Water Table Effects on Histosol Drainage Water Carbon, Nitrogen, and Phosphorus


ABSTRACT

Water table (WT) levels in Histosols may influence release of nutrients such as C, N, and P into drainage water. This study examined the effects of water table level on redox potential; C, N, and P release; and inorganic and organic P concentrations in soil columns from an Everglades Histosol. Soils were from two fields of Pahokee muck (eutic, hyperthermic, lithic Medisaprists) in the Everglades Nutrient Removal Project in Florida. Soil columns were subjected to four WT levels: 0 (flooded), 10, 20, and 35 cm below the soil surface. Each 30 d for 5 mo, porewater was drained and columns were leached with deionized water, followed by reestablishment of water table levels. Redox potential tended to stabilize 6 to 10 d after onset of each flooding-drainage cycle and displayed substantial spacial heterogeneity below WTs. Release of NH$_3$–N and TKN were not influenced by WT depth, but release of NO$_3$–N substantially increased with increasing WT depth in both soils. In soil from a previously flooded field, total P release increased with increasing WT depth. With soil from a previously drained field, total P release decreased with increasing WT depth. Differences in P release response appeared to be related to previous flooding-drainage history of the site fields. Fractionation of soil P following the study revealed that labile pools of P were influenced by WT depth but sizes of bulk inorganic and organic P pools were not. From <0.7 to 2.3% of total soil P was leached.

Histosols in the Everglades Agricultural Area (EAA) of south Florida are normally flooded when not under crop production, to decrease the rate of organic matter decay (Reddy and Rao, 1983; Snyder, 1987). Summer flooding reportedly increases loss of plant nutrients from these soils, resulting in enrichment of drainage waters with C, N, and P (Lucas, 1980; Reddy, 1982a, 1983b; Reddy and Rao, 1983). Terry et al. (1980) observed that Histosols released 20 times more P with a 30 cm water table depth than with a 90 cm water table depth. Reddy (1982a) observed that Histosols from Central Florida released four to six times more P into drainage effluent when they had been flooded for 25 d than when they had not been flooded. Soluble P released to drainage water from Histosols may be detrimental to water quality of the water bodies into which the drainage water flows (Terry et al., 1980) because of promotion of phytoplankton growth and resulting enhancement of the eutrophic status of the water body (Pollman and Brezonik, 1979; Reddy, 1987) and decreases in plant diversity in the sawgrass (Cladium jamaicense L.) marsh of the Everglades (Amador et al., 1992; Lake Okeechobee Technical Advisory Committee, 1989). Phosphorus is often considered the limiting factor restricting primary productivity in aquatic systems (Boström et al., 1982). The primary productivity of sawgrass-dominated habitats is P-limited (Steward and Ornes, 1983). Phosphorus enrichment of northern Everglades habitats may lead to displacement of sawgrass by cattail (Typha domingensis), elimination of native oligotrophic algal mat communities, and lower dissolved oxygen concentrations in standing water (South Florida Water Management District, 1992; Belanger et al., 1989).

During wet seasons, excess drainage water from the cultivated Histosols of the EAA is routed through a network of canals into Lake Okeechobee, the Atlantic Ocean, the Gulf of Mexico, and several water conservation areas bordering the EAA to the east and south (Izuno et al., 1991; Qualls and Richardson, 1995; Snyder, 1987). Some of the water in these water conservation areas flows into Everglades National Park. Thus, excessive P release from cultivated Histosols in the EAA could pose an environmental threat to Lake Okeechobee and the Everglades National Park.

The Everglades Nutrient Removal Project (ENRP) is being constructed by the South Florida Water Management District (SFWMD) to bioremediate EAA drainage water that is high in N and P due to engineering and agricultural activities in the developed northern part of the Everglades (South Florida Water Management District, 1990; South Florida Water Management District, 1992). This N- and P-enriched drainage water is to be bioremediated prior to reaching the Water Conservation Areas (WCAs) downstream, in order to maintain low nutrient levels in waters entering the Everglades. The ENRP is a large-scale wetland engineering scheme designed to provide vegetated water-cleansing buffer


Abbreviations: EC, electrical conductivity; DI, deionized water; ENRP, Everglades Nutrient Removal Project; EAA, Everglades Agricultural Area; i.d., internal diameter; NH$_3$–N, ammonium-nitrogen; NO$_3$–N, nitrate-nitrogen; SRP, soluble reactive phosphorus; TKN, total Kjeldahl nitrogen; TOC, total organic carbon; TP, total phosphorus; WT, water table.
areas separating the EAA from Water Conservation Area 1 (WCA 1), which drains into three other WCAs before draining into Everglades National Park. The ENRP has been taken out of agricultural production and will be flooded to its natural water table level so growth of natural sawgrass and other plant communities can be re-established. The intent is to use the ENRP to deliver to the WCAs, water that is close to natural levels of P, N, and soluble C. Although N and C processes are important, emphasis is usually placed on P because in many systems, P effluent quality dictates the design criteria for constructing wetland phytoremediation and soil-sink nutrient removal systems.

Upon full operation of the ENRP marsh, hydrologic fluctuations can result in a variable water table, thus influencing the breakdown of wetland soil organic P. More information is needed on the effect of water table levels on the potential for P release from these soils. This study was part of a series of studies conducted to describe biogeochemical nutrient cycling in the ENRP prior to full initiation of the ENRP (Reddy and Graetz, 1991).

