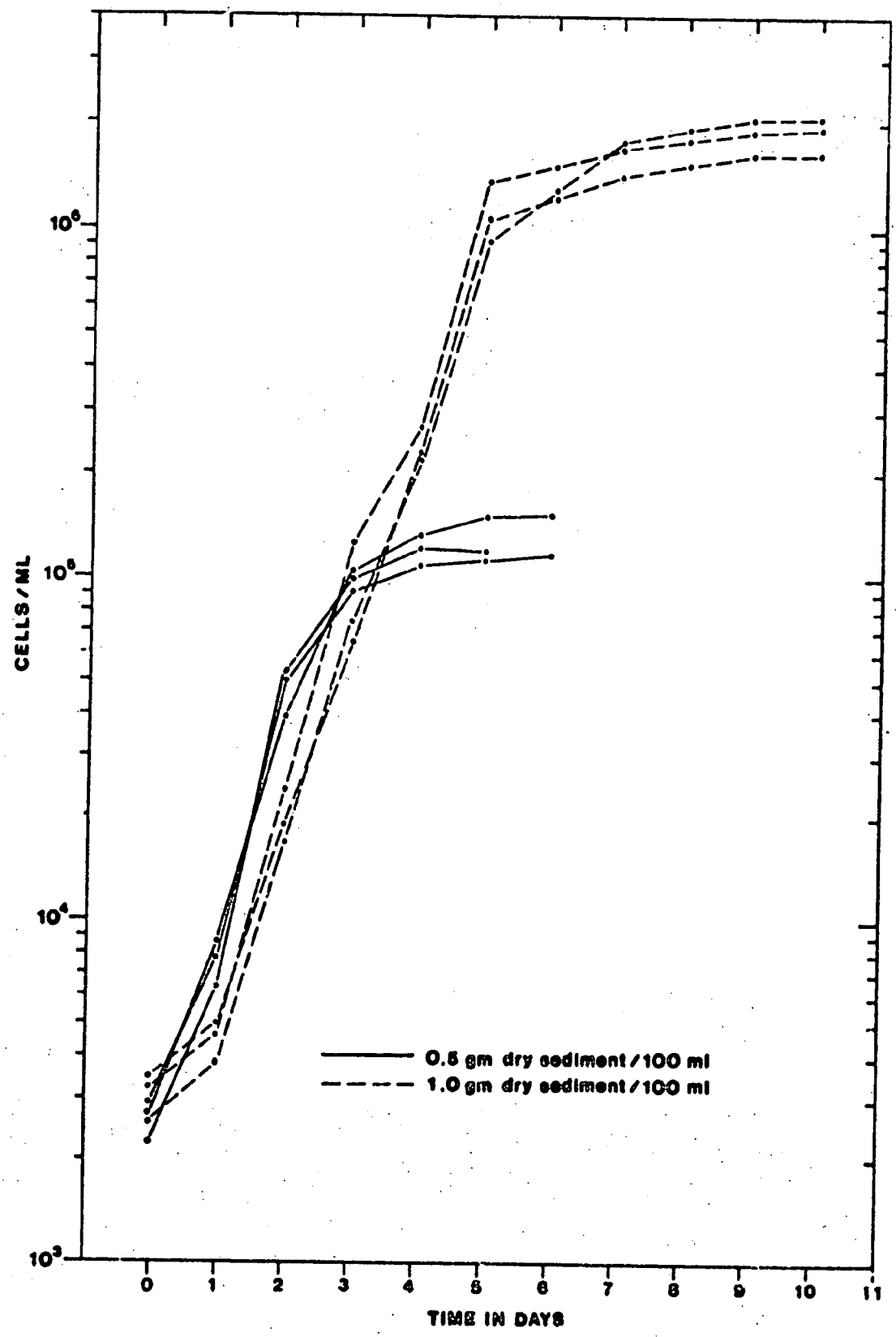


Figure 36. Selenastrum growth in dry sediment suspensions

free, it mixes immediately with the algal population indigenous to Lake Apopka. Furthermore, the test pools received no nutrient input other than that provided by rainfall and sediment-water exchange. In the lake, the water overlying dried sediment will exchange freely with water overlying wet unconsolidated sediment. Furthermore, nutrient sources other than the sediment and rainfall will exist following drawdown. The most important nutrient source will probably be runoff. Figure 37 shows that 10 weeks after refill, the diversity indices for test and control tanks are quite similar. It was mentioned earlier that turbidity in the test pools was lower than in the control pools. Hopefully, increased water clarity would occur in the lake following drawdown and favor the establishment of submerged macrophytes which would out compete the phytoplankton for nutrients or exert a negative influence on the floating algae. Again, however, the presence of unconsolidated sediments will add suspended solids and nutrients to the overlying water as the lake refills. At this time, no quantitative estimate of the influence of consolidated sediments on refill water quality can be made. Table 20 provides further detail on the phytoplankton results. Averages of the test versus control pool means presented in Table 20 indicate the test pools had lower total numbers of algae, a lower percentage of blue-greens, and the same number of blooms (defined as 500/ml or greater for one species). Chlorophyll a was measured weekly for 36 weeks following refill (Figure 38). The analysis of variance showed that the test pools had significantly lower chlorophyll values than the control pools when only the last two months were considered. Thirty-six weeks after refill, however, chlorophyll levels were similar in the control and test tanks. Considering all the data relating directly or indirectly to phytoplankton, it appears unlikely that a lake-wide pulse of phytoplankton growth will occur following refill.

In addition to phytoplankton and chlorophyll a, the pool simulations were monitored for sediment-water interface adenosine triphosphate levels and total bacterial counts. These parameters were chosen to determine whether microbial metabolic activity was higher in drying sediments. The data for ATP and total plate counts fluctuated greatly with time for both test and control sediments. No trends were apparent. These results strengthen the conclusion that significant breakdown of organic matter in drying sediment does not occur

Benthic invertebrate and macrophyte populations were also determined and are presented in Chapter VI.