The mechanism of water table effects on amounts of C, N, and P released from Histosols is not well understood. The purpose of this study was to determine the effect of water table depth in previously drained and previously flooded agricultural organic soils on (i) soil oxidation-reduction potential; (ii) release of total organic C (TOC), NO$_3^-$-N, NH$_4^+$-N, total Kjeldahl N (TKN), soluble reactive P (SRP), and total P (TP) into drainage and leachate waters; and (iii) labile and nonlabile pools of inorganic and organic P in the soil.

**MATERIALS AND METHODS**

**Soils and Location**

Soils used in this study were Pahokee mucks (euic, hyperthermic, lithic Medisaprists) from Fields A and D of the ENRP site (Fig. 1). Both fields had been used for sugarcane (**Saccharum** spp.) and/or vegetable crop production since the early 1950s. At the time of sampling, Field A had been used for sugarcane production prior to being flooded for about 10 mo. Field D was drained and under sweet corn (**Zea mays** var. **Saccharata** (Sturt.) Bailey) production at the time of sampling.

Intact soil columns were obtained on 21 Sept. 1991 by slowly driving a polyvinyl chloride (PVC) pipe (6.2 m internal diam., i.d., with one end beveled) to a depth of about 45 cm with a sledge hammer. Cores were then retrieved using a shovel. Columns were sealed at both ends with rubber stoppers and
stored at 23°C for 3 mo prior to initiation of the flooding-drainage treatments.

Experimental Design
The study used a 2 by 4 factorial design with three replications. The first treatment (unreplicated) was soil core source, that is, Field A that had been previously flooded and Field D that had been previously drained. The second treatment was water table level imposed on three replicate columns. These levels were 0 (continuously flooded), 10, 20, and 35 cm below the soil surface. While these water table level differences are relatively small compared to what occurs under some wetland field conditions, they are appropriate to a column study system and with some caution, results from such a study can be extrapolated to some field conditions.

Incubation (Flooding-Draining-Leaching Cycles) Procedure
Each PVC column was equipped with a fill tube and a water table sighting/hydraulic balance tube (Fig. 2). The bottom of each column was lined with glass wool and capped with a no. 13 rubber stopper. The bottom stopper was equipped with a glass outlet tube connected to (i) a 1 cm i.d. plastic tube that was used to monitor and control water table level; and (ii) a smaller i.d. tube (1.5 mm i.d.) to allow air to escape during flooding. Connections were secured with clear silicone rubber sealant.

To impose each cycle of water table level treatments, simulated rainwater was added from below [as opposed to the studies of Reddy (1982b) and Pavan et al. (1984)] to avoid leaching C, N, and P from drained portions of the soil, to minimize entrapment of gas pockets that would prevent complete soil saturation, and to mimic the natural upward movement of the groundwater table in the field. Columns also were periodically tapped lightly on the sides to release entrained gas pockets. Soil columns were incubated in a temperature-controlled (21 ± 6°C) greenhouse. Simulated rain water minus P was prepared to mimic Florida rain water chemistry according to Gholz (1991, personal communication). The composition of this water was 628, 195, 492, 133, 435, 142, 421, and 43 μM L−1 of Cl−, NO3−N, SO42−S, NH4−N, Ca2+, Mg2+, Na+, and K+, respectively.

Initially, each soil column was leached by applying 2 L (approx. 8 pore volumes) of 0.01 M CaCl2 to the top of the soil columns, followed by 1 L of deionized water to flush soluble nutrients from the soil. To collect leachate from each column, the water fill tube and the water table sighting/hydraulic balance tube were disconnected (Fig. 2), the tube exiting the bottom of the column was connected to a side arm suction flask, and each flask was connected to a vacuum manifold. Soil columns were then allowed to drain naturally followed by applying 100 cm water (0.1 bar) suction. Suction of 100 cm water was maintained by means of a controlled bleed valve and a 100 cm water manometer connected to vacuum lines. Suction values were tested at several points between the vacuum pump and soil columns and found to be homogeneous within 1.5 cm Hg. Drainage rates during leaching varied considerably between columns. Thus it was necessary to apply >100 cm water suction (in some cases as high as 5000 cm water) to some ponded columns for short periods of time to approach the target leaching rate of 3 cm h−1 and to keep the interval between cycles at 7 d or less.

Water table treatments were initiated on 21 Dec. 1990. Once every 30 d for a period of 5 mo, porewater in the columns was collected with 100 cm water suction in suction flasks underneath the columns. This water constituted free-drainage porewater and constituent concentrations were included in release calculations. Columns were then leached for about 10 h with 1 L of deionized water and were allowed to drain naturally into suction flasks underneath the columns followed by 100 cm water suction once more (Reddy, 1982b). Greater than 100 cm water suction was applied to some slow-leaching columns as described above. Some of these slow-leaching columns required 48 h or more to pass 1 L of surface-applied simulated rainfall even under elevated suction. Free-drainage porewater plus leachate constituents for each cycle were expressed on a mg m−2 per day basis.

Upon collection, water samples were stored at 5°C immediately after collection and just prior to analysis. In the interim, samples were frozen at −5°C or colder.

Redox Measurements
On one of the three replicate columns for each field and water table level treatment, four 0.64-mm diam. (22 gauge) platinum wire electrodes were inserted via small holes drilled in the sides. Each electrode wire was 3.7 cm in length and insulated with 1.5 cm of heat-shrinkable tubing (Mueller et al., 1985). Electrodes were sealed to the column wall with silicone glue. These electrodes were positioned 5, 15, 25, and 35 cm below the initial soil surface. Redox potentials were measured with a portable redox meter and a calomel reference electrode, and corrected to SHE values by adding +244 mV to each reading.

Analytical Methods
Water
Each porewater and leachate sample was divided into three subsamples. The first subsample was analyzed for pH and conductivity (EC) without filtration or acidification (APHA,
A second subsample was filtered through a 0.45-μm Gelman membrane filter, acidified with H₂SO₄ to pH < 2, and analyzed for soluble NH₄⁻N, NO₃⁻N, and SRP using a Technicon AutoAnalyzer II EPA method 365.2 (EPA, 1983). The filtered samples likely contain some suspended particles smaller than 0.45 μm. These very small particles likely contain some organic and inorganic P (APHA, 1989) but it was operationally assumed that only dissolved inorganic P was detected by this method.

A third subsample of acidified unfiltered water was analyzed for total organic carbon (TOC) by wet oxidation using APHA method 5310 D (APHA, 1989). A separate aliquot of this acidified filtered subsample was subjected to micro-Kjeldahl digestion and analysis for total N (TKN) by an automated colorimetric method (EPA, 1983). To determine total P (TP), an additional aliquot of this acidified filtered subsample was digested with sulfuric acid-peroxidase in a 380°C block digester to convert suspended and dissolved organic P and suspended inorganic P to dissolved inorganic P (EPA, 1983). The digestate was analyzed for soluble inorganic P using the AutoAnalyzer method described above using singl azotometric acid reagent. Porewater and leachate P from the columns was fractionated into inorganic SRP and suspended and dissolved organic P. Organic P was calculated as the difference between TP and SRP.

Soil

At the conclusion of the study period, soil was sampled from all columns from each 10-cm soil column depth increment. Soil bulk density was determined from the mass of dry matter (105°C for 24 h) in each core section. Ash content of 1-g subsamples of soil core increments was determined by ignition at 550°C. Total C and N were determined on finely ground oven-dried soil that was sifted through 0.15-mm screen (100 mesh) and analyzed with a Carlo-Erba NA-1500 Carbon-Nitrogen-Sulfur Analyzer (Haak-Buchler Instruments, Saddlebrook, NJ). Soil was dried and ground to pass a 2-mm (10 mesh) screen and total P was determined by ashing and HCl hydrolysis (Andersen, 1974). A 1-g dry soil sample was ignited at 550°C in a muffine furnace for 1 h. The ash was dissolved in 6 M HCl while heating to near dryness. The solution was then rehydrated with 1 M HCl, filtered through qualitative filter paper, then analyzed for P on an AutoAnalyzer. Certified standard reference materials (SRM) were used to check the accuracy of the data resulting from the digestion process.

Soil Phosphorus Fractionation

At the conclusion of the study period, soil was sampled as described above from each water table treatment. The P in these soil samples was fractionated by a sequential extraction scheme modified from Bowman and Cole (1978) and Reddy and Ivanoff (1991), and depicted graphically in Newman and Reddy (1993). The general sequence of extractions was 0.5 M NaHCO₃, then 1 M HCl, then 0.5 M NaOH.

In the first extraction step, wet soil samples were extracted with 25 mL of 0.5 M NaHCO₃ at pH 8.5 for 16 h on a reciprocating shaker, followed by centrifugation at 7000 rpm for 10 min. The concentration and volume of bicarbonate solution were adjusted to account for dilution by soil pore water in each soil sample, to obtain a uniform concentration of 0.5 M NaHCO₃, and a uniform volume of 25 mL of extractant, adjusted to pH 8.5. All extracts were filtered with Whatman no. 41 filters to exclude large particulates, then analyzed for SRP and TP. Extractable organic P was determined as the difference between TP and SRP. The difference between 0.5 M NaHCO₃-extractable TP and 0.5 M NaHCO₃-extractable SRP was considered to be a form of labile organic P (Bic-P). Bicarbonate-extractable inorganic P (Bic-P) represents free SRP in soil water and weakly adsorbed inorganic P on surfaces (Qualls and Richardson, 1995), thus is labile.

In the second extraction step, the residual soil was extracted with 1 M HCl for 3 h with shaking; then centrifuged, decanted, and filtered as described for the first extract. The HCl extractable SRP was considered to be a form of relatively nonlabile inorganic P (HCl-P). The HCl-extractable organic P was considered to be a form of moderately labile organic P (HCl-P).

In the third extraction, residual soil was extracted with 0.5 M NaOH for 16 h with shaking; then centrifuged, decanted, and filtered as described for the first extract. The NaOH-extractable TP was designated NaOH-TP. To separate the operationally defined fulvic acid (FA-P) and humic acid (HA-P) P pools, a measured volume of the NaOH extract was acidified to pH 0.2 with concentrated HCl (resulting in precipitation of acid insoluble P), centrifuged, decanted, and filtered as described above, and analyzed for TP to determine FA-TP. The SRP in this acidified NaOH extract (alkali hydrolyzable and acid soluble) SRP was considered to be a form of moderately labile organic P (FA-P) recoverable as inorganic P. The FA-P, in the organic soils from these two fields was considered to be derived primarily from organic P. The organic P in this acidified NaOH extract was considered to be a form of moderately resistant organic P (FA-P). Humic acid (alkali hydrolyzable and acid insoluble) P (HA-P) was estimated by subtracting FA-TP from NaOH-TP was operationally considered to be a highly resistant form of organic P. Following the first and second extractions, the total weight of soil and extractant solution was determined. For the second and third extractions, the volume of the extractant used was equivalent to the amount of liquid removed from the previous extraction. After the last extraction, the remaining total soil P was quantified by being ashed at 550°C, dissolved in 1 M H₂SO₄, and analyzed for residual total P. This was considered to be a form of highly resistant organic P (Resid. P).