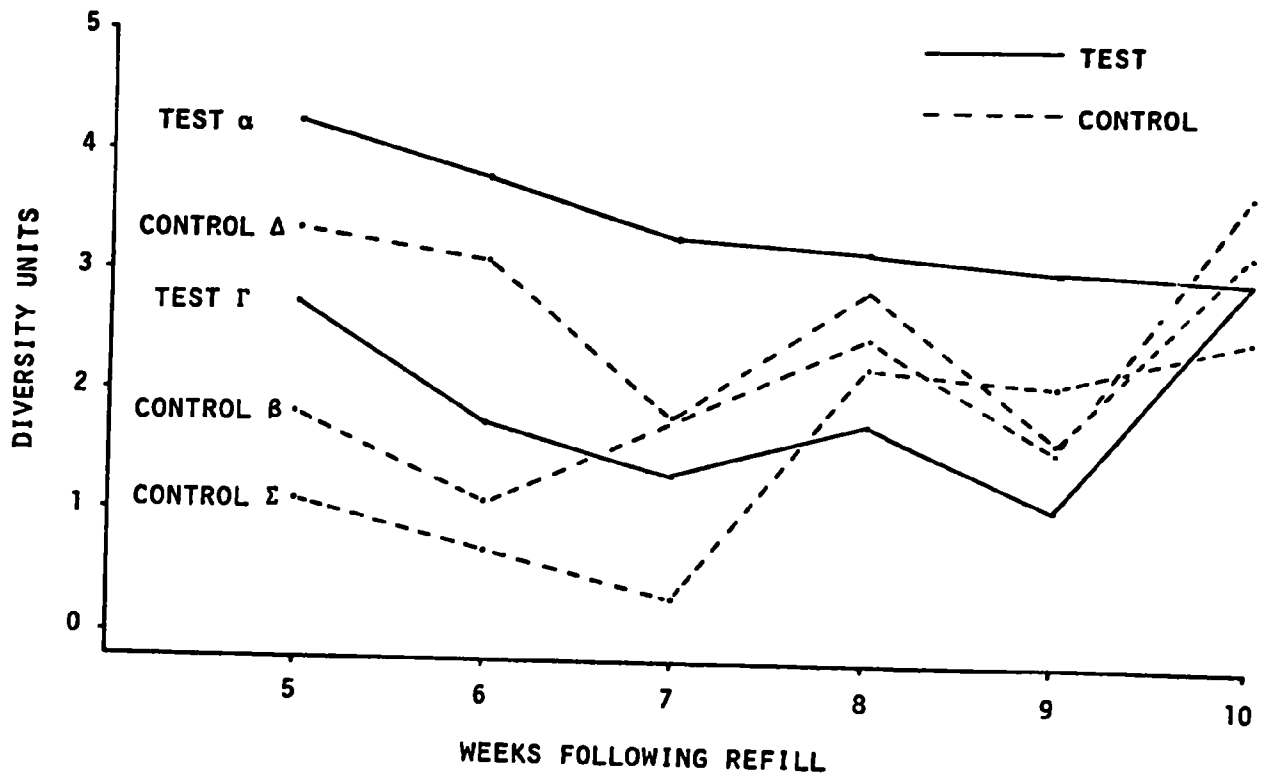


Figure 37. Diversity indices for test and control pool simulation phytoplankton populations

TABLE 20. POST DRAWDOWN POOL SIMULATION PHYTOPLANKTON RESULTS

| Date | Test Pools | | | | | | Control Pools | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | α | | | γ | | | β | | | Δ | | | ε | | |
| | Count[a] | B[b] | BG[c] | Count | B | BG | Count | B | BG | Count | B | BG | Count | B | BG |
| 4/12/74 | 1,953 | 0 | 20 | 4,792 | 2 | 66 | 6,379 | 2 | 9 | 2,742 | 1 | 60 | | | |
| 4/20/74 | 10,206 | 8 | 27 | 8,306 | 2 | 5 | | | | 3,680 | 2 | 36 | 19,090 | 1 | 3 |
| 4/24/74 | 7,268 | 4 | 16 | 3,410 | 2 | 3 | | | | 6,442 | 2 | 19 | 40,883 | 1 | 1 |
| 5/02/74 | 23,942 | 10 | 55 | 10,465 | 4 | 9 | 3,540 | 3 | 45 | 18,170 | 6 | 51 | 8,970 | 2 | 33 |
| 5/10/74 | 3,864 | 2 | 53 | 1,967 | 1 | 1 | 5,865 | 3 | 17 | 29,120 | 3 | 18 | 6,323 | 3 | 74 |
| 5/31/74 | 4,990 | 3 | 39 | 1,081 | 0 | 6 | 4,068 | 2 | 32 | 11,146 | 2 | 55 | 1,326 | 0 | 37 |
| 6/07/74 | 12,060 | 2 | 3 | 510 | 0 | 18 | 5,580 | 2 | 44 | 47,992 | 16 | 27 | 3,060 | 1 | 55 |
| MEAN | 9,183 | 4 | 30 | 4,362 | 2 | 15 | 3,633 | 2 | 29 | 17,041 | 5 | 38 | 13,275 | 1 | 34 |