To determine microbial biomass P, a separate soil subsample was extracted with chloroform. One milliliter of chloroform was added to a wet soil subsample and left to react for 12 h under a hood. Then the soil was extracted with 0.5 M NaHCO₃, as described above. Microbial biomass P (MP) was calculated by taking the difference in the bicarbonate extractable total P from chloroform-treated and chloroform-unextracted soils (Hedley and Stewart, 1982). Since the chloroform acts as a cell lysing agent, increased amounts of P in the extracts of chloroform-treated soils should originate from P in microbial biomass. Microbial biomass P was considered to be a form of labile P and was operationally assumed to be all organic.

On the basis of this soil fractionation scheme, soil inorganic P was operationally grouped into: (i) labile inorganic P (Bic-P) and (ii) nonlabile inorganic P (HCl-P). Soil organic P was operationally grouped into: (i) labile organic P including bicarbonate-extractable organic P (Bic-P), and microbial biomass-assiated organic P (MP-P); (ii) moderately labile organic P including acid soluble organic P (HCl-P) and alkali hydrolyzable acid soluble organic P; (iii) moderately resistant organic P including alkali hydrolyzable acid soluble fulvic acid organic P (FA-P); and (iv) highly resistant organic P including alkali extractable but acid insoluble organic P associated with humic acid (HA-P) and residual P (Resid. P). Labile pools of both organic and inorganic P are assumed to be bioavailable.

Statistical Methods

The effects of water table level treatment were tested with linear regression while the differences between field A soil
Table 1. Selected characteristics of soils from Fields A and D of the Everglades Nutrient Removal Project site upon termination of the column study.

<table>
<thead>
<tr>
<th>Soil depth</th>
<th>Bulk density</th>
<th>Ash</th>
<th>Total C</th>
<th>Total N</th>
<th>Total P</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>g cm⁻³</td>
<td>g kg⁻¹</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Field A soil</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>0-10 cm</td>
<td>0.252</td>
<td>140</td>
<td>474</td>
<td>28.3</td>
<td>0.532</td>
</tr>
<tr>
<td>10-20 cm</td>
<td>0.275</td>
<td>111</td>
<td>486</td>
<td>27.8</td>
<td>0.388</td>
</tr>
<tr>
<td>20-30 cm</td>
<td>0.268</td>
<td>93</td>
<td>492</td>
<td>28.2</td>
<td>0.254</td>
</tr>
<tr>
<td>&gt;30 cm</td>
<td>0.175</td>
<td>79</td>
<td>514</td>
<td>23.0</td>
<td>0.160</td>
</tr>
<tr>
<td>Field D soil</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>0-10 cm</td>
<td>0.266</td>
<td>82</td>
<td>494</td>
<td>29.5</td>
<td>0.337</td>
</tr>
<tr>
<td>10-20 cm</td>
<td>0.279</td>
<td>89</td>
<td>500</td>
<td>28.6</td>
<td>0.296</td>
</tr>
<tr>
<td>20-30 cm</td>
<td>0.238</td>
<td>86</td>
<td>497</td>
<td>27.2</td>
<td>0.240</td>
</tr>
<tr>
<td>&gt;30 cm</td>
<td>0.193</td>
<td>89</td>
<td>492</td>
<td>26.4</td>
<td>0.206</td>
</tr>
</tbody>
</table>

and field D soil were examined with a t-test. Interaction between these two factors was not tested due to the lack of replication of the field soil variable. All tests, as well as the calculation of standard deviations, were conducted with SAS (SAS Institute, 1985).

RESULTS AND DISCUSSION

Soil Characterization

Soil bulk densities for the surface 20 cm from both fields was similar (Table 1), with values for the >30-cm depth increment noticeably lower than those of the surface soil. Ash content decreased with depth in Field A, while total C increased slightly. Total N concentrations were similar in both soils. Total P decreased with depth, particularly in the soil from Field A.

Soil Redox Potential

Redox potential values were generally higher in Field D soil than in Field A soil, and showed a large degree of variation while generally reflecting the location of the water table (Table 2). Soil redox potential below the water table generally changed to moderately reduced values as the water table treatments were imposed. With the onset of each new flooding cycle, redox potential values generally did not drop to stable values for at least 6 to 10 d. This is much longer than previously reported (Ponnampерuma, 1972; Reddy, 1987), and even longer than the slow rate reported for mineral wetland soil by Cogger et al. (1992).

Reasons for the apparently slow kinetics (poise) of redox changes in these soils might include: (i) a limited supply of readily-available C for microbial metabolism, despite these being organic soils; (ii) the redox buffering capacity of nitrate; and (iii) low concentrations of redox species or dominance of the system by redox species that are not amenable to accurate or precise redox monitoring by Pt electrodes, thereby inhibiting electron transfers detectable by the Pt electrodes. Recent work indicates that in some environments, dissolved H₂ may give a more accurate and precise description of soil and groundwater redox potential and zones of discrete biogeochemical activity than use of platinum electrodes (Chapelle et al., 1996).

In some cases, redox potentials >0 mV were maintained below the water table. While some reports have attributed such extreme variability in redox potential readings from soil-installed platinum electrodes to electrode failures such as poisoning (Bailey and Beauchamp, 1971; Bohn, 1971), leakage (Mann and Stolzy, 1972), and epoxy breakdown (Mueller et al., 1985), Cogger et al. (1992) found that such variability is due almost entirely to soil microscopic effects and not electrode malfunction. This could be due, in part to pockets of gas trapped below the water table.

Light tapping on the sides of the columns on about a weekly basis released some of the gas, resulting in substantial water level drops in the manometer and fill tubes of some columns. Formation of gas pockets was also minimized by reducing the rate at which water was added to the columns from below during initial flooding. Redox potential values at the 35-cm depth were also generally less negative than expected, likely due in part to diffusion of O₂ from oxygenated water in the manometer tubing and/or production of O₂ by algae in the glass outlet tubing.

At the conclusion of our study, all platinum wires were removed from the columns, tested with buffered quinhydrone (Dirasian, 1968), and found to be performing properly.

Table 2. Soil redox potentials. Values are means of all measurements made 10 or more days after (30-d) cycle initiation.

<table>
<thead>
<tr>
<th>Water table depth</th>
<th>Electrode depth, cm</th>
<th>Field A soil</th>
<th>Field D soil</th>
</tr>
</thead>
<tbody>
<tr>
<td>cm</td>
<td>5</td>
<td>15</td>
<td>25</td>
</tr>
<tr>
<td></td>
<td>redox potential, mV</td>
<td></td>
<td></td>
</tr>
<tr>
<td>0</td>
<td>-94 (68)†</td>
<td>-114 (27)</td>
<td>-71 (90)</td>
</tr>
<tr>
<td>10</td>
<td>681 (27)</td>
<td>385 (258)</td>
<td>-110 (41)</td>
</tr>
<tr>
<td>20</td>
<td>674 (23)</td>
<td>639 (109)</td>
<td>-189 (40)</td>
</tr>
<tr>
<td>35</td>
<td>702 (20)</td>
<td>723 (21)</td>
<td>131 (302)</td>
</tr>
</tbody>
</table>

† Standard deviations in parentheses.

Nutrient Release Patterns

Carbon

Release of TOC was not influenced by water table depth with either soil (Table 3). More TOC was released from Field D soil than from Field A soil. Leachates from Field D soil usually appeared darker and redder in color than those from Field A soil. The amount of TOC released decreased during the fifth leaching cycle with Field D soil.

Nitrogen

Water table depth did not significantly affect NH₄⁺-N release from either soil (data not shown) (Martin et al., 1991), though somewhat more NH₄⁺-N was released from Field D soil than from Field A soil. Average values for all water table depths were 3.1 and 5.5 mg NH₄⁺-N
Table 3. TOC released from two Everglades Nutrient Removal Project organic soils as affected by water table (WT) depth.

<table>
<thead>
<tr>
<th>WT depth (cm)</th>
<th>1</th>
<th>2</th>
<th>3</th>
<th>4</th>
<th>5</th>
<th>Mean</th>
<th>SD†</th>
</tr>
</thead>
<tbody>
<tr>
<td>Field A soil</td>
<td>514</td>
<td>306</td>
<td>285</td>
<td>225</td>
<td>-</td>
<td>269</td>
<td>180</td>
</tr>
<tr>
<td>10</td>
<td>246</td>
<td>356</td>
<td>282</td>
<td>260</td>
<td>-</td>
<td>237</td>
<td>140</td>
</tr>
<tr>
<td>20</td>
<td>233</td>
<td>270</td>
<td>249</td>
<td>245</td>
<td>-</td>
<td>237</td>
<td>103</td>
</tr>
<tr>
<td>35</td>
<td>287</td>
<td>353</td>
<td>293</td>
<td>291</td>
<td>-</td>
<td>249</td>
<td>134</td>
</tr>
<tr>
<td>Linear</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
</tr>
<tr>
<td>Mean</td>
<td>520</td>
<td>321</td>
<td>277</td>
<td>255</td>
<td>-</td>
<td>248</td>
<td>NS</td>
</tr>
<tr>
<td>Field D soil</td>
<td>363</td>
<td>580</td>
<td>539</td>
<td>489</td>
<td>253</td>
<td>445</td>
<td>167</td>
</tr>
<tr>
<td>10</td>
<td>418</td>
<td>517</td>
<td>501</td>
<td>609</td>
<td>304</td>
<td>460</td>
<td>133</td>
</tr>
<tr>
<td>20</td>
<td>456</td>
<td>559</td>
<td>397</td>
<td>488</td>
<td>154</td>
<td>407</td>
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<td>35</td>
<td>377</td>
<td>546</td>
<td>367</td>
<td>431</td>
<td>87</td>
<td>361</td>
<td>184</td>
</tr>
<tr>
<td>Linear</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
</tr>
<tr>
<td>Mean</td>
<td>404</td>
<td>550</td>
<td>451</td>
<td>504</td>
<td>200</td>
<td>418</td>
<td>NS</td>
</tr>
<tr>
<td>Foll A vs. D</td>
<td>NS</td>
<td>***</td>
<td>***</td>
<td>***</td>
<td>***</td>
<td>***</td>
<td>***</td>
</tr>
</tbody>
</table>

*** Significant at P = 0.001L  
† Standard deviation.
‡ Total organic carbon.

Release of NO$_3^-$-N increased with increasing water table depth in both soils (Table 4), with more released from Field A soil than from Field D soil. Since Field A was flooded for 10 mo prior to sampling, the greater NO$_3^-$-N release from this soil is puzzling. The effect of water table depth on the amount of NO$_3^-$-N formed during incubation was linear (Field A soil: NO$_3^-$-N = 3.65 + 1.59 WT, P = 0.0001, r$^2$ = 0.67; Field D soil: NO$_3^-$-N = 3.69 + 0.716 WT, P = 0.0001, r$^2$ = 0.41, where NO$_3^-$-N = mg NO$_3^-$-N m$^{-2}$ d$^{-1}$ and WT = water table depth in centimeters. Short-term flooding decreased leaching of NO$_3^-$-N as reported previously by Terry et al. (1980) and Reddy (1982a). Under flooded or water-saturated conditions, NO$_3^-$ is commonly used as a terminal electron acceptor during microbial respiration in wetland soils (Ponnamperuma, 1972).