a Count in organisms per ml

b Number of blooms, defined as 500 or more cells of one species per ml

c Percent of total count belonging to the phylum Cyanophyta

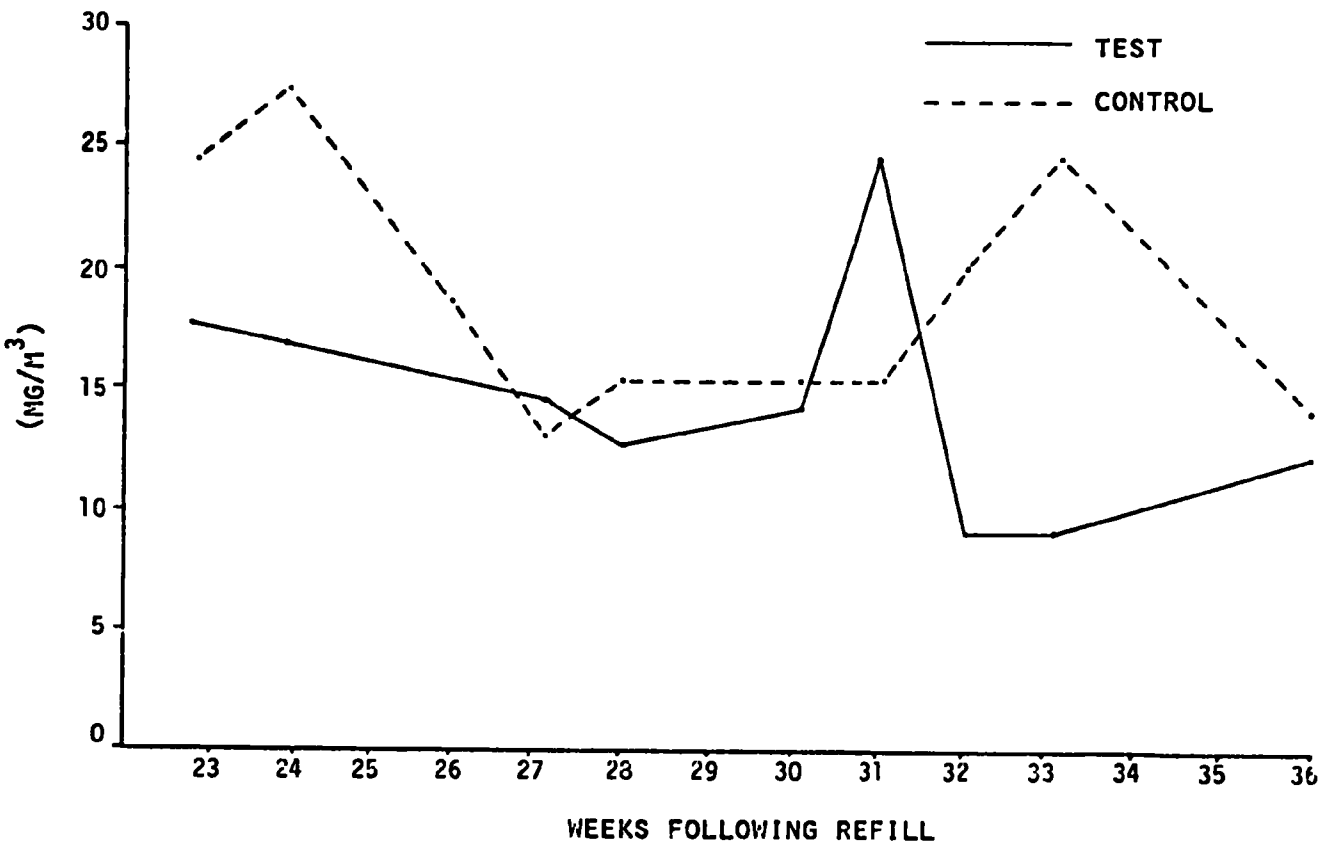


Figure 38. Chlorophyll concentrations in test and control pool simulations following refill

## SECTION VIII

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## SECTION IX

## APPENDIX

## METHODS

Whenever possible, the methods used followed those described in either Standard Methods for the Examination of Water and Wastewater (A.P.H.A., 1971), Methods for Chemical Analysis of Water and Wastes (E.P.A., 1971b), or Biological Field and Laboratory Methods for Measuring the Quality of Surface Waters and Effluents (E.P.A., 1973). The following is a summary of methods used for water, sediment and statistical analyses:

### WATER ANALYSES

1. Algal Assay - Performed according to the U.S. Environmental Protection Agency's (1971a) basic bottle algal assay procedure. Details are provided in the text.