Release of TKN was not influenced by water table depth with either soil (data not shown) (Martin et al., 1991). More TKN was released from Field D soil (mean of 31 mg N m$^{-2}$ d$^{-1}$, or 113 kg N ha$^{-1}$ yr$^{-1}$) than from Field A soil (mean of 22 mg N m$^{-2}$ d$^{-1}$, or 80 kg N ha$^{-1}$ yr$^{-1}$). No attempt was made to determine soluble organic N by difference (TKN - NH$_4^+$-N) because TKN commonly includes some NO$_3^-$-N, particularly with organic soils (Bremmer and Mulvaney, 1982).

Phosphorus

The quantity of SRP mineralized during five flooding-draining cycles increased as water table depth increased in Field A soil (Table 5), with the opposite trend observed in Field D soil. The effect of water table depth on the amount of soil SRP released was linear with both soils as indicated by the equations: (Field A soil: SRP = 0.267 + 0.0341 WT, P = 0.0001, r$^2$ = 0.26; Field D soil: SRP = 3.56 - 0.0560 WT, P = 0.0001, r$^2$ = 0.12), where SRP = mg P released m$^{-2}$ d$^{-1}$ and WT = water table depth in cm). More SRP was released from Field D soil than from Field A soil for water table depths of 0, 10, and 20 cm. Averaged over the 0, 10, and 20-cm water table depths, Field A soil released 0.5 mg SRP m$^{-2}$ d$^{-1}$ (1.8 kg SRP ha$^{-1}$ yr$^{-1}$) compared to 2.9 mg SRP m$^{-2}$ d$^{-1}$ (10.6 kg SRP ha$^{-1}$ yr$^{-1}$) for Field D soil.

The unusual behavior of Field A soil with flooding may have been due to the low Fe content in this soil (Porter and Sanchez, 1992; Qualls and Richardson, 1995) and/or the control of its P chemistry by Ca, CO$_3^-$ (Porter and Sanchez, 1992), and/or microbiological activities (Amador et al., 1992). Everglades Histosols are generally rich in calcium carbonate and the concentrations of Ca$^{2+}$ and CO$_3^-$ in porewater and overlying

Table 4. Nitrate-nitrogen released from two Everglades Nutrient Removal Project organic soils as affected by water table (WT) depth.

<table>
<thead>
<tr>
<th>WT depth (cm)</th>
<th>1</th>
<th>2</th>
<th>3</th>
<th>4</th>
<th>5</th>
<th>Mean</th>
<th>SD†</th>
</tr>
</thead>
<tbody>
<tr>
<td>Field A soil</td>
<td>0</td>
<td>9</td>
<td>10</td>
<td>5</td>
<td>-</td>
<td>7</td>
<td>8</td>
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<td>10</td>
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<td>14</td>
<td>11</td>
<td>5</td>
<td>14</td>
<td>10</td>
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<td>70</td>
<td>20</td>
</tr>
<tr>
<td>Linear</td>
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<td>***</td>
<td>***</td>
<td>***</td>
<td>***</td>
<td>***</td>
<td>***</td>
</tr>
<tr>
<td>Mean</td>
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<td>32</td>
<td>29</td>
<td>32</td>
<td>30</td>
<td>30</td>
<td>20</td>
</tr>
<tr>
<td>Field D soil</td>
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<td>6</td>
<td>4</td>
<td>4</td>
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<td>NS</td>
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<td>NS</td>
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<td>Mean</td>
<td>8</td>
<td>17</td>
<td>16</td>
<td>14</td>
<td>20</td>
<td>15</td>
<td>15</td>
</tr>
<tr>
<td>Foll A vs. D</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
</tr>
</tbody>
</table>

*** Significant at P = 0.05, 0.01, and 0.001, respectively.
† Standard deviation.

Table 5. Soluble reactive P released from two Everglades Nutrient Removal Project organic soils as affected by water table (WT) depth.