2. Alkalinity - Run colorimetrically using the methyl orange brom-cresol green mixed indicator (A.P.H.A., 1971)

3. Chemical Oxygen Demand - Determined using the dichromate-sulfuric acid technique described in Standard Methods (A.P.H.A., 1971).

4. Chlorophyll a - Samples were Millipore filtered (0.45 μ), extracted in 90 percent acetone, and analyzed spectrophotometrically as described by Strickland and Parsons (1968).

5. Conductivity - Determined in the laboratory using a Beckman Model RC 16BZ conductivity bridge.

6. Dissolved Oxygen - For most field and simulation measurements, a YSI oxygen electrode was used. For the diurnal productivity measurements, the Winkler-Azide modification technique (A.P.H.A., 1971) was used.

7. Nutrients -Inorganic nitrogen and phosphorus forms were determined with a Technicon Auto Analyzer on samples preserved with one ml saturated mecuric chloride per liter of sample. Ammonia was determined using modifications (Yorton, 1971) of the phenol-hypochlorite method (Technicon Corporation, 1969). Nitrate was determined by the automated hydrazine reduction technique (Henriken, 1965), and nitrite by adaption of the Standard Methods (A.P.H.A.) diazotization procedure to the Auto Analyzer. Orthophosphate was measured by the Murphy and Riley (1972) single reagent method. Total organic nitrogen was determined by the macro-Kjeldahl method according to Standard Methods (A.P.H.A., 1971). Recovered ammonia was measured by titration or by the Auto Analyzer. Particulate organic nitrogen was measured by micro or macro-Kjeldahl on solids scraped from an $MgCO_3$ mat placed on a 0.45 μ Millipore filter. Total phosphate analyses were performed by persulfate-sulfuric acid autoclaving for one hour at 15 psi followed by manual determination of orthophosphate by the Murphy and Riley (1962) method.

8. pH - Field measurements were made with Orion Model 401 Meter. A Corning Model 12 was used in the laboratory.

9. Productivity - Field and simulation determinations were made utilizing the diurnal free water oxygen method described by Odum and Hoskin (1958) and the radioisotopic carbon bottle technique as presented by Strickland and Parsons (1968).

10. Turbidity - Determined in the laboratory with a Hach Model 2100A Turbidimeter.

SEDIMENT ANALYSES

1. Adenosine Triphosphate - A spatula was used to sample sediments from test and control simulations. The sample consisted of about 10 $cm^2$ of surface sediment (0.5 cm deep). A sediment-filtered lake water suspension was Millipore filtered and the ATP extracted in boiling TRIS buffer. A Dupont luminescence biometer was used to detect the amount of light emitted by the sample ATP in the presence of luciferin and luciferase (the firefly reaction).

2. Bacterial Counts - Samples were taken as described above for ATP. Samples were suspended in Millipore filtered (0.45 μ) lake water and serial dilutions plated on tryptone glucose extract agar. Incubation was at 25°C for 72 hours. Colony counts were related to sediment dry weight.

3. Consolidation - The decrease in sediment height, as measured by a meter stick, was expressed as the percentage of the original sediment height.

4. Nutrients - Total organic nitrogen and free ammonia determinations were made on fresh-sediments diluted with demineralized water into a thin slurry. Ammonia was distilled from the slurry buffered at pH 8.3 with sodium bicarbonate and the recovered ammonia measured by titration. Total organic nitrogen was determined similarly by micro-Kjeldahl digestion. Results were converted to a dry weight basis by drying and weighing a known volume of the sediment slurry. Total phosphate was run on fresh sediment slurries using persulfate oxidation and the Murphy and Riley (1962) manual method for orthophosphate.

5. Weights - Wet weight, dry weight and ashed weight determinations were made according to Standard Methods (A.P.H.A., 1971). From those data, appropriate calculations were made for percent water and volatile solids content.

STATISTICAL ANALYSIS

An analysis of variance (ANOVA) was used to analyze the experimental pool results. ANOVA (Sokal and Rohlf, 1969) is a test of whether two or more sample means could have been obtained from populations with the same parametric mean with respect to a given variable. The variables in this experiment were the physical, chemical and biolo-

gical parameters. The two populations are a population of drawdown lakes versus a population of control lakes, and the two sample means upon which we based our population conclusions were the test and the control pools. ANOVA determines whether the sample means from the test and control pools differ from each other to such an extent that we must assume (with 95% confidence) that they were sampled from different populations. Two ANOVA designs were considered: a single classification ANOVA and a block design ANOVA. The single classification ANOVA is used to determine if there is a significant mean variable difference between the control and test pools during the time-week period. The block design is similar except it removes the weekly (block) variation, making it a more exact test of treatment (test versus control) effects. Weekly variation in the pools could be caused by climatic factors and it is assumed that these factors affect the test and control pools in a similar fashion.

References cited in this appendix are listed in Section VIII.

## TECHNICAL REPORT DATA
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| | | |
|---|---|---|
| 1. REPORT NO<br>EPA-600/3-77-005 | 2. | 3. RECIPIENT'S ACCESSION NO |
| 4. TITLE AND SUBTITLE<br>Lake Drawdown as a Method of Improving Water Quality | | 5. REPORT DATE<br>January 1977<br>6. PERFORMING ORGANIZATION CODE |
| 7. AUTHOR(S)<br>Jackson L. Fox, Patrick L. Brezorik and Michael A. Keirn | | 8. PERFORMING ORGANIZATION REPORT NO |
| 9. PERFORMING ORGANIZATION NAME AND ADDRESS<br>University of Florida<br>Gainesville, Florida 32611 | | 10. PROGRAM ELEMENT NO.<br>1BA031<br>11. CONTRACT/GRANT NO<br>EPA R-800305 |
| 12. SPONSORING AGENCY NAME AND ADDRESS<br>U.S. Environmental Protection Agency<br>Corvallis Environmental Research Laboratory<br>200 S.W. 35th Street<br>Corvallis, Oregon | | 13. TYPE OF REPORT AND PERIOD COVERED<br>Final<br>14. SPONSORING AGENCY CODE<br>EPA/600/02 |

15. SUPPLEMENTARY NOTES

16. ABSTRACT

Investigations were made to determine the feasibility of radical drawdown as a restoration technique for Lake Apopka, Florida, a 12,545 hectare lake in central Florida. Field studies showed the lake to be hypereutrophic with continual algal blooms, mats of floating water hyacinths, and a flocculent organic muck bottom rich in interstitial water nutrients. Sediments were dredged from the lake bottom and placed in aquaria, columns, tanks and pools. Following dewatering and varying drying periods, the containers of sediment were refilled. A large number of physical, chemical and biological parameters were monitored before, during and following sediment drying. Results indicate that drawdown improves subsequent refill water quality. In muck sediments, drying causes significant water loss and shrinkage. Loss of organic material is minimal. During and following refill, sediment is colonized by two macroscopic aquatic plants, Typha (cattail) and the alga, Chara. Drying results in only minor chemical changes in muck sediment. Refill water in the pool test simulations has the same or lower nutrient content, lower turbidity, higher dissolved oxygen, lower temperature, fewer algae and a more diverse benthic invertebrate population. Based on these laboratory scale investigations, drawdown appears to be an effective restoration technique for Lake Apopka.

17. KEY WORDS AND DOCUMENT ANALYSIS

| a. DESCRIPTORS | | b. IDENTIFIERS/OPEN ENDED TERMS | c. COSATI Field/Group |
|---|---|---|---|
| Sediments* | nitrogen | | 08M |
| renovating | aquatic weeds* | | 07B |
| lakes | aquatic biology | | 06F |
| limnology | benthos | | |
| phosphorus | bioassay | | |
| *Major descriptors | | | |

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