<table>
<thead>
<tr>
<th>WT depth (cm)</th>
<th>1</th>
<th>2</th>
<th>3</th>
<th>4</th>
<th>5</th>
<th>Mean</th>
<th>SD†</th>
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</thead>
<tbody>
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<td>0.7</td>
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<tr>
<td>10</td>
<td>0.5</td>
<td>0.1</td>
<td>0.1</td>
<td>0.3</td>
<td>0.5</td>
<td>0.3</td>
<td>0.3</td>
</tr>
<tr>
<td>20</td>
<td>0.3</td>
<td>0.3</td>
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<td>1.0</td>
<td>1.8</td>
<td>0.8</td>
<td>0.8</td>
</tr>
<tr>
<td>35</td>
<td>1.6</td>
<td>1.6</td>
<td>1.6</td>
<td>1.6</td>
<td>1.8</td>
<td>1.6</td>
<td>1.0</td>
</tr>
<tr>
<td>Linear</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
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<td>***</td>
</tr>
<tr>
<td>Mean</td>
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<td>0.8</td>
<td>1.2</td>
<td>0.8</td>
<td>0.8</td>
</tr>
<tr>
<td>Field D soil</td>
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<td>4.8</td>
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<td>3.4</td>
<td>3.9</td>
<td>1.8</td>
</tr>
<tr>
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<td>1.8</td>
<td>1.7</td>
<td>1.4</td>
<td>1.6</td>
<td>2.4</td>
<td>1.9</td>
</tr>
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<td>6.3</td>
<td>2.6</td>
<td>1.1</td>
<td>1.1</td>
<td>1.6</td>
<td>2.5</td>
<td>2.3</td>
</tr>
<tr>
<td>35</td>
<td>4.0</td>
<td>1.9</td>
<td>1.1</td>
<td>0.5</td>
<td>0.9</td>
<td>1.7</td>
<td>1.5</td>
</tr>
<tr>
<td>Linear</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>**</td>
<td>**</td>
</tr>
<tr>
<td>Mean</td>
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<td>2.8</td>
<td>1.7</td>
<td>1.4</td>
<td>1.9</td>
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<td>2.6</td>
</tr>
<tr>
<td>Foll A vs. D</td>
<td>***</td>
<td>***</td>
<td>***</td>
<td>***</td>
<td>***</td>
<td>***</td>
<td>***</td>
</tr>
</tbody>
</table>

*** Significant at P = 0.05, 0.01, and 0.001, respectively.
† Standard deviation.
Table 6. Chemistry of free drainage (porewater) collected from soil columns. Values are means of three replicates of five 30-d incubation and leaching cycles.

<table>
<thead>
<tr>
<th>WT depth cm</th>
<th>Vol. mL</th>
<th>pH</th>
<th>EC † dS m⁻¹</th>
<th>TOC</th>
<th>TKN</th>
<th>NH₄⁻N</th>
<th>NO₃⁻N</th>
<th>SRP</th>
<th>TP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Field A soil</td>
<td>184</td>
<td>148</td>
<td>179</td>
<td>33</td>
<td>3.2</td>
<td>0.44</td>
<td>0.28</td>
<td>0.04</td>
<td>0.08</td>
</tr>
<tr>
<td>Field D soil</td>
<td>164</td>
<td>124</td>
<td>200</td>
<td>49</td>
<td>4.8</td>
<td>0.70</td>
<td>0.02</td>
<td>0.58</td>
<td>0.63</td>
</tr>
</tbody>
</table>

**Linear regression**

Mean

Water are often near saturation with respect to calcite (Swift, 1984). It is also possible that the preleaching with 0.01 M CaCl₂ resulted in some apatite formation, altering the P release characteristics of the soil. With the soil from Field D, flooding increased leaching of SRP. This trend is consistent with other reports in the literature (Terry et al., 1980; Reddy, 1982a).

Trends for total P released during incubation were similar to those for SRP but less pronounced (data not shown) (Martin et al., 1991). Total P released increased with decreasing water table depth for Field A soil (TP = 0.503 + 0.0300 WT, P = 0.0006, r² = 0.17) and decreased with WT depth in Field D soil although the latter effect was not statistically significant. Mean SRP comprised 43 to 106% of leachate TP in Field A soil and 89 to 118% in Field D soil. This suggests that most of the P in the leachate was present as SRP. In about one third of the cases, SRP exceeded TP concentration. This indicates that the sulfuric acid-persulfate digestion method for total P may underestimate total P in some organic soils. It is also possible that with these organic soil leachates, SRP measured by the ascorbic acid method (EPA, 1983) includes significant amounts of dissolved and <0.45 μm-diameter suspended organic P.

Soil Phosphorus Fractions

Fractionation of soil P at the end of five 30-d flooding-drainage cycles revealed that labile pools of P were influenced by water-table depth. Percent distribution is based on the total mass of P in the 40-cm soil cores. Although total P concentration in the soil decreased with depth, relative distribution of pool sizes remained the same with depth (Table 7). Thus, data for whole soil columns were used in the calculation of pool sizes. Means of the various P pools with different water table depths are summarized in Table 8. Leachate SRP and TP in Table 8 represent cumulative leachate plus porewater P from the greenhouse column study prior to column soil sectioning. Although differences between the two soil types were apparent, no noticeable effect of water table depth was observed on sizes of the bulk inorganic and organic P pools.

Cumulative P leached represented < 0.7% of total P in soil from Field A, compared to 1.1 to 2.3% in soil from Field D. Labile P (Bio-P), which comprises 5 to 6% of total P, was not affected by water-table depth. Nonlabile P (HCl-P) was higher in soil from Field A than Field D. In Field D soil, labile P, represented a...
Table 7. Labile and nonlabile pools of soil P as influenced by water table (WT) depths.

<table>
<thead>
<tr>
<th>WT depth (cm)</th>
<th>Soil depth (cm)</th>
<th>Inorganic P</th>
<th>Organic P</th>
<th>Moderately labile P₀</th>
<th>Moderately resistant P₀</th>
<th>Highly resistant P₀</th>
<th>Resid. P</th>
<th>Total P (mg/kg)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>BiP₀</td>
<td>HCl-P₀</td>
<td>BiP₀</td>
<td>HCl-P₀</td>
<td>FA-P₀</td>
<td>FA-P₀</td>
<td>HA-P₀</td>
</tr>
<tr>
<td>0</td>
<td>0-10</td>
<td>4.8</td>
<td>18.8</td>
<td>3.1</td>
<td>6.9</td>
<td>12.7</td>
<td>4.4</td>
<td>15.9</td>
</tr>
<tr>
<td>10</td>
<td>0-10</td>
<td>5.3</td>
<td>25.5</td>
<td>3.0</td>
<td>16.7</td>
<td>4.0</td>
<td>2.8</td>
<td>12.2</td>
</tr>
<tr>
<td>20-30</td>
<td>0-10</td>
<td>4.7</td>
<td>22.6</td>
<td>4.8</td>
<td>6.8</td>
<td>0.4</td>
<td>3.5</td>
<td>14.2</td>
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<tr>
<td>30-40</td>
<td>0-10</td>
<td>7.9</td>
<td>16.5</td>
<td>4.7</td>
<td>11.7</td>
<td>0.0</td>
<td>2.5</td>
<td>11.3</td>
</tr>
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<td>4.2</td>
<td>6.0</td>
<td>7.8</td>
<td>4.2</td>
<td>17.6</td>
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<td>6.0</td>
<td>17.2</td>
<td>4.4</td>
<td>8.5</td>
<td>2.8</td>
<td>7.7</td>
<td>12.9</td>
</tr>
<tr>
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<td>0-10</td>
<td>5.7</td>
<td>14.6</td>
<td>5.7</td>
<td>10.3</td>
<td>1.1</td>
<td>3.6</td>
<td>9.0</td>
</tr>
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<td>0-10</td>
<td>5.8</td>
<td>17.5</td>
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<td>9.0</td>
<td>0.6</td>
<td>2.9</td>
<td>12.5</td>
</tr>
</tbody>
</table>

larger proportion of total inorganic P than in Field A soil. This is likely one reason why Field D soil released more P than Field A soil.

Labile organic P (BiP₀ and MP₀) was higher in Field D soil (17.4-18.6% of total P) compared to Field A soil (12.4-13.8% of total P). This may be another reason why Field D soil released more P than Field A soil.

Acid hydrolyzable P (HCl-P₀) was higher in Field A soil than Field D soil and organic P that was present in the moderately resistant pool was in the range of 11 to 15% of total P. The P present in the highly resistant pool was higher in Field D soils (up to 51% of total P) compared to Field A soils (up to 41% of total P).

CONCLUSIONS

The draining of wetland soils, whether permanently or periodically, increases release of NO₃⁻-N to drainage water, though permanent flooding minimizes release of NO₃⁻-N. Under natural conditions, water table fluctua-

Table 8. Distribution of labile and nonlabile pools of P as influenced by water table (WT) depth in Fields A and D of the Everglades Nutrient Removal Project site.

<table>
<thead>
<tr>
<th>WT depth (cm)</th>
<th>Leachate</th>
<th>Inorganic P</th>
<th>Organic P</th>
<th>Moderately labile P₀</th>
<th>Moderately resistant P₀</th>
<th>Highly resistant P₀</th>
<th>HA-P₀</th>
<th>Resid. P</th>
<th>Total P storage (mg P m⁻²)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>SRP</td>
<td>BiP₀</td>
<td>HCl-P₀</td>
<td>BiP₀</td>
<td>HCl-P₀</td>
<td>FA-P₀</td>
<td>FA-P₀</td>
<td>HA-P₀</td>
<td>Resid. P</td>
</tr>
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<td>0</td>
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<td>5.0</td>
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<td>7.2</td>
<td>11.6</td>
<td>0.7</td>
<td>4.0</td>
<td>11.4</td>
</tr>
</tbody>
</table>
tions occur in wetland soils, resulting in periodic release of NO\textsubscript{3}--N to drainage waters.

Field D soil at the time of sampling had much higher levels of SRP in the porewater than Field A soil. The lower levels of SRP in Field A soils at the time of sampling may be due to previous losses via diffusion to the overlying water or movement of P in lateral surface water and/or groundwater flow; or medium-term (10 mo) flooding may have transformed some labile forms of P into less labile forms. Results suggest that previous flooding of Field A depleted labile P or rendered it less labile, thus resulting in lower P release into the drainage water when this soil was subjected to frequent flooding and draining. These opposing results suggest that long-term flooding can decrease P concentration in the drainage water, even if the soils are drained and reflowed again. In both soils, preleaching with 0.01 M CaCl\textsubscript{2} and D.I. water likely flushed some fraction of P that subsequently would have become labile.

In Field A soil, more P was released from soil cores maintained at lower water table depths than from soil cores maintained under near-flooded conditions. At lower water table depths, it is likely that aerobic decomposition resulted in more rapid breakdown of labile pools of organic P, resulting in greater release of P; Results suggest that microbial P was lower in Field A soil with a 35-cm water table depth, compared to soil with a 0-cm water table depth. With the water table at 35 cm, aerobic decomposition likely prevailed over anaerobic decomposition, with the result that much of the microbial P was broken down and recovered as HCl-P. Though Field A soil had previously been flooded for 10 mo, during this period hydrolysis and incomplete breakdown of soil organic matter probably resulted in the accumulation of labile organic forms, which then were readily decomposed when the cores were exposed to aerobic conditions. Field D soil, however, released more SRP to the drainage water under flooded conditions than under drained conditions. Field D had been fertilized and maintained under drained conditions prior to soil sampling. Subsequent flooding of these soils likely resulted in solubilization of inorganic P.

